Fish for the future

The effect of western rock lobster fishing on the deepwater ecosystems of the west coast of Western Australia

Final FRDC Report – Project No. 2004/049

Dr Lynda Bellchambers

Government of **Western Australia**
Department of **Fisheries**

Australian Government

Fisheries Research and Development Corporation

Fish for the future

Contents

2004/049 The effect of western rock lobster fishing on the deepwater ecosystems of the west coast of Western Australia

Principal Investigator: Dr L. M. Bellchambers

Address: Department of Fisheries Western Australia
PO Box 20
North Beach WA 6020
Telephone: 08 92030111   Fax: 08 92030199

Objectives

1. To identify gradients in the density/size distribution of western rock lobster to enable selection of representative areas.

2. To assess the catchability of western rock lobster and its relationship with population abundance and size structure.

3. To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water.

Non Technical Summary

Outcomes achieved to date

This study has resulted in the collection of biological, population and habitat data that improves our understanding of the role of western rock lobster in the deepwater ecosystem. The relationship between western rock lobster size and abundance and habitat established in this study is important for the effective management of the fishery in an ecosystem based fisheries management (EBFM) framework. For fisheries managers this information has application in two main areas. Firstly, it enables the examination of the spatial distribution of lobster abundance and size by habitat type and in combination with fine resolution fishing effort data may allow the assessment of standing biomass and harvest rate. Secondly, it provides invaluable information for the design and implementation of marine protected areas as a tool for research and species conservation. For researchers this study provides an assessment of different techniques to assess the benthic habitat of deepwater ecosystems that has previously been a costly exercise requiring specialised equipment. For industry this information is the first step in the process of assessing the impact of lobster biomass removal in the deepwater and provides detailed information for the ecological risk assessment (ERA) process required to meet Department of Environment, Water, Heritage and the Arts (DEWHA) regulations and maintain marine stewardship certification (MSC).

Similarly, depletion estimates for western rock lobster in shallow and deep water at the Abrolhos can be used to improve the stock assessments by using length-based models that will lead to a more robust assessment of the management of the fishery that will be of benefit to fisheries managers and industry alike. The Results confirm a high level of pot saturation at the Abrolhos, particularly in shallow water. This result suggests that there may be some economic benefit from further pot reductions in shallow waters at the Abrolhos Islands and these are currently being examined for the 2008/09 season.

The western rock lobster fishery was awarded Marine Stewardship Council (MSC) certification as a well-managed fishery in 2000 and has since been successfully re-certified in 2006. It was the first fishery in the world to receive this certification, which was awarded after an extensive review of the sustainability of the fishery and its impact on the marine environment. This certification largely reflects the significant data available on the shallow (<40 m) water ecology of lobsters and the impact of fishing and biomass removal. However, there remains a knowledge gap with respect to the deep-water ecology of western rock lobsters. Most western rock lobster undertake a substantial offshore migration as they approach sexual maturity which coincides with the size at which they reach legal minimum size and are targeted by the commercial and recreational fishery. This study seeks to fill in some of the knowledge gaps to assist with answering the question "what is the effect of lobster biomass removal on the ecocsystem?"

The benthic biota of deepwater ecosystems (35-75 m) was classified using towed video and diver sampling at Dongara, Jurien and Lancelin. While differences in sponge/algal assemblages and macroinvertebrate community composition were detected between study locations, a direct link between sponge/algal assemblage structure and macroinvertebrate community composition could not be established. Macroinvertebrates are important prey items for western rock lobsters, therefore differences in macroinvertebrate community composition will have implications for prey available to western rock lobsters. The two methods used to classify assemblage structure were also compared. Both methods had similar outcomes, suggesting a single method of classifying habitat can be employed in future studies.

Two components of this report were focused on examining the relationship between the abundance and size of western rock lobster and the habitats in which they are found. In the first component the relationship between lobster abundance and size, from the annual western rock lobster independent breeding stock survey (IBSS), and habitat variables were quantified using towed video transects at Dongara, Jurien and Lancelin. All three locations vary both with respect to the composition of habitats and the abundance and size of lobster, with Dongara being significantly different from the other two sites. The largest lobsters were found at Dongara while the highest abundance of lobsters was at Lancelin. In addition, lobsters vary both in abundance and size with habitat. The type of habitat benthos rather than the density of a particular habitat type determined lobster size.

In the second component western rock lobster size and abundance data from the annual western rock lobster independent breeding stock survey at the five subregions of the IBSS at Jurien was used. Two habitat datasets were used: (1) a towed video survey to derive benthic habitat data and (2) a full coverage habitat map derived from a multibeam hydroacoustic survey and towed video. Abundance and size of lobsters varied significantly across the five subregions of the IBSS at Jurien. Habitat classification from both towed video and the habitat map indicated that subregions also varied significantly in habitat composition. Multivariate statistical techniques show a strong association between western rock lobster abundance and size and habitat types the combination of *Ecklonia* and sponges appeared to be driving the differences observed.

Both components of the study illustrated that low numbers of larger lobsters are present in areas associated with sponge and high numbers of smaller lobsters are present in more structurally complex *Ecklonia* dominated habitats.

We used stable isotope analysis and gut content analysis to determine the diet and trophic position of western rock lobsters at three locations (Dongara, Jurien and Lancelin). Lobsters were primarily carnivorous, and no consistent differences in diet were detected with varying lobster size, sex or among locations. The main components of the diet were bait (from the fishery) and small crustaceans such as crabs and amphipods/isopods. Foliose red algae, bivalves, gastropods and sponges were minor contributors to diet. The diet of lobsters in deepwater differed to results of previous studies of lobsters from shallow water ecosystems. Coralline algae and molluscs are important prey in studies of lobsters from shallow water but were minor components of the diet. These differences are likely to reflect differences in food availability and potentially differences in choice of prey by lobsters that inhabit deeper water. Given the high contribution of bait to lobster diet, bait is likely to be subsidising lobster production in deep coastal ecosystems during the fishing season.

A desk top study was conducted to examine changes in the catchability of western rock lobster. The Abrolhos Islands region was used to examine catchability dynamics between shallow (< 20 fm) and deep (>= 20 fm) water western rock lobster using the DeLury depletion model. This region was chosen as a significant decline in CPUE due fishing in a relatively short period (3.5 months) occurs in this location. As a result of the high exploitation and the large number of pots operating in a relatively restricted area, the catchability of rock lobsters in shallow water was significantly increased by the 18% pot reduction in 1993/94. This study indicates that the increase is confined to the shallow water component of the fishery. This indicates that there is a significant level of pot saturation occurring in the shallow waters of the Abrolhos. This result suggests that there may be some economic benefit from further pot reductions in shallow waters at the Abrolhos Islands and these are currently being examined for the 2008/09 season.

In summary this project did not definitively answer the question what is the effect of biomass

removal of lobsters in deepwater. However it has substantially increased our understanding of deepwater lobsters and their use and interaction with the environment they inhabit. In accordance with the EcoSRG recommendations as the gradient approach adopted by this project did not result in the ability to examine the ecosystem impacts of biomass removal the next step is to implement closed areas.

The real value of this study lies in the increased knowledge of the ecology and biology of deepwater western rock lobster that allows scientists, fisheries managers and industry to design in partnership closed areas, as a research and fisheries management tool, with a good understanding of the relationships between lobster abundance and size structure and habitat variables. Designing closed areas without an adequate knowledge of habitat preferences and use and how it influences species distribution has, in several cases, resulted in areas that are either insufficient in size to encompass the range of habitats used by the target species or simply have failed to incorporate key habitats. This study provides invaluable information to ensure that the next step in the process i.e. negotiating, implementing and monitoring closed areas delivers a useful outcome for all stakeholders

Acknowledgements

1.0   Background

In 2000 the Western Rock Lobster Managed Fishery became the world's first fishery to receive Marine Stewardship Council (MSC) certification, the fishery was successfully re-certified in 2006. Since then the management process has moved on to address the MSC's annual audit requirements. As part of this process an ecological risk assessment (ERA) and more recently an environmental management strategy (EMS) have been completed. The ERA rated the effects of lobster fishing on the ecosystem as a low risk. However, the lack of research data about the ecological impacts of removing rock lobster biomass from the environment, particularly from deep water was noted, and a recent ERA has rated this aspect as a moderate risk. The Department of Environment, Water, Heritage and the Arts (DEWHA) will require similar information in the next assessment of the Western Rock Lobster Managed Fishery. A scientific reference group (SRG) was convened to provide advice on a research program that would address the ecosystem effects of fishing. This gap in the knowledge of deep-water rock lobster ecology was identified as a priority area of research. It was recognized that shallow water projects being planned by both the Department of Conservation (DEC) and CSIRO's Strategic Research Fund for the Marine Environment (SRFME) in the Jurien Marine Park would be complementary to any deepwater rock lobster research. There are significant opportunities for increased collaboration and co-operation across the respective agencies and institutions to ensure results will lead to a comprehensive regional understanding of ecological processes that relate to the western rock lobster fishery. The SRG devised a strategic framework to evaluate future research. The aim of the framework is to ensure that future research is focused primary on the question "What is the effect of lobster biomass removal on the ecosystem?"

1.1   Need

As a result of DEWHA and MSC strategic assessment processes, a number of areas of uncertainty in the western rock lobster fishery have arisen. The DEWHA recommendations to the Department of Fisheries explicitly state that the Department should examine mechanisms for monitoring potential ecosystem impacts of the fishery, including the appropriateness of reference areas that would allow comparison of fished and unfished areas. These recommendations also outline the need to assess options for ecosystem based fishery management including an assessment of the role and catchability of large western rock lobsters. In response to the recommendations of the DEWHA and MSC, the SRG identified that there is a major gap in the understanding of the interactions between the rock lobster fishery and the ecosystem in deep water. Although significant data is available on the shallow (<40 m) water ecology of lobsters and the fishing impact, there is gap in the knowledge of the deep-water ecology of lobster. They recognised that the major change in abundance and size structure of the western rock lobster population would have occurred in deep water (40-100 m) where most of the adult population resides, and concluded this was a priority area of research. The SRG also recognized that closed areas could not be put in place without research on the habitat structure and relative abundance and size structure of lobsters There is a need to collect basic ecological information to determine if changes in lobster density and size structure, due to fishing has caused significant changes in habitat structure and benthic community composition in deep water. This will provide information on the level of ecosystem impact of removing lobsters from deep-water habitats to improve the assessment of risk to the ecosystem to ensure that the western rock lobster fishery maintains MSC certification and complies with DEWHA requirements for export permits. However, the SRG recognised that research needs to occur in a structured manner and have highlighted the need for research proposals which sit within

the strategic framework which they have devised. The SRG recognises that the provision of a strategic framework and related scientific research will ultimately allow management of deep-water stocks in a more sophisticated ecosystem-based manner. If removal of lobster biomass by the deepwater fishery has resulted in detectable changes in the ecosystem, management options such as reduction in the fishing effort, minimum size changes and area closures will be considered by RLIAC to reduce the removal of biomass from areas of the deepwater fishery.

1.2   Objectives

1. To identify gradients in the density/size distribution of western rock lobster to enable selection of representative areas.

2. To assess the catchability of western rock lobster and its relationship with population abundance and size structure.

3. To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water.

2.0   General Introduction

Overfishing has resulted in the rapid decline of many species in coastal habitats (Dayton *et al.* 1995, Jackson *et al.* 2001). Fishing is the most widespread human impact on the marine environment (Pauly *et al.* 1998, Jackson *et al.* 2001). However, measuring the impacts of fishing can be difficult if habitat quality varies either naturally or as a result of anthropogenic impacts. Besides directly affecting the target populations, fishing may also result in indirect effects such as habitat modification (Bergman and Hup 1992, Turner *et al.* 1999, Kaiser *et al.* 2000, Kaiser *et al.* 2003) or trophic effects (Jennings and Kaiser 1998, Tegner and Dayton 2000). The removal of target species may result in a decrease in predation pressure on lower trophic levels (Shears and Babcock 2002). Depending on the role of the organisms that occupy these trophic levels, removal of predation pressure can have implications for the rest of the ecosystem (Shears and Babcock 2002). Trophic cascades are strong interactions between trophic levels where variation in the abundance or biomass of one trophic level alters the abundance, diversity or productivity of other trophic levels (Pace *et al.* 1999, Polis *et al.* 2000). Trophic cascades may involve top-down control where lower trophic levels are regulated by the abundance of predators or bottom-up control where higher-trophic levels are regulated by availability of nutrients or primary producers (Hall 1999).

Spiny lobsters are the targets of valuable fisheries worldwide (Butler *et al.* 2006) and like the majority of fishery stocks worldwide, lobster stocks are nearly all fully or overexploited (National Research Council 1994). All panulirus fisheries are now considered fully exploited (Phillips and Melville-Smith 2005). Western rock lobsters are ecologically important in the coastal marine environment they inhabit and form the basis of substantial commercial and recreational fisheries (Phillips and Melville-Smith 2005). The large size, predatory habits and high local abundance of spiny lobsters, suggest their impact on benthic prey communities should be profound, yet evidence of their influence on prey community structure appears to be limited largely to rocky temperate ecosystems (Butler *et al.* 2006). Predation by spiny lobsters has been demonstrated to lead to differences in invertebrate assemblage structure in New Zealand (Shears and Babcock 2002, Shears and Babcock 2003, Langlois *et al.* 2005, Langlois *et al.* 2006), Tasmania (Johnson *et al.* 2004, Pederson and Johnson 2006), South Africa (Mayfield and Branch 2000) and California (Tegner and Levin 1983).

The most extensive evidence that spiny lobsters alter prey communities with cascading effects on benthic community structure comes from studies on *Panulirus interruptus* and the predatory fish *Semicossyphus pulcher*. These two species controlled the abundance and distribution of two species of sea urchin (*Strongylocentrotus franciscanus* and *S. purpuratus*). In turn the urchins' grazing altered the abundance of giant kelp (*Macrocystis pyrifera*), the dominant species in Californian kelp forests (Tegner and Dayton, 1981). Similarly, predation by *P. interruptus* along with octopuses and whelks controls the distribution of bivalve and gastropod prey on rocky reefs in Southern California (Schmidt 1982, 1987).

In South Africa predation by *Jasus landii* and *Panulirus homarus* alters the abundance and size structure of mussels, urchins and gastropods. Conversely, lobster growth may be limited by the availability of prey items (Mayfield *et al.* 2000). Predation by lobsters and fish on benthic invertebrates is thought to have cascading effects in the ecosystem, including impacts on the macroalgal community structure and abalone recruitment.

Studies in New Zealand also suggest that lobsters (*Jasus edwardsii*) together with predatory fish control the abundance of sea urchins that in turn alter macroalgal structure in subtidal rocky

communities (Andrew and MacDiarmid 1991, Shears and Babcock 2002). For example when urchins (*Evechinus chloroticus*) were removed from urchin barrens dominated by crustose coralline algae those areas became macroalgal-dominated within 12 months. In general, macroalgal dominated areas are more abundant in marine protected areas where lobsters and large fish are protected and therefore more abundant (Babcock *et al.* 1999)

In Tasmania, *Jasus edwardsii* were more important than fish in controlling abundance of the sea urchin *Heliocidaris erthrogramma* (Pederson and Johnson 2006). Grazing by *H. erthrogramma* on the native fleshy macroalgae creates urchin barrens that allow the establishment of the introduced kelp *Undaria pinnatifida* (Johnson *et al.* 2004). Johnson *et al.* (2004) suggest that fishing of lobsters may account for increases in urchin densities. Urchin mortality is higher inside marine protected areas where predators, primarily lobsters, are more abundant than in adjacent fished areas.

Previous studies provide evidence of top down control of community structure by spiny lobster in rocky subtidal temperate ecosystems around the world. In contrast, there is little evidence to indicate that panulirids have the same defining effect on benthic community structure in tropical or soft sediment habitats. However, predation by spiny lobsters can reduce prey densities and alter prey size structure and species composition in seagrass and soft sediment habitats adjacent to dens where lobsters are aggregated but the effects are localized and diminish with distance from the den (Joll and Phillips 1984, Jernakoff 1987, Jernakoff *et al.* 1987, Edgar 1990). Worldwide the vast majority of previous research has focused on shallow water lobster populations. Similarly in Western Australia research effort has been focused on shallow inshore reefs. While juvenile *P. cygnus* are found in shallow inshore areas, the bulk of the adult population is in deeper water (>40m). *P. cygus* is one of the panulirid species that undertakes a substantial offshore migration as they reach sexual maturity, which is around the same time it reaches minimum legal size and is therefore subjected to substantial fishing pressure.

Therefore, there is a need to collect basic ecological information to determine if changes in lobster density and size structure due to fishing on the west coast of Western Australia has caused significant changes in habitat structure and benthic community composition in deep water. This research will provide information on the level of ecosystem impact of removing lobsters from deep-water habitats to improve the assessment of risk to the ecosystem to ensure that the western rock lobster fishery maintains MSC certification and complies with DEWHA requirements for export permits.

2.1   Reporting format

The majority of the research in this document has been, or is in the process of being submitted for publication in peer-reviewed scientific journals. The project objectives are outlined in Chapter 3.2 (see above). The links between project objectives and chapters of the final report are explained below and are also highlighted along with the main conclusions in the boxed section at the start of each chapter.

Chapter 5 is titled Assessment of the benthic biota of deep-coastal ecosystems associated with western rock lobster (*Panulirus cygnus*) populations along the temperate west coast of Australia. This paper partially addresses Objective 3 (To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water) by using two different techniques to assess benthic habitats at three coastal locations.

Chapter 6 is titled Abundance and size of western rock lobster (*Panulirus cygnus*) as a function of benthic habitat: implications for ecosystem based fisheries management (EBFM). This paper partially addresses Objective 1 (To identify gradients in the density/size distribution of western rock lobster to enable selection of representative areas) by using Independent Breeding Stock Survey (IBSS) data to identify gradients in the density and size of lobsters at three coastal locations. Secondly this paper addresses Objective 3 (To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water) by using IBSS data and towed video habitat data to determine patterns in the relationship between habitat type and lobster population dynamics at three coastal locations in the centre of the fishery.

Chapter 7 is titled Predicting western rock lobster (*Panulirus cygnus*) abundance and size based on habitat characteristics at Jurien, Western Australia. Similarly this chapter addresses the same objectives as Chapter 6 but uses two different methods to asses benthic habitat and focused at only one coastal site.

Chapter 8 is titled Diet and trophic position of western rock lobsters (*Panulirus cygnus* George.) in Western Australian deep-coastal ecosystems (35-60 m) indicates they are more carnivorous than their counterparts in shallow water ecosystems. This paper addresses Objective 3 (To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water) by determining the trophic role of western rock lobster in deepwater to determine the role lobsters may have in structuring benthic habitat and faunal assemblages.

Chapter 9 is titled Depletion-based shallow and deep-water estimates of catchability of western rock lobsters at the Abrolhos Islands and addresses Objective 2 (To assess the deep-water catchability of western rock lobsters and its relationship with population abundance and size structure).

3.0 Assessment Of The Benthic Biota Of Deep-Coastal Ecosystems Associated With Western Rock Lobster (*Panulirus Cygnus*) Populations Along The Temperate West Coast Of Australia

Waddington, K., Meeuwig, J., Evans, S. and Bellchambers L. M.

3.1 Summary

This study aims to characterise the benthic biota of deepwater ecosystems at three locations (Dongara, Jurien and Lancelin), central to the western rock lobster fishery, along coast of Western Australia to determine the importance of sponge and algal assemblages to western rock lobsters. The research contained in this chapter was initiated to addresses objective 1 which aimed to identify areas of representative habitat and lobster populations to enable the selection of representative areas on which to focus latter components of the research.

Differences in algal and sponge assemblages were detected between the study locations; Dongara is a sponge dominated system, Jurien Bay is a mixture of sponge and algae while Lancelin is algal dominated. Differences in algal and sponge abundance between the three locations have implications for western rock lobster. The presence of sponge and algae increases habitat complexity, which may lead to reduced predation and competition on lobsters or may provide additional nutrients and food sources for macroinvertebrates that are important dietary items for lobsters.

The current study provided detailed descriptions of benthic habitats at three coastal locations and highlights the importance of incorporating the full range habitats used by lobsters in future components of the larger study. This may be achieve either by using more than one study location (i.e. Dongara, Jurien and Lancelin) or by ensuring that both algal and sponge habitats are represented within the one location.

3.2 Abstract

Deep coastal ecosystems (35–75 m) along the temperate west coast of Australia are important habitats for adult populations of western rock lobsters (*Panulirus cygnus*). Knowledge of benthic biota in these areas is therefore important for understanding processes occurring in these ecosystems. The current study classified the benthic biota of these ecosystems using towed video and diver sampling, and compared the results provided by both methods of habitat classification. Deep-coastal ecosystems have significant algal and sponge assemblages, suggesting a reduction in irradiance with depth is not constraining algal distribution at these depths. Differences in sponge, algal, and macroinvertebrate community composition were detected at a regional scale between study locations. However, a direct link between sponge/ algal assemblage structure and macroinvertebrate community composition could not be established. Macroinvertebrates are important prey items for western rock lobsters (*Panulirus cygnus* George.), meaning differences in macroinvertebrate community composition will have implications for prey available to western rock lobsters. Both methods of classifying

assemblage structure yielded similar outcomes, suggesting a single method of classifying habitat can be employed in future studies to determine assemblage structure.

3.3   Introduction

The coastal ecosystems along the temperate west coast of Australia support a number of invertebrate fisheries including the socially and economically important western rock lobster (*Panulirus cygnus*). This region is oligotrophic and strongly influenced by the Leeuwin current – a low nutrient current that flows through this region (Cresswell, 1991; Johannes *et al.*, 1994; Hanson *et al.*, 2005). Given the presence of the Leeuwin current and low runoff of terrestrial water in this area (Li *et al.*, 1999), pelagic production is low and benthic production is important in supporting fisheries in this region (Lenanton *et al.* 1991). The abundance and size of western rock lobsters in this region has recently been shown to relate to the distribution of benthic assemblages (Bellchambers *et al.* In Press). Habitats dominated by sponges were demonstrated to contain small numbers of large lobsters, while habitats dominated by *Ecklonia* sp. were shown to contain large numbers of small lobsters (Bellchambers *et al.* In Press).

The temperate west coast of Australia is characterised by limestone reefs running parallel to the coastline between one and twenty kilometers offshore (Searle and Semeniuk, 1985). The biota occupying shallow water (<20 m) reefs in this area have been extensively described. Shallow water reefs are dominated by the kelp *Ecklonia radiata* (C. Agardh) J. Agardh (Phillips *et al.*, 1997; Wernberg *et al.*, 2003a; Kendrick *et al.*, 2004) which reach 1-2 metres in length (Wernberg *et al.*, 2003b) and form extensive areas of habitat termed kelp beds (Steinberg and Kendrick, 1999). Other algae also occur on these reefs, both associated with *E. radiata* and in distinct patches (Kendrick *et al.*, 1999; Kendrick *et al.*, 2004, Vanderklift and Kendrick, 2004). Algal assemblages support significant macroinvertebrate (0.5-20 mm) assemblages including various crustaceans, molluscs, polychaetes, and echinoderms (Edgar, 1990; Edgar and Shaw, 1995). As these species are commonly consumed by higher order consumers such as fish and lobsters (Joll and Phillips, 1984; Edgar and Shaw, 1995), it is likely algal assemblages and associated macroinvertebrate fauna are important to coastal food webs.

The composition of benthic assemblages at depths greater than 20 metres in this region remains largely unknown. Irradiance is reduced as water depth increases, which may affect macroalgal communities (Kirkman, 1989). An increase in depth may result in a decrease in wave action (Brey, 1991), also potentially leading to differences in benthic algal assemblages (Molloy and Bolton, 1996; Phillips *et al.*, 1997). As deep-coastal ecosystems yield significant catches of adult western rock lobsters, benthic production is likely significant at these depths. Sources of benthic production may be from *in situ* primary production, from benthic-pelagic coupling (eg. Graf, 1989; Waite *et al.*, 2000), or from import of detrital material (eg. Wernberg *et al.* 2006). As lobsters reside in deep-coastal ecosystems sources of benthic production must be sufficient to support the high level of lobster production at these depths; over 40% of total lobster catches (~4000 tonnes) are taken from the deep-coastal ecosystems that are the focus of this study (unpublished catch and effort statistics, Department of Fisheries Western Australia). Indeed, recent dietary studies (Waddington *et al.* 2008) have demonstrated that lobsters prey on a variety of macroinvertebrates associated with these deep-coastal ecosystems.

One of the reasons that deep-coastal ecosystems are less well known is that they are logistically difficult and expensive to sample compared to shallow water ecosystems. Detailed abundance data and ecological samples must be collected using diver based samples as small macroinvertebrate species on hard bottom communities are not amenable to collection using

trawl or sled. Due to the depth of these ecosystems, diver based sampling must be undertaken using commercial divers, which makes collection of relatively few ecological samples expensive. Remote sampling using towed video systems or AUV provides an alternative but is restricted in that ecological samples are not provided and the level of taxonomic resolution is reduced. An important component of the current study is to investigate the degree to which information is lost if one method of classifying habitat is chosen over another – i.e. to what degree can one method of habitat classification act as a surrogate for the other method of habitat classification. Surrogates have been extensively used to estimate species diversity from occurrence of higher taxonomic levels (Gaston and Williams, 1993; Andersen 1995; Cardoso *et al.*, 2004). The rationale of this approach is if patterns in diversity at the species level can be captured at higher taxonomic levels, costs associated with collecting and processing samples can be reduced (Andersen, 1995).

This study aims to characterise the benthic biota of deep coastal ecosystems at three locations along the temperate west coast of Australia as a basis for understanding productivity of these deep coastal systems. Specifically, this study will compare two methods used to classify benthic sponge and algal assemblages – towed video and diver sampling. Here, the applicability of one method as a surrogate for the other will allow investigation of whether the two methods of habitat classification provide comparable data, and the degree to which information is lost if one method alone is employed to classify sponge and algal assemblages. To determine the likely importance of benthic sponge and algal assemblages to western rock lobsters, the relationship between benthic sponge and algal assemblages and those macroinvertebrates known to be important prey items for western rock lobsters will be investigated.

3.4   Materials and methods

3.4.1   Study locations

Biota of deep-coastal ecosystems was classified at three locations Lancelin, Jurien Bay, and Dongara (Figure 1). These locations span 200 km of the Western Australian coastline near the center of the distribution of western rock lobsters. Habitat at each location was assessed using two techniques. Towed video was used to classify habitat on a broad scale (kilometres), while diver-harvested quadrats were used to classify habitat on a fine scale (metres). Further, the composition of benthic macroinvertebrate communities was determined from the diver-harvested quadrats. These two techniques vary in both the scale of the sampling unit and the metric of classification. Each dive site was paired with a nearby towed video transect, allowing the applicability of one method to act as a surrogate for the other to be examined.

3.4.2   Determining broad-scale habitat characteristics using towed video

Broad-scale habitat classification using towed video was carried out between March 2005 and May 2007. Between 8 and 11 transects (depths 35 m to 75 m) ranging in length from 1075 m to 3725 m (mean ~2500 m) were surveyed at each of the three locations. An underwater video apparatus, consisting of one forward and one downward facing video camera was towed at speeds of 1-2 knots between one and three metres above the substratum. Footage was recorded to video and classified in the laboratory. To classify habitat, video was paused every 25 metres and substratum in the field of view of the downward facing camera (~1 m diameter) was identified, providing a series of habitat snapshots along the transects. Habitat was classified on

the basis of assemblage type and coverage, using the categories described in Table 1. For bare reef, rubble/limestone, and sand no measure of habitat coverage was recorded.

3.4.3　Data analysis

The method of habitat classification employed provided a description of assemblage type and coverage (eg. 1a, 4c, 5b) at various points along each transect. The proportional contribution of each habitat category along each transect was then calculated. Data were analysed using the PRIMER v5® statistical package. To account for transects of differing lengths, habitat categories were standardized to percent coverage along a transect. Data were then square root transformed to reduce the influence of dominant habitat classes and a Bray Curtis similarity matrix calculated. Non metric MDS plots were constructed for sponge/ algal assemblages. Constrained ordinations were then plotted using Bray Curtis similarity and untransformed data, allowing the dispersion of data points to be investigated. Differences in assemblage structure between sites and locations were investigated using analysis of similarity (ANOSIM), while species responsible for driving observed differences in assemblage structure were identified using similarity percentages (SIMPER).

3.4.4　Determining fine-scale and macroinvertebrate characteristics by diver sampling

Habitat samples were collected by divers for fine-scale classification during March and April 2006. Between four and five sites (35-60 m depth) were selected for sampling at each of the three locations. Divers breathing mixed gas (Enriched Air Nitrox, Trimix) from SCUBA collected habitat samples at each site. When collecting habitat samples, the entire contents of a 0.25 m^2 quadrat were removed using a paint scraper and placed in a calico bag, ensuring no material was lost (n=2 sub-samples at each site at Dongara and Jurien Bay and n=3 sub-samples at each site at Lancelin). In the laboratory, all collected material was sorted to lowest possible taxonomic level then weighed, yielding the biomass of alga, sponges, and macroinvertebrates per unit area.

3.4.5　Data analysis

Diver collection of samples provided the biomass of different benthic groups at various sites and locations. Biomass data were square root transformed and Bray Curtis similarity coefficients calculated. Non metric MDS plots were constructed separately for sponge/ algal assemblages and macroinvertebrate community composition. Further, constrained ordinations were plotted using Bray-Curtis similarity and untransformed data, allowing the dispersion or "shape" of these data to be investigated. Differences in sponge/algal assemblage structure and macroinvertebrate community composition depending on the factors site and location were investigated using ANOSIM. Species responsible for driving differences in assemblage structure were identified using SIMPER. Following analysis of community composition using multivariate techniques, taxa from diver sampling were aggregated into functional groups (algae, sponges, macroinvertebrates) and biomass of functional groups compared for the factors site and location using two-way analysis of variance (ANOVA).

3.4.6　Comparison of broad-scale and fine-scale methods of habitat classification

Following habitat classification using broad-scale and fine-scale techniques, these two methods of habitat classification were compared. While the same locations were sampled

by both methods, they can be considered independent as sampling within locations was not at the same sites, and sampling occurred at different scales and used different variables (e.g. as relative occurrence of habitat type for broad-scale and biomass of species for fine-scale). This is analogous but not identical to surrogacy studies that ask whether a location can be characterized by one set of variables (species) or another (family or order) (Andersen, 1995; Phillips et al. 1997; Olsgard and Somerfield, 2000), although these variables are typically collected from the same sites within locations.

Conventional methods of comparing methods of classification require a correspondence between samples (eg. BIOENV, Relate analyses; Whitman *et al.*, 2004) or variables (eg. 2nd Stage MDS; Clarke *et al.*, 2006). As such, these methods were not applicable for comparison of these two datasets. As an alternative approach, we built on the concepts behind these techniques by asking the question "To what degree do the spatial distribution of samples correspond between the two techniques?" e.g. Are the Lancelin samples similarly spatially distributed relative to each other and the samples from the other location in space regardless of the technique used? Thus, a comparison of broad-scale and fine-scale methods for habitat classification was based on the overlap of the constrained ordination for both datasets. For each location, the dive site was coupled with an adjacent video transect. Each of these areas was then treated as a "site" with respect to method of habitat classification. PERMANOVA was used to test for differences in the distribution of points according to site and method of habitat classification. Site was treated as a random factor with 12 levels while method of classification was treated as a fixed factor with two levels. The dependant variables were the x, y - coordinates of the sampling points on the constrained ordination. This analysis gave an indication of whether the points differed in their mean position by site. As the scale of constrained ordinations is determined by the maximum dissimilarity between two data points, differences in assemblage structure between each site or method of classification can be compared to this maximum dissimilarity of points in the data set. This gives a relative measure of differences in assemblage structure between sites or method of classification.

3.5   Results

Deep-coastal reef ecosystems at the three locations support considerable biomasses of sponges and algae. Biomass of sponge and algae at the three locations were between 1.2 and 2.2 kg 0.25m^{-2}. Macroinvertebrate biomasses of between 2.5 and 3.0 g.0.25m^{-2} were also observed, with polychaetes, small crabs and amphipods the most commonly encountered invertebrate fauna. Sponge/ algal assemblages and invertebrate community composition differed between locations.

3.5.1   Broad-scale patterns in sponge/algal assemblage structure determined from towed video

Differences in assemblage structure were apparent from towed video data (Figure 2). Significant differences in sponge and algal assemblages were observed between locations based on Analysis of Similarity (ANOSIM) tests (Clarke's R = 0.55, p = 0.001, permutations = 999). Pairwise tests indicated these differences were apparent between all locations (Table 2). Similarity percentages analysis indicated it was primarily the high occurrence of habitat categories containing sponge fauna at Dongara responsible for observed differences in assemblage structure between study locations. Between study locations, it is low occurrence of mixed assemblage with *E. radiata*, no sponge (class 4b) and mixed assemblage, no *E. radiata*, no sponge (class 3b) along with the higher occurrence of mixed assemblage with sponge, no

E. radiata (class 2c) contributing to observed differences between Dongara and the other two locations. Differences in assemblage structure between Lancelin and Jurien Bay were primarily driven by high occurrence of mixed assemblage with *E. radiata*, no sponge (class 4b) at Lancelin relative to Jurien Bay, reflecting the lower occurrence of sponge at Lancelin relative to Jurien Bay.

3.5.2 Fine-scale patterns in sponge/algal assemblage structure determined by diver sampling

A range of taxa were identified from samples collected by divers. Collected samples were dominated by sponges at Dongara, while the kelp *Ecklonia radiata* accounted for a high proportion of sample biomass at Lancelin. Jurien Bay had high biomass of both *E. radiata* and sponges. Across the three locations 149 sponge species, 34 red algal species, 8 coralline algal species, 5 green algal species, 7 brown algal species and a single seagrass species (*Thalassodendron pachyrhizum* Hartog) were recorded.

Collection of biota using divers allowed species biomass to be assessed on a fine-scale (e.g. metres). ANOSIM tests detected differences in algal and sponge assemblages between locations (Figure 3) (Clarke's R = 0.47, p = 0.001, permutations = 999) with pairwise tests indicating these differences were apparent between all locations (Table 2). Differences in assemblage structure were also apparent between sites within locations (Lancelin, Clarke's R = 0.63, p = 0.002, permutations = 9999; Jurien Bay, Clarke's R = 0.67, p = 0.004, permutations = 9999; Dongara, Clarke's R = 0.58, p = 0.008, permutations = 945).

SIMPER analysis revealed high biomass of three sponge species (Class Calcarea, Order Clathrinida, *Clathrinida* sp.; Class Demospongiae, Order Dictyoceratida, Family Irciniidae, *Sarcotragus* sp.; and Class Demospongiae, Order Dictyoceratida, Family Thorectidae, *Cacospongia* sp.) and low biomass of the kelp *Ecklonia radiata* at Dongara relative to Jurien Bay and Lancelin were responsible for driving observed differences in assemblage structure between Dongara and Lancelin/Jurien Bay.

3.5.3 Macroinvertebrate community composition determined by diver sampling

Macroinvertebrate community composition was also assessed from diver samples. Across the three locations, 20 families of macroinvertebrates and 27 separate species of sessile invertebrates (excluding sponges) were identified. The invertebrate groups, echinoderms, crustaceans, molluscs, polychaetes, sipunculids, bryozoans, ascidians, and corals were all represented in samples. Of the macroinvertebrate fauna, crabs (<15 mm carapace width), polychaetes, and amphipods were most commonly encountered.

The invertebrate community composition did not differ between sites within locations (Clarke's R = 0.17, p = 0.09, permutations = 999), but did differ between locations (Figure 4) (Clarke's R = 0.28, p = 0.001, permutations = 999). Pairwise tests indicated all locations significantly differed in macroinvertebrate community composition (Table 2). SIMPER analysis indicated it was higher abundances of decapod crustaceans (excluding crabs) and the hammer oyster (*Malleus meridianus*) at Dongara relative to Lancelin and Jurien Bay driving observed differences in invertebrate fauna between Dongara and the other study locations. Higher abundances of crabs and gastropods and lower abundances of amphipods at Jurien Bay relative to Lancelin are responsible for driving the differences in invertebrate fauna observed between these locations.

3.5.4 Comparison of total sponge, algal and macroinvertebrate abundances

Taxa were combined into functional groups (algae, sponges and invertebrates) and total biomass of functional groups compared on the basis of site and location. No difference in algal biomass was detected between sites within locations (Table 3). However, significant differences in algal biomass were detected between locations (Table 3; Figure 5). Further analysis using post hoc Tukey tests indicated algal biomass was significantly higher at Lancelin relative to Dongara ($p=0.009$). No difference in algal biomass were apparent between Lancelin and Jurien Bay ($p=0.792$), or Jurien Bay and Dongara ($p=0.069$).

Sponge biomass did not significantly differ between sites within locations (Table 4), however significant differences in sponge biomass were observed between locations (Table 4; Figure 5). Post hoc Tukey tests indicated that sponge biomass at Dongara was significantly higher than sponge biomass at Lancelin ($p=0.043$). No difference in sponge biomass was apparent between Dongara and Jurien Bay ($p=0.99$) or Lancelin and Jurien Bay ($p=0.07$).

No differences in macroinvertebrate biomass were observed between sites within locations or between locations (Table 5; Figure 6).

3.5.5 Comparison of broad-scale and fine-scale methods of habitat classification

Both methods of habitat classification were found to provide similar results with respect to differences in algal and sponge assemblages between locations. Both methods identified Dongara as a sponge dominated system, Jurien Bay as sponge and algal dominated and Lancelin as algal dominated. Figure 7 shows the overlap of constrained ordinations from the broad-scale classification (white symbols, n=12) and the fine-scale classification (black symbols, n=12). A PERMANOVA on the x and y coordinates demonstrates that the points on the ordination plots are significantly different according to site (Table 6) but not method of classification (Table 7). This indicates that the relative position of sites on the constrained ordinations is consistent between methods of analysis.

3.6 Discussion

Deep-coastal reef ecosystems along the temperate west coast of Australia support significant algal assemblages. These assemblages, dominated by *Ecklonia radiata*, are comparable in biomass and composition to shallow water assemblages in this region (Kendrick *et al.*, 1999; Wernberg *et al.*, 2003a). The occurrence of algal assemblages of similar composition and biomass to those encountered in shallow water indicates irradiance does not constrain the distribution of *E. radiata* and other algae to depths shallower than 60 metres. Observed biomasses and coverage of *E. radiata* and associated algae suggests macroalgae are the dominant source of primary production in these deep-coastal ecosystems. In contrast, detached macroalgae were rarely observed during sampling and are unlikely to be a significant source of secondary production in these systems, unlike in shallow water systems (Wernberg *et al.* 2006). Sponges were also frequently observed in deep-coastal ecosystems, particularly at Dongara and Jurien Bay, suggesting benthic pelagic coupling (eg. Graf, 1989; Waite et al. 2000) may contribute to observed secondary production of these deep-coastal ecosystems. However, further studies are required to establish the strength of any such link between the pelagic and benthic zones.

Differences in algal and sponge assemblages were detected between the study locations. These differences in habitat assemblages were driven by high biomasses and occurrence of sponge at Dongara relative to Lancelin where macroalgae was more abundant. Differences in macroalgal assemblage structure have been observed on comparable scales in shallow water ecosystems (Wernberg *et al.*, 2003a). Few other studies have used biomass as a metric for describing sponge assemblages in temperate sub-tidal systems, with coverage and species richness more commonly employed. However, a study by McQuillan (2006) determined that sponges occupy between 30-50% of space on limestone reefs in shallow water ecosystems (<12 m) in nearby Marmion Lagoon (31° 44' S, 115° 40' E). This is comparable to sponge coverage observed in the current study.

Differences in algal and sponge abundance between the three locations have implications for associated macroinvertebrate fauna. Taxa that are important in the structuring of marine ecosystems, and have implications for other species within ecosystems are termed ecosystem engineers (Jones *et al.*, 1994; Lawton and Jones, 1995). The presence of sponge and algae increases habitat complexity, available space and nutrients for associated macroinvertebrate fauna. A positive correlation between habitat complexity and invertebrate abundance has previously been demonstrated for a number of systems (Heck and Orth, 1980; Robertson and Lenanton 1984; Jernakoff and Nielsen, 1998). This positive relationship may be driven by specific resource requirements of invertebrate fauna or may be influenced by other factors such as predation and competition (Heck and Orth, 1980; Langlois *et al.*, 2005; Pederson and Johnson, 2006). As the current study is descriptive in nature, a direct link between variation in sponge and algal assemblages and macroinvertebrate abundance cannot be established. However, differences in macroinvertebrate community composition may have implications for food available to higher order consumers such as the western rock lobster. A recent study of lobster diet in these deep-coastal areas demonstrated that differences in western rock lobster diet between sampling locations largely reflected differences in prey availability (Waddington et al. 2008). Differences in prey availability has also been shown to affect the diet of western rock lobsters in shallow water ecosystems (Joll and Phillips, 1984; Edgar 1990) and among other spiny lobster species (Barkai and Branch, 1988; Mayfield *et al*. 2000). Observed differences in macroinvertebrate community composition between locations observed in the current study will influence prey available to lobsters between study locations. Those taxa that occur in high abundance (eg. polychaetes, decapods, amphipods/isopods) will likely be more important to lobster diet than taxa infrequently observed on the benthos such as sea urchins and sipunculids.

3.6.1 Towed video as a surrogate for diver sampling in classification of sponge and algal assemblages

While surrogacy approaches have been used to determine the extent to which information is lost if species are classified at higher taxonomic levels (Gaston and Williams 1993; Andersen 1995; Cardoso *et al.,* 2004), to our knowledge this is the first study to examine the degree to which one method of habitat classification may act as a surrogate for a different method of habitat classification. Results indicate towed video provides a reliable surrogate to diver sampling when classifying sponge and algal assemblages in temperate Western Australian ecosystems. Method of classification had no determinable influence on classification of sponge and algal assemblages, and the differences between locations were preserved. The observed correlation between classification methods indicates fine-scale patterns in sponge and algal assemblages can be inferred from broad-scale patterns determined by towed video. Towed video is not depth limited, is cost-effective (relative to diver sampling), and allows classification of large

areas of habitat cheaply, making it an effective technique for classifying sponge and algal assemblages. Diver sampling is a useful measure when sample collection and examination of macroinvertebrate community composition is desired (such as for trophodynamic studies). Since these two techniques of habitat classification are highly correlated, techniques employed to classify sponge and algal assemblage structure in future studies should also reflect the broader aims of the relevant research projects.

3.6.2   Conclusions

Deep-coastal ecosystems in temperate Western Australia support significant sponge and algal assemblages. Significant biomasses of macroinvertebrates were also observed to be associated with these habitats which are likely to be productive enough to support observed secondary production in this region (Waddington and Meeuwig 2009). While the current study was largely descriptive, it provides a valuable starting point for further investigation of processes occurring in these deep-coastal ecosystems, particularly those involving the commercially important western rock lobster. The two methods of habitat classification employed in this study were also determined to be comparable for classifying sponge and algal assemblages. Thus, one method may be sufficient in the future when classifying benthic habitats.

3.7   Tables

Table 1.   Categories used to classify habitat assemblage type.

Habitat code	Habitat type	Coverage Code	Coverage	Coverage
1	Mixed assemblage with *Ecklonia radiata* and sponge	a	High	>70%
2	Mixed assemblage with sponge/ no *Ecklonia radiata*	b	Medium	30-70%
3	Mixed assemblage no *Ecklonia radiata* no sponge	c	Low	<30%
4	Mixed assemblage with *Ecklonia radiata* no sponge			
5	Brown algae (no *Ecklonia radiata*)			
6	Reef structure with no visible algae			
7	Rubble/ limestone with no visible algae			
8	Sand			

* Coverage refers to biotic coverage estimated within field of view.

Table 2.   Pairwise comparisons showing differences in assemblage structure between locations by towed video, diver surveys and in macroinvertebrate assemblages. Results considered significant at $p<0.05$ and are based on 999 permutations.

Comparison	Towed Video		Diver		MacroInvert	
	Clarke's R	p-value	Clarke's R	p-value	Clarke's R	p-value
Lancelin, Jurien Bay	0.46	0.002	0.24	0.014	0.16	0.04
Lancelin, Dongara	0.82	0.001	0.68	0.001	0.41	0.001
Jurien Bay, Dongara	0.36	0.002	0.41	0.001	0.214	0.002

Table 3.   Two-way ANOVA testing the effect of the factors location and site on algal biomass.

Source of Variation	df	SS	MS	F value	p-value
Location	2	7.82	3.91	5.89	0.009
Site	4	7.35	1.84	2.77	0.052
Residual	23	15.27	0.66		
Total	29	32.43	1.12		

Table 4.   Two-way ANOVA testing the effect of the factors location and site on sponge biomass.

Source of Variation	df	SS	MS	F value	p-value
Location	2	0.31	0.15	4.09	0.03
Site	4	0.15	0.04	0.99	0.432
Residual	23	0.86	0.04		
Total	29	1.25	0.04		

Table 5. Two-way ANOVA comparing total biomass of invertebrates between sites and locations.

Source of Variation	df	SS	MS	F value	p-value
Location	2	0.03	0.01	0.12	0.88
Site	4	0.08	0.02	0.18	0.94
Residual	23	2.56	0.11		
Total	29	2.67	0.09		

Table 6. PERMANOVA comparing differences in the co-ordinates of points on constrained ordinations depending on the factors location and site. Site nested within location. p-values were generated from 999 permutations of raw data.

Source of Variation	df	SS	MS	Pseudo F	p-value
Location	2	5114	2557	5.44	0.006
Site (Location)	9	4267	470	1.20	0.34
Residual	12	4680	390		
Total	23	14021			

Table 7. PERMANOVA comparing differences in the co-ordinates of points on constrained ordinations depending on the factor method of classification. p-values were generated from 999 permutations of raw data.

Source of Variation	df	SS	MS	Pseudo F	p-value
Method of Classification	1	0	0	0	1
Residual	22	14021	637.3		
Total	23	14021			

3.8   Figures
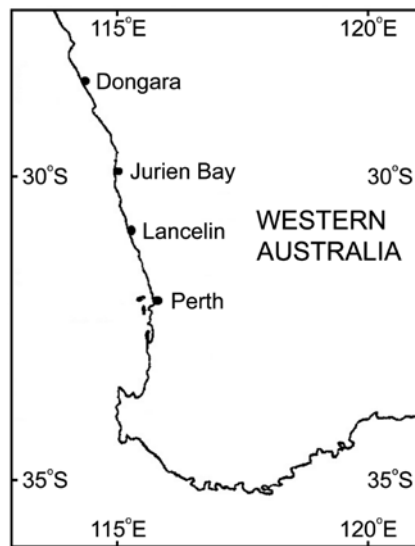
Figure 1.   Map of locations investigated in this study relative to Perth, Western Australia.
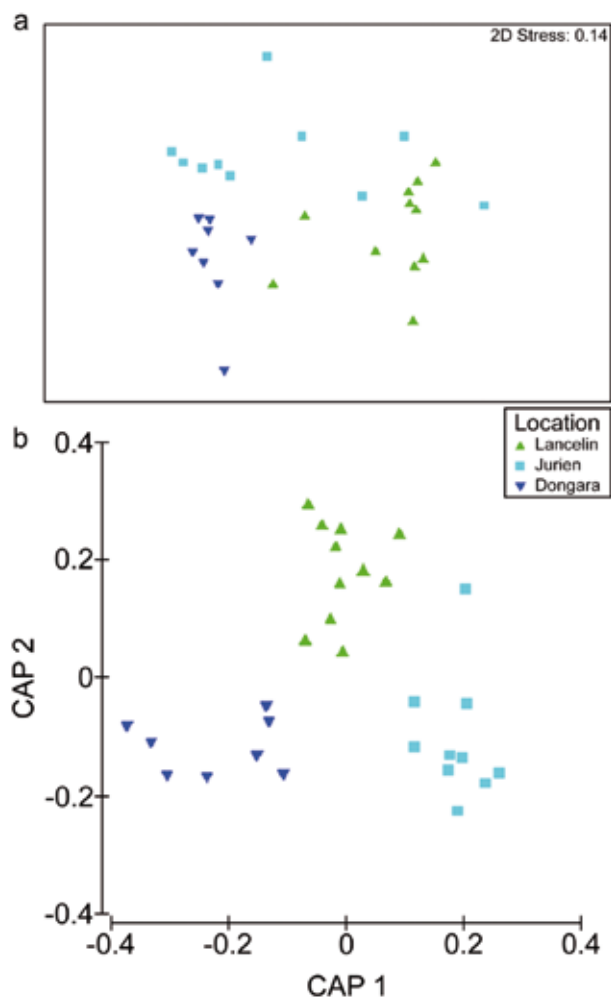
Figure 2.   (a) MDS plot (calculated from square root transformed data) and (b) constrained ordination (untransformed data) of sponge and algal assemblages at the three study locations determined by towed video. Similarity determined using Bray-Curtis coefficient.
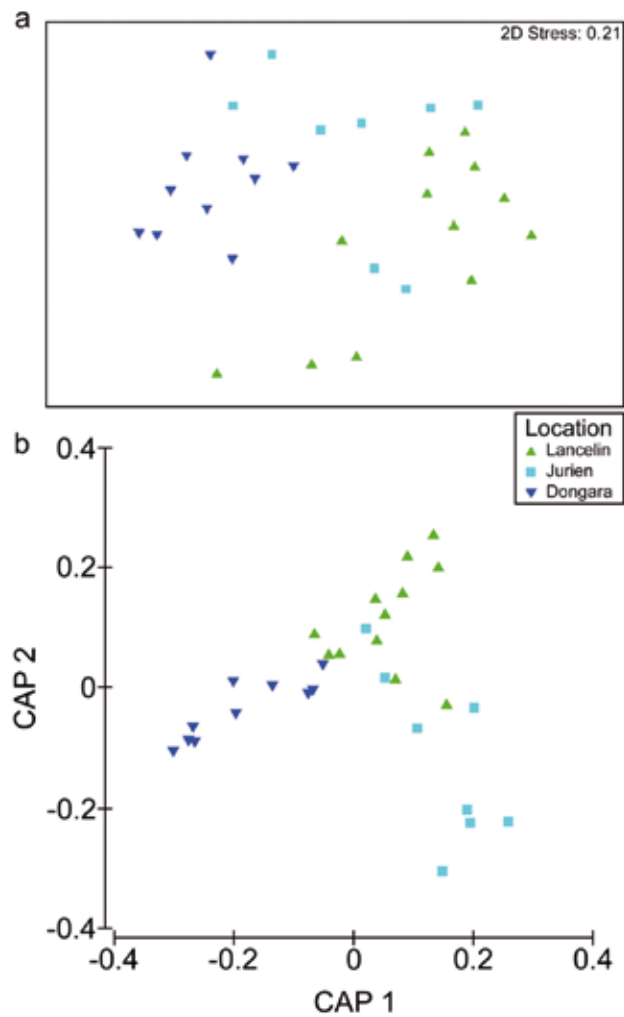
Figure 3. (a) MDS plot (calculated from square root transformed data) and (b) constrained ordination (untransformed data) of sponge and algal assemblages at the three study locations determined by diver sampling. Similarity determined using Bray-Curtis coefficient.
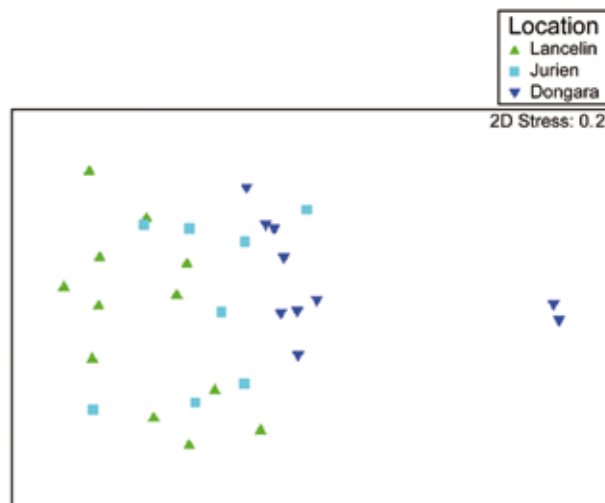
Figure 4. MDS plot of invertebrate community composition at Lancelin, Jurien Bay and Dongara. Data were square root transformed and similarity determined using Bray-Curtis coefficient.
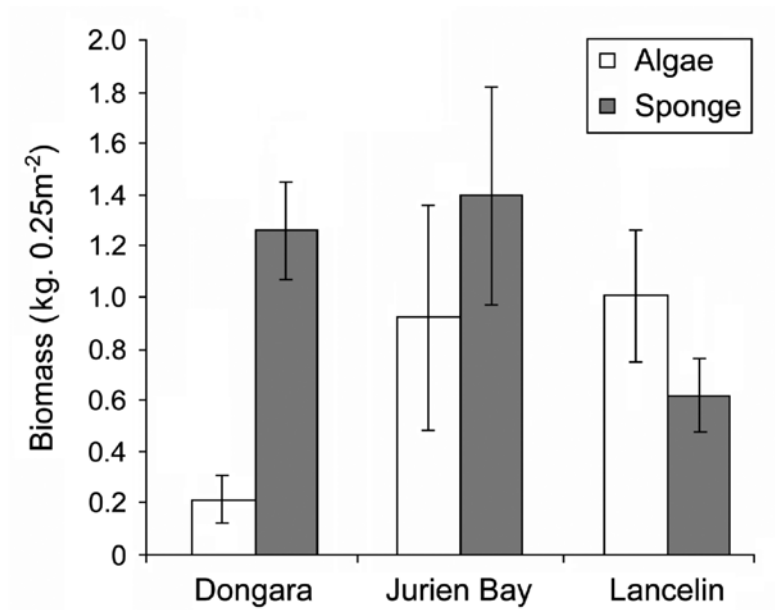
Figure 5. Mean biomass (± se) of algae and sponge at the three study locations.
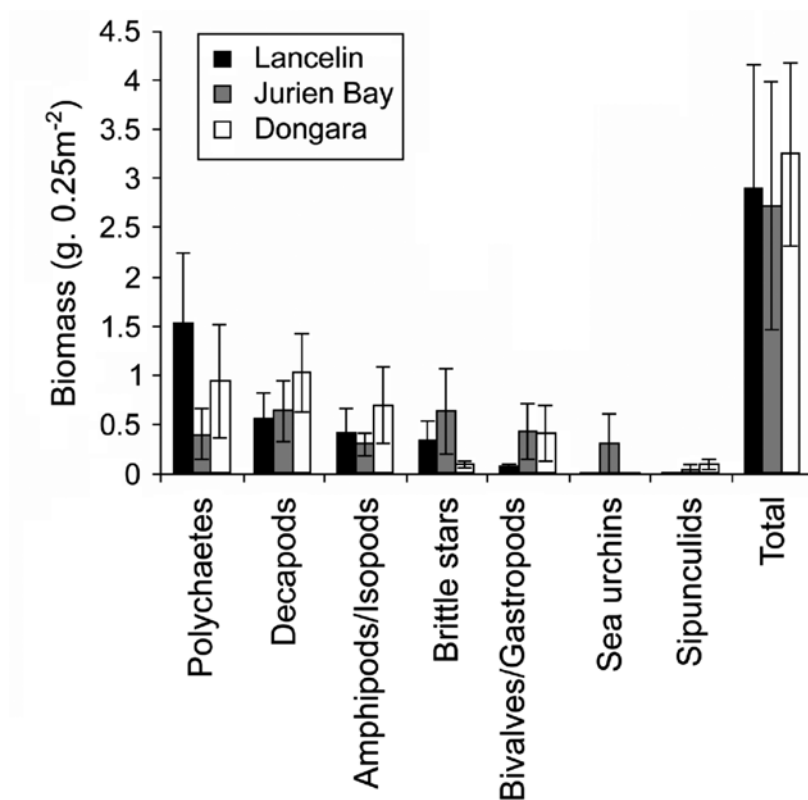
Figure 6. Mean biomass (± se) of macroinvertebrate groups at the three study locations.
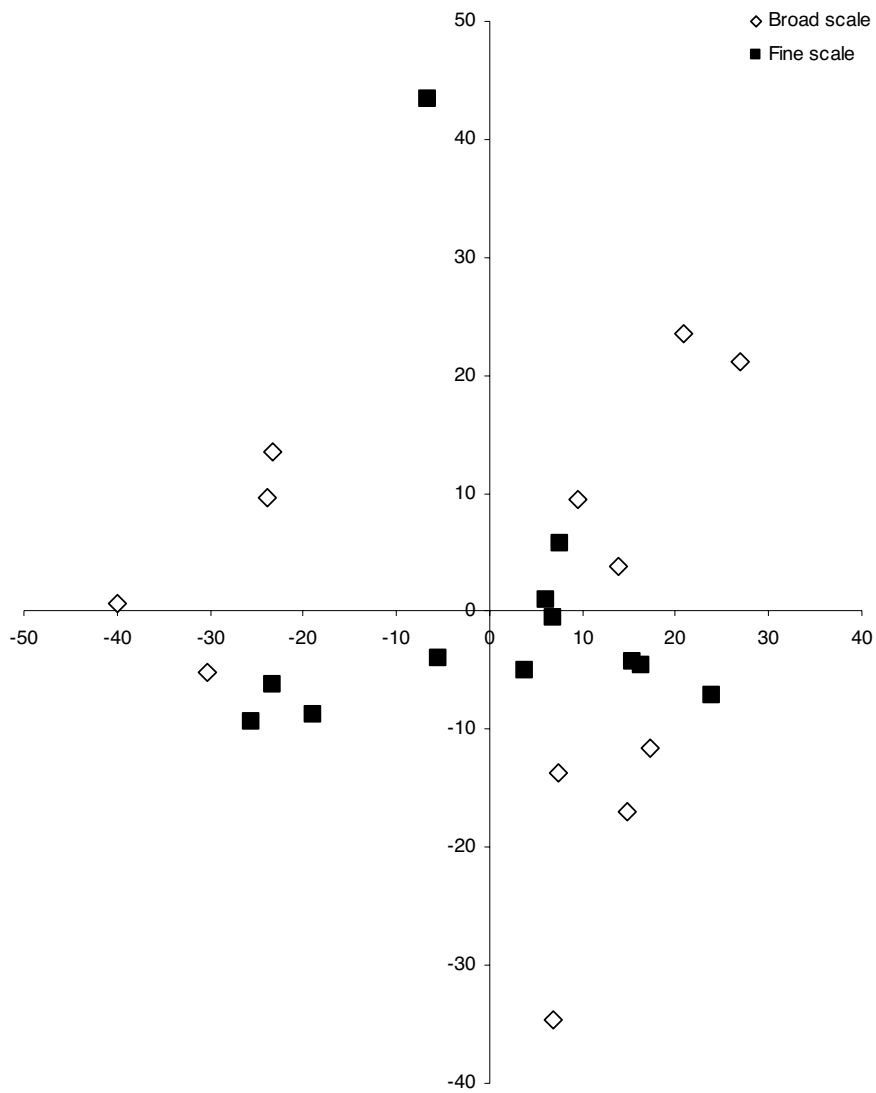
Figure 7. Relative distribution of points from constrained ordination plots.

4.0 Abundance and size of western rock lobster (*Panulirus cygnus*) as a function of benthic habitat: implications for ecosystem based fisheries management (EBFM)

Bellchambers, L. M., Evans, S. N. and Meeuwig J. J.

4.1 Summary

This study aims to characterise the relationship between deepwater habitats and the abundance and size of lobster at three locations (Dongara, Jurien and Lancelin) central to the western rock lobster fishery. The research contained in this chapter was initiated to address Objectives 1 and 3. Objective 1 aimed to identify gradients in the density/size distribution of western rock lobster to enable selection of representative areas. While Objective 3 aimed to identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water. Towed video transect were used to assess benthic habitat types while data from the Independent Breeding Stock Survey (IBSS) was used to determine the size and abundance of lobsters. This study found that both the type and amount of habitat varied significantly between the three locations with Dongara dominated by sponges and Lancelin dominated by macroalage. There was a relationship between habitat type and lobster size and abundance with low numbers of larger lobsters present in areas associated with sponges and high numbers of smaller lobsters present in more structurally complex habitats dominated by *Ecklonia sp.* These results indicate that different habitats may be important to different life stages which highlights the importance of any future study incorporating either a range of study locations or ensuring that each study site encompasses the full range of habitat types i.e. sponge, sand, macroalgae. This information is an important first step in assessing the habitat specific effects of biomass removal of target species by fishing.

4.2 Abstract

Ecosystem-based fisheries management (EBFM) requires the expansion of fisheries research programs to include the relationship between target species and their habitats such that trophic and other ecological interactions can be assessed. The western rock lobster (*Panulirus cygnus*) is an ecologically important species that supports Australia's most valuable single-species fisheries. We tested the relationship between abundance and size of western rock lobster and benthic habitats based on the annual independent breeding stock survey (IBSS) and benthic towed video transects. The work was undertaken at Dongara, Jurien Bay and Lancelin, Western Australia between 2005 and 2007. Abundance of western rock lobster was significantly but moderately related to benthic habitat (adjR^2=0.28), with high abundances associated with high cover of mixed assemblage and *Ecklonia sp.* Size was effectively predicted by habitat (adjR^2=0.65) with larger lobsters found in mixed assemblages with sponge and smaller lobsters associated with mixed assemblage with *Ecklonia sp.* Our study has shown that understanding the influence of habitat and fishing pressure on the abundance and size of targeted species is a critical step in the effective implementation of EBFM.

4.3   Introduction

Fisheries worldwide are typically managed on a stock-by-stock basis, with varying degrees of consideration given to by-catch and other environmental concerns such as habitat destruction (Jennings 2004, Fletcher 2007). However, overfishing and declining world fisheries have led to debate over the effectiveness of single-stock management and its inability to take into account social, economic and ecological factors (Link 2002; Jennings 2004; Zeller and Pauly 2004). Ecosystem-based fisheries management (EBFM) has emerged as a more comprehensive approach to natural resource management, as it deals with not only the wider ecological impacts of fishing at the spatial scale of an ecosystem but also the social and economic aspects of fishing activities (Fletcher 2007; Smith *et al*. 2007).

While the need for ecosystem-based approaches to fisheries management is recognised (Zeller and Pauly 2004; Field and Francis 2006), implementation has been slow due to a lack of agreement on the best approach and the associated increased data needs (Mace 2004; Smith *et al*. 2007). Marine ecosystems are complex and often poorly understood, with lack of knowledge one of the main obstacles to effective decision making (Mace 2004). The successful implementation of EBFM would require the expansion of fisheries-related research programs to include consideration of habitat characteristics, environmental variables, trophic interactions and non-target species (Link 2002; Link *et al*. 2002). Knowledge of the spatial distributions of marine populations and their interactions with their environment is also important in the assessment of fishing resources and in implementing effective management strategies (Lipcius *et al*. 1998; Acosta 1999; Rios-Lara *et al*. 2007). Specifically, greater knowledge of preferred habitats throughout a species' geographic range and life history can assist in understanding variability in its distribution and aid in designing more effective management strategies (Mai and Hovel 2007; Rios-Lara *et al*. 2007).

The western rock lobster, *Panulirus cygnus* George, lives along the west coast of Australia from the North West Cape south to Cape Leeuwin. As the most abundant benthic consumer in a primarily oligotrophic system, western rock lobsters are ecologically important (Lenanton *et al*. 1991). They are also the target of substantial commercial and recreational fisheries: the West Coast Rock Lobster Fishery (WCRLF), which fishes with baited pots between Shark Bay and Cape Leeuwin, has an annual production of around 11,000 t valued at US$200 million. It is Australia's most valuable single-species fishery and is generally considered a well-managed fishery (Phillips and Melville-Smith 2005). The recreational fishery landed 214 tonnes in 2005/06, a figure that has been growing in recent years with the increase in recreational fishers (Phillips and Melville-Smith 2005).

Most western rock lobster are targeted when they migrate offshore as they approach sexual maturity, at which stage they reach legal minimum size (Phillips and Melville-Smith 2005). Therefore, increasing the understanding of the deep-water ecology of this species is essential not only to ensure sustainability of the species but also to ensure ecosystems are healthy and stocks are managed in a more sophisticated, ecosystem-based manner. As a first step to increase our understanding of the deep-water ecology of western rock lobster, the present study tests for the presence of relationships between western rock lobster abundance and size, and the habitats in which they are found.

4.4 Methods

4.4.1 Study locations and the Independent Breeding Stock Survey (IBSS)

Patterns in western rock lobster abundance and size and their associations with different habitats were studied at three locations: Lancelin (115º 00'E, 30º 57'S), Jurien Bay (114º 44'E, 30º 15'S), and Dongara (114º 33'E, 29º 20'S) (Figure 1). These locations, which span 200 km of the Western Australian coastline near the centre of the western rock lobster fishery, are also within the area surveyed by the western rock lobster independent breeding stock survey (IBSS). This annual, standardised, ten-day fishery independent survey has been conducted at five coastal and one offshore location in depths between 36 and 73 m since the early 1990s. Its aim is to assess the state of breeding stock without the bias of changes in fishing industry efficiency and fishing regulations (Melville-Smith *et al*. 1998).

Each year, the IBSS deploys standard commercial-size pots with closed escape gaps from chartered commercial western rock lobster vessels. At each IBSS location, 800 pots are deployed in lines of 16 to 20 pots, each with its own GPS coordinate so the locations are consistently sampled between years (Figure 1). The data collected include the number of lobsters per pot, carapace length in millimetres, sex, reproductive state and general condition of lobsters caught in each pot. Throughout this paper, CPUE is used as a proxy for lobster abundance, and carapace length as a proxy for lobster size.

The Jurien IBSS location, which is in the centre of the western rock lobster fishery, is approximately 27-37 kilometres offshore from Jurien. The 800 pot sites are in five subregions, with each subregion containing ten lines with 16 pots per line (i.e. 50 lines in this location). The Dongara and Lancelin IBSS locations are 27 and 46 kilometres offshore from their respective ports. At these locations, 800 pot sites are in four subregions, each of which contains ten lines of 20 pots per line (i.e. 40 lines in each location). In this study, each subregion was analysed line-by-line so mean lobster catch rates both within and between subregions could be compared.

4.4.2 Habitat classification by towed video transects

The habitat was classified from towed videos conducted between March 2005 and May 2007 along IBSS lines. Between 8 and 15 habitat transects running roughly from north to south were videoed at each of the three locations (Figure 1). GPS co-ordinates from the IBSS were used so that the benthic habitat identified in each video transect could be directly correlated to the corresponding abundance and size frequency of lobsters in the IBSS pots. The transects ranged in length from 2.48 km to 6.20 km; all were between 35 and 75 m deep (Table 1). Throughout this paper, all IBSS lines videoed for habitat classification are referred to as transects while lobster data is referred to as lines due to the method of capture being lines of pots.

Video imagery of benthic habitat was collected by two high-resolution colour video cameras attached to a 500-kg weight hydraulically suspended from a reinforced wire towed behind a vessel. One camera faced forward at an angle of about 45° while the other, a live-feed analogue camera fitted with a wide-angle lens and auto iris lens for low light, was directed vertically downwards. A drogue was fitted to the vessel to ensure that the video apparatus was towed at speeds between 1.0 and 2.0 knots (1.852-3.704 kilometers per hour) with the cameras positioned 1.0 to 2.0 m above the substrate. Depth, position, speed and time were continuously recorded onto mini-digital video recorders for analysis in the laboratory.

To classify the habitat, the video footage was transferred to BTV Pro (www.bensoftware.com/btv/index.html), with the video stream set to pause every 0.013 seconds of latitude. Substrate and the main benthic groups within the field of view (~1 m diameter) were then identified, and the frame was assigned to a habitat class. The eight habitat classes were based on substrate and assemblage as well as percentage cover (Table 2). Four categories of mixed assemblage (a combination of red, brown and green algae) with or without sponges and with or without *Ecklonia* sp. were identified, with a separate category for habitats dominated by non-*Ecklonia* sp., or other brown algae. *Ecklonia* sp. was distinguished from other low profile brown algae due to its canopy-forming structure, abundance, and ease of identification through video imagery. In addition, habitats that consisted of bare sand, limestone rubble or reef were classified separately; only presence/absence was recorded for these three habitat types.

4.4.3 Statistical analysis

The habitat data were analysed with the multivariate statistical package PRIMER. As the habitat transects were different lengths, counts were standardised to give percentages of cover along a given transect. The data were square-root transformed and the modified Gower dissimilarity matrix calculated. A principal coordinates ordination was then run along with a multivariate ANOVA (PERMANOVA). The ANOVA design nested subregions (13 levels, random) within location (3 levels, fixed).

Mean lobster abundance (the number of lobsters per pot) and size were calculated for each of the IBSS lines. The mean number of lobsters was calculated for each line and for each of the three years (2004-2006) based on the individual values for each line's 16-20 pots. The mean abundance and standard deviation were then calculated from the three annual values. The means of the subregions were calculated from (1) the mean values for all ten IBSS lines within the subregion and (2) for the subset of lines for which habitat data were collected. Mean size, based on lobster carapace length (Morgan 1977), was calculated for each pot. The average was then calculated for the mean pot values for each line in each of the three years, which in turn was used to generate a mean line length across the three years. Subregion mean sizes and standard deviations were then estimated as for abundance. The abundance and size (log10) for lobsters from all lines and the subset of lines associated with the habitat transects were compared using a paired t-test at the level of subregions. The lobster abundance and size data were analysed with PRIMER Permutational ANOVAs (PERMANOVA) with location and subregion nested within location.

The relationship between lobster abundance and size and habitat was tested using stepwise multiple regression. Second- and third-order polynomials of the habitat variables were also calculated on the centred estimates of percentage of cover (Legendre and Legendre 1998). Neither abundance nor size was transformed as the assumptions associated with general linear models were met. Models were considered across all three locations (n = 34 lines) and within each location (Dongara = 8, Jurien = 15, Lancelin = 11 lines respectively). Dummy variables were also used to code for location so that the consistency of lobster habitat models could be assessed across locations. Model strength was assessed from the adjusted coefficient of determination due to a different number of independent variables being included in models and the model standard error due to different sample sizes. We also ran the models by aggregating habitat classes across habitat types (e.g. summing the high, medium, and low percentage cover values for each habitat type).

4.5   Results

4.5.1   Habitat

Principal coordinates ordination showed that 48% of the variation in habitat data could be explained by the first and second axes (Figure 2). Multivariate habitat characteristics varied significantly both among locations (pseudo-F = 4.08; df = 2; p = 0.0002) and among subregions within locations (pseudo-F = 1.65; df = 9; p = 0.0002). Statistically, the multivariate characterisation of habitats indicates that Dongara is different from Jurien (p = 0.001) and Lancelin (p = 0.006), which do not differ from each other (p = 0.10). Dongara is distinct in having higher levels of sand than the other two sites as well as high levels of low percentage cover of mixed assemblage with sponge (MA_SpLow). It is also distinct in having no high or medium percentage cover of mixed assemblage with *Ecklonia* sp. (MA_Eck), unlike the other two sites. Jurien is distinct only in having higher levels of medium percentage cover of undifferentiated mixed assemblage (MA), while Lancelin is distinct from the other two sites in having lower levels of low percentage cover of brown algae (BA) (Table 3).

4.5.2   Lobster

The average abundance of lobsters across the three locations ranged from 0.3 to 3.6 lobsters per pot, both the highest and lowest abundance of lobsters occurred at Jurien, the average abundance of lobsters was 1.5 ($\pm$ 0.28 SE) per pot (Figure 3). The size of lobsters ranged from 75 mm to 93 mm, both sizes were found at Lancelin, while the average size was 85 mm ($\pm$ 1.5 SE) (Figure 3). The paired t-test indicated that mean abundance is slightly, but significantly, greater on the IBSS lines associated with the video transects as opposed to all IBSS lines (p = 0.03), whereas there was no significant difference in size between the two sets of IBSS lines (p = 0.39). Lobsters abundance and size were also negatively correlated (r = 0.66, n = 34, p < 0.05; Figure 4).

Permutational nested ANOVA indicated that lobster abundance did not vary among locations (pseudo-F = 1.64; df = 2; p = 0.251), but did vary among subregions (pseudo-F= 9.98; df = 10; p = 0.001). Size showed the same pattern for locations (pseudo-F = 1.14; df = 2; p =0.361) and subregions (pseudo-F = 2.74; df = 10; p = 0.011). This result was largely an outcome of the high variability among subregions within the Jurien location. When lines were considered independently of subregions and a single-factor (location) ANOVA used, there were significant differences in abundance (pseudo-F = 9.50; df = 2; p = 0.001), with differences in size marginal with respect to locations (pseudo-F = 2.84; df = 2; p = 0.054) (Figure 5). Abundance was significantly higher at Jurien and Lancelin than at Dongara, while lobsters were largest at Dongara and smallest at Lancelin.

4.5.3   Modelling lobsters and their habitats

Abundance was difficult to predict quantitatively as a function of habitat variables across the three locations and 34 habitat transects. The best relationship was that between high percentage cover of mixed assemblage with *Ecklonia* sp. (Table 4), which explained 28% of the variation in abundance based on the adjusted R^2. Lobster abundance at Dongara was not predicted by any habitat variable, which likely reflects the lack of variability in abundance at the site (Figure 3). Low percentage cover of mixed assemblages were the best predictors of abundance at Jurien and Lancelin, with *Ecklonia* sp. being positively correlated to abundance and sponge negatively correlated. Although both relationships were significant and stronger than that

for all locations combined, neither is particularly powerful from a predictive perspective. Aggregating habitat classes across low, medium and high percentage cover categories did not improve predictive capacity for abundance.

Size was much more effectively predicted from habitat than was abundance. Moreover, aggregated habitat classes were better predictors, suggesting that size is determined more by qualitative characteristics of the benthos than by whether a habitat type is present at low, medium or high percentage cover. The best model predicted size as a function of habitat, with larger lobsters present in areas with sponge and smaller lobsters present in areas with *Ecklonia* sp. The multiple regression with dummy variables for locations indicated that overall lobsters at Lancelin were larger for the given habitat than at Dongara and Jurien. As with abundance, individual models could be developed for Jurien and Lancelin, with the best models predicting lobster size from the presence of *Ecklonia* sp. However, the predictive power of these models, as indicated by the larger or equivalent standard error of the estimate was not an improvement on the general model.

4.6   Discussion

This study established patterns between habitat and lobster abundance and size based on baited lobster pots. We trialled several sampling techniques, including deepwater divers and tangle nets. However, only pots successfully sampled lobsters at depths greater than 36 m. One effect of using baited pots is that the habitats in which the lobsters are caught may not reflect their shelter habitat, but rather their foraging area (Polovina *et al.* 1995). *Panulirus cygnus,* tracked in shallow waters, are active throughout the night and forage until dawn, travelling distances of up to 300 m (Jernakoff *et al.* 1987) therefore it is likely that bait from pots acts as an attractant to foraging lobsters (Jernakoff and Phillips 1988). Recent diet studies in deep water (>40m) have illustrated that bait from pots has become a large component of their diet (Waddington *et al.* 2008; Waddington and Meeuwig 2009). The second effect of using pots is that size and abundance data may be biased; larger lobsters present in or around pots may reduce the entrapment of smaller lobsters (Miller 1990; Frusher and Hoeing 2001). Thus, the presence of large lobsters in pots does not necessarily mean that smaller lobsters are absent from the habitat, whereas the presence of smaller lobsters in pots may indicate the absence of larger lobsters.

Bearing these caveats in mind, this study has demonstrated that lobster abundance and sizes differ according to the nature of the habitat. Benthic habitats dominated by sponges generally contain low numbers of large lobsters while habitats dominated by *Ecklonia* sp. generally contain higher numbers of smaller lobsters. Consequently Dongara, which had the largest percentage of habitat dominated by sponge (25%) had the fewest and largest lobsters, while Lancelin had the highest percentage of *Ecklonia* sp. dominated habitats (62%), with the highest abundance of small lobsters. Jurien was intermediate with 31% of habitat dominated by *Ecklonia* sp. and moderate abundances and sizes of lobsters. Therefore, despite the potential biases associated with the sampling technique, it would appear that the abundance and size of lobsters is influenced by habitat that may reflect the availability of suitable shelter or preferred food or both (Lipcius *et al.* 1998; Acosta 1999). These results are generally consistent with previous studies that have demonstrated that spiny lobsters are closely associated with structured habitats such as seagrass meadows, coral and rocky reefs that may offer food or refuge from predators (Acosta 1999; Mai and Hovel 2007) and that the balance between food and refuge requirements also change with lobster size (Acosta 1999; Rios-Lara *et al.* 2007), despite variability in the behaviour of different lobster species.

In our study, lobster abundance was best predicted across all three locations by the presence of high percentage cover of mixed assemblages dominated by *Ecklonia sp*. The relationship, while statistically significant, is not particularly strong, with only 28% of the variation in lobster abundance accounted for by habitat. However, the presence of high numbers of small lobsters in areas dominated by *Ecklonia sp*. is consistent with anecdotal reports from fishers that these areas are used for shelter and is also consistent with reports for other lobster species such as *Jasus edwardsii* (MacDiarmid *et al*. 1991).

Lobster size was much more effectively predicted from habitat variables than abundance. Larger lobsters were present in areas associated with sponges and smaller lobsters present in more structurally complex habitats dominated by *Ecklonia sp*. The size models were also more consistent across locations with mixed assemblage dominated by *Ecklonia sp*. the best predictor of size at both Jurien and Lancelin. As larger lobsters may be less vulnerable to predators, their distributions are not restricted to structurally complex habitats that provide protection from predators, they may move over a wider range and the habitat where they are trapped is not necessarily representative of their preferred shelter habitat (Polovina *et al*. 1995).

While this study has shown that there is a relationship between habitat and lobster abundance and size, lobster abundance and size may also be indicative of levels of fishing pressure. Commercial fishing statistics in Western Australia, averaged between 1996-2006 for the deepwater (>36m) portion of the western rock lobster fishery, show that Lancelin and Jurien, with an average of 160,066 potlifts and 266,658 kg landed per annum had similar catch rates of approximately 1.69 and 1.67 kg per pot respectively (unpublished catch and effort statistics, Department of Fisheries Western Australia 2008). These two sites had the highest abundances and smallest sizes of lobsters, which are generally under the minimum legal size for the commercial fishery. This is in contrast to Dongara, which had the lowest levels of fishing pressure with an average of 120,049 potlifts and 173,488 kg landed per annum, or a catch rate of 1.45 kg per pot. Dongara also had the lowest abundance of lobsters but on average these were the largest observed. The pattern observed in this study of large lobsters present in areas of low fishing effort is consistent with evidence from the spotted spiny lobster (*Panulirus guttatus*) in Auguilla (British West Indies) where large lobsters were present only at sites fished at low intensity (Wynne and Côté 2007).

Lobster abundance and size are correlated with both habitat and fishing pressure and the challenge is to disentangle these drivers. Wynne and Côté (2007) concluded that while fishing pressure affected the mean size of spotted spiny lobster through the removal of the largest lobsters from populations under the heaviest fishing pressure, they could find no clear link between fishing pressure and lobster abundance. Our results show that the largest lobsters are present in areas of low fishing effort but are also present in low abundance. The low abundance of lobsters, despite relatively low fishing effort, suggests that habitat may have a role in determining lobster demographics. Therefore, any effort at managing this species should incorporate knowledge of habitat quality.

The outcomes of our study indicate that stock assessments might be improved by incorporating spatial analysis of habitats and estimates of the distribution and abundance of the resource. In a broader sense, with the implementation of EBFM, understanding relationships between habitat variables and such attributes as abundance and size of commercially exploited species can also help to establish benchmarks against which the impacts of anthropogenic and natural impacts can be assessed. For example, it has been suggested that climate change will increase water temperatures, which may result in lobsters beginning their deepwater migration at a smaller

mean size (Caputi *et al.* unpublished data). If this occurs, smaller individuals may become more abundant in deepwater, which would increase the demand for preferred habitat, i.e. *Ecklonia*-dominated habitats. The concentration of particular size classes in small number of preferred habitats could also result in the formation of "hot spots", which might be preferentially targeted by fishers (Bello *et al.* 2005; Wynne and Côté 2007). These scenarios highlight the need for a better understanding of the spatial distribution of resources that can underpin management tools such as spatial closures.

4.7 Tables

Table 1. Description of IBSS sites and video transects.

Location	IBSS lines (*n*)	Video transects (*n*)	Mean Latitude	Mean Longitude	Mean max. depth (m)	Mean IBSS line length (km)	Mean distance from shore (km)
Dongara	40	8	29°20'00"	114°32'00"	50	6.20	41
Jurien	50	15	30°14'00"	114°43'00"	48	2.48	30
Lancelin	40	11	30°57'00"	115°00'00"	42	2.52	28

Table 2. Classification of habitat types from towed video. [a]Mixed assemblage consisted chiefly of red, brown and green algae. [b]High >70%, Medium 31–69% and Low <30%.

Label	Habitat type	Percentage cover
MA_Eck_Sp	[a]Mixed assemblage; sponges, *Ecklonia sp.*	[b]High, Medium and Low
MA_Sp	Mixed assemblage; sponges, no *Ecklonia sp.*	High, Medium and Low
MA	Mixed assemblage; no *Ecklonia sp.,* no sponges	High, Medium, Low
MA_Eck	Mixed assemblage; *Ecklonia sp.,* no sponges	High, Medium and Low
BA	Other Brown algae	High, Medium and Low
Reef	Reef structure	Presence/Absence
Rub/Lime	Rubble / Limestone	Presence/Absence
Sand	Sand	Presence/Absence

Table 3. Mean percentage (%) abundance of habitat type at each location. (For explanation of labels, see Table 2).

Labels	Dongara	Jurien	Lancelin
Reef	0.00	1.40	0.26
Rub/Lime	1.25	5.71	5.79
Sand	40.18	5.58	12.20
MA_Eck_SpHi	0.00	0.73	0.53
MA_Eck_SpMed	0.14	8.12	11.67
MA_Eck_SpLow	0.87	1.80	3.64
MA_SpHi	1.15	0.27	0.61
MA_SpMed	5.28	7.63	6.17
MA_SpLow	18.56	3.45	3.23
MAHi	0.00	0.00	0.16
MAMed	2.08	8.86	2.17
MALow	14.49	17.43	5.79
MA_EckHi	0.00	6.54	6.52
MA_EckMed	0.00	10.86	30.71
MA_EckLow	1.45	2.88	8.72
BAHi	0.00	0.00	0.45
BAMed	0.34	2.21	0.37
BALow	14.21	16.16	1.02

Table 4. Regression equations and associated statistics for predicting western rock lobster abundance and size as a function of habitat, including number of observations (n), adjusted R^2 (due to varying numbers of independent variables), p and model equation.

	Location	n	Adj R^2	SEE	p	Model
Abundance	All	34	0.28	1.26	0.007	1.61 + 8.44 (MA Eck HI)
	Dongara					none significant
	Jurien	15	0.32	1.64	0.01	1.72 + 29.6 (MA Eck LO)
	Lancelin	11	0.42	0.63	0.02	2.15–9.89 (MA SP Lo)
Size	All	34	0.65	4.13	<0.000001	81.7 + 20.7 (MA SP)–15.3 (MA ECK) + 5.3 (Lancelin = 1)
	Dongara	8				none significant
	Jurien	15	0.43	5.21	0.004	84.7–18.5 (MA Eck)
	Lancelin	11	0.76	4.11	0.0003	98.6–36.1 (MA Eck)

4.8   Figures
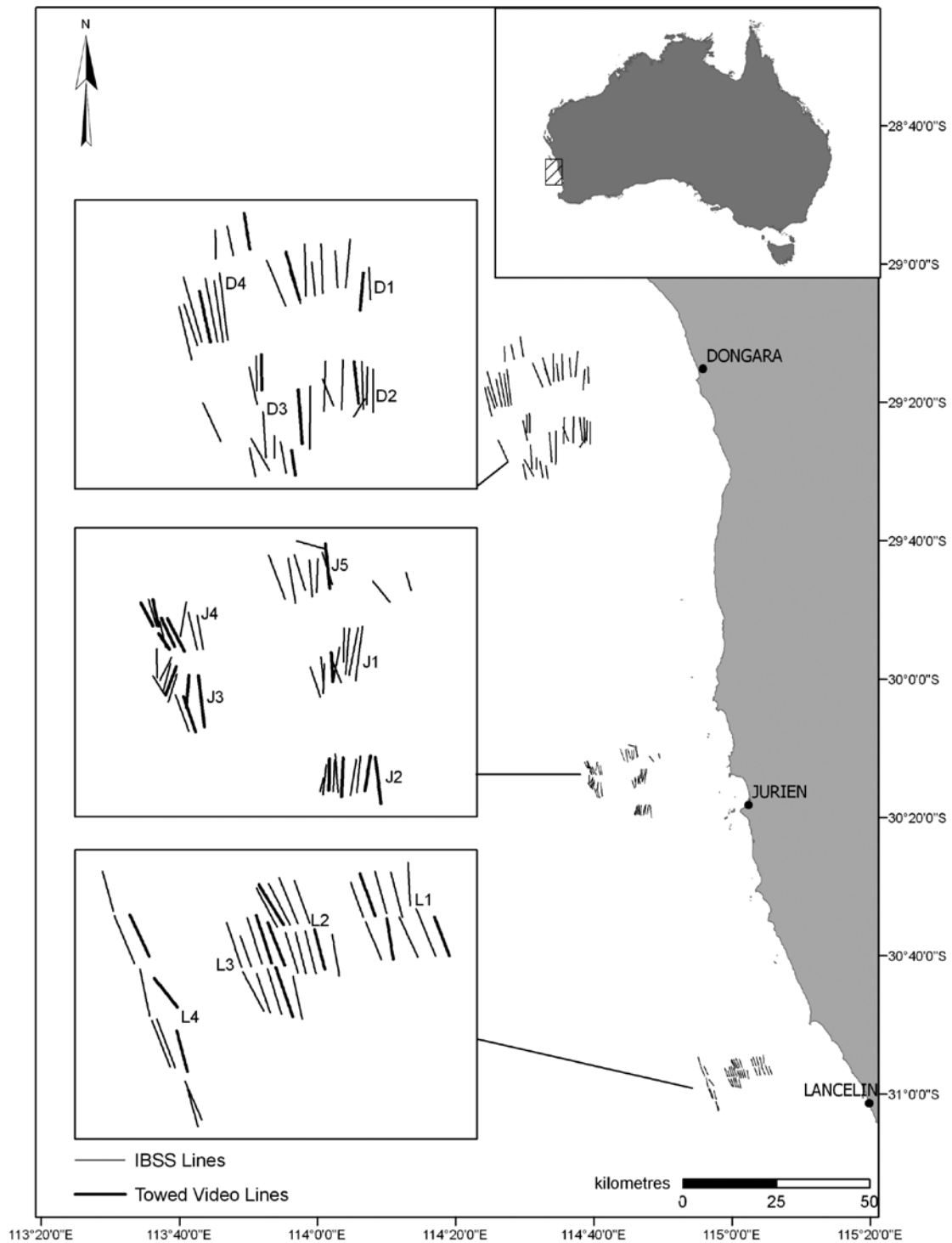
Figure 1.   Map of west coast of Western Australia showing the IBSS locations (Dongara, Jurien and Lancelin) and subregions at each location. Lobster size and abundance data was collected from all the lines while the bold lines are where towed video was collected.
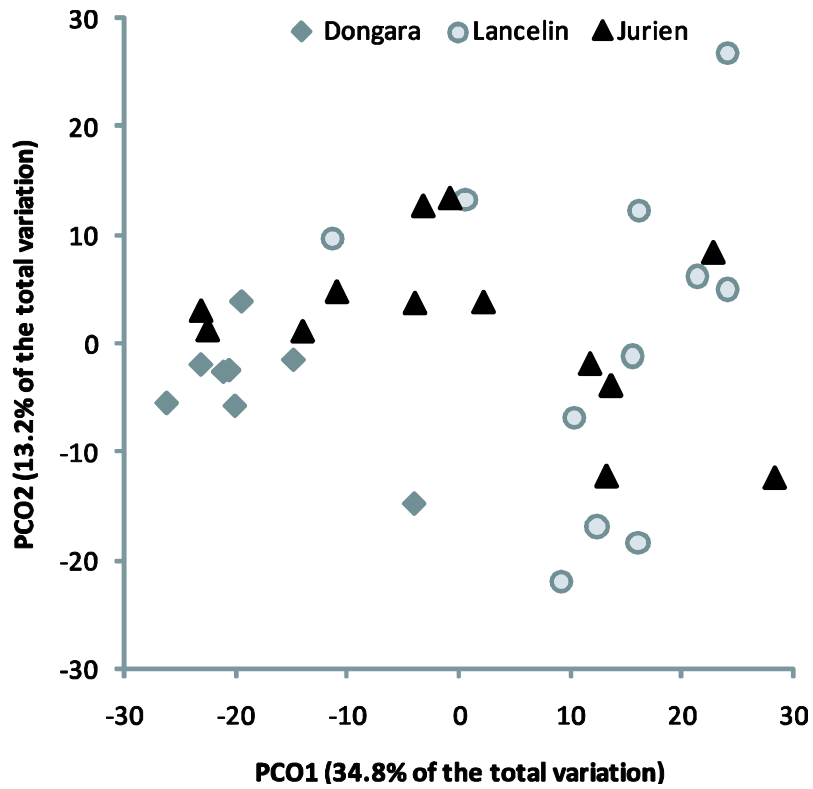
Figure 2. Principal coordinates ordination of habitat characteristics across the three study locations where Dongara is different from Jurien (p = 0.001) and Lancelin (p = 0.006), which do not differ from each other (p = 0.101).
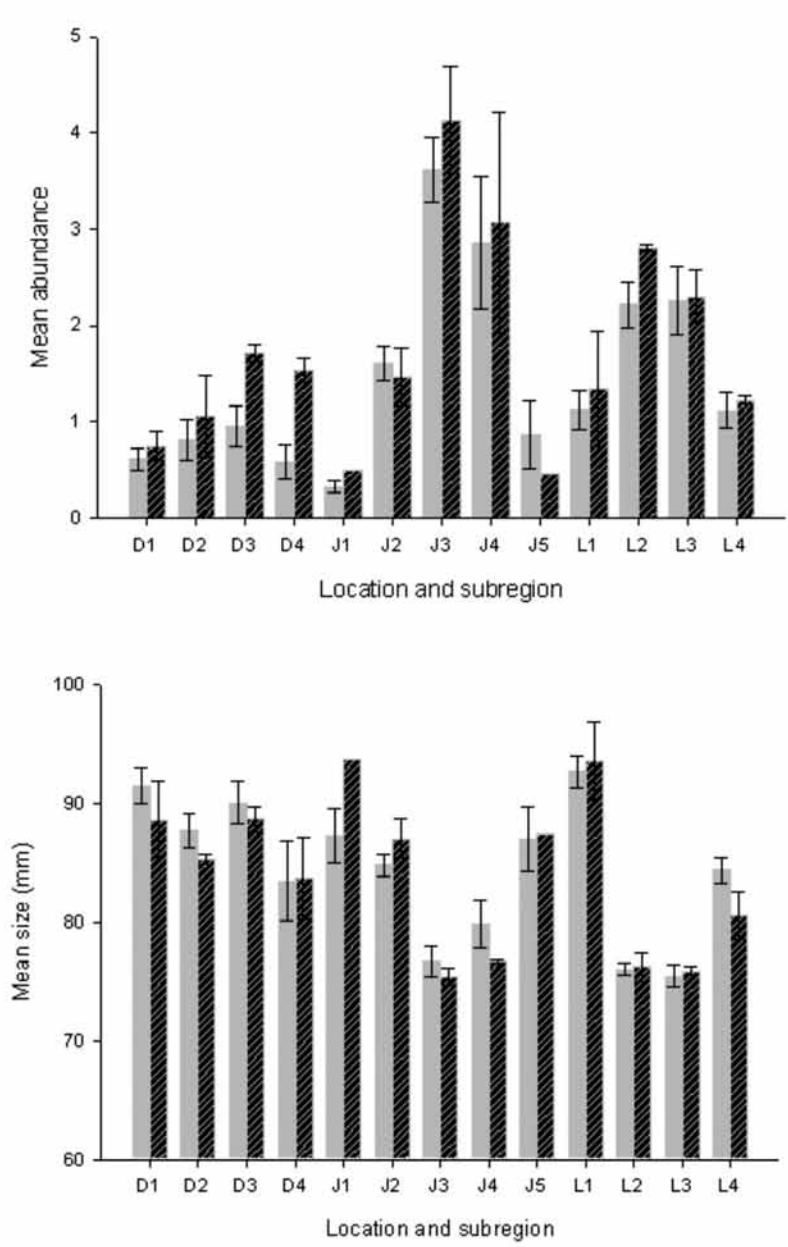
Figure 3. Mean abundance and size of western rock lobster (+/–se) for the period 2004–2006. Gray bars represent the mean values for all transects within a subregion at each location (D = Dongara, J = Jurien and L = Lancelin) and striped bars represent mean values for transect with towed video data.
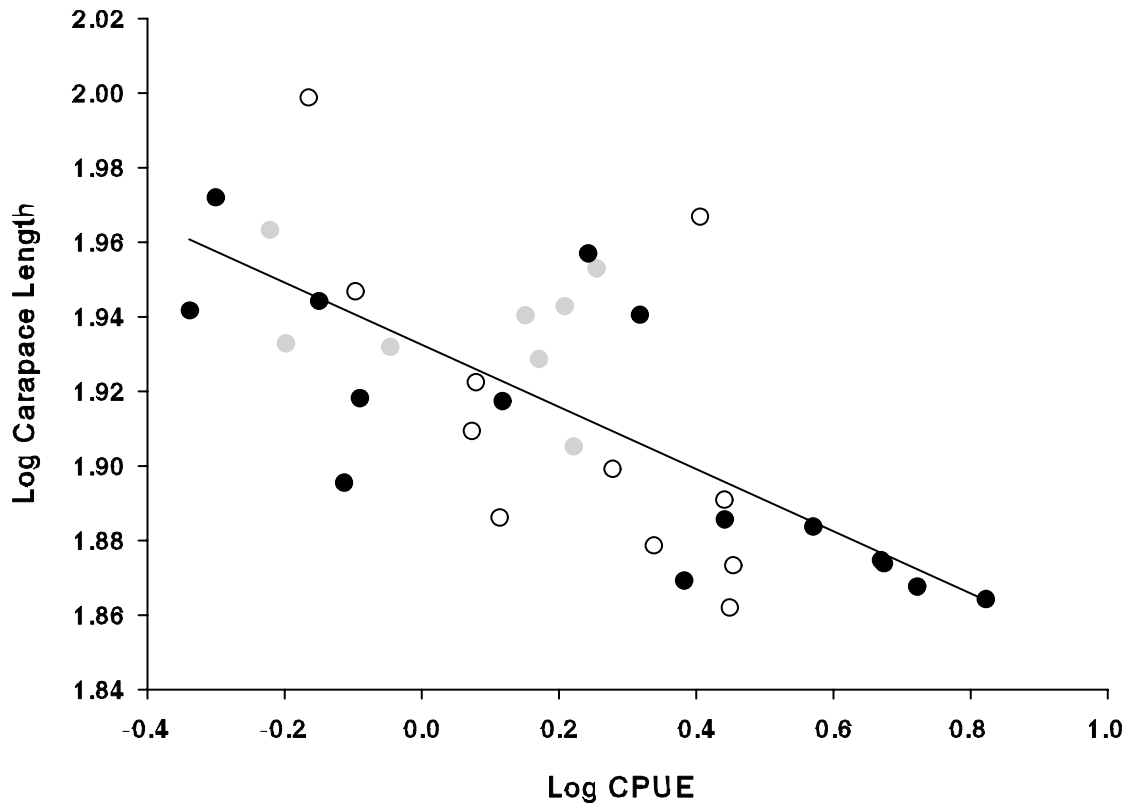
Figure 4. Relationship between lobster abundance (CPUE) and size (carapace length) at the three coastal locations (grey circles = Dongara, black circles = Jurien and white circles = Lancelin).
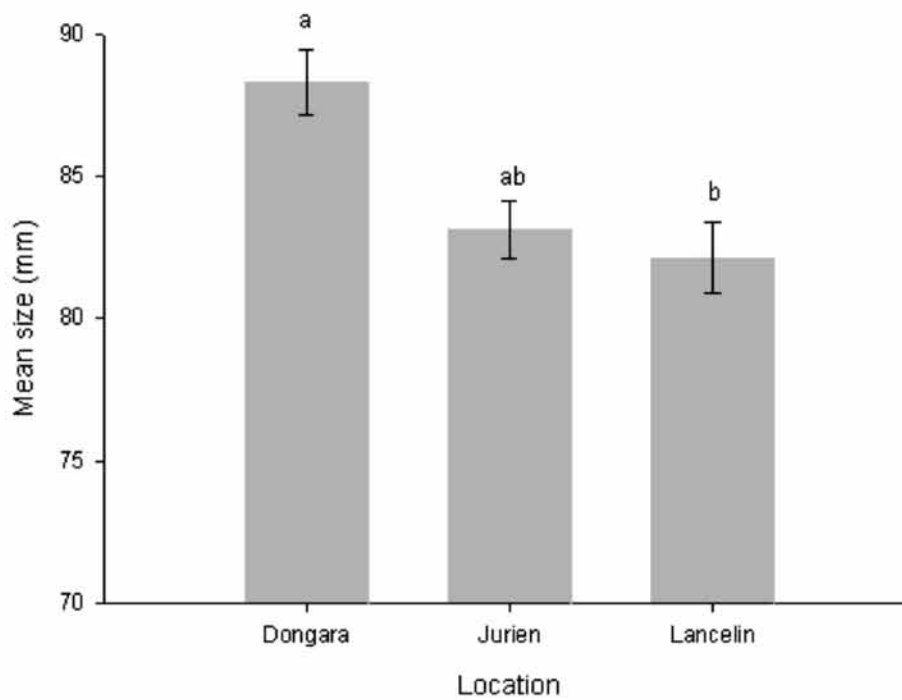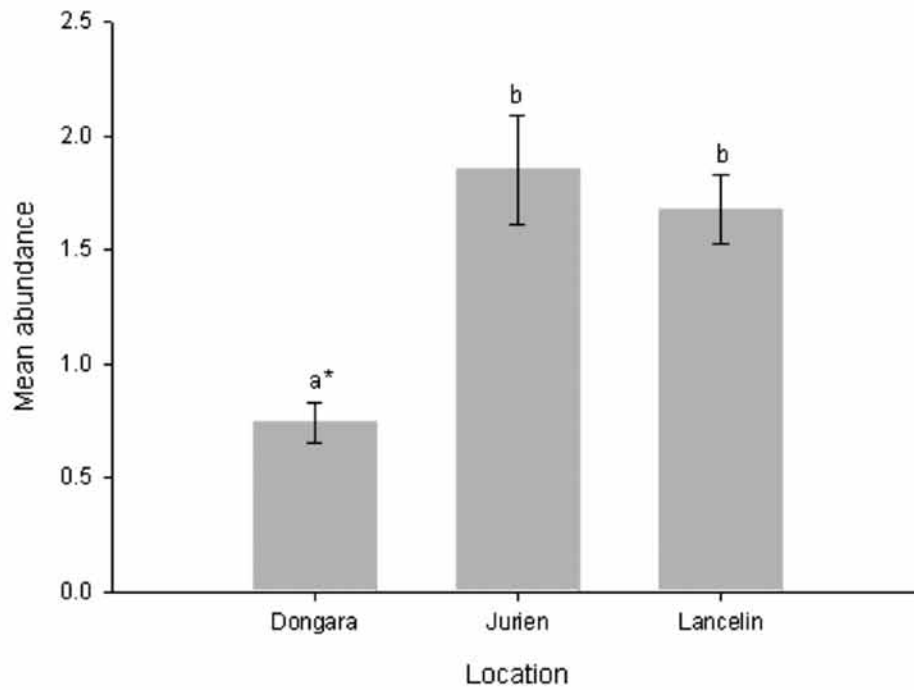
Figure 5. Mean abundance and size of western rock lobster (+/–se) at the three IBSS locations, with letters indicating significant differences at $p = 0.001$ (abundance) and $p = 0.003$ (size).

5.0 Predicting western rock lobster (*Panulirus cygnus*) abundance and size based on habitat characteristics at Jurien, Western Australia

Bellchambers, L. M., Meeuwig, J. J., Evans, S. N. and Radford, B.

5.1 Summary

This study aims to characterise the relationship between deepwater habitats and the abundance and size of lobster at Jurien. The research contained in this chapter was initiated to address Objective 3. Two different techniques, towed video and a full coverage hydroacousitc map were used to assess benthic habitat while data from the Independent Breeding Stock Survey (IBSS) was used to determine the size and abundance of lobsters.

This study found that both the habitat type and the size and abundance of lobsters varied significantly between subregions at the study site. There was a relationship between habitat type and lobster size and abundance with low numbers of larger lobsters present in areas associated with sponges and high numbers of smaller lobsters present in more structurally complex habitats dominated by *Ecklonia sp* which reflected the results found across the three sites examined in chapter 3. Similarly, these results indicate that different habitats may be important to difference life stages and highlight the importance of encompasses the full range of habitat types i.e. sponge, sand, macroalgae in any assessment of effects of biomass removal of target species by fishing.

5.2 Abstract

The present study aims to quantify the relationship between the abundance and size of western rock lobster (*Panulirus cygnus*) and the habitats in which they are found. Western rock lobster size and abundance data was derived from the annual western rock lobster independent breeding stock survey at five subregions of the IBSS. Two habitat datasets were used: (1) a towed video survey to derive benthic habitat data, conducted at Jurien, north of Perth, Western Australia between 2005 and 2007; and (2) a full coverage habitat map derived from a multibeam hydroacoustic survey and towed video.

Abundance and size of lobsters varied significantly across the five subregions of the IBSS at Jurien. Habitat classification from both towed video and the habitat map indicated that subregions also varied significantly in habitat composition. Our results show a strong association between western rock lobster abundance and size and habitat types with 74% of the variation being explained by the combination of *Ecklonia* and sponges. In general, low numbers of larger lobsters are present in areas associated with sponge and high numbers of smaller lobsters are present in more structurally complex *Ecklonia* dominated habitats.

This study has illustrated that both the amount of habitat and the configuration of habitat in deepwater influences the size and abundance of western rock lobster off the west coast of Western Australia. Our research also illustrates that incorporating a knowledge of the distribution and densities of preferred habitats is essential to ensure the effective assessment of the effects of biomass removal and to ensure appropriate habitats are included in the design and implementation of long term monitoring programs and marine protected areas.

5.3   Introduction

Spatial distribution and abundance of fisheries resources depend on the processes that are inherent to their life cycle such as recruitment, migration and mortality as well as on the distribution of the habitats used by these species (Lipcius *et al.* 1998, Acosta 1999, Butler *et al.* 2006). Knowledge of these processes and habitat use can more accurately predict distribution patterns of commercially important species and assist with developing effective management. In recent years the need to incorporate spatial analyses into stock and fisheries assessments has been emphasised in order to understand the dynamics of natural resources and fisheries (Bello *et al.* 2005, Rios-Lara *et al.* 2007).

Overfishing has resulted in the rapid decline of many species in coastal habitats (Dayton *et al.* 1995, Jackson *et al.* 2001). However, measuring the impacts of fishing can be difficult if habitat quality varies either naturally or as a result of anthropogenic impacts. Previous studies have illustrated that, for some species, their distributions and abundance are determined by the availability of preferred habitats (Polovina *et al.* 1995, Mai and Hovel 2007, Rios-Lara *et al.* 2007). The American lobster (*Homarus americanus*) is associated with structurally complex habitats in particular it has been closely associated with kelp beds (Bologna and Steneck 1993). The Caribbean spiny lobster (*Panulirus argus*) is strongly associated with sponges in Florida (Hernkind *et al.* 1997, Butler *et al.* 2005) and corals in Mexico (Bello *et al.* 2005). The abundance of the California spiny lobster (*Panulirus interruptus*) was higher in the presence of understorey kelp in California (Mai and Hovel 2007). While apparent habitat preferences may change with location, and even within the same species, these habitats contain higher densities of target organisms than less preferred habitats, reflecting naturally high recruitment rates and/or low mortality (Wynne and Côté 2007). Preferred habitats may also sustain higher fishing pressure before showing signs of overfishing (Wynne and Côté 2007).

Spiny lobsters are the targets of valuable fisheries worldwide (Butler *et al.* 2006), and like the majority of fishery stocks world wide, lobster stocks are nearly all fully or overexploited (National Research Council 1994). Indeed, a more recent review identifies all Panulirid fisheries as fully exploited (Phillips and Melville-Smith 2006). In recent years, marine protected areas have been increasingly established as a means of providing protection for exploited species and as research areas to study aspects of the population dynamics such as carrying capacity. While the debate over the design and usefulness of marine protected areas in fisheries management continues, spiny lobsters have been shown to benefit from protected areas (see Kelly *et al.* 2000, Davidson *et al.* 2002, Babcock *et al.* 2007). However for marine protected areas or harvest refuges to effectively protect exploited species, the interaction of the organisms with their habitats need to be understood to ensure that appropriate habitats are included (Acosta 1999, Acosta and Robertson 2003, Cox and Hunt 2005).

The western rock lobster *Panulirus cygnus* occurs along the west coast of Australia from Cape Leeuwin to North West Cape. Western rock lobsters are ecologically important in the coastal marine environment and form the basis of substantial commercial and recreational fisheries (Phillips and Melville-Smith 2005; de Lestang *et al*. 2007; refer to previous chapter for fishery details). The western rock lobster fishery was awarded Marine Stewardship Council (MSC) certification as a well-managed fishery in 2000 and has since been successfully re-certified in 2006. This certification largely reflects the significant data available on the shallow (<40 m) water ecology of lobsters and the impact of fishing and biomass removal. However, there remains a knowledge gap with respect to the deep-water ecology of western rock lobster. Most western rock lobster undertake a substantial offshore migration as they approach sexual

maturity which coincides with the size at which they reach legal minimum size and are targeted by the commercial and recreational fishery (Phillips and Melville-Smith 2006). The major issue of concern is the ecosystem impacts of biomass removal in deepwater where the bulk of the adult population resides and how to best assess and mitigate the potential impacts.

The present study aims to identify the preferred habitat of western rock lobster and ultimately predict spatial distribution of western rock lobster based on habitat variables. We also consider the use of two different and independent approaches to the characterisation of habitats: towed video transects and full cover habitat maps. Knowledge of the abundance and size structure of western rock lobster as related to habitat is a critical first step to understanding the impacts of fishing practices and is a crucial component of designing successful management measures to ensure the sustainability of the species and the ecosystem they inhabit. It also provides the basis for identifying potential marine protected (closed) areas that can be used for further study on the ecosystem effects of fishing and other population dynamic relationships.

5.4 Methods

5.4.1 Study locations and the Independent Breeding Stock Survey (IBSS)

See Chapter 4, Section 4.4.1 for details

5.4.2 Habitat classification using towed video

See Chapter 4, Section 4.4.2 for details

5.4.3 Habitat classification using multibeam hydroacoustics and towed video

Multibeam data were acquired for the Jurien location as part of the Marine Futures project (Kendrick *et al*. in prep) and as the base layer for subsequent habitat mapping (Radford and Van Niel 2008). Full cover multibeam bathymetry and acoustic backscatter (snippets) to map water depths and to delineate seabed features and potential benthic habitats were acquired using the Kimberley Quest 1 (KQ1). The KQ1 is 26 m long vessel with a draft of 1.98 m, allowing it to operate safely to a minimum depth of 5 m. Surveys were conducted around the clock subject to weather and safety concerns. Processing occurred concurrently as data was acquired.

The survey equipment used included an inertial motion and position system to correct for vessel heave, pitch and roll (Applanix F180), an on line navigation and data logging system (Starfix Seis navigation software), a velocimeter / temperature salinity probe, the multibeam echo sounder (Reson Seabat 8101 with associated multibeam acquisition software), and aquatic tide gauges. Processing was completed on the vessel using ArcGIS 9.1++ and the Starfix Processing software (Fugro Survey Pty, 2006).

Coordinates are referenced to the Geocentric Data of Australia 1994 (GDA94) following conversion from the International Terrestrial Reference Frame 2000 which is essentially identical to the World Geodetic System 1984. Vertical positioning was controlled by deploying a temporary tide gauge to evaluate whether there were significant tidal gradients and lag across the survey area and relative to established tide gauge positions (Fugro Survey Pty, 2006).

Multibeam bathymetry processing was completed using the Starfix Multibeam and Proc

system. Each line was processed individually, filtered, despiked and reduced for effects of refraction, draft and tide before being merged into a gridded dataset for the survey area. Bathymetry was gridded to a resolution of 2.5 m. Data was exported to *.pnt files for gridding in Starfix Workbench. Backscatter data was also processed using Starfix Proc system and also rasterised to a resolution of 2.5m (Fugro Survey Pty, 2006).

A towed video sampling plan was developed based on the full coverage hydroacoustic data (Radford and Van Niel. 2008). Effectively, video sampling is stratified across the area of interest to maximise coverage over acoustically complex areas as well as boundaries between areas of varying complexity. Video imagery was then classified using standard techniques into a series of predetermined habitat classes (Radford and Van Niel. 2008; Table 1). These classes are not identical to those previously established for the targeted western rock lobster study but are roughly consistent.

A habitat map for Jurien was then produced using classification and regression tree (CART) techniques to predict the probability of presence/absence of each habitat type as a function of the physical data generated from the hydroacoustic data, including primary and secondary derivatives (Radford and Van Niel. 2008).

The IBSS survey lines were then overlaid on the habitat map. At the mean coordinates of each pot location (2004-2006), the probability of presence for each of the habitat classes was generated. Where the probability was greater than 50%, the habitat class was scored as "present" with probabilities less than or equal to 50% scored as "absent". Scores were then summed for each of the 16 pots along each line and treated as estimates of relative abundance of each habitat type.

5.4.4   Statistical analysis

5.4.4.1   Lobster characteristics

Mean abundance (the number of lobsters per pot) and size were estimated for each of the transects associated with the habitat data derived from (1) towed video (n=15) and (2) the multibeam map (n=20). With respect to abundance, the mean number of lobsters was calculated for each transect and for each of the three years (2004-2006) based on the individual values for each of the 16-20 pots per line. The transect mean and standard deviation were then calculated from the three annual values. Subregional means were calculated using (1) the mean values for all ten transects within the subregion and (2) for the subset of transects for which habitat data exists.

Mean lobster size, based on lobster carapace length (Morgan, 1977), was calculated for each pot. The average was then calculated for the mean pot values for each transect in each of the three years, which in turn were used to generate a mean carapace length for each transect across the three years. As with abundance, subregion mean lengths and standard deviations were then estimated as (1) the average of the ten transects within the subregion and (2) the average of those transects for which habitat data exists.

The abundance and size estimates of lobsters for the towed video transects and the map transects were compared at the level of subregion using a paired t-test with abundance and size transformed to log10 to meet assumptions about equal variance.

The lobster abundance and size data were analysed using the PRIMER package (Clarke and Gorley 2006) in order to take advantage of the permutational hypothesis testing, in part because variance across locations and among subregions was heteroscedastic, even under transformation

(Clarke and Gorley 2006). Lobster abundance and size were treated as a multivariate dependent variable with a single permutational multivariate ANOVA applied to test for the effect of subregion on lobster attributes (Anderson 2001). This choice was made rather than testing abundance and size separately to take into account variability with respect to the relationship between abundance and size. Type III PERMANOVA with subregion as a single fixed factor were performed based on 999 unrestricted permutations on the non-transformed abundance and size data.

5.4.4.2   Habitat

The habitat datasets derived from towed video and the map were also analysed using the multivariate statistical package PRIMER. As transects varied in length, the counts were standardised so that they represented percentage cover along a given transect. No transformations were applied to the habitat data, as the influence of rare habitat classes was not the central focus. The modified Gower dissimilarity matrix was applied as the habitat data are treated as environmental (Euclidean) predictors however the presence of joint absences was not considered relevant (Clarke and Gorley 2006). A principal component analysis was then run along with a multivariate ANOVA based on 999 unrestricted permutations (PERMANOVA) to evaluate the effect of subregion on habitat composition with subregion as a fixed factor (Anderson 2001).

5.4.5   Modelling the relationship between western rock lobster and habitat

The relationship between western rock lobster and habitat characteristics, both those derived from towed video and full coverage habitat maps, were modelled using distance based linear models in Primer, a multivariate form of linear regression (McCardle and Anderson 2001). These models maximise the relationship between the multivariate response variables and the predictor variables as a continuous form of redundancy analysis. The western rock lobster abundance and size data were retained as normalised data in order to place both variables on the same scale and with the Euclidean distance resemblance measure calculated as the abundance and size data are effectively continuous variables. This approach again effectively treats the dependent variable as multivariate, integrating information on both abundance and size.

The towed video derived habitat data were aggregated so that the low, medium and high cover classes were summed within each abiotic or biotic category. This aggregation was done as exploratory analyses indicated that models had difficulty distinguishing between levels of cover. Specifically, the same habitat class entered the model as, for instance, *Ecklonia* – medium cover and *Eckonia* – low cover with the direction of the relationship typically the same (e.g. positive). Given the small number of samples (n=10 for towed video data and n=20 for habitat derived data), the models risked being overfitted by habitat data if the relationship between more than one habitat class was to be explored.

In running the DistLM routine in Primer, the adjusted R^2 was used as the criterion of fit. This choice was made to facilitate comparison of models with varying numbers of independent variables. The selection procedure chosen was "Best" in order to allow comparison of various models. Pearson correlations were then run between western rock lobster abundance and size and the habitat characteristics identified most strongly with western rock lobster in order to interpret the direction of the relationship. Redundancy analysis plots were also produced to demonstrate the associations between transects with respect to western rock lobster characteristics and habitats.

5.5   Results

Abundance and size of lobsters differ significantly across the five subregions of Jurien for which towed video derived habitat data exists (Table 2; Figure 2). Monte Carlo tests indicated that SR3 was significantly different from SR1 (p=0.032), SR2 (p=0.003) and SR5 (p=0.040), and that SR2 was significantly different from SR4 (p=0.047). No other pairwise comparisons among regions yielded statistically significant differences in western rock lobster characteristics. With respect to the lobster transects for which full coverage map derived habitat data exists, there was no significant difference between SR 3 and SR4 (Table 2, p=0.23), consistent with a lack of difference between SR3 and SR4 in the towed video associated transects. The average distances within and among subregions with respect to abundance and size of western rock lobster indicate that SR1 is most different from SR3 and SR4 (Table 3) followed by differences between SR3 and SR5. The average distances within subregions indicate that within subregion differences are relatively small compared to between subregions with the exception of SR4, which is relatively variable (Table 3). In summary, subregions 1, 2 and 5 typically had low abundance (<2/pot) of larger lobsters (>85mm) while subregions 3 and 4 typically had higher abundance (>2/pot) of smaller lobsters (<80mm).

Paired t-tests indicated no statistically significant differences between log10 transformed lobster abundance and size in the two subregions (SR3 and SR4) for which both towed video and map derived habitat data exist (p=0.18 and 0.24 respectively). This in part likely reflects the small sample size (n=2). Paired t-tests also indicated no significant differences between the subsets of transects associated with the towed video habitat data and the entire IBSS data set (50 transects; p=0.92, p=0.66, n=5). The values for the entire IBSS dataset and the map derived transects are identical as all IBSS transects were used. The lack of significant differences suggest that the subsets of transects used for the towed video appropriately reflect the larger IBSS dataset.

Subregions differ significantly in habitat composition both with respect to habitat data derived from towed video and from the full coverage habitat map (Table 4; Figures 3-5). Monte Carlo tests on the towed video data indicated that SR3 was significantly different from SR1 (p=0.035), SR2 (p=0.005) and SR5 (p=0.038). No other pairwise comparisons among regions indicated statistically significant differences in habitat. The mapped data indicated significant differences between SR 3 and SR 4. *Ecklonia* strongly dominates SR3 while a more mixed assemblage dominates SR1 and SR2. Brown algae is the most dominant habitat class in SR4 and SR5 but both of these subregions characterised by a more even distribution of habitat classes than the other regions. The principal component analyses both indicate the high variability within SR4 relative to the other subregions.

Strong relationships exist between western rock lobster abundance and size and habitat types (Figure 6 and 7). Considering the habitat data derived from towed video, 74% of the adjusted variation in western rock lobster abundance and size is explained by the *Ecklonia* / Sponge, *Ecklonia* and Sponge classes (Table 6). At a univariate level, abundance is most strongly and positively correlated to the presence of *Ecklonia* while size is significantly and positively correlated to the presence of sponge (Table 7).

In considering just SR3 and SR4 for which map derived habitat data are available, western rock lobster abundance and size is most strongly associated with "other algae" (Table 6). Restricting the towed video data to these two regions, brown algae is strongly associated with western rock lobster abundance and size (Table 6) and it is likely that these two classes are reasonably

synonymous. The strength of these two relationships is also similar (Table 6) and the direction of the relationships the same (Table 7).

5.6   Discussion

Previous authors have suggested that lobsters are associated with specific habitats (Bologna and Steneck 1993, Acosta and Butler 1997, Eggleston and Dahlgren 2001, Butler *et al*. 2005, Wynne and Côté 2007) and that habitat type and structure plays a crucial role in determining lobster size (Acosta 1999, Acosta and Robertson 2003, Cox and Hunt 2005, Mai and Hovel 2007) and abundance (Polovina *et al*. 1995, Lipcius *et al*. 1998, Cox and Hunt 2005). Our study is consistent with these results, demonstrating that western rock lobster abundance and size in Jurien deepwater is strongly driven by habitat, with the abundance of relatively small lobsters higher in areas of *Ecklonia* and mixed assemblage while the larger, less abundant lobsters were strongly associated with areas of sponge.

Previous authors have also suggested that the abundance of particular size classes of lobsters may be limited by the size specific availability of suitable shelter and protection from predators, which in turn is determined by habitat structure (Eggleston *et al*. 1990, Eggleston and Lipcius 1992). The abundance of *P. argus* in Key West National Wildlife Refuge is related to the density and volume of large sponges in channel habitats in Key West National Wildlife Refuge (Eggleston and Dahlgren 2001) with juvenile lobsters in seagrass and hard bottom habitats while adult lobsters were found in offshore patch reefs. Similarly *P. argus* in Belize were more abundant on mangrove and coral islands surrounded by seagrass with higher proportions of juveniles in these habitats while islands surrounded by sand and rubble were dominated by adult lobsters (Acosta 1999). Therefore even within a species the habitat preference of different size classes can vary with location. However, there appears to be a general trend for smaller lobsters in structurally complex habitats that offer them adequate food and shelter from predators (Lipcius *et al*. 1998, Acosta 1999). While larger lobsters may be less vulnerable to predators and therefore may not be as dependent on habitat for shelter (Polovina *et al*. 1995). This may also be the case for the western rock lobster as smaller individuals were found in areas of higher structural complexity *i.e. Ecklonia* that may offer shelter from potential predators while the larger lobsters were found in the more open areas dominated by sponges and sessile invertebrates.

Two key issues need to be considered with respect to the observed pattern between size and habitat given commercial rock lobster pots were used to collect western rock lobster data. Firstly, the habitats in which the western rock lobsters are caught may not reflect their shelter habitat but rather their foraging area (Polovina *et al*. 1995). Previous studies have shown that lobsters remain active throughout the night and continue to forage until dawn and that the median distance travelled by western rock lobster in a single foraging trip is approximately 310m (150m radius from their den) although distances may be as large as 800m (Jernakoff *et al*. 1987). Bait from pots has been shown to be a major component of western rock lobster diet (Waddington *et al*. 2008) and a likely attractant (Jernakoff and Phillips 1988) indicating that lobsters caught in pots are foraging and therefore may not be in their shelter habitat.

Secondly, previous studies have suggested that larger lobsters present in or around pots may reduce the entrapment of smaller lobsters thus biasing the size classes of lobster caught upward (Miller 1990, Frusher and Hoeing 2001). The presence of large lobsters in pots does not necessarily mean that smaller lobsters are absent from the habitat whereas the presence of smaller lobsters in pots may indicate the absence of larger lobsters.

The strong associations between western rock lobsters and habitats have three major applications to management. The first application lies in improving the fisheries independent breeding stock surveys. In general, the *a priori* classification of transects into relatively homogeneous subregions for the IBSS survey is reasonable. However, subregion 4 shows greater variability in abundance and slightly more variability in size than is observed in the other subregions. Both the towed video data (Figure 4) and the habitat map show that subregion 4 is also very heterogenous with respect to habitats, including areas of *Ecklonia*, mixed assemblage and brown algae. This is an important result from a monitoring perspective as changes in the independent breeding stock survey within subregion 4 may be in part masked by habitat-driven variability. This outcome is also relevant to the design of any monitoring programs associated with assessing changes over time in population structure after removal of fishing effort: monitoring must occur on comparable locations (Childress 1997).

The second application lies in the design of marine protected areas. Marine protected areas are an important research tool to understand ecosystem effects of fishing and important population dynamics characteristics. They also have the potential to protect juveniles to maturation increasing the densities of mature individuals and higher reproductive output. However, information regarding the distribution of habitats, abundance and size of lobsters and biomass removal is essential in the design of fished areas or marine protected areas (Acosta 1999, Eggleston and Dahlgren 2001, Acosta and Robertson 2003). The current study has suggested that if lobster stocks are being recruitment overfished, that protecting habitats dominated by sponges might be important, as this is the preferred habitat type of larger lobsters. While if the population is growth overfished then protecting habitats dominated by *Ecklonia* may be more important has these are the preferred habitats of smaller lobsters. For reseach purposes both habitat types would need to be protected to understand the effects of fishing. Similarly in planning marine protected areas or harvest refuges it is important to include a diversity of habitats to encompass different life stages (Cox and Hunt 2005). Previous studies have illustrated that failure to incorporate habitat preference and habitat use for shelter and foraging can result in areas that are inadequate in size or location to protect exploited species (Eggleston and Dahlgren 2001, Cox and Hunt 2005). Marine protected areas effectively establish a natural controlled experiment that provides the ability to examine changes due to natural variability and environmental cycles from those due to anthropogenic effects. These types of investigations, conducted at realistic spatial scales, can usually only be conducted in marine reserves, and highlight the critical importance of MPAs as reference areas for understanding the effects of fishing (Buxton *et al*. 2006).

The third application is in the development of cost effective methods for habitat assessment such that species habitat relationships can be established. By using data derived from towed video transect and data derived from a full coverage habitat map, we have been able to compare the outputs and effectiveness of these tow techniques. Both methods generate strong correlations between western rock lobster and habitats. Although slightly different habitat categories were used in the two different methods, the results for the subregions were generally consistent and the methods are robust if habitat is classified using the dominant species present and the conclusions are consistent. Larger lobsters like sessile invertebrates and sponge while smaller lobsters like *Ecklonia*. Previous studies using only towed video habitat classification have found that the same correlations exist for both Dongara and Lancelin on the West coast of Western Australia (Bellchambers *et al*. in press.).

The primary advantage of the towed video is that it is a cost effective method to obtain a good understanding of habitat / western rock lobster relationships. The primary advantage of the full

coverage maps is that they allow greater visualisation of the spatial representation of sampling programs and they also may allow more accurate estimates of standing stock by weighting catch rates and biomass by availability of habitat. Full coverage habitat maps are however relatively expensive, and require specialised equipment and a high level of expertise. Fine scale commercial fishing data can also be used to provide an estimate of biomass removal for each particular habitat type to examine the distribution of the impacts of fishing on particular habitats and corresponding size classes of lobsters.

This study has allowed the comparison of towed video transects and full coverage habitat maps as methods of habitat assessment. Both techniques successfully classified the benthic habitat and found correlations between lobster abundance and size structure and habitat variables. The difference in the two techniques lies in the expense and level of detail required, towed video transects are cost effective and give a general overview of the habitat type present and its extent, while full coverage habitats provide fine spatial detail that in combination with fishing effort data allow the opportunity to assess the potential impacts of fishing pressure. This study has also increased the knowledge of the relationship between the abundance and size structure of lobsters and habitat variables, which is essential for the effective design, and implementation of MPAs. Effectively designed MPAs as a research and management tool can have important implications can have for exploited species for example western rock lobster populations in sanctuary zones at Rottnest Island, Western Australia, display higher levels of density, biomass and egg production than those in adjacent fished areas (Babcock *et al*.2007). Therefore, an increased knowledge of species and habitat allows an assessment of the impacts of fishing and is a crucial component of designing successful management measures to ensure the sustainability of the species and the ecosystem they inhabit.

5.7 Tables

Table 1. Classification of habitat types from towed video captured as part of the (a) western rock lobster project and (b) Marine Futures project. The latter habitat classes are presented next to the most comparable habitat class from the towed video.

Label	Habitat type – towed video	Habitat type – Full coverage map
MA_Eck_Sp	Mixed assemblage with sponges and Ecklonia	
MA_Sp	Mixed assemblage with sponges and without Ecklonia	Invertebrates
MA	Mixed assemblage without Ecklonia and without sponges	Other algae, Rhodoliths
MA_Eck	Mixed assemblage with Ecklonia and without sponges	Ecklonia
BA	Brown algae	
Reef	Reef structure	Reef (high, medium and low profile)
Rub/Lime	Rubble / Limestone	
Sand	Sand	Sand

Table 2. PERMANOVA for western rock lobster characteristics, based on Euclidean distance resemblance matrix for abundance and size for (a) western rock lobster lines associated with towed video data and (b) western rock lobster lines associated with map data. Type III fixed factor model with 997 and 990 unrestricted permutations.

Data	Source	df	SS	MS	Pseudo-F	P(perm)
(a) Towed	SR	4	16,811	4,203	4.53	0.012
Video	Res	10	9,283	928		
	Total	14	26,094			
(b) Map	SR	1	2.7	2.7	1.38	0.227
	Res	18	35.3	2.0		
	Total	19	38.0			

Table 3. Average distances in western rock lobster characteristics within/between groups based on PERMANOVA for data associated with towed video subset.

	SR1	SR2	SR3	SR4
SR2	1.13	0.77		
SR3	3.22	2.20	0.77	
SR4	2.90	1.96	1.29	1.81
SR5	0.91	0.64	2.53	2.13

Table 4. PERMANOVA for habitat derived from (a) towed video for five subregions and (b) mapped data for SR 3 and 4. Analyses based on modified Gower resemblance matrix for aggregated habitat classes in the case of towed video (a) and predicted habitat classes for Map (b). Type III fixed factor model with 999 and 994 unrestricted permutations respectively. Monte Carlo tests on the towed video data indicated that SR 3 was significantly different from SR1 (p=0.035), SR2 (p=0.005) and SR5 (p=0.038).

Data	Source	df	SS	MS	Pseudo-F	P(perm)
(a) Towed Video	SR	4	5,451	1,363	2.37	0.015
	Res	10	5,754	575		
	Total	14	11,205			
(b) Map	SR	1	3,275	3,275	4.08	0.021
	Res	18	14,434	802		
	Total	19	17,709			

Table 5. Average similarities within/between groups based on PERMANOVA for the towed video data.

Towed video	SR1	SR2	SR3	SR4
SR2	76.5	77.0		
SR3	48.6	55.4	75.2	
SR4	59.2	64.6	61.4	58.2
SR5	69.4	70.4	50.2	57.1

Table 6. Habitat variables driving "best" models with respect to western rock lobster abundance and size for the towed video and map data sets.

Data Set	Variable 1	Variable 2	Variable 3	adj R^2
Towed video - all SR	Ecklonia (0.001)	Ecklonia + Sponge (0.011)	Sponge (0.006)	0.74
Towed video - SR 3, 4	Brown Algae (0.006)			0.60
Map - SR 3, 4	Other Algae (0.001)			0.56

Table 7. Pearson correlation coefficients and associated p –values for western rock lobster characteristics (mean abundance and mean size) and habitat classes for the models provided in Table 6 for the two data subsets (towed video and map). The towed video model results for SR 3 and 4 and the results for brown algae across all towed video subregions are provided by way of contrast to the results from the mapping data set.

Data Set	Habitat Class	n	Mean Abundance		Mean Size	
			r	p	r	p
Towed video - all SR	Ecklonia Sponge	15	0.54	0.040	-0.61	0.015
	Sponge	15	-0.49	0.066	0.78	0.001
	Ecklonia	15	0.79	0.000	-0.71	0.003
Towed video - all SR	Brown Algae	15	-0.60	0.017	0.28	0.310
Towed video - SR 3, 4	Brown Algae	9	-0.81	0.009	0.80	0.009
Map - SR 3, 4	Other algae	20	-0.69	0.0007	0.83	0.0000

5.8   Figures
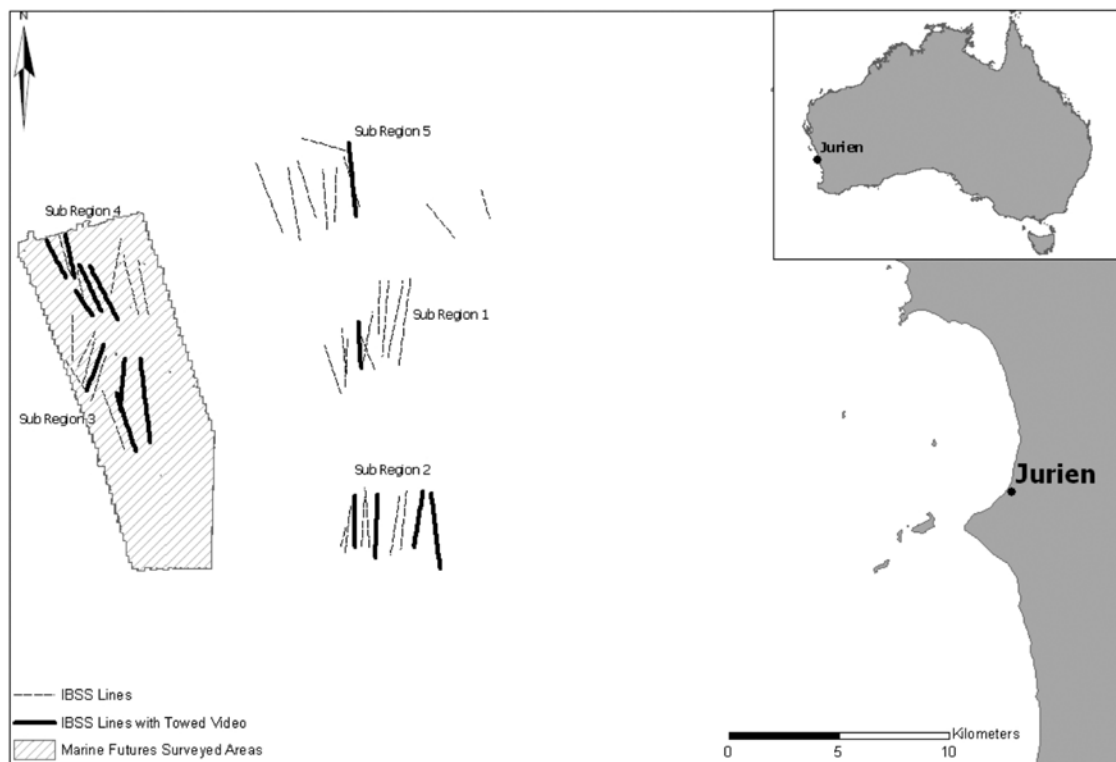
Figure 1. Map of Jurien Bay, Western Australia, showing the of the Independent Breeding Stock Survey (IBSS) subregions and areas surveyed by multibeam hydroacoustics.
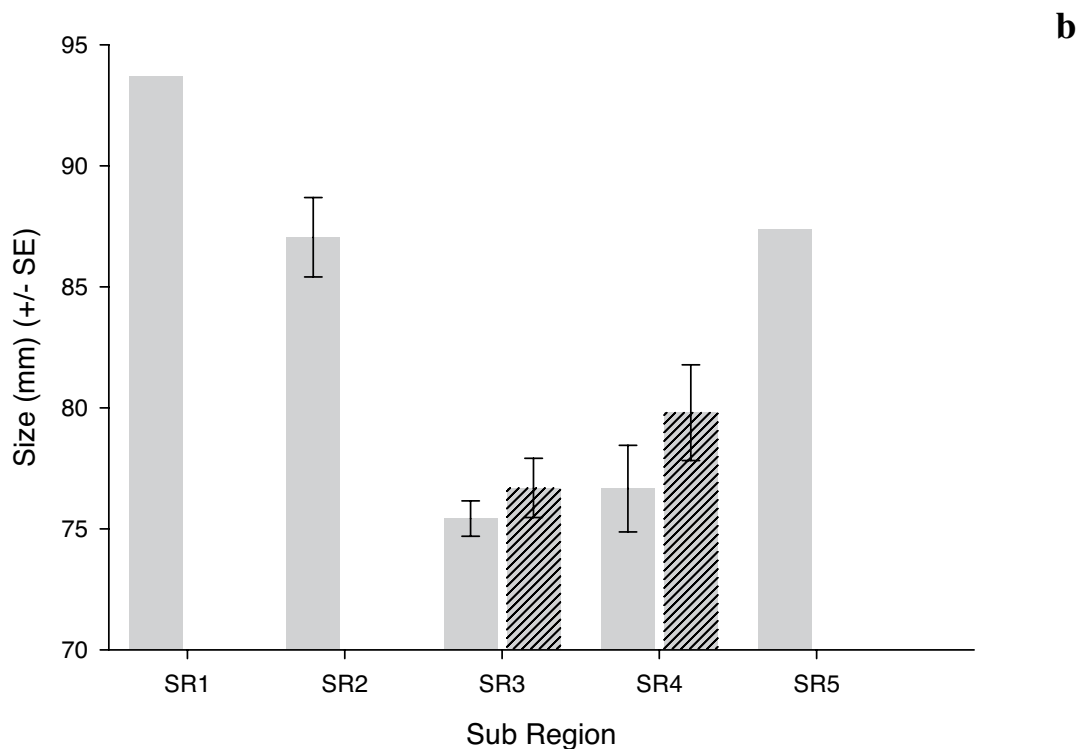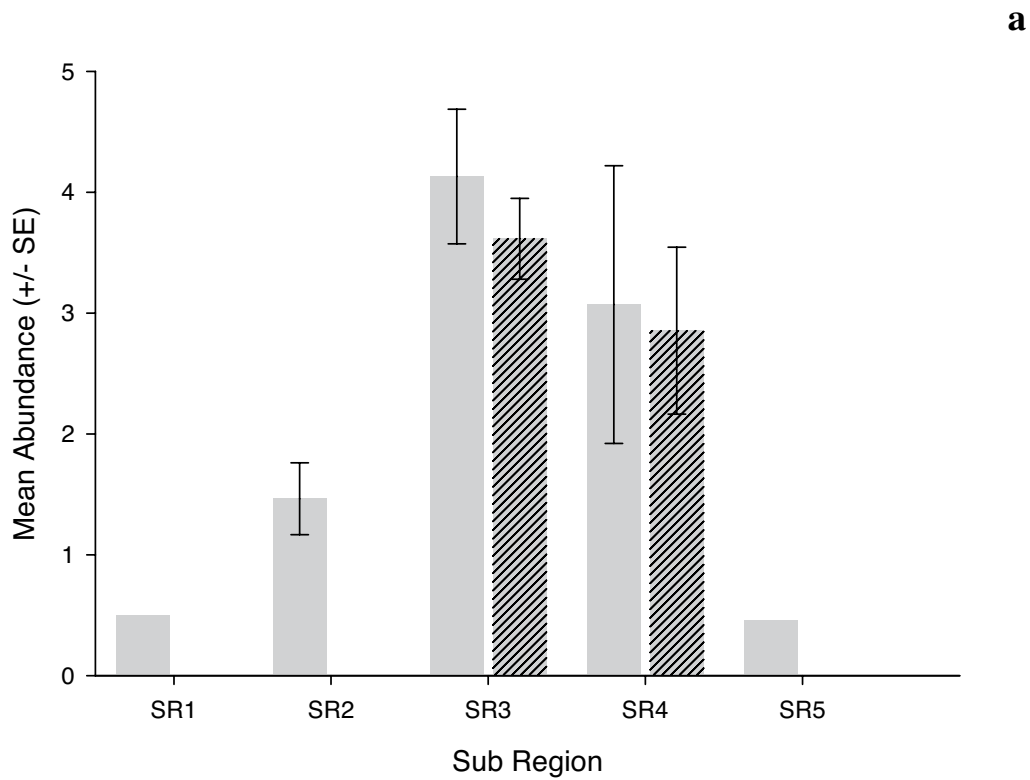
Figure 2. a) Mean abundance and b). size of western rock lobster (+/- s.e.) by subregions at Jurien Bay IBSS. Grey bars refer to lobster data derived from IBSS transect lines along which towed video classifications were undertaken. Hatched bars include all pot transects in sub-regions that were extensively surveyed by multibeam hydroacoustics (subregions 3 and 4).
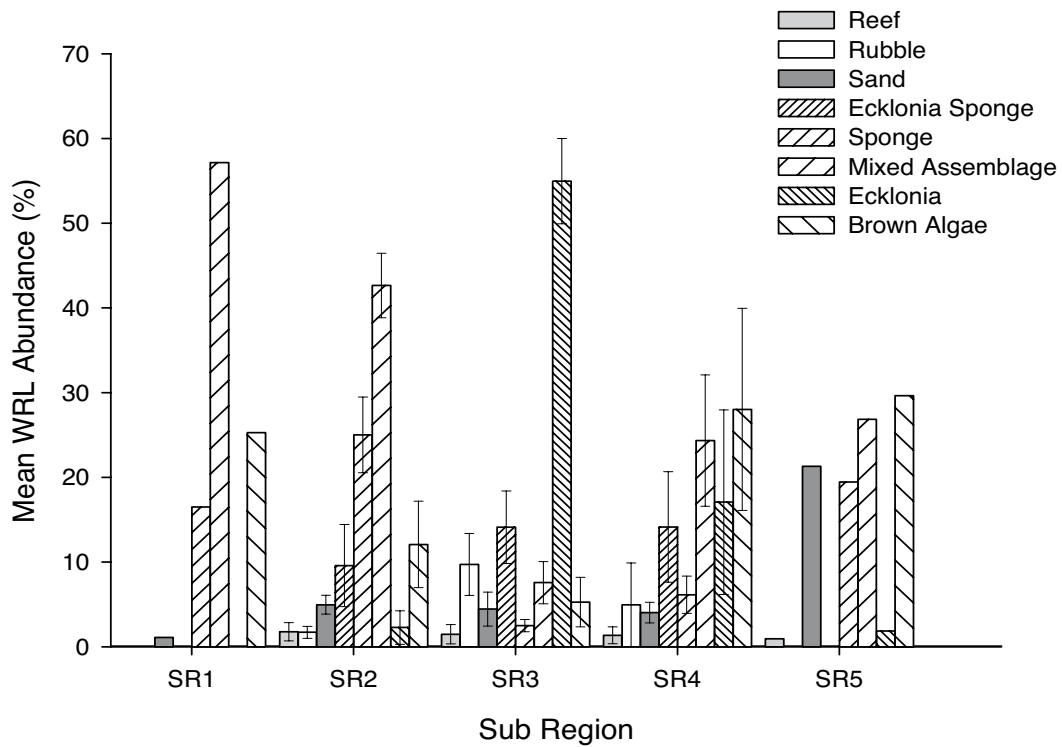
Figure 3. Mean % abundance of western rock lobster by aggregated habitat classes from towed video surveys (+/- se) for subregions 1-5.
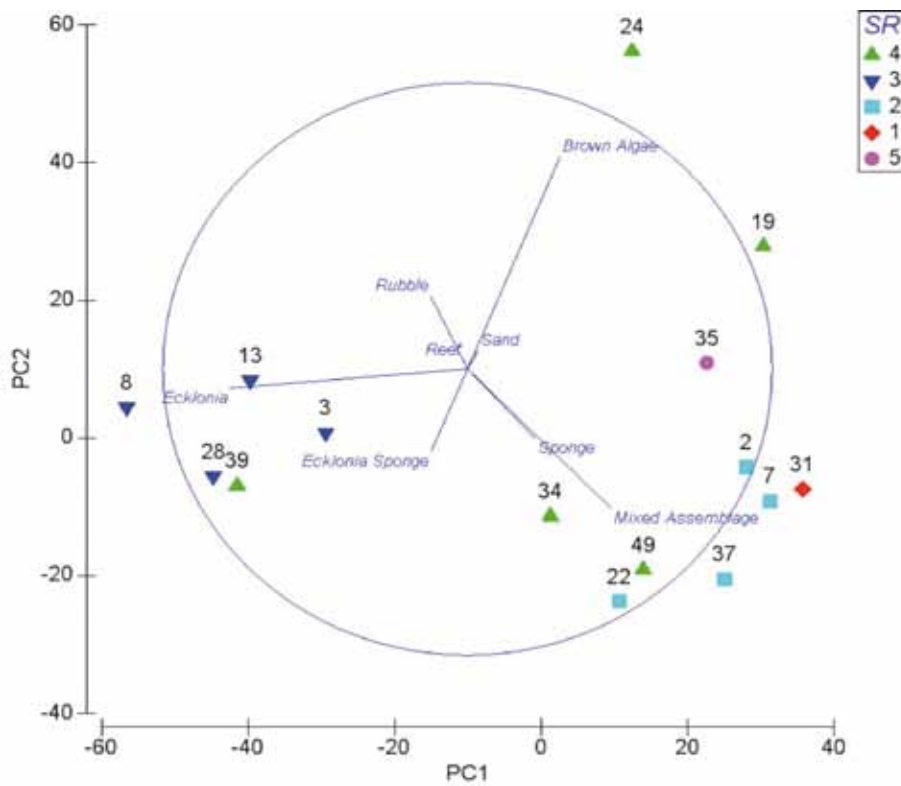
Figure 4. PCA of habitat data for towed video. The first two components account for 62.4% and 24.5% of the variance respectively, totalling 86.9%.
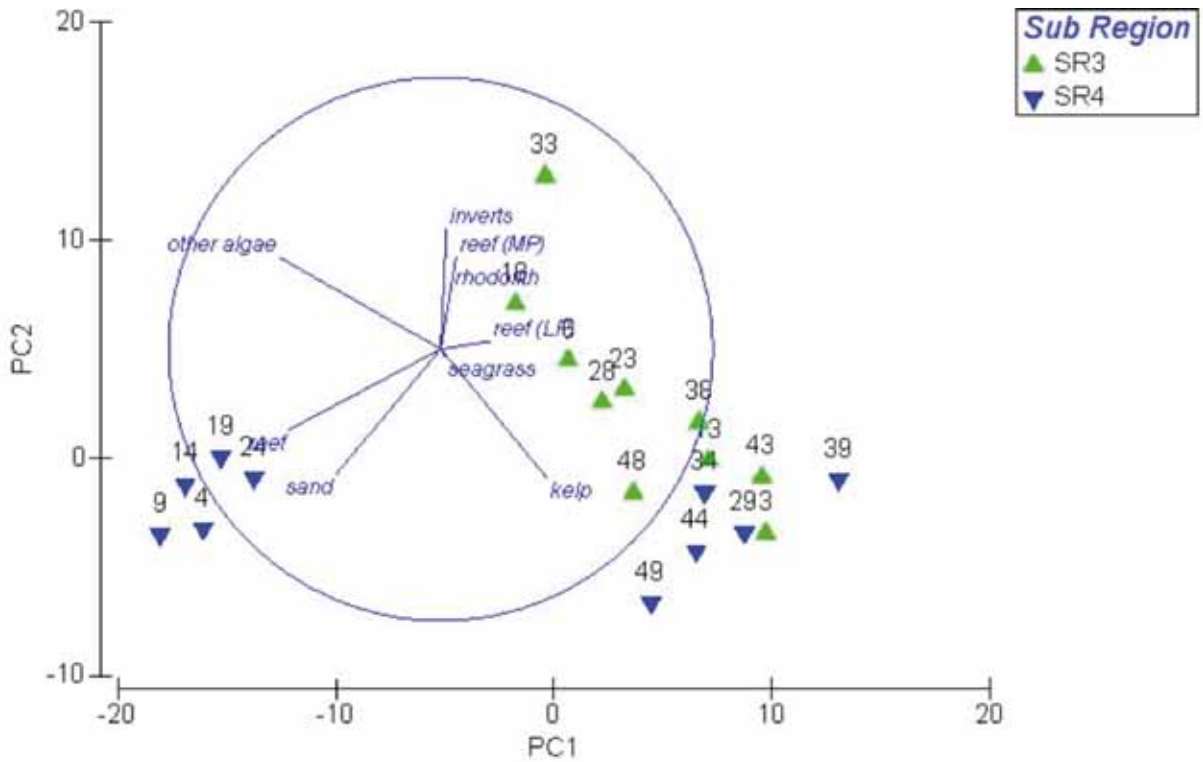
Figure 5 PCA of habitat data generated from habitat map. The first two components account for 71.6% and 13.7% of the variance respectively, totalling 85.3%.
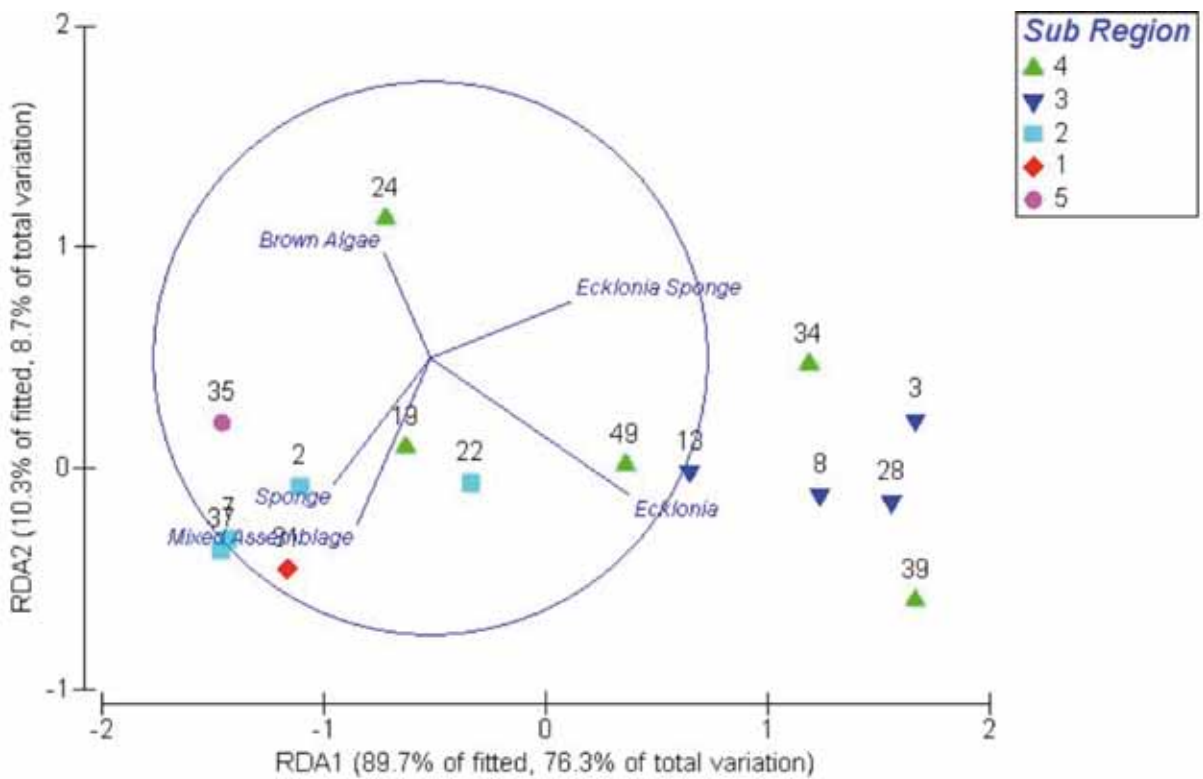
Figure 6. Lobster abundance and size as a function of habitat derived from towed video. A total of 85.1% of the variation in the data is accounted for by the first two RDA axes.
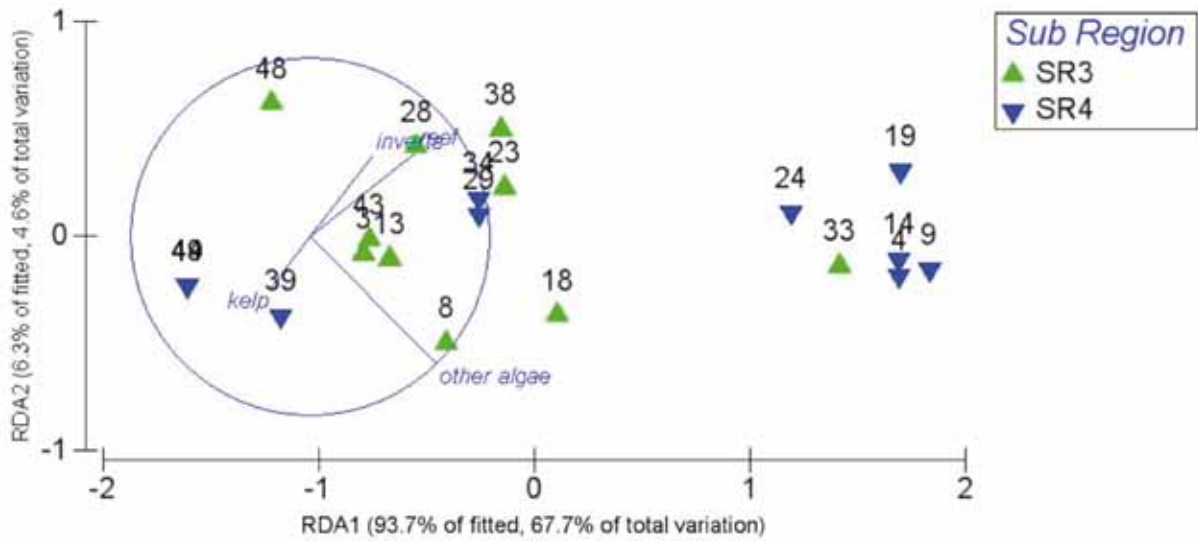
Figure 7. Lobster abundance and size as a function of habitat derived from Marine Futures. A total of 85.1% of the variation in the data is accounted for by the first two RDA axes.

6.0 Diet and trophic position of western rock lobsters (*Panulirus cygnus* George.) in Western Australian deep-coastal ecosystems (35-60 m) indicates they are more carnivorous than their counterparts in shallow water ecosystems

Waddington, K. I, Bellchambers, L. M., Vanderklift, M. A. and Walker, D. I.

6.1 Summary

The research contained in this chapter was initiated to address Objective 3: To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water. This study aims to provide preliminary identification of the important dietary items which will allow a better understanding of what types of habitat are important in terms of providing a food source for deep water lobsters. Understanding the trophic position of western rock lobster will also allow a better understanding of the direct and indirect interactions between lobsters and their predators and prey and therefore allow an evaluation of the impact of lobster biomass removal in the deep-water. In this study lobsters were primarily carnivorous, and no consistent differences in diet were detected with varying lobster size, sex or among locations. The main components of the diet were bait (from the fishery) and small crustaceans –crabs and amphipods/isopods. In contrast previous studies of the diet of lobster in shallow water environments have suggested that coralline algae and molluscs are important dietary items. Differences in lobster diet between shallow water and deep-coastal ecosystems have implications when assessing the effect of lobster removal on these ecosystems As lobsters inhabiting deep-coastal ecosystems are mostly carnivorous, removal of lobsters from these ecosystems will most strongly impact the macroinvertebrate community composition.

6.2 Abstract

The western rock lobster (*Panurilus cygnus*) is a conspicuous consumer in the coastal ecosystems of temperate Western Australia. We used stable isotope analysis and gut content analysis to determine the diet and trophic position of western rock lobsters from mid-shelf coastal ecosystems (35-60 m depth) at three locations. Lobsters were primarily carnivorous, and no consistent differences in diet were detected with varying lobster size, sex or among locations. The main components of the diet were bait (from the fishery) and small crustaceans –crabs and amphipods/isopods. Foliose red algae, bivalves/gastropods and sponges were minor contributors to diet. The diet of lobsters in deep-coastal ecosystems differed to results of previous studies of diets of lobsters from shallow water ecosystems. In particular, coralline algae and molluscs–important prey in studies of lobsters from shallow water–were minor components of the diet. These differences are likely to reflect differences in food availability between these systems and potentially, differences in choice of prey by lobsters that inhabit deeper water. Given the high contribution of bait to lobster diet, bait is likely to be subsidizing lobster production in deep coastal ecosystems during the fishing season.

6.3   Introduction

Knowledge of species' diets and trophic position is fundamental to understanding food webs. The composition of a consumer's diet provides insights into the transfer of energy through food webs, and into the ultimate sources of production supporting food webs (Polis and Strong 1996). Trophic position provides a general framework for understanding the direct and indirect interactions between predators and prey (Polis and Strong 1996).

Spiny lobsters are abundant consumers in many coastal ecosystems and an understanding the diet and trophic position of spiny lobsters is important as their feeding ecology can be important in determining ecosystem structure (Tarr *et al*. 1996; Tegner and Dayton 2000; Shears and Babcock 2002; Langlois *et al*. 2005). Predation by spiny lobsters has caused differences in the abundance and size structure of their prey in New Zealand (Shears and Babcock 2002; Langlois *et al*. 2005; Langlois *et al*. 2006b), Tasmania (Pederson and Johnson 2006), South Africa (Tarr *et al*. 1996; Mayfield and Branch 2000) and California (Tegner and Levin 1983). These changes in prey abundance can have indirect effects on other elements of the ecosystem (e.g. Babcock *et al*. 1999).

The diet of spiny lobsters can change with lobster size (Goni *et al*. 2001; Mayfield *et al*. 2001; Langlois *et al*. 2006b). Differences in choice of prey have been demonstrated for *Jasus edwardsii,* with larger lobsters tending to choose large prey and smaller lobsters tending to choose small prey (Langlois *et al*. 2006b). Such patterns may relate to an increased ability of larger lobsters to consume larger, hard-shelled prey (Robles *et al*. 1990), although prey choice may also be influenced by a relationship between energetic value of prey and energetic costs of prey capture and consumption (Hughes 1980). Changes in choice of prey with increases in lobster size was shown to affect prey community composition inside marine reserves where large lobsters are more abundant (Langlois *et al*. 2006a).

The western rock lobster (*Panulirus cygnus*) is conspicuous along the west coast of Australia (Phillips 1990). Previous studies have found that juvenile *P. cygnus* consume a wide range of benthic biota including molluscs, polychaetes, small crustaceans and coralline algae (Joll and Phillips 1984; Edgar 1990; Jernakoff *et al*. 1993). However, these investigations have focused on shallow water ecosystems (<5 m depth). The diet of lobsters in deeper coastal ecosystems (>35 m depth) has been poorly studied. The size structure of lobsters in these deep-coastal ecosystems differ significantly from those in shallow water. Deeper coastal ecosystems are occupied by a greater proportion of adult lobsters; approximately 25% of *P. cygnus* in deeper water (>35 m) are >80 mm carapace length (unpublished catch and effort statistics, Department of Fisheries Western Australia 2007), while the proportion of >80 mm *P. cygnus* in shallow water (LD MacArthur, unpublished data) is approximately 4%. In addition, approximately 40% of the commercial catch of *P. cygnus* is taken from depths >35 m (unpublished catch and effort statistics, Department of Fisheries Western Australia 2007). The differences in lobster size structure between deep and shallow coastal ecosystems may therefore result in differences in diet, and so differences in trophic interactions by lobsters. Because of this, the potential indirect effects of fishing between shallow and deep-coastal ecosystems may differ in important ways.

In this study, we used stable isotope and gut content analyses to determine the diet and trophic position of *Panulirus cygnus* in deep-coastal (35-60 m depth) ecosystems. Stable isotopes of carbon and nitrogen can help unravel complex food webs and identify important trophic relationships within ecosystems (Fry 1988; Jennings *et al*. 1997; Davenport and Bax 2002; Post 2002). Analyses of gut contents provides dietary information on a shorter time scale —

between ingestion and assimilation of food (Overman and Parrish 2001) — and are also useful in verifying results from stable isotope analyses (Whitledge and Rabeni 1997). The aim of this study was to determine diet and trophic position of *P. cygnus*, focusing on whether these varied spatially, or according to lobster size or sex.

6.4   Methods

6.4.1   Study area

This study was conducted at three locations on the west coast of Australia: Lancelin (30° 58.2 S, 114° 57.1 E), Jurien Bay (30° 12.5 S, 114° 39.1 E) and Dongara (29° 18.9 S, 114° 38.5 E). These locations span 200 km of coast near the centre of the distribution of *P. cygnus*. Four sites were selected at Lancelin and Jurien Bay and five sites were selected at Dongara, with sites separated by at least 2 km. Sites contained higher relief than the surrounding reef habitat, and were selected to maximize probability of encountering lobsters. The sites were located 20-40 km from the shore in 35-60 m depth. The sea floor is comprised of limestone reefs, which are remnants of Pleistocene/Holocene coastal sand dunes (Seddon 1972; Searle and Semeniuk 1985). Offshore reefs are typically low relief (<1 m relief) and are dominated by kelp, *Ecklonia radiata*, and sponges (Kris Waddington, unpublished data)

6.4.2   Collection

Divers breathing mixed gas (Enriched Air Nitrox, Trimix) from SCUBA apparatus collected biota at each site between 28th March and 10th April 2006. For reef biota, the entire contents of a 0.25m^2 quadrat were removed using a paint scraper and placed in a calico bag, ensuring no material was lost (n=2 per site for Dongara and Jurien Bay and n=3 per site for Lancelin). For sediment biota, cores (100 mm diameter × 200 mm deep) were collected from sediment adjacent to the reef (n=2 for each site). Sample sizes are small, reflecting the difficulty of sampling at these depths. At the completion of each dive, samples were frozen for later sorting in the laboratory.

The divers collected lobsters from three sites at each location. Lobsters were collected within 2 hours of sunrise using a noose and were between 53.7 and 144.6 mm carapace length (CL). Collection occurred soon after sunrise to minimise error associated with variable evacuation rates of gut contents (Waddington, unpublished data). Following collection, lobsters were immersed in an ice-slurry to induce a chill coma. Lobster size, sex and moult stage were recorded. Lobster foreguts were then removed and frozen for later gut content analysis. A sample of muscle tissue for stable isotope analysis was dissected from the tail and frozen. Additional lobsters were collected from Jurien Bay using unbaited pots. Pots were set overnight and retrieved within one hour of sunrise and foreguts and tail muscle removed as described above. Baited pots were unsuitable for collecting lobsters for gut content analysis as the lobsters fed on bait in the pots, and so gut contents would be biased. Exclusion of the bait using 'bait savers' attracted isopods (*Natatolana* sp.), which the lobsters fed on, also causing bias (Kris Waddington personal observation, 2005). However, baited pots were suitable for collecting lobsters for stable isotope analysis, and were used to collect additional lobsters from Lancelin and Dongara between 20th and 30th April 2006.

6.4.3   Stable isotope analyses

In the laboratory, biota collected from quadrats and cores were defrosted, sorted, and identified to at least family level. Sediment cores were sieved and potential lobster prey removed. Bulk

tissue of macroalgae, muscle tissue from tails of lobsters, and whole (or multiple whole) polychaetes, crabs, amphipods and isopods were used for stable isotope analysis. The flesh of imported mackerel (*Scomber* spp.) and Australian pilchards (*Sardinops sagax* Jenyns) – two baits commonly used in the fishery – were also analysed as they were possible lobster dietary items. All samples were rinsed in de-ionised water, dried in an oven at 60 °C until completely dry, then ground to a fine powder using a ball mill grinder. Samples containing non-dietary carbonates (crabs, amphipods, isopods, coralline algae) were treated with 1M HCl to dissolve these non-dietary carbonates (Bunn *et al*. 1995).

Continuous-flow isotope ratio mass spectrometry using Europa Scientific (Roboprep-CN/ Tracermass and ANCA-NT/20-20 units) and Isogas Sira 9 Instruments were used to measure $\delta^{15}N$ and $\delta^{13}C$. Most samples were analysed in dual isotope mode, allowing $\delta^{15}N$ and $\delta^{13}C$ to be measured simultaneously. Samples containing non-dietary carbonates were analysed for $\delta^{15}N$ prior to acid treatment, and analysed for $\delta^{13}C$ after acid treatment. Analytical precision of the instruments was 0.081 ‰ and 0.046 ‰ (± se) for $\delta^{15}N$ and $\delta^{13}C$ respectively. Cornflour, lobster muscle tissue and turnip calibrated against IAEE reference materials (IAEA-CH-6, IAEA-N-1, IAEA-N-2, USGS40, USGS41, USGS24) were used as internal standards for stable isotope analysis.

6.4.4   Defining lobster dietary sources

The mixing model software IsoSource (Phillips and Gregg 2003) was used to determine the contribution of each potential prey to lobster diet for each location (Lancelin n=25 lobsters, Jurien Bay n=19 lobsters, Dongara n=35 lobsters) (source increment 1%, tolerance 0.1). To reduce variability in mixing model outputs, we sought to combine similar diets prior to analysis. Only taxonomically related groups with similar life histories and feeding strategies were considered for combination (Phillips *et al*. 2005). The K nearest-neighbour randomization test was used to test for differences in $\delta^{15}N$ and $\delta^{13}C$ isotope signatures of those groups considered for combination (Rosing *et al*. 1998), and taxa were combined if $\delta^{15}N$ and $\delta^{13}C$ were not significantly different (p<0.05).

The IsoSource method is appropriate when the number of dietary sources = *i*+2, where *i* is the number of stable isotopes (Phillips and Gregg 2003). While no unique solution for the contribution of dietary sources exists, calculations yield the range of possible dietary source contributions to lobster diet (Phillips and Gregg 2003). Values for consumer-diet discrimination (2.57‰ for $\delta^{15}N$ and 3.20‰ for $\delta^{13}C$), determined from a separate experiment (Waddington and MacArthur, in press) were used to 'adjust' stable isotope values before input to IsoSource. Sites within locations were pooled for these analyses.

6.4.5   Trophic position of lobsters

A continuous measure of trophic position of *P. cygnus* was calculated. A continuous measure of trophic position is useful in ecological studies as assigning organisms to discrete levels ignores processes such as omnivory and diet shifts (Polis and Strong 1996; Vanderklift *et al*. 2006). The following formula modified from (Vander Zanden *et al*. 1997) was used to determine trophic position of lobsters:

$$\text{Trophic Position} = \frac{\text{lobster } \delta^{15}N - \text{macroalgae } \delta^{15}N}{2.57} + 1,$$

where 2.57 is the average consumer-diet discrimination between lobster tail muscle tissue and diet (Waddington and MacArthur, in press).

6.4.6   Gut content analyses

Lobster foreguts were defrosted, blotted dry and weighed. After removing the gut contents, the foregut membrane was blotted dry and re-weighed. A quantitative index of gut fullness (GFI) was calculated for all lobsters collected by divers and using unbaited pots as Mayfield et al., (2000)

$$GFI = \frac{\text{total foregut weight (g) - foregut membrane weight (g)}}{\text{total foregut weight (g)}} \times 100$$

The contents of lobster foreguts were rinsed into a 9.5 cm diameter petri dish and placed over a sheet with 60 randomly marked dots. The item over each dot was then identified to lowest possible taxonomic level using a dissecting microscope ($6.4\times – 40\times$ magnification), yielding a score out of a possible 60 for each prey (note that according to binomial probability, 60 points gives a 95% chance of recording a prey that makes up 5% or more of the gut contents (Vanderklift *et al*. 2006). The score for each prey was then multiplied by 100/60 to give percentage of each prey in the gut. Prey observed in the gut but not recorded using this method were assigned a value of 1%. Due to breakdown of dietary items in lobster guts, it was not always possible to identify prey to species level, and prey were more frequently identified to family level. Amphipods and isopod fragments could not be separated during identification so were combined.

All lobsters used in gut content analyses were in intermoult, and were caught by divers or using unbaited pots. Analyses were further restricted to lobster foreguts with GFI >10 to avoid biases introduced by individuals with guts containing few dietary items. Distance-based multivariate multiple regression, DISTLM (Legendre and Anderson 1999; Mcardle and Anderson 2001) was used to test for relationships between gut content composition and lobster size, sex, location of capture (Lancelin, Jurien Bay, Dongara), method of capture (unbaited pot, diver) and gut fullness (GFI). The analysis was based on Bray-Curtis dissimilarities and significance was determined by 4 999 permutations of the raw data.

6.4.7   Prey electivity

Ivlev's index of prey electivity (Ivlev 1961) was used to calculate electivity by *P. cygnus*. Ivlev's index of electivity (E) relates the proportional abundance of a prey on the benthos (determined from quadrats and sediment cores collected by divers) relative to the proportional abundance of that prey within a lobster gut as:

$$\text{Ivlev's index (E)} = \frac{r_i - p_i}{r_i + p_i},$$

where r_i represents the proportion of prey i on the benthos, and p_i represents the proportion of prey i in the gut of the lobster. Electivity of -1 indicates the prey is inaccessible, or there is total selection against the prey, while electivity of +1 indicates there is complete selection for the prey. A value near 0 indicates the item is consumed in proportion to its abundance on the benthos. Electivity was calculated for prey making up >1 % of gut contents. Due to insufficient sample size, lobsters at Lancelin and Dongara were not considered. Ivlev's index can be biased by different availability of food to predators, and by differences in prey digestion rates (Kohler and Ney 1982).

6.5   Results

6.5.1   Determination of diet using stable isotopes

The stable isotope values of the potential diets of western rock lobster were generally consistent between locations (Figure 1a-c). The spread of values was similar for all three locations (between -28 ‰ and -12 ‰ for δ^{13}C and between 4 ‰ and 11 ‰ for δ^{15}N). However at Dongara (figure 1c), δ^{13}C and δ^{15}N of amphipods/isopods were higher than the other two locations. Foliose red algae from Lancelin had lower δ^{13}C, and higher δ^{15}N than foliose red algae from the other locations.

At all locations gut content analysis indicated that lobsters were omnivorous, preying on amphipods/isopods, crabs, bait, foliose red algae and sponges. However, the proportional contribution of each diet, as estimated by IsoSource, differed among the three locations (Figure 2a-c). One consistent pattern was that bait, crabs, and amphipods/isopods were likely to be important components of the diet at all locations. Bait was estimated to have contributed between 30 and 57% of the diet of lobsters at Lancelin, between 62 and 79% at Jurien Bay, and between 4 and 70% of diet of lobsters at Dongara. Crabs (Lancelin 0-50%; Jurien Bay 0-26%; Dongara 0-76%), and amphipods/isopods (Lancelin 0-54%; Jurien Bay 0-23%; Dongara 0-52%) were also likely to be important diets at all three locations. Foliose red algae (Lancelin 6-25%; Jurien Bay 2-13%; Dongara 0-13%) and sponges (Lancelin 0-16%; Jurien Bay 0-11%; Dongara 0-15%) were likely to be of lesser importance. IsoSource also estimated that lobsters at Dongara might also prey on molluscs (bivalves and/or gastropods: 0-24%). These taxa were not observed in benthic samples collected from Lancelin or Jurien Bay and so were not included in the IsoSource analyses.

6.5.2   Trophic position

The trophic position of lobsters was calculated relative to the δ^{15}N value for macroalgae. The δ^{15}N values of red, green, and brown algae were consistent. Lobsters occupied the trophic positions expected by a first-order predator, with trophic position at each location between 1.90 and 2.18. Trophic positions varied significantly among locations (ANOVA: $F_{75, 2} = 7.724$, $p < 0.001$), although the magnitude of differences was small. Post hoc Tukey tests indicated that lobsters from Lancelin occupied a significantly higher trophic level (2.18 ± 0.06, n=25) than lobsters from Dongara (1.90 ± 0.05, n=35) ($p < 0.001$) while lobsters from Jurien Bay were intermediate (2.01 ± 0.04, n=18) and were not significantly different to lobsters from either Lancelin or Dongara.

6.5.3   Gut content analysis

Neither size, sex, location of capture, method of capture, nor gut fullness index were found to be significantly related to the composition of gut contents (n=30) (Tables 1 and 2). Thus all locations were combined for further analysis. The composition of food items in lobster foreguts were dominated by crabs (61.8%): bait (13.9%) and amphipods/isopods (9.1%) were other important diet items (Figure 3). Bivalves/gastropods, foliose red algae, sponges, and polychaetes each comprised less than 2% of gut contents (Figure 3).

6.5.4   Electivity

Lobsters at Jurien Bay exhibited clear electivity for some prey (Figure 4). Such differences may result from selection for or against prey, differences in accessibility of prey, or differences

in evacuation rates of prey from lobster foreguts. Amphipods/isopods and crabs were selected for by lobsters and/or were highly available to lobsters when foraging. Conversely, lobsters selected against polychaetes and/or polychaetes were less accessible to the lobsters during foraging. Bivalves/gastropods were selected for, although not as strongly as amphipods/ isopods and crabs.

6.6 Discussion

At the locations sampled, western rock lobsters in deep coastal (35-60 m) ecosystems were primarily carnivorous, with a diet consisting mainly of crabs, amphipods/isopods and bait, and small quantities of bivalves/gastropods, sponges and red algae. The diet of lobsters did not vary with sex, size or among locations. While conclusions drawn in the current study are based on data from a small number of quadrats and lobsters for gut content analyses, results of this study provide important quantitative information regarding the diet of western rock lobsters in previously inaccessible ecosystems.

Stable isotope analysis and gut content analysis indicated that bait is an important component of lobster diet, contributing up to 80%. Bait is available to the lobsters while the lobsters are in the pots as well as in the form of discards from the fishing fleet. The high relief sites we targeted will presumably also be targeted by fishermen, perhaps increasing the bait input on a localized scale. In addition, our surveys occurred during the months of April and May. While the commercial fishing season operates between 15th November and 30th June, maximum fishing effort in deep-coastal ecosystems occurs between January and May. Our survey occurred during peak fishing times in deep-coastal ecosystems, suggesting the average contribution of bait to lobster diet over the entire year may be lower. Considering the tissue turnover rate for decapod crustaceans is less than three months (Fantle *et al.* 1999; Waddington and MacArthur, in press), the high contribution of bait determined from stable isotope analysis might reflect higher consumption of bait during the fishing season. Nevertheless, given the likelihood of high bait contribution to lobster diet, and the known positive relationship between growth rate and food availability for *Panulirus cygnus* (Chittleborough 1976), bait input is likely to provide a significant subsidy to lobster growth in these ecosystems during the ~5 months that the fishing fleet is present. Bait has also been shown to subsidise production of American Lobsters (*Homarus americanus*) in the Gulf of Maine (Saila *et al.* 2002). In the Gulf of Maine, bait was estimated to meet between one-quarter and one-third of lobster food requirements (Saila *et al.* 2002), which is comparable to the results from the current study. When present, bait appears to be highly elected for by the lobsters. High electivity of bait by lobsters may be because lobster pots provide a ready source of food and shelter, two important resources for spiny lobsters (Chittleborough 1975; Eggleston and Lipcius 1992). However, due to spatial and temporal variability in bait addition, formal comparison of electivity is not possible using Ivlev's electivity index (Ivlev 1961).

Bait is imported from outside the study area, meaning it represents a direct subsidy to lobster production in these ecosystems (Saila *et al.* 2002). Addition of organic matter to marine ecosystems has been shown to have consequences for the functioning of marine ecosystems worldwide, primarily through the enhancement of secondary production from trawl discards (Groenewold and Fonds 2000; Ramsay *et al.* 1997). It is highly likely that addition of organic matter in the form of bait may be having similar effects in Western Australian ecosystems, particularly given the oligotrophic nature of these systems (Lenanton *et al.* 1991). Further studies should be undertaken to investigate the potential effects of bait input in these systems.

The natural diet of western rock lobsters was dominated by crabs and amphipods/isopods, with sponges, algae, gastropods/bivalves and polychaetes less important. Crabs and amphipods/isopods were strongly selected for – or alternatively highly available to lobsters – relative to gastropods/bivalves, polychaetes, sponges and foliose red algae. Given the high biomass of sponges and red algae on the reef where lobsters were collected, the low importance of red algae and sponges to lobster diet is likely due to low selection for these taxa. Polychaetes and gastropods have previously been shown to be important lobster prey in shallow water (Joll and Phillips 1984; Edgar 1990; Jernakoff *et al*. 1993), suggesting the low proportion of these items observed in diet of lobsters in this study might be the result of low availability. Bivalves and gastropods were not frequently observed in benthic samples collected by divers in this study. While polychaetes were observed in samples collected by divers, they were most frequently observed within sponges collected from the benthos. A hypothesis for the low importance of polychaetes in lobster diet may relate to the burrowing habit of polychaetes (Netto *et al*. 1999; Abdo 2007), providing a refuge from predation by lobsters.

The two techniques of dietary analysis employed in the current study gave differing outcomes for the proportional contribution of prey to lobster diet. Analysis using stable isotope data indicated bait was the most important component of lobster diet whereas gut content analysis indicated crabs were more important. Observed differences likely reflect the different time scales over which the two techniques calculate dietary composition (Overman and Parrish 2001) and the variability in evacuation rates of prey from lobster guts (Waddington, in press). Gut content analysis provides an indication of lobster diet between ingestion and assimilation of prey whereas stable isotope analysis provides a time integrated description of lobster diet over the time scale equivalent to the tissue turnover rate of the tissue analysed (Kling *et al*. 1992; Overman and Parrish 2001). At the time lobsters were collected for gut content analysis crabs were the most important lobster prey. When the diet of lobsters was integrated over a longer period, bait was more important to lobster diet reflecting the spatial and temporal variability of bait input to these ecosystems. The relative contribution of crabs to lobster diet determined from gut content analysis may also be overestimated due to variability in evacuation rates of prey from lobster foreguts (Waddington, in press). A recent study indicated that prey with hard components were more slowly evacuated from lobster foreguts relative to diet items without these hard components (Waddington, in press). This suggests the relative contribution of crabs and bait may be overestimated in the current study relative to prey such as foliose red algae that are rapidly evacuated from lobster guts.

Diet of western rock lobsters in the current study differs to diet of western rock lobsters from shallow water ecosystems (<10 m) (Joll and Phillips 1984; Edgar 1990; Jernakoff *et al*. 1993). Observed differences in diet between these ecosystems occur despite overlap in the lobster size range. The size range of lobsters in the current study (between 53.7 and 144.6 mm CL) overlaps the size range of lobsters previously investigated from shallow water (25 – 90 mm CL) (Edgar 1990). Since differences in diet occur between shallow and deep-coastal ecosystems (despite overlap in lobster size) it is unlikely lobster size/ontogenetic stage is driving observed differences in lobster diet. Lobsters in deep-coastal ecosystems predominantly consume animal prey (crabs, amphipods/isopods, and bait), with algae less important to lobster nutrition. Observed differences in lobster diet between shallow water and deep-coastal ecosystems are mirrored by the trophic positions occupied by lobsters in these ecosystems. In deep-coastal ecosystems at Jurien Bay, lobsters had a trophic position of 2.01 reflecting their role as secondary consumers in these ecosystems. Lobsters from shallow water ecosystems at Jurien Bay had a trophic position between 1.50 and 1.60 (Lachlan MacArthur,

Edith Cowan University, personal communication) reflecting the importance of plant sources (primarily coralline algae) to lobsters in shallow water ecosystems (Joll and Phillips 1984; Edgar 1990; Jernakoff *et al.* 1993).

Differences in lobster diet between shallow and deep-coastal ecosystems may reflect differences in prey availability or prey choice between these ecosystems. Molluscs comprise a high proportion of gut contents of lobsters from shallow water ecosystems (Joll and Phillips 1984; Edgar 1990; Jernakoff *et al.* 1993), but were poorly represented in gut contents in the current study. This low consumption of molluscs likely reflects low abundances of molluscs in benthic samples collected from Lancelin and Jurien Bay. Two species of coralline algae commonly consumed by lobsters in shallow water ecosystems (*Jania affinis* and *Amphiroa anceps*) (Joll and Phillips 1984) were not observed to be consumed in the current study, despite being present in deep-coastal ecosystems (average biomass ~80 g. m^{-2}). This indicates differences in diet may also reflect differences in prey choice between shallow and deep-coastal ecosystems.

Differences in lobster diet between shallow water and deep-coastal ecosystems have implications when assessing the effect of lobster removal on these ecosystems. Removal of spiny lobsters (through fishing) reduces predation pressure on lower trophic levels (Tegner and Dayton 2000). Differences in spiny lobster abundance due to differences in exploitation rates have been shown to have detectable effects on abundance of spiny lobster prey in California (Tegner and Levin 1983), South Africa (Mayfield and Branch 2000), New Zealand (Shears and Babcock 2002; Langlois *et al.* 2005), and Tasmania (Pederson and Johnson 2006). As lobsters inhabiting deep-coastal ecosystems are mostly carnivorous, removal of lobsters from these ecosystems will most strongly impact macroinvertebrate community composition. In contrast, the omnivorous feeding behaviour of lobsters in shallow water ecosystems means removal of lobsters from these ecosystems will likely equally affect both the macroinvertebrate community composition and abundance of coralline algae in these ecosystems.

6.7 Tables

Table 1. Distance-based multivariate multiple regression (DISTLM) testing for the significance of the amount of variation in the composition of lobster guts contents that was explained by lobster size, lobster sex, location of capture, and method of capture. p-value determined by 4,999 permutations of the raw data.

Source of Variation	df	SS	MS	pseudo F	p value
Factors	5	12 268	2 453	1.30	0.216
Residual	19	35 749	1 881		
Total	24				

Table 2. Percentage of diet categories in lobster foreguts at each location. All lobsters were caught by divers or in unbaited pots.

	Lancelin n=5 53.7-114.5 mm CL	Jurien Bay n=19 54.1-81.9 mm CL	Dongara n=6 64.1-144.6 mm CL
Prey Item	mean (± se)	mean (± se)	mean (± se)
Crab	79.19 ± 10.61	54.90 ± 8.61	79.88 ± 7.16
Amphipods / Isopods	6.11 ± 4.73	10.01 ± 5.56	3.33 ± 3.12
Bait	8.02 ± 3.22	18.27 ± 7.42	8.9 ± 3.05
Sediment	0	9.37 ± 4.48	0
Bivalves/Gastropods	0.56 ± 0.43	1.61 ± 0.62	7.78 ± 4.84
Algae	2.22 ± 1.72	0.70 ± 0.34	0
Sponge	2.22 ± 1.72	0.94 ± 0.43	0

6.8   Figures
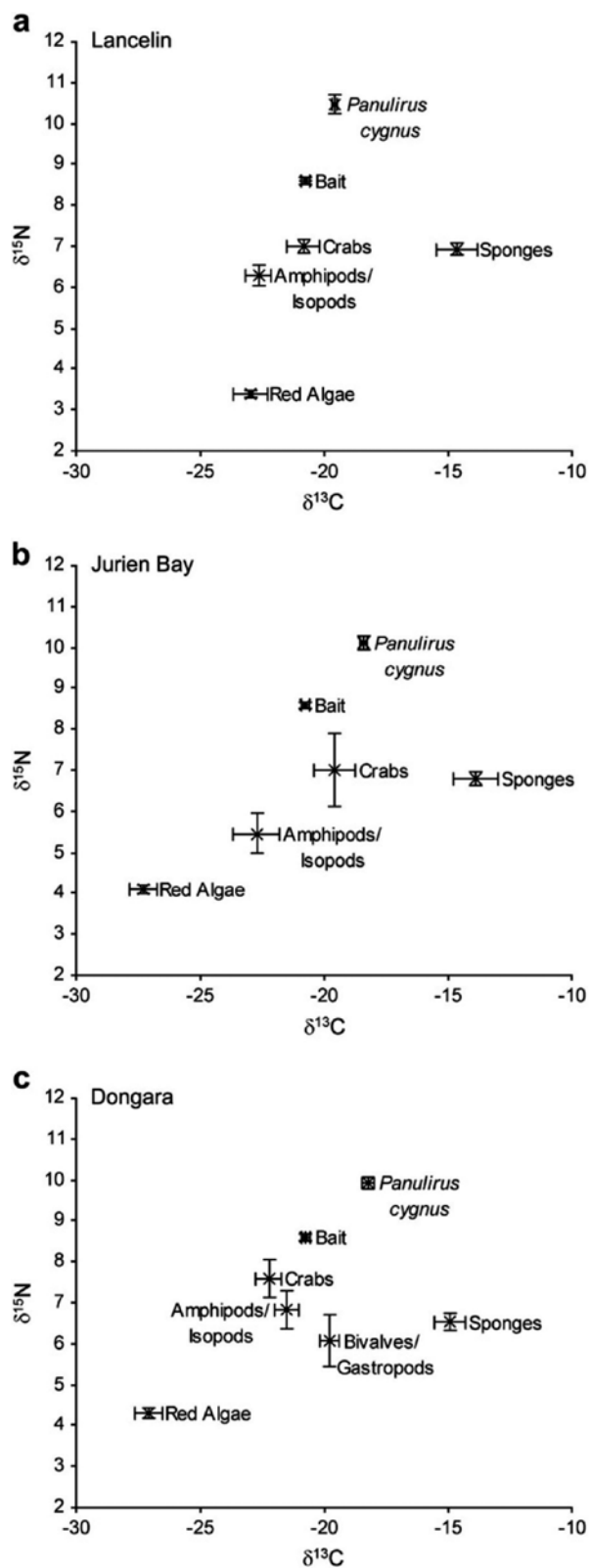
Figure 1.   δ¹³C and δ¹⁵N of western rock lobsters and potential prey in deep coastal ecosystems off (a) Lancelin, (b) Jurien Bay, and (c) Dongara, Western Australia. Prey comprising <1% of diet (determined by gut content analysis) are not shown.
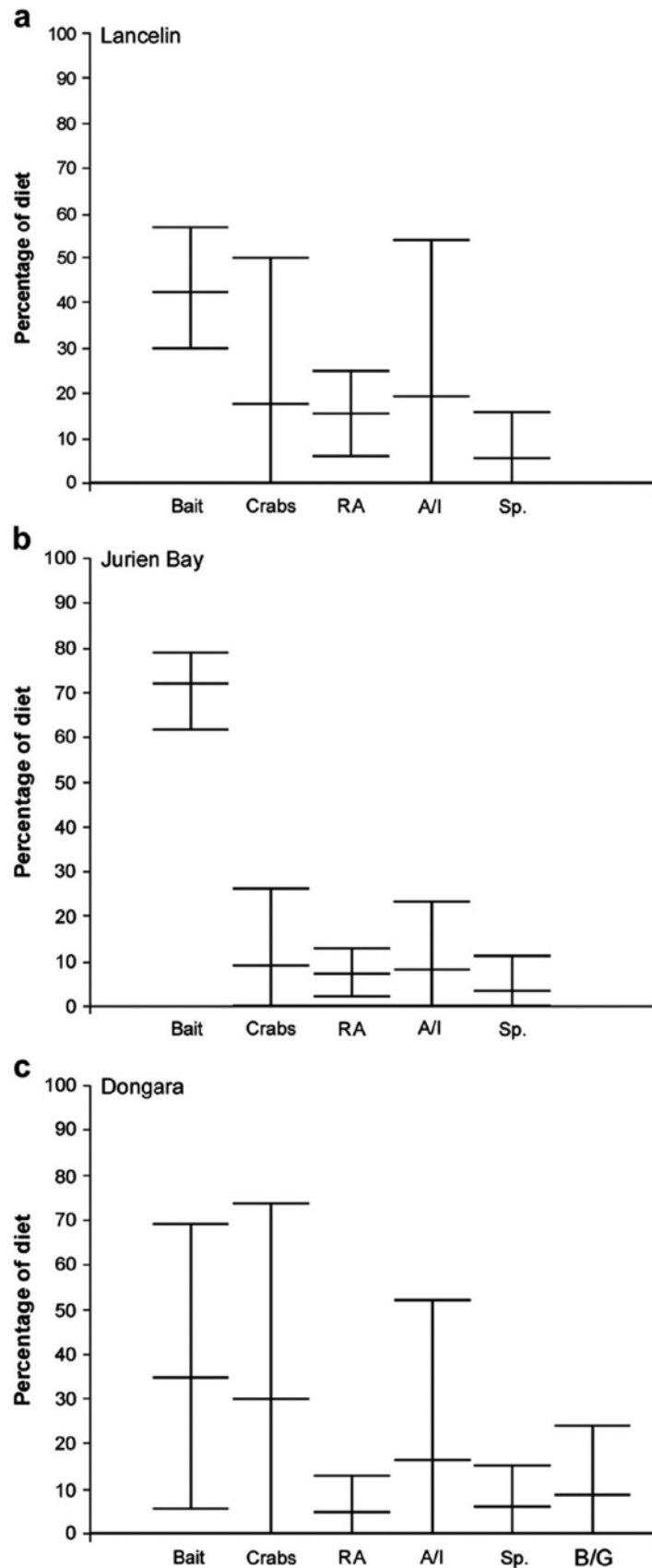
Figure 2. Distribution of feasible proportions of contribution of potential prey to diet of lobsters collected from (a) Lancelin (b) Jurien Bay and (c) Dongara. Outside tick marks represent range of feasible proportions (1-99%). Midline represents mean of feasible proportions. RA = Red Algae, A/I = Amphipods/Isopods, Sp. = Sponge, B/G = Bivalves/Gastropods.
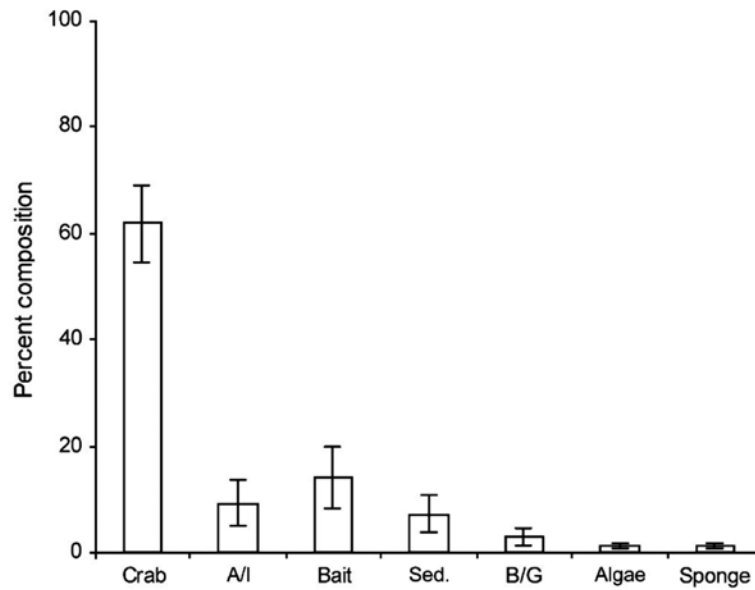
Figure 3. Percentage (mean ± se, n=30) of diet categories in lobster foreguts at all locations. All lobsters were caught by divers or in unbaited pots. Diet categories comprising <1% of diet are not shown on graph. A/I = Amphipods/Isopods, Sed. = Sediment, B/G. = Bivalves/Gastropods.
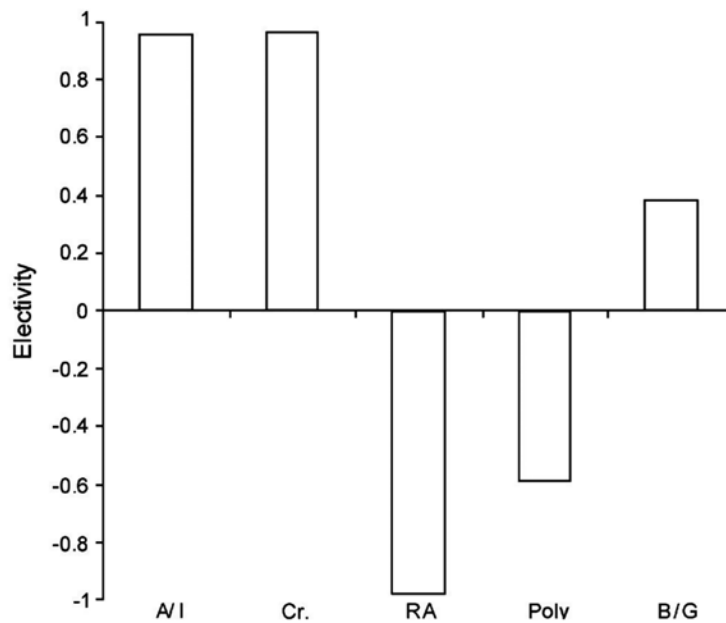
Figure 4. Ivlev's index of prey electivity for taxa observed in the guts of lobsters collected from Jurien Bay (n=19). A/I = Amphipods/Isopods, Cr. = Crabs, RA = Red Algae, Poly = Polychaetes, B/G = Bivalves/Gastropods.

7.0 Depletion-based shallow and deep-water estimates of catchability of western rock lobsters at the Abrolhos Islands

Wright, I., de Lestang, S., Caputi, N. and Bellchambers, L. M.

7.1 Summary

This chapter aims to address Objective 2: To assess the catchability of western rock lobster and its relationship with population abundance and size structure. Deepwater and shallow water catchability of western rock lobster was examined at the Abrolhos Islands using a DeLury depletion model. This study indicates that the catchability of lobsters in the shallow waters at the Abrolhos Islands was significantly increased by the 18% pot reduction in 1993/94 This indicates that there is a significant level of pot saturation occurring in the shallow waters of the Abrolhos. This result suggests that there may be some economic benefit from further pot reductions in shallow waters at the Abrolhos Islands and these are currently being examined for the 2008/09 season. The progressive decline in catchability in the deeper waters of the Abrolhos Islands may reflect a slow increase in competition between pots within this area as more of the fleet look towards deeper waters to retain high catch rates.

7.2 Introduction

The basic DeLury method for estimating catchability and population size is based on the relationship between the decline in CPUE and the cumulative catch that is removed from the fishery (Ricker 1975). The slope of this (linear) relationship provides an estimate of the catchability of the unit of fishing that also reflect the efficiency of effort applied by the fishing fleet.

Depletion techniques can be used in a controlled experimental approach to estimate catchability of the fishing gear such as prawn and scallop trawl (Penn and Joll 1990) and stock abundance. It can also be used with data from a normal fishing operation provided the basic assumptions of the technique are satisfied and there is a significant catch removed that has an impact on the abundance as measured by the CPUE (Ricker 1975). Wright *et al.* (2006) provided an estimate of the catchability for each of the three zones of the western rock lobster (WRL) fishery. As there is a difference in the size structure and abundance of lobsters in deep water and there are also differences in fishing efficiencies between the two depths, this study examines the depletion analysis by depth to compare the catchabilities. Since the offshore Abrolhos Islands region of the WRL fishery results in significant decline in CPUE due fishing in a relatively short period (3.5 months), this location has been chosen to examine catchability dynamics between shallow (< 20 fm) and deep (>= 20 fm) water using the DeLury depletion model.

The WRL fishery has two distinct catch phases: a migratory phase (November–January), "Whites", and a non-migratory phase (February–June), "Reds", with a synchronous moult occurring at the start of each phase when large numbers of lobsters reach legal size (Caputi *et al.* 1994). The non-migratory phase after the February moult provides an opportunity for using the depletion assessment as it generally satisfies the key assumptions. It has been undertaken using the mandatory monthly catch and effort of the Abrolhos Islands zone during the months of March–June for each fishing season since 1983/84.

The catchability of the western rock lobster is affected by environmental factors such as temperature, swell, and moon phase (Morgan 1974, Srisurichan 2001, Srisurichan et al 2005). These factors vary considerably within seasons and often have a different pattern between seasons. The depletion model depends on CPUE changes that reflect changes in abundance and which must not significantly distorted by environmental or biological effects on catchability. The assessment does not produce reliable results unless these catchability changes are allowed for, and the CPUE is standardised for environmental changes between months and between years. Another complicating factor is the timing of the February moult and its impact on the availability of recruiting lobsters during March. If the moult is late in any year, the assumption of no recruitment during the period is not met and this needs to be taken into account for that year's depletion assessment.

7.3 Methods

7.3.1 Depletion model

The classical DeLury depletion model has:

$$C_t/F_t = q\, B_t$$

where B_t is the biomass present at time t.

For our estimation procedure we use a reformulation of this model based on the following quantities:

C_t = catch in month t in tons;

F_t = effort in month t in thousands of pot lifts;

B_t = average biomass in month t;

q = catchability standardised for temperature and swell;

B_0 = total biomass in zone at start of exploitation (March);

K = scale factor that correctly sets the coefficient of variation of the monthly catches in this model and through this the standard errors in our parameter estimates. The method for determining this parameter is considered presently in the assumptions section.

$$\text{Taking } B_t = B_0 + 0.5C_t - \sum_{i=1}^{t} C_i$$

where the index i ranges over the time period defined, our model becomes:

C_t has the probability distribution K Poisson ($q\, F_t\, B_t / K$) for month $t = 1,2,3,4$ (i.e. March –June).

The parameters B_0 and q in the above equations were estimated using the program WINBUGS 3.0. By formulating the problem in a Bayesian framework, we were able to avoid optimisation fine-tuning problems (and more importantly automatically obtain standard errors) by using Bayesian estimation via the Markov Chain Monte Carlo (MCMC) method. The rationale for using the MCMC method was the automatic estimation of standard errors for our parameters, regardless of the model type and complexity.

There were 23 depletions to be estimated during the period 1984-2006. For this production phase, the estimation was performed by the program WINBUGS 3.0.

The parameter estimates were effectively the medians of a sample of 10 000 from the simulated posterior distributions based on over-dispersed priors. The priors were exponential distributions with large means. The standard errors for the estimates were the observed standard deviations of the posterior distributions of the parameters. The sample of 10 000 consecutive simulated values used for our estimates followed a "burn-in" period of 20 000 samples. Following this burn-in we confirmed that the convergence diagnostics were comfortably within required ranges. Further information can be obtained from Wright *et al.* (2006).

Approximately 30% of fishers operating in the Zone fill in voluntary daily logbooks and we assume that these are a random sample of all fishers in the zone. The logbook fishers report their catches and the effort in various depth categories for each day from which we determine the total monthly catches of logbook holders in deep and in shallow water, and the average catch per unit effort of those catchers in the two depths.

From the relative proportions of the catches of the logbook fishers catches in deep and shallow water we can divide the total monthly zone catch for all fishers in the same proportion into deep and shallow. The effort attributed to these catches is estimated by dividing the catches in each depth category by the relevant logbook CPUE in that depth. These catches and efforts are then used in a depletion analysis to determine catchability for each year. The monthly bottom temperatures and swell values used for the whole zone to standardize the CPUE are separately applied to the deep and shallow parts of the zone in these depletion estimations.

7.3.2   Assumptions and some consequences

The assumptions of the depletion method are: (1) closed population: i.e., no migration, recruitment or natural mortality; and (2) sample is homogenous with effort, i.e., probability of given individual being caught when population subject to f units of effort is $k\Delta f + o(\Delta f)$ where k is a constant independent of time; (3) all individuals have the same chance of being caught in the ith sample.

The result of these assumptions is that if N_t is the average size of the population while the catch of the ith fisher in a specified month is being acquired through the application of effort $f_{i,t}$, then, following the standard arguments used by Cox and Miller (1965) or Ross (1985), it is readily shown that the probability distribution of $C_{i,t}$ the number caught by the ith fisher in that month has a Poisson distribution with mean $q\,f_{i,t}\,N_t$ where q is the constant catchability. In fisheries parlance q is the (constant within season) catchability of each animal in relation to one potlift, equal to a unit of nominal effort.

If there are n fishers operating in a zone the total number of animals caught in month t is $C_t = \partial_i\,C_{i,t}$ and their total effort is $F_t = \partial_i\,f_{i,t}$. If their catches are independent, then following the standard arguments above, it follows that the total monthly catch has an approximately Poisson distribution with mean $q\,F_t\,N_t$. Conversely, if the catches are perfectly correlated (which is a much more likely scenario and which we assumed) the total monthly number of animals caught is approximately distributed as n times Poisson with mean $q\,F_t\,N_t\,/n$. The mean of this Poisson distribution is the average monthly catch of animals per fisher in the Abrolhos Islands zone (10 000). The resulting distribution has coefficient of variation around 1%.

When considering the catch in terms of biomass (tonnes), it can be readily shown that with a conversion rate of 2000 animals per ton, the monthly catch in tons will be distributed as K times a Poisson distribution with parameter equal to average monthly catch (ton) divided by K, where K is determined so that the coefficient of variation of the biomass distribution is equal to that for the distribution of animals described above. In Zone A the average monthly catch is 480 t so that K = 1/21. This scale factor has an important influence on the standard errors of our parameter estimates, but does not affect the median values.

When the t-th sample (zone monthly catch) represents a significant proportion of the remaining population, the value of N_t is taken to be the average value of the population during the sampling period. This improvement owing to Braaten (1969) was endorsed by Ricker (1975). Our estimation process includes this feature in both its frequentist and Bayesian versions.

7.3.3   Environmental influence on catchability

We consider three main environmental influences on catchability: temperature, swell and moon phase. The swell index is the average throughout the month of swell levels recorded in fishers' daily logbooks on the scale: (0=none, 10=Low, 20=Moderate, 30=High). The average monthly water temperature is determined by fisheries staff in conjunction with other local monitoring activities.

In Wright *et al.* (2006) we derived corrections for the impact on catchability of: temperature alone (with all other environmental variables held constant); swell alone (with all other environmental variables held constant); and temperature and swell together (with all other environmental variables held constant).

There is widespread anecdotal evidence that the catchability of western rock lobster increases with swell, and this matter was investigated in Srisurichan (2001) and Srisurichan *et al.* (2005). We also investigated the effect of swell on catchability using fishers daily logbook data and examined the linear regression between the daily values of the natural log of catch rate within each zone and both the swell size and moon brightness within single months between March and June, when it was reasonable to assume that the water temperature within each month was constant. The possible temperature dependence of the result was thus removed. However the results showed significant volatility in estimates of the coefficient of the swell term. For these reasons it was decided to use the zone averages of the swell coefficient over the relevant months of several years to make the swell correction of catchability needed in our depletion model.

Because we were using monthly catch and effort data, the moon phase influence on catchability was averaged across each full month and thus was not explicitly required for months where fishing takes place over the whole month. Over incomplete fishing months we needed to take the effect of reduced catchability on the full moon into account.

7.4   Results

7.4.1   Environmentally corrected catchability

The temperature and swell corrections for catchability derived in Wright et al 2006 provide the following temperature and swell correction formulae for Zone A:

$q_1 = q_0 \times \exp(0.097 \times \Delta T + 0.010 \times \Delta S)$

where ΔT = temperature increment above 21.4°C and ΔS = swell index increment above 9.5.

The combination of full moon and cold period in June may result in fishers ceasing fishing early. This can result in an elevated catch rate for the influential final catch, which may be reduced by some animals beginning to moult. Also the March catch at the Abrolhos, which covers a 17 day period, is at the start (and thus less influential part) of the depletion curve (Ricker 1975).

7.4.2   Depletion based estimates for catchability

The depletion method was applied to all licensed fisher's monthly catch and effort returns in < and >= 20 fm of water for each of the months between March and June of each season to estimate the catchability in the period 1984-2006 (Figure 2). A weighted-smoothing (3-year moving average weighted by 1/variance) was used to minimise the effect of the variability in annual estimate and that of standard errors in estimates in assessing the underlying trend.

Estimates of catchability in the two depths regions of the Abrolhos Islands displayed markedly different trends. In the shallow (< 20 fm) waters the catchability coefficient remained low throughout the 1980s and early 1990s before increasing rapidly from 1994 to its maximum in 2000. In the following years this estimate has declined to a level halfway between its maxima and minima (Figure 2). In the deeper (>= 20 fm) waters of this region the catchability coefficient progressively declined over the entire time series from being above that estimated for the shallow waters in the mid 1980s, to well below the shallow waters estimates in the late 2000s (Figure 2).

7.5   Discussion

One of the advantages of the depletion approach to estimating catchability and exploitation is that it takes into account changes in fishing efficiency over the years. Although fishing power studies have estimated the significant improvements in fishing power in the WRL fishery owing to technology changes such as colour echo sounders and GPS (Brown *et al.* 1995; Fernandez *et al.* 1997), these techniques cannot be used to estimate the continuous smaller improvements in efficiency that are occurring each year. These include improvements in skipper ability, increased bait usage, more mobile fleet, and accumulating good fishing locations in computer plotters.

This issue is particularly important in the management of limited-entry fisheries that require regular adjustment to the nominal effort to maintain the harvest rate at a consistent level. Estimation of effective effort and harvest rates from age or length-based modelling approaches requires an independent estimate of the annual efficiency rate increases (Hall and Chubb 2001).

The depletion method cannot be used in all fisheries, as it requires that some key assumptions are met or adjustments are made to the raw data. In the WRL fishery it was assessed that the technique was applicable only during the non-migrating phase of the fishery March–June after the synchronous moult in February when large numbers of sub-legal animals reach legal size. The growth during the March–June period is also expected to be minimal. However in some years it was observed that the moult appeared to be slightly later and some moulting was delayed until March. In these years the depletion from April to June was examined.

Although some natural mortality would occur over the 4 months this can be considered minimal in a relatively long-lived species and would generally be consistent between years and thus would not affect the trends in exploitation over the years. The depletion model can be adjusted

to take into account natural mortality if this is regarded as a significant bias. Natural mortality of 1.5% per month on numbers of animals would be offset by size and temperature dependent monthly weight gains by individuals which preliminary estimates from tagging studies place as high as 3% per month.

The need for standardised catch rates that take into account environmental factors affecting catchability is thus apparent. Water temperature decreases during the March–June period and exacerbates the decline in catch rate during this period owing to the lower catchability associated with lower water temperatures (Morgan 1974). If the correction for water temperature change is not undertaken it results in a lower estimate of residual biomass and higher estimate of harvest and catchability. On the other hand, catchability is enhanced with increases in swell conditions (Srisurichan *et al.* 2005), and this may bias the parameter estimates if not taken into account. Moon phase is a third factor affecting catchability and the use of monthly catch rates eliminates the impact of this factor except for March at the Abrolhos Islands where fishing is for the last 17 days.

Despite the adjustments made to standardise the monthly catch rates, there was still a significant variation in the annual parameter estimates for most of the parameters. The use of the weighted-smoothed estimates was employed to improve the signal to noise ratio of parameters used for assessment of the trends in the fishery.

As a result of the high exploitation and the large number of pots operating in a relatively restricted area, the catchability of rock lobsters in shallow water was significantly increased by the 18% pot reduction in 1993/94 (Figure 2). This jump in catchability was not apparent in the deeper waters that are generally larger in fishable area and where the competition between pots is not as intense as in the shallows. This increase in catchability at the time of the pot reduction was also apparent when the whole zone was analysed (Wright *et al.* 2006). This study indicates that this increase is confined to the shallow water component of the fishery. This indicates that there is a significant level of pot saturation occurring in the shallow waters of the Abrolhos. This result suggests that there may be some economic benefit from further pot reductions in shallow waters at the Abrolhos Islands and these are currently being examined for the 2008/09 season. The progressive decline in catchability in the deeper waters of the Abrolhos Islands may reflect a slow increase in competition between pots within this area as more of the fleet look towards deeper waters to retain high catch rates. There has been a significant steady increase in catch in the Abrolhos region in recent years with the three highest catches occurring in 2004/05 to 2006/07. Much of this increase occurred in the deep water as a result of migration of whites at a smaller size due to warming temperatures and due to an increase in the legal minimum size in the whites in 1993/94 (Caputi *et al.* in prep). This increase in abundance would have the effect of spreading the Abrolhos deep water fishing effort over a larger area and this would result in a decline in catchability.

These depletion estimates can be used to improve the stock assessments undertaken using alternative stock assessment methods such as length-based models (Hall and Chubb 2001), which have used a constant catchability coefficient. Although Hall and Chubb (2001) assumed an increase in efficiency of 1-2% per year, these increases could be improved using the trends in catchability in the depletion assessment. The jump in catchability it the shallow waters of the Abrolhos Islands since 1993/94 could also be used in the length-based assessment leading to a more robust assessment of the future management of the fishery. There is a new spatial population model under development that takes into account depth and these differences in catchability by depth are valuable information for these models.

7.6   Figures
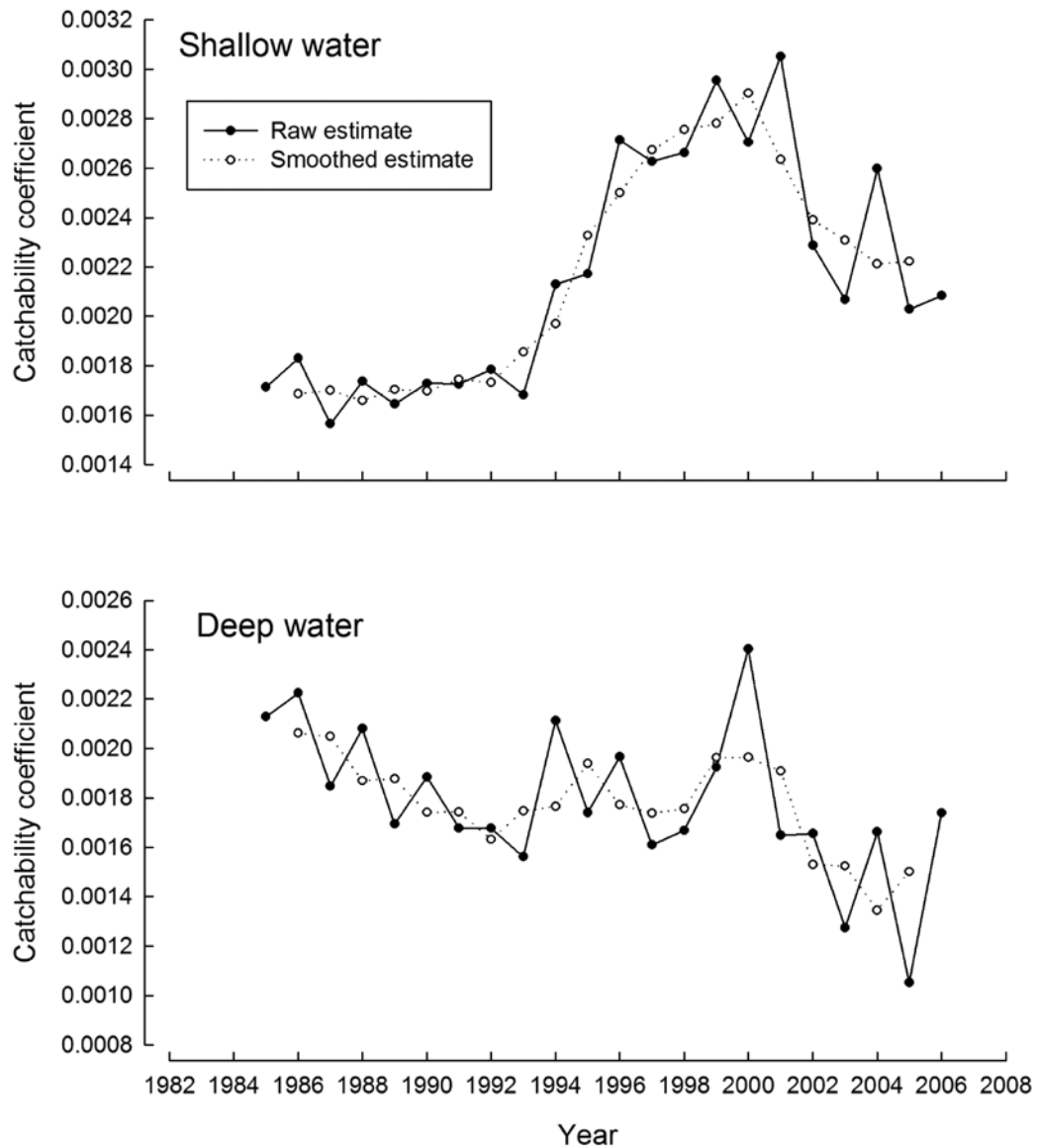
Figure 1.   Raw and weighted-smoothed depletion-based estimates of lobster catchability in shallow (< 20 fm) and deep (>= 20 fm) waters of the Abrolhos Islands fishing zone.

8.0   Benefits

1. Incorporating a knowledge of the distribution and densities of preferred habitats of western rock lobster and fishing effort is essential to ensure the effective implementation of a more holistic ecosystem based fisheries management scheme.

2. The increased knowledge of the relationship between western rock lobster abundance and size structure and preferred habitats will ensure appropriate habitats are included in the design and implementation of long term monitoring programs and marine protected areas.

3. The development of cost effective methods for deepwater benthic habitat assessment such that species habitat relationships can be established is invaluable in being able to effectively monitor ecosystem changes over time.

4. Depletion estimates by depth can be used to improve the stock assessments undertaken using alternative stock assessment methods such as length-based models, which have traditionally used a constant catchability coefficient.

9.0   Further Development

The EcoSRG in their structured plan for research on western rock lobster and their interaction with deepwater ecosystems noted observing ecosystem patterns based on gradients was the important first step for ecosystem research. If this approach proved unviable then the EcoSRG considered that the next step would be to close an area(s) to conduct research on fished versus unfished areas i.e. marine protected areas.

This study has identified a relationship between the abundance and size of western rock lobsters off the west coast of Western Australia and habitat variables. Our research illustrates that incorporating a knowledge of the distribution and densities of preferred habitats is essential to ensure the effective assessment of the effects of biomass removal and to ensure appropriate habitats are included in the design and implementation of long term monitoring programs and marine protected areas. However this study has not provided a definitive answer to the question "what is the effect of lobster biomass removal on the ecosystem?"

Therefore the next logical step in the process, in accordance with the EcoSRG's and MSC recommendations, is the negotiation and establishment of marine protected area(s). With the information from this study on the relationship between lobsters and the ecosystem they inhabit and the assessment of accurate and cost effective techniques for assessing benthic habitats this process has commenced. An industry-working group, comprised of fishers, industry representative bodies (Western Rock Lobster Council and Rock Lobster Industry Advisory Committee) and Department of Fisheries scientists, has been established. The closed areas working group has assessed several areas as potential locations for a marine protected area and negotiation with industry are underway on the exact location and size of the area to be closed to fishing. This is an on-going process and an FRDC funded project will commence in 2009 to collect baseline data in the nominate area.

There is also a new spatial population model under development that takes into account depth the differences in catchability with depth identified in this project will be valuable information for these models.

10.0   Planned Outcomes

There were a number of planned outcomes defined in the original application:

1. For the project to provide ecological information on western rock lobster that will allow management of deepwater breeding stocks in a more sophisticated ecosystem-based manner.

   This study has resulted in the collection of biological, population and habitat data that improves our understanding of the role of western rock lobster in the deepwater ecosystem. The relationship between western rock lobster size and abundance and habitat variables established in this study are important for the effective management of the fishery in an ecosystem based fisheries management (EBFM) framework.

2. The provision of information on the ecosystem impact of removing lobsters from deepwater habitats that can be readily used by managers to meet the requirements of DEWHA to ensure the export of lobster product.

   While this project did not definitively answer the question what is the effect of biomass removal of lobsters in deepwater it has substantially increased our understanding of deepwater lobsters and their use and interaction with the environment they inhabit. This study has identified a relationship between the abundance and size of western rock lobsters off the west coast of Western Australia and habitat variables. Our research illustrates that incorporating a knowledge of the distribution and densities of preferred habitats is essential to ensure the effective assessment of the effects of biomass removal and to ensure appropriate habitats are included in the design and implementation of long term monitoring programs and marine protected areas.

3. The provision of information that will aid maintenance of DEWHA certification.

   This research is the first step in the process of assessing the impact of lobster biomass removal in the deepwater and provides detailed information for the ecological risk assessment (ERA) process required to meet the Department of Environment, Water, Heritage and the Arts (DEWHA) regulations and maintain marine stewardship certification (MSC). A better understanding of the spatial distribution of lobster, habitat and fishing effort is essential for the assessment and design of management and research tools such as spatial closures or protected areas and for the assessment of fishing practices, which are a critical component of continuing to meet DEWHA regulations and MSC certification requirements.

11.0   Conclusions

The following summary highlights the objectives of the project and illustrates that the project has successfully achieved its objectives.

Objective 1: To identify gradients in the density/size distribution of western rock lobster to enable selection of representative areas.

Prior to the commencement of this project The EcoSRG held the view that there was a general lack of knowledge or information on the interaction of the Western Rock Lobster Fishery with the deepwater ecosystem. Therefore it was necessary for initial work to be focused on identifying and observing ecosystem patterns on which to base the design of future studies comparing fished and unfished fished areas. The EcoSRG believed that targeted studies could determine relationships between fishing pressure, lobster population size structure and benthic structure, and provide a quick and cost-effective way of determining some of the impacts of fishing on benthic ecosystems.

The original focus of this objective was to use existing data, from both the Department of Fisheries and other sources, to compile a comprehensive database on the abundance and distribution of rock lobsters and the associated habitat types. This objective was primarily a desktop study to use existing data to identify areas, within the scope of the rock lobster fishery, where gradients in lobster density and/or size structure can be investigated. However at the time of this project the majority of habitat data available for deepwater habitats (>40m) belonged to resource companies and private consultancies who were unable or unwilling to release the data for public use. To combat this issue a survey of deepwater habitats at three coastal locations (Dongara, Jurien and Lancelin) central to the western rock lobster industry was conducted by towed video and deepwater divers. The results of these surveys are discussed in Chapter 2. Similarly, western rock lobster fishers were approached to obtain detailed catch information which would allow the investigation of the density and size of lobsters along the west coast of Western Australia. However, the commercial fishers were unwilling to release detailed catch data so detailed lobster data from the Independent Breeding Stock Survey (IBSS) was used to determine gradients at the three coastal locations listed above. Therefore this project focused on observing ecosystem patterns based on gradients in areas of known fishing effort.

Objective 2: To assess the catchability of western rock lobster and its relationship with population abundance and size structure

An assessment of the catchability of rock lobsters in the Abrolhos Islands (A Zone) showed that catchability in shallow water significantly increased following the 18% pot reduction in 1993/94. This study indicates that this increase is confined to the shallow water component of the fishery. This indicates that there is a significant level of pot saturation occurring in the shallow waters of the Abrolhos. This result suggests that there may be some economic benefit from further pot reductions in shallow waters at the Abrolhos Islands and these are currently being examined for the 2008/09 season. These depletion estimates can be used to improve the stock assessments undertaken using alternative stock assessment methods such as length-based models, which have used traditionally used a constant catchability coefficient.

Objective 3: To identify the relationship between the deep-water habitat and the density/size distribution of western rock lobster to enable a preliminary evaluation of the impact of lobster biomass removal in the deep-water

Two separate studies, have quantified the relationship between the abundance and size of western rock lobster and habitats in which they are found.

The first study used towed video transects for benthic habitat classification and lobster size and abundance data from the annual western rock lobster independent breeding stock survey (IBSS) at three coastal locations, Dongara, Jurien Bay and Lancelin. The data were analysed to examine the relationship between lobster population characteristics and benthic habitat variables. The sites varied both with respect to the composition of habitats and the abundance and size of lobster, with Dongara being significantly different from the other two sites. The largest lobsters were found at Dongara while the highest abundance of lobsters was at Lancelin. In addition, lobsters vary both qualitatively and quantitatively with habitat. Qualitative characteristics of the benthos rather than the density of a particular habitat type determined lobster size, with low numbers of larger lobsters present in areas associated with sponge and high numbers of smaller lobsters present in more structurally complex *Ecklonia* dominated habitats.

The second study compared two different techniques for assessing benthic habitat classification using (1) a towed video survey and (2) a full coverage habitat map derived from a multibeam hydroacoustic survey and towed video at Jurien IBSS. Habitat classification from both techniques indicated that the IBSS subregions varied significantly in habitat composition. There was a strong association between western rock lobster abundance and size and habitat types that mirrored those found in the regional study i.e. low numbers of larger lobsters associated with sponge and high numbers of smaller lobsters present in *Ecklonia* dominated habitats. This study also illustrated that towed video transects are a simple, cost effective and reliable method of assessing benthic habitat. Full coverage maps are expensive and require specialised equipment however, in combination with fine scale fishing effort data (if available) they can be used to provide habitat specific estimates of standing biomass and an estimate of biomass removal.

Both studies illustrated that both the amount of habitat and the configuration of habitat in deepwater influences the size and abundance of western rock lobster off the west coast of Western Australia. Our research also illustrates that incorporating a knowledge of the distribution and densities of preferred habitats and fishing effort is essential to ensure the effective implementation of a more holistic ecosystem based fisheries management scheme including the design, implementation and monitoring of marine protected areas.

12.0   References

Abdo, D. 2007. Endofauna differences between two temperate marine sponges (Demospongiae; Haplosclerida; Chalinidae) from southwest Australia. *Marine Biology*. 152, 845–854.

Acosta, C. 1999. Benthic dispersal of Caribbean spiny lobsters among insular habitats: Implications for the conservation of exploited marine species. *Conservation Biology.* 13, 603–612.

Acosta, C., Butler, M. J. IV. 1997. The role of mangrove habitat as nursery for juvenile spiny lobster, *Panulirus argus,* in Belize. *Marine and Freshwater Research*. 48, 721–728.

Acosta, C. A., Robertson, D. N. 2003. Comparative spatial ecology of fished spiny lobsters *Panulirus argus* and an unfished congener *P. guttatus* in an isolated marine reserve at Glover's Reef atoll, Belize. *Coral Reefs*. 22, 1–9.

Andersen, A. N. 1995. Measuring more of biodiversity: Genus richness as a surrogate for species richness in Australian ant faunas. *Biological Conservation*. 73, 39–43.

Anderson, M. J. 2001. A new method for non–parametric multivariate analysis of variance. *Austral Ecology.* 26, 32–46.

Andrew, N. L., MacDiarmid, A. B. 1991. Interrelations between sea urchins and spiny lobsters in northeastern New Zealand. *Marine Ecology Progress Series*. 70, 211–222.

Babcock, R. C., Kelly, S., Shears, N. T., Walker, J. W., Willis, T. J. 1999. Changes in community structure in temperate marine reserves. *Marine Ecology Progress Series*. 189, 125–134.

Babcock, R. C., Phillips, J. C., Lourey, M., Clapin, G. 2007. Increased density, biomass and egg production in an unfished population of Western Rock Lobster (*Panulirus cygnus*) at Rottnest Island, Western Australia. *Marine and Freshwater Research*. 58, 286–292.

Barkai, A., Branch, G.M. 1988. Energy requirements for a dense population of rock lobsters (*Jasus lalandii)*: novel importance of unorthodox food sources. *Marine Ecology Progress Series*. 50, 83–96.

Bellchambers, L. M., Evans, S. N., Meeuwig, J. J. (In Press). Abundance and size of western rock lobster (*Panulirus cygnus*) as a function of benthic habitat: implications for ecosystem–based fisheries management (EBFM). *Marine and Freshwater Research.*

Bello, P. J., Rios, L. V., Liceaga, C. M. A., Zetina, M. C., Cervera, C. K., Arceo, B. P., Hernandez, N. H. 2005. Incorporating spatial analysis of habitat into spiny lobster (*Panulirus argus*) stock assessment at Alacranes reef, Yucatan, Mexico. *Fisheries Research*. 73, 37–47.

Bergman, M. J. N., Hup M. 1992. Direct effects of beam trawling on macrofauna in a sandy sediment in the southern North Sea. *ICES Journal of Marine Science*. 49, 5–11.

Bologna, P. A. X., Steneck, R. S. 1993. Kelp beds as habitat for American lobster (*Homarus americanus)*. *Marine Ecology Progress Series*. 100, 127–134.

Bowen, B. K., Hancock, D. A. 1989. Effort limitation in the Australian rock lobster fisheries. In: Caddy, J. F (Ed). *Marine invertebrate fisheries: Their assessment and management*. New York, Wiley. pp. 375–393.

Braaten, D. O. 1969. Robustness of DeLury population estimation. *Journal of Fisheries Research, Board of Canada*. 26, 339–355.

Brey, T. 1991. The relative significance of biological and physical disturbance: An example from intertidal and subtidal sandy bottom communities. *Estuarine Coastal and Shelf Science*. 33, 339-360.

Brown, R. S., Caputi, N., Hall, N. G. 1994. Measurement of catch and fishing effort in the western rock lobster fishery. In: Phillips B. F., Cobb J. S., Kitaka J. (Ed). *Spiny lobster management*. London, Fishing News Books. pp. 262–282.

Bunn, S.E., Loneragan, N.R., Kempster, M.A. 1995. Effects of acid washing on stable isotope ratios of C and N in penaeid shrimp and seagrass: Implications for food–web studies using multiple stable isotopes. *Limnology and Oceanography*. 40, 622–625.

Butler, M. J., Steneck, R. S., Herrnkind, W. F. 2006. Juvenile and adult ecology. In: Phillips, B. (Ed) *Lobsters: Biology, Management, Aquaculture and Fisheries*. Blackwell Publishing Ltd., UK. pp. 263–309.

Butler, M. J. IV, Dolan, T., Hunt, J. H., Hernkind, W. F., Rose, K. 2005. Recruitment in degraded marine habitats: a spatially explicit, individual based model for spiny lobster. *Ecological Applications*. 15, 902–918.

Buxton, C., Barrett, N., Haddon, M., Gardner, C., Edgar. G. 2006. Evaluating the effectiveness of marine protected areas as a fisheries management tool. FRDC Project No. 1999/162.

Caputi, N., Brown, R.S., Chubb, C. F. 1995. Regional prediction of the western rock lobster (*Panulirus Cygnus)* catch in western Australia. *Crustaceana*. 68, 245–256.

Cardoso, P., Silva, I., de Oliveira, N.G., Serrano, A. R. M. 2004. Higher taxa surrogates of spider (*Araneae*) diversity and their efficiency in conservation. *Biological Conservation*. 117, 453–459.

Childress, M. J. 1997. Marine reserves and their effects on lobster populations: report from a workshop. *Marine and Freshwater Research*. 48, 1111–1114.

Chittleborough, R. G. 1975. Environmental factors affecting growth and survival of juvenile western rock lobsters *Panulirus longipes* (Milne–Edwards). *Australian Journal of Marine and Freshwater Research*. 26, 177–196.

Chittleborough, R. G. 1976. Growth of juvenile (*Panulirus longipes Cygnus)* George on coastal reefs compared with those reared under optimal environmental conditions. *Australian Journal of Marine and Freshwater Research*. 27, 279–295.

Clarke, K. R., Gorley, R. N. 2006. Primer v6: User Manual / Tutorial. Primer–E Ltd.

Clarke, K. R., Somerfield, P. J., Airoldi, L., Warwick, R. M. 2006. Exploring interactions by second–stage community analyses. *Journal of Experimental Marine Biology and Ecology*. 338, 179–192.

Cox, C., Hunt, J. H. 2005. Change in size and abundance of Caribbean spiny lobster (*Panulirus argus)* in a marine reserve in the Florida Keys National Marine Sanctuary, USA. *Marine Ecology Progress Series*. 294, 227–239.

Cox, D. R., Miller H. D. 1965. *The theory of stochastic processes*. London, Chapman and Hall.

Cresswell, G. R. 1991. The Leeuwin Current – observations and general models. *Journal of the Royal Society of Western Australia*. 74, 1–14.

Davenport, S. D., Bax, N. J. 2002. A trophic study of a marine ecosystem off southeastern Australia using stable isotopes of carbon and nitrogen. *Canadian Journal of Fisheries and Aquatic Sciences*. 59, 514–530.

Davidson, R. J., Villouta, E., Cole, R. G., Barrier, R. G. F. 2002. Effects of marine reserve protection on spiny lobster (*Jasus edwardsii*) abundance and size at Tonga Island Marine Reserve, New Zealand. Aquatic Conservation Marine and Freshwater Ecosystems. 12, 213–227.

Dayton, P. K., Trush, S. F., Argardy, M. T., Hofman, R. J. 1995. Viewpoint. Environmental effects of marine fishing. *Aquatic Conservation: Marine and Freshwater Ecosystems*. 5, 205–232.

de Lestang S., Melville–Smith, R., Thomson, A., Rossbach, M. 2007. West Coast Rock Lobster Fishery Status Report. In: Fletcher, W. J, Santoro, K. (Eds). State of the Fisheries Report 2006/07, Department of Fisheries, Western Australia. pp. 22–25.

Delury, D. B. 1947. On the estimation of biological populations. *Biometrics*. 3, 145–167.

Edgar, G. J. 1990a. Predator–prey interactions in seagrass beds. Distribution and diet of the blue manna crab (*Portunus pelagicus*) Linnaeus at Cliff Head, Western Australia. *Journal of Experimental Marine Biology and Ecology.* 139, 23–32.

Edgar, G. J. 1990b The influence of plant structure on the species richness, biomass and secondary production of macrofaunal assemblages associated with Western Australian seagrass beds. *Journal of Experimental Marine Biology and Ecology*. 137, 215–240.

Edgar, G. J., Shaw, C. 1995. The production and trophic ecology of shallow–water fish assemblages in southern Australia. III. General relationships between sediments, seagrasses, invertebrates and fishes. *Journal of Experimental Marine Biology and Ecology*. 194, 107–131.

Eggleston, D. B., Dahlgren, C. P. 2001. Distribution and abundance of Caribbean spiny lobsters in the Key West National Wildlife Refuge: relationship to habitat features and impact of an intensive recreational fishery. *Marine and Freshwater Research*. 52 ,1567–1576.

Eggleston, D. B., Lipcius, R. 1992. Shelter selection by spiny lobster under variable predation risk, social conditions and shelter size. *Ecology*. 73, 992–1011.

Eggleston, D. B., Lipcius, R., Coba–Centina, L., Miller, D. 1990. Shelter scaling regulates survival of juvenile spiny lobster (*Panulirus argus)*. *Marine Ecology Progress Series*. 62, 79–88.

Fantle, M. S., Dittel, A. I., Schwalm, S. M., Epifanio, C. E., Fogel, M.L. 1999. A food web analysis of the juvenile blue crab (*Callinectes sapidus)* using stable isotopes in whole animals and individual amino acids. *Oecologia*. 120, 416–426.

Fernandez, J. A., Cross, J. M., Caputi, N. 1997. The impact of technology on fishing power in western rock lobster (*Panulirus cygnus*) fishery. *Proceedings of the International Congress on modelling and simulation*. University of Tasmania. pp. 1605–1610.

Field, J. C., Francis, R. C. 2006. Considering ecosystem–based fisheries management in the California Current. *Marine Policy*. 30, 552–269.

Fletcher, W. J. 2007. A guide to implementing an ecosystem approach to fisheries management (EBFM) within the western and central Pacific region. Forum Fisheries Agency, Honiara, Solomon Islands.

Frusher, S. D., Hoeing, J. M. 2001. Impact of size on selectivity of traps for southern rock lobster (*Jasus edwardsii*). *Canadian Journal of Fisheries and Aquatic Sciences*. 58, 2482–2489.

Fry, B. 1988. Food web structure on Georges Bank from stable C, N, and S isotopic compositions. *Limnology and Oceanography*. 33, 1182–1190.

Fugro Survey Pty Ltd. 2006. WA Marine Futures: hydroacoustic survey plan. Technical Report. Fugro Survey Pty.

Gaston, K. J., Williams, P. H. 1993. Mapping the world's species – The higher taxon approach. *Biodiversity Letters*. 1, 2–8.

Goni, R., Quetglas, A., Renones, O. 2001. Diet of the spiny lobster *Panulirus elephas* (Decapoda: Palinuridea) from the Columbretes Islands Marine Reserve (north–western Mediterranean). *Journal of the Marine Biological Association of the United Kingdom*. 81, 347–348.

Graf, G. 1989. Benthic–pelagic coupling in a deep–sea benthic community. *Natur*e. 341, 437–439.

Groenewold, S., Fonds, M. 2000. Effects on benthic scavengers of discards and damaged benthos produced by the beam–trawl fishery in the southern North Sea. *ICES Journal of Marine Science*. 57, 1395–1406.

Hall, S. J. 1999. The effects of fishing on marine ecosytems and communities. Blackwell Science. Oxford.

Hall, N. G., Chubb C. F. 2001. The status of the western rock lobster (*Panulirus Cygnus)* fishery and the effectiveness of management controls in increasing the egg production of the stock. *Marine and Freshwater Research*. 52, 1657–1667.

Hanson, C. E., Pattiaratchi, C. B., Waite, A. M. 2005. Seasonal production regimes off south–western Australia: influence of the Capes and Leeuwin Currents on phytoplankton dynamics. *Marine and Freshwater Research*. 56, 1–16.

Heck, K. L., Orth, R. J. 1980. Seagrass habitats: The roles of habitat complexity, competition and predation in structuring associated fish and motile macroinvertebrate assemblages. In: V.S Kennedy (Ed) *Estuarine Perspectives: International Estuarine Research Conference*. Academic Press, New York. pp. 449–464.

Hernkind, W. F., Butler, M. J., IV, Hunt, J. H., Childress, M. 1997. The role of physical refugia: implications from a mass sponge die–off in a lobster nursery. *Marine and Freshwater Research*. 48, 759–570.

Hilborn, R., Walters C. J. 1992. *Quantitative Fisheries Stock Assessment: Choice Dynamics and Uncertainty*, Chapman and Hall, London.

Hughes, R. N. 1980. Optimal foraging theory in a marine context. *Oceanography and Marine Biology Annual Review*. 18, 423–481.

Ivlev, V. S. 1961. Experimental ecology of the feeding of fishes. Yale University Press, New Haven, Connecticut. pp. 302.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., Warner, R. R. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science*. 293, 629–637.

Jennings, S. 2004. The ecosystem approach to fishery management: a significant step towards sustainable use of the marine environment? *Marine Ecology Progress Series*. 274, 269–303.

Jennings, S., Kaiser, M. J. 1998. The Effects of Fishing on Marine Ecosystems. *Advances in Marine Biology*. 34, 201–352.

Jennings, S., Renones, O., Morales–Nin, B., Polunin, N.V.C., Moranta, J., Coll, J. 1997. Spatial variation in the ^{15}N and ^{13}C stable isotope composition of plants, invertebrates and fishes on Mediterranean reefs: implications for the study of trophic pathways. *Marine Ecology Progress Series*. 146, 109–116.

Jernakoff, P. 1987. Foraging patterns of juvenile western rock lobster (*Panulirus cygnus)* George. *Journal of Experimental Marine Biology and Ecology*. 1131, 25–144.

Jernakoff, P., Nielsen, J. 1998. Plant–animal associations in two species of seagrasses in Western Australia. *Aquatic Botany*. 60, 359–376.

Jernakoff, P., Phillips, B. F. 1988. Effect of baited trap on the foraging movements of juvenile western rock lobster (*Panulirus cygnus)* George. *Australian Journal of Marine and Freshwater Research*. 39, 185–192.

Jernakoff, P., Phillips, B.F., Fitzpatrick, J.J. 1993. The diet of post–puerulus western rock lobster (*Panulirus Cygnus)* George, at Seven Mile Beach, Western Australia. *Australian Journal of Marine and Freshwater Research*. 44, 649–655.

Jernakoff, P., Phillips, B. F., Maller, R. A. 1987. A quantitative study of nocturnal foraging distances of the western rock lobster (*Panulirus cygnus)* George. *Journal of Experimental Marine Biology and Ecology*. 113, 9–21.

Johannes, R.E., Pearce, A.F., Wiebe, W.J., Crossland, C.J., Rimmer, D.W., Smith, D.F., Manning, C. 1994. Nutrient characteristics of well–mixed coastal waters off Perth, Western Australia. *Estuarine, Coastal and Shelf Science*. 39, 273–285.

Johnson, C. R., Valentine, J. P., Pederson, H. G. 2004. A most unusual barrens: Complex interactions between lobsters, sea urchins and algae facilitates spread of an exotic kelp in eastern Tasmania. *Proceedings of 11th International Echinoderm Conference*.

Johnson, R. A., Wichern, D. W. 1982. *Applied multivariate statistical analysis*. Prentice Hall, New York, Chapter 7. pp. 291–357.

Joll, L. M., Phillips, B. F. 1984. Natural diet and growth of juvenile western rock lobster (*Panulirus cygnus)* George. *Journal of Experimental Marine Biology and Ecology*. 75, 145–169.

Jones, C. G., Lawton, J. H., Shachak, M. 1994. Organisms as ecosystem engineers. *Oikos*. 69, 373–386.

Kaiser, M. J., Collie, J. S., Hall, S. J., Jennings, S., Poiner, I. R. 2003. Impacts of fishing gear on marine benthic habitats. In: Sinclair, M. and Valdimarsson, G. (Eds) *Responsible fisheries in the marine ecosystem*. FAO and CABI Publishing, pp. 197–218.

Kaiser, M. J., Ramsay, K., Richardson, C. A., Spence, F. E., Brand, A. R. 2000. Chronic fishing disturbance has changed shelf sea benthic community structure. *Journal of Animal Ecology*. 69, 494–503.

Kelly, S. Scott, D., MacDiarmid, A. B., Babcock, R. C. 2000. Spiny lobster (*Jasus edwardsii)* recovery in New Zealand marine reserves. *Biological Conservation*. 92, 359–369

Kendall, M., Stuart, A. 1973. *The advanced theory of statistics*. Volume 2 (3rd ed). Charles Griffin, London. Chapter 27, pp. 330–360.

Kendrick, G. A., Lavery, P. A., Phillips, J. C. 1999. Influence of *Ecklonia radiata* kelp canopy on structure of macro–algal assemblages in Marmion Lagoon, Western Australia. *Hydrobiologia*. 399, 275–283.

Kendrick, G. A., Harvey, E. S., Wernberg, T., Harman, N., Goldberg, N. 2004. The role of disturbance in maintaining diversity of benthic macroalgal assemblages in southwestern Australia. *The Japanese Journal of Phycology*. 52, 5–9.

Kirkman, H. 1989. Growth, density and biomass of *Ecklonia radiata* at different depths and growth under artificial shading off Perth, Western Australia. *Australian Journal of Marine and Freshwater Research*. 40, 169–197.

Kling, G. W., Fry, B., O'Brien, W. J. 1992. Stable isotopes and planktonic trophic structure in arctic lakes. *Ecology*. 73, 561–566.

Kohler, C. C., Ney, J. J. 1982. A comparison of methods for quantitative analysis of feeding selection of fishes. *Environmental Biology of Fishes*. 7, 363–368.

Langlois, T. J., Anderson, M. J., Babcock, R. C. 2005. Reef–associated predators influence adjacent soft–sediment communities. *Ecology*. 86, 1508–1519.

Langlois, T. J., Anderson, M. J., Babcock, R. C., Kato, S. 2006a. Marine reserves demonstrate trophic interactions across habitats. *Oecologia*. 147, 134–140.

Langlois, T. J., Anderson, M. J., Brock, M., Murman, G. 2006b. Importance of rock lobster size–structure for trophic interactions: choice of soft–sediment bivalve prey. *Marine Biology*. 149, 447–454.

Lawton, J. H., Jones, C. G. 1995. Linking species and ecosystems: Organisms as ecological engineers. In: Jones, C. J., Lawton, J. H. (Eds), *Linking species and ecosystems*. Chapman and Hall, New York. pp. 141–150.

Legendre, P., Anderson, M. J. 1999. Distance–based redundancy analysis: testing multispecies responses in multifactorial ecological experiments. *Ecological Monographs*. 69, 1–24.

Legendre, P., Legendre, L. 1998. Numerical Ecology. Elsevier Science: Amsterdam.

Lenanton, R. C., Joll, L., Penn, J., Jones, K. 1991. The influence of the Leeuwin Current on coastal fisheries of Western Australia. *Journal of the Royal Society of Western Australia*. 74, 101–114.

Leslie, P. H., Davis, D. H. S. 1939. An attempt to estimate the absolute number of rats on a given area. *Journal of Animal Ecology*. 8, 94–113.

Li, Q., James, N. P, Bone, Y., McGowran, B. 1999. Palaeoceanographic significance of recent foraminiferal biofacies on the southern shelf of Western Australia: a preliminary study. *Palaeogeography, Palaeoclimatology, Palaeoecology*. 147, 101–120.

Link , J. S. 2002. What does ecosystem–based fisheries management mean? *Fisheries*. 27, 18–21.

Link, J. S., Brodziak, J. K. T., Edwards, S. F., Overholtz, W. J., Mountain, D., *et al.* 2002. Marine ecosystem assessment in a fisheries management context. *Canadian Journal of Fisheries and Aquatic Science*. 59, 1429–1440.

Lipicus, R. N., Eggleston, D. B., Miller, D. L., Luhrs, T. C. 1998. The habitat–survival function for Caribbean spiny lobster: an inverted size effect and non–linearity in mixed algal and seagrass habitats. *Marine and Freshwater Research*. 49, 807–816.

MacDiarmid, A. B., Hickey, B., Maller, R. A. 1991. Daily movement patterns of the spiny lobster *Jasus edwardsii* (Hutton) on a shallow reef in northern New Zealand. *Journal of Experimental Marine Biology and Ecology*. 147, 185–205.

Mace, P. M. 2004. In defence of fisheries scientists, single–species models and other scapegoats: confronting the real problems. *Marine Ecology Progress Series*. 274, 269–303.

Mai, T. T., Hovel, K. A. 2007. Influence of local–scale and landscape–scale habitat characteristics on California spiny lobster (*Panulirus interruptus*) abundance and survival. *Marine and Freshwater Research*. 58, 419–428.

Mayfield, S., Branch, G. M. 2000. Interrelations among rock lobsters, sea urchins, and juvenile abalone: implications for community management. *Canadian Journal of Fisheries and Aquatic Sciences*. 57, 2175–2185.

Mayfield, S., Branch, G. M., Cockcroft, A. C. 2000. Relationships among diet, growth rate, and food availability for the South African rock lobster, *Jasus Lalandii* (Decapoda, Palinuridae). *Crustaceana*. 73, 815–834.

Mayfield, S., de Beer, E., Branch, G. M. 2001. Prey preference and the consumption of sea urchins and juvenile abalone by captive rock lobsters (*Jasus lalandii*). *Marine and Freshwater Research*. 52, 773–780.

McArdle, B. H., Anderson, M. J. 2001. Fitting multivariate models to community data: A comment on distance–based redundancy analysis. *Ecology*. 82, 290–297.

McQuillan, L. (2006). Species richness, density and cover of sponge assemblages on temperate reefs off Perth, Western Australia. MSc. Thesis, School of Natural Sciences. Edith Cowan University, Perth, Australia.

Melville–Smith, R., Chubb, C. F., Caputi, N. Cheng, Y. W., Christianopoulos, D., Rossbach M. 1998. Fishery independent survey of the breeding stock and migration of the western rock lobster (*Panulirus cygnus*). Fisheries Research and Development Corporation. 96/108 Final Report. Canberra. pp. 47.

Melville–Smith, R., Thomson, A. W., Caputi, N. 2004. Improved forecasts of recreational western rock lobster (*Panulirus cygnus*) catches in Western Australia, by predicting licence usage. *Fisheries Research*. 68, 203–208.

Miller, R. J. 1990. Effectiveness of crab and lobster traps. *Canadian Journal of Fisheries and Aquatic Sciences*. 47, 1228–1251.

Molloy, F.J., Bolton, J. J. 1996. The effects of wave exposure and depth on the morphology of inshore populations of the Namibian kelp, (*Laminaria schinzii*) Foslie. *Botanica Marina*. 39, 525–531.

Morgan, G. R. 1974. Aspects of the population dynamics of the western rock lobster (*Panulirus cygnus*) George II, seasonal changes in the catchability coefficient. *Australian Journal of Marine and Freshwater Research*. 25, 249–259.

Morgan, G. R. 1977. Aspects of the population dynamics of the western rock lobster and their role in management. PhD thesis, University of Western Australia, Australia.

National Research Council. 1994. Priorities for Coastal Ecosystem Science. National Academy Press, Washington, DC.

Netto, S.A., Warwick, R.M., Attrill, M.J. 1999. Meiobenthic and macrobenthic community structure in carbonate sediments of Rocas Atoll (North–east, Brazil). *Estuarine Coastal and Shelf Science*. 48, 39–50.

Olsgard, F., Somerfield, P.J. 2000. Surrogates in marine benthic investigations, which taxonomic unit to target? *Journal of Aquatic Ecosystem Stress and Recovery*. 7, 25–42.

Overman, N. C., Parrish, D. L. 2001. Stable isotope composition of walleye: ^{15}N accumulation with age and area–specific differences in ^{13}C. *Canadian Journal of Fisheries and Aquatic Sciences*. 58, 1253–1260.

Pace, M. L., Cole, J. J., Carpenter, S. R., Kitchell, J. F. 1999. Trophic cascades revealed in diverse ecosystems. *Trends in Ecology and Evolution*. 14, 483–488.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., Torres, J. 1998. Fishing down marine food webs. *Science*. 279, 860–863.

Pederson, H. G., Johnson, C. R. 2006. Predation of the sea urchin (*Heliocidaris erythrogramma)* by rock lobsters (*Jasus edwardsii*) in no–take marine reserves. *Journal of Experimental Marine Biology and Ecology*. 336, 120–134.

Penn, J. W., Joll, L. M. 1990. The application of high resolution navigation systems to Leslie–DeLury depletion experiments for measurement of trawl efficiency under open sea conditions. *Fisheries Research*. 9, 41–55.

Penn, J. W., Fletcher, W. J., Head F. 2004. State of the Fisheries Report 2002/2003, Department of Fisheries, Government of Western Australia.

Phillips, B. F. 1990. Estimating the density and mortality of juvenile western rock lobsters (*Panulirus cygnus*) in nursery reefs. *Canadian Journal of Fisheries and Aquatic Science*. 47, 1330–1338.

Phillips, B. F., Brown, R. S. 1989. The West Australian Rock Lobster Fishery: Research for Management, In: Caddy, J. F. (Ed). *Marine Invertebrate Fisheries: Their Assessment and Management*. Wiley Interscience, New York. pp. 159–81.

Phillips, D. L., Gregg, J. W. 2003. Source partitioning using stable isotopes: coping with too many sources. *Oecologia*. 136, 261–269.

Phillips, B. F., Melville–Smith, R. 2005. Sustainability of the western rock lobster fishery: a review of past progress and future challenges. *Bulletin of Marine Science.* 76, 485–500.

Phillips, B. F., Melville–Smith, R. 2006. Panulirus species. In: Phillips, B. (Ed.), Lobsters: Biology, Management, Aquaculture and Fisheries. Blackwell. pp. 359–384.

Phillips, J. C., Kendrick, G. A., Lavery, P. A. 1997. A test of a functional group approach to detecting shifts in macroalgal communities along a disturbance gradient. *Marine Ecology Progress Series*. 153, 125–138.

Phillips, D. L., Newsome, S. D., Gregg, J. W. 2005. Combining sources in stable isotope mixing models: alternative methods. *Oecologia*. 144, 520–527.

Polis, G. A., Strong, D. R. 1996. Food web complexity and community dynamics. *The American Naturalist*. 147, 813–846.

Polis, G. A., Sears, A. L. W., Huxel, G. R., Strong, D. R., Maron, J. 2000. When is a trophic cascade a trophic cascade? *Trends in Ecology and Evolution*. 15, 473–475.

Polovina, J. J., Haight, W. R., Moffitt, R. B., Parish, F. A. 1995. The role of benthic habitat, oceanography, and fishing on the population dynamics of the spiny lobster, *Panulirus marginatus* (Decapoda, Palinuridae) in the Hawaiian archipelago. *Crustaceana*. 68**,** 203–212.

Post, D. M. 2002. Using stable isotopes to estimate trophic position: models, methods and assumptions. *Ecology*. 83, 703–718.

Ramsay, K., Kaiser, M. J., Moore, P. G., Hughes, R. N. 1997. Consumption of fisheries discards by benthic scavengers: utilization of energy subsidies in different marine habitats. *Journal of Animal Ecology*. 66, 884–896.

Radford, B., Van Niel, K. 2008. WA Marine Futures: Benthic modelling and mapping final report. University of Western Australia.

Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations. *Bulletin of Fisheries Research, Board of Canada*. 191, 149–161.

Rios–Lara, V., Salas, S., Bello–Pineda, J., Irene–Ayora, P. 2007. Distribution patterns of spiny lobster (*Panulirus argus*) at Alacranes reef, Yucatan: Spatial analysis and inference of preferential habitat. *Fisheries Research*. 87, 35–45.

Robertson, A. I., Lenanton, R. C. J. 1984. Fish community structure and food chain dynamics in the surf–zone of sandy beaches: the role of detached macrophyte detritus. *Journal of Experimental Marine Biology and Ecology*. 84, 265–283.

Robles, C., Sweetnam, D., Eminike, J. 1990. Lobster predation on mussels: shore–level differences in prey vulnerability and predator preference. *Ecology*. 71, 1564–1577.

Rosing, M. N., Ben–David, M., Barry, R. P. 1998. Analysis of stable isotope data: A K nearest–neighbours randomization test. *Journal of Wildlife Management*. 62, 380–388.

Ross, S. M. 1985. Introduction to probability models. 3rd ed. Orlando Florida, Academic Press.

Saila, S.B., Nixon, S.W., Oviatt, C.A. 2002. Does lobster trap bait influence the Maine inshore trap fishery? *North American Journal of Fisheries Management*. 22, 602–605.

Schmidt, R. S. 1982. Consequences of dissimilar defenses against predation in a subtidal marine community. *Ecology*. 63, 1588–1601.

Schmidt, R. S. 1987. Indirect interactions between prey: apparent competition, predator aggregation and habitat segregation. *Ecology*. 68, 1887–1897.

Searle, D. J., Semeniuk, V. 1985. The natural sectors of the inner Rottnest Shelf coast adjoining the Swan Coastal Plain. *Journal of the Royal Society of Western Australia*. 67, 116–136.

Seddon, G. 1972. A Sense of Place. University of Western Australia Press, Perth, Western Australia. pp. 274.

Shears, N. T., Babcock, R. C. 2002. Marine reserves demonstrate top–down control of community structure on temperate reefs. *Oecologia*. 132, 131–142.

Shears, N. T., Babcock, R. C. 2003. Continuing trophic cascade effects after 25 years of no–take marine reserve protection. *Marine Ecology Progress Series*. 246, 1–16.

Smith, A. D. M., Fulton, E. J., Hobday, A. J., Smith, C., Shoulder, P. 2007. Scientific tools to support the practical implementation of ecosystem–based fisheries management. *ICES Journal of Marine Science*. 64, 633–639.

Spiegelhalter, D. J., Thomas, A., Best, N., Gilks, W. R. 1996. BUGS 0.5, Bayesian inference using Gibbs sampling Manual. Medical Research Council Biostatistics Unit, Institute of Public Health. Cambridge, England.

Srisurichan, S. 2001. Time series modelling of the environmental factors affecting the daily catch rate of western rock lobster. Masters thesis, Edith Cowan University, Perth, Western Australia.

Srisurichan, S. Caputi, N., Cross, J. 2005. Impact of lunar cycle and swell on the daily catch rate of western rock lobster using time series modelling. *New Zealand Journal of Marine and Freshwater Research*. 39(3), 749–764.

Steinberg, P., Kendrick, G. A. 1999. Kelp forests. In: Under Southern Seas – the Ecology of Australia's Rocky Reefs (ed. N. Andrew), pp. 60–71. University of New South Wales Press Ltd., Sydney.

Tarr, R. J. Q., Williams, P. V. G., MacKenzie, A. J. 1996. Abalone, sea urchins and rock lobster: A possible ecological shift may affect traditional fisheries. *South African Journal of Marine Science*. 17, 319–323.

Tegner, M. J., Levin, L. A. 1983. Spiny lobsters and sea urchins: analysis of a predator–prey interaction. *Journal of Experimental Marine Biology and Ecology*. 73, 125–150.

Tegner, M. J., Dayton, P. K. 1981. Population structure, recruitment and mortality of two sea urchins (*Strongylocentrotus franciscannus* and *S. purpuratus*) in a kelp forest. *Marine Ecology Progress Series*. 5, 255–268.

Tegner, M. J., Dayton, P. K. 2000. Ecosystem effects of fishing in kelp forest communities. *ICES Journal of Marine Science*. 57, 579–589.

Turner, S. J., Thrush, S. F., Hewitt, J. E., Cummings, V. J., Funnell, G. 1999. Fishing impacts and the degradation or loss of habitat structure. *Fisheries Management and Ecology*. 6, 401–420.

Vander Zanden, M. J., Cabana, G., Rasmussen, J. B. 1997. Comparing trophic position of freshwater fish calculated using stable isotope ratios (δ^{15}N) and literature dietary data. *Canadian Journal of Fisheries and Aquatic Science*. 54, 1142–1158.

Vanderklift, M. A., Kendrick, G. A. 2004. Variation in abundances of herbivorous invertebrates in temperate subtidal rocky reef habitats. *Marine and Freshwater Research*. 55, 93–103.

Vanderklift, M. A., Kendrick, G. A., Smit, A. J. 2006. Differences in trophic position among sympatric sea urchin species. *Estuarine, Coastal and Shelf Science*. 66, 291–297.

Waddington, K. I. In Press. Variation in evacuation rates of different foods skew estimates of diet in the western rock lobster (*Panulirus cygnus*). *Marine and Freshwater Research*.

Waddington, K., MacArthur, L. In Press. Diet quality and muscle tissue location influence diet–consumer discrimination in captive reared rock lobsters (*Panulirus cygnus*). *Marine Biology*.

Waddington, K. I., Meeuwig, J. J. 2009. Contribution of bait to lobster production in an oligotrophic marine ecosystem. *Fisheries Research*. 99, 1–6.

Waddington, K. I., Bellchambers, L. M., Vanderklift, M. A., Walker, D. I. 2008. Western rock lobsters (*Panulirus cygnus*) in Western Australian deep coastal ecosystems (35–60 m) are more carnivorous than those in shallow water ecosystems. *Estuarine, Coastal and Shelf Science*. 79, 114–120.

Walters, C. J., Hall, N., Brown, R., Chubb, C. 1993. Spatial model for the population dynamics and exploitation of the western Australian rock lobster (*Panulirus cygnus*). *Canadian Journal of Fisheries and Aquatic Sciences*. 50, 165–062.

Whitledge, G.W., Rabeni, C.F. 1997. Energy sources and ecological role of crayfishes in an Ozark stream; Insights from stable isotopes and gut analysis. *Canadian Journal of Fisheries and Aquatic Science*. 54, 2555–2562.

Waite, A. M., Safi, K. A., Hall, J. A., Nodder, S. D. 2000. Mass Sedimentation of Picoplankton Embedded in Organic Aggregates. *Limnology and Oceanography*. 45, 87–97.

Wernberg, T., Coleman, M., Fairhead, A., Miller, S., Thomsen, M. 2003a. Morphology of *Ecklonia radiata* (Phaeophyta, Laminarales) along its geographic distribution in south–western Australia and Australasia. *Marine Biology*. 143, 47–55.

Wernberg, T., Kendrick, G. A., Phillips, J. C. 2003b. Regional differences in kelp–associated algal assemblages on temperate limestone reefs in south–western Australia. *Diversity and Distributions*. 9, 427–441.

Wernberg, T., Vanderklift, M. A., How, J., Lavery, P. S. 2006. Export of detached macroalgae from reefs to adjacent seagrass beds. *Oecologia*. 147, 692–701.

Whitman, R. L., Nevers, M. B., Goodrich, M. L., Murphy, P. C., Davis, B. M. 2004. Characterization of Lake Michigan coastal lakes using zooplankton assemblages. *Ecological Indicators*. 4, 277–286.

Wright, I. W., Caputi, N., Penn, J. W. 2006. Depletion–based population estimates for western rock lobster (*Paniluris cygnus*) fishery in Western Australia. *New Zealand Journal of Marine and Freshwater Research*. 40(1), 107–122.

Wynne, S. P., and Côté, I. M. 2007. Effects of habitat quality and fishing on Caribbean spotted spiny lobster populations. *Journal of Applied Ecology*. 44, 488–494.

Zeller, D., Pauly, D. 2004. The future of fisheries: from 'exclusive' resource policy to 'inclusive' public policy. *Marine Ecology Progress Series*. 274, 295–303.

13.0   Appendices

Appendix 1: Intellectual Property

There are no identifiable intellectual property arising from the project

Appendix 2: Project Staff

Dr Lynda Bellchambers	Principal Investigator	(DoF)[*]
Mr Scott Evans	Senior Technical Officer	(DoF)[*]
Mr Kris Waddington	PhD Student	(UWA)[*]
Dr Nick Caputi	Supervising Scientist	(DoF)[#]
Dr Simon de Lestang	Research Scientist	(DoF)[#]
Dr Ian Wright	Senior Research Scientist	(DoF)[#]
Dr Jessica Meeuwig	Senior Lecturer	(UWA)[#]
Dr Ben Radford	Postdoctoral Student	(UWA)[#]
Casual Field Assistants	Assorted Technical Staff	(DoF)[#]
Deepwater Divers	Contract divers*	

*Staff employed for parts of the project under FRDC funding

In kind staff contributions by DoF and other institutes