Assessing the ecological impact of the western rock lobster fishery in fished and unfished areas
Final FRDC Report – Project 2008/013

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227/14

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Contents

1.0   Non-technical summary

> **2008/013 Assessing the ecological impact of the Western Rock Lobster Fishery in fished and unfished areas**

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Objectives

1. Identification and assessment of suitable unfished reference areas to exclude rock lobster fishing in deep water.
2. Development of a qualitative trophodynamic model that will provide a conceptual framework for determining sampling protocols, indictors and targets.
3. To provide cost effective methods to measure deep water ecosystems in both fished and unfished reference areas.

Outcomes achieved to date

This project, in consultation with an industry working group, has been instrumental in establishing a closed area in the western rock lobster fishery to assess the potential ecosystem effects of removing lobster biomass from deep water (40-100 m). The major outcome of this research is an improved understanding of the ecosystem that supports the western rock lobster. These types of information for improve the Department of Fisheries' ability to manage the fishery in an ecosystem based manner and provide scientifically defensible data to more precisely assess the level of risk of the fishery to deep water communities. In early 2013, data from this project was used to reassess the risk of potential ecosystem effects of fishing on deep water communities at the Ecological Risk Assessment (ERA) for the fishery. The increased knowledge gained though this project reduced the risk level from moderate to low for deep water communities and the continued accreditation of the Western Rock Lobster Managed Fishery (WRLMF) by the Marine Stewardship Council (MSC).

This project has also provided a unique opportunity to study aspects of lobster population dynamics such as growth, natural mortality and carrying capacity under different levels of exploitation. This has led to an increased understanding of lobster population dynamics and their interactions with the ecosystem which will ultimately result in an increased capacity to ensure the long-term sustainability of the fishery, particularly in the face of increased environmental variability and climate change.

An additional outcome of this project is the development of appropriate techniques to monitor the structure and function of deep water ecosystems. Through the refinement of these sampling techniques this project has developed a cost effective, long-term monitoring program to identify changes in the risk to the ecosystem due to the on-going removal of lobster biomass, thereby supporting on-going third party certification and the maintenance of Department of Environment certification for the export of lobster product.

The need for additional ecological data to assist with the assessment of the WRLMF was highlighted by the ERA in 2007, where the effect of removing western rock lobster (here after referred to as lobster) biomass in deep water (> 40m) was given a "moderate" risk. Shallow water ecosystems where already considered low risk due to numerous studies. It was also noted that re-assessment of the level of risk required the gaps in the knowledge of deep water ecosystems to be addressed. Research undertaken during FRDC 2004/049 increased the understanding of the relationship between lobster and their deep water habitats; however the approach was ineffective in addressing MSC requirements. Therefore, the Ecosystem Scientific Reference Group (EcoSRG) concluded that there was a need for future research in deep water to use research closures.

The project first had to identify an appropriate area to close to lobster fishing so the potential impacts of lobster fishing on deep water (>40m) ecosystems could be compared with nearby fished areas. Potential areas for closure were identified using an industry-based closed area working group. Initially, the working group nominated and assessed a number of potential areas using selection criteria established by the EcoSRG. Two areas were short-listed as potential closed areas i.e. the south eastern corner of the Abrolhos zone and the 30°S latitude line offshore from Leeman.

On the basis of the habitat information, collected by towed video surveys, the Leeman site was found to be the most representative of nearby fished areas. Surveys also found that the abundance and size structure of lobster populations in this area were representative of the fishery in general. Importantly they were directly comparable to those present at a nearby fished research area at Jurien for which detailed habitat information and long-term lobster data are available. Following negotiations with the Rock Lobster Industry Advisory Council (RLIAC) and EcoSRG an area *ca.* 3900 ha (6 nm x 2 nm) that straddles the 30°S latitude line offshore from Leeman was officially closed to lobster fishing on the 15 March 2011.

While the direct impacts of fishing on target stocks are usually easy to monitor and effects may be observed after short periods of protection, indirect ecosystem effects may be diffuse, difficult to detect and take a long time to manifest. Ecosystems are complex and the ability to measure and monitor all components in the ecosystem is frequently restricted by both time and available funds. Therefore, monitoring programs must be focussed to maximise the likelihood that potential changes to the ecosystem will be detected. To assist with the process of determining what aspects to monitor qualitative models were used. Qualitative modelling can be used to increase the understanding of ecosystem dynamics by simplifying trophic systems. The models predicted that fishing for rock lobster would most likely positively impact small fish, such as old wife (*Enoplosus armatus*), footballer sweep (*Neatypus obliquus*), and king wrasse (*Coris auricularis*) as they compete for the same food source. Therefore, these small fish were identified as potential indicators of the effects of rock lobster fishing. Small crustaceans (amphipods and isopods) were identified as potential indicators of bait effects.

The ecosystem components to be monitored were divided into three categories; target species (lobsters), benthic habitats and indirect ecosystem indicators (small fish). This project trialled different sampling methods to assess their ability to detect change due to fishing and to developed a cost effective long term monitoring program and establish ecosystem baselines.

There was an increase in the abundance of lobsters in both fished and closed areas in deep water areas over the five years of this study. The sampling period (2008-2012) corresponds to a period of below average recruitment and substantial management changes in the fishery. Therefore, the increase in lobster abundance in deep water illustrates the effectiveness of management

measures implemented during this period. While there was an increase in both fished and unfished areas the increase in abundance was larger in the closed area with a significant increase evident after only 18 months of closure. The increase in abundance in the closed area was most pronounced in legal sized lobsters, particularly males.

An autonomous underwater vehicle (AUV) was used to sample benthic assemblages in fished (Jurien) and closed (Leeman) areas. The use of the AUV enabled the collection of benthic information at different scales and the ability to precisely re-sample geo-referenced areas so that each site can be followed through time to determine the impacts of fishing on the benthic assemblages. The first year of data (collected in April 2011, immediately post closure) has been collected providing a baseline for each of the sites against which any future changes can be quantified.

The small fish, suggested by qualitative modelling as potential indicators of ecosystem effects of fishing for lobster, were sampled along with the rest of the fish assemblage using baited remote underwater videos (stereo BRUVs) in fished (Jurien) and closed (Leeman) areas. The fish assemblages of the fished and closed areas were comparable and a number of small fish were observed at abundances high enough to use as possible indicators. In all cases, sampling of fish in macroalgal habitats provided the most statistical power to detect change. Limiting fish sampling to macroalgal habitats would therefore provide the most cost effective monitoring scenario.

The results of this project indicate that all three components (target species, benthic habitats and indirect ecosystem indicators i.e. fish) and the sampling methods used are sensitive enough to detect changes. It is important to note that the full impact of fishing on the ecosystem may take an extended period to manifest (i.e. >10 years). It remains to be seen if any changes detected in the scope of this study are simply natural perturbations or can be attributed to the impact of fishing e.g. the changes in lobster abundance detected 18 months after the implementation of the closed area. However, establishing a baseline, as done by this study, and implementing a long term monitoring program for a range of different ecosystem components increases the likelihood that any potential changes will be detected.

The real value of this research is the increased understanding of the deep water ecosystem which will allow future assessments of the risk to the ecosystem of the removal of lobster biomass by fishing to be based on scientific research. In addition, the capacity to detect change in the ecosystem that may be driven or influenced by fishing in a timely manner allows for adaptive management and if required allows mitigation measures to be implemented to minimise the potential impacts. An area closed to lobster fishing also allows the research of lobster populations in an unexploited state to investigate factors such as carrying capacity, density dependent growth and natural mortality. The refinement of these aspects of lobster biology and ecology can improve existing stock assessments and modelling.

KEYWORDS: Western rock lobster, benthic habitats, effects of fishing, qualitative modelling, ecosystem monitoring.

2.0   Acknowledgments

3.0   Background

The initial certification process for the Western Rock Lobster Fishery (WRLF) to obtain accreditation by the Marine Stewardship Council (MSC) in 2000 required an ecological risk assessment (ERA) to be undertaken. Although that process, which was completed in 2001, rated the effects of lobster fishing on the overall ecosystem as a low risk, the lack of research data about the ecological impacts of removing lobster biomass from the environment, particularly from deep water remained a concern. An Ecosystem Scientific Reference Group (EcoSRG) was formed in 2003 to provide advice on research directions on determining the effects of western rock lobster fishing on the ecosystem. Among a range of priority information gaps, they identified the need to collect basic ecological information to determine if changes in lobster density and size structure, due to fishing, had caused significant changes in habitat structure and benthic community composition in deep water. A second risk assessment completed at this time identified that the potential ecological impacts of lobster fishing whilst remaining a low risk within shallow waters, was a moderate risk within deep water regions and therefore additional research was required to address these knowledge gaps.

The EcoSRG recognised that any new research in deep water needed to occur in a structured manner and devised a strategic framework which recommended that the initial work should focus on identifying and observing any ecosystem patterns associated with levels of fishing pressure, lobster population size structure and benthic structure. The patterns observed across these gradients were expected to provide information on these relationships and assist in determining if research using fished versus unfished areas was necessary.

An FRDC (2004/049) project provided the critical baseline data on the relationships between the abundance and size distributions of lobster and the different benthic habitats located in deeper waters (Bellchambers 2010). This project also provided preliminary information on the trophic role of lobster within these depths. However, despite the identification of gradients in the abundance of lobsters within similar habitats, this technique ultimately proved ineffective in providing sufficient information to reduce the risk level. Thus, a risk assessment of the WRLF completed in 2007 still determined that there was a moderate risk that the removal of lobster biomass may be altering the relative abundance of species within deep water communities.

To meet the 2006 action plan for MSC recertification, which identified the need to have an adequate understanding of the impacts of the fishery on trophic linkages between lobsters and their predators and prey at the main stages of lobster life history, it was recognized that research in deep water would now have to be based on comparing fished and unfished areas using research closures. Consequently, this would require the establishment of suitable fished and unfished areas plus the collection of baseline information to enable on-going comparisons to occur. Based upon the observations in other temperate reef communities (e.g. Tasmania) it was recognized that the maintenance and monitoring of closed areas may take a decade or more before any definitive conclusions about the impacts of lobster fishing on community structure will be possible.

4.0   Need

Continuation of the Marine Stewardship Council's (MSC) accreditation of the Western Rock Lobster Fishery (WRLF) depends on the fishery addressing issues outlined within the 2006 re-certification action plan. This plan states "no substantive evidence has been presented for assessment about the impacts of the fishery on ecosystem structure, function, diversity, productivity or habitats caused by the removal of target stocks." The re-certification plan also identified the need to understand the impacts of the fishery on trophic linkages between lobsters and their predators and prey, at each of the main stages of lobster life history.

The need for these types of ecological data was highlighted by the most recent risk assessment (2007), where the effects of removing lobster biomass in deep water was given a "moderate" risk. Re-assessment of the level of risk will require the gaps in the knowledge of deep water ecosystems to be addressed.

Research undertaken during FRDC 2004/049 increased the understanding of the relationship between lobster and their deep water habitats. However, the approach was ineffective in addressing MSC requirements. Therefore the Ecosystem Scientific Reference Group (EcoSRG) concluded that there was a need for future research in deep water to use research closures.

Based on the outcomes of FRDC 2004/049, the risk levels and the EcoSRG workshop, this project used closed areas to examine the potential impacts of the WRLF in deep water, providing baseline information essential for the assessment of the impacts of the fishery on the ecosystem.

The project was conducted in two phases. Phase one established fished and unfished areas within deep water in collaboration with industry. Phase two established baseline information on lobster stocks, habitat and community structure.

5.0   Objectives

5.1   Identification and assessment of suitable unfished reference areas to exclude rock lobster fishing in deep water.

Spatial management in the form of closed or unfished areas is often used to meet both fisheries management and biodiversity or conservation objectives (FAO 2011, Lozano-Montes *et al.* 2012). In Australia, closed areas are one of a suite of fisheries management tools that are used for a range of purposes such as habitat protection (by excluding destructive gear types such as trawling), protecting targeted individuals or important nursery and/or spawning grounds. The establishment of closed areas are not a replacement for traditional fisheries management as they do not address key issues beyond their boundaries nor do they address lack of management or unsuccessful management that may have led to over fishing (FAO 2011). However, the establishment of closed areas does provide the opportunity to study marine ecosystems and target stocks in reference areas that are free from the effects of fishing (Tegner and Dayton 2000, Jackson and Sala 2001). Closed areas may also provide an increased understanding of target stocks. For example, the study of populations protected from fishing may provide valuable information on population parameters such as natural mortality and carrying capacity (Buxton *et al.* 2004, Willas and Millar 2005, Babcock *et al.* 2007, Barrett *et al.* 2009a, b) which informs population models and allows for more adaptive management of the fishery.

The majority of previous studies focused on target species have illustrated that closed areas may result in an increase in the abundance, length and biomass of target species (Babcock *et al.* 1999, Watson *et al.* 2007, Barrett *et al.* 2009a, b). For example, Watson *et al.* (2007) found that the abundance of four targeted fish species at the Abrolhos Islands, Western Australia, were between 1.13 to 8 times higher in closed areas than in fished areas. Similar results have been shown for invertebrates, with abundance of western rock lobster inside closed areas at Rottnest Island, Western Australia, 30 times higher than at that fished locations (Babcock *et al.* 2007), and the abundance and biomass of southern rock lobster (*Jasus edwardsii*) in closed areas around Tasmania significantly increased compared to pre-closure assessments (Barrett *et al.* 2009a, b).

With the implementation of Ecosystem Based Fisheries Management (EBFM), closed areas are also useful research tools to improve our understanding of the structure and function of marine ecosystems and determine the ecosystem effects of fishing (Babcock *et al.* 1999, Buxton et al .2004, Barrett *et al.* 2009a, FAO 2011, Lozano-Montes *et al.* 2012). Previous studies have focused on closed areas to examine the indirect or ecosystem effects of fishing, i.e. trophic interactions or cascades (for reviews see Tegner and Dayton 2000, Sumaila *et al.* 2000). Lobsters are ecologically and economically important in temperate and tropical marine ecosystems worldwide. Previous authors have illustrated that several species play an important role in structuring benthic communities (Robles *et al.* 1990, Babcock *et al.* 1999, Mayfield and Branch 2000, Mayfield *et al.* 2001, Shears and Babcock 2001) and that they may induce cascading ecosystem effects (Barrett *et al.* 2009a, Ling *et al.* 2009, Blamey and Branch 2012). In South Africa, predation by *Jasus lalandii* and *Panulirus homarus* altered the abundance and size structure of mussels, urchins and gastropods (Mayfield *et al.* 2001). While, *Panulirus interruptus* and sheepshead (a predatory fish) control the abundance of two species of sea urchins that alter the abundance of giant kelp (*Macrocystis pyrifera*) in Southern Californian kelp forests (Tegner and Dayton 1981). Similarly, *Jasus edwardsii* in Tasmania control populations of *Centrostephanus rodgersii* (long-spined sea urchin). In areas with low abundances of *J. edwardsii*, overgrazing by urchins has caused a phase shift from kelp forests to

urchin barrens (Ling *et al.* 2009). These types of information are required to address ecological concerns surrounding capture fisheries to satisfy the requirements of government and non-government regulatory bodies (i.e. Environmental Protection and Biodiversity Act (EPBC Act) and third party certification such as MSC) (Department of Fisheries 2012, Lozano-Montes *et al.* 2012). Considerations of cumulative impacts on the environment are also an important part of the implementation of Ecosystem Based Fisheries Management (EBFM) (Fletcher *et al.* 2010).

Previous studies have provided critical baseline data on the relationship between the abundance and size of western rock lobster and benthic habitats in deep water (> 40 m) (Bellchambers 2010, Bellchambers *et al.* 2010, Bellchambers *et al.* 2013). However, there is currently insufficient information to clarify the ecosystem impacts of lobster fishing (Bellchambers 2010, Department of Fisheries 2012). Therefore, in accordance with the Ecosystem Scientific Reference Group's (EcoSRG) strategic framework, it was recognized that the next step in the process was that research in deep waters would now be based on comparing fished and unfished (closed) areas using research closures. Consequently, this would require the establishment of suitable fished and unfished (i.e. closed) areas. This objective outlines the process taken to identify and validate an appropriate closed area.

5.2 Development of a qualitative trophodynamic model that will provide a conceptual framework for determining sampling protocols, indictors and targets.

The value of 'no take' closed areas in determining the ecological effects of fishing derives from the fact that they provide contrast and act as controls against which to assess human impacts (Jackson and Sala 2001, Ballantine and Langlois 2008, Pelletier et al. 2008). However, the ability of researchers to demonstrate ecosystem change depends on the appropriateness of the sampling methods employed, i.e. the temporal and spatial scale of monitoring and the taxa sampled (Halpern 2003, Barrett *et al.* 2007, Ojeda-Martinez *et al.* 2007). The capacity to measure and monitor all aspects of ecosystems is unrealistic due to their complexity (Kremen 1992). Time and funding constraints dictate that monitoring programs must be focus on maximising the likelihood that potential changes to ecosystem structure and function may be detected.

The direct impacts of fishing on target stocks are typically easy to monitor and may be observed after relatively short periods of protection. In contrast, indirect effects of fishing, i.e. impacts on the wider ecosystem, may be more complex and take longer to perpetuate. Techniques that conceptualise our understanding of the relationships in these complex systems, such as ecosystem modelling, can indicate how systems may react to perturbations. One such modelling technique that can be useful in providing an understanding of ecosystem structure and dynamics by simplifying trophic systems is qualitative modelling (Puccia and Levins 1985).

Qualitative modelling (also known as loop analysis) uses feedbacks to investigate the impacts of perturbation on system stability and produce predictions of change in ecological, social and economic aspects of systems. This technique may be useful in data-limited situations as it requires only the signs of interactions (positive, negative or no effect) between species or variables in an ecosystem and not quantitative data on the strength of the individual interactions (Levins 1974, 1975).

These models allow the prediction of ecosystem responses to perturbations such as fishing or climate change and provide an understanding of the ecological relationships that drive those responses (Hosack et al. 2009). As such, qualitative methods can be used to identify indicators of ecosystem change and may be useful in informing monitoring programs (Dambacher *et al.* 2009).

5.3 To provide cost effective methods to measure deep water ecosystems in both fished and unfished reference areas.

The effectiveness of no take marine reserves in providing contrast between exploited and non-exploited ecosystems depends on reserve age (McClanahan 2000, Guidetti and Sala 2007, Kramer and Heck 2007), size (Eggleston and Dahlgren 2001, Guidetti and Dulcic 2007, Barrett *et al.* 2007) and the appropriateness of the habitats protected (Goni *et al.* 2001, Mayfield et al. 2005, Prado *et al.*2008). However, the ability of researchers to demonstrate ecosystem change will depend on the appropriateness of the sampling methods employed, i.e. the taxa targeted, and the temporal and spatial scale of monitoring (Halpern 2003, Barrett *et al.* 2007, Ojeda-Martinez *et al.* 2007).

While greater sampling effort may increase the probability of detecting change, time and funding constraints prohibit excessive monitoring. Methods must be focused to maximise the likelihood that any changes to ecosystem structure and function will be observed. The design of monitoring strategies for deep water habitats (>40m) is complicated by the fact that many conventional techniques, such as visual censes, rely on scuba and are not practical in deep water. Deep water habitats are also further from shore and monitoring typically costs more due to the requirement for larger boats and specialist staff and equipment. The cost effectiveness of methods is a important consideration in the development of monitoring programs in deep water. The merits of a wide range of methods available for monitoring the various ecological groups in deep water habitats have been considered.

Meta-analysis based on long-term data sets from multiple marine reserves reveal that direct effects involving target species typically manifest far more rapidly than indirect ecosystem effects, i.e. *ca* 5.1 vs 13.1 years (Babcock *et al.* 2010). While this highlights the requirement for appropriate temporal scale in any monitoring program, it also demonstrates the logic in focusing a portion of monitoring resources on the groups targeted by fishing, in this case the lobsters themselves. Furthermore, a thorough understanding of the effect of protection from fishing on lobster abundance and size structure is fundamental to the development of any hypotheses relating to the potential for lobster fishing to have indirect effects on the ecosystem.

If the removal of lobster biomass by fishing does trigger widespread trophic effects, impacts may cascade down to primary producers resulting in lasting changes to ecosystem structure and function. Therefore, any monitoring regime in a closed area might aim to quantify change in the abundance or cover of macro-benthic primary producers. Obviously marine ecosystems are complex and there are many more taxa that may potentially react to changes in lobster fishing. This project also utilised the outputs of qualitative models in an attempt to predict ecosystem responses and the key trophic relationships driving responses. This allowed the identification and evaluation of potential indicators of ecosystem change which may become components of a comprehensive and cost effective long-term monitoring program.

6.0   Methods

6.1   Identification and assessment of suitable unfished reference areas to exclude rock lobster fishing in deep water.

6.1.1   Site nomination and selection

The EcoSRG recognised that for a closed area to be successful in addressing ecological concerns of the WRLF, the area would have to satisfy key selection criteria (Department of Fisheries 2012):

1. Be representative in terms of lobster demographics, i.e. have the potential for high adult biomass (relative to undersize biomass), as indicated by good or high catch rates of mature lobster;

2. Be central to and generally representative of the fishery (e.g. region between Lancelin and Dongara);

3. Have optimal accessibility – needs to be as close to shore/a 'port' as practical while satisfying other criteria;

4. Be representative of lobster habitat based on information obtained from previous habitat mapping (structure and function);

5. Have replicates of closed areas in different locations;

6. Be in an optimum location for enforcing compliance of the closure;

7. Meet the following size of site criteria:
   a. Complementary to the size of the lobster's foraging area;
   b. Large enough to allow measurements of indicator responses (both up/down the lobster food web, e.g. predators of lobster and key prey for lobster;
   c. Must encompass representative habitats; and

8. Minimise the relative level of economic loss to industry.

Negotiations with industry to establish a closed area within the WRLF commenced in 2007 with the formation of a Closed Area Working Group (CAWG), comprised of RLIAC (Rock Lobster Industry Advisory Committee) members, formed with the specific aim of identifying and ranking areas on their potential to become closed areas. The CAWG originally nominated six locations between the Abrolhos Islands and the Capes (see Table 6.1). Using the selection criteria formulated by the EcoSRG (see above). CAWG agreed that the two most suitable locations were the south eastern corner of the Abrolhos zone and the 30°S latitude line offshore from Leeman (Figure 6.1).

However, there were two significant unknowns in addressing the EcoSRG criteria that the area needs to be representative in terms of (1) lobster demographics and (4) habitat – (structure and function) (see Table 6.1). It was noted that additional information on lobster demographics and benthic habitats were required to make a final selection, and the CAWG requested that the Department of Fisheries (DoF) Research Division address (1) and (4) by investigating catch data from the two nominated areas and conducting short camera tows over the areas in early 2008.

Subsequently, a Scientific Advisory Group (SAG) with representatives from Universities (UWA, ECU and Murdoch) and Department of Environment and Conservation was also formed to independently review the methods to be used in this project, including the size and position of the closed area and provide recommendations to DoF for the implementation of a closed area.

6.1.2 Habitat surveys for site validation

Towed videos were conducted at the two shortlisted sites (Abrolhos and Leeman 30°S latitude line) in October 2008. At the Abrolhos, seven habitat transects running roughly from north to south were videoed, while at Leeman nine habitat transects were videoed (Figures 7.2 and 7.3). The videoed transects ranged in length from 2.48 to 6.20 km; all were between 35 and 75 m deep. The towed underwater video was a 'live feed' system consisting of a Dallmeier DF3000AS-DN HiRes UWDR Cam_inPix® colour box progressive scan camera with a F0.95/2.8-8.8 mm lens in an underwater housing attached to towed vane. The system was connected to the vessel by 10 mm rope and a reinforced video umbilical cable. The live feed video, with GPS overlay, was recorded onto a Sony DCRHC21 Mini DV Progressive Scan HandyCam. The camera was towed at between one and two knots, approximately 0.5 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 (Ben Software, London, UK), with the video stream set to pause every 0.013 s of latitude. Substrate and the main benthic groups within the field of view (1m diameter) were then identified, and the frame was assigned to a habitat class. The seven habitat classes were based on substrate and assemblage as well as measure of cover, i.e. medium, high (as in Figures 7.2 and 7.3). 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., non-sponge. *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 sand and limestone/rubble were classified separately; only presence or absence was recorded for these habitat types.
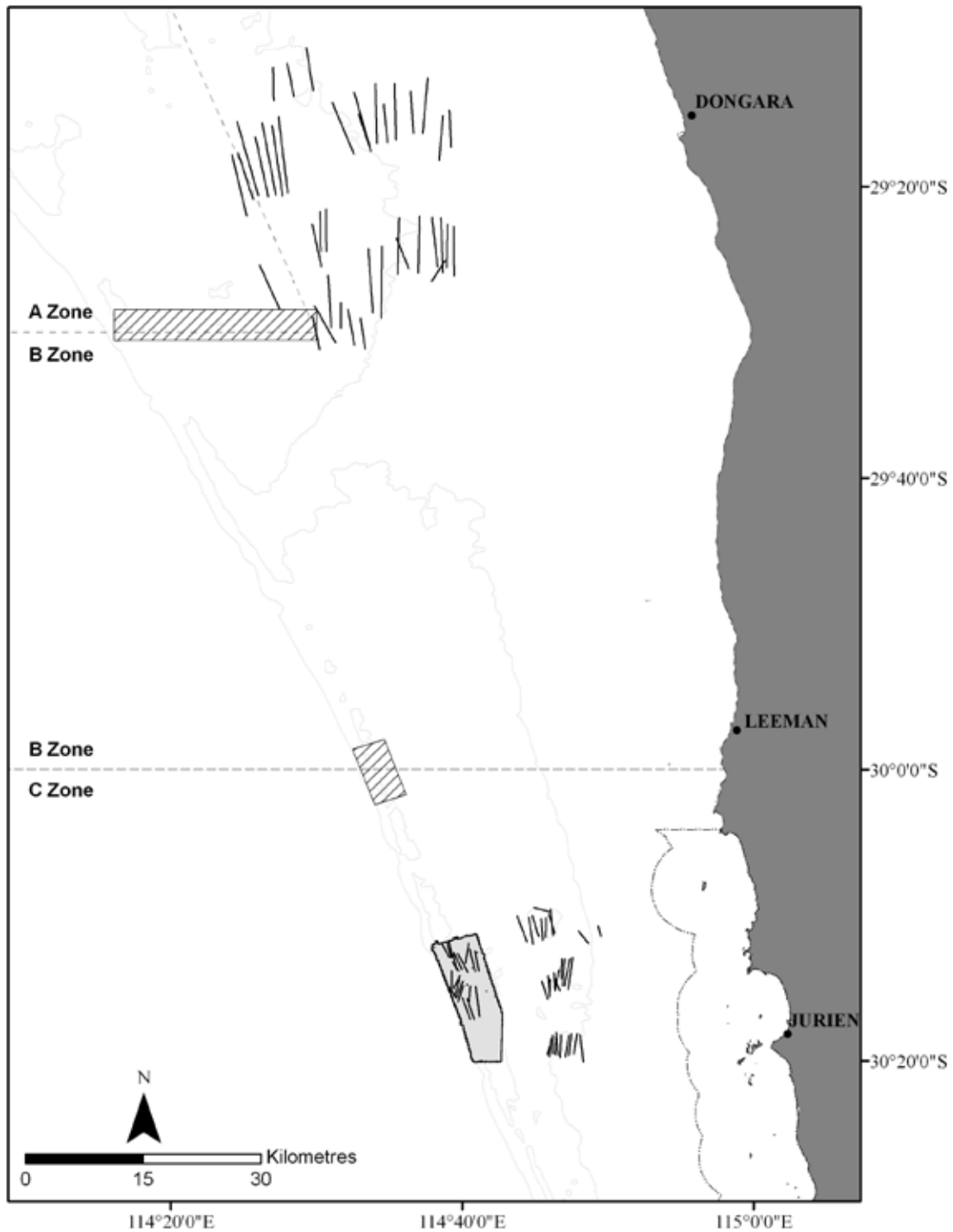
Figure 6.1. Location of potential areas short-listed to close to lobster fishing (hatched boxes) where further research was conducted to validate habitats and/or lobster demographics. Dashed lines indicate the boundaries between different management zones in the fishery. Short solid lines illustrate the approximate location of the independent breeding stock survey (IBSS) sites offshore from Dongra and Jurien. Shaded box indicates the extent of the high-resolution habitat map at the Jurien IBSS site, produced during the Securing Our Marine Futures project. The broken line inshore near Jurien shows the boundary of the Jurien Bay Marine Park. The 50 m and 100 m bathymetric contours are shown (grey).

Table 6.1. Details including the advantages and disadvantages of six potential research locations assessed against each of the closed area selection criteria formulated by the EcoSRG

EcoSRG Criteria	Location					
	Jurien	Dongara	Lancelin	30°S boundary	South of Abrolhos	Capes
Location details	Portion of Jurien IBSS	Near Dongara IBSS	Near Lancelin IBSS	Boundary between B and C zones	Boundary of A Zone	In Capes region
Background information	Focus area for current ecological project (FRDC) IBSS location – ca. 12yrs of data. IBSS can be used for representative fished comparison. Marine Futures study site – habitats already mapped.	Focus area for current ecological project (FRDC) IBSS location – ca. 12 yrs of data. IBSS can be used for representative fished comparison.	Focus area for current ecological project (FRDC) IBSS location – ca. 12 yrs of data. IBSS can be used for representative fished comparison.		Area subject to existing closure for part of year	Within proposed Capes Marine Park
Criteria 1: Representative	Yes	Unknown	Unknown	Unknown	Unknown	Unlikely as at southern-most limit
Criteria 2: Central to fishery	Yes	Yes	Yes	Yes	Unknown	Unlikely as at southern-most limit
Criteria 3: Optimal accessibility	18 nm from shore Research vessel access to harbour when dredged. Commercial vessel access good	25 nm from shore Research vessel access from Geraldton Commercial vessel access good	18 nm from shore Research vessel access from Fremantle. Commercial vessel access good	22 nm from shore. Closest access is Jurien	35 nm from shore Poor access	Coastal Research vessel access from Bunbury 20-30 nm
Criteria 4: Representative habitat	Yes Multibeam full coverage map	Unknown	Unknown	Unknown	Unknown	Unlikely to cover large enough areas in deep water
Criteria 6: Optimal compliance	FMO via Jurien Patrol Vessel	On patrol vessel schedule	On patrol vessel schedule	Already a compliance presence	Already a compliance presence for part of year	Unknown

EcoSRG Criteria	Location					
	Jurien	Dongara	Lancelin	30°S boundary	South of Abrolhos	Capes
Location details	Portion of Jurien IBSS	Near Dongara IBSS	Near Lancelin IBSS	Boundary between B and C zones	Boundary of A Zone	In Capes region
Criteria 7 :Size of site-criteria	Yes to a, b and c	Unknown, but should be similar to Jurien site	Unknown, but should be similar to Jurien site	Unknown, but should be similar to Jurien site	Unknown, but should be similar to Jurien site	Unknown
Advantages	Close to IBSS for direct comparison IBSS can be used as fished site. Full coverage habitat map available Close to Jurien – reduced compliance costs Some accessibility of research vessel to harbour.	Close to IBSS for direct comparison IBSS can be used as fished site. Close to Dongara – reduced compliance costs	Close to IBSS for direct comparison IBSS can be used as fished site. Close to Lancelin – reduced compliance costs	No interaction with IBSS (i.e. effort) Greater equity between zones Compliance already have a presence on the 30° S line Further away from recreational/charter fishing effort Less displacement of commercial fishing effort	No interaction with IBSS (ie effort) Compliance already have a presence part of the year. Further away from recreational/charter fishing effort Less displacement of commercial fishing effort?	No additional displacement of fishing effort

EcoSRG Criteria	Location					
	Jurien	Dongara	Lancelin	30°S boundary	South of Abrolhos	Capes
Location details	Portion of Jurien IBSS	Near Dongara IBSS	Near Lancelin IBSS	Boundary between B and C zones	Boundary of A Zone	In Capes region
Disadvantages	May cause an increase in effort in IBSS area. Due to its proximity to IBSS site may distort some of the results. Will displace large amount of fishing effort. Inequity between zones (i.e. effects C Zone only). Will displace charter and recreational effort being directed out from Jurien	May cause an increase in effort into the IBSS area. Will displace large amount of fishing effort. Inequity between zones (i.e. effects B Zone only). Will displace charter and recreational effort being directed out of Dongara. No habitat map. Higher start-up costs in terms of habitat validation and mapping requirements.	May cause an increase in effort into the IBSS area. Will displace large amount of fishing effort. Inequity between zones (i.e. effects C Zone only). Will displace charter and recreational effort being directed out of Lancelin. No habitat map. Higher start-up costs in terms of habitat validation and mapping requirements.	Not adjacent to IBSS sites, not comparable. Further offshore (ca. 22 nm). Less knowledge of catches. No habitat map. Higher start-up costs in terms of habitat validation and mapping requirements. Will require sampling of representative fished areas (similar to IBSS sampling). Represents a long-term cost beyond project life	Not adjacent to IBSS sites – not comparable. Further offshore (ca. 35 nm). Less knowledge of catches. No habitat map. Higher start-up costs in terms of habitat validation and mapping requirements. Will require sampling of representative fished areas (similar to IBSS sampling). Represents a long-term cost beyond project life	Not central to fishery. Unrepresentative in terms of lobster population dynamics (adult biomass) and migration patterns. Sporadic recruitment and catch. No habitat map. Higher start-up costs in terms of habitat validation and mapping requirements. Will require sampling of representative fished areas (similar to IBSS sampling). Represents a long-term cost beyond project life
Related costs		Habitat mapping (Once off cost of ca. $ 50k)	Habitat mapping (Once off cost of ca. $ 50k)	Habitat mapping (Once off cost of ca. $ 50k)	Habitat mapping (Once off cost of ca. $ 50k)	Habitat mapping (Once off cost of ca. $ 50k)

6.1.3 Lobster Surveys

Abundance and demographics of lobsters at Leeman were validated by lobster potting. Sampling was initially conducted in October 2008 during the western rock lobster independent breeding stock survey (IBSS). This annual ten day fishery independent survey has been conducted at a number of sites along the Western Australian coast since the early 1990s and employs standardised methods including pot type (commercial pots with closed escape gaps), soak time (two days) and bait type, i.e. 1.5 kg of skipjack tuna heads and Atlantic herring (Melville-Smith *et al.* 1998).

Leeman was surveyed in combination with the nearby Jurien IBSS site which is located 27–37 km offshore from Jurien (114º 44′E, 30º 15′S) (Figure 6.1). Jurien IBSS consists of five subregions (SR), two on the outer bank and three on the inner bank (Figure 6.1). For the purpose of this study, only lobster data from SR3 and SR4 (on the outer bank) were used as these subregions have a pre-existing full coverage habitat map and have consistently supported high numbers of lobsters. While the sub-region numbers have been provided here to be consistent with the previous FRDC study (Bellchambers 2010), for clarity SR3 and SR4 are referred to hereafter as Jurien South (JS) and Jurien North (JN), respectively.

At both Jurien sub-regions, 10 lines of 16 pots were set over 10 days (giving a total of 160 pot lifts for each sub-region). Pots were deployed from chartered commercial western rock lobster vessels in waters 45-70 m deep with lines running north to south roughly parallel with the coast. Individual pots were approximately 200 m apart and each pot has its own GPS coordinate so the pot locations are consistently sampled between years (Bellchambers 2010).

The data collected include the number of lobsters, carapace length (mm), sex, reproductive state (presence/absence of ovigerous setae and visible gonads) and egg stage where applicable. Damage to each lobster was recorded as the number of legs or antennae that were lost or newly regenerated. Evidence of the presence of predators was noted as this is known to affect catch rates (Brock *et al.* 2006), as was any condition of individual pots that may have affected catch (i.e. soak time, open escape gap or gate, missing batten). Occurrences of by-catch were also recorded. All lobsters were released following data collection. While the original sampling locations at Jurien were selected without the aid of detailed habitat information, outputs of FRDC 2004/049 have informed of the main habitats found at each line and the results have been interpreted accordingly where appropriate.

The proposed closed area at Leeman near the 30°S latitude line (Figure 6.1) was added to the existing survey in 2008 and then sampled again in 2009. With no detailed habitat map having been produced for the Leeman area in 2008, a systematic potting regime was implemented across the research area with 270 pots deployed *ca.* 400m apart, in nine lines of 30 pots. The lines ran north to south parallel with the coast in depths of *ca.* 45 m to 70 m. In 2009, the most eastern line was not re-sampled, but instead a further two lines were sampled to the west (in deeper water up to *ca.* 85 m). On each sampling occasion biological data was again collected and recorded as described above for Jurien.

6.1.4 Habitat Mapping

6.1.4.1 Bathymetry and biological data collection

To complete full coverage information on terrain, substrates and benthic biota, detailed information of the seafloor was captured. A full coverage dataset for the study area consisted

of bathymetry from hydroacoustic surveys collected in 2010, using a SEABAT 8101 Reson Multibeam (for detailed methods see Hovey *et al.* 2012). Substrates and benthic biota were observed using video footage from an underwater camera towed behind a boat travelling 1–2 knots per hour. The camera was held at approximately 1 m above the seafloor and the position geo-referenced using an Ultra Short Base Line (USBL) acoustic positioning system linked to a GPS with satellite differential correction. Sampling was designed to ensure a broad geographic coverage of the study area with sufficient numbers of geo-referenced data for modelling and mapping.

A total of 55 km of seafloor was sampled with the video over an area of 52 km^2. Transects were designed to run perpendicular and parallel to the coastline to cover ecological gradients, and areas -with high and low variability were expected. Video classification involved identifying primary and secondary substrates and benthic biota, as well as biota density. Primary substrate was classified as sand, rhodoliths (hard structures of coralline algae typically at low points on sandy substrates), obscured reef (hard substrate covered with sand veneer), flat reef, or low, medium, or high profile reef. Biota categories were *Ecklonia* sp. (kelp), other macroalgae and sessile invertebrates.

6.1.4.2 *Predictive modelling of habitats*

Bathymetry and derived terrain datasets were used as input to species distribution models for predicting substrate type and biota at unsampled locations (Table 6.2). Classification trees (CTs) were developed in S Plus H 8.2 for Windows (TIBCO Software Inc, Palo Alto, California, USA) and used to predict (in order) substrates, benthic biota, and finally lobster distributions. CTs explain the variation of a response variable by one or more predictor variables and are constructed by recursively partitioning data and splitting into mutually exclusive groups. The objective is to partition the response into homogeneous groups while keeping the tree size small. Splitting continues until the stopping criterion (e.g. minimum deviance) is reached, then the tree is pruned back to an optimal size using cross-validation (for detailed methods see Hovey *et al.* 2012). Separate models were developed for each type of substrate and biota category. Video observations of medium and high relief reef were merged together with low relief reef creating a comprehensive reef class. Classes modelled to create a full coverage substrate map were sand, rhodoliths, obscured reef, flat reef and reef. Full models for biota categories were developed using bathymetry, terrain datasets and substrate types.

The same predictive methods were employed using existing bathymetry and derived terrain data sets for the Jurien sites. These data were collected as part of the Securing our Marine Futures Project and used in a previous FRDC project (FRDC 2004/049) (see Bellchambers 2010 and Bellchambers *et al.* 2010).

Table 6.2. Datasets derived from bathymetry used as predictors of substrate and biota (Adapted from Holmes *et al.* 2008).

Predictor datasets	Definition
Bathymetry	Depth relative to the Australian Height Datum
Aspect	Direction of the steepest slope (0-360°), calculated on 3 x 3 pixel area
Slope	Average change in elevation with distance calculated on 3 x 3 pixel area
Profile curvature	Measure of concave/convexity parallel to the slope (e.g., hill cross-section), calculated on 3 x 3 pixel area
Plan curvature	Measure of concave/convexity perpendicular to the slope (e.g., contour lines), calculated on 3 x 3 pixel area
Focal analysis	Statistical operation that computes a value for each cell as a function of cells that are in a specified neighborhood around a focal cell, calculated as standard deviation of surface area with kernel radius of 7 m and 21 m.
Curvature	Combined index of profile and plan curvature
Hypsometric index	Indicator of whether a cell is a local high or low point within a neighborhood of 12.5, 25 and 50 m kernel radius
Range (local relief)	Maximum minus the minimum depth in the local neighborhood of 12.5, 25 and 50 m kernel radius
Standard deviation	Standard deviation of depth within a neighborhood of 12.5, 25 and 50 m kernel radius
Rugosity (surface area)	Actual surface area of local neighborhood

6.2 Development of a qualitative trophodynamic model that will provide a conceptual framework for determining sampling protocols, indictors and targets.

6.2.1 Qualitative modelling

The application of qualitative modelling in conceptualising ecological and social systems and the general methodologies are summarised in Metcalf et al. (2009). In this project, qualitative trophodynamic models were developed for the deep water (40-70 m) benthic ecosystem off the west coast, i.e. near Jurien. The precise methods and rational behind the models constructed for Jurien are given in detail in Metcalf *et al.* (2011). In brief, models were constructed using signed digraphs to represent the signs of interactions between species or groups (variables) in the system. For example, the direct effect of a predator on its prey is represented by a negative link, whereas the direct effect of a prey on its predator gives rise to a positive link (Figure 6.2). Negative self-effects were used to represent a reliance on factors external to the modelled system, such as nutrients or light, as well as density dependent effects.
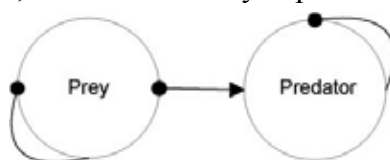
Figure 6.2. Signed digraph representing the relationship between a predator and its prey. The predator has a negative direct effect on the prey (dot and line), whereas the prey has a positive direct effect on the predator (arrowed line). Negative self-effects represent a reliance on factors external to the system or density-dependence.

These direct interactions between variables (+, –, 0) in a model, i.e. system structure, can also be represented in the community matrix, and the signs of the adjoint of the negative community matrix was used to detail qualitative predictions of response (direction of change in population abundances) to press (long-term) perturbation using both direct and indirect effects among community members. In complex systems, variables may be impacted via multiple pathways. If the potential pathways have opposing signs, the effects cancel each other and create ambiguity in predicting a response. Obviously, the overall effect predicted will be dependent on the balance (or net) of the various pathways (positive or negative). To assess ambiguity, predictions can be given weights, based on the net of the pathways divided by the total number of pathways (Dambacher *et al.* 2003). The methods of Hosack *et al.* (2008) where used to provide an index of sign determinacy and indicate the reliability of the prediction signs.

6.2.2   Study site

Qualitative trophodynamic models were developed for the deep water (40-70 m) benthic ecosystem off Jurien and Leeman. This region was selected as it is located toward the centre of the western rock lobster fishery and has been demonstrated to be representative of the wider fishery in terms of habitats, fishing effort and lobster catch (Bellchambers 2010, Bellchambers *et al.* 2010). This region is located near the centre of the West Coast Bioregion of Western Australia which extends between Kalbarri and Augusta. The West Coast bioregion is characterised by exposed sandy beaches with a limestone reef line approximately 5 kilometres off the coast. Sea floors further offshore on the continental shelf are typically composed of coarse sand interspersed with low limestone reef, which are remnant of old shorelines. A common submerged limestone shoreline is a major bathymetric feature of the Jurien and Leeman study sites. A common set of trophodynamic models were developed for the Jurien and Leeman sites as these sites were assumed to be analogous prior to the introduction of a lobster fishing closure. These models were then modified to assess the different ecosystem questions, i.e. the effect of bait addition or closure to lobster fishing (see 6.24 below).

6.2.3   Collation and review of trophic information

Qualitative trophodynamic models were developed using dietary information on a range of species that reside within each ecosystem, collected from the published literature and researchers. If available, dietary information from the specific ecosystem was used as input in the qualitative models (Table 6.3). However, when site-specific information was not available, dietary data from similar areas in south-western Australia were used. Both the published literature and expert scientific opinion was used to determine the species or groups to be included in the complex model. Species selected for inclusion were also assumed to spend the majority of their time within the ecosystem. For example, dolphins may enter these areas occasionally, yet were thought to be too transient to have a large impact on organisms within the ecosystem. A workshop was held in June 2009 to review the modelling methods, the ecological information and assumptions and how these models could be used to provide a conceptual framework for determining the appropriate sampling protocols, indictors and targets to evaluate the ecological impacts of fishing. Workshop attendees included international experts in the areas of qualitative modelling, modelling of marine systems, identification of ecological indicators, ecological effects of exploitation and the biology and ecology of lobsters (Table 6.4).

6.2.4   Model construction

A 'complex' trophic model was constructed initially to include all of the available dietary information for the deep water ecosystem. Species or groups were selected for input into the

complex deep water model if they were thought to be prey, predators, competitors or habitat for western rock lobster (33 variables; Figure 6.3a). The complex model was then simplified to reduce the uncertainty of model predictions (Metcalf *et al.* 2008). Simplification was undertaken using an iterative process to aggregate variables or remove variables and/or links from the models. Three methods were used in the simplification process: logic, expert opinion, and the REGE algorithm (Luczkovich *et al.* 2003, www.analytictech.com/downloaduc6.htm).

Table 6.3. Dietary references used to construct the qualitative trophic model of the deep water benthic ecosystem off Jurien, Australia.

Species/group	Dietary references
Isopods	K. Waddington pers. comm.
Crabs	
Amphipods	Crawley *et al.* 2009
Polychaetes	Hutchings 1998
Small gastropods	Edgar 2000
Large herbivorous gastropods	Edgar 2000
Large carnivorous gastropods	Edgar 2000
Echinoids	Vanderklift 2002, Vanderklift *et al.* 2006
Old wife	Howard 1988
Foxfish	Cossington 2006
Small fish (e.g. black spot wrasse (*Austrolabrus maculatus*), rough bullseye (*Pempheris klunzingeri*)	Froese and Pauly 2012, Platell and Potter 2001
Footballer sweep	Froese and Pauly 2012
Old wife	Howard 1988
Foxfish	Cossington 2006.
Small sharks and rays	Kailola *et al.* 1993, Last and Stevens 1994, Platell and Potter 2001
Breaksea cod	Moore 2005.
Dhufish	Lek 2004
Pink snapper	Peng 2003
Baldchin groper	Lek 2004, E. Lek unpublished data
Octopus	Joll 1977
Sea lions	Costa and Gales 2003, Fletcher *et al.* 2005

Table 6.4. Participants in deep water rock lobster qualitative modelling workshop held 4 June 2009

Attendees	Institution
Dr Jeff Dambacher	CSIRO Mathematical and Information Sciences
Dr Verena Trenkel Dr Marie-Joelle Rochet	Laboratoire Mahera, IFREMER (Institut Français de Recherche pour l'Exploitation de la MER)
Dr Michael Fogarty	Northeast Fisheries Science Centre, NOAA (National Oceanic and Atmospheric Administration)
Assoc/Prof Stewart Frusher	Tasmanian Aquaculture and Fisheries Institute
Dr Dan Gaughan Dr Nick Caputi Dr Simon de Lestang Dr Lynda Bellchambers Dr Sarah Metcalf Dr Matt Pember	Department of Fisheries, Western Australia Department of Fisheries, Western Australia Department of Fisheries, Western Australia Department of Fisheries, Western Australia Department of Fisheries, Western Australia Department of Fisheries, Western Australia

For example, the complex model was first simplified using logic by removing variables that interact with only one other variable (i.e. octopus fishery only interacts with cephalopods in Figure 6.3a), since these do not produce any net feedback that affects the rest of the system (Puccia and Levins 1985). These variables were collapsed into a single variable (i.e. cephalopods), where a change in the fishery is represented through a negative perturbation (decrease in abundance) (Figure 6.3b; 11 variables).

Alternative simplified models were produced for each case study to deal with uncertainty in how the models should be structured. The comparison of results from models with different structures is useful as it allows the identification of species or groups in the model that are sensitive to structural uncertainty (i.e. results differ between models) and those that are predicted to occur despite this uncertainty (i.e. results are consistent between models) (Hayes *et al.* 2008). These two types of results can be useful in different ways. Results that differ because of model structure can provide an indication of the role the altered link plays in the dynamics of the system and the responses that may be observed in such a situation. Results that are consistent between different models suggest the response will occur regardless of structural uncertainty and this species/group may be a good indicator of change in the perturbed variable. Potential indicators of change were therefore selected only if the sign of the predicted response was consistent in the adjoint matrices from different models.

Due to similar ecology and life-history, the three gastropod variables (small herbivorous gastropods, large herbivorous gastropods and large carnivorous gastropods) were aggregated (Gastropods) into one variable. Footballer sweep (*Neatypus obliquus*), old wife (*Enoplosus armatus*), king wrasse (*Coris auricularis*), foxfish (*Bodianus frenchii*) and small fish (unidentified) were also aggregated into one variable (general fish) for the same reason. Amphipods and isopods (small crustaceans) were aggregated due to similar ecology and life-history and the potential benefit to both groups from the input of bait by the rock lobster fishery. Prior to aggregation, small fish (unidentified) was the only variable with links to predators included in the complex model; this was due to references including 'unidentified teleosts' as a major dietary item of species such as dhufish (*Glaucosoma herbraicum*), breaksea cod (*Epinephilides armatus*) and Australian sea lions (*Neophoca cinerea*), while predators for specific species (e.g. footballer sweep) were unknown. Similar to the aggregation of species into general fish, dhufish, pink snapper (*Pagrus auratus*), baldchin groper (*Choerodon rubescens*) and breaksea cod were aggregated into demersal fish. All of these species have comparable functional roles in the trophic web and the rock lobster fishery (i.e. are common bycatch species; Fletcher and Santoro 2009), as well as similar life histories and habitat requirements (Nardi *et al.* 2006, Moore *et al.* 2007, Fletcher and Santoro 2009, Froese and Pauly 2012).
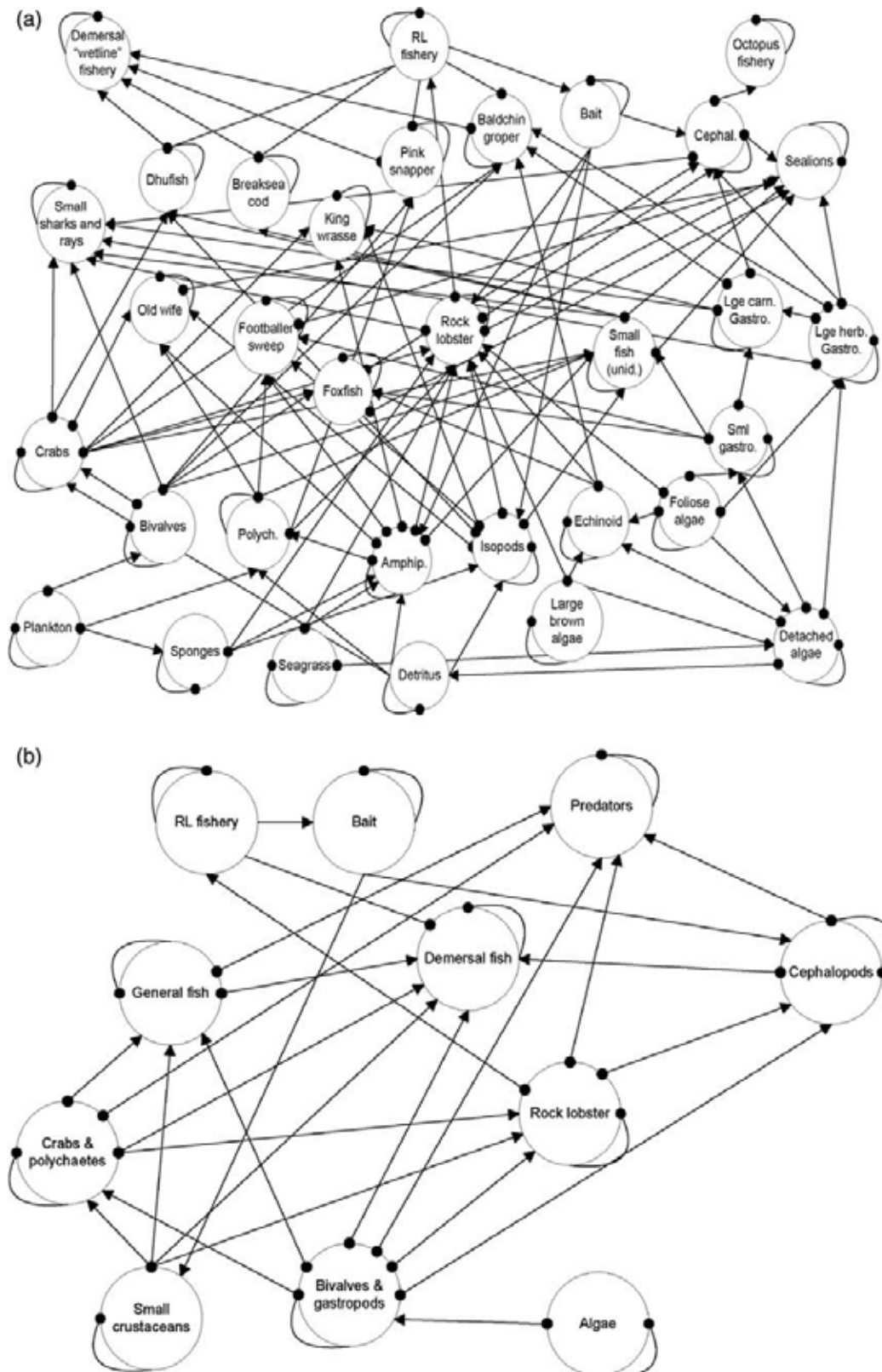
Figure 6.3. The (a) complex Jurien model with 33 variables before simplification, and (b) the simplified model with 11 variables. Variations on this model that can be attributed to structural uncertainty are complex.

Expert opinion was used to remove 'weak' links from the complex model. 'Filtering' of links

must occur in qualitative models because all links are assumed to be equal. If this is not the case in reality, weaker links need to be removed from the system to ensure a greater importance is not attributed to these ecosystem dynamics and relationships than is warranted. Echinoids were removed from the complex model, as the predatory link from rock lobster to echinoids was considered to be inconsequential in the Jurien deep water ecosystem (K. Waddington, pers. comm.). Similarly, predation on sponges is thought to occur incidentally while rock lobster prey on amphipods and isopods that live within sponges (K. Waddington, pers. comm.), and the predatory link between rock lobster and sponges was therefore removed.

The REGE algorithm was used in a similar fashion to Metcalf *et al.* (2008) to objectively identify 'like' variables for aggregation (Figure 6.4). This algorithm includes predators and prey in the assessment of similarity (regular equivalence) between variables and has been shown to produce relatively small aggregation error (Metcalf *et al.* 2008). An understanding of the ecological system needs to be applied to the results obtained using the REGE algorithm, as this method cannot take into account different functions or life-history. For instance, rock lobster, general fish and cephalopods were identified as similar using this method; however, these variables were not aggregated, as they play a different role in the rock lobster fishery (i.e. octopus prey directly on rock lobster in pots) and cephalopods may also benefit directly from the fishery through the consumption of bait.

The model was then further simplified using logic by removing variables that have only one-way interactions (e.g. plankton, detritus). These variables do not affect the overall dynamics of the system and were instead treated as parameters of the variables they connected to (Puccia and Levins 1985) (Figure 6.3b). Further simplification was not undertaken, as an 11 variable model was considered to be an appropriate size for the identification of indicator groups. Any additional aggregation may have resulted in the forced aggregation of dissimilar groups and may have confounded the identification of indicators.

Structural uncertainty exists in the simplified deep water trophic model for two reasons. Firstly, the overall impact of bait on cephalopods and the deep water ecosystem in general is unknown. Waddington and Meeuwig (2009) suggest that the input of bait may have a substantial impact on deep water (> 36 m depth) trophic systems by providing an additional resource to cephalopods and small crustaceans, particularly isopods. Links from bait to these variables (cephalopods and small crustaceans) have therefore been retained in some simplified models (Models A, B and C; Table 6.5). To assess the possibility that octopus do not feed on the bait within pots, the link from bait to cephalopods was removed in Models B and C. In addition, to include the possibility that the impact of bait is relatively small for both cephalopods and small crustaceans, all bait links were removed in Models D and E. A second source of structural uncertainty occurred because the extent to which, if any, the rock lobster fishery injures octopus during pot lifting is unknown. To deal with this uncertainty, a negative link from the fishery to octopus was included in Models C and D and excluded in Models A, B and E. Discrepancies in the results between structurally different models were then used to highlight specific impacts that bait and/or fishery impacts on cephalopods may have on ecosystem dynamics.

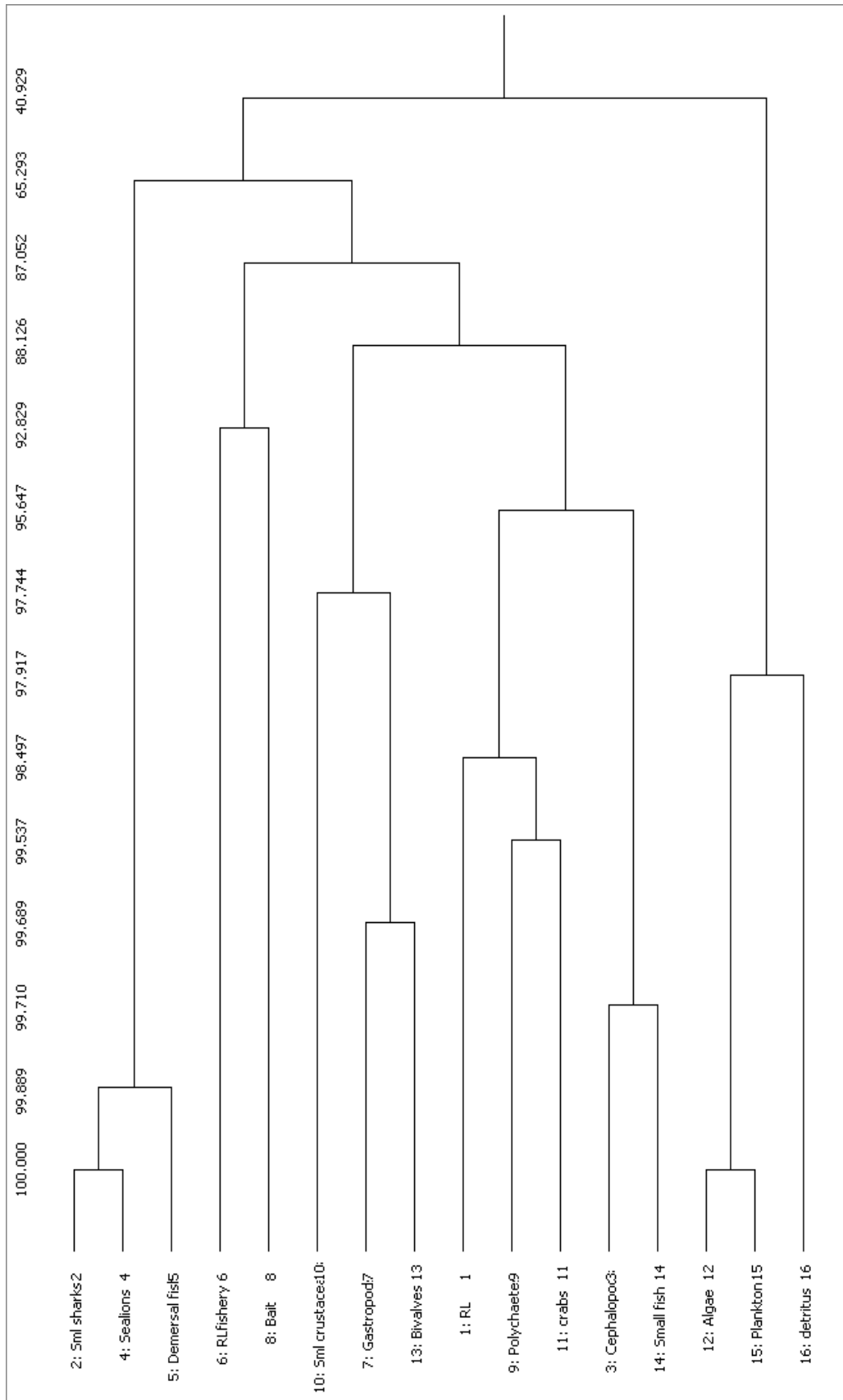**Figure 6.4.** Dendrogram showing the similarity (regular equivalence shown in numbers at the top) using the REGE algorithm between variables in the partially simplified deep water ecosystem model (Figure 6.3b). Lines surround the variables that were aggregated into groups.

Table 6.5.   Variations on the simplified Jurien models (Figure 6.3b) to allow investigation of structural uncertainty on model predictions.

Model	Links included/excluded
A	Bait links to small crustaceans and cephalopods included
B	Bait link to bivalves and gastropods included, but bait link to cephalopods excluded
C	Same as model B, plus a negative link from the rock lobster fishery to cephalopods
D	All bait links excluded, but negative links from the rock lobster fishery to cephalopods included
E	All bait links and negative link from rock lobster fishery to cephalopods excluded

6.2.5   Identification of indicators

Following the production of structurally different models, the 'average proportion of correct sign' (Hosack *et al.* 2008) was used to provide an index of sign indeterminacy. Sign indeterminacy refers to the possibility that the predicted response to a perturbation may be overwhelmed by a strong response in the opposing sign (Dambacher *et al.* 2002). For example, if there is one negative and three positive feedback cycles contributing to the response of a variable, the predicted response in the adjoint matrix will be positive. This occurs because the negative cycle cancels a positive cycle, leaving two positive feedbacks in the adjoint matrix. If, however, the negative feedback is very strong, it may overwhelm the positive cycles, and a reduction in the abundance of the variable would be observed instead of the predicted increase. An average proportion of correct sign > 0.80 is thought to be reasonably high (Hayes *et al.* 2008) and assumed to be reliable.

Variables with consistent prediction signs in the adjoint matrices of the different models and a consistently high average proportion of correct sign (> 0.80) in response to a change in the rock lobster fishery were identified as potential indicators (Hayes *et al.* 2008). The specific variables that were perturbed to investigate the impact of change were not assessed as potential indicators because change in these variables is known and does not directly indicate whether change in the broader trophic system is occurring. These perturbed variables were bait, rock lobster and the rock lobster fishery.

Once the indicators had been determined, these variables were disaggregated to determine which, if any, specific groups/species could provide a better indication of change than the aggregated variable. The disaggregation of the indicators produced models of an intermediate complexity in comparison to the simplified and complex models. For example, general fish was disaggregated into four variables: sweep and wrasse, foxfish, old wife and small fish (Figure 6.5). The separation of species into disaggregated variables was undertaken based on their similarity of prey and predators. The predatory links to small fish from predators and demersal fish were retained as these relationships were highlighted in the literature. In addition, bait links to small crustaceans and cephalopods (as in Model A) were retained, as this scenario was thought to be the most likely alternative.

Aggregation error (Gardner *et al.* 1982, Cale *et al.* 1983, Auger *et al.* 2000) was used to provide an indication of the level of uncertainty that arose through the aggregation of variables (in the simplified and intermediate models) and the reliability of the results in comparison to the complex models. Aggregation error was calculated following the methods of Metcalf *et al.* (2008), whereby the number of predictions in the simplified and intermediate models that differed from the complex model (Figure 6.3a) were expressed as percentages. Investigation into aggregation error in models of intermediate complexity was used here to investigate

whether the disaggregation of identified indicator variables would reduce error and increase the reliability of the results.

The average proportion of correct sign was assessed for all variables disaggregated from the previously identified indicator. This was undertaken to determine whether one of these variables would provide a more reliable indicator than the other disaggregated variables or the aggregated variable, general fish.

6.3 Provision of cost effective methods to measure deep water ecosystems in both fished and unfished reference areas.

6.3.1 Lobster

6.3.1.1 Underwater visual census and potting survey comparison

Lobster demographics in deep water were derived from catches made with lobster pots. As described below, two types of pots were used; a) standard commercial pots and b) small meshed pots. Small meshed pots are commercial batten pots modified to have all gaps between the battens reduced to <15 mm (see Bellchambers *et al.* 2009). Due to the water depth at the Jurien and Leeman deep water sites potting was more appropriate than other methods, such as under water visual census (UVC) using divers, to sample lobsters. To understand the biases associated with relying on pot derived data, a comparison of methods was undertaken in medium depth waters. By limiting the comparison to waters of *ca* 15-25m, UVC could be included. A region near Leeman with a similar latitudinal range to that of the deep-water closed area was sampled in 2011 (Figure 6.5). Side-scan sonar and scuba divers were used to find areas that had appropriate habitats and depths. Sites were selected with importance placed on continual stretches of lobster habitat that would be conducive to conducting ten minute timed UVC survey transects.

Paired divers on SCUBA conducted a total of 18 UVC over two days in April 2011. Each survey consisted of a ten minute timed transect that followed areas likely to shelter lobsters. The first diver recorded lobster abundance and estimated carapace length (CL). The second diver followed and attempted to collect any observed lobsters by hand-loop. All diver-caught lobsters were measured using callipers to the nearest millimetre.

A potting survey, using both standard commercial pots and small meshed pots, was conducted using a commercial fishing vessel following the UVC surveys. Potting concentrated on areas of high relief in the same depth range as the visual census. The commercial pots used in the medium depth comparison differed from those used in deep water (see section 6.1.3 and below) by not having the escape gap closed. A total of 72 pot lifts were performed (35 – commercial; 37 – meshed). The bait used was consistent with those used in deep water and biological data was recorded as described in section 6.1.3. All lobsters collected on the first day of sampling were tagged with an external antennae cable tie tag to identify recaptures.
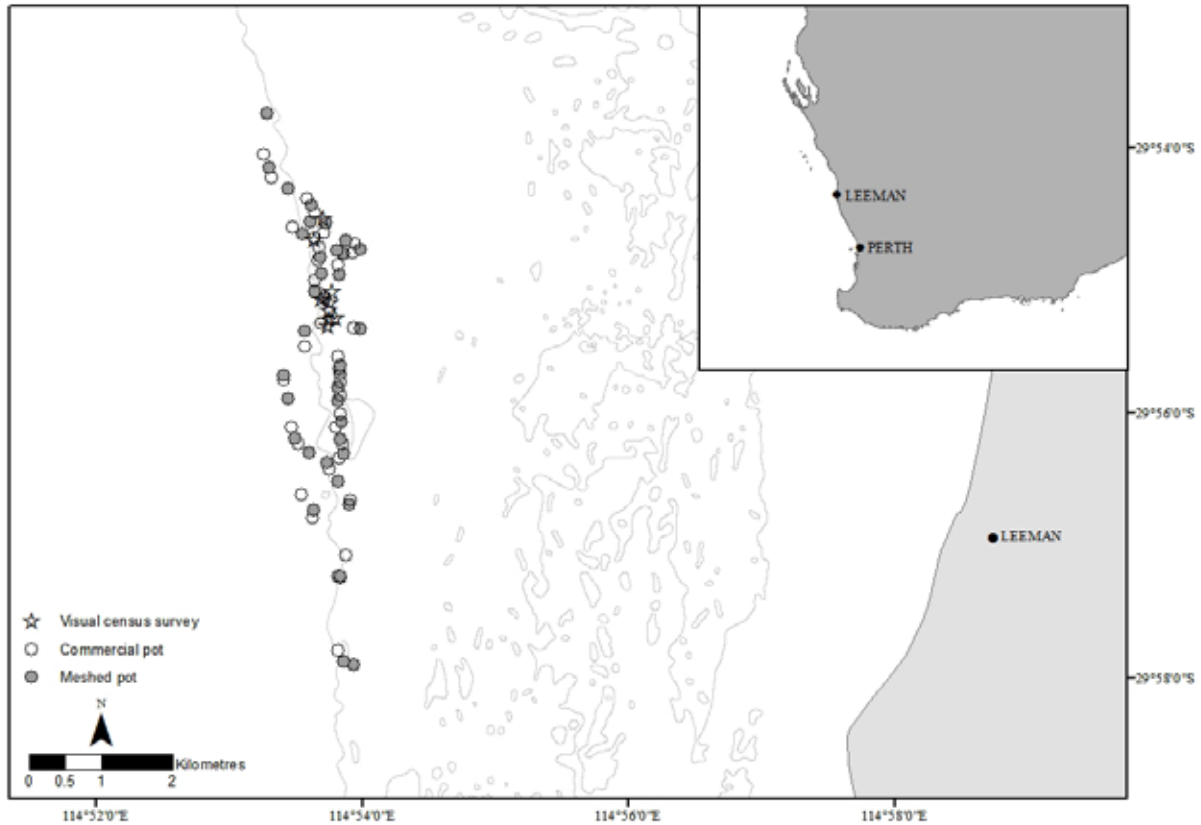
Figure 6.5. Map of medium depth (15-25m) sites west of Leeman where the sampling method comparison was conducted.

6.3.1.2 Deep water lobster sampling

Lobsters were primarily sampled in deep water at Jurien and Leeman using standard commercial lobster pots with closed escape gaps, as described in section 6.1.3. In addition to the catch data for 2008 and 2009 (reported in section 7.1.3), lobsters continued to be sampled annually at both Jurien and Leeman between 2010 and 2012, giving abundance and demographic data for five consecutive years.

The northern and southern sub-regions at Jurien, both of which have stayed open to fishing, were treated as separate sites in analyses (JN and JS). Each of the Jurien sites was sampled with 160 pots. Pot catches at Leeman were also allocated to two sites; L1: the areas that stayed open to fishing (64 pots) and L2: the area that eventually became closed to fishing. Thus four sites were sampled in each of the five years (three fished and one that became closed to fishing) (Figure 6.6).

At each location, sampling was spatially consistent with pots being set on the same predetermined marks each year. This produced geo-referenced lobster data and allowed a) samples to be easily categorised by geo-physical characteristics, substrates and biota and b) catch data to parameterise spatial modelling (see lobster distribution modelling below).
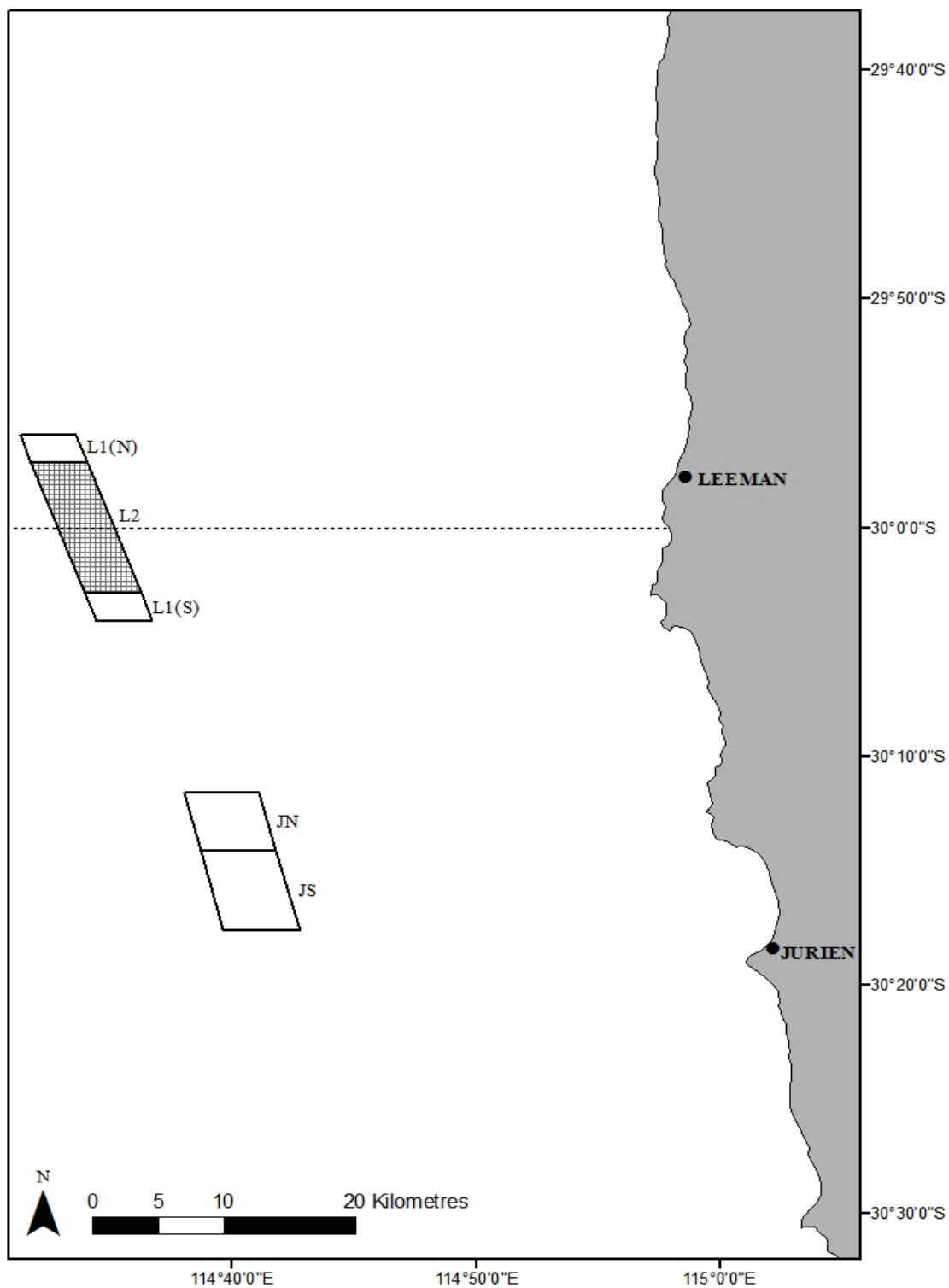
Figure 6.6. Map showing sites used in analysis of commercial and small meshed pot catch data at Jurien and Leeman. JN and JS refer to Jurien north and Jurien south, respectively. Note that both JN and JS stayed open to lobster fishing. The two L1 sites (N & S) depict the northern and southern areas surveyed at Leeman that stayed open to lobster fishing. L2 is the area sampled at Leeman that became closed to fishing in 2011.

6.3.1.3 Analyses of catch data

The catch rates of all (total) lobsters, legal sized lobsters, legal sized male lobsters and undersized lobsters were subjected to ANOVA with site, year and habitat treated as factors. The daily means of the catch across pots in each of these combinations were used as replicates. Protection from lobster fishing was not treated as a factor (as it only came into effect at one site after the third year) and any potential effect of protection may be identified through significant interactions between site x year. In each case above, only the two-way interactions were interpreted as the three-way interactions were found to be highly non-significant (i.e. p>0.9). The data was treated in the same way for corresponding general linear models (GLMs).

Examination of the relationship between the means and standard deviations of catch rates for the different lobster groups (total, legal, legal males and undersize) suggested that in each case, prior to analysis of variance (ANOVA) being undertaken, a square root transformation of the data was appropriate (see Clarke and Gorley 2001).

Habitat categories used in the analyses were generated from point predictions (for each individual pot location) from the habitat models in objective 1. To provide the greatest consistency between the different areas general substrate categories were used, i.e. reef, rhodoliths and sand. However, a portion of the pots from site L1 (Leeman fished) fell outside the region hydro-acoustically mapped. Habitat information for these pots was augmented with video imagery collected with pot mounted cameras as part of the field work for a related research project (FRDC 2011/021 Development of an industry-based habitat mapping/monitoring system). Video of habitats was collected from baited remote underwater videos set in this location (Section 6.3.3).

6.3.1.4 Lobster distribution modelling

The geo-referenced (commercial) pot catch data collected at Leeman from 2008 to 2010 was used to derive a matrix of lobster presence/absence, corresponding with the area mapped in Chapter 6. This presence/absence information was then used to parameterise a predictive model of lobster occurrence. The classification tree for the lobster model was developed as for other biota classes using 75% (chosen using the set seed function in S Plus) of the pot catch data over three years as the dependent variable with predictor variables from bathymetry, terrain, substrate and benthic biota classes (see objective 1). Like the substrate and biota models, the 10- fold cross-validation method was used to determine key predictors and optimal tree size. The remaining 25% of the data were used to validate the model. The final model was applied to the predictor datasets to create a full coverage map of predicted lobster presence/absence (see Hovey *et al.* 2012).

6.3.1.5 Small meshed pot data

Small meshed pots were introduced in 2010 to sample a wider size-range of lobsters in deep water. As described above, these pots are commercial batten pots modified so that all gaps between the battens are reduced to <15 mm. Small meshed pots were sampled concurrently to the normal sampling regime but were set <400 m from other pots. The type and quantity of bait and data recorded during the meshed pot sampling was consistent with the commercial pot sampling. However, soak time varied as the meshed pots were set for one day (weather permitting) rather than two. Mesh pots were not typically set on the same location each year but a record was kept of the precise location of each pot pulled so that the associated lobster catch data was geo-referenced and matched against geo-physical and biotic habitat characteristics. The only year where catch rates from small meshed pots were subjected to ANOVA was 2012 as sampling intensity was unbalanced between sites and habitats in previous years.

6.3.1.6 Lobster tagging

When practical, all males without extensive damage (i.e. multiple leg loss) and with a carapace length greater than 60mm were tagged. An internal T-Bar anchor tag was inserted in the ventral side of the lobster into the muscle between the first and second abdominal segments as described in Melville-Smith and Chubb (1997). A small number of non-setose females with a carapace length greater than 60mm were also tagged.

After lobsters were tagged, they were returned to the water as close as practical to the location the geo-referenced pot was retrieved from. Some individuals were subsequently recaptured from geo-referenced (research) pots enabling the small-scale movement and habitat use of these lobsters to be explored. Lobsters that were caught within three months of being tagged were removed from analysis.

6.3.1.7 Lobster dietary studies

Sample collection

Samples of lobsters for dietary studies were collected as by-catch from the west coast demersal gillnet fishery. Samples were collected on two separate occasions: summer (Nov/Dec 2010) and autumn (Mar/Apr 2011), in depths ranging from 34 to 54 metres between the latitudes of 29.00° and 29.75°. Any lobsters that became entangled in the demersal gill net immediately adjacent to an entangled fish or shark were not retained for dietary studies as they may have been attracted to and feeding on the fish prior to entanglement. The lobsters were kept frozen until the foreguts were removed in the laboratory and stored in 70% ethanol. The same biological information was recorded for each lobster as in section 6.1.3 including sex and carapace length.

Stomach analysis

The foregut was removed from the ethanol solution then blotted dry with absorbent paper. Percentage fullness of the gut was estimated visually (Edgar 1990, Jernakoff *et al.* 1993, MacArthur *et al.* 2011) and an index of gut fullness (GFI) was also determined using the following equation from Mayfield *et al.* (2000):

$$GFI = \frac{\text{total stomach weight - stomach membrane weight x 100}}{\text{total stomach weight}}$$

Stomachs that had either an estimated gut fullness or a GFI < 10% were omitted from analyses to avoid biases associated with items that may have a higher retention time in the gut and also to maintain consistency with previous diet studies on this species (Joll and Phillips 1984, Jernakoff *et al.* 1993, Waddington *et al.* 2008, MacArthur *et al.* 2011). Twenty-nine samples (15 summer; 14 autumn) remained for analysis.

The contents of the stomach were flushed with 70% ethanol into a 9.5 cm petri dish and the evacuated stomach membrane was blotted dry and weighed to determine the stomach membrane weight. The contents of the stomach were spread evenly over the dish and examined under a Nikon SMZ800 dissecting microscope (1x − 6.3x magnification) to identify prey items. The dish was placed onto grid paper and the percentage contribution was estimated by determining the proportion of the total area covered by each prey item (MacArthur *et al.* 2011).

Molecular techniques

In addition to the traditional stomach content analyses outlined above, dietary samples from six individuals were also subjected to next generation sequencing (NGS). Lobster stomach contents were homogenised and subsampled for DNA extraction. The ethanol in each sample was gently removed to a second vial and approximately 1 ml of homogenised gut contents was transferred to a 2 ml eppendorf tube. Ethanol was poured back in each vial. All samples were stored frozen at -20 °C. Extractions were performed using QIAamp DNA Stool Mini Kit (QIAGEN) as per manufacturer's instructions. DNA was eluted in 50 µL of AE buffer and DNA extracts stored at -20°C until further analysis. Dilutions of 1:10 and 1:100 were made to DNA extracts using PCR-grade water (Fisher Biotec) for subsequent real-time PCR amplification.

Table 6.6. The identity, target and source of primers evaluated during preliminary molecular dietary studies of western rock lobster.

Taxonomic Group	Primers		Amplicon size (bp)	Target Region	Target taxon	Reference
	Forward	Reverse				
Plants	psbA	trnH	300 – 450	plastid trnH-psbA intergenic spacer	seagrass	Budarf et al. 2011
	p23SrV_f1	P23SnewR		23S rDNA plastid marker	eukaryotic algae and Cyanobacteria	Sherwood & Presting 2007 Clarkston & Saunders 2010
	rbcl fw	rbcl rev	183	chloroplast gene encoding the large subunit of ribulose-bisphosphate carboxylase (rbcL)	plants	Poinar et al. 1998
	trnLc	trnLh		trnL (UAA) Group 1 intron in chloropast DNA	plants	Taberlet et al. 2007
Animals	ZBJ-ArtF1c	ZBJ-ArtR2c	100-300	COI barcode	arthropods	Zeale et al. 2010
	LCO1490	HC02198	710	COI barcode	invertebrates	Folmer et al. 1994
	Uni-MinibarF1	Uni-MinibarR1	120-150	COI barcode	invertebrates	Meusnier et al. 2008
	LCO1490	Uni-MinibarR1	120-150	COI barcode	invertebrates	Folmer et al. 1994 Meusnier et al. 2008
	16S1F degenerate	16S2R degenerate	180–270	mitochondrial 16S rDNA	fish	Deagle et al. 2007

Each extract was screened using real-time PCR and series of primer pairs available at the Murdoch University Ancient DNA research laboratory and from the literature (Table 6.6). Each extract was amplified at neat, 1:10 and 1:100 dilutions using the ABI Step One Real Time PCR machine. Each reaction was made up to 25 µL, containing 1x PCR Gold Buffer, 2 mM MgCl2, 0.4 mg BSA, 0.25 mM dNTPs, 0.4 mM of each primer, 0.6 µL SybrGreen (1:2000), 0.2 units of AmpliTaq Gold (Applied Biosystems), 14.45 µL of water and 2 µL DNA. Reaction conditions consisted of initial heat denaturation at 95°C for 5 min, followed by 40 cycles of 95°C for 30s, 50 to 54°C for 30s (annealing temperatures varied according to references for each primer set) and 72°C for 45s, followed by a final extension at 72°C for 10 min and a 1°C melt curve to assist in the identification of primer dimer and nonspecific amplification. From all the primer pairs tested only the rbcl, trnL, ZBJ-Art, LCO1490/Uni-MinibarR1 and 16S primer sets generated amplicons. For each sample, the DNA dilution that generated the best compromise of amplicon DNA yield/PCR inhibition was assigned a unique tagged primer set. Fusion tagged real-time PCR was carried out in 25 µL reactions containing 1x PCR Gold Buffer, 2 mM MgCl$_2$, 0.4 mg BSA, 0.25 mM of each dNTP, 0.4 mM of each primer, 0.2 units of AmpliTaq Gold (Applied Biosystems), 14.45 µL of water and 2 µL of DNA. The cycling conditions consisted of an initial heat denaturation at 95°C for 5 min, followed by 50 cycles of 95°C for 30s, 54°C for 30s and 72°C for 45s, followed by a final extension at 72°C for 10 min.

Only the tagged primer sets rbcl, LCO1490/Uni-MinibarR1 and 16S generated amplicon quantities suitable for Next-Generation Sequencing. Reactions were conducted in duplicate and amplicons pooled together to minimise the effects of PCR stochasticity. The resultant pooled amplicons were purified using Agencourt AMPure XP PCR Purification Kit (Beckman

Coulter Genomics, NSW, Australia), and eluted in 40 µL of water. Purified amplicons were electrophoresed on 2% agarose gel and amplicons were pooled in approximately equimolar ratios based on band intensity. All procedures involved in the setup of the sequencing run (emulsion PCR and bead recovery), including the sequencing run itself, were carried out according to the Roche GS FLX Junior (Roche) protocols for amplicon sequencing. (http://www.454.com).

6.3.2   Benthic assemblages

6.3.2.1   Benthic assemblage sampling

Surveys to collect qualitative data on the composition of benthic assemblages in deep water were conducted with an autonomous underwater vehicle (AUV) '*Sirius*' (Figure 6.7). This AUV is operated by the Australian Centre for Field Robotics (ACFR, University of Sydney) and supported by the Integrated Marine Observing System (IMOS). *Sirius* is equipped with a full suite of oceanographic instruments, including multibeam sonar, depth, conductivity and temperature sensors, Doppler Velocity Log (DVL) including a compass with integrated roll and pitch sensors, Ultra Short Baseline Acoustic Positioning System (USBL), forward looking obstacle avoidance sonar and other sensors to measure various biological variables (see Smale *et al.* 2012 for a more thorough description).

A high resolution stereo camera pair and strobes enable the collection of benthic imagery with high positional accuracy. The AUV works un-tethered, collecting data along a precise flight path programmed prior to deployment. A series of preliminary surveys were conducted with the AUV at Leeman in April 2011, primarily to test the appropriateness of this technology. The ability of the AUV to revisit precise locations in deep water allows it to undertake spatially repeatable surveys which can facilitate the detection of change over time (Smale *et al.* 2012) Thus a wider survey was initiated in April 2012 to collect benthic data at Jurien and Leeman. At both locations, two transects were surveyed at northern (N), middle (M) and southern (S) sites giving a total of 12 transects retained for analyses in 2012 (Figures 6.8 and 6.9). Note that the middle and southern sites at Leeman (LM and LS) were located inside the area closed to lobster fishing. The location of sites was selected based on the substrate and biota maps produced in Chapter 6 and aimed to target moderate to high relief reef.
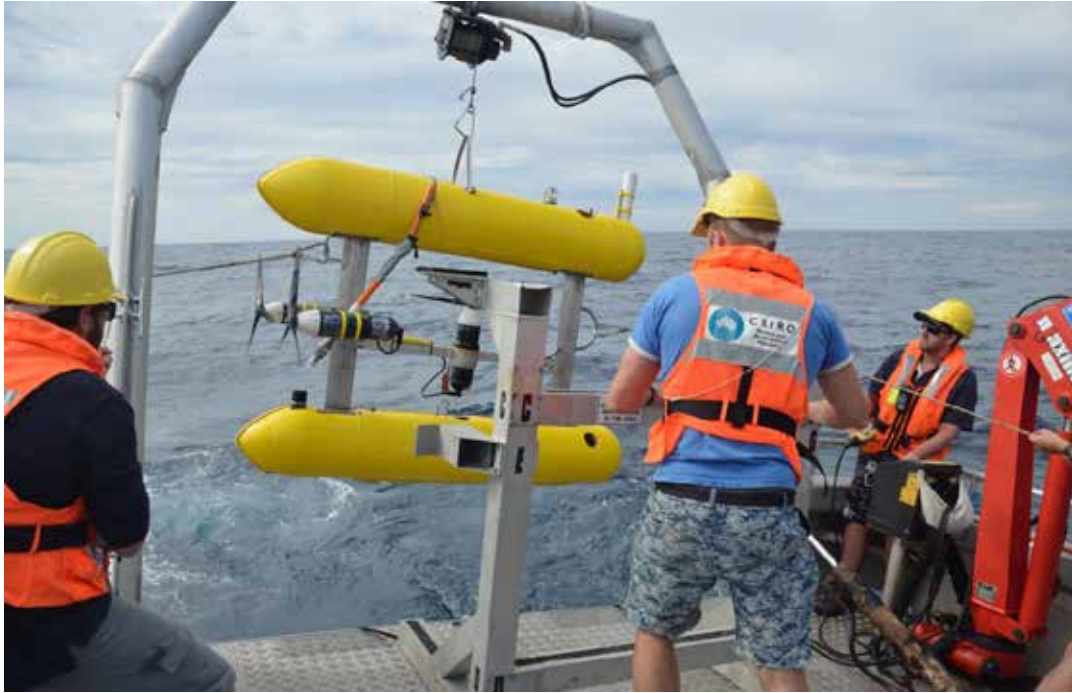
Figure 6.7.   Autonomous underwater vehicle (AUV), Sirius, being retrieved on to the RV Linnaeus following a successful deployment off Jurien.

Each 'transect' consisted of a series of parallel overlapping 25m long mini-transects which, when combined, sampled a 625 m^2 area of seabed in a grid formation (i.e. 25 x 25 m). In addition, each transect terminated in a single straight transect or 'tail' of *ca* 100m which was orientated to provide additional information about the landscape surrounding the fully mapped grid. Along each transect, the AUV captured overlapping geo-referenced stereo images of the benthos, as well as bathymetric data at 2 resolutions and physiochemical data. Note that only the data for the 25 x 25 m grids has been included in the analyses described below. Information gathered from the transect 'tails' and an additional irregularly shaped transect flown at most sites were primarily analysed separately and the information used to further validate the accuracy of habitat models (see further development in Chapter 9).
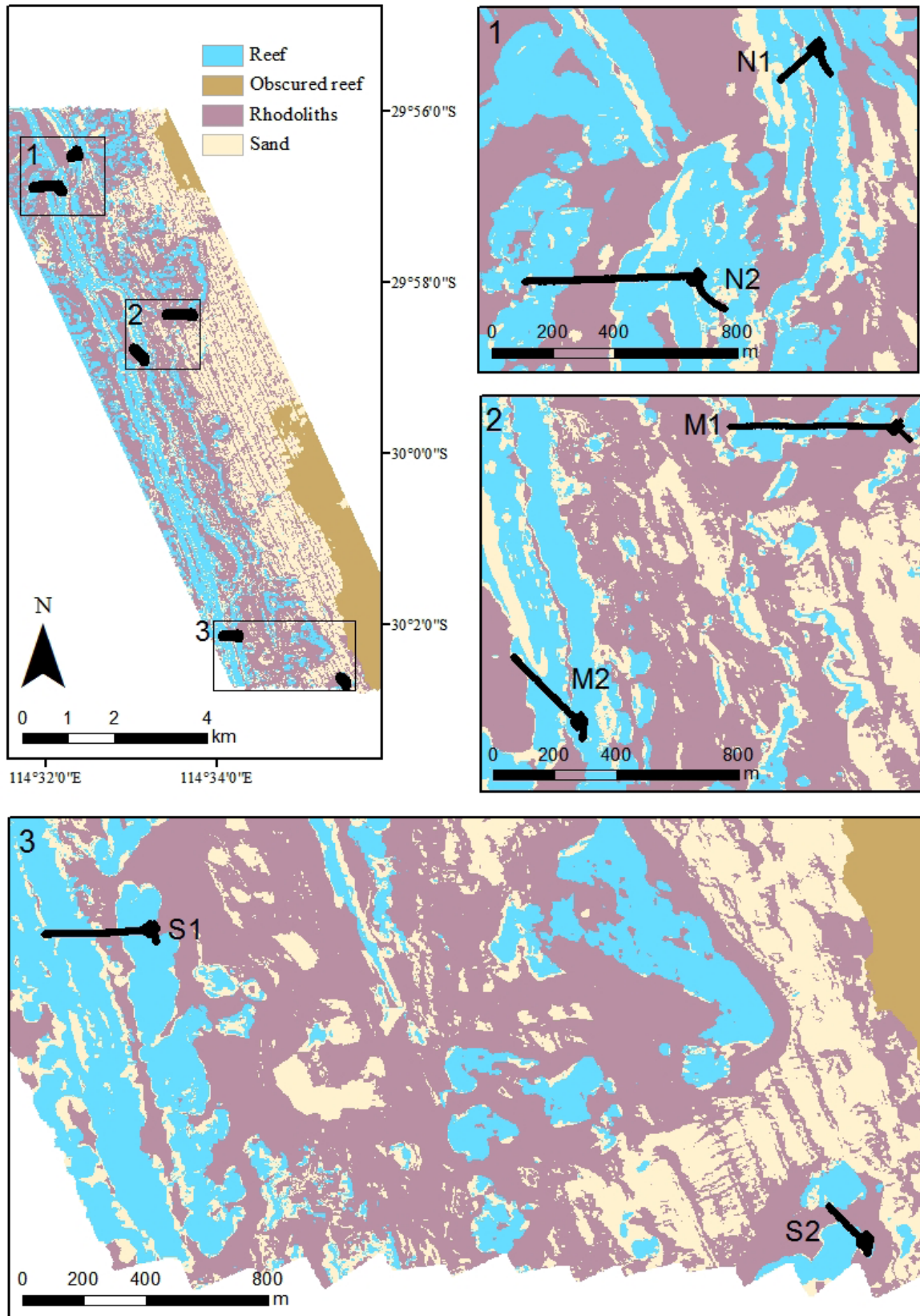
Figure 6.8. Location of transects (including 25m x 25m grids) surveyed at Leeman by the AUV in April 2012.
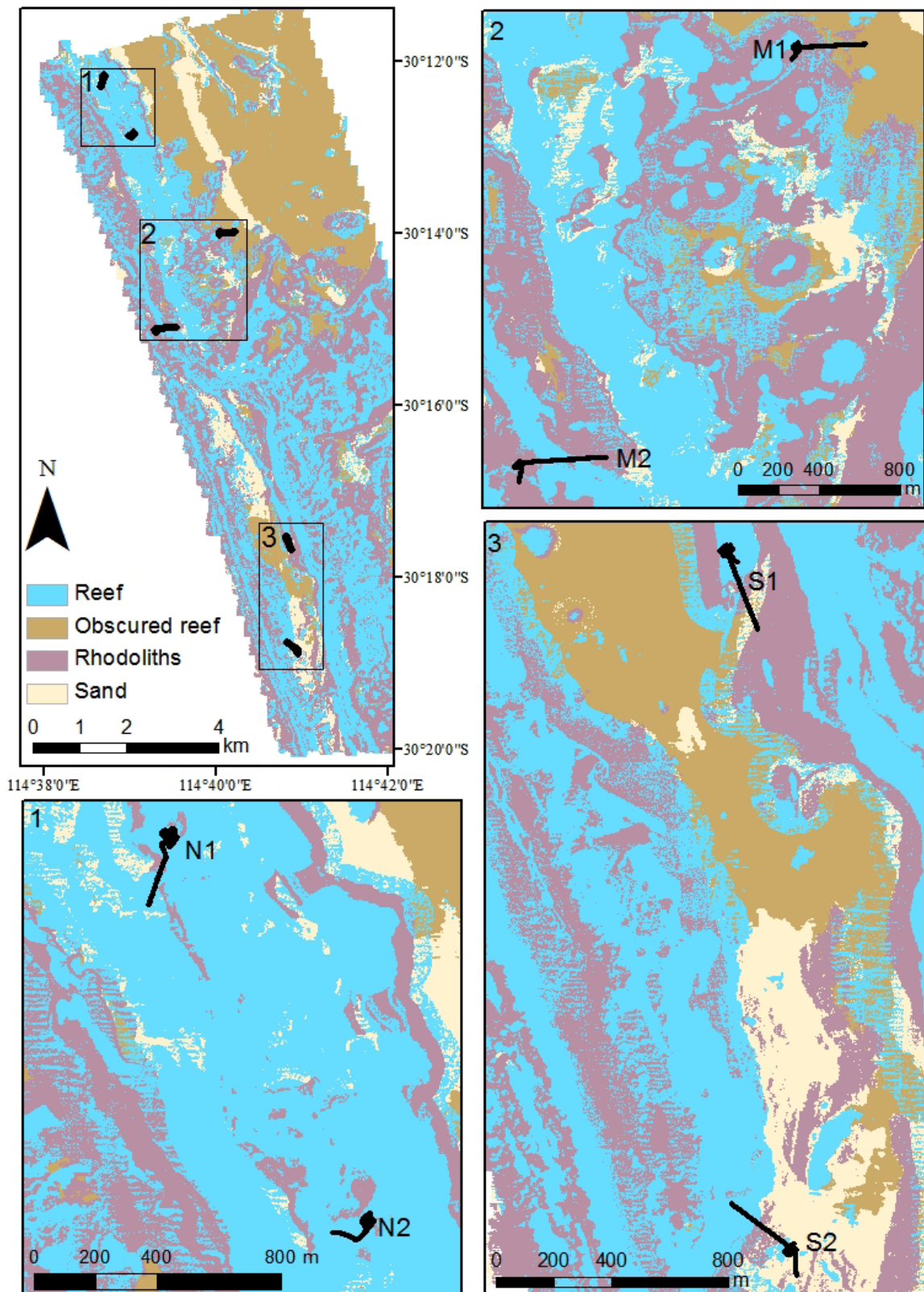
Figure 6.9. Location of transects (including 25m x 25m grids) surveyed at Jurien by the AUV in April 2012.

6.3.2.2   Analysis of benthic assemblage data

Over 1000 geo-referenced stereo image pairs were collected at each transect grid. Images were stitched together to generate composite 'meshes' of the entire 25 x 25 m patch of seabed. These meshes provided a complete overview of each reference site, and were used to assess broad-scale ecological structure, such as kelp canopy cover. For this study, individual high-resolution images (each capturing approximately 1.5 x 1.0 m of seabed) were subsampled from each grid to quantify assemblage structure using image analysis techniques. Subsamples of 20 non-overlapping images were selected to maximise the spatial coverage at each grid.

For benthic analysis, initially each image was subjectively scored for a number of physical and biological habitat attributes including: rugosity (flat, low, moderate, high); substrate (sand, gravel, cobble, rock, reef); form (none, ripple, wave, low slope, moderate slope, steep); sponge height (none, low, moderate, high) and algal canopy height (none, low, moderate, high). In addition the number of kelp recruits and mobile invertebrates were counted in each image.

To investigate benthic community composition, 50 random points were digitally overlaid onto each image and the number of points covering each benthic category was counted using CPCe software (see Kohler and Gill 2006). Numbers were expanded to give a proxy of percent cover for each category. The benthic categories were determined a priori based on those used in previous research (Smale *et al.* 2010, 2012).

6.3.3   Fish indicators

6.3.3.1   Fish community sampling

Fish communities in deep water at Leeman and Jurien were sampled with stereo baited underwater videos (stereo BRUVs). Sampling was conducted on a single occasion in Autumn 2011 shortly after the closed area came into effect. The sampling sites at the two locations Leeman (closed) and Jurien (fished) were selected to be as comparable as possible in terms of habitat and depth. Sites were selected on or near rugose-ground and reef habitat associated with the old, submerged coastline feature adjacent to the coastline, and all sites were within a depth range of 47-58 m. From the two locations sampled, demersal fish data from 233 stereo BRUV deployments were used in the analyses, with between 4-5 replicate deployments per site.

The stereo BRUV method used in the present study is the same as that used by Langlois *et al.* (2010). Detailed information on the design and photogrammetric specifics are presented in Harvey and Shortis (1996). Stereo BRUV systems were comprised of two Sony CX12 full high-definition (1920x1080) video cameras mounted 0.7 m apart on a base bar inwardly converged at seven degrees to gain an optimized field with stereo-coverage from 0.5 m to the maximum visibility. For this study we only include fish within six meters of the cameras, which was the minimum visibility recorded. A synchronising diode and bait basket was positioned in front of the cameras. Up to ten systems were deployed simultaneously from a commercial lobster boat and left to film on the sea floor for a period of one hour. With multiple systems in use, a single stereo BRUV could be deployed at one site, followed by another at a second site and so on, maximising sampling efficiency. Previous research in temperate regions has found that >36 minutes is required to obtain measures of the majority of fish species, and that 60 minutes is advisable to include high risk fished species (Watson *et al.* 2005). Systems were baited with 800 grams of pilchards (*Sardinops sagax*) in a plastic-coated wire mesh basket, suspended 1.2 m in front of the two cameras. The pilchards were crushed to maximise dispersal of their oil. Adjacent replicate stereo BRUV deployments were separated by at least 250 m to avoid overlap

of bait plumes and reduce the likelihood of fish moving between stereo BRUVs within the sampling period.

The use of bait as an attractant has been suggested to cause biases in the representation of the fish fauna samples. However, analyses have found that bait attracts greater numbers of predatory and scavenging species without decreasing the abundances of herbivorous or omnivorous fishes observed (Harvey *et al.* 2007). Baited video was also found to have greater statistical power to detect spatial and temporal changes in the structure of fish assemblages and the relative abundances of individual species compared to unbaited video (Harvey *et al.* 2007). The stereo BRUV method has been shown to be comparable to, and in some case more efficient and cost effective than other visual methods, such as diver-based stereo-video census methods (Langlois *et al.* 2010). Stereo BRUV is also less susceptible to inter observer variability than traditional visual census methods, as the images can be repeatedly reviewed and provide a permanent record.

6.3.3.2   Habitat classification

The fine scale maps of substrate and biota were used to choose locations for BRUV deployments so that sampling could be balanced across key habitats. The classification of final habitats used in analyses was amended, if required, based on the habitat observed in each camera deployment. Habitats were classified into three benthic categories: 1) macroalgae; 2) rhodoliths or 3) sessile invertebrate, where this habitat type made up over 80% of the field of view. An estimate of the field of view was made for each deployment. Sites where benthos obscured over 35% of the view were removed from the analysis. Water visibility was consistently good at all sites and greater than 6 m.

6.3.3.3   Video analysis

Stereo BRUV videos were analysed with the help of the program EventMeasure (Stereo) (www. seagis.com.au). Data were recorded capturing the timing of events and reference images of the seafloor and fish in the field of view and the maximum number of any one species seen at one time during the recording (MaxN, Priede *et al.* 1994, Cappo *et al.* 2003). Estimates of MaxN avoid double counting of individuals that may re-enter the field of view. They are a conservative estimate as on occasion only a portion of the total number of individuals of a species in the area may be present at any one time. The range of each fish from the camera system was measured and standardised using EventMeasure (Stereo) and all individuals beyond a 7 m limit were excluded from species richness and relative abundance analyses.

6.3.3.4   Analysis of fish data

Multivariate analyses were done using a permutational analysis of variance (PERMANOVA with 4999 permutations) on a Modified Gower Log 2 similarity matrix constructed from the fish assemblage data using the PERMANOVA + add on (Anderson *et al.* 2008) to PRIMER v.6 (Plymouth Routines in Multivariate Ecology Research; PRIMER-E 2009). The data was analysed according to the following three factor design with the continuous variable Depth as a covariate:

1) Area (two levels, fixed: fished versus closed)

2) Habitat (three levels, fixed: macroalgae, rhodolith and sessile invertebrate)

3) Site (random, nested in area and habitat)

Where main effects or interactions were significant, a posteriori pairwise comparisons were explored (Anderson 2008). A principle coordinate analysis (PCO) was used to illustrate the unconstrained grouping of sites and to examine the overall patterns of variation in the assemblages between the two locations. To investigate the significant factors evident from the PERMANOVA a constrained canonical analysis of principle coordinates (CAP) was used. A CAP identifies an axis through the multivariate data cloud that is best at separating a priori groups and thus illustrates real differences that cannot be seen in an unconstrained PCO ordination (Anderson *et al.* 2008). Correlation vectors were overlaid on the CAP plot and used to identify the species that were correlated to these a priori differences in the assemblage composition, even in the presence of potentially high variation in the data cloud that may be due to other factors (Anderson and Robinson 2003, Anderson and Willis 2003). The number of PCO axes (m) used to construct the CAP was chosen by plotting the residual sum of squares and choosing m at the first large drop in relation to the other values (Anderson and Robinson 2003).

All of the species found to correlate strongly ($r2 > 0.4$) with the CAP axes were analysed using univariate permutational analysis of variance (herein referred to as ANOVA) using the PRIMER v6 computer program (Clarke and Gorley 2006) with the PERMANOVA+ add-on (using 4999 permutations, Anderson *et al.* 2008) to test the hypotheses stated above. All ANOVA analyses were preceded by Levene's test for homogeneity of variance (Anderson *et al.* 2008). Where the test showed significant heterogeneity, variables were transformed to $x' = \ln(x + 1)$. Analyses of variance were followed by a posteriori Student-Newman-Keuls pair-wise tests (SNK tests) on terms of the model found to be significant with $P < 0.05$. Stereo BRUV estimates of all metrics are relative given the variable distance of attraction of bait, as discussed above, and therefore were not expressed per meter but were assumed comparable across all areas.

6.3.3.5 Cost-benefit analysis

The logistics of a possible monitoring program for the indicator species were investigated using a cost–benefit optimization, as described by Underwood (1981) and as previously employed for stereo BRUV data by Langlois *et al.* (2010). In this case, the percentage change detectable was estimated for increasing numbers of sites sampled per closed and fished location for each habitat sampled.

7.0   Results and Discussion

7.1   Identification and assessment of suitable unfished reference areas to exclude rock lobster fishing in deep water

7.1.1   Initial site nomination

Six sites were initially nominated by the closed area working group as potential areas that could be closed to lobster fishing (see Table 6.1). After careful consideration of the advantages and disadvantages of each site nominated, two sites were short-listed, i.e. the bottom of the Abrolhos zone and the 30°S latitude line near Leeman (Figure 6.1). Validation of benthic habitats was undertaken for both sites. As benthic habitats appeared most suitable at the Leeman site (see section 7.1.2), information was then gathered on lobster demographics at that site (see section 7.1.3).

7.1.2   Habitat surveys

The benthic habitats in more than half of the transects surveyed near Leeman contained substantial amounts of mixed assemblages with *Ecklonia* sp. (Figure 7.1a). Mixed assemblages dominated by sessile invertebrates were not prevalent. Those transects with substantial areas of sand were located to the east of the study area (Figure 7.2). The composition of benthic habitats at Leeman was similar to those found at the nearby Jurien lobster monitoring (IBSS) sites, Jurien North and Jurien South (Figure 7.1c)

In contrast, all of the transects surveyed at the Abrolhos had high proportions of sessile invertebrate dominated mixed assemblage (primarily sponges) (Figure 7.1b). Only three transects had significant amounts of mixed assemblage with *Ecklonia* sp. and these transects were restricted spatially (Figure 7.3).

The prevalence of reef habitats dominated by *Ecklonia* sp. rather than corals and sponges suggested that the Leeman site contained habitats more representative of those
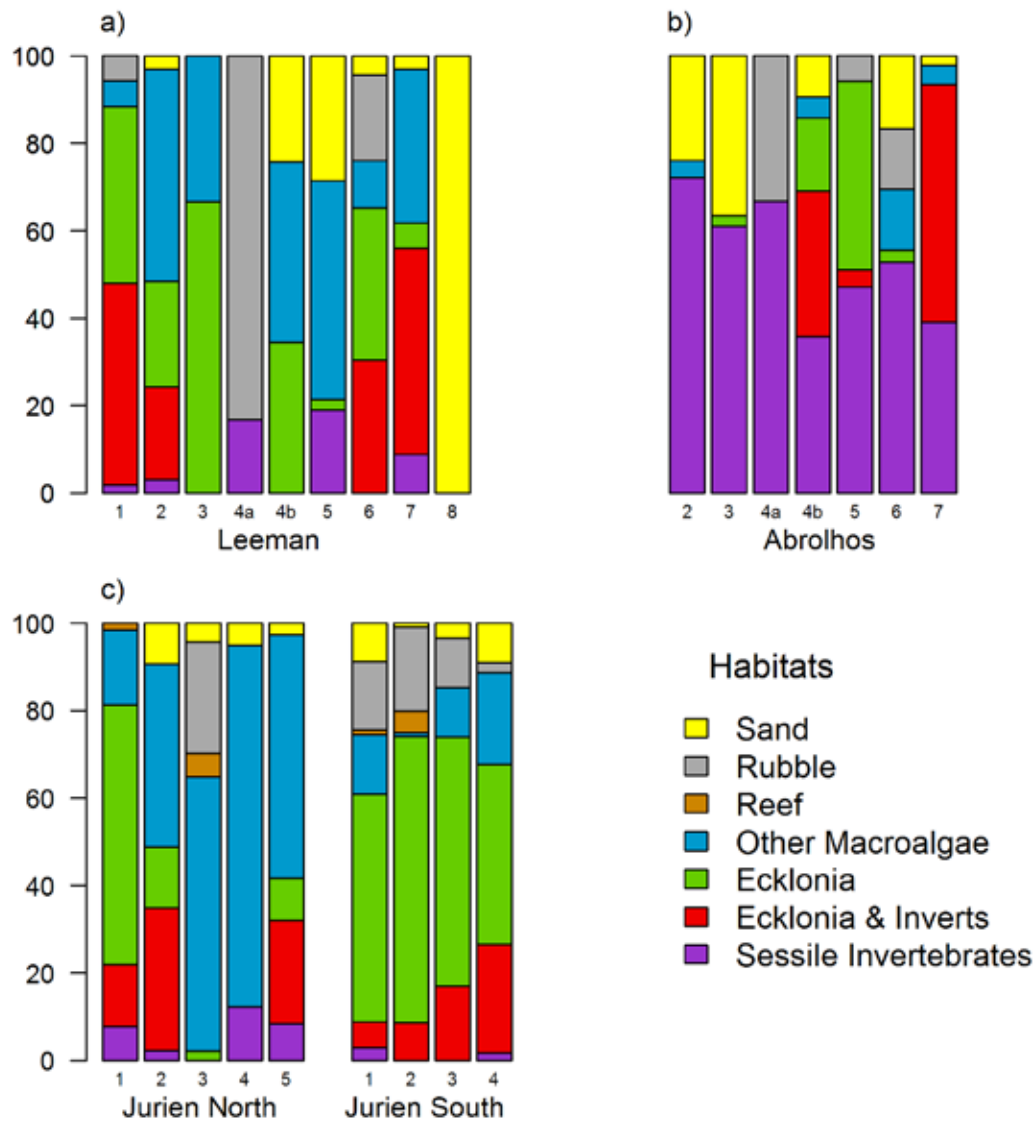
Figure 7.1. Contribution of major benthic habitats types to towed video transects at the two locations proposed as potential research areas to close to lobster fishing, i.e. Leeman (30°S latitude boundary) and the Abrolhos (south eastern corner of A zone) and Jurien (north and south). Habitat information for the Jurien site was collected during FRDC 2004/049. Numbers above bars at Leeman and Abrolhos refer to transects represented spatially in Figures 7.2 and 7.3.
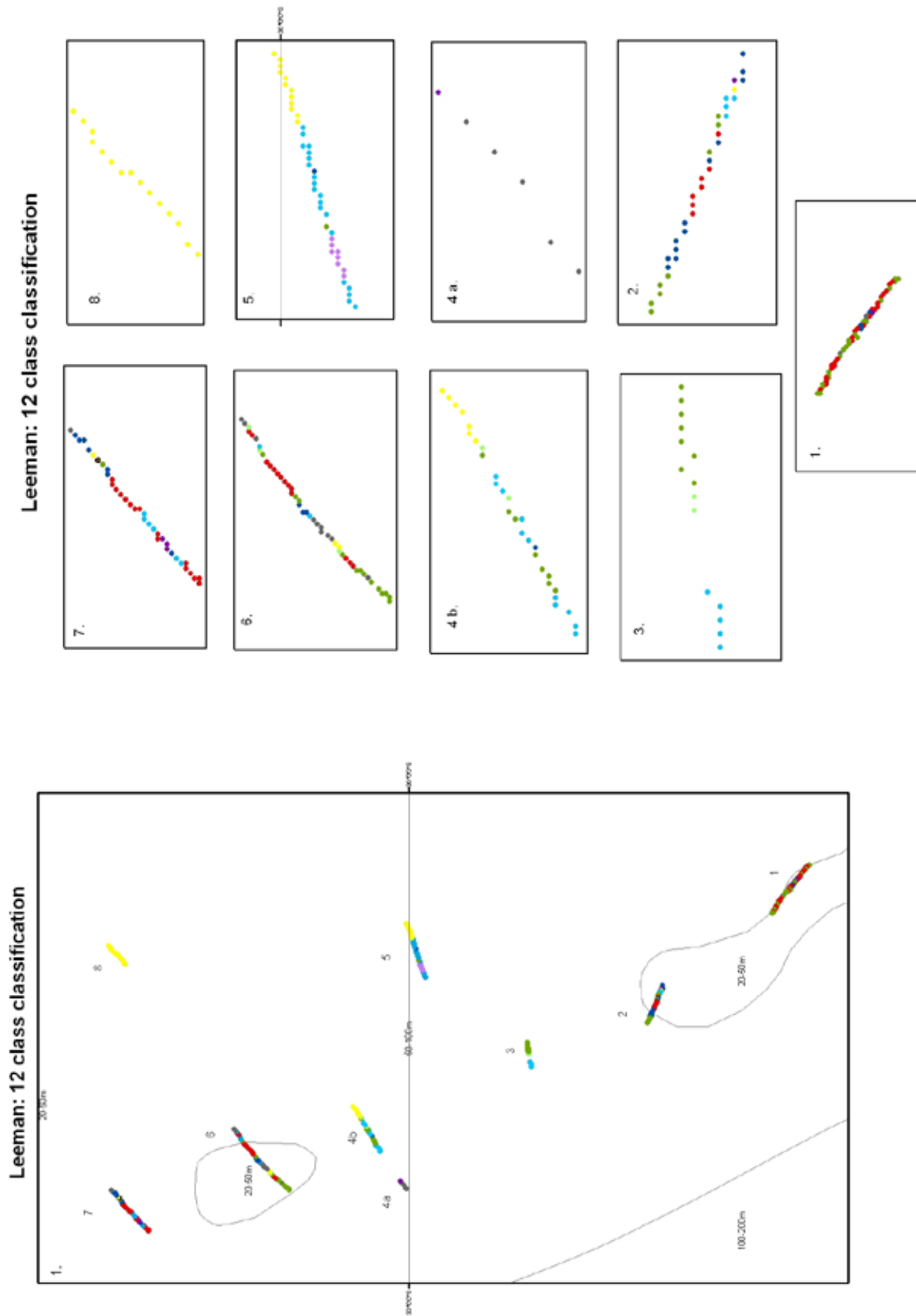
Figure 7.2. Spatial distribution of transects at Leeman (30°S latitude boundary) where the composition of benthic habitats were validated using towed video. The large box is an enlargement of the area indicated in Figure 6.1. Coloured points refer to habitat classifications for individual video frames. Colours of major habitat types and transect numbers are consistent with Figure 7.1.

Dongara: 12 class classification
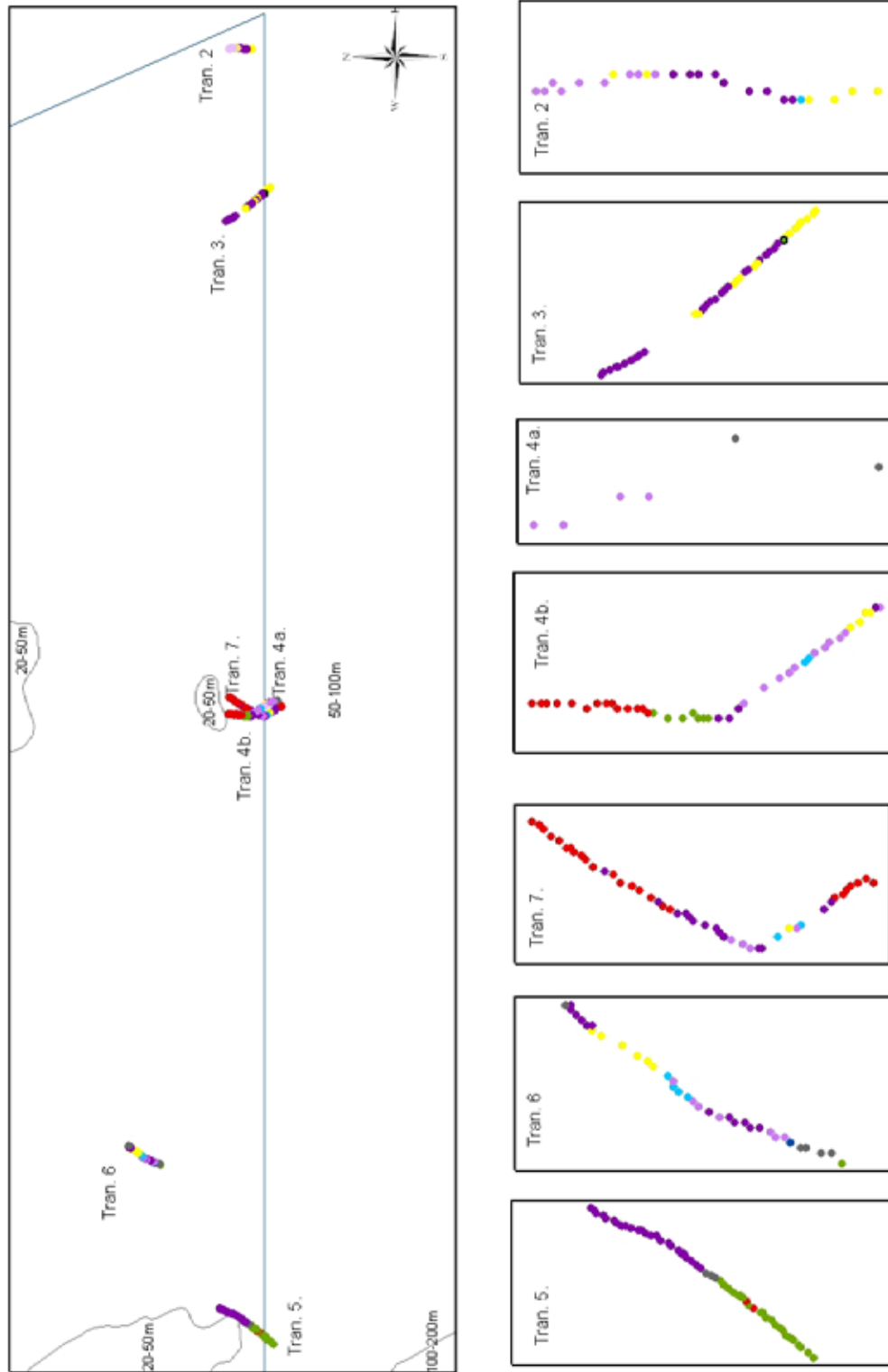
Figure 7.3. Spatial distribution of transects at Abrolhos (south eastern corner of A zone) where the composition of benthic habitats were validated using towed video. The large box is an enlargement of the area indicated in Figure 6.1. Coloured points refer to habitat classifications for individual video frames. Colours of major habitat types and transect numbers are consistent with Figure 7.1.

important to *Panulirus cygnus* (Bellchambers 2010, Bellchambers *et al.* 2010). The comparable habitats between Leeman and the Jurien IBSS sites and their close proximity made Leeman a favourable site, as Jurien could be sampled as a "fished" control site. High resolution habitat mapping and long-term lobster demographic data are available for Jurien (Bellchambers 2010, Bellchambers *et al.* 2010, Bellchambers *et al.* 2013). Surveys were then undertaken to access the representativeness of lobster demographics at the Leeman site.

7.1.3   Lobster demographics

Rather than being randomly distributed across the survey area, the greatest catches of western rock lobster at Leeman in 2008 were concentrated into a few smaller areas, mainly between the 50 and 100 m depth contours (Figure 7.4). The spatial distribution of lobster catches was likely influenced by benthic habitats. While the mean catch rate across all pots at Leeman was 2.16 lobsters pot^{-1}, catches were higher in the western areas, e.g. *ca.* 4.2 lobsters pot^{-1} in sub region A (Figure 7.5). The mean total catch rates at Jurien North and South in 2008 were 2.44 and 2.79 lobsters pot^{-1}, respectively. The mean catch rates from *Ecklonia* dominated lines at the two Jurien sites were comparable with catch rates from the western sub-regions (A-C) at Leeman (Figure 7.5).

In 2009, catches at Leeman were more concentrated spatially (Figure 7.4). Catch rates were lower at all sites sampled, i.e. 1.57, 1.6 and 2.41 lobsters pot^{-1} at Leeman, Jurien North and Jurien South, respectively. The mean catch rates at all sites in 2008 and 2009 were well within the bounds of historical survey data for the Jurien sites, which ranged from 0.86 lobsters pot^{-1} at Jurien North in 1995 to 5.83 lobsters pot^{-1} at Jurien South in 2004 (Figure 7.6).

The size compositions of lobsters in 2008 were comparable between Leeman and both Jurien sites with the majority of lobsters between 70 and 80 mm carapace length (CL) (Figure 7.7a). There were relatively smaller numbers of sub-legal size lobsters sampled at all sites in 2009 and the majority of lobsters were between 76 and 86 mm CL (Figure 7.7a).

7.1.4   Closed area implementation

After reviewing the recommendations of the closed area working group, and information provided by the DoF on the habitat and lobster demographics in the nominated area, the Scientific Advisory Group (SAG) was confident that the length compositions and abundance of lobsters in the proposed area are representative of the fishery and comparable to those found in the nearby Jurien IBSS sites (Jurien North and Jurien South). As the DoF has long term catch data and detailed habitat information for the Jurien IBSS sites, which were the focus area for studies during FRDC 2004/049, these sites have high potential as "fished" control areas.

The SAG indicated that the final size and boundary orientation of the closed area will have a significant impact on the potential of the closed area in offering protection to rock lobsters and ultimately toward the success of the project. They considered that the key to size is adequate coverage of important habitat types (i.e. *Ecklonia* dominated mixed assemblage) and adequate catches of lobster (including different size classes). The range of lobster movements also controls the size of the area required. Similarly, edge effects arising from the placement of reserve boundaries on or close to continuous habitats will act to reduce the effective size of any potential area. Based on the information provided, the committee suggested that an area approximately 8x3 nm would be required.

In 2010, based on the recommendations of both the CAWG and SAG further negotiations between DoF and RLIAC resulted in the closure of an area (6 nm x 2 nm, see Figure 7.8) on

the 30°S latitude line demarcating the boundary between B and C zones. The total area of the closure was *ca.* 3900 ha, with the long boundaries of the closure orientated parallel with the bathymetry to cover the greatest amount of suitable lobster habitats. The area was gazetted and closed to lobster fishing on the 15th March 2011 for a five year period (to be reviewed after 5 years).
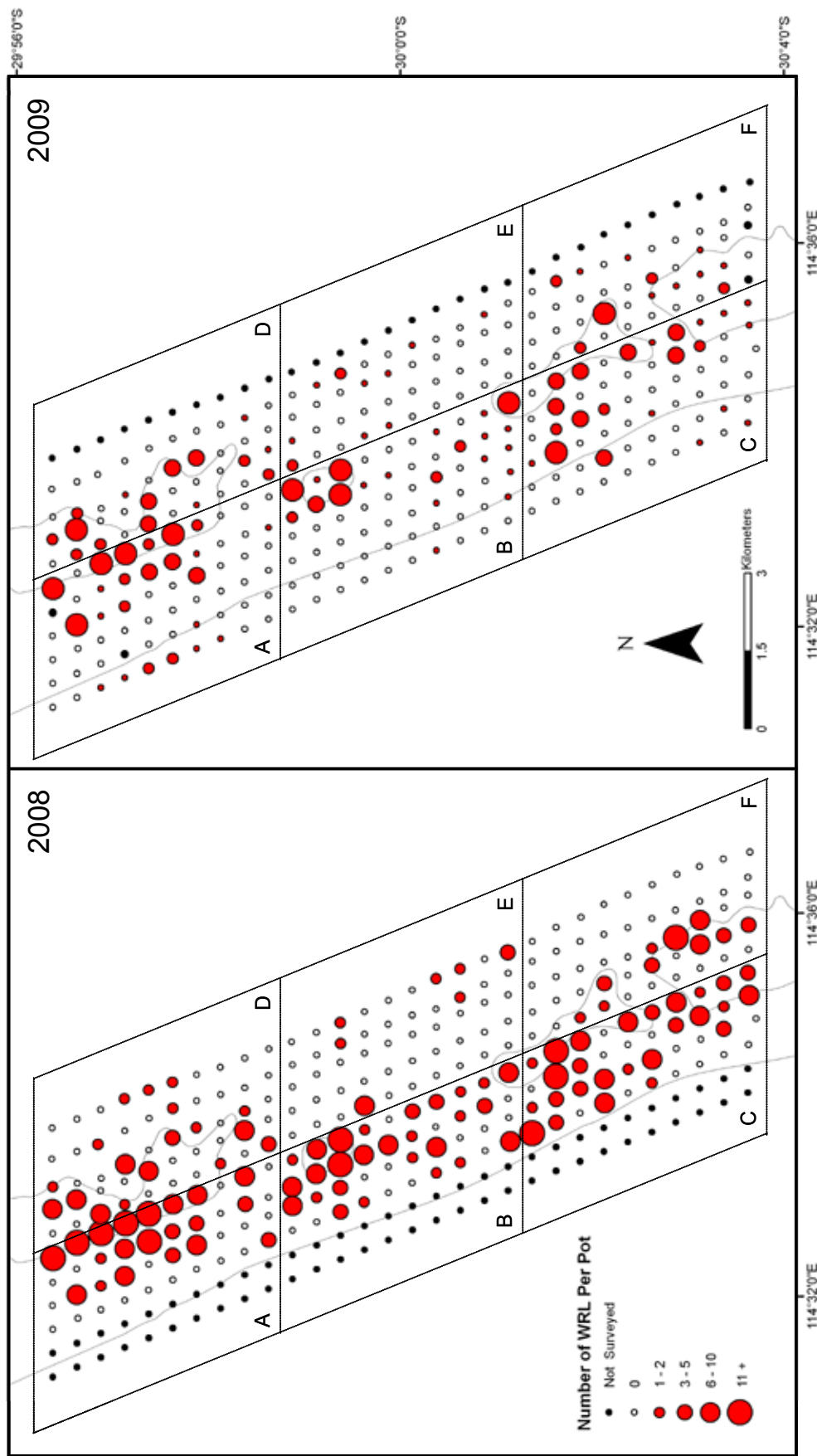
Figure 7.4 Spatial distribution of western rock lobster catches (total lobsters pot⁻¹) in 2008 and 2009 at the Leeman research area adjacent to the 30°S latitude line.
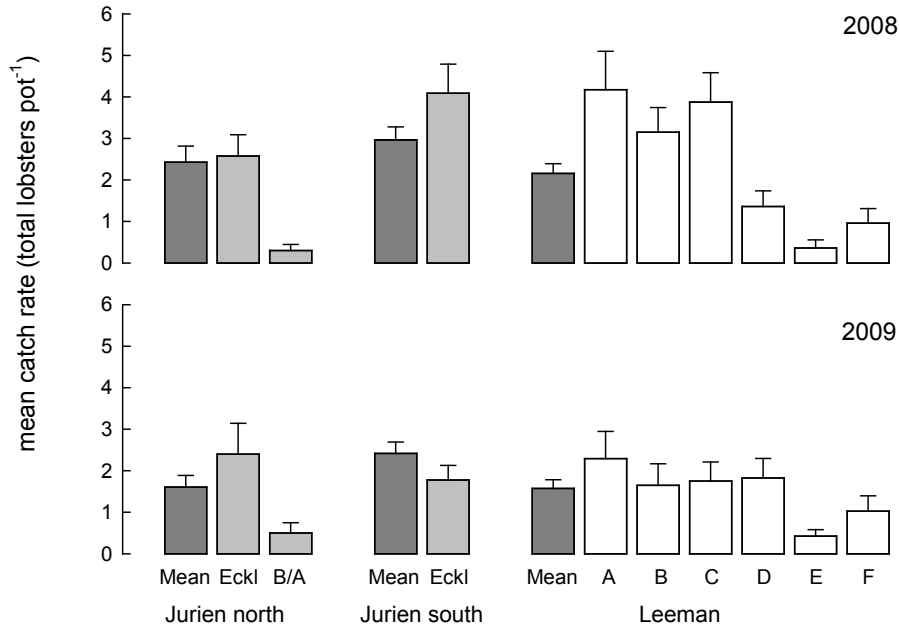
Figure.7.5. Catch rates of western rock lobsters (total lobsters pot^{-1}) at Leeman (30°S latitude line) and at Jurien (north and south) during 2008 and 2009. Dark grey bars show overall means for each location. Catch rates by major habitats are given for Jurien (light grey). Habitats: Eckl - mixed assemblage dominated by *Ecklonia*; B/A - mixed assemblage dominated by other brown algae. Catch rates at Leeman are shown for the six separate sub-regions (open bars) illustrated in Figure 6.5.
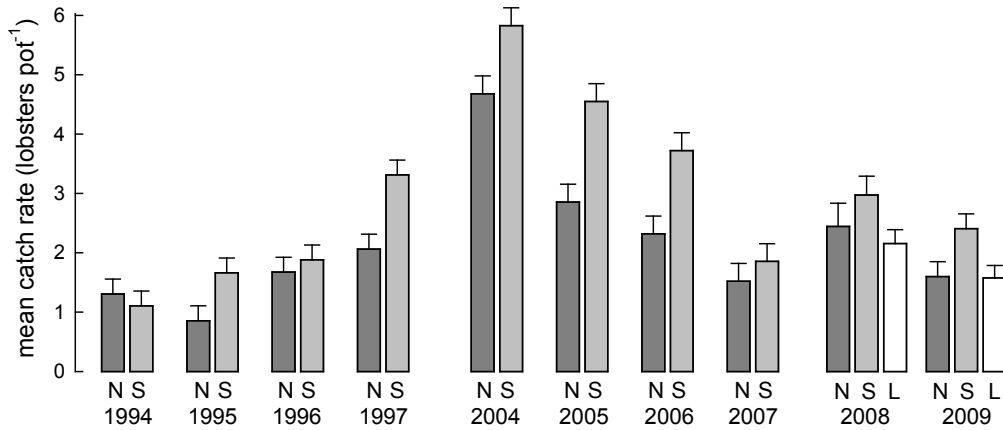
Figure. 7.6. Mean catch rates of western rock lobsters (total lobsters pot^{-1}) at the Jurien north (dark grey) and south (light grey) areas for the periods 1994-1997 and 2004-2009. Mean catch rates at the Leeman research area in 2008 and 2009 are also shown (open bars).
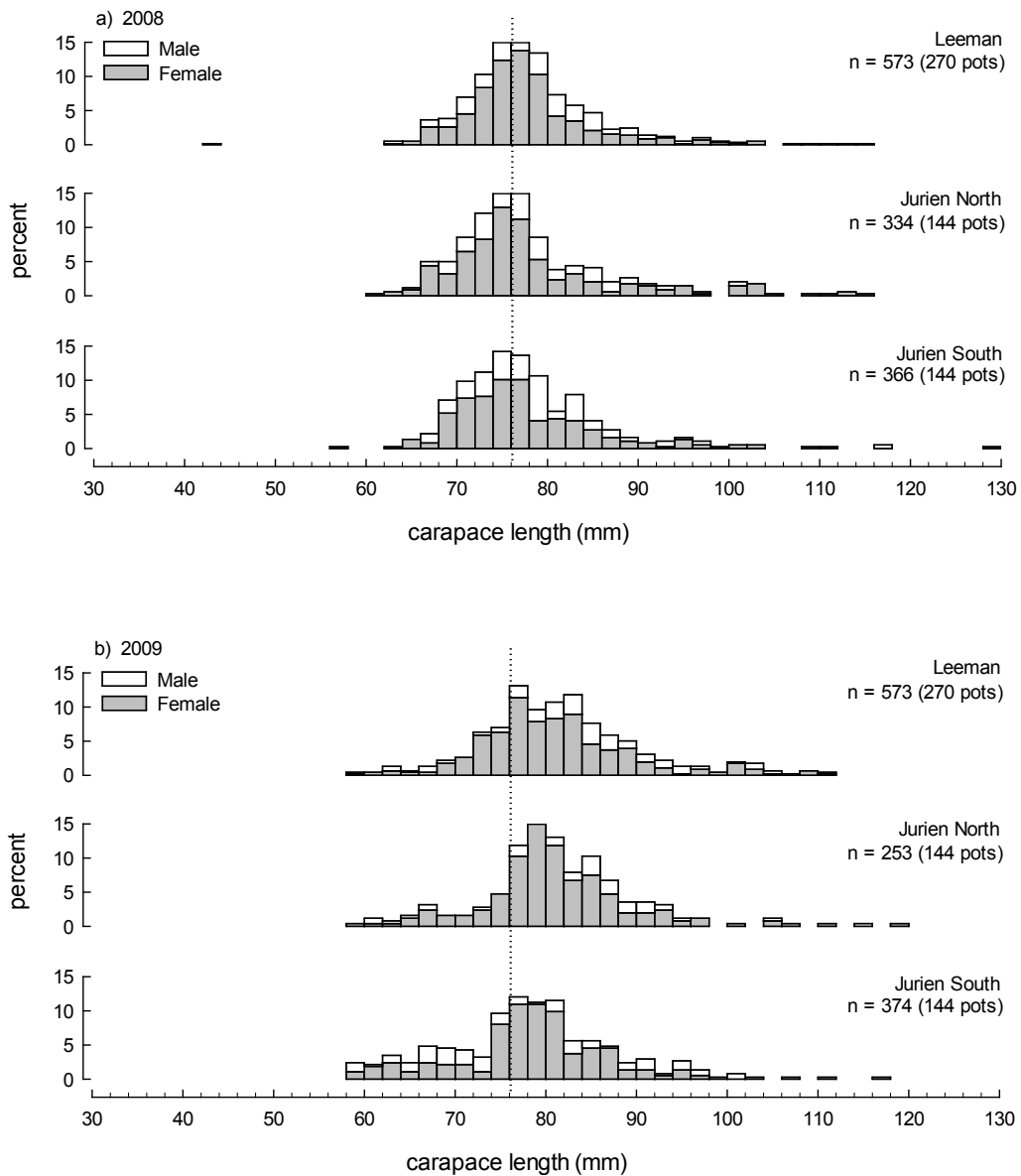
Figure. 7.7. Size distribution of male (open) and female (shaded) western rock lobsters caught at Leeman (30°S latitude line) and at Jurien (north and south) during (a) 2008 and (b) 2009.
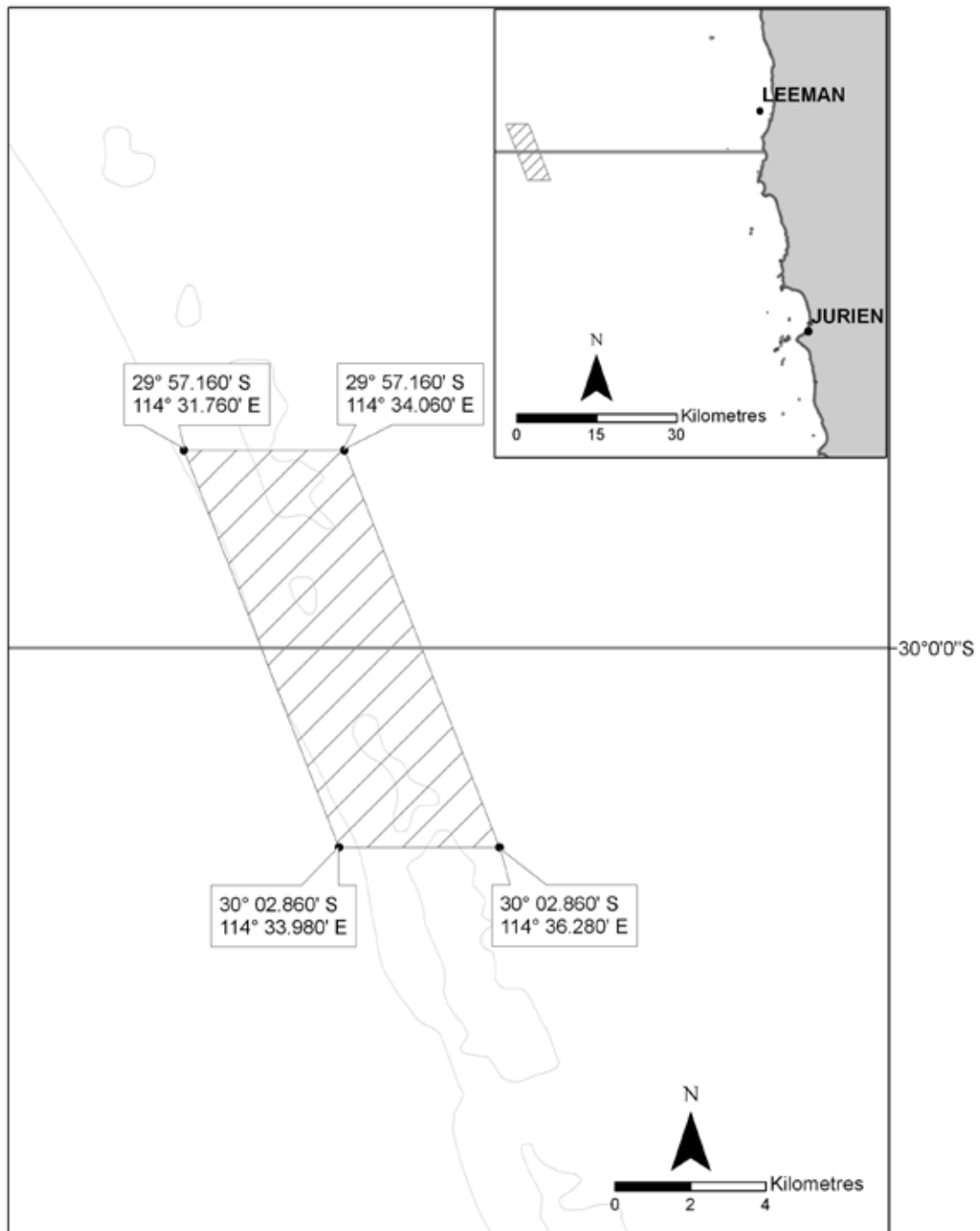
Figure 7.8. Size and position of research area off Leeman that was closed to commercial lobster fishing in March 2011. Light and dark bathymetric lines refer to refer to 50 m and 100 m depth contours, respectively.

7.1.5   Predictive models of substrate and biota

The hydro-acoustic bathymetry data collected for the Leeman study site is presented in Figure 6.10. Depths ranged across the area surveyed from < 85 m in the west to *ca.* 45-50 m on top of the bank, which runs roughly north-south through the centre of the area. This bank is part of an ancient submerged coastline and also forms the primary bathymetric structure at the Jurien study sites (Figure 7.9). In general, there is a greater area of shallow (< 55 m deep) habitats at Jurien than at the Leeman site.

A total of 3122 video frames from the 55 km of transects conducted at the Leeman site were classified for substrate and biota occurrence. Rhodoliths were the most prevalent substrate (*ca.* 31 %), followed by flat reef (Table 7.1). High and medium relief reef combined (> 1 m in height) contributed < 10 % to substrates. The dominant biota were non-*Ecklonia* macroalgae (29 %) and sessile invertebrates/sponges (20 %). *Ecklonia* was observed in 16 % of all video frames, which is consistent with the values recorded earlier for individual transects (Figure 7.1a).

Table 7.1.   Occurrence of primary substrate and biota categories observed in frames from towed video footage.

Category	% of frames
Substrate	
Sand	14
Rhodoliths	31
High Reef	2
Medium Reef	5
Low Reef	15
Flat Reef	19
Obscured Reef	13
Biota	
Ecklonia (kelp)	16
Other macroalgae	29
Sessile invertibrates	20
Hard coral	<1
None	34

These data were used to parameterise substrate and biota models. Substrate models explained 53 % to 83 % of the total deviance (Adjusted D^2 in Table 7.2). These high values indicated strong associations with geophysical environmental variables. The strongest association was for reef, and the main predictors of reef were bathymetry (depth) and the standard deviation of the surface area (Table 7.2). These two predictors, along with detrended bathymetry were the most influential variables for the substrate models. The individual contributions of different variables are described in Hovey *et al.* 2012.

For biota categories, sessile invertebrates had the highest deviance explained by the model with an Adjusted D^2 of 87 %, followed by other macroalgae (70 %) and *Ecklonia* (48 %). *Ecklonia* and sessile invertebrates showed strong associations with depth and reef whereas the contribution of variables for other macroalgae was spread more evenly among multiple predictors (Table 7.2).

Table 7.2. Predictive performance of substrate and biota models and model predictors which were used to create distribution maps.

	Area under curve (Bootstrap)	% Sensitivity	% Specificity	% Correct	Adjusted D2	Predictors
Obscured Reef	0.72 (0.67-0.78)	48	94	88	53	Bathymetry, focal analysis (surface area, mean, 21 m)
Reef	0.77 (0.73-0.81)	70	76	75	83	Bathymetry, focal analysis (surface area, st dev, 7 and 21 m), standard deviation (12.5 m), Hypsometric index (12.5 m), rugosity and curvature
Sand	0.74 (0.69-0.78)	76	65	66	68	Detrended bathymetry, range (25 and 62.5 m), focal analysis (surface area, mean, 21 m), slope and aspect
Rhodoliths	0.77 (0.71-0.85)	66	78	74	81	Bathymetry, focal analysis (surface area, mean, 21 m), hypsometric index (12.5 and 62.5 m), aspect, focal analysis (surface area, st dev, 7 m), Standard deviation (12.5 and 62.5 m)
Ecklonia (Kelp)	0.94 (0.89-0.97)	90	84	85	48	Bathymetry, focal analysis (surface area, mean, 21 m), range (25 m), reef
Other macroalgae	0.70 (0.66-0.74)	60	68	66	70	Bathymetry, focal analysis (surface area, mean, 21 m), Range (12.5 and 62.5 m), reef, obscured reef, sand, slope, aspect, rugosity, hypsometric index (12.5), standard deviation (62.5 m)
Sessile inverts	0.80 (0.77-0.84)	60	87	81	87	Bathymetry, reef, flat reef, range (62.5 m), focal analysis (surface area, st dev, 7 and 21 m)

The substrate models all had acceptable predictive power (AUC range between 0.72 and 0.77, Table 7.2). The model for Obscured reef had the lowest ability to correctly predict presence of obscured reef with a sensitivity of 48 %; however, this model had a high specificity (correct absence) value, i.e. 94 %. Of the biota models, those for both *Eckonia* and sessile invertebrates had high predictive power (AUC = 0.94 and 0.80, respectively). *Ecklonia* had the best predictive performance with both high sensitivity (90 %) and specificity (84 %). Other macroalgae model performance was poorer (AUC = 0.70), with low sensitivity and specificity values (Table 7.2)

The spatial distribution of dominant substrates and biota at Leeman are represented in maps (Figures 7.9). Vegetated habitats, including those with *Ecklonia,* are strongly correlated with the ridge and lee of reef substrates. Rhodoliths were mainly found on the shallow side of the reef and extended toward the sandy substrates to the east. Maps of dominant substrates developed for the Jurien area showed similar patterns in spatial distributions of reef, rhodoliths and sand (Figure 7.10). As expected, there is a strong relationship between vegetated habitats and reef substrates at Jurien and generally more vegetated habitats at Jurien than Leeman. This fact likely reflects the slightly shallower depths at Jurien. In addition the prevalence of macroalgae as a dominant habitat type at Jurien may, at least in part, be related to video classification artefacts.

For most of the biota categories, much of the variation in distribution could be explained by depth and geophysical variables. This confirms that depth and geomorphology are the principle drivers of biota distribution in these deep water ecosystems. Moreover, these variables are ecologically relevant as they likely reflect important physiological (e.g. light requirements), environmental or ecological (e.g. hard substrates to attach holdfasts) limitations (Anderson and Miller 2004, Moore *et al.* 2009).

Though time consuming and expensive to produce, habitat maps are critical for assessing the spatial relationship of important marine resources to their environment. The development of spatially explicit, detailed habitat maps in this study has allowed us to accurately represent the benthic environment in an area of known importance for the western rock lobster. The models can be used to explore complex geomorphic characteristics and the major drivers in benthos distributions in deep water lobster habitats.
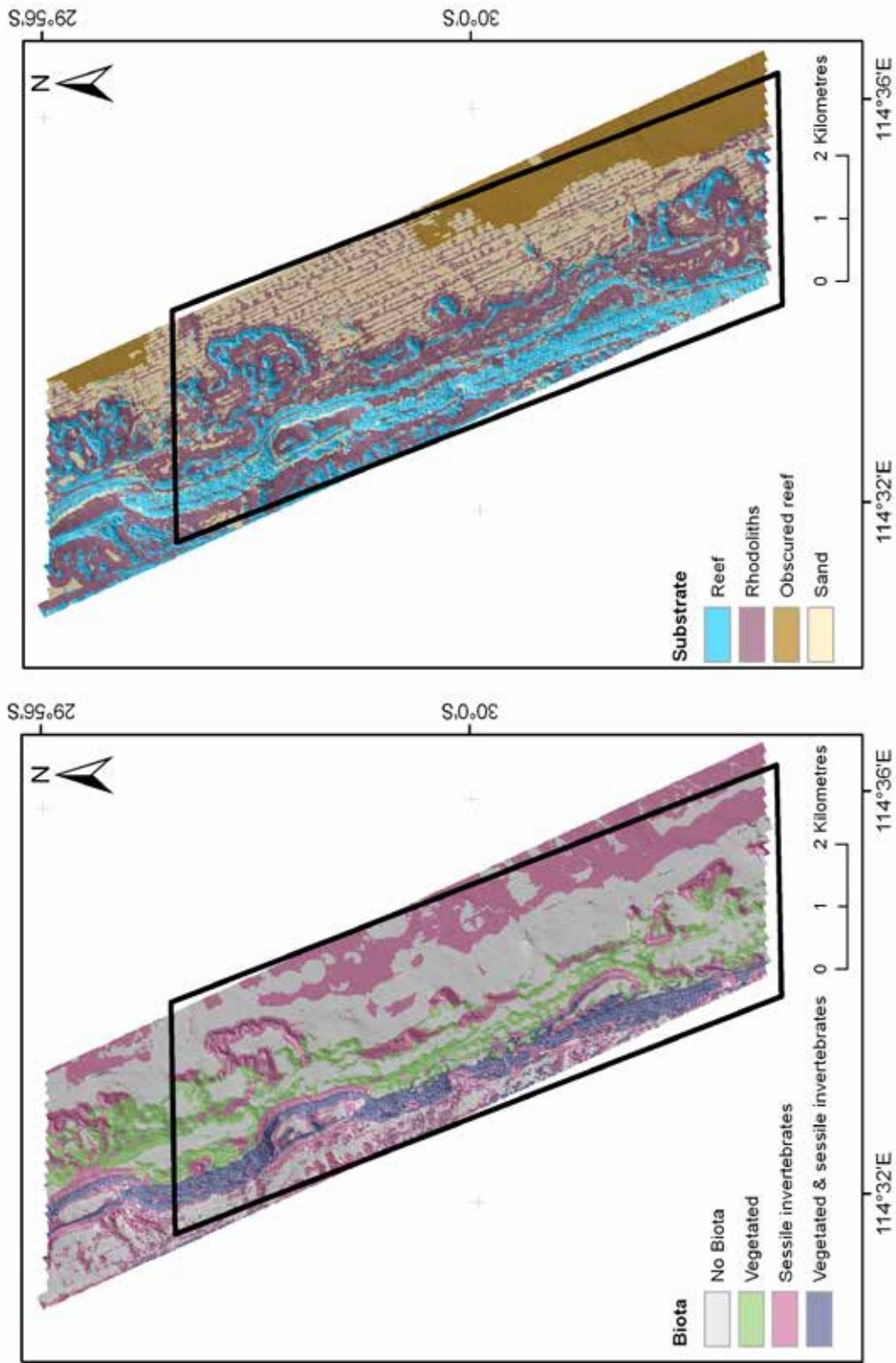
Figure 7.9. Extent of substrate and biota types modelled at Leeman on the basis of hydro-acoustic data and ground truthed with towed video. Closed area is designated by black box.
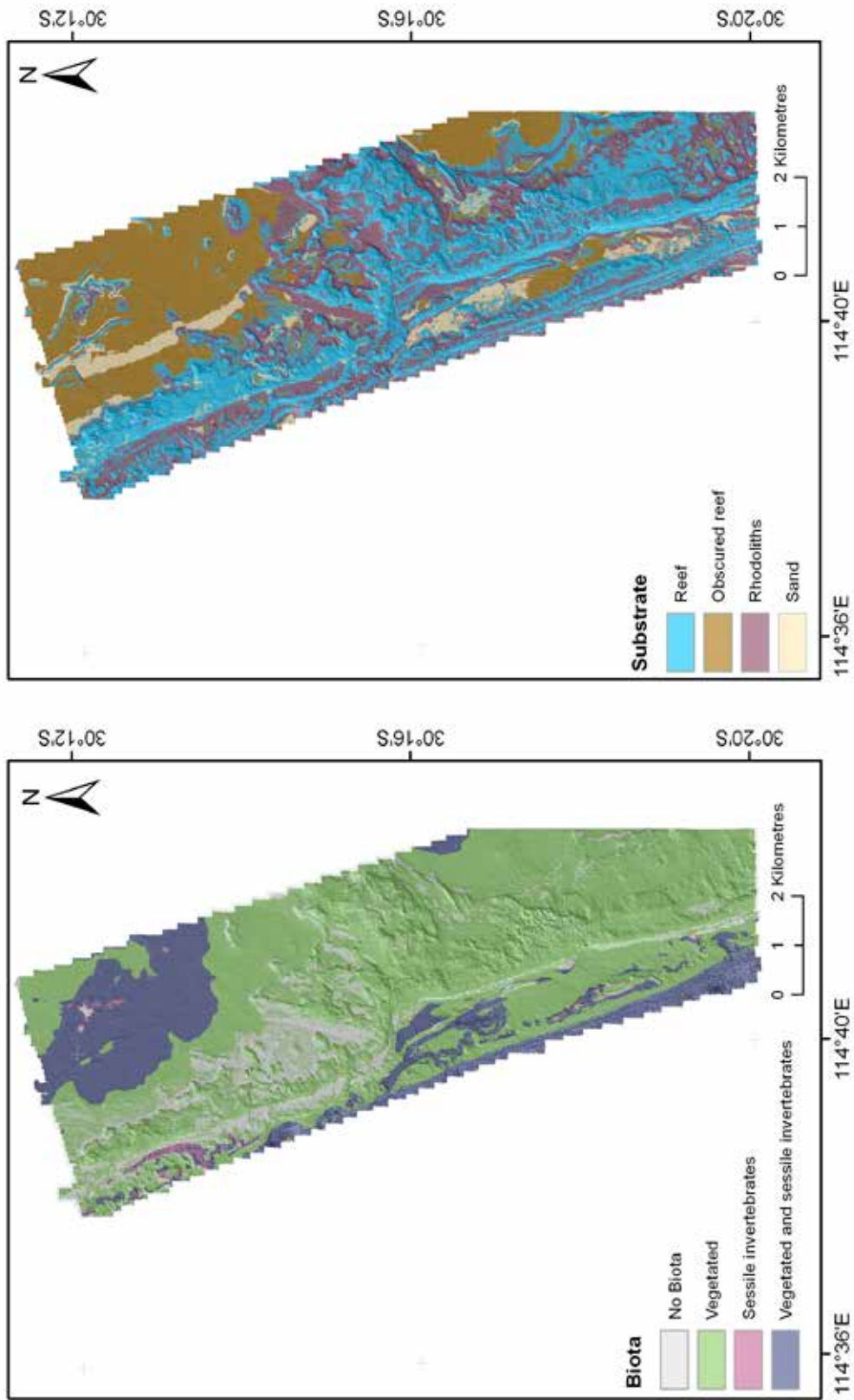
Figure 7.10. Extent of substrate and biota types modelled at Jurien on the basis of hydro-acoustic and derived terrain data sets collected as part of the Securing our Marine Futures Project.

7.2 Development of a qualitative trophodynamic model that will provide a conceptual framework for determining sampling protocols, indictors and targets.

7.2.1 Results

Table 7.3 details the results of the adjoint matrices and the average proportion of correct sign for each simplified model. General fish were identified as the best indicator of overall change in the impact of the fishery on rock lobster because this variable had both consistency of response signs between different models and the highest average proportion of correct sign.

Table 7.3. Predicted sign of responses to change in the impact of the fishery on the rock lobster population with the index of sign determinacy (ISD) in parenthesis to indicate the reliability of the predicted response. An indicator must not be directly impacted by the fishery (i.e. cannot be the rock lobster fishery, rock lobster or bait). Results from general fish have both consistent signs between different models and high average proportion of correct sign (bold text) and may be used as an indicator of change in the fishery. Small crustaceans may be used as an indicator of the effect of bait on the ecosystem (dotted line).

Group	Model A	Model B	Model C	Model D	Model E
Predators	+ (0.68)	+ (0.58)	- (0.52)	- (0.66)	- (0.55)
Cephalopods	+ (0.58)	- (0.82)	- (0.97)	- (0.94)	- (0.72)
Demersal fish	+ (0.63)	- (0.64)	- (0.85)	- (0.94)	- (0.83)
Crabs and polychaetes	+ (0.80)	+ (0.66)	- (0.55)	+ (0.68)	+ (0.83)
Algae	+ (0.80)	+ (0.66)	- (0.55)	- (0.77)	- (0.62)
General fish	+ (0.94)	+ (0.98)	+ (0.99)	+ (0.99)	+ (0.97)
Bivalves/gastropods	- (0.80)	- (0.66)	+ (0.55)	+ (0.77)	+ (0.62)
Small crustaceans	+ (0.99)	+ (0.97)	+ (0.95)	- (0.72)	- (0.58)

In addition, small crustaceans may be a useful indicator of the specific influence of bait in the system, as this group was predicted to increase in abundance when bait was included in the model (Models A-C) and decline in abundance when bait was assumed to have little influence on the system (Models D and E) (Table 7.3). Following the identification of general fish as the most reliable indicator for change in the rock lobster fishery in Models A-E, this variable was disaggregated to form an intermediate model with four new variables (old wife, sweep & wrasse, foxfish and small fish) based on prey groupings (Figure 7.3). Similar to the simplified model (Model A), the majority of these variables (old wife, sweep & wrasse and small fish) were predicted to increase in response to a perturbation to rock lobster fishing (Table 7.4). These variables also had very high average proportion of sign determinacy (Table 7.4) and may therefore be good indicators of change in the rock lobster fishery.

Table 7.4. Predicted sign of responses with the index of sign determinacy (ISD) in parenthesis for general fish variables in the Jurien intermediate model (Figure 7.4); old wife and sweep and wrasse were identified as the most appropriate indicators.

Variable	Predicted response
Old wife	+ (1.00)
Foxfish	- (0.85)
Sweep and wrasse	+ (1.00)
Small fish	+ (1.00)
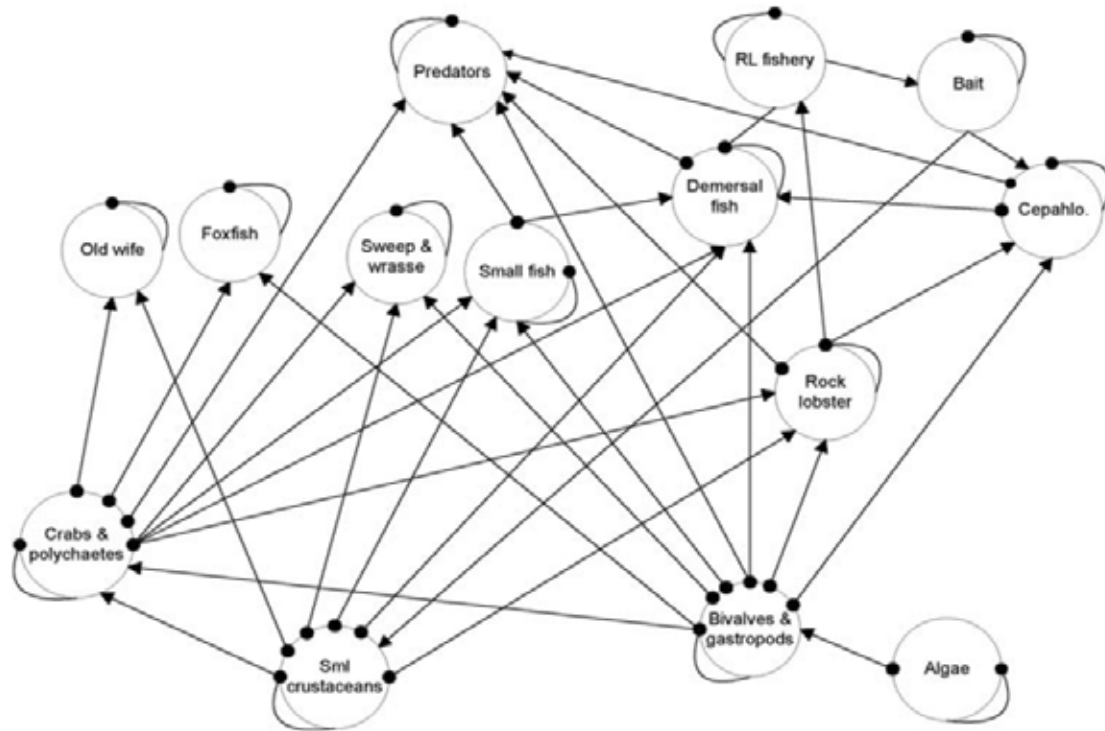
Figure 7.11. The intermediate model where the indicator variable identified in the simplified model, general fish, has been disaggregated into the four variables old wife, foxfish, sweep and wrasse, and small fish.

Foxfish was the only 'general fish' variable that was predicted to decline due to a perturbation to the fishery. This decline was predicted as foxfish do not consume small crustaceans as a major food source in the model and therefore do not receive an indirect benefit from the fishery through bait. In contrast, old wife, small fish and sweep and wrasse all received an indirect positive impact from the fishery through the input of bait. Small fish were assumed to be an inappropriate indicator as they were not adequately defined and, in addition to the blackspot wrasse and bullseye, contained 'unidentified teleosts' from the diets of dhufish, breaksea cod and sea lions. Monitoring for change in this variable would therefore be difficult, as the species in this variable cannot be adequately identified.

Investigation into the percent aggregation error was undertaken for the simplified model (Model A) and the intermediate model. Investigation into the aggregation error of Models B-E was not undertaken as these situations were not thought to be as likely to occur as Model A. Model A was found to have 21.49 % aggregation error. The intermediate model had a moderate level of aggregation error (9.18 %) and was more likely to result in the same predictions as the complex model. This reduced aggregation error occurred in the intermediate model as a large amount of feedback involving the 'general fish' variables was retained, and their aggregation in the simplified model resulted in the loss of this information.

7.2.2 Discussion

The complexity and data limitations associated with ecosystems are often a major source of uncertainty in scientific studies, and researchers must be able to work with this uncertainty in order to produce meaningful results. Qualitative models can easily incorporate structural uncertainty into ecosystem investigations (Hayes *et al.* 2008), and this uncertainty can actually

be made useful by aiding the identification of indicators of change. For instance, alternative models were produced to represent different scenarios for the deep water ecosystem (i.e. with and without the impact of bait and the fishery on cephalopods) and demersal ecosystem (i.e. with and without a significant impact from the rock lobster fishery). The comparison of results highlighted general fish as variables that were robust to these uncertainties and may therefore provide good indicators of change due to fishing. The capacity to investigate structural uncertainty using qualitative models is important because parameter uncertainty is often taken into account by fisheries scientists, yet structural uncertainty is often neglected (Hill *et al.* 2007). Ignoring uncertainty or attempting to monitor every aspect of an ecosystem would likely produce useless results and the swift exhaustion of financial resources.

The use of qualitative models also allowed the assessment of aggregation error, which aided model selection by highlighting the lower aggregation error in models of intermediate-complexity. This lower aggregation error occurred because the disaggregation of the indicator variable allowed the models to retain more complex information than the simplified model. Models of intermediate complexity can reduce the disadvantages associated with both very simple, highly aggregated models and very detailed, disaggregated models. Less detailed models have been criticised for being unable to produce realistic behaviours (Fulton *et al.* 2003) and, if they include aggregated variables, are subject to aggregation error (Gardner *et al.* 1982, Cale *et al.* 1983, Auger *et al.* 2000). Yet, forced model complexity can be problematic when data is limited, as it may increase uncertainty (Kimmins *et al.* 2008). The intermediate model was found to have lower aggregation error than the simplified model, yet retained the relatively high sign determinacy generally associated with very simple models. It must be noted here that lower aggregation error does not indicate that the results are 'correct'. Rather, if we assume that the complex models are an appropriate representation of the system, the results from models with low aggregation error should be useful for assessing ecosystem dynamics and identifying indicators for monitoring.

The lobster fishery was found to have an indirect influence on the abundance of the disaggregated variables: old wife, small fish and sweep and wrasse. These groups benefit from reduced competition for food resources by lobsters targeted by the fishery. In addition, these groups may receive an indirect benefit from the addition of bait to the system. This result lends support to the suggestion that the addition of bait may have a substantial impact on the trophic dynamics of the ecosystem (Waddington and Meeuwig 2009). Accordingly, small crustaceans were identified as a potential indicator of the impact of bait on the ecosystem. Empirical data collected for small crustaceans in areas open and closed to commercial fishing (i.e. with or without bait input) may increase the capacity of researchers to quantify the ecological effects of bait input hypothesised by Waddington and Meeuwig (2009). However, the operational difficulties in collecting small invertebrates from deep water, in combination with the high level of taxonomic expertise required to analyse samples, may make monitoring small invertebrates too costly. Furthermore, the recent transition of the WRLF toward a quota managed fishery has led to a marked reduction in total bait usage (e.g. 10 900 tonnes in 2008/09 to 4500 tonnes in 2009/10) and presumably a lower risk of adverse ecological impacts of bait input into deep water systems.

While qualitative models can be useful for aiding the selection of indicators in data-limited situations, the selection of indicators must also take into account the efficiency with which populations can be monitored (Langlois *et al.* 2012). While some captured species are currently used as indicators by the Department of Fisheries due to the ease of monitoring through catch data, the use of both captured species and non-retained species as a 'suite' of indicators may prove to be more effective at providing an indication of the broader ecosystem effects of fishing

(Cury and Christensen 2005). The cost of monitoring, both in terms of time and resources, is critical in the selection of indicators for fisheries management (de Jonge *et al.* 2006) as the majority of fisheries agencies cannot afford to support expensive field-based monitoring regimes without significant return for their investment. Nonetheless, pressure remains to provide accurate and sensitive predictions for change in target stocks and ecosystems. An assessment of different monitoring methods must therefore follow the identification of indicators to ensure the necessary data is collected at a reasonable cost (time and resources). The collection of data regarding the abundance of small fish, which were highlighted in both the demersal and deep water ecosystems as potential indicators, may be difficult due to depth (deep water ecosystem) making dive transects impossible or the need for substantial resources to fund the analysis of video data. While qualitative models can be used to guide research towards species likely to be effective indicators, further analyses is required to ensure that these indicators are monitored using the most efficient methods.

The identification of a 'suite' of potential indicators, as suggested by Cury and Christensen (2005), when only commercial catch data is available remains a significant problem for fisheries management. Long-term, fishery-independent time series are often few and far between (e.g. Shin *et al.* 2010), particularly for non-commercial species (Fletcher 2005). The use of qualitative models has been shown in this study to be useful for combating this lack of data and allowing the identification of specific relationships and variables that will be affected by perturbation (through the examination of feedback). Qualitative models can therefore aid the identification of useful indicators, where 'usefulness' is defined as the ability to link a change in the indicator to a specific perturbation as well as the availability of data (measurability) and the sensitivity to detect change (Link *et al.* 2009). Furthermore, it is widely acknowledged that the social and economic aspects of fisheries must be included in Ecosystem Based Fisheries Management (EBFM); however, the consideration of these aspects is generally only undertaken in very broad terms (i.e. Christie *et al.* 2007, Vieira *et al.* 2009). Qualitative models can be beneficial in progressing EBFM through the identification of social and economic indicators of change in fishery systems, in addition to ecological indicators. The capacity for incorporating uncertainty and assessing aggregation error determines that qualitative modelling can be particularly beneficial in data-limited situations and may prove to be useful in many future studies as a precursor to quantitative analyses focussing on specific indicators.

7.3 Provision of cost effective methods to measure deep water ecosystems in both fished and unfished reference areas.

7.3.1 Results

7.3.1.1 Lobster

7.3.1.1.1 Visual survey and commercial pot comparison

Twenty-five lobsters were recorded by UVC giving an average encounter rate of <1.4 lobsters 10 min^{-1}. The estimated size range of lobsters observed by UVC was 25 to 95 mm (Figure 7.12). Only one lobster was captured by divers and the estimated CL (85 mm) closely matched the calliper measurement recorded underwater (83.6 mm).

Commercial pots (with open escape gaps) and meshed pots caught 64 and 92 lobsters, respectively. This corresponds to catch rates of *ca* 1.83 and 2.49 lobsters pot^{-1}. Although the mode in the size range was similar for the two potting methods, the size ranges were vastly different with

commercial pots not retaining any lobsters with CLs <70 mm (72.5 – 117.1 mm) (Figure 7.12). In contrast, meshed pots caught a large number of smaller lobsters (44.3 - 109.2 mm).
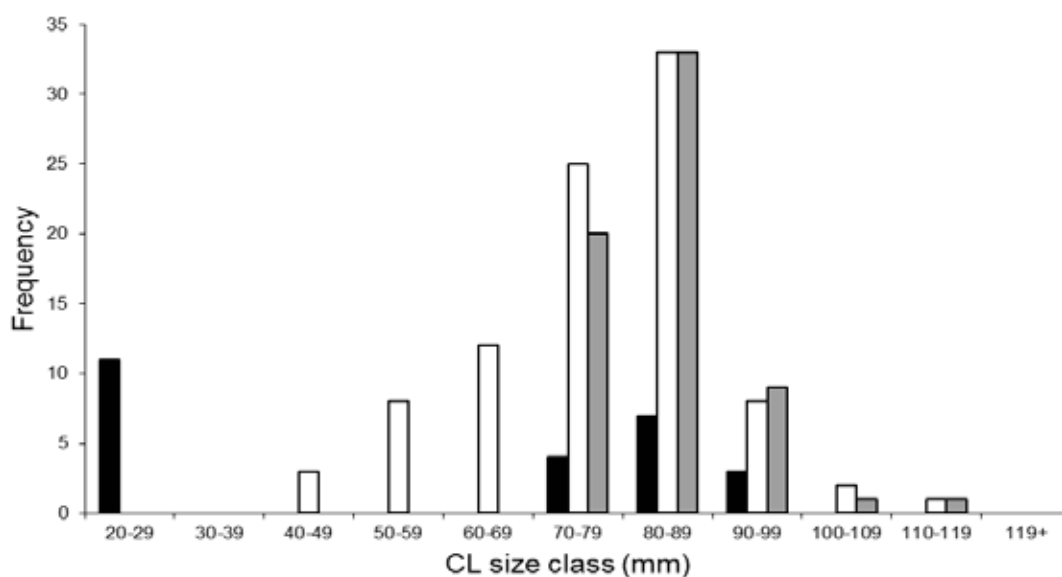
Figure 7.12. Comparative size frequency distributions of UVC (black) and potting (commercial – grey; meshed - white) surveys

7.3.1.1.2 Lobster abundance in deep water

A total of 9034 modified commercial pots were pulled between 2009 and 2012 at Jurien and Leeman as part of the lobster sampling regime, resulting in the capture of 19388 lobsters. The average total catch rate at the locations varied markedly between years from a low of 1.57 lobsters pot^{-1} at Leeman in 2009 to a high of 10.3 lobsters pot^{-1} at Jurien south in 2012 (Table 7.5). The pot catches were not spatially uniform across the regions. Catch rates were further analysed to investigate the effect of habitat (reef, rhodoliths and sand) and closure to fishing at Leeman (sites L1 v L2).

Table 7.5. Effort (pot lifts), catch (total lobsters) and CPUE (lobsters pot^{-1}) over five years at the two Jurien sub-regions and Leeman combined.

		2008	2009	2010	2011	2012	Totals
	Jurien North	142	160	159	160	156	2785
Pots	Jurien South	144	159	160	158	160	2789
	Leeman	270	296	295	294	297	3460
	Jurien North	347	257	647	611	1107	4978
Catch	Jurien South	427	384	737	900	1648	6105
	Leeman	582	466	1039	1285	2924	8305
	Jurien North	2.44	1.61	4.07	3.82	7.1	
CPUE	Jurien South	2.97	2.42	4.61	5.7	10.3	
	Leeman	2.16	1.57	3.52	4.37	9.85	

ANOVA of the total catch rates of lobsters caught in the three habitats (reef, rhodoliths and sand) at the four sites (two each at Jurien and Leeman) over five years showed that catch rates differed significantly among years ($p<0.001$), habitats ($p<0.001$) and sites ($p<0.01$) (Table. 7.6).

A GLM based on the same data found 2012 to be significantly greater (*p*<0.001) than 2008 and there was a significant interaction between year and site with L2 (the closed area at Leeman) being significantly different (*p*<0.01) to the other sites in 2012. The influence of year on total catches rates can be seen in all habitats with catch rates generally rising at all sites between 2009 and 2012. This was particularly true on reef where median catch rates went from between 2.4 and 3.6 lobsters pot[-1] in 2009 to between 6.0 and 8.0 lobsters pot[-1] in 2011 (Figure 7.13). All sites recorded a median >8.0 lobsters pot[-1] on reef in 2012 and this value was the highest for L2 (19.25 lobsters pot[-1]) demonstrating the significant year x site interaction for L2 in 2012 in the GLM.

Table 7.6. Mean squares (MS) and significance levels (*p*) for the ANOVAs of the catch rates of the total number of lobsters, legal sized lobsters and legal sized male lobster in three habitats at four sites over five years.

	df	Total catch		Legal sized		Legal males	
		MS	p	MS	p	MS	p
Year (Y)	4	41.6	<0.001	46.7	<0.001	8.09	<0.001
Site (S)	3	3.04	0.007	2.36	0.007	2.10	<0.001
Habitat (H)	2	15.8	<0.001	8.66	<0.001	2.63	<0.001
Y x S	12	0.80	0.388	1.24	0.011	0.74	<0.001
Y x H	8	0.65	0.550	0.65	0.325	0.33	0.118
S x H	6	1.19	0.150	0.68	0.306	0.21	0.385
Residuals	418	0.75		0.56		0.20	

When the same analyses were performed on catch rates of lobsters above the legal size (including setose females), very similar patterns were observed with catch rates again differing significantly between years (*p*<0.001), habitats (*p*<0.001) and sites (*p*<0.01). ANOVA also demonstrated a significant (*p*<0.05) interaction between year and site (Table 7.6). The GLM found that both 2012 and 2011 were significantly greater than 2008 (*p*<0.001) and the 2012 x L2 interaction was still significant (*p*<0.01). The effect of year on catch rates was marked on reef habitats where median catch rates of legal lobsters remained <3.5 lobsters pot[-1] at all sites between 2008 and 2010 (Figure 7.14). In contrast, median catch rates at each site were greater than 4.7 and 6.5 legal lobsters pot[-1] in 2011 and 2012, respectively. In addition, the median catch rate at L2 on reef in 2012 was more than twice the next nearest site, L1 (16.75 vs 8.25 legal lobsters pot[-1]) (Figure 7.14).

A component of the increase observed in the abundance of legal sized lobsters in the closed area was attributable to males. All main effects and a site x year interaction were each highly significant (*p*<0.001) when the catch rates of legal males were subjected to ANOVA (Table 7.6). Median catch rates of legal males on reef at site L2 increased from a median of ≤1.0 lobsters pot[-1] prior to closure to 2.5 and 3.5 lobsters pot[-1] in 2011 and 2012, respectively (Figure 7.15). In contrast, the highest catch rate of legal males observ3ed at any other site was *ca* 1.3 lobsters pot[-1] at L1 in 2011.
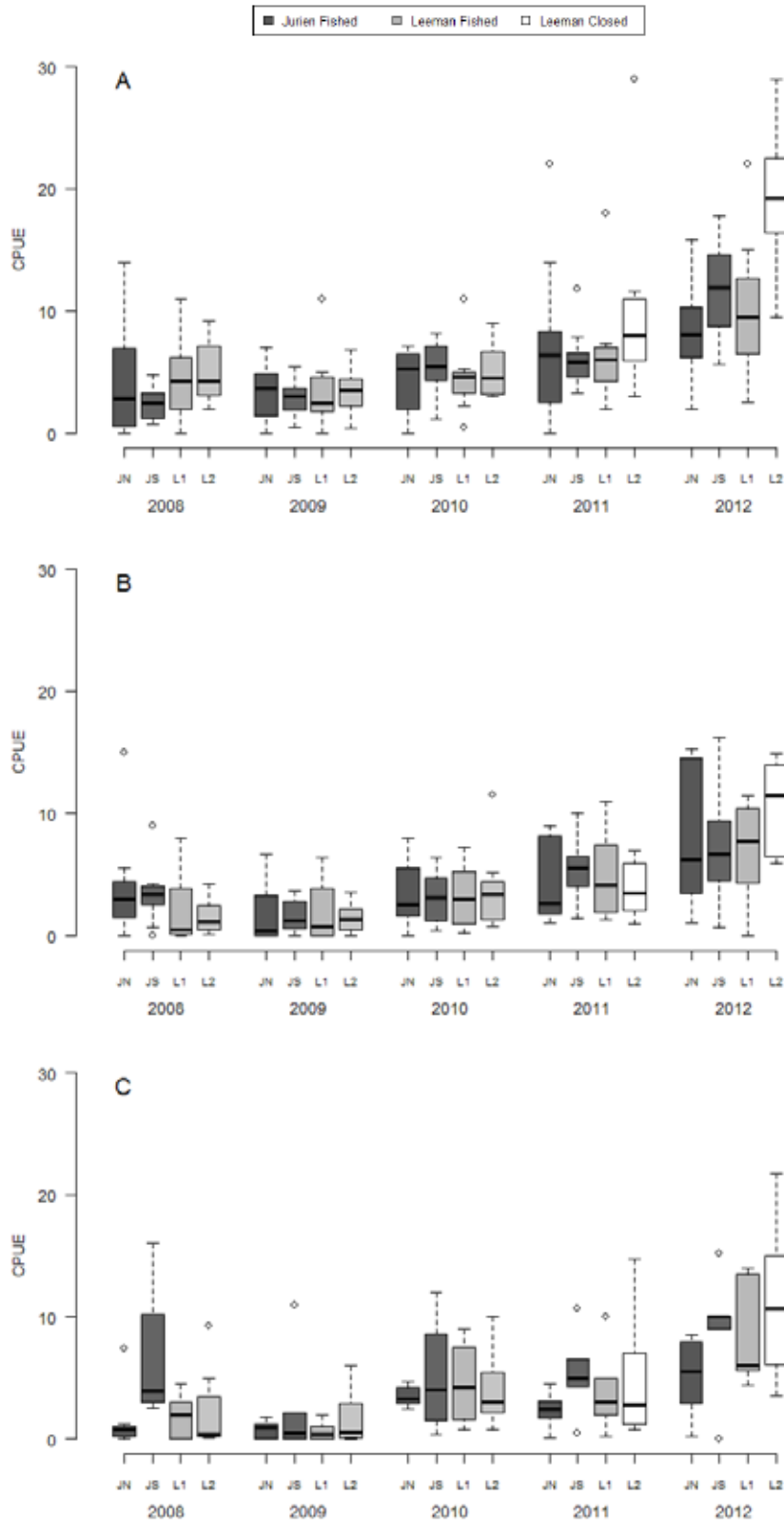
Figure 7.13. CPUE (Total lobsters pot^{-1}) on a) reef, b) rhodoliths and c) sand between 2008 and 2012 at Jurien; (JN and JS: dark grey bars) and Leeman; (L1 and L2; light grey bars). Note that site L2 at Leeman became closed to fishing in 2011 and is differentiated with an unfilled bar in 2011 and 2012. In this and subsequent figures, the median of each sample is displayed with a black band inside the box which represents the inter-quartile range. Whiskers denote 95% confidence limits and outliers are plotted.
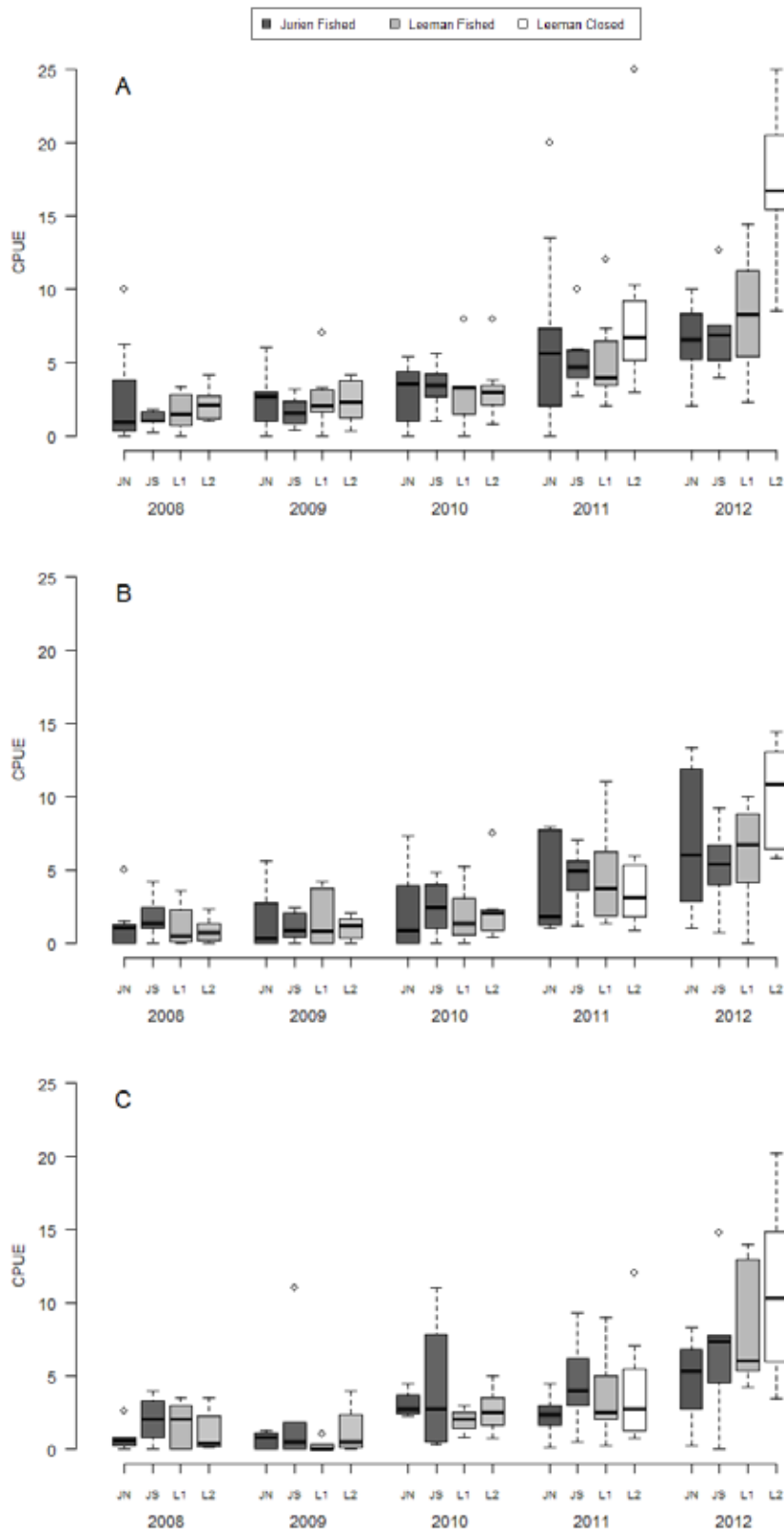
Figure 8.14. Median catch rates (Legal lobsters pot^{-1}) on a) reef, b) rhodoliths and c) sand between 2008 and 2012 at Jurien; (JN and JS: dark grey bars) and Leeman; (L1 and L2; light grey bars). Note that site L2 at Leeman became closed to fishing in 2011 and is differentiated with an unfilled bar in 2011 and 2012.
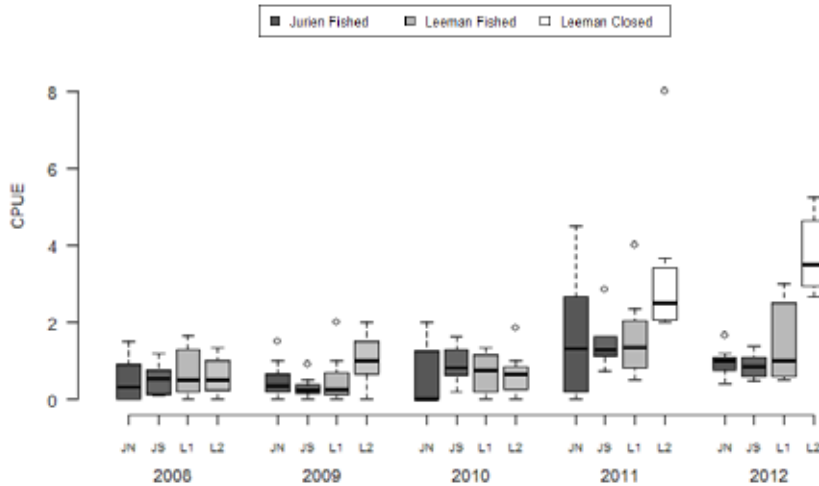
Figure 7.15. Median catch rates (Legal male lobsters pot^{-1}) on reef between 2008 and 2012 at Jurien; (JN and JS: dark grey bars) and Leeman; (L1 and L2; light grey bars). Note that site L2 at Leeman became closed to fishing in 2011 and is differentiated with an unfilled bar in 2011 and 2012.

7.3.1.1.3 Lobster size composition in deep water

The length and sex compositions of the lobsters caught with commercial pots in 2008 were similar at all four sites with each site having a mode between 74 and 78 mm (Figure 7.16). At each site, numbers dropped off sharply at sizes >78 mm. Patterns in size composition remained relatively comparable between sites over the next three years. Over this time the modes typically became less defined and size distributions became biased toward greater numbers of larger lobsters. This shift in lobster demographics between the two periods is demonstrated by the fact that *ca* 72% of lobsters in 2008 had a CL of <80mm. In contrast, by 2011 more than 75% of lobsters were >80 mm CL.

The size distributions observed in 2012 at Jurien differed from previous years with two clearly discernable modes at 86-90m and 60-64 mm (Figure 7.16). This newly recruited secondary mode was observed at both Jurien north and south. In contrast, only small numbers of lobsters in this size range were caught at Leeman and those observed were predominantly from L1 (both L1 north and L1 south). The primary modes at L1 and L2 were at 82-86 and 84-88 mm, respectively. Legal sized males also had a relatively greater contribution at L2 in 2012 than at the other sites or years.

7.3.1.4.4 Small meshed pots

A total of 114 small meshed pots were sampled at Jurien and Leeman in 2012, resulting in the capture of 996 lobsters. The size distribution of lobsters collected at the two Jurien sites demonstrated two modes at similar length ranges to those observed from commercial pot catches (Figure 7.16 vs Figure 7.17). The main differences were that the modes were less defined and the larger mode comprised of smaller sized lobsters, *i.e.* between 52-66 mm CL (Figure 7.17). Mesh pots also retained substantial numbers of lobsters with CLs <50 mm at Jurien which was not the case for the commercial pots at these sites in 2012.
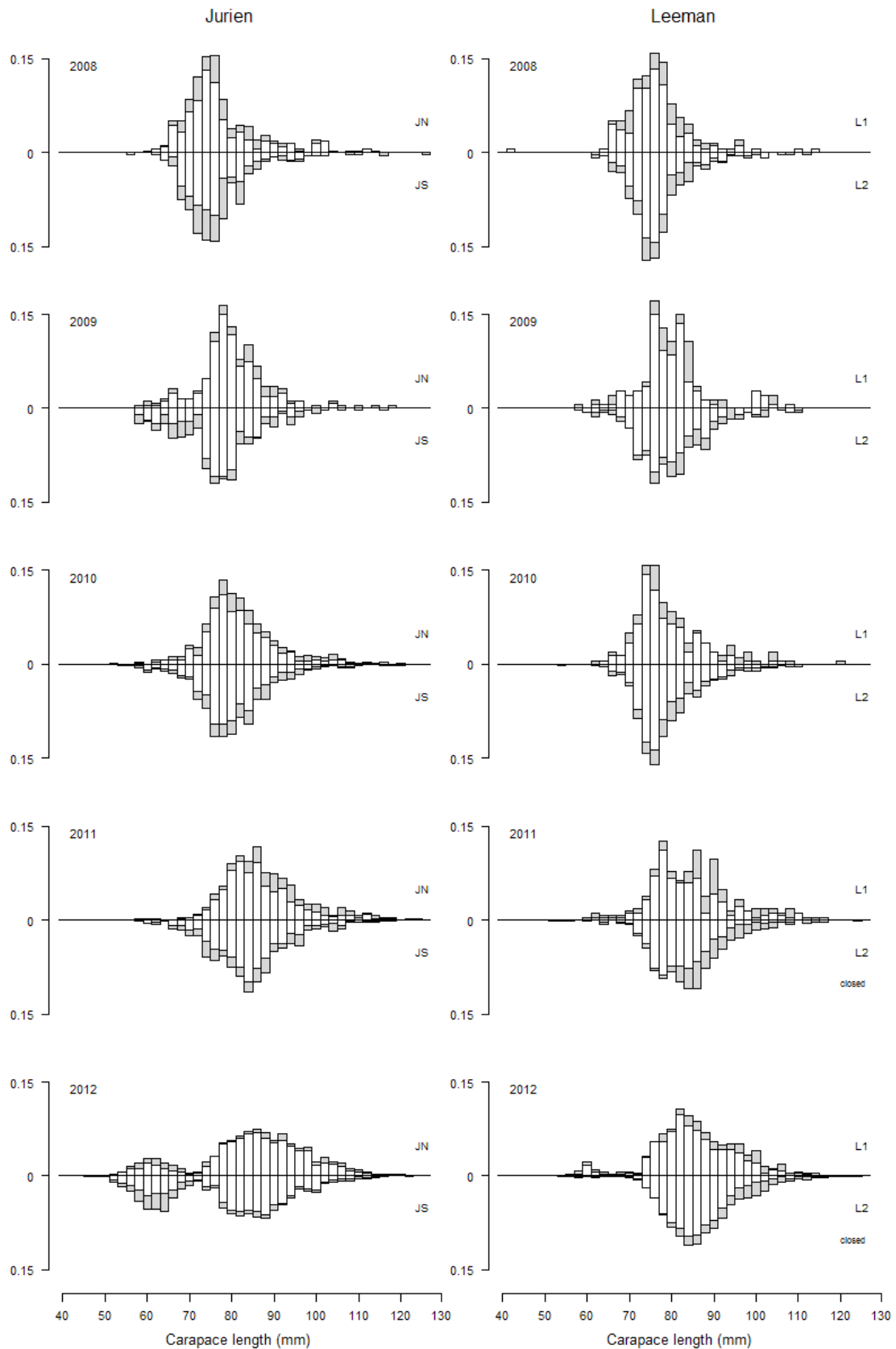
Figure 7.16. Size composition (CL mm) of male (filled bars) and female (un-filled bars) lobsters at Jurien (left) and Leeman (right) between 2008 and 2012. In each year the two sites at each location (i.e JN and JS or L1 and L2) have been plotted above or below the x axis. Note that L2 became closed to fishing prior to sampling in 2011.
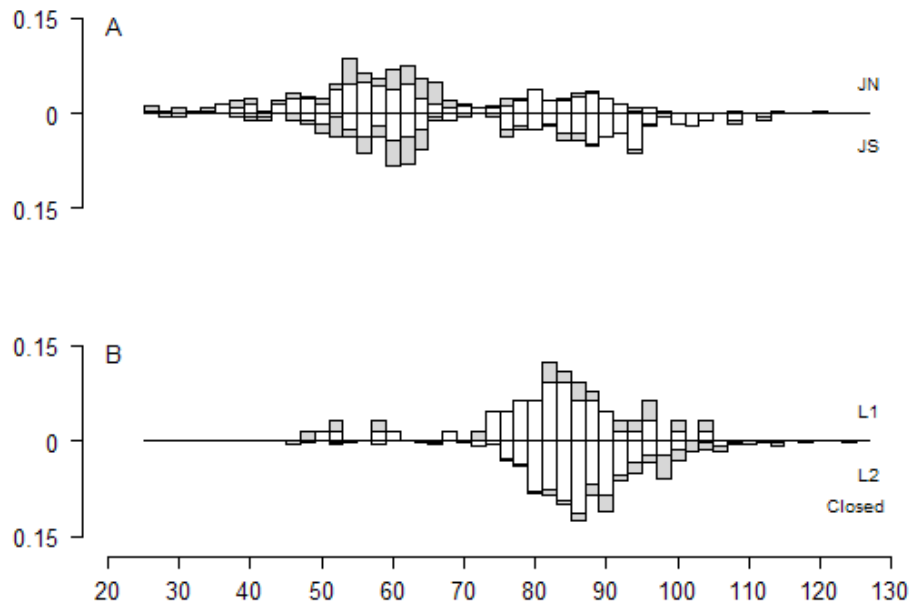
Figure 7.17. Size composition of lobsters caught in meshed pots in 2012 at a) Jurien and b) Leeman. The total number of lobsters (and effort in brackets) at each site in 2012 were: JN, 264 (20 pots); JS, 190 (18 pots); L1, 64 (11 pots), L2, 478 (65 pots).

The size composition of mesh pot catches at Leeman was very similar to that observed from the commercial pots in 2012. Only very small numbers of lobsters from the smaller size cohort were recorded and those that were primarily came from mesh pots set in L1. The mean catch rate did not differ between the four sites for lobsters >70 mm CL. In contrast, catch rates of lobsters <70 mm CL at L2 were significantly ($p<0.05$) lower than at the other three sites.

7.3.1.1.5 *Lobster distribution modelling*

The final classification tree model for the presence/absence of western rock lobster explained 64% of total deviance. The model retained only geophysical variables, including three based on local neighbourhood measures; hypsometric index with 12.5 m kernel radius, range with 62.5 m kernel radius and focal analysis using standard deviation statistic, as well as depth (Figure 7.18). Detrended bathymetry was the most influential variable, contributing 47% of the variation explained, followed by focal analysis (33%), range (10%) and hypsometric index (10%). Figure 7.19 illustrates the lobster distribution map produced for Leeman from the model, based on presence/absence catch data collected between 2008 and 2010.
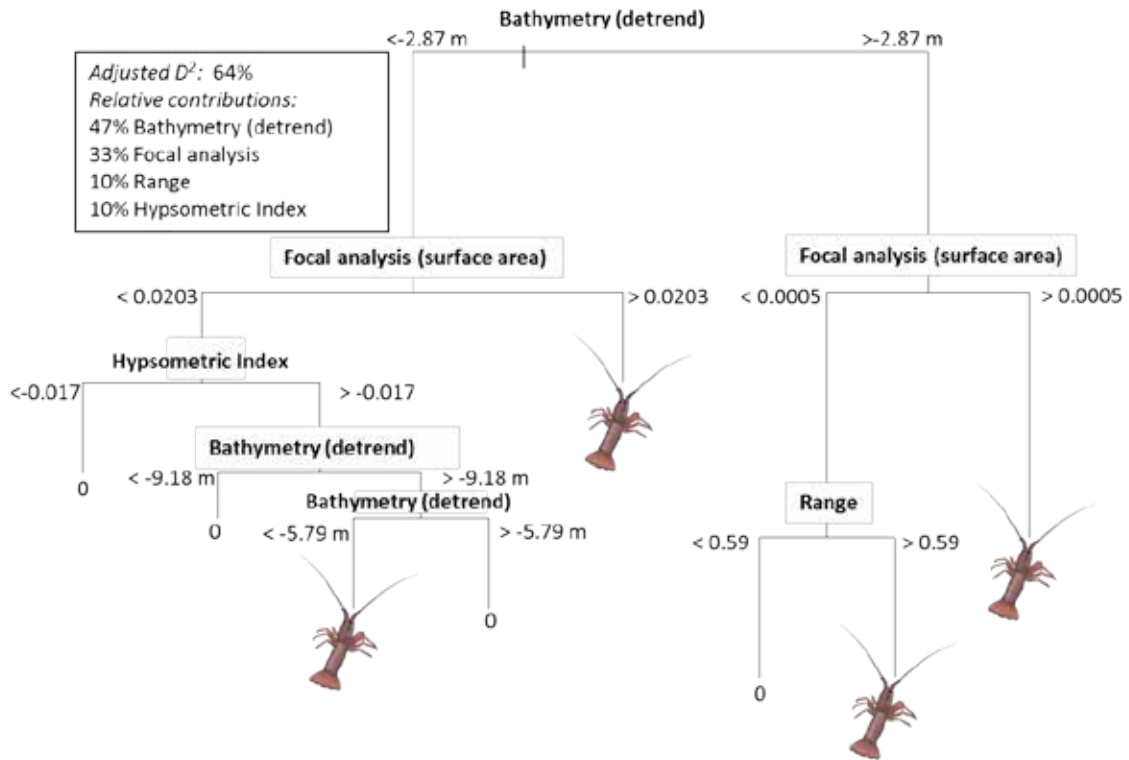
Figure 7.18. Final classification tree model for the presence/absence of lobster,. Focal analysis was calculated based on standard deviation of surface area over a 7 m kernel radius. Hypsometric Index and Range were calculated over a 12.5 m and 62.5 m kernel radius, respectively. Relative contributions to total variation are given in the insert.
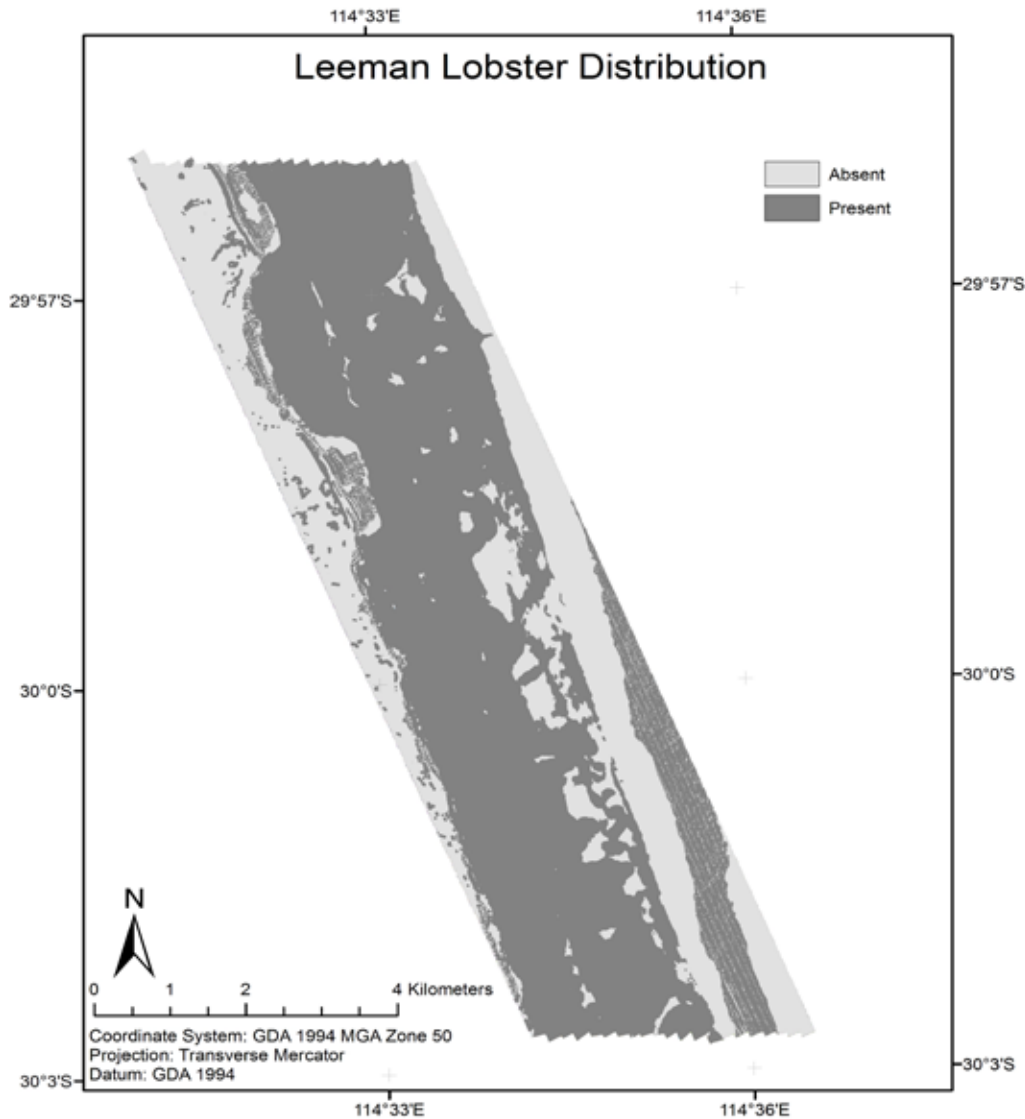
Figure 7.19.   Map of the modelled presence/absence of western rock lobster at Leeman

7.3.1.1.6 Lobster tagging

A combined total of 2021 lobsters were tagged between 2009 and 2012 at Jurien and Leeman (Table 7.7). Recapture rates were generally low with only 69 recaptures in total. The greatest recapture rate (8.5%) was observed at Jurien in 2011 for lobsters tagged in 2012. In contrast, only 1.0% were recaptured in the corresponding period at Leeman, were most of the tagged lobsters were released inside the closed area.

In general, there were no strong relationships between the distance moved and either the size of the lobster or time at liberty. The majority of lobsters were recaptured <2 km from their release position (Figures 7.20 and 7.21). In general the lobsters that had the greatest movement estimates were industry recaptures (Figures 7.20 and 7.21). A single industry recaptured lobster was recaptured *ca* 9.4 km from the release site and all others were estimated to have moved <4 km. The greatest movements recorded for research recaptured lobsters were 1.3 and 1.0 km for two individuals that had both been at liberty for two entire years. The remainder of research recaptured lobsters were at liberty for one year and moved a maximum of 537 m (Figures 7.20 and 7.21). Four individuals were caught in exactly the same location (the same research pot) in two consecutive years.

Overlaying the substrate maps for each region with the release and recapture positions of individual lobsters demonstrates that small-scale movements of lobster were predominantly along the bank or across the bank between areas of similar reef habitat (Figures 7.20 and 7.21).

Table 7.7. Numbers of lobsters tagged each year and the number of those lobsters recaptured each year in either research pots (no parentheses) or in commercial pots (numbers in parentheses).

		2009	2010	2011	2012	Total
Jurien						
No.Tagged		86	262	294	248	890
Recaptured	2009	–				0(0)
	2010	0(2)	1(0)			1(2)
	2011	–	5(4)	4(0)		9(4)
	2012	–	1(6)	0(25)	–	1(31)
	Total	0(2)	7(10)	4(25)	0(0)	11(37)
Leeman						
No. Tagged		118	198	298	517	1131
Recaptured	2009	–				0(0)
	2010	0(1)	–			0(1)
	2011	–	3(7)	–		3(7)
	2012	–	1(1)	0(3)	1(4)	2(8)
	Total	0(1)	4(8)	0(3)	1(4)	5(16)
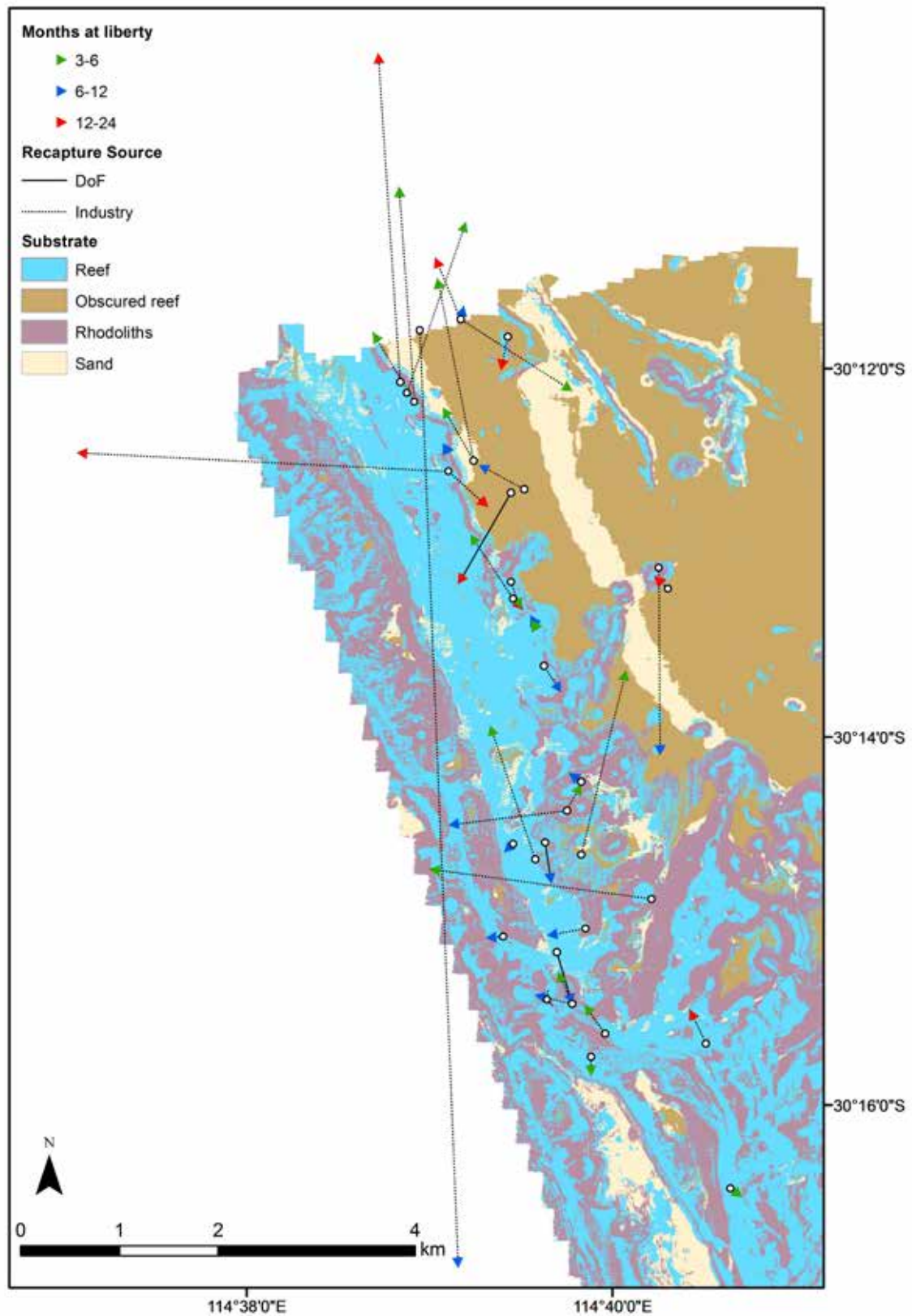
Figure 7.20. Spatial representation of the release (open circle) and recapture (triangle) locations for recaptured lobsters at Jurien.
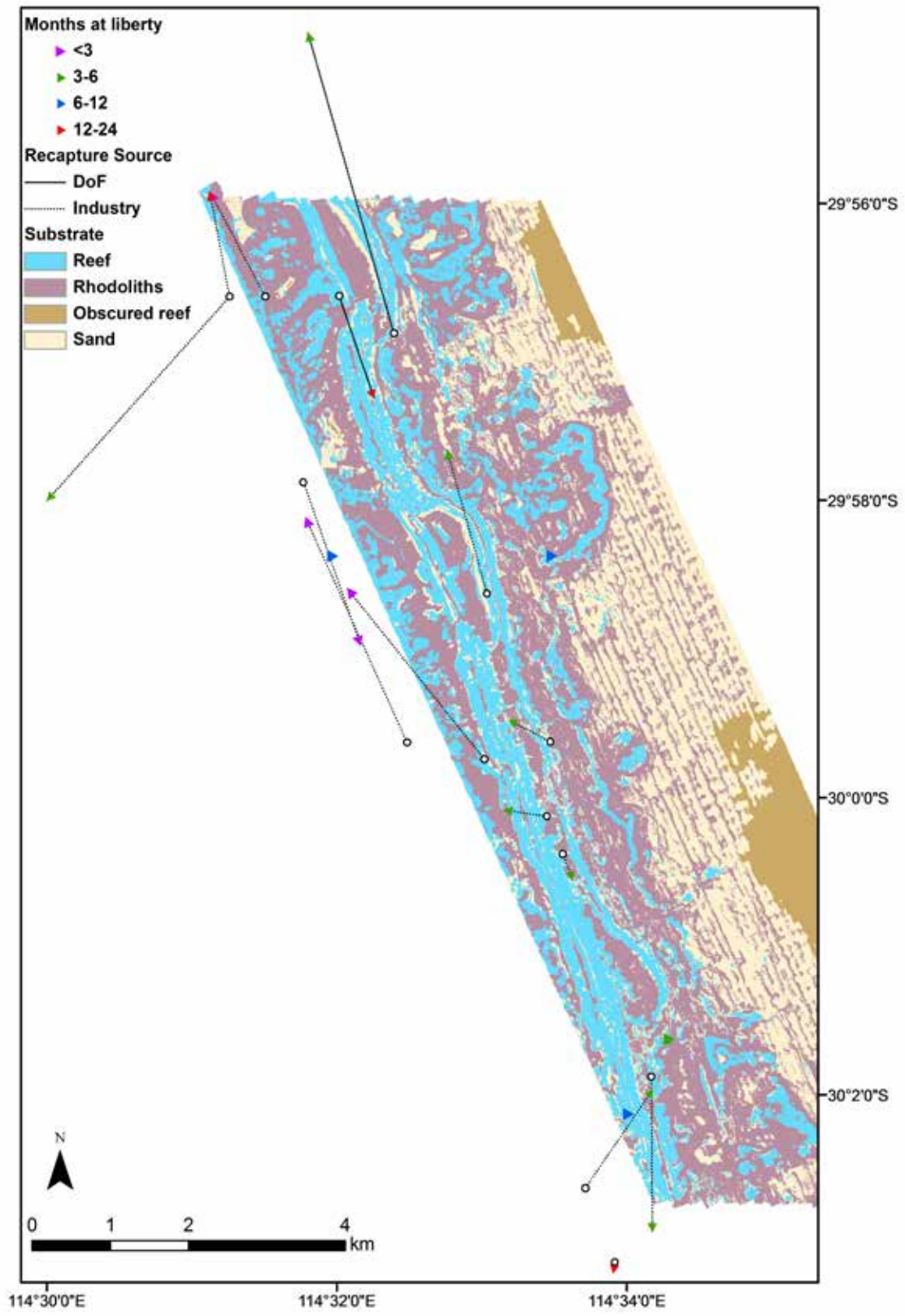
Figure 7.21. Spatial representation of the release and recapture locations for recaptured lobsters at Leeman

7.3.1.1.7 Lobster dietary studies

Sampling of demersal gill net bycatch resulted in 42 lobsters (41 male; 1 female) being collected for dietary studies. The lobsters had a size range of 75.4 to 141.0 mm CL however there were few <90 mm and the overall mean was 108 mm. Only 29 samples (15 summer; 14 autumn) remained after those with <10% GFI or gut fullness were omitted from analyses.

Traditional visual analyses identified 29 dietary items in lobster stomach contents. Two were unidentified groups (unidentified flesh and unidentified other) which contributed 9.4 and 6.9% to the overall total, respectively (Table 7.8). Of the dietary items that could be identified, crustaceans made the greatest contribution (35.4%). This total mainly consisted of Brachyurans (31.6%) followed by Paguroid hermit crabs (3%). Gastropods contributed <8.5% of a total of 11.37% for molluscs. The only other group that approached a contribution of 10% were the polychates (Table 7.8). The contribution of crustaceans was consistent between the two sampling occasions, autumn and summer (Table 7.8). In contrast gastropods, and to a lesser degree polychates, were observed more frequently in autumn than summer. The reverse was true for algae (Table 7.8). Crustaceans were the primary dietary item observed in the stomachs of lobsters both smaller and larger than 100 mm CL (Figure 7.22).
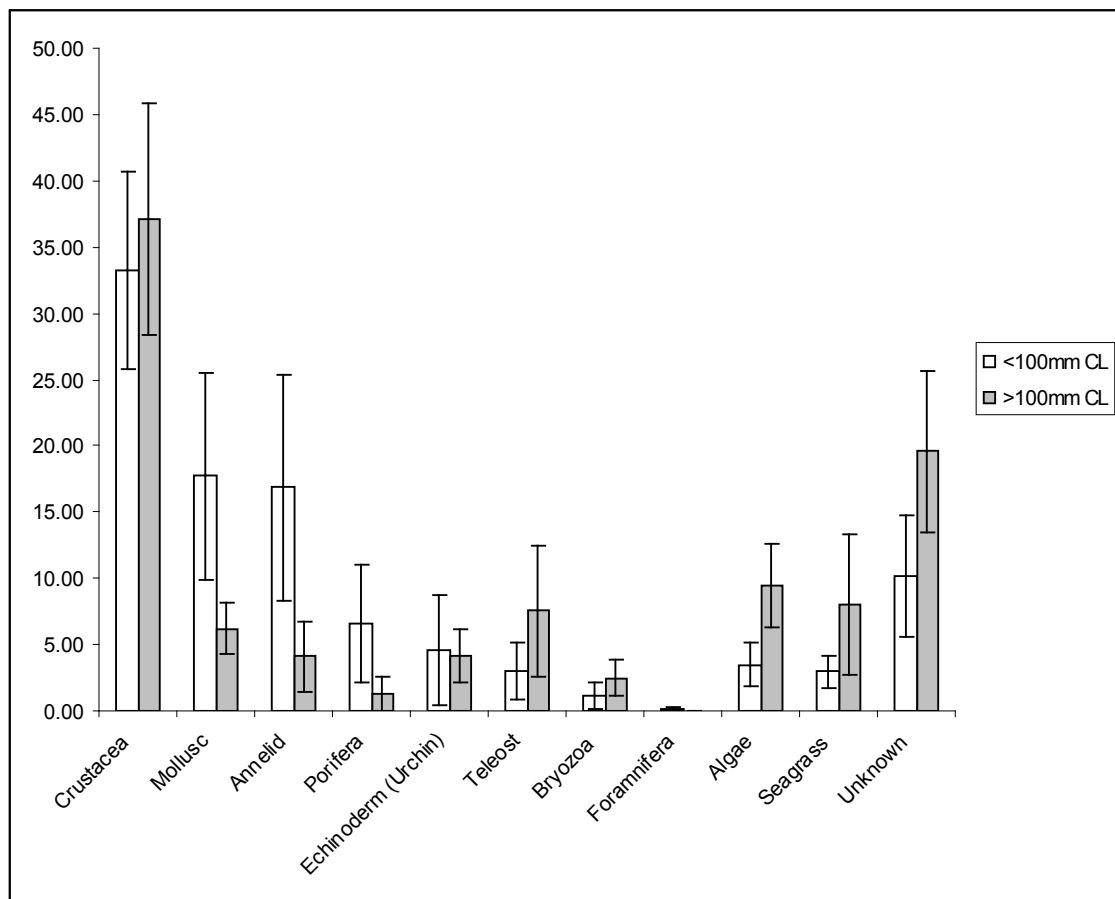
Figure 7.22.   Mean percentage contribution of the main dietary items to stomach contents of lobsters collected from deep water as bycatch from demersal gill nets which had a CL less or greater than 100 mm (open and shaded bars, respectively).

Table 7.8. Mean percentage contribution of the main dietary items to the stomach contents of lobsters collected from deep water as bycatch from demersal gill nets in autumn and spring.

Prey item	Mean contribution to total prey items (%)					
	Autumn (n = 14)		Summer (n = 15)		Total	
Crustacean	31.79 ± 8.30		38.69 ± 8.24		35.36 ± 5.78	
Brachyurans (Majidae)	10.33 ± 3.59		9.08 ± 4.42		9.68 ± 2.82	
Brachyurans (Other)	17.97 ± 8.21		25.68 ± 7.34		21.96 ± 5.44	
Paguroidea (Hermit crab)	3.25 ± 2.50		2.80 ± 1.25		3.02 ± 1.34	
Isopoda	0.02 ± 0.02		0.86 ± 0.68		0.46 ± 0.35	
Ostracoda (Seed shrimp)	0.04 ± 0.04		0.04 ± 0.03		0.04 ± 0.02	
Cirripedia (Barnacle)	0.19 ± 0.19		0.24 ± 0.24		0.21 ± 0.15	
Mollusc	17.56 ± 7.25		5.59 ± 1.84		11.37 ± 3.73	
Gastropod (Cerithiidae)	0.08 ± 0.05		0.00 ± 0.00		0.04 ± 0.02	
Gastropod (Phasianotrochus)	0.27 ± 0.27		0.00 ± 0.00		0.13 ± 0.13	
Gastropod (Phasianella)	1.07 ± 0.51		0.00 ± 0.00		0.52 ± 0.26	
Gastropod (Trochidae)	0.86 ± 0.84		0.00 ± 0.00		0.42 ± 0.41	
Gastropod (Columbellidae)	0.24 ± 0.21		0.00 ± 0.00		0.12 ± 0.10	
Gastropod (Aplysiidae)	0.06 ± 0.06		0.00 ± 0.00		0.03 ± 0.03	
Gastropod (unidentified)	11.74 ± 7.02		3.18 ± 0.92		7.31 ± 3.45	
Bivalves	2.77 ± 1.27		2.41 ± 1.27		2.58 ± 0.89	
Chiton	0.47 ± 0.33		0.00 ± 0.00		0.23 ± 0.16	
Annelida (polychaetes)	12.21 ± 7.93		7.59 ± 3.51		9.82 ± 4.18	
Porifera	2.53 ± 2.32		4.72 ± 3.51		3.66 ± 2.11	
Echinoderm (urchin)	4.48 ± 3.85		4.21 ± 2.18		4.34 ± 2.13	
Teleost	6.21 ± 5.20		4.92 ± 2.94		5.54 ± 2.88	
Bryozoa	2.30 ± 1.13		1.52 ± 1.32		1.90 ± 0.86	
Foraminifera	0.03 ± 0.02		0.15 ± 0.11		0.09 ± 0.06	
Algae	1.35 ± 0.97		11.83 ± 3.18		6.77 ± 1.95	
Ecklonia	0.06 ± 0.04		0.00 ± 0.00		0.03 ± 0.02	
Phaeophyta	0.10 ± 0.10		0.00 ± 0.00		0.05 ± 0.05	
Hypnea	0.02 ± 0.02		0.00 ± 0.00		0.01 ± 0.01	
Articulated Red Corallines	0.99 ± 0.96		2.56 ± 1.45		1.80 ± 0.88	
Algae (unidentified)	0.19 ± 0.12		9.27 ± 3.32		4.89 ± 1.89	
Seagrass	8.39 ± 5.46		3.28 ± 2.69		5.75 ± 2.96	
Unidentified	13.15 ± 5.11		17.51 ± 6.13		15.40 ± 3.96	
Unidentified (flesh)	9.08 ± 4.01		9.70 ± 4.77		9.40 ± 3.08	
Unidentified (other)	4.07 ± 3.95		7.81 ± 4.51		6.00 ± 2.98	

The six lobsters that had stomach contents analysed using next generation sequencing ranged in size between 96 and 125 mm CL. The GS FLX Junior (Roche) sequencing run generated a total of 5465 reads for this study consisting of 3084 reads from the rbcl primers, 2265 reads from the 16S primers and 163 reads from the Folmer primers. Sequences were obtained for only four of the six rock lobster samples. Sequences were searched using BLASTN against the NCBI GenBank nucleotide database. This was automated in the Internet-based bioinformatics workflow environment, YABI (https://ccg.murdoch.edu.au/yabi/). The BLAST results that were obtained using YABI were imported into MEtaGenome Analyzer (MEGAN) where they were taxonomically assigned. Sequences obtained with the marine plant rbcl primers were all matched to land plants (this has been seen in previous runs and is thought to result from airborne contamination) and were discarded from further analysis.

Visual analysis of stomach contents carried out prior to NGS analysis showed a varied diet for the four individual lobsters which latter successfully produced sequences. All four individuals contained brachyuran crustaceans, two contained hermit crabs and one recorded isopods and ostrocods (Table 7.9). Other groups well represented with multiple occurrences included molluscs, algae, seagrass, bryozoans and polychates. Echinoderms and unidentified fish flesh were observed in one lobster each (Table 7.9).

Table 7.9   Comparison of techniques for the detection of prey items for the four samples analysed using next generation sequencing (NGS). Ticks denote a positive identification for either method.

Prey item		1 Visual	1 NGS	2 Visual	2 NGS	3 Visual	3 NGS	4 Visual	4 NGS
Crustacean	Brachyura	✓	X	✓	✓	✓	X	✓	✓
	Majidae	–	–	✓	X	✓	X	✓	✓
	Micippa thalia	–	–	–	–	–	–	X	✓
	Pilumnidae	–	–	X	✓	–	–	X	✓
	Pilumnus hirtellus	–	–	X	✓	–	–	–	–
	Tiaramedon spinosum	–	–	–	–	–	–	X	✓
	Anomura	–	–	✓	✓	–	–	✓	X
	Diogenidae	–	–	X	✓	–	–	–	–
	Dardanus lagopodes	–	–	X	✓	–	–	–	–
	Isopoda	✓	X	–	–	–	–	–	–
	Ostracoda	✓	X	–	–	–	–	–	–
Mollusc	Gastropoda	–	–	✓	X	–	–	✓	X
	Cerithiidae	–	–	–	–	–	–	✓	X
	Trochidae	–	–	–	–	–	–	✓	X
	Phasianothrochus	–	–	–	–	–	–	✓	X
	Phasianellidae	–	–	✓	X	–	–	✓	X
	Phasianella	–	–	✓	X	–	–	✓	X
	Columbellidae	–	–	–	–	–	–	✓	X
	Columbella	–	–	–	–	–	–	✓	X
	Aplysiidae	–	–	✓	X	–	–	–	–
	Bivalvia	–	–	–	–	–	–	✓	X
Annelida		✓	X	✓	X	–	–	–	–
Echinodermata	Echinoidea	–	–	✓	X	–	–	–	–
Osteichthyes	Teleostei	–	–	–	–	✓	✓	–	–
	Glaucosoma hebraicum	–	–	–	–	X	✓	–	–
Chondrichthyes	Carcharinus plumbeus	X	✓	–	–	–	–	–	–
Bryozoan	Polychaeta	–	–	✓	X	✓	X	–	–
Phaeophyceae		–	–	✓	X	–	–	✓	X
	Ecklonia	–	–	✓	X	–	–	–	–
	Other	✓	X	–	–	–	–	✓	X
Angiospermae	Posidoniaceae	✓	X	✓	X	–	–	✓	X
	Posidonia	✓	X	✓	X	–	–	✓	X

The 16S degenerate primers were designed to amplify DNA from fish species and were therefore not surprisingly able to identify the fish prey in one of the four rock lobster samples to the species level (*Glaucosoma hebraicum*) (Figure 7.23). Sandbar shark *Carcharhinus plumbeus*

was also identified in one lobster which was not observed visually (Table 7.9). These primers also amplified species of crustacea (shrimps, crabs and lobsters) in two of the lobster samples, again down to species in some cases. The LCO1490/ Uni-Minibar R1 primers did not generate any sequences of other invertebrate taxa. (Table 7.9). The taxa identified by both morphology-based and NGS analysis has been compiled in Table 7.9. No sequences were generated by any primer for molluscs, polychates, echinoderms, bryozoans, algae or seagrass taxa. Sequences of human DNA shown in Figure 7.23 were not included as they are thought to result from contamination during the capture of lobsters or handling of gut content samples during visual analysis of gut contents without gloves.
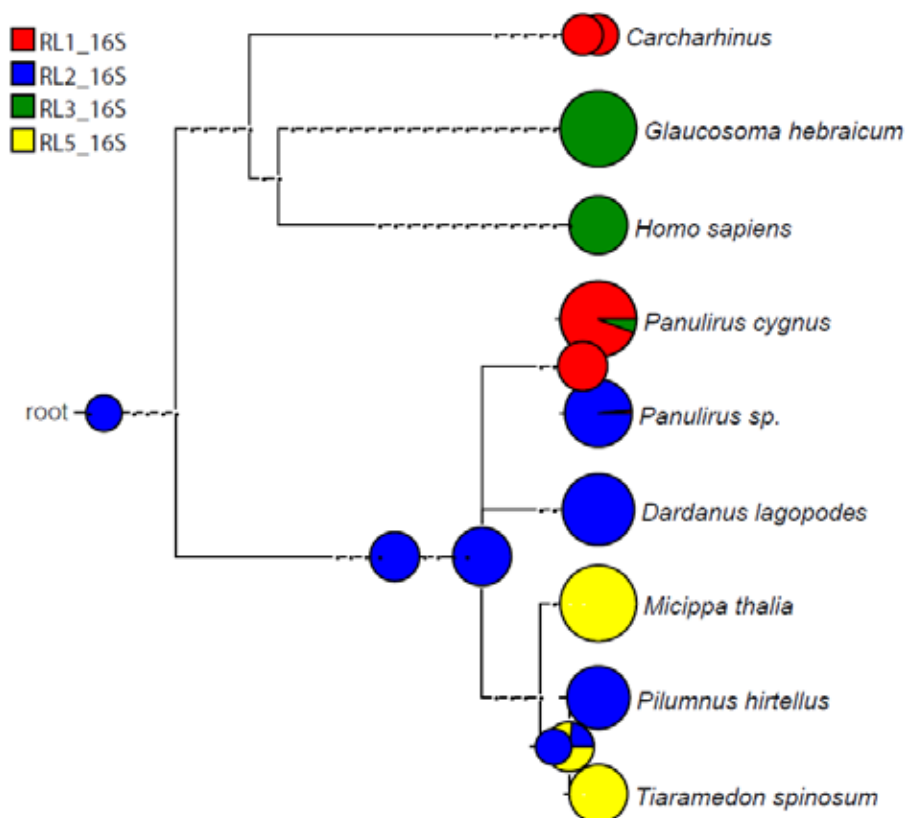
Figure 7.23.   MEGAN phylogram of four lobster samples. The phylogram depicts taxonomic affiliations of the sequences after comparison to GenBank. Size of nodes is proportional to the number of sequence reads within each taxon level.

7.3.1.2 Benthic assemblage

7.3.1.2.1 Assemblage Composition

Each of the transects flown by the AUV at Jurien and Leeman in 2012 were successful in collecting benthic imagery. The overlapping images for the 25 x 25 m grids were spliced together to form mosaics in which characteristics of the sites can be observed at a landscape scale (Figure 7.24). These geo-referenced mosaics retain sufficient resolution at a high zoom for point analyses (Figure 7.25). However, in this study, benthic community composition was analysed at the individual image level. All of the grids mapped contained substantial amounts of hard bottom and included some reef.

The average contribution of all algae varied between the sites from a high of 82.5% at Jurien N1 to 24.8% at Leeman N2 (Table 7.10). The most abundant algal taxa were the kelp *Ecklonia radiata* (17.3%), red foliose algae (14.3%) followed by encrusting coralline algae (7.9%). The mean percentage cover of each of these varied across the sites. For example *Ecklonia* cover accounted for over half of the points analysed at Jurien N1 (52.7%) but was not observed at Leeman N2 (Figure 7.26). The cover of articulated red coralline algae ranged between 0.1% and 5.4% at Leeman N2 and M2, respectively (Figure 7.26d). The contribution of rhodoliths to the benthic assemblage at Leeman N1 was 21.1% but was less than *ca* 3.5% at all other sites.

The sessile invertebrate fauna was dominated by sponges (Table 7.10). The percentage cover of sponges was highest at Leeman N2 (5.7%) but typically <2% at most other sites (Figure 8. 20e). Abiotic categories accounted for a total of 40.9% of the overall composition. Sand inundated rock made the largest overall contribution (25.9%) however the individual contributions differed between sites. For example the highest mean contributions for sand inundated rock (42.1%), bare rock (16.6%) and sand (27.4%) were observed at Leeman M1, Leeman M2 and Leeman N2, respectively (Table 7.10).

There was a large variation in the density of kelp recruits between sites with only Jurien N2 and Leeman M2 recording mean values >1.0 recruits m^{-1}, whereas there were no recruits observed at Leeman N2 (Figure 7.26f).

Mosaics of 25 x 25 m grids

Leeman S1

Leeman S2

Jurien Bay S1

Jurien Bay S2

0   5   10   20 Meters

Figure 7.24. Examples of mosaics of 25m x 25m grids compiled from individual overlapping geo-referenced images collected by the AUV.

Mosaics of 25 x 25 m grids

Leeman S1

Leeman S2

Jurien Bay S1

Jurien Bay S2

0   0.5   1   2 Meters

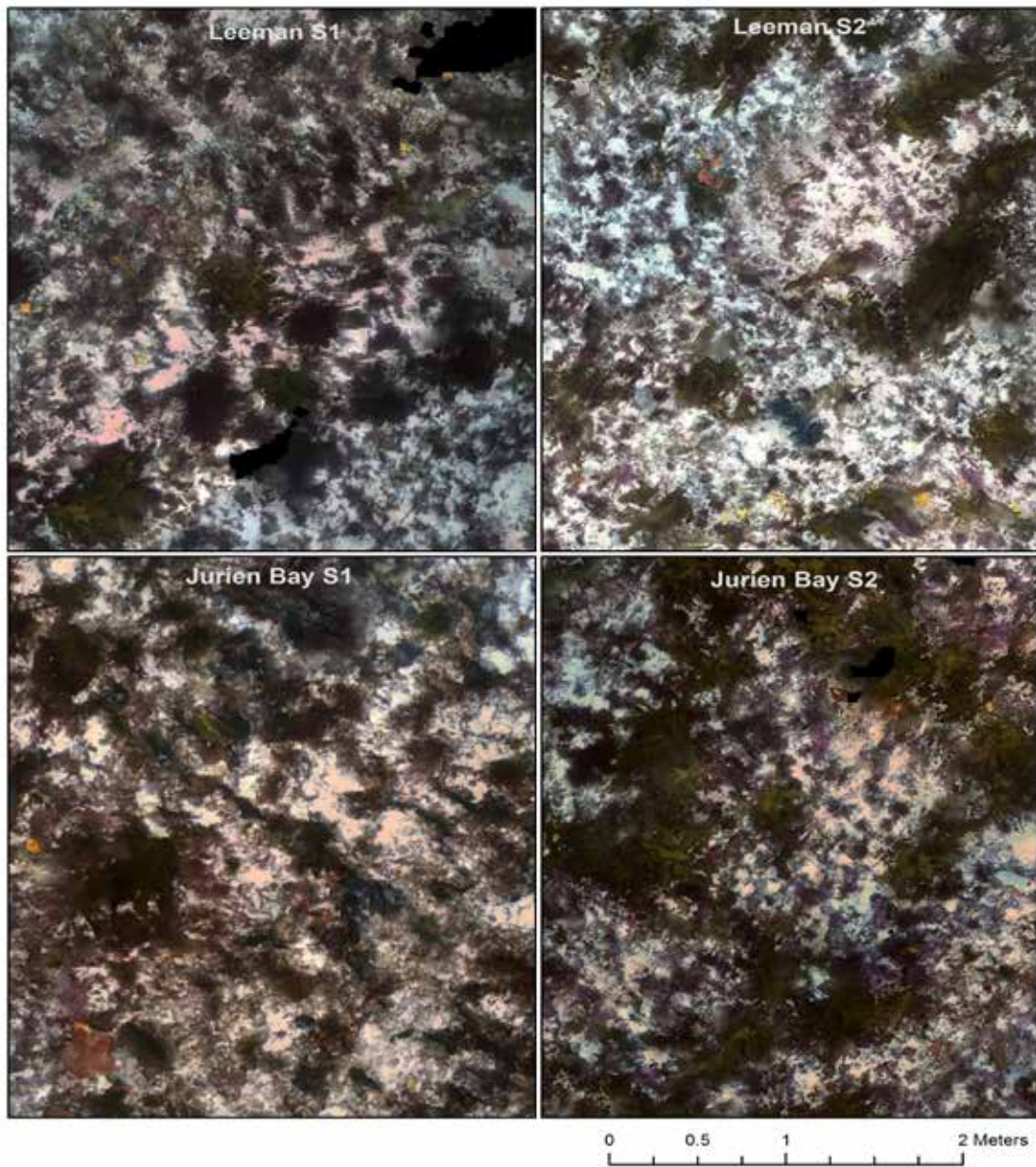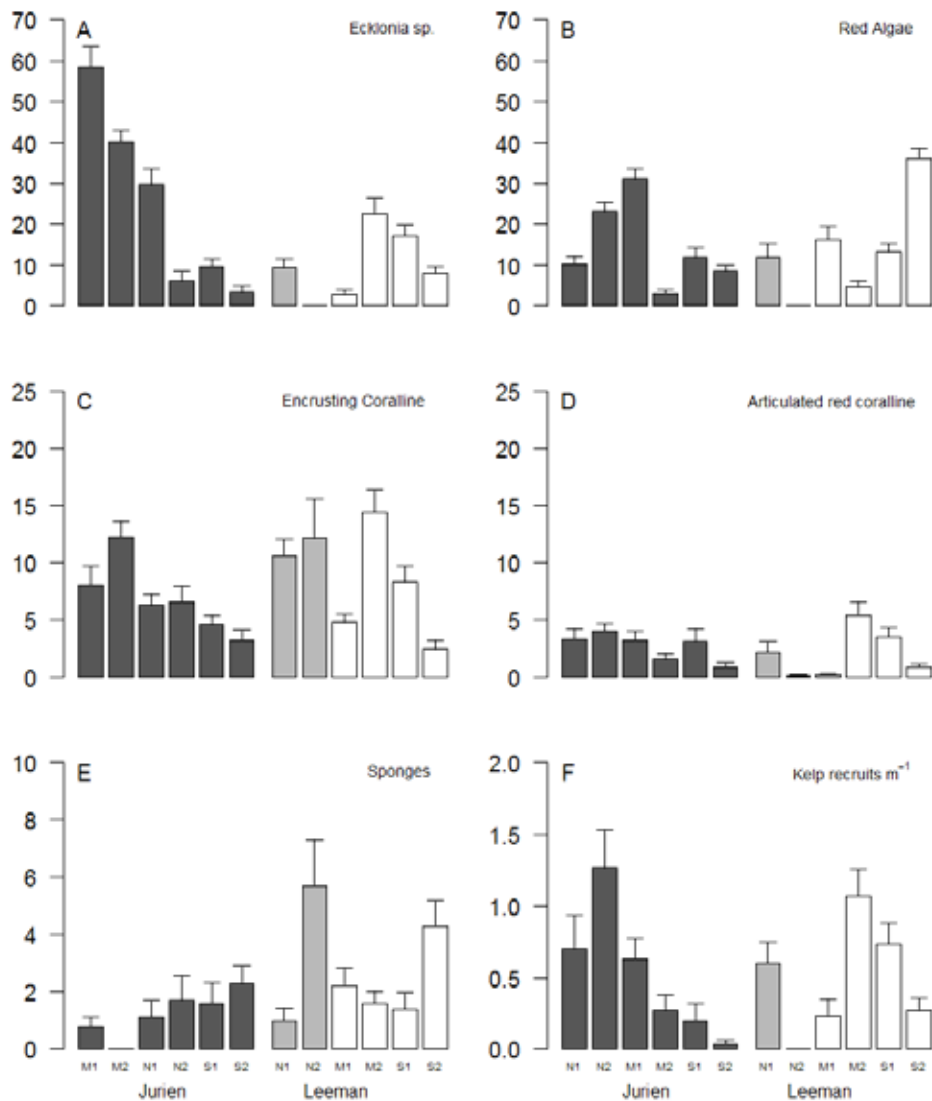**Figure 7.25.** Zoomed in examples of mosaics from previous figure showing detail of biota

Figure 7.26. Percentage contribution a) *Ecklonia radiata*, b) red foliose algae, c) encrusting coralline algae, d) articulated red coralline algae and e) sponges at each site (mesh) sampled with the AUV at Jurien and Leeman. f) The mean density of kelp recruits (number of m⁻¹) is also shown for each site.

Table 7.10 . Mean percentage contribution of the different algal and invertebrate taxa and abiotic classifications at each site (mesh) sampled with the AUV at Jurien and Leeman.

		Jurien						Leeman						Total (%)
		M1	M2	N1	N2	S1	S2	M1	M2	N1	N2	S1	S2	
Algae	Total	73.5	31.6	82.5	81.5	44.5	49.1	34.7	54.4	60.9	24.8	50.7	55.1	53.6
	Ecklonia radiata	29.8	6.2	58.7	40.2	9.6	3.5	2.9	22.6	9.4	0	17.1	7.9	17.3
	Red foliose	31.2	3.2	10.3	23.2	11.8	8.6	16.4	4.8	11.9	0.1	13.3	36.1	14.3
	Encrusting coralline algae	6.3	6.6	8.1	12.3	4.6	3.3	4.8	14.4	10.6	12.2	8.4	2.5	7.9
	Turf algae	0.8	9.9	0.9	0.6	6.1	18.0	0.3	3.5	2.1	10.0	4.0	2.3	4.9
	Articulated red coralline	3.3	1.6	3.4	4.0	3.2	0.9	0.2	5.4	2.2	0.1	3.6	0.9	2.4
	Rhodoliths	0	0	0	0	0.1	0.1	3.3	0.1	21.1	0	0	0	2.1
	Brown foliose	1.0	0.7	0	0.1	0.8	1.3	0.5	0.2	1.4	0.8	0.9	1.1	0.7
	Encrusting algae other	0	0	0	0	0	0	0.1	0.6	0.2	0.5	0.4	0.1	0.2
	Filamentous algae	0	0.1	0	0	0.5	0.2	0.2	0	0.1	0	0	0	0.1
	Green foliose	0	0.2	0	0	0	0	0	0	0	0	0	0	0.0
	Unknown foliose (image quality)	1.0	3.0	1.0	1.1	7.7	13.2	6.0	2.7	1.8	1.1	2.9	4.2	3.8
Inverebrates	Total	1.4	3.6	0.8	0	1.9	3.4	2.4	3.1	1.5	7.2	2.2	4.6	2.7
	Sponge	1.1	1.7	0.8	0	1.6	2.3	2.2	1.6	1.0	5.7	1.4	4.3	2.0
	Bryozoa	0	0.9	0	0	0.3	1.1	0	1.0	0.2	1.1	0.6	0.2	0.5
	Ascidians	0	0	0	0	0	0	0.1	0.4	0	0.1	0	0	0.1
	Encrusting coral	0	0.5	0	0	0	0	0	0	0	0	0	0	0.0
	Other inverts	0.3	0.5	0	0	0.4	0.1	0.1	0.1	0.3	0.3	0.2	0.1	0.2
Abiotic	Total	24.0	61.5	15.3	17.8	50.3	43.4	59.5	38.4	36.0	63.0	43.9	37.9	40.9
	Sand inundated rock	20.8	35.0	6.4	15.1	27.0	31.7	42.1	16.0	32.5	22.4	32.5	29.0	25.9
	Bare rock	3.2	4.8	7.1	2.4	5.5	2.8	13.6	16.6	3.2	12.8	11.1	8.8	7.7
	Sand	0	21.7	1.2	0	17.4	8.8	3.5	4.1	0.3	27.4	0.1	0	7.0
	Gravel	0	0	0.6	0.3	0.4	0.1	0.3	1.7	0	0.4	0.2	0.1	0.3
Unknown	Total	1.1	3.3	1.3	0.7	3.3	4.0	3.3	4.0	1.6	4.9	3.2	2.4	2.8

7.3.1.3 Fish indicators

7.3.1.3.1 Species density and total number of individuals

A total of 17,874 individuals from 140 species and 55 families were recorded from the 173 stereo BRUV deployments. The species density per stereo BRUV ranged from 1 to 31 with an average of 12.02 ± 0.34 SE species observed. A significant difference was recorded between habitats (p <0.001), with the greatest in macroalgae habitats followed by Sessile Invertebrates and the lowest in rhodoliths for both the closed (Leeman) and fished (Jurien) areas (Figure 7.27a). No significant differences in the species richness were detected between the two areas ($p = 0.07$). There were no significant differences in the average total number of individuals per stereo BRUV deployment between the two areas, closed and fished. There was a significant difference in the total number of individuals between habitats (p <0.001), with the greatest in Macroalgae habitats followed by Sessile Invertebrates and the least in Rhodoliths for both the Closed and Fished areas (Figure 7.27b).

7.3.1.3.2 Assemblage Composition

There were significant effects of Area, Habitat and the covariate Depth, with no interactions (Table 7.11). However, these significant effects were not obvious in the unconstrained ordination (Figure 7.28). The significant effects of Area and Depth were investigated using two separate CAP analyses that have been plotted against one another in Figure 7.29. The snapper *Pagrus auratus* was found to be strongly correlated with closed sites, whereas the pigfish *Bodianus vulpinus* and the schooling butterflyfish *Chaetodon assarius* were correlated with Closed sites in deeper water. Four of the proposed indicator species (*Coris auricularis*, *Notolabrus parilus*, *Pseudolabrus biserialis* and *Neatypus obliquus*) were found to be strongly correlated with the shallower water sites in both areas. The significant effect of Habitat was investigated using two CAP axes plotted in Figure 7.30. The CAP axis 1 for habitat found the same four indicator species (*C. auricularis*, *N. parilus*, *P. biserialis* and *N. obliquus*) along with *Labroides dimidiatus*, *Chromis westaustralis* and the sweetlip *Plectorhinchus flavomaculatus* to be strongly correlated with macroalgal dominated sites, whereas the blowfish *Torquiner pallimaculatus* was correlated with rhodolith sites. CAP axis 2 for habitat found *C. assarius* to be strongly correlated with Invertebrate dominated sites.

7.3.1.3.3 Cost-benefit analyses of indicator species

Coris auricularis was the most abundant fish sampled in the study and its abundance was not found to differ significantly between either Area or Habitat (Figure 7.31a). Cost-benefit analyses indicated that relatively small changes of less than 25% could be detected using less than 10 sampling sites inside and outside the closed area. *Neatypus obliquus* was also relatively abundant but significantly more abundant in macroalgal dominated sites, and cost benefit analyses indicated that within macroalgal sites it would require less effort to detect a ~25% change but that up to 15 sites would be required (Figure 7.31b). Although *P. biserialis* was generally rare and Area x Habitat interaction found it to be significantly more abundant at macroalgal sites within the Closed area, and cost-benefit analyses found that despite the low abundance of this specie its low variance meant that 10 sites would be sufficient to detect a change of less than 25% in sites dominated by macroalgae (Figure 7.31c).
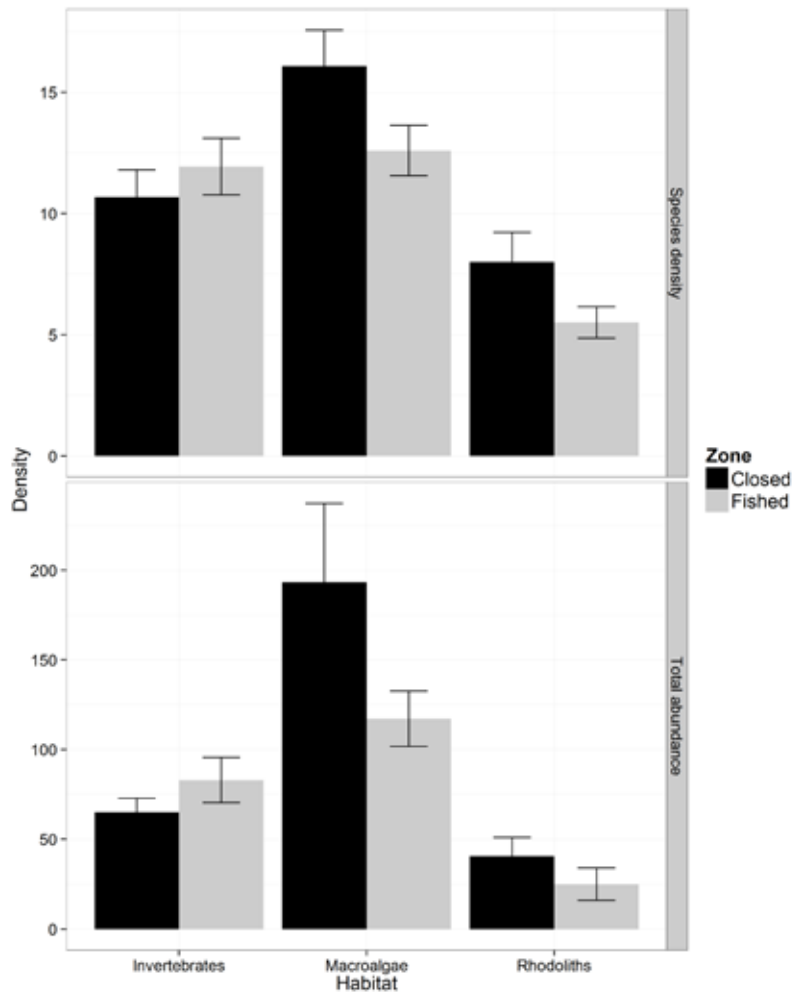
Figure 7.27. Mean species density (top) and total number of individuals (bottom) per stereo BRUV deployment for sessile invertebrates, macroalgae and rhodolith habitats between the closed (Leeman) and fished (Jurien) areas. Significant effect of habitat in the ANOVA model is indicated along with the results of pair-wise comparisons, with significant differences indicated with lower case letters

The proposed indicator species *Enoplosus armatus* was found to be extremely rare in the current study (Figure 7.31d) and cost-benefit analyses could not be calculated. *Notolabrus parilus* was not found to be abundant in the current study, it was significantly more abundant within Macroalgal dominated sites and cost-benefit analyses indicated 10 sites would be sufficient to detect less than 25% change in this habitat (Figure 7.31e). *Pagrus auratus* was not considered to be an indicator species in this study, however, it was significantly more abundant in the closed area and had low variance and the cost-benefit analyses found that a change of less than 25% could be detected by as few as 5 sites within each Area (Figure 7.31f).

The abundance of some non-indicator species was found to correlate with Area or Habitat (Table 7.12). For example, *Bodianus vulpinus* and *C. assarius* were found to be significantly more abundant in the closed area (Figure 7.32a-b). *Chromis westaustralis*, *L. dimidiatus* and *P. flavomaculatus* were all significantly more abundant within macroalgae dominated sites (Figure 7.32c-e), whereas *T. pallimaculatus* was significantly more abundant within rhodolith dominated sites within the fished area (Figure 7.32f).
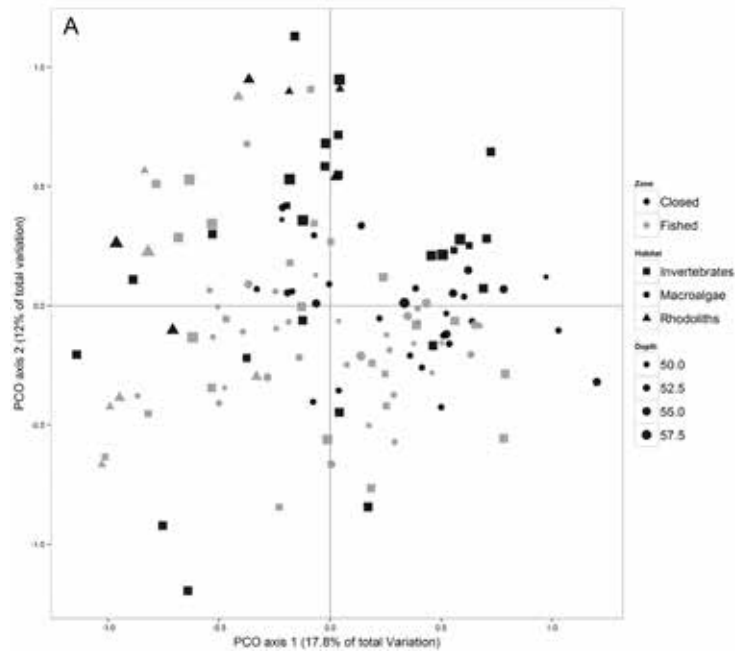
Figure 7.28. First two axes of the principle coordinate ordination (PCO) of samples from the closed and fished areas, from each habitat (sessile invertebrates, macroalgae and rhodoliths) and with depth indicated by the size of the symbol. % of total variation explained is shown for each axis. Significant effects in the PERMANOVA model are indicated.
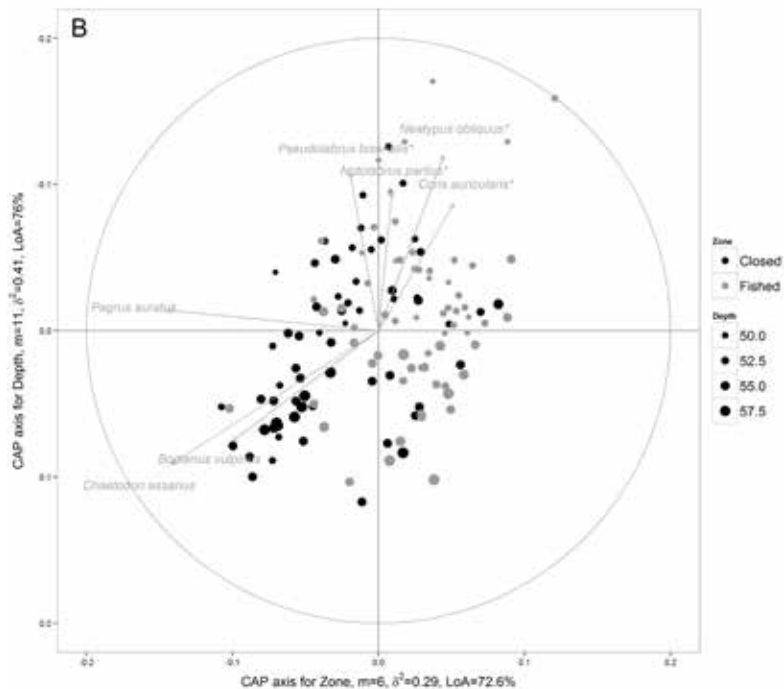
Figure 7.29. Constrained ordination for discriminant Canonical analysis of principle coordinates (CAP) between areas and correlative CAP for depth. depth indicated by the size of the symbol.
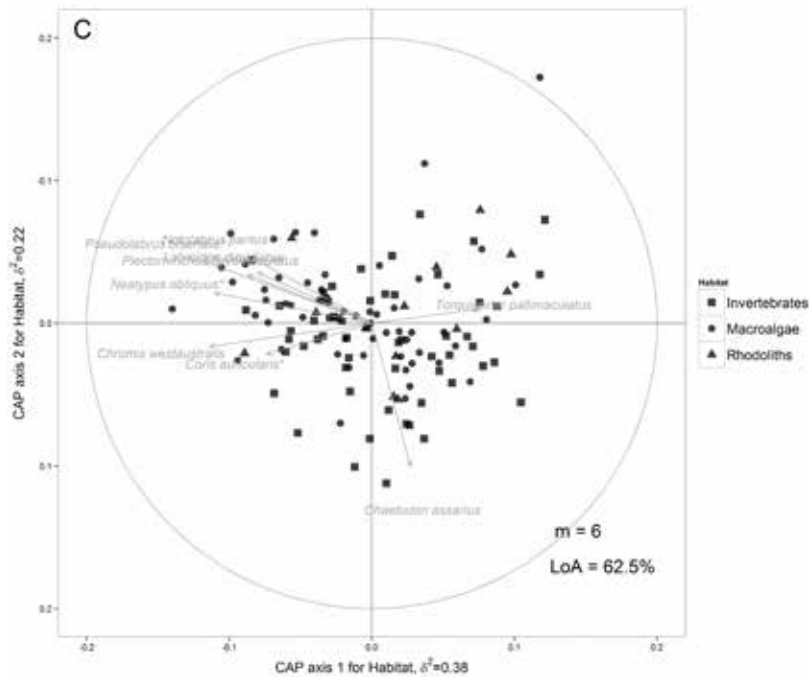
Figure 7.30. Discriminant CAP ordination for habitat (sessile invertebrates, macroalgae and rhodoliths). Species correlated (>0.4) with both canonical axes are represented as vectors and the circle represents a Spearman Rank correlation of 1, the proposed indicator species are indicated (*). The number of PCO axes used in each CAP is indicated (m) along with the squared canonical correlation (δ2) and the leave-one-out-allocation success (LoA).
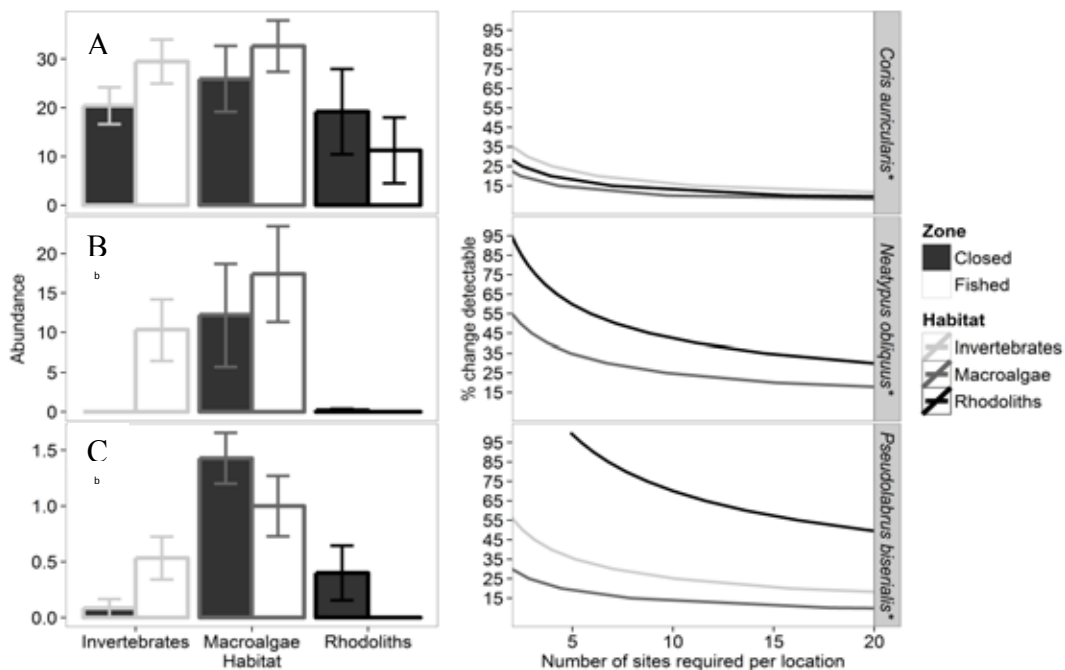
Figure 7.31. Average abundance and cost-benefit optimization of the five proposed indicator species (*) and *Pagrus auratus* for each Habitat sampled. Significant effects in the ANOVA model are indicated along with the results of pair-wise comparisons, with significant differences indicated with lower case letters.
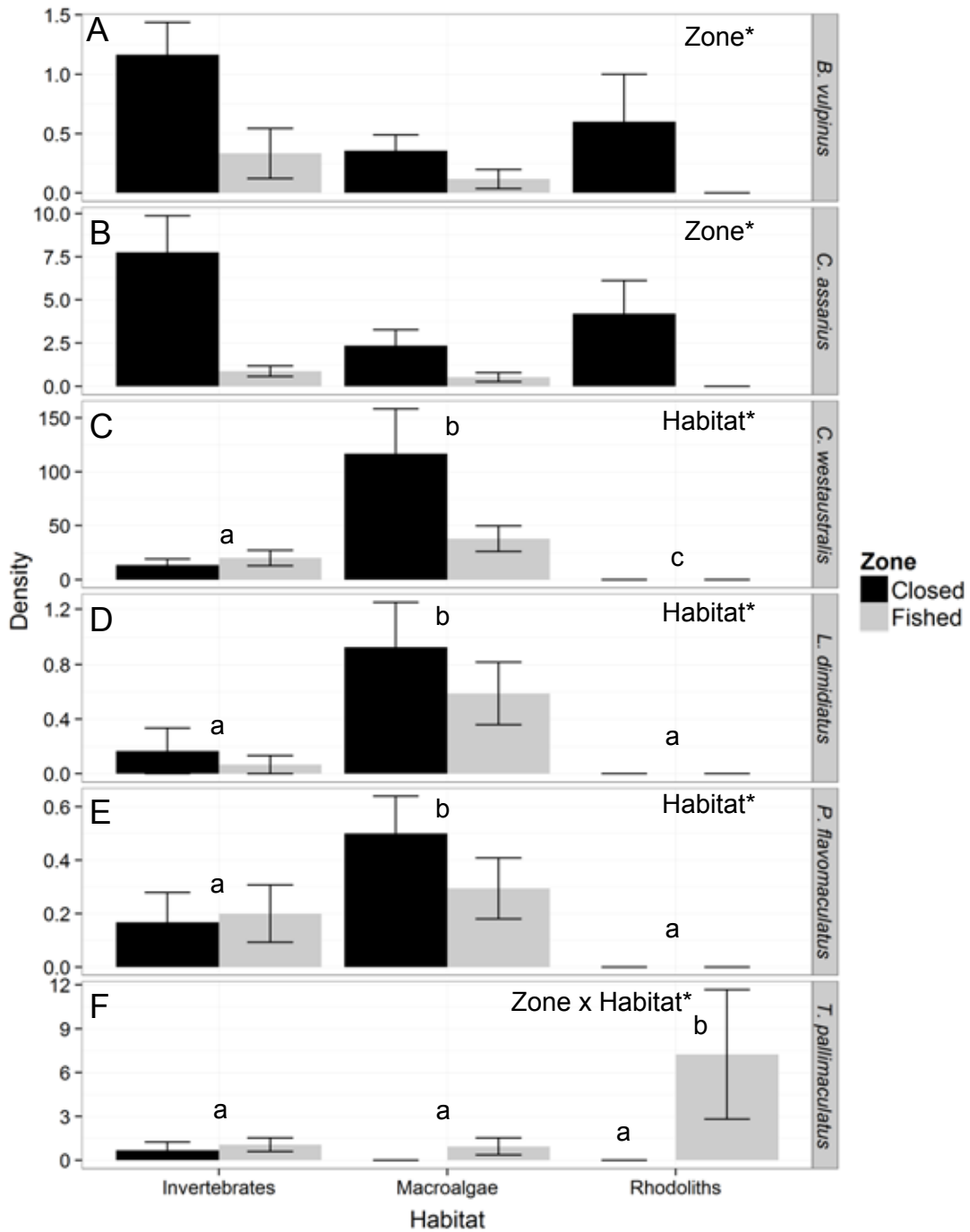
Figure 7.32. Average abundance of non-indicator species found to be strongly correlated (>0.4) with any of the CAP axes. Significant effects in the ANOVA model are indicated along with the results of pair-wise comparisons, with significant differences indicated with lower case letters.

Table 7.11 A. PERMANOVA results for assemblage composition and B. pair-wise comparisons between Habitat. Significant P values for effects are shown in bold

A. Assemblage composition					B. Pair-wise comparison of Habitat		
Source	df	MS	F	P	Comparison	t	P
Depth	1	5.655	3.829	**<0.001**	Macroalgae vs. Invertebrates	1.473	**0.032**
Area	1	5.918	3.212	**0.001**	Macroalgae vs. Rhodoliths	2.481	**<0.001**
Habitat	2	4.973	3.391	**0.000**	Invertebrates vs. Rhodoliths	1.724	**0.008**
DexZo	1	0.748	0.528	0.894			
DexHa	2	1.915	1.370	0.143			
ZoxHa	2	1.440	1.013	0.427			
Si(HaxZo)	61	1.424	1.099	0.247			
DexZoxHa	1	0.707	0.545	0.760			
DexSi(HaxZo)	20	1.102	0.850	0.822			
Res	36	1.296					
Total	127						

Table 7.12. ANOVA results for species density, total abundance, four of the five proposed indicator species (*) and other taxa found to be strongly correlated with the CAP axes (*E. armatus** was not analysed as it was only found at one site). Significant P values for effects are shown in bold.

Source	df	Species density MS	F	P	Total abundance MS	F	P	C. westaustralis MS	F	P	C. auricularis* MS	F	P
Area	1	64.27	2.691	0.103	9782	1.279	0.258	4343	0.905	0.370	1477	2.423	0.122
Habitat	2	369.56	18.816	**<0.001**	83257	11.335	**<0.001**	39473	8.315	**0.002**	631	1.030	0.369
ZoxHa	2	33.146	1.709	0.188	5144	0.731	0.482	7532	1.587	0.241	377	0.615	0.543
Si(HaxZo)	62	19.034	1.309	0.170	7129	0.982	0.507	4530	0.851	0.609	625	1.076	0.421
Res	60	14.538			7259			5324			581		
Total	127												

Source	df	N. obliqus* MS	F	P	C. asarius MS	F	P	N. parilus* MS	F	P	P. aratus MS	F	P
Zo	1	7.609	0.018	0.898	259.640	18.434	**<0.001**	0.017	0.063	0.806	306.900	15.785	**<0.001**
Ha	2	2348.200	5.456	**0.008**	47.590	3.193	0.054	2.716	9.886	**<0.001**	5.121	0.264	0.767
ZoxHa	2	563.030	1.308	0.294	23.513	1.578	0.223	0.154	0.562	0.581	2.298	0.118	0.881
Si(HaxZo)	62	416.980	0.895	0.598	18.326	3.153	**0.006**	0.264	0.872	0.617	19.221	0.967	0.534
Res	60	466.080			5.813			0.303			19.874		
Total	127												

7.3.2 Discussion

7.3.2.1 Lobster

7.3.2.1.1 Shallow water comparison

The use of pots to sample lobster in mid depths (*ca.* 20m) off Leeman provided more demographic data than visual census using scuba over a comparable time frame. Even at mid depths *ca* 20 m, visual census was not cost effective and costs would increase in deep water study areas (45-85 m). Visual census did reveal the presence of some very small lobsters (*ca* 25mm CL) at the study site that were not caught in either commercial or meshed pots but is unlikely that lobsters of this size would be attracted to or retained in pots. The size distributions of larger lobsters observed with visual census were comparable to the potting data. This is consistent with the results of a study conducted in shallow water at Rottnest which demonstrated that potting produces data directly comparable to visual census when a combination of both commercial and meshed pots are used (How and de Lestang, unpub. data). The benefit of adding meshed pots to any lobster sampling regime was also demonstrated during this snap shot study in mid depths at Leeman (15-25m), as meshed pots retained lobsters between 44 and 72 mm which would not have been caught with commercial pots alone. The difference would presumably not have been as great if the escape gaps of the commercial pots were closed as they were in the deep water sampling.

7.3.2.1.2 Abundance and demographics of lobsters in deep water

There was an increase in lobsters in deep water over the five year period of this study, at all sites sampled. The sampling period (2008-2012) corresponds with a period of below average recruitment and substantial management changes in the fishery, including a move from input control to individual quotas. The general increase in lobster numbers in deep water demonstrates the success of management initiatives to retain mature biomass. Over this time there has been a substantial shift in the demographics of lobsters toward a deep water population comprised of large mature lobsters.

Catch rates of lobsters in both closed and fished areas increased over the sampling period but the increase was larger in the closed area suggesting that the closure has led to an increase in lobster abundance in the deep water closed area. While the difference between closed and fished could be observed in the total abundance of lobsters, it was more distinct for lobsters over the legal size and particularly legal sized males. This finding is consistent with a closure effect as the fishery selects for mature males. Mature females can only be retained at certain times of the year when they are not reproductive. Caution is required when interrupting these results as the closed area has only been gazetted since March 2011. However, these results serve to highlight the importance of the continued closure and monitoring of the area in providing useful data to assist with assessing the potential ecosystem effects of fishing.

The size and sex compositions of lobsters at Jurien and Leeman were similar between 2008 and 2011, suggesting that the sites were comparable in terms of suitability for lobsters. However, in 2012 there was a substantial recruitment of sub-legal sized lobsters at the Jurien sites which was not observed in the Leeman closed area. This size class corresponds with near average settlement of juveniles two years previous. The failure of this size class to be observed in the closed area is unlikely to be related to available habitats as sampling with both commercial and meshed pots involved multiple habitats including vegetated reef which has been shown to be important for sub-legal lobsters (Bellchambers, 2010, Bellchambers *et al.* 2010). It is

also unlikely to be explained by the small difference in distance from the coast as there were some small lobsters observed in the Leeman L1 (fished) areas directly to the north and south of the closed area. It is more likely that either a) few sub-legal sized lobsters recruited into the closed area due to density dependent factors or b) the high number of lobsters, particularly large males may deter sub-legal lobsters from entering pots. There has been little research into size or density dependent catchability and further analysis of the demographic composition of individual pot catches may help resolve this.

Lobster distribution modelling

The species distribution model (SDM) produced in this study for the western rock lobster was based on the presence/absence of lobster in pot catches prior to the introduction of the closed area. It is common for species distribution models (SDMs) to use presence/absence data to predict the distribution of a species as a first step to understanding species presence and absence in a landscape (e.g. in this case, seafloor habitats).

More than 60% of the variation in lobster distribution could be explained by depth and geophysical variables, suggesting that depth and geomorphology are the principle drivers of lobster distribution in the Leeman study area.

While a strong relationship between lobster presence and the presence of reef and *Ecklonia radiata* (kelp) has been demonstrated recently (Bellchambers 2010, Bellchambers *et al.* 2010), the current lobster model did not select these variables. However, the variables selected in the model are ecologically relevant to reef and kelp distribution, reflecting important physiological (e.g., light requirements), environmental or ecological (e.g., hard substrates to attach holdfasts) limitations that drive their distribution.

The significance of the current model only selecting geophysical variables is that only bathymetric data is needed to predict lobster distribution; the distribution of reef and kelp habitats are not required, therefore eliminating the considerable investment of time and money that goes into collecting benthic habitat information via towed video and the modelling of benthic habitats.

Thus future work will investigate how these relationships can be extrapolated over larger spatial scales using existing, but coarser resolution, bathymetric data sets. In addition, the fit of the Leeman lobster SDM to a) lobster catch data collected for Jurien and b) Leeman post closure need to be assessed.

Lobster diets in deep water and the utility of next generation sequencing

Crustaceans, particularly crabs, made by far the largest contribution to the diets of deep water lobsters in this study, irrespective of season or lobster size. Crabs were also the most commonly occurring dietary item in the only other study of deep water lobster diets by Waddington *et al.* (2008) who suggested that deep water lobster tended to be more carnivorous than lobsters in shallow water. The importance of crabs in the diets of deep water lobsters is consistent with recent laboratory feeding trials conducted by Dumas *et al.* (2013) who demonstrated that sub-legal, medium and large (<115mm CL) *P. cygnus* all show a preference for crab prey. Dumas *et al.* (2013) were able to demonstrate that the type and size of the prey selected and consumed was related to the size of the lobsters and that large lobsters had a preference for large mussels in addition to crabs. These findings suggest that large lobsters could presumably have a stronger influence on trophic structure than their smaller counterparts if they occur in high concentrations.

While the frequent occurrence of crabs suggests they are an important prey item, visual analysis can tend to over-estimate the contribution of hard bodied prey such as crabs due to their long retention times in the foregut (Waddington 2008). This is also likely to be true of gastropods which were the next most common prey taxa consumed. Gastropods may be over-represented as they were identified by shell which could be contributed by other means i.e. hermit crabs or sediment.

In contrast to the previous deep water dietary study of Waddington *et al.* (2008), only a relatively small contribution was made by small invertebrates such as isopods and amphipods. This difference is presumably related to differences in the foraging habitats of the lobsters prior to capture including the fact that none appeared to have been feeding on bait, or had presumably been foraging in or near lobster pots as was the case in the previous study. It may also reflect the larger size of the lobsters, in general, in this study.

There was also a very high percentage (15.5) of stomach contents that could not be identified. In contrast, for example, Macarthur *et al.* (2011) only recorded around 2.0 − 6.0% unknown. The difference is most likely related to not being able to control the time of sample collection in the present study, i.e. being able to restrict samples analysed to those collected first thing in the morning and had presumably actively fed in the proceeding hours. Foregut contents from lobsters that were collected later in the day or entangled for an extended period would have more advanced decomposition.

This study also explored the possible application of a molecular technique, i.e. next generation sequencing (NGS), as a tool to assess trophic questions relating to lobster. In the preliminary examination undertaken in this study, NGS was only successful in recording the presence of selected crustaceans and fish. A number of major taxa known to be present in the samples through visual analysis were missed by NGS. The presence or absence of taxa recorded using NGS depends on the successful amplification of specific portions of molecular material. It is therefore dependent on a) genetic material of a suitable quality, b) the use of primers appropriate to discriminate the taxa present c) suitable sequences archived in a reference database against which to compare results (Deagle *et al.* 2007, Pompanon *et al.* 2012).

The genetic information available for replication in these dietary samples may have been quite badly degraded as their origin was a lobster's foregut, they had varying degrees of digestion and sample preparation did not happen straight away. In a more targeted study, timing of sample collection and preparation methods may make a difference (Deagle *et al.* 2007). Inaccuracy surrounding detection of molluscs and echinoderms may be explained by the biases of visual survey to identify the hard parts of these animals that may have a higher retention time in the foregut and may be present after any genetic material has been digested.

Many of the primers tested were possibly not targeted enough to identify appropriate taxa. A larger study would likely involve the creation of specific sets of primers. The primers that did work worked well, for example the 16S degenerate primers which are designed to amplify DNA from fish species, were able to identify the fish prey to species level. The fact that two typically large fish, the Dhufish *Glaucosoma hebraicum* and the Sandbar shark *Carcharhinus plumbeus* were identified in the diet samples suggests that lobsters may have been scavenging from fish entangled in the demersal gill net prior to becoming entangled themselves. While the lobster samples collected from gill net bycatch may be better for dietary studies than pot caught samples, the use of NGS has demonstrated that these samples may not be as unbiased as originally assumed. Similarly, amplification of human DNA is more likely a reminder to promote good laboratory practices rather than an indication of lobsters predating on fishermen.

Given the primers tested, NGS was effective at identifying crabs, a number to the species or genus level. As described above crabs are obviously an important prey item for lobsters and NGS may be an appropriate and cost-effective technique for screening large numbers of lobster dietary samples to assess patterns in predation of certain crab taxa. However there is not a lot of information available on the abundance of crab species in deep water and how they may be influenced by changes in lobster demographics. The utility of this technique will only improve as the gaps in barcode reference databases are filled for Western Australian species of molluscs and crustaceans, something that should be an additional aim of any future studies.

7.3.2.2 Benthic assemblages

The use of the AUV during this study enabled the collection of benthic information at multiple scales. 1) Collection of high resolution imagery which can be analysed using traditional point analyses techniques provides community composition data at the level of species in some cases. 2) The high confidence in the location information accompanying each image means that imagery can be spliced together to form geo-referenced mosaics which inform at a patch scale, i.e. form, slope, rugosity. 3) Overlaying this information on high resolution bathymetry and habitat maps provides insight into how the surrounding landscape may influence patterns in assemblage structure and dynamics.

The analysis undertaken in this study concentrated at the image level, providing baseline information on the composition of the benthic community at 12 sites, four of which are located inside the closed area. Overall, the assemblage composition was relatively similar between different sites with all sites containing a mixture of algae and invertebrates. As the kelp *Ecklonia* was the most abundant taxa, its presence or absence tended to drive much of the observed variation between the sites. However, transects with either low or high *Ecklonia* cover were balanced across both fished and unfished sites. Similarly, articulated red coralline alage, a known food source for western rock lobster (Waddington *et al.* 2008), was present in almost all sites.

The greatest benefit of using the AUV to sample benthic assemblages is that each site is geo-referenced and therefore the precise location can be revisited and resampled (Smale *et al.* 2012). This adds power to any future analyses investigating possible changes in assemblage structure. Each precisely mapped area (of 625 m^2) can be followed through time, negating to some degree the need for excessive replication that would be required to overcome small scale patchiness which is inherent in macroalgal assemblages (Smale *et al.* 2010). These analyses do not have to be limited to assemblage composition but could concentrate on certain life history stages such as the density estimates derived in this study for *Ecklonia* recruits. Similarly, revisiting precise locations enables individual organisms or colonies to be followed through time and the stereo image pairs generated by the AUV allow growth estimates to be generated (Smale *et al.* 2012).

Smale *et al.* (2012) were able to successfully revisit transects over multiple years with the same AUV and demonstrated the data produced provided the power required to detect relatively small changes (<35%) in assemblage structure. It would therefore make sense to include the sites sampled in this study into any long-term program designed to monitor for ecosystem change (Williams *et al.* 2012). Cost-effectiveness could be increased if sampling frequency was reduced once an initial understanding of annual variability was established (Hewitt and Thrush 2007).

7.3.2.3   Fish indicators

The fish assemblages within the two areas (Leeman closed vs Jurien fished) were comparable, however, three species were a significantly more abundant in the closed area. Two of these species the pigfish *Bodianus vulpinus* and the butterflyfish *Chaetodon assarius* were associated with deeper sites within the closed area. Whereas the snapper *Pagrus auratus* was found to be approximately twice as abundant across all habitats and depths in the closed area compared to the reference area.

Pagrus auratus has been recognised as a high risk demersal fisheries species in the region and is currently assessed recovering with recommendations that catches should be reduced by 50% (Fairclough *et al.* 2012). The fished area is also closer to Jurien township and boat ramp therefore less fishing effort for *P. auratus* may occur in the closed area resulting in higher relative densities. Equally the increased abundance of snapper in the closed area may be due to habitat availability or a number of other factors.

The current study found strong habitat associations in the fish assemblage and three of the five small bodied invertivore indicator species, highlighted by qualitative modelling (Objective 2), were found to be more abundant in macroalgal dominated sites. Cost-benefit analyses suggested that smaller percentage changes in abundance would be detectable with fewer sites if sampling for these species focused on macroalgal dominated sites.

The western king wrasse was the most abundant species across all habitat types and areas and cost-benefit analyses suggested changes in this indicator species would be the first to be detected by any monitoring program. However, one of the proposed indicator species, the old wife *Enoplosus armatus*, was found in very low abundances and therefore very unlikely to be a useful indicator species.

Our baseline study confirms the supposition made by Metcalfe *et al.* (2011) and suggests that several small bodied invertivore fish species can be effectively monitored across these deeper water habitats but also finds that, given strong habitat associations, macroalgae dominated sites are likely to be the most cost-effective to monitor.

The strong habitat associations and species specific patterns characterised by the current study can help design future studies to investigate patterns and processes influencing the structure of these deeper water fish assemblages. For studies focusing on the ecosystem effects of rock lobster fishing, the small bodied invertivore and wrasse species are likely to be most cost-effectively studied in macrolagal dominated habitats.

No evidence of ecosystem effects of fishing for demersal fish species has been found by previous correlative studies in the region (Langlois *et al.* 2012). However, if any changes in populations of *P. auratus* occur in the areas of interest, cost-benefit analyses suggest they will be relatively easy to detect.

The closure of one area only to rock lobster fishing, should allow the ecosystem effects of that activity to be separated from ecosystem effects due to other forms of fishing.

8.0   Benefits and adoption

This project has established a closed area within the western rock lobster fishery to address concerns relating to possible ecosystem effects of removing lobster biomass from the deep water (40-100 m). The research conducted in the closed area will, over time, provide the ecological knowledge required to detect changes to the level of risk to deep water communities associated with removing lobster biomass. The major outcome of this research will be the continued accreditation of the WRLMF by the MSC. In addition, the information will aid the maintenance of Department of Environment certification for the export of lobster product.

A key benefit of this research will be an improved understanding of the ecosystem that supports the western rock lobster. These types of information will be critical in improving the Department of Fisheries ability to manage the fishery in an ecosystem based manner.

The implementation of a closed area also provides a unique opportunity to study aspects of lobster population dynamics without fishing pressure, which may include research into lobster growth, natural mortality, population interaction and carrying capacity. An increased understanding of lobster population dynamics and lobster interactions with the ecosystem will lead to better management of the fishery as a whole, particularly in the face of increased environmental variability and climate change. For example, the closed area maybe valuable in assessing the effect of the recent series of below average puerulus settlement and subsequently effort reduction in the fishery.

An additional outcome of this project has been the development of cost-effective methods and techniques to monitor the structure and function of deep water ecosystems. Through the refinement of these sampling techniques this project will contribute in developing a financially sustainable, long-term programme to monitor the ecosystem attributes of the closed area beyond the life of the project.

9.0   Further Development

The sampling techniques outlined in Chapter 6 and implemented during this study have been successful in providing a baseline against which to quantify ecosystem change. Lobster populations have been observed to have already undergone some changes in abundance and size structure within the closed area. However, in order to understand if these changes in lobster demographics are enduring or have significant ecological consequences, monitoring of lobsters, benthic assemblages and any potential indicators needs to continue as part of a long-term monitoring plan.

The ability to continue to monitor lobsters in the closed area will also give researchers the opportunity to explore various biological questions for an unexploited population. For example, are there density dependent differences in growth or mortality of mature lobsters? The results of this continuing work will be used to parameterise stock assessment models and ultimately lead to improved management of the fishery.

This project combined geo-referenced lobster catch data and high resolution geo-physical data to produce a predictive model of lobster occurrence. In this work bathymetric data sets alone accounted for <60% of variation in lobster occurrence. Future modelling work will attempt to incorporate backscatter outputs from the multibeam data into the models to further refine relationships between lobsters and habitat. The lobster distribution model outlined in this study was based on presence/absence data. Work is currently underway to create models that incorporate size structure, sex and abundance data. The presence/absence data was just for the Leeman area from the period prior to closure. Thus future work will also assess the fit of the Leeman lobster SDM to a) lobster catch data collected for Jurien and b) Leeman post closure.

This work comprises the first step in a strategy to produce a fishery wide understanding of the relationship between lobsters and benthic habitats. The next step is to investigate how these relationships can be extrapolated over larger spatial scales using bathymetric data sets of various resolutions. For example, how fine does bathymetric data need to be to provide meaningful insight into species distributions.

Beyond providing a baseline of benthic assemblages against which to quantify ecosystem change, use of the AUV has produced an inexhaustible source of potential data, as illustrated by the ability to derive density estimates for *Ecklonia* recruits during this study. Similarly, as images are collected in stereo pairs, individual organisms can be measured and their growth potentially estimated over time. The images collected during this project are currently being analysed for lobsters. The size and number of lobsters in individual frames can be related to habitat types and key geographic features. Another key output of the AUV work has been the highly precise multibeam sonar and habitat information it provides which can be used to validate the habitat maps

10.0   Planned Outcomes

This project put in place the research framework required to address concerns relating to possible ecosystem effects of removing lobster biomass from the deep water (40-100 m). The research framework provides the ecological knowledge required to reassess the level of risk to deep water communities associated with removing lobster biomass. The major outcome of this research is the continued accreditation of the WRLF by the MSC with outputs of the project contributing towards the 2013 annual surveillance audit and Ecological Risk Assessment (ERA).

In addition, a key benefit of this research is an improved understanding of the ecosystem that supports the western rock lobster. This project has provided a unique opportunity to study aspects of lobster population dynamics under different levels of exploitation including research into lobster growth, natural mortality, population interaction and carrying capacity. An increased understanding of lobster population dynamics and lobster interactions with the ecosystem will lead to better management of the fishery as a whole, particularly in the face of increased environmental variability and climate change.

An additional outcome of this project has been the development of cost-effective methods to monitor the structure and function of deep water ecosystems. Through the refinement of these sampling techniques this project will contribute to developing a financially sustainable, long-term endeavour to monitor the ecosystem attributes of the closed area beyond the life of the project.

The project results have been communicated in a number of peer reviewed journal articles, at national and international conferences, industry and advisory group meetings and have formed an integral part of the MSC and ERA process for the fishery.

11.0   Conclusion

The need for ecological data was highlighted by the ERA in 2007, where the effect of removing lobster biomass in deep water (> 40m) was given a "moderate" risk. The risk was based primarily on a precautionary approach due to lack of scientifically defensible data on the potential ecosystem impacts of lobster biomass removal due to fishing. It was also noted that re-assessment of the level of risk required the gaps in the knowledge of deep water ecosystems to be addressed. Research undertaken during FRDC 2004/049 increased the understanding of the relationship between lobster and their deep water habitats; however the approach was ineffective in addressing MSC requirements. Therefore, the Ecosystem Scientific Reference Group (EcoSRG) concluded that there was a need for future research in deep water to use research closures. Therefore the aim of this project was to provide scientifically defensible data which would assist to lower the risk in future reassessments by establishing an area closed to lobster fishing and establishing a long term monitoring program.

Objective 1: Identification and assessment of suitable unfished reference areas to exclude rock lobster fishing in deep water

A deep water closed area was identified, negotiated and implemented to enable the potential impacts of lobster fishing on deep water ecosystems to be assessed by comparing areas of contrasting lobster abundance.

The involvement of commercial fishers through the creation of an industry-based closed area working group was instrumental in the closed area identification and negotiation process. Initially, the industry working group assessed a number of potential areas against various selection criteria. Two areas were short-listed as potentially favourable, i.e. the south eastern corner of the Abrolhos zone and the 30°S latitude line offshore from Leeman. Video surveys were undertaken to validate the representative of the benthic habitats at each site. On the basis of the habitat information, the Leeman site was deemed most representative and surveys of lobster demographics were then undertaken. An area straddling the finally 30°S latitude line offshore *ca.* 3900 ha (6 nm x 2 nm) was closed to lobster fishing on the 15 March 2011.

Objective 2: Development of a qualitative trophodynamic model that will provide a conceptual framework for determining sampling protocols, indicators and targets

Qualitative models were developed for the west coast deep water ecosystem that supports the fishery for the western rock lobster (*Panulirus cygnus*) in an attempt to identify potential indicators of ecosystem change that would be both cost effective to monitor and capable of detecting change.

The models predicted that lobster fishing may positively impact small fish, such as old wife (*Enoplosus armatus*), footballer sweep (*Neatypus obliquus*), and king wrasse (*Coris auricularis*). These small fish were therefore identified as potential indicators of the effects of rock lobster fishing. Small crustaceans (amphipods and isopods) were also identified as potential indicators of bait effects. Therefore the models indicated that monitoring small fish and crustaceans may detect ecosystem change caused by the rock lobster fishery.

Objective 3: To provide cost effective methods to measure deep water ecosystems in both fished and unfished reference areas

This project trialled different sampling methods to assess their ability to detect change at the required spatial and temporal scales and to develop a cost effective long term monitoring

program. To establish ecosystem baselines and devise a cost-effective strategy for long term monitoring the components to be monitored were divided into three categories; target species (lobsters), benthic habitats and indirect ecosystem indicators (small fish)

The lobster monitoring regime implemented has demonstrated that protection from fishing has led to a marked increase in the abundance of lobsters in the closed area. The increase was primarily attributable to mature lobsters, particularly large males which would have otherwise been available for capture.The increases in abundance have occurred in a short time; less than two years, and in the case of males, after only three months of protection. It is likely that in the future the closed area will provide the anticipated contrast in lobster abundance and size structure required to fully explore the potential ecological consequences of lobster fishing. Therefore, it is important to continue to monitor lobsters in the closed area to see if the initial trends persist and to understand how management initiatives are impacting deep water lobster populations.

The ecosystem implications of altered lobster demographics may take much longer to manifest. If the removal of lobster through fishing has consequences that are reflected in the benthic assemblages the baseline collected during this study, with an AUV and full coverage habitat map inside and outside the closed area, will provide a background against which potential changes can be quantified. This work comprises the first step in a strategy to produce a fishery wide understanding of the relationship between lobsters and benthic habitats. The next step is extrapolate over larger spatial scales using bathymetric data sets of various resolutions this is an important step in not only understanding potential ecosystem impacts throughout the extent of fishery but for addressing criteria required for the ongoing certification.

The qualitative modelling suggested that small fish may be potential indicators of ecosystem change associated with removal of lobsters through fishing. The subsequent BRUV sampling provided a baseline for fish communities inside and outside the closed area and investigated capacity to quantify change in populations of fish indicators. This work confirmed that several small bodied invertivore fish species can be effectively monitored. The study also found that the indicator species displayed strong habitat associations and therefore macroalgae dominated sites were the most cost-effective to monitor. However, such indirect trophic consequences of changes in lobster abundance would presumably take time to become apparent and may be subtle. Future monitoring efforts, concentrating on macro-algal habitats would likely be repeated at multiyear timeframes to be cost-effective.

In conclusion, the project achieved its objectives of implementing an area closed to lobster fishing and has established baselines for the closed and nearby fished areas against which the potential impacts on deep water ecosystem of lobster biomass removal by fishing can be quantified. Initial lobster data from the closed area suggests a rapid increase in lobster abundance, particularly mature males. However, as ecosystem impacts of fishing can often diffuse and full impact of fishing on the ecosystem may take an extended period to manifest (i.e. >10 years) therefore it is essential that a range of ecosystem components i.e. target species (lobsters), benthic habitats and indirect ecosystem indicators (small fish), continue to be monitored through time. The success of this project is illustrated by the recent ERA assessing the risk of ecosystem impacts due to the removal of lobster biomass to deep water communities as low. This was due primarily due to the ERA technical panel having confidence that this research had increased the understanding of deep water ecosystems and has established a scientifically rigours monitoring framework capable of detecting any future changes in the ecosystem that may be attributable to fishing. This project has also been instrumental in ensuring the on-going MSC certification of the fishery by providing research to address conditions under Principal 2 of the MSC's Fishery Assessment Methodology.

12.0   References

Anderson M. J., R. N. Gorley and Clarke K. R.. 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER–E, Plymouth, United Kingdom.

Anderson, M.J. and Miller, R.B. 2004. Spatial variation and effects of habitat on temperate reef fish assemblages in northeastern New Zealand. *Journal of Experimental Marine Biology and Ecology* 305: 191–221.

Auger P., Charles, S., Viala M. and Poggiale J.C. 2000. Aggregation and emergence in ecological modelling: integration of ecological levels. *Ecological Modelling* 127: 11–20.

Babcock R.C., Kelly S., Shears N.T., Walker J.W. and 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. and 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.

Ballantine, W.J. and Langlois, T.J. 2008. Marine reserves: the need for systems. *Hydrobiologia* 606: 35–44.

Barrett N., Buxton C. and Gardner C. 2009b. Rock lobster movement patterns and population structure within a Tasmanian marine protected area inform fishery and conservation management. *Marine and Freshwater Research*. 60: 417–425.

Barrett N.S., Buxton C.D. and Edgar, G.J. 2009a. Changes in invertebrate and macroalgal populations in Tasmanian marine reserves in the decade following protection. *Journal of Experimental Marine Biology and Ecology* 370(1–2): 104–119.

Barrett N.S., Edgar G.J., Buxton C.D. and Haddon M. 2007. Changes in fish assemblages following 10 years of protection in Tasmanian marine protected areas. *Journal of Experimental Marine Biology and Ecology* 345(2): 141–157.

Bellchambers L., Bridgwood S., How J., Lewis P., de Lestang S., Mackie M. and Coutts, T. 2009. Development of a long–term program to monitor coastal communities within the Swan region. Fisheries. Research Report No. 183. Department of Fisheries, Western Australia. 144p.

Bellchambers L.M. 2010. The effect of western rock lobster fishing on the deep water ecosystems of the west coast of Western Australia. Fisheries Research Report 199. Department of Fisheries, Western Australia. 96 p.

Bellchambers L.M., Evans S.N and Meeuwig, J.J. 2010. Abundance and size of western rock lobster (*Panulirus cygnus*) as a function of benthic habitat: implications for ecosystem–based fisheries management. *Marine and Freshwater Research* 61(3): 279–287.

Bellchambers L.M., Evans S.N. and Meeuwig J. J. 2013. Assessing the effectiveness of two methods of habitat characterisation for understanding species habitat relationships, using the western rock lobster (*Panulirus cygnus* George). Fisheries Research. 139: 5–10.

Blamey L.K. and Branch G.M. 2012. Regime shift of a kelp–forest benthic community induced by an 'invasion' of rock lobster *Jasus lanlandii*. *Journal of Experimental Marine Biology and Ecology* 420–421: 33–47.

Brock D.J., Saunders T.M., Ward T.M. and Linnane A.J. 2006. Effectiveness of a two–chambered trap in reducing within–trap predation by octopus on southern spiny lobster. *Fisheries Research* 77(3): 348–355.

Budarf A.C., Burfeind D.D. Loh W.K.W. and Tibbetts I.R. 2011. Identification of seagrasses in the gut of a marine herbivorous fish using DNA barcoding and visual inspection techniques. *Journal of Fish Biology* 79: 112–121.

Buxton C.D., Barrett N.S., Haddon M., Gardner C. and Edgar G.J. 2004. Evaluating the effectiveness of marine protected areas as a fisheries management tool. Report to FRDC, TAFI, University of Tasmania, Australia.

Cale W.G., O'Neill R.V. and Gardner, R.H. 1983. Aggregation error in non–linear ecological models. *Journal of Theoretical Biology* 100: 539–550.

Cappo M., Harvey E., Malcolm H., Speare P. 2003. Potential of video techniques to monitor diversity, abundance and size of fish in studies of Marine Protected Areas. Aquatic Protected Areas–what works best and how do we know, 455–464

Christie P., Fluharty D.L., White A.T., Eisma–Osorio L., and Jatulan W. 2007. Assessing the feasibility of ecosystem–based fisheries management in tropical contexts. *Marine Policy* 31: 239–250.

Clarke K.R. and Gorley R.N. 2001. *PRIMER* v5: User manual/tutorial. Primer–E: Plymoth, United Kingdom.

Clarkston B.E. and Saunders G.W. 2010. A comparison of two DNA barcode markers for species discrimination in the red algal family Kallymeniaceae (Gigartinales) with a description of *Euthora timburtonii*. *Botany* 88: 119–131.

Cossington S. 2006. The biology of the Foxfish (*Bodianus frenchii*) in the marine coastal waters of south–western Australia. Honours Thesis, Murdoch University, Perth, Australia.

Costa D.P. and Gales N.J., 2003. Energetics of a benthic diver: seasonal foraging ecology of the Australian sea lion, *Neophoca cinerea. Ecological Monographs* 73: 27–43.

Crawley K.R., Hyndes G.R., Vanderklift M.A., Revill A.T. and Nichols P.D. 2009. Allochthonous brown algae are the primary food source for consumers in a temperate, coastal environment. *Marine Ecology Progress Series* 376: 33–44.

Cury P.M. and Christensen, V. 2005. Quantitative ecosystem indicators for fisheries management. *ICES Journal of Marine Science* 62: 307–310.

Dambacher J.M., Gaughan D.J., Rochet M.-J., Rossignol P.A. and Trenkel, V.M. 2009. Qualitative modelling and indicators of exploited ecosystems. *Fish and Fisheries* 10(3): 305–322.

Dambacher J.M., Li H.W. and Rossignol P.A. 2002. Relevance of community structure in assessing indeterminacy of ecological predictions. *Ecology* 83:1372–1385.

Dambacher J.M., Luh H.K., Li H.W. and Rossignol P.A. 2003. Qualitative stability and ambiguity in model ecosystems. *The American Naturalist* 161(6): 876–888.

De Jonge V.N., Elliott M. and Brauer V.S. 2006. Marine monitoring: its shortcomings and mismatch with the EU Water Framework Directive's objectives. *Marine Pollution Bulletin* 53(1–4): 5–19.

Deagle B.E., Gales N.J., Evans K. *et al.* 2007. Studying seabird diet through genetic analysis of faeces: a case study on macaroni penguins (*Eudyptes chrysolophus*). *PLoS One*, 2, e831

Department of Fisheries 2012. A Report of the Effects of Fishing Advisory Group (EFAG) Meeting (2–3 November 2010) and The Western Rock Lobster Ecological Effects of Fishing Research Plan Revised and Updated by the Effects of Fishing Advisory Group (2 – 3 November 2010). Fisheries Occasional Publication 91. Department of Fisheries Western Australia.

Dumas J-P, Langlois T.J., Clarke K.R. and Waddington K.I. 2013. Strong preference for decapod prey by the western rock lobster Panulirus cygnus. *Journal of Experimental Marine Biology and Ecology* 439:25–34.

Edgar G.J. 2000. Australian Marine Life: the plants and animals of temperate waters. Reed New Holland, Sydney, Australia.

Fairclough D., Lai E., Sawyer M., Holtz M. and Nicholas T. 2012.West Coast Demersal Scalefish Resource Status Report In : Status Reports of the Fisheries and Aquatic Resources of Western Australia 2011/2012: The state of the Fisheries eds. W.J. Fletcher and K. Santoro, Department of Fisheries Western Australia pp 100–105.

FAO 2011. Fisheries Management 4. Marine protected areas and fisheries. FAO technical guidelines for responsible fisheries. No. 4, Supplement 4, Rome, 198p.

Fletcher W.J. 2005. The application of qualitative risk assessment methodology to prioritise issues for fisheries management. *ICES Journal of Marine Science* 62: 1576–1587.

Fletcher W.J. and Santoro K. 2009. *State of the fisheries report 2007/2008*. Department of Fisheries, Government of Western Australia, Perth, Australia. 308 p.

Fletcher W.J., Chubb C., McCrea J., Caputi N., Webster F., Gould R. and Bray T. 2005. ESD Report Series No. 4. Western rock lobster fishery. Department of Fisheries, Western Australia, Perth.

Fletcher W.J., Shaw J., Metcalf S.J. and Gaughan D.J. 2010. An Ecosystem Based Fisheries Management framework: the efficient, regional–level planning tool for management agencies. *Marine Policy* 34(6): 1226–1238.

Folmer O., Black M., Hoeh W., Lutz R., and Vrijenhoek R. 1994. DNA primers for amplification of mitochondrial cytochrome c oxidase subunit I from diverse metazoan invertebrates. *Molecular Marine Biology and Biotechnology* 3: 294–299.

Froese R. and Pauly D. (eds) (2012). Fishbase. Electronic publication. www.fishbase.org, version. (Accessed 12/2012).

Fulton E.A., Smith A.D.M., and Johnson C.R. 2003. Effect of complexity on marine ecosystem models. *Marine Ecology Progress Series* 253: 1–16.

Gardner R.H., Cale W.G., and O'Neill R.V. 1982. Robust analysis of aggregation error. *Ecology* 63: 1771–1779.

Halpern B.S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? *Ecological Applications* 13: 117–137.

Harvey E.S., Cappo M., Butler J.J., Hall N. and Kendrick G.A. 2007. Bait attraction affects the performance of remote underwater video stations in assessment of demersal fish community structure. *Marine Ecology Progress Series* 350: 245–254

Harvey E.S. and Shortis M.R. 1996. A system for stereo–video measurement of sub–tidal organisms. *Marine Technology Society Journal* 29(4): 10–22.

Hayes K.R., Lyne V., Dambacher J.M., Sharples R. and Smith R. 2008. Ecological indicators for the exclusive economic zone waters of the south west marine region. Final report (08/82) for the Australian Government, Department of the Environment and Heritage, CSIRO Marine and Atmospheric Research, Hobart. 152 p.

Hewitt J. and Thrush S. 2007. Effective long–term ecological monitoring using spatially and temporally nested sampling. *Environmental Monitoring and Assessment* 133: 295–307.

Hill S.L., Watters G.M., Punt A.E., McAllister M.K., Le Quere C. and Turner J. 2007. Model uncertainty in the ecosystem approach to fisheries. *Fish and Fisheries* 8: 315–336.

Holmes K.W., Van Niel K.P., Radford B., Kendrick G.A. and Grove S.L. (2008). Modelling distribution of marine benthos from hydroacoustics and underwater video. *Continental Shelf Research* 28(14): 1800–1810.

Hosack G.R., Hayes K.R. and Dambacher J.M. 2008. Assessing uncertainty in the structure of ecological models through a qualitative analysis of system feedback and Bayesian belief networks. *Ecological Applications* 18: 1070–1082.

Hosack G.R., Li H.W. and Rossignol, P.A. 2009. Sensitivity of system stability to model structure. *Ecological Modelling* 220(8): 1054–1062.

Hovey R.K., Van Niel K.P., Bellchambers L.M. and Pember M.P. 2012. Modelling deep water habitats to develop a spatially explicit, fine scale understanding of the distribution of the Western Rock Lobster, *Panulirus cygnus*. *PLoS ONE* 7(4): e34476. doi:10.1371/journal.pone.0034476.

Howard R.K. 1988. Fish Predators of the Western Rock Lobster (*Panulirus cygnus* George) in a Nearshore Nursery Habitat. *Australian Journal of Marine and Freshwater Research* 39(3): 307–316.

Hutchings P. 1998. Biodiversity and functioning of polychaetes in benthic sediments. *Biodiversity and Conservation* 7: 1133–1145.

Jackson J.B.C. and Sala E. 2001. Unnatural Oceans. *Scientia Marina* 64(S2): 273–281.

Jernakoff P., Phillips B.F. and 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.

Joll L.M. 1977. The predation of pot–caught western rock lobster (*Panulirus longipescygnus*) by octopus. Western Australian Department of Fisheries and Wildlife Report 29: 1–58.

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

Kailola,P.J., Williams,M.J., Stewart,P.C., Reichelt,R.E., McNee, A. and Grieve C. 1993. Australian Fisheries Resources. Bureau of Resource Sciences, Department of Primary Industries, and Fisheries Research and Development Corporation. Canberra, Australia.

Kimmins J.P., Blanco, J.A., Seely B., Welham C. and Scoullar, K. 2008. Complexity in modelling forest systems: How much is enough? *Forest Ecology and Management* 256: 1646–1658.

Kohler K. E. and Gill S. M. 2006. Coral Point Count with Excel extensions (CPCe): a Visual Basic program for the determination of coral and substrate coverage using random point count methodology. *Computers and Geosciences*, 32: 1259–1269.

Kremen C. 1992. Assessing the Indicator Properties of Species Assemblages for Natural Areas Monitoring. *Ecological Applications* 2(2): 203–217. doi: 10.2307/1941776

Langlois T.J., Harvey, E.S. and Meeuwig, J.J. 2012. Strong direct and inconsistent indirect effects of fishing found using stereo–video: Testing for indicators from fisheries closures. *Ecological Indicators* 23: 524–534.

Langlois T.J., Harvey E.S., Fitzpatrick B., Meeuwig J.J., Shedrawi G. and Watson D.L. 2010. Cost–efficient sampling of fish assemblages: comparison of baited video stations and diver video transects. *Aquatic Biology* 9 (2): 155–168

Last P.R. and Stevens J.D. 1994. Sharks and Rays of Australia. CSIRO, Australia.

Lek E. 2004. Diets of three carnivorous fish species in marine waters of the west coast of Australia. Honours Thesis, Murdoch University, Perth.

Levins R. 1974. The qualitative analysis of partially specified systems. *Annals of the New York Academy of Science* 231: 123–138.

Levins R. 1975. Evolution in communities near equilibrium. *Ecology and Evolution of Communities*. pp. 16–50.

Ling S.D., Johnson C.R., Frusher S.D. and Ridgway K.R. 2009. Overfishing reduces resilience of kelp beds to climate–driven catastrophic phase shift. *Proceedings of the National Academy of Sciences* 106: 22341–22345.

Link J.S., Yemane D., Shannon L.J., Coll M., Shin Y.–J., Hill L. and Borges M.F. 2009. Relating marine ecosystem **indicator**s to fishing and environmental drivers: an elucidation of contrasting responses. *ICES Journal of Marine Science* 62: 000–000. doi: 10.1093/icesjms/fsp258

Lozano–Montes H.M., Babcock R. and Lonergan N.R. 2012. Exploring the effects of spatial closures in a temperate marine ecosystem in Western Australia: A case study of the western rock lobster (*Panulirus cygnus*) fishery. *Ecological Modelling* 245(12): 31–40.

Luczkovich J. J., Borgatti S. P., Johnson J. C. and Everett M. G. 2003. Defining and measuring trophic role similarity in food webs using regular equivalence. *Journal of Theoretical Biology* 220: 303–321.

MacArthur L.D., Phillips D.L., Hyndes G.A., Hanson C.E. and Vanderklift M.A., 2011. Habitat surrounding patch reefs influences the diet and nutrition of the western rock lobster. *Marine Ecology Progress Series* 436: 191–205.

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

Mayfield S., Branch G.M. and Cockroft 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., and Branch G.M. 2001. Prey preference and the consumption of sea urchins and juvenile abalone by captive rock lobster (*Jasus lanlandii*). *Marine and Freshwater Research* 52: 773–780.

Melville–Smith R., Chubb C. F., Caputi N., Cheng Y. W., Christianopoulos D., *et al.* 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.

Melville–Smith R. and Chubb C.F. 1997. Comparison of dorsal and ventral tag retention in western rock lobsters, *Panulirus cygnus* (George). *Marine and Freshwater Research*. 48: 577–580.

Metcalf S.J., Dambacher J.M., Hobday A.J. and Lyle, J.M. 2008. Importance of trophic information, simplification and aggregation error in ecosystem models. *Marine Ecology Progress Series* 360: 25–36.

Metcalf S.J., Gaughan D.J. and Shaw J. 2009. Conceptual models for Ecosystem Based Fisheries Management (EBFM) in Western Australia. Fisheries Research Report No. 194. Department of Fisheries, Western Australia. 42 p.

Metcalf S.J., Pember M.B. and Bellchambers L.M. 2011. The identification of indicators of the effects of fishing using alternative models, uncertainty, and aggregation error. *ICES Journal of Marine Science* 68(7): 1417–1425. doi:10.1093/icesjms/fsr050

Meusnier I., Singer G.A., Landry J.F., Hickey D.A., Hebert P.D. and Hajibabaei M. 2008. A universal DNA mini–barcode for biodiversity analysis. BMC Genomics 12: 9–214.

Moore S.E. 2005. The biology of the Breaksea cod (*Epinephelides armatus*) in coastal marine waters of south–west Australia. Honours Thesis, Murdoch University, Perth, Australia.

Moore C.H., Harvey E.S. and Van Niel K.P. 2009. Spatial prediction of demersal fish distributions: enhancing our understanding of species–environment relationships. *ICES Journal of Marine Science: Journal du Conseil* 66: 2068–2075.

Moore S.E., Hesp A.S., Hall N.G. and Potter I.C. 2007. Age and size compositions, growth and reproductive biology of the breaksea cod *Epinephelides armatus*, a gonochoristic serranid. *Journal of Fish Biology* 71(5): 1407–1429.

Nardi, K. Newman S.J., Moran M.J. and Jones, G.P. 2006. Vital demographic statistics and management of the baldchin groper (*Choerodon rubescens*) from the Houtman Abrolhos Islands. *Marine and Freshwater Research* 57: 485–496.

Ojeda–Martinez C., Bayle–Sempere J.T., Sánchez–Jerez P., Forcada A. and Valle, C. 2007. Detecting conservation benefits in spatially protected fish populations with meta–analysis of long–term monitoring data. *Marine Biology* 151(3): 1153–1161.

Pelletier D., Claudet J., Ferraris J., Benedetti–Cecchi L. and Garcìa–Charton, J.A. 2008. Models and indicators for assessing conservation and fisheries–related effects of marine protected areas. *Canadian Journal of Fisheries and Aquatic Sciences* 65(4): 765–779.

Peng A.H. 2003. Comparisons of the diets of four species of sparid on the central and lower west coast of Australia. Honours Thesis, Murdoch University, Perth, Australia.

Platell M.E. and Potter I.C. 2001. Partitioning of food resources amongst 18 abundant benthic carnivorous fish species in marine waters on the lower west coast of Australia. *Journal of Experimental Marine Biology and Ecology* 261(1): 31–54.

Poinar H.N., Hofreiter M., Spaulding W.G., Martin P.S., Stankiewicz B.A., Bland H., Evershed R.P., Possnert G. and Pääbo S. 1998. Molecular coproscopy: dung and diet of the extinct ground sloth *Nothrotheriops shastensis*. *Science* 17;281(5375):402–6.

Pompanon F. Deagle B. E. Symondson W. O. C. Brown D S. Jarman S. N. and Taberlet P. 2012. Who is eating what: diet assessment using next generation sequencing. *Molecular Ecology* 21(8) 1931–1950.

Priede I.G., Bagley P.M., Smith A., Creasey S. and Merrett N.R. 1994. Scavenging deep demersal fishes of the Porcupine Seabight, North–east Atlantic: observations by baited camera, trap and trawl. *Journal of Marine Biological Association* UK 74:481–498

Puccia C.J. and Levins R. 1985. *Qualitative modeling of complex systems*. Harvard University Press, Cambridge, Massachusetts. 259 pp.

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

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

Sherwood A.R. and Presting G.G. 2007. Universal primers amplify a 23S rDNA plastid marker in eukaryotic algae and cyanobacteria. *Journal of phycology* 43: 605–608.

Shin Y.-J., Shannon L.J., Bundy A., Coll M., Aydin K., Bez N., Blanchard J.L., Borges M. F., Diallo I., Diaz E., Heymans J.J., Hill L., Johannesen E., Jouffre D., Kifani S., Labrosse P., Link J.S., Mackinson S., Masski H., Mollmann C., Neira S., Ojaveer H., Abdallahi K. M., Perry I., Thiao D., Yemane D. and Cury P.M. 2010. Using indicators for evaluating, comparing, and communicating the ecological status of exploited marine ecosystems. 2. Setting the scene. *ICES Journal of Marine Science* 67: 692–716.

Smale D.A., Kendrick G.A., Harvey E.S., Langlois T.J., Hovey R.K., Van Niel K.P., Waddington K.I., Bellchambers L.M., Pember M.P., Babcock R.C., Vanderklift M.A., Thomson D.P., Jakuba M.V. Pizarro O. and Williams S.B. 2012. Regional–scale benthic monitoring for Ecosystems–Based Fisheries Management (EBFM) using an Autonomous Underwater Vehicle (AUV). *ICES Journal of Marine Science*. 69(6):1108–1118 doi:10.1093/icesjms/fss082

Smale D. A., Kendrick G. A., Waddington K. I., Van Niel K. P., Meeuwig J. J. and Harvey, E. S. 2010a. Benthic assemblage composition on subtidal reefs along a latitudinal gradient in Western Australia. *Estuarine Coastal and Shelf Science* 86: 83–92.

Sumaila U.R., Guenette S., Alder J. and Chuenpagdee, R. 2000. Addressing ecosystem effects of fishing using marine protected areas. *ICES Journal of Marine Science* 57: 752–760.

Taberlet P., Coissac E., Pompanon F., Gielly L., Miquel C. *et al*. 2007. Power and limitations of the chloroplast trnL (UAA) intron for plant DNA barcoding. *Nucleic Acids Research* 35: e14.

Tegner M.J. and Dayton P.K. 1981. Population structure, recruitment and mortality of two sea urchins (*Strongylocentrotus franciscanus* and *S. purpuratus*) in a kelp forest. *Marine Ecology Progress Series* 5: 225–268.

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

Underwood A.J. 1981. Techniques of analysis of variance in experimental marine biology and ecology. *Annual Reviews of Oceanography and Marine Biology* 16: 513–605.

Vanderklift M.A. 2002. Interactions between sea urchins and macroalgae in southwestern Australia: testing general predictions in a local context. Ph.D. Thesis, University of Western Australia, Perth, Australia.

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

Vieira S., Schirmer J. and Loxton E. 2009. Social and economic evaluation methods for fisheries: a review of the literature. Fisheries Research Contract Report No. 21, Department of Fisheries, Government of Western Australia, and the Western Australian Marine Science Institution (WAMSI). Perth, Australia, 94 p.

Waddington, K I. (2008). Variation in evacuation rates of different foods skew estimates of diet in the western rock lobster *Panulirus cygnus*. *Marine and Freshwater Research* 59, 347-350.

Waddington K.I. and Meeuwig J.J. 2009. Contribution of bait to lobster production in an oligotrophic marine ecosystem as determined using a mass balanced model. *Fisheries Research* 99(1): 1–6.

Waddington K.I., Bellchambers L.M., Vanderklift M.A. and 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 coastal ecosystems. *Estuarine, Coastal and Shelf Science* 79: 114–120.

Watson D.L., Harvey E.S., Anderson M.J. and Kendrick G.A. 2005. A comparison of temperate reef fish assemblages recorded by three underwater stereo–video techniques. *Marine Biology* 148 (2), 415–425

Watson D.L., Harvey E.S., Kendrick G.A., Nardi K. and Anderson M.J. 2007. Protection from fishing alters the species composition of fish assemblages in a temperate–tropical transition zone. *Marine Biology* 152(5): 1197–1206.

Willas T.J. and Millar R.B. 2004. Using marine reserves to estimate fishing mortality. *Ecology Letters* 8: 47–52.

Williams S. B., Pizarro O., Jakuba M. V., Johnson C. R., Barrett N., Babcock R. C., Kendrick G. A., *et al.* 2012. Monitoring of benthic reference sites. *IEEE Robotics and Automation Magazine* 19: 73–84.

Zeale M.R.K., Butlin R.K., Barker G.L.A., Lees D.C. and Jones G. 2011. Taxon-specific PCR for DNA barcoding arthropod prey in bat faeces. *Molecular Ecology Resources* 11: 236–244.

13.0   Appendices

Appendix 1: Intellectual Property

There is no identifiable intellectual property arising from this report

Appendix 2: Project staff

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*Staff employed for parts of the project under FRDC funding

#In-kind staff contributions by DoF and other institutes