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**Spatial models and risk assessments to inform
marine planning at ecosystem-scales:
seagrasses and dugongs as a case study**

Thesis submitted by

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B.Env.Sc. (Hons) University of Adelaide

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Abstract

Informing marine planning and the management of species at ecosystem-scales is difficult because data are generally lacking at that scale. Collecting empirical information on the distribution and/or abundance of species across broad spatial scales is expensive and logistically difficult. Accurate and efficient monitoring programmes that assess the response of species to management actions often cannot be conducted at ecosystem-scales due to time, expertise and cost constraints.

The Great Barrier Reef World Heritage Area (GBRWHA) of Queensland, Australia, is the world's largest World Heritage Area (approximately 348,000 km²) and second largest marine protected area (MPA). The region supports a variety of habitats and species including coastal seagrasses and globally significant populations of the dugong (*Dugong dugon*), a threatened marine mammal. Seagrasses, dugongs and their habitats are exposed to multiple anthropogenic threats along much of the 2,300 km coastline of the GBRWHA. Assessing the effectiveness of the current management arrangements for seagrasses and dugongs and informing the design of new regimes is challenging due to the difficulties associated with data collection and monitoring at the scale of the coastal GBRWHA.

My thesis goal was to overcome the difficulties associated with informing the management of coastal seagrasses and dugongs in the GBRWHA by using spatial models and risk assessments in geographical information systems (GIS). My objectives were to: (1) develop spatial models of seagrasses and dugongs at the scale of the coastal GBRWHA; and, (2) use these models to estimate the risk to coastal seagrasses and dugongs from their anthropogenic threats. This approach allowed me to compare and rank the threats to identify the most severe risks, and to locate specific sites that require conservation actions.

I used spatial information on the distribution of coastal seagrasses and predictor variables along with ecological theory and expert knowledge to inform the design of a Bayesian belief network, and to develop a predictive seagrass habitat model. The Bayesian belief network quantified the relationship (dependencies) between seagrass habitats and eight environmental drivers: relative wave exposure, bathymetry, spatial

extent of flood plumes, season, substrate, region, tidal range and sea surface temperature. The outputs of the modelling exercise were probabilistic GIS-surfaces of seagrass habitat suitability for the entire GBRWHA coast in both the wet and dry seasons at a planning unit of 2 km * 2 km.

Quantitative information on the relative impact of the anthropogenic threats to coastal seagrasses is incomplete or unavailable, and the cumulative impact of multiple threats is difficult to measure and predict. In the light of this uncertainty, I used expert knowledge to evaluate the relative risk of coastal seagrass habitats to their hazards. Vulnerability scores derived from expert opinion, spatial information on the distribution of threats and the probabilistic GIS-surfaces of seagrass habitat suitability were used to delineate areas of low, medium and high relative risk to coastal seagrass habitats. I found that whilst most planning units in the remote Cape York region of the GBRWHA are classified as low risk, almost two thirds of coastal seagrass habitats along the urban coast are at high or medium risk from multiple anthropogenic activities. Reducing the risk to coastal seagrass habitats in 13 sites identified for conservation action would require: (1) improving the quality of terrestrial water that enters the GBRWHA; (2) mitigating the impacts of urban and port infrastructure development and dredging; and, (3) addressing the hazards of shipping accidents and recreational boat damage.

I derived a spatially explicit dugong population model from spatial information on the abundance and distribution of dugongs collected by a 20 year time-series of aerial surveys. Data from the aerial surveys were corrected for differences in sampling intensity and area sampled between surveys prior to the development of the model. I interpolated the corrected data to the spatial extent of the aerial surveys using the geostatistical estimation method of universal kriging. The model estimated the relative density of dugongs across the GBRWHA at the scale of 2 km * 2 km dugong planning units (the same spatial scale as the seagrass habitat model). I classified each dugong planning unit as of low, medium, or high conservation value on the basis of the relative density of dugongs estimated from the model and a frequency analysis.

I compared the spatially explicit dugong population model with information on the distribution of commercial gill-netting activities to estimate the risk of dugong bycatch in the GBRWHA. I found that new management arrangements introduced in the

GBRWhA in 2004 appreciably reduced the risk of dugong bycatch by reducing the total area where commercial netting is permitted. Restructuring of the industry further reduced the total area where netting is conducted. Netting is currently prohibited in 67% of dugong planning units of high conservation value, a 56% improvement over the former management arrangements. I identified four sites where netting is still conducted in dugong planning units of high and medium conservation value. Conservation actions including area closures or modified fishing practices should be considered for these regions.

In addition to commercial gill-netting, dugongs are threatened by Indigenous hunting, trawling, vessel traffic, and poor quality terrestrial runoff. I developed a rapid approach to assess the risk to dugongs from multiple anthropogenic threats in the GBRWhA, and evaluated options to ameliorate that risk. Expert opinion and a Delphi technique were used to identify and rank anthropogenic threats with the potential to adversely impact dugongs and their habitats. I quantified and compared the distribution of these threats with the spatially explicit model of dugong distribution and found that almost all dugong planning units of high (96%) and medium (93%) conservation value in the GBRWhA are at low risk from human activities. Decreasing the risk to dugongs from anthropogenic threats in four sites that I identified for conservation action would require netting or Indigenous hunting to be banned in the remote Cape York region, and the impacts of vessel traffic, terrestrial runoff and commercial netting to be reduced in urban areas.

The approach I developed in this thesis was able to overcome the difficulties associated with informing marine planning and management at ecosystem-scales by using spatial models and risk assessments in GIS to: (1) quantify the spatial distribution of species; and, (2) assess the risk to species and identify sites for conservation action. I was able to achieve this outcome in a data-inadequate environment by combining qualitative assessments on the relative impact of multiple anthropogenic threats with spatial models of species and threat distributions. Implementing conservation actions at the sites that I identified for management will provide the greatest positive result for coastal seagrasses and dugongs at the scale of the GBRWhA. Future research should be directed at understanding the constraints and opportunities for management in the region to ensure that effective implementation of conservation actions can be achieved.

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Publications produced during my PhD candidature

Publications

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Lukoschek, V., Heatwole, H., **Grech, A.**, Burns, G. and Marsh, H. 2007. Distribution of two species of marine snakes, *Aipysurus laevis* and *Emydocephalus annulatus*, in the southern Great Barrier Reef: metapopulation dynamics, marine protected areas and conservation. *Coral Reefs* **26**: 291–307.

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Chapter 1

General introduction

Acquiring the necessary data to inform marine planning at ecosystem-scales is difficult because data are generally lacking at that scale. In this chapter, I identify an approach that can inform the planning of conservation actions and the management of species at ecosystem-scales. I also provide an outline of the rationale for my thesis, its objectives and structure.

Marine planning and management at ecosystem-scales

Marine ecosystems are characterised by high productivity and biodiversity values (Gray 1997) and provide a variety of ecosystem services (*see* Peterson and Lubchenco 1997). The annual value of these services is almost twice that of the terrestrial biome (Costanza et al. 1997). People use marine areas for shipping, recreation, and aquaculture and extract marine resources including: (1) oil, natural gas, minerals, sand and other non-living natural resources; and, (2) fisheries and genetic resources (Agardy and Alder 2005). The value of marine ecosystem services has inevitably led to a rapid increase in the world's coastal population, and it is estimated that almost 40% of people live within 100 km of the coast (Agardy and Alder 2005).

For most marine ecosystems the outcome of human use are stress from pollution, degradation and depletion of marine resources, and damage and destruction of habitat (Kelleher and Kenchington 1992). Anthropogenic activities along the coast have an indirect impact on adjacent marine ecosystems by modifying sediment regimes and nutrient and pollution inputs (Galloway et al. 2008), and a direct impact by removing or destroying habitats for infrastructure. Marine-based anthropogenic activities deplete fisheries resources (Jackson et al. 2001), pollute waters, alter habitats and change species composition (Halpern et al. 2008a). Anthropogenic climate change has profound implications for marine ecosystems across the globe (IPCC 2001). Halpern et al. (2008a) estimate that no marine area is unaffected by anthropogenic activities and that almost 41% of marine areas are seriously impacted by multiple threats. As a result, humans are compromising the delivery of marine ecosystem services that are crucial to the well-being of human communities across the world (Agardy and Alder 2005).

Mitigating the impact of anthropogenic activities on marine ecosystems can be achieved by management procedures that: (1) develop institutions and incentives that encourage conservation; (2) build public awareness through education of the value of marine ecosystem services; and, (3) protect species and ecosystems through management strategies and protected areas (Leslie 2005; Salafsky et al. 2002). Management strategies can be implemented via a diverse range of management tools that include gear restrictions, quotas and closed areas. Combinations of these tools have helped to achieve reductions in exploitation rates in global fisheries (Worm et al. 2009).

Marine protected areas (MPAs) are an effective tool for maintaining ecological processes and conserving marine biodiversity and fisheries (Agardy et al. 2003; Lubchenco et al. 2003; Pikitch et al. 2004). MPAs manage or protect defined areas of sea through legislative arrangements and management infrastructure (Kelleher and Kenchington 1992), and are typically established to increase the likelihood of sustainable fisheries, biodiversity conservation, species conservation, the preservation of cultural values or some combination of these factors (Kelleher et al. 1995). MPAs can provide protection at various levels that range from restrictions on one or more anthropogenic activities, to the comprehensive protection of an area from all extractive effects. The highest level of protection afforded by MPAs is 'no-entry' marine reserves that restrict the entry of people and vessels and protect species and habitats from extractive activities (Fernandes et al. 2005). The next highest level of protection is 'no-take' marine reserves that exclude extractive activities from an area while allowing the entry of people and vessels. MPAs and marine reserves are demonstrated to: have a positive impact on biological components of marine ecosystems including density, biomass, and size of organisms; initiate recovery of species by providing refuges for over fished stocks; restore community structure and biodiversity; protect important habitat features; and increase ecosystem resilience (Halpern et al. 2003; Russ et al. 2008; Worm et al. 2009). Marine reserves are also demonstrated to have positive impacts on areas outside the reserve by enhancing the performance of adjacent fisheries (Roberts et al. 2001).

Safeguarding the delivery of marine ecosystem services requires the maintenance of ecological processes that underpin the functioning of marine ecosystems (Agardy 1994; Daily 1997; Roberts et al. 2003). Ecological processes within marine systems differ from terrestrial systems in their scale and variability (Steele 1985). Marine productivity is heterogeneous and highly variable over space and time (Agardy 1994). The transport of materials, nutrients and organisms by the convective forces of ocean waves and currents extends the spatial scale of many ecological processes (Carr et al. 2003). Maintaining ecological processes that underpin the functioning of marine ecosystems requires the management of marine resources to occur at an appropriate spatial scale. Planning at the broad spatial scale of ecosystems alleviates the impact of human activities on the delivery of ecosystem services as activities would be managed at a scale similar to that of the associated ecological processes (Halpern et al 2008b).

Ecosystem-scale networks of MPAs are a collection of individual MPAs or reserves that operate cooperatively at '*various spatial scales and with a range of protection levels that are designed to meet objectives that a single reserve cannot achieve*' (UNEP-WCMC 2008). Ecosystem-scale networks of MPAs are increasingly favored over small, isolated MPAs, as demonstrated by the 2002 World Summit on Sustainable Development target to establish a global representative network of MPAs by 2012. This target includes a recommendation that 20 – 30% of each marine ecosystem type should lie within strictly protected reserves (Sutherland et al. 2009).

Networks of MPAs that occur over ecosystem-scales should sample or represent the species of a region, and separate these species from anthropogenic activities that threaten their existence. Achieving this goal requires a systematic approach to locating and designing networks of MPAs. Margules and Pressey (2000) formulate a six stage approach for systematic conservation planning: (1) compile data on biodiversity and species in the planning region; (2) identify conservation goals; (3) review existing conservation areas; (4) select additional conservation areas; (5) implement conservation actions; and, (6) maintain the required values of conservation areas. MPAs permit or restrict anthropogenic activities within a given area and so information on ecosystems and species, conservation goals, quantitative targets and conservation actions are primarily spatially explicit (Margules and Pressey 2000). Stage 6 requires the accurate and efficient monitoring of MPAs to assess the response of ecosystems and species to conservation actions that attempt to ameliorate the impact of anthropogenic activities (Margules and Pressey 2000; Pressey and Botrill 2008; Joseph et al. *in press*).

The data necessary to inform a systematic approach to planning at ecosystem-scales are far less organised and available for most marine environments than for many terrestrial environments (Carr et al. 2003). Spatial information on the distribution of habitats and species is difficult to collect at the scale of marine ecosystems as it is expensive and logistically difficult (Ban 2009). Accurate and efficient monitoring programmes that assess the responses of habitats and species to management actions are generally unavailable at ecosystem-scales due to multiple factors, including time, expertise, and cost constraints. For many marine species it is impossible to detect even large changes in their populations with current levels of investment in surveys, survey technology, and survey design (Taylor and Gerrodette 1993; Field et al. 2004; Taylor et al. 2007).

Spatially explicit models incorporating the distribution of species and anthropogenic impacts (Pull and Dunning 1995) are a valuable tool that can inform the design and management of ecosystem-scale networks of MPAs (Turner et al. 1995; Rogers-Bennett et al. 2002). Predictive habitat distribution models quantify species-environment relationships to create spatially explicit models of species distribution and habitat structure (Guisan and Zimmermann 2000) that allow information from areas where data are readily available to be extended to predict distribution in areas where data are limited or non-existent. Decision-support tools, such as spatial risk assessments in geographical information systems (GIS), can assist the rapid assessment of risks to ecosystems and species by incorporating spatially explicit models of species distribution with qualitative and quantitative information on the distribution of anthropogenic threats (Pull and Dunning 1995). A spatial risk assessment approach provides managers with maximum return for minimal investment in data collection by identifying areas where management intervention should provide the greatest positive result for the resources of concern (Theobald 2003).

Great Barrier Reef World Heritage Area

The Great Barrier Reef World Heritage Area (GBRWHA) of Queensland, Australia, (Figure 1.1) is a complex and diverse collection of tropical marine ecosystems of globally significant environmental, cultural, social and economic value. The GBRWHA is approximately 348,000 km² making it the world's largest World Heritage Area and second largest MPA. The coastline bordering the GBRWHA stretches ~ 2,300 km from the tip of the Cape York Peninsula (142°E, 10°S) south to Bundaberg (152°E, 24°S). The values of the GBRWHA are threatened by a range of anthropogenic activities that occur both within the boundaries of the World Heritage Area and its adjacent land catchments (*see* Chapter 2). Multiple levels of government control the anthropogenic threats that occur within and adjacent to the GBRWHA. Management of the region involves cross-jurisdictional partnerships between the Australian Government, Queensland State Government, local councils and Indigenous groups. A range of spatial (e.g. reserves) and non-spatial (e.g. fisheries quotas) strategies are employed by these governments to mitigate the impact of anthropogenic activities to the ecosystems that occur within the GBRWHA. Strategies to mitigate the impact of land based activities include the Australian and Queensland Government's *Reef Water*

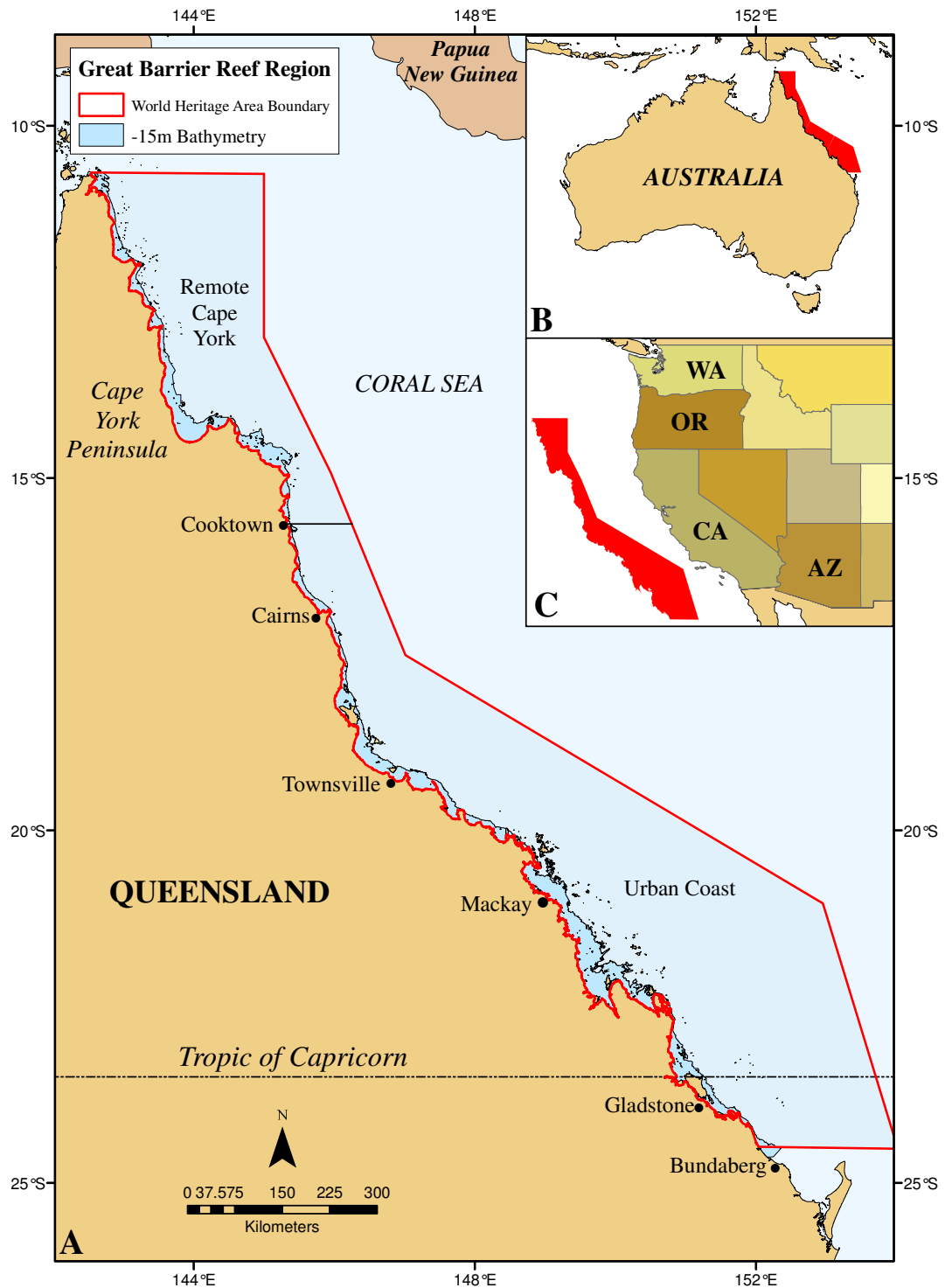


Figure 1.1: (A) Extent of the Great Barrier Reef World Heritage Area (GBRWHA) off the coast of Queensland, Australia. Major regional cities are shown. The coastal waters of the GBRWHA are approximately -15 m below mean sea level as illustrated; (B) the GBRWHA relative to Australia; and (C) extent of the GBRWHA relative to the west coast of the United States.

Quality Protection Plan (The State of Queensland and Commonwealth of Australia) that recommends specific water quality targets for individual river systems that flow into the GBRWHA.

The GBRWHA was rezoned in 2004 to improve marine biodiversity protection through a comprehensive and representative multiple-use zoning regime that established an ecosystem-scale network of ‘no-take’ areas covering > 33% of the Great Barrier Reef Marine Park and the contiguous Great Barrier Reef Coast Marine Park (Day et al. 2000; Fernandes et al. 2005). ‘No-take’ areas in the GBRWHA are sites that are protected from extractive uses such as commercial and recreational fishing. An independent expert Scientific Steering Committee established 11 Biophysical Operational Principles to guide the Australian Government’s Great Barrier Reef Marine Park Authority in developing the ‘no-take’ network (Fernandes et al. 2005 and 2009). Principle 8 addressed the need to represent a minimum amount of each community type and physical environment type in the overall network. The objectives to implement this principle included a commitment to ensure that about 10% of coastal seagrass habitats and 50% of 29 high priority sites of the dugong, *Dugong dugon*, were zoned as ‘no-take’ areas (Fernandes et al. 2005).

In the GBRWHA, coastal seagrass habitats are both inter-tidal and sub-tidal to approximately -15 m below mean sea level, and are characterised by high diversity and productivity (Lee Long et al. 2000). Coastal seagrasses are spatially and temporally dynamic (ephemeral), provide food for green sea turtles (*Chelonia mydas*) and dugongs, and are an important nursery grounds for commercial, recreational and Indigenous fisheries. Management of coastal seagrasses in the GBRWHA is complex; almost fifty pieces of legislation listed for Queensland alone could have some bearing on its protection (McGrath 2003). Dugongs are specialist feeders in seagrass communities and are a threatened species of very high cultural value to Indigenous Australians and high intrinsic value to non-Indigenous Australians. Significant populations of dugongs inhabit the shallow, protected inshore waters of the GBRWHA (Marsh et al. 2002) and were an explicit reason for the region’s World Heritage listing (GBRMPA 1981). Dugongs are protected under the Queensland Government’s *Nature Conservation Act* 1992, and listed as ‘vulnerable wildlife’ under Schedule 3 of the *Nature Conservation (Wildlife) Regulation* 1994. They are also protected under the Australian Government’s

Environment Protection and Biodiversity Conservation Act 1999 as a ‘listed marine’ and a ‘listed migratory’ species.

Informing marine planning at the scale of the entire GBRWHA (348,000 km²) is challenging because collecting information on the spatial distribution of species at that scale is expensive, time-consuming and logistically difficult. Fernandes et al. (2005) and Dobbs et al. (2008) assess the effectiveness of new zoning arrangements to protect coastal seagrasses and dugongs by using expert scientific knowledge and information on seagrass and dugong spatial distribution collected as point localities. Both of these studies are limited in their ability to inform marine planning in the GBRWHA because they do not use information that accurately delineates species distribution. When compared with most of the world information on seagrass and dugong distribution in the GBRWHA is data rich. Nonetheless, the data sets used by Fernandes et al. (2005) and Dobbs et al. (2008) are inadequate to set and/or test quantitative targets. The seagrass point locality datasets do not: (1) cover the entire range of coastal seagrass habitats of the GBRWHA; or, (2) effectively represent spatio-temporal changes in seagrass distribution. The dugong point locality data sets do not identify sites where dugongs are most abundant over time. Furthermore, expert scientific knowledge is often biased to areas along the urban coast where human population densities are high and coastal areas are readily accessible (Figure 1.1).

Assessing only those anthropogenic activities that are within the regulatory control of ‘no-take’ areas is insufficient to inform the management of coastal seagrasses and dugongs in the GBRWHA. In addition to commercial and recreational fishing activities, coastal seagrass and dugong habitats are threatened by climate change, poor quality terrestrial runoff from adjacent land catchments, and habitat modification as a result of land reclamation, dredging and infrastructure development (Coles et al. 2007). Dugongs are also directly threatened by vessel strike, bycatch in commercial gill-nets and Indigenous hunting (Marsh et al. 2002).

Estimating the protection afforded to coastal seagrasses and dugongs from multiple anthropogenic threats and informing the design of new conservation actions in the GBRWHA is challenging. The relative impact of multiple threats is difficult to measure and predict because monitoring the response of coastal seagrasses and dugongs at the

scale of the GBRWHA is impossible. Some tropical seagrasses are ephemeral, and responds both spatially and temporally to environmental change. This characteristic makes it difficult to determine if changes in seagrass distribution and dugong distribution and abundance are a result of natural causes or in response to anthropogenic activities. The number of dugongs injured or killed by any one of the major causes of human-induced mortality in the GBRWHA cannot be ascertained under the existing monitoring programme (Marsh et al. 2002). Even if this number was known, uncertainty about the absolute size of the dugong population in the GBRWHA and their spatial and temporal variability in the dugong's life history parameters makes it difficult to conduct quantitative evaluations such as Potential Biological Removal modelling (Wade 1998; Marsh et al. 2004) or Population Viability Analysis (Heinsohn et al. 2004). A new approach to assess the effectiveness of the current management regime and inform the design of future management actions is required.

Thesis objectives

The Queensland and Australian Governments are committed to using the best available scientific information to underpin the management of the GBRWHA (GBRMPA 2009). The goal of this thesis was to contribute to the scientific basis for optimising the conservation of coastal seagrasses and dugongs by informing their planning and management at the scale of the GBRWHA. To achieve this goal, I identified the following objectives:

1. Quantify the spatial distribution of coastal seagrass habitats and dugongs at the scale of the GBRWHA;
2. Estimate the risk of coastal seagrass habitats and dugongs from their anthropogenic threats;
3. Inform the management of coastal seagrass habitats and dugongs at the scale of the GBRWHA.

Objective 1: Quantify the spatial distribution of coastal seagrass habitats and dugongs at the scale of the entire GBRWHA.

Marine planning at the scale of the GBRWHA requires spatial information on the distribution of species at that scale. A combination of expert scientific knowledge and maps of seagrass and dugong distribution as point localities have been used in previous

studies to inform marine planning at the scale of the coastal GBRWHA (Lewis et al. 2003; Fernandes et al. 2005; Dobbs et al. 2008). However, point locality data sets are limited in their capacity to inform marine planning over large areas as they are spatially and temporally biased. Predictive models of seagrass habitats and dugongs at the scale of the coastal GBRWHA are required because of the lack of resources available to collect complete and adequate data sets at that scale. I have addressed this need by using a spatially explicit modelling approach in Chapters 3 and 5.

Objective 2: Estimate the risk of coastal seagrass habitats and dugongs from their anthropogenic threats.

Coastal seagrass habitats and dugongs in the GBRWHA are exposed to anthropogenic threats that are both terrestrial and marine based. Assessing the impact of one and/or more threats and the effectiveness of conservation actions at the scale of the GBRWHA is made difficult by the uncertainties in the data available, and the costs associated with collecting information at that scale. It is impossible to determine the impact of one threat in the presence of multiple threats due to logistical, ethical and political difficulties. Risk assessments in GIS overcome the constraints associated with evaluating the impact of multiple anthropogenic threats at the spatial scale of the GBRWHA by incorporating spatially explicit models of species distribution with qualitative and quantitative information on the distribution of anthropogenic threats (Pull and Dunning 1995; Andersen et al. 2004).

Spatial risk assessments require information on the relative impact of various threats to ecosystems and species. The relative impact of anthropogenic threats on coastal seagrasses in the GBRWHA is difficult to measure and predict, and quantitative information on anthropogenic dugong mortality is biased to populated areas.

Determining the relative impact of anthropogenic threats on coastal seagrass habitats and dugongs in the GBRWHA can only be achieved by qualitative assessments that are informed by expert opinion (e.g. Halpern et al. 2007; Selkoe et al. 2008). I estimated the risk to coastal seagrass habitats and dugongs from their anthropogenic threats in Chapters 4, 6 and 7.

Objective 3: Inform the management of coastal seagrass habitats and dugongs at the scale of the GBRWHA.

Managing anthropogenic threats at the scale of the GBRWHA is constrained by the cost associated with implementing conservation actions at that scale. Effective management in the GBRWHA therefore requires strategic deployment of conservation resources (Cleary 2006). Conservation actions that target all anthropogenic threats are unfeasible as limited funding and resources restricts the number of activities that can be managed (Halpern et al. 2007). An alternative approach is to prioritise conservation action by ranking the relative impact of anthropogenic threats and identifying vulnerable habitats (Crain et al. 2009). This approach allows for the identification of sites that if targeted would have the greatest conservation benefit to coastal seagrass habitats and dugongs. I informed the management of coastal seagrass habitats and dugongs at the scale of the entire GBRWHA by identifying priority sites for conservation actions in Chapters 4, 6, and 7. The constraints and opportunities for implementing conservation actions in these priority areas are discussed in Chapter 8.

Thesis outline

This thesis is presented as a series of chapters that have been written in a format to facilitate publication in peer review journals. Figure 1.2 illustrates the overall structure of the thesis. Authorship of chapters for publication (Chapters 3 – 8) is shared with two members of my thesis committee, Helene Marsh (Chapters 4 – 8) and Rob Coles (Chapters 3, 4, 6 and 8). Helene Marsh and Rob Coles contributed to the development of the approach, analysis of data and the interpretation of results, training, funding and the preparation of chapters and manuscripts for publication. Several government and research organisations contributed spatial information that featured as layers in the data analysis. I have identified the custodians of spatial information within the relevant chapters.

Tables and figures are shown throughout the text; additional supporting figures are provided in the appendices. I created all the tables and figures that are shown in this thesis and the appendices, unless stated otherwise.

Chapter 1 provides an introduction to the limitations associated with informing marine planning at ecosystem-scales.

Chapter 2 describes the study region and species.

Chapter 3 quantifies the spatial distribution of coastal seagrass habitats. This chapter is *in press* at *Aquatic Conservation: Marine and Freshwater Ecosystems*. The relative wave exposure index that I described in this chapter is published in *Coral Reefs* (Lukoschek et al. 2007) and *Aquatic Conservation: Marine and Freshwater Ecosystems* (Santana Garcon et al. *in press*); I am a co-author on both publications. Rob Coles and his group at Fisheries Queensland collected data on seagrass distribution that informed the model. I conducted the analysis and wrote the chapter, and Rob Coles and Helene Marsh assisted with model interpretation and editing.

Chapter 4 estimates the risk to coastal seagrass habitats from multiple anthropogenic threats. A preliminary assessment that used the Dry and Wet Tropics of the GBRWHA as a case study was produced as a report for the Marine and Tropical Sciences Research Facility (Grech et al. 2008a). This chapter will be submitted for publication in a peer reviewed journal. I conducted the analysis and wrote the chapter and Rob Coles and Helene Marsh assisted with the interpretation of results and editing.

Chapter 5 prioritises areas for dugong conservation in the GBRWHA from a spatially explicit model of dugong distribution and relative abundance. This chapter has been published in *Applied GIS* (Grech and Marsh 2007). The spatially explicit model also features in reports for the Marine and Tropical Sciences Research Facility (Marsh et al. 2007) and Fisheries Queensland (Coles et al. 2007); I am a co-author on both publications. Helene Marsh and her group in the School of Earth and Environmental Sciences collected the data on dugong distribution and abundance that informed the model. I conducted the analysis and wrote the chapter, and Helene Marsh assisted with model interpretation and editing.

Chapter 6 estimates the risk to dugongs from bycatch in nets of the Queensland East Coast Inshore Fin Fish Fishery. This chapter has been published in *Aquatic Conservation: Marine and Freshwater Ecosystems* (Grech et al. 2008b). The approach I

outlined in this chapter is used by Dryden et al. (2008); I am a co-author. I conducted the analysis and wrote the chapter and Helene Marsh and Rob Coles assisted with the interpretation of results and editing.

Chapter 7 rapidly assesses the risk to dugongs from multiple anthropogenic threats in the GBRWHA. This chapter has been published in *Conservation Biology* (Grech and Marsh 2008). I conducted the analysis and wrote the chapter and Helene Marsh assisted with the development of the rapid assessment approach, interpretation of results and editing.

Chapter 8 provides a summary of the previous chapters and a discussion on the constraints and opportunities for implementing conservation actions in the areas that I identified as a priority for management. This chapter will be further developed before it is submitted for publication in a peer reviewed journal. I wrote the chapter and Helene Marsh and Rob Coles assisted with the development of the approach and editing.

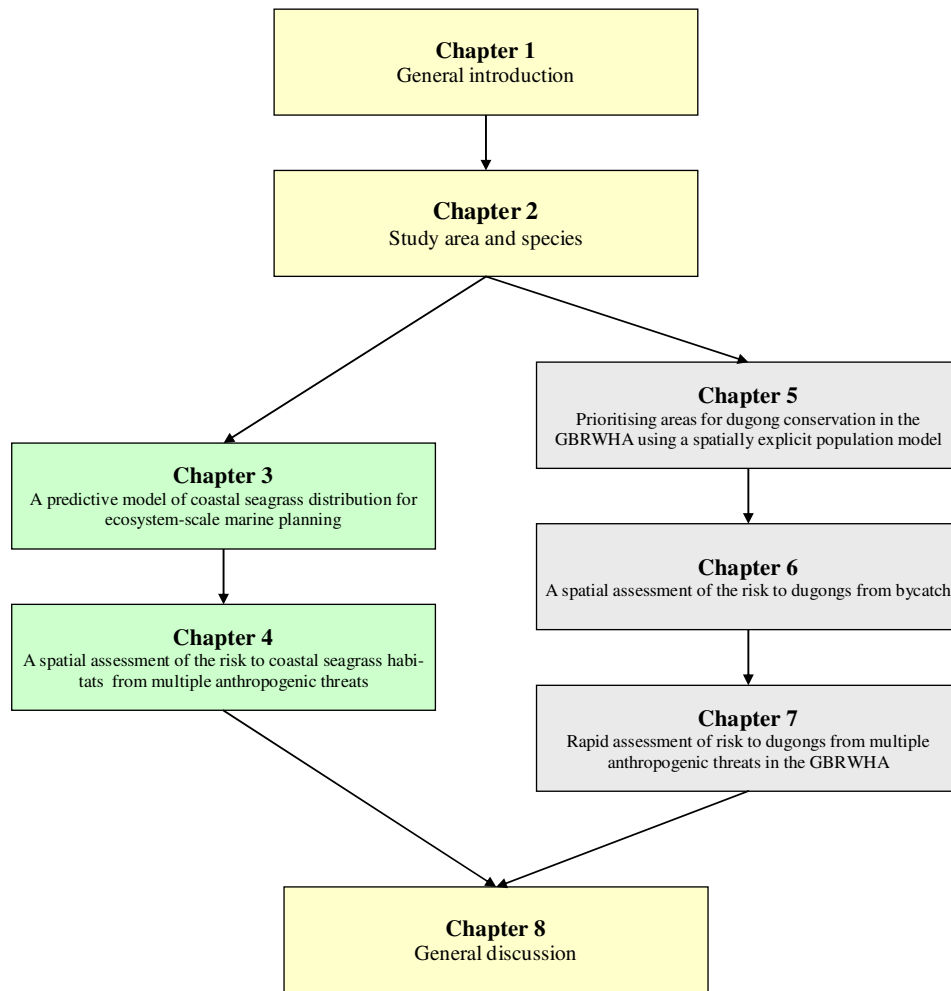
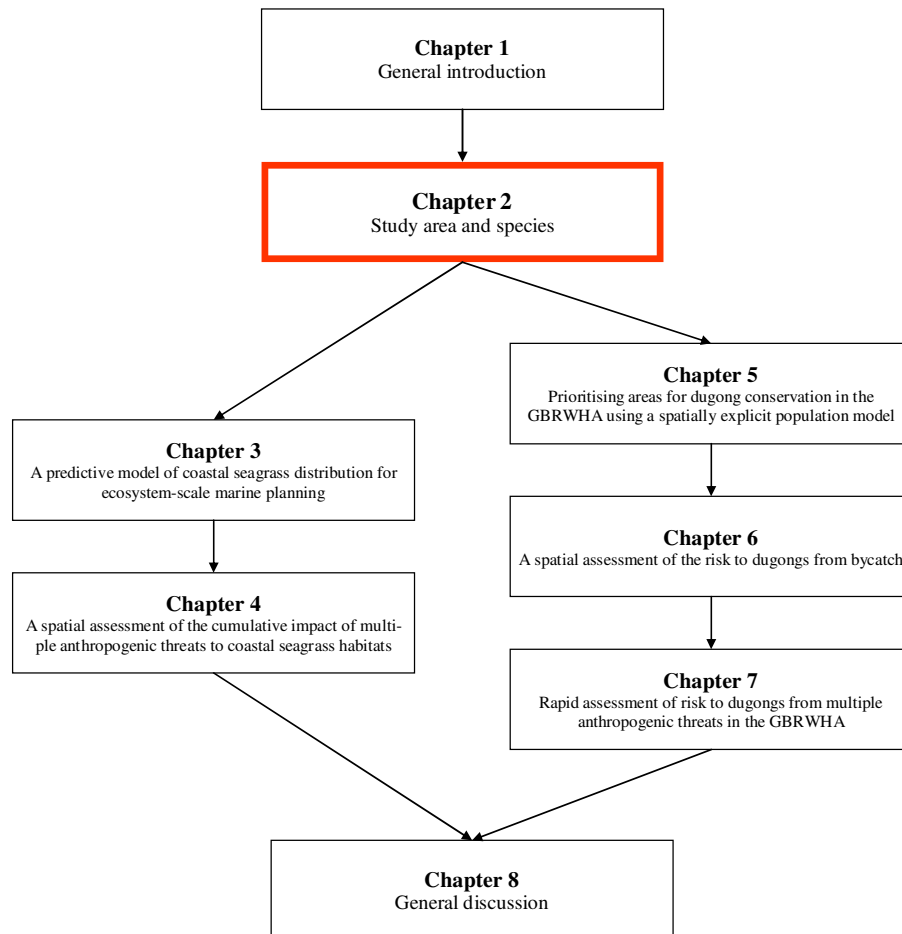


Figure 1.2: Chapter structure of this thesis.

Chapter 2

Study area and species

In this chapter, I summarise the relevant information on the environment and management of the Great Barrier Reef World Heritage Area and the ecology, status and management of seagrasses and dugongs as the background to my study.



The Great Barrier Reef World Heritage Area

The Great Barrier Reef World Heritage Area (GBRWHA) stretches from 10.7°S to 24.5°S (~ 2,300 km) along Australia's north-eastern seaboard (Figure 1.1), and extends 70 to 250 km from the coast. Five major habitat types occur in the GBRWHA (GBRMPA 2009): coral reefs (7% of the region); seagrass, shoals and sandy or muddy seabed (61%); continental slope habitats (15%); deep oceanic waters (16%); and mangrove and island habitats (1%). The habitats of the GBRWHA support a variety of biota including: 1500 species of fish; 400 species of corals; 30 species of whales, dolphins and porpoises; and around 125 species of sharks and rays (Hutchings et al. 2008). The GBRWHA is populated by many vulnerable and threatened species including the dugong (*Dugong dugon*) and six of the seven species of marine turtles.

The biodiversity and ecosystem functions of the GBRWHA are strongly affected by the physical driving forces of oceanic, wind driven and tidal currents. The South Equatorial, Hiri and East Australian currents are the most significant currents that influence the oceanography of the GBRWHA. The South Equatorial Current flows westward across the Pacific and Coral Sea, and when it reaches the Australian continental shelf at about 14°S divides into two currents: the Hiri Current (north) and the East Australian Current (south). In some areas these currents cause upwelling of deep, cold, nutrient-rich water onto the continental shelf. While oceanic currents have a strong influence on the movement of water on the continental shelf, in shallow waters currents are also driven by wind. Oceanic and wind-driven currents primarily drive water parallel to the coast and tidal currents (which operate on a 12-hour cycle) drive water across the continental shelf perpendicular to the coast. Tidal ranges in the coastal waters of the GBRWHA can be very large and range from 2.5 – 9 m (Hopley et al. 2007). Together, oceanic, wind-driven and tidal currents create a complex pattern of water movement in the GBRWHA.

Climate in the GBRWHA is dominated by two large scale global circulation systems: (1) the Australian summer monsoon westerly circulation; and, (2) the south-easterly trade wind circulation. The two systems effectively divide the year into a warm summer wet season (November to March) and a cooler dry season (April to October). Approximately 80% of the annual rainfall occurs during the summer wet season. Rainfall is highly variable both temporally and spatially within the summer monsoon

season and usually occurs in several bursts of activity. There is also considerable inter-annual variability in rainfall. Monthly mean air and sea surface temperatures reach maxima from January to February and minima in August. Monthly mean sea surface temperatures range from greater than 29°C in summer in the northern latitudes to less than 22°C in winter in the southern latitudes. Seasonal wind patterns generally shift from a predominance of north-easterlies to south-easterlies from summer to winter, respectively. The conditions suitable for tropical cyclone development occur during the summer wet season; peak tropical cyclone activity occurs between January and March. Tropical cyclones often result in elevated sea levels and destructive storm waves (storm surge), high rainfall and rapid increases in river flows.

In northeast Australia generally and the GBRWHA specifically, there is high inter-annual variability in climate, especially in rainfall and river flow. The major source of this variability is the El Niño Southern Oscillation (ENSO) (McPhaden 2004). ENSO has two phases: El Niño and La Niña. El Niño conditions are associated with a cooling of sea surface temperatures in the GBRWHA and a reduction in rainfall off eastern Australia. La Niña conditions result in increased sea surface temperatures and higher than average rainfall. The level of disturbance appears to be greater during La Niña events due to heightened tropical cyclone activity and enhanced rainfall and river flows (Lough 2007).

The GBRWHA receives runoff from the catchments of eastward-draining streams and rivers between the Cape York Peninsula and Fraser Island (Figure 2.1). The combined area of the 35 drainage basins of the GBRWHA catchment is 423,070 km², and encompasses 25% of the land area of Queensland and 5.6% of Australia. About 30 significant rivers and many hundreds of small, usually ephemeral streams drain into the coastal waters of the GBRWHA. The majority (~ 80%) of total river flow into the GBRWHA occurs between 17°S and 23°S with greatest annual flow in March (a month after the rainfall maxima).

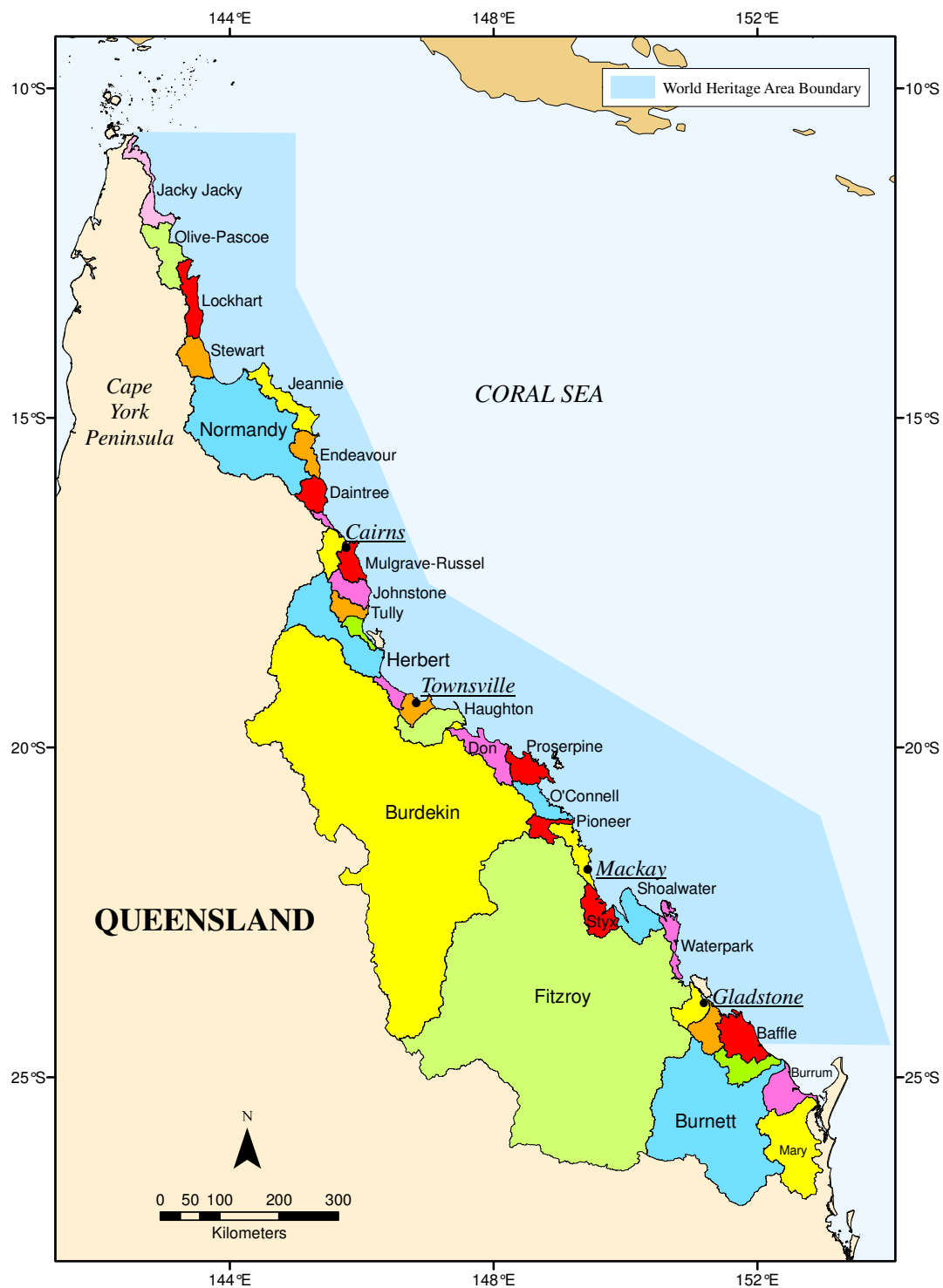


Figure 2.1: The name and spatial location of the 35 mainland drainage basins of the GBRWHA catchment. Major regional centres are underlined.

The GBRWHA catchment supports a human population of almost 1,115,000, which is projected to grow at a rate of about 2% per annum (OESR 2008). There are 68 urban centres (population > 200) adjacent to the GBRWHA coast and four large regional centres (population > 50,000; Figure 2.1). The major land uses of non-urban areas in the GBRWHA catchment are cattle grazing, cropping, dairying, horticulture, forestry and protected areas (Furnas et al. 2003). Significant mine enterprises are also a feature of the GBRWHA catchments; however, oil drilling, mining and exploration have been prohibited inside the Great Barrier Reef Marine Park since the proclamation of the Australian Government's *Great Barrier Reef Marine Park Act* in 1975.

The goods and services that the GBRWHA provides are important to the wellbeing of many people living within its catchments and beyond. The GBRWHA is a major recreational area, an internationally important scientific resource, and an important area for military training. The GBRWHA supports significant commercial industries, especially marine tourism and fishing. Shipping activity through the GBRWHA services the regional centres in the catchments and is vital for the trade of commodities.

Industries in the GBRWHA are important to the Australian economy. By way of illustration, in 2006/07 industries in this region directly and indirectly contributed an estimated AU\$5.4 billion to the country's economy (GBRMPA 2009). This value includes \$5.1 billion from the tourism industry, \$153 million from recreational activity, and \$139 million from commercial fishing. The economic activity in the GBRWHA generates about 66,000 jobs, mostly in the tourism industry.

Aboriginal people and Torres Strait Islanders have a 40,000 year history along the east coast of Queensland, and the habitats and species of the GBRWHA are a significant part of their culture, spirituality and livelihoods. Archaeological excavation of dugong bones in adjacent Torres Strait reveals that dugong hunting has an antiquity of at least 4000 years (Crouch et. al. 2007). Currently there are about 70 Traditional Owner¹ groups with connections to the GBRWHA. Traditional Owners maintain connections with their sea country in multiple ways that can include fishing and Indigenous hunting of dugongs, marine turtles and other species. Traditional Owners can conduct Indigenous hunting in all areas of the GBRWHA as Indigenous hunting rights have been affirmed

¹ Traditional Owners are Aboriginal and Torres Strait Islander people who are descendents of the tribe or ethnic group that occupied a particular region before European settlement.

by the Australian Government's *Native Title Act* 1993, subsequent judgments in the High Court of Australia and the Australian Government's *Environmental Protection and Biodiversity Conservation Act* 1999 (see Havemann et al. 2005).

Management arrangements

The GBRWHA is functionally divisible into two distinct management regions: the developed urban coast and the remote Cape York region (Figure 1.1). Large regional centres including the cities of Cairns, Townsville, Mackay, and Gladstone, are situated along the urban coast of the GBRWHA, which supports a total population of greater than 1,000,000 people (Figure 2.1). Land-use in catchments along the urban coast is dominated by agriculture, industry, mines and urban areas. Ten of the twelve ports in the GBRWHA are located along the urban coast.

In the remote Cape York region, the population is much smaller (< 7,000 people) and scattered amongst communities separated by large expanses with very little infrastructure. The majority of land tenure in catchments of the remote Cape York region is held by cattle grazing properties, Aboriginal communities and the State of Queensland. A significant proportion of land catchments (25% of ~ 43,000 km²) in the remote Cape York region are managed by the State of Queensland as protected areas. The waters of the remote Cape York region are the most biologically diverse in the entire GBRWHA (Hutchings and Kingsford 2008). The high diversity in the region is most likely due to the warmer sea temperatures and extensive reef habitats across the latitudinal gradient. Most of the waters in the remote Cape York region are designated as a 'Remote Natural Area' in recognition of the environmental significance of the region. Activities such as motorised water sports, dredging, reclamation, harbour works and construction are prohibited within the waters designated as a 'Remote Natural Area' (Dobbs et al. 2008).

The GBRWHA is jointly managed by the State of Queensland and Australian (Commonwealth) Governments as a multiple-use area to maximise biodiversity protection while also maintaining the cultural, social, and economic attributes of the region. The Australian Government's *Great Barrier Reef Marine Park Act* 1975 established both the Great Barrier Reef Marine Park and the Great Barrier Reef Marine

Park Authority. The Great Barrier Reef Marine Park Authority is a Commonwealth Statutory Authority and principal adviser to the Australian Government on the planning and management of the Marine Park. The Great Barrier Reef Marine Park is a Commonwealth marine park that protects all the waters below mean low water mark and some islands or part of islands that are Commonwealth-owned. Most coastal and island waters above mean low water mark are within Queensland's Great Barrier Reef Coast Marine Park. Marine areas within the boundaries of Port Authorities are excluded from the Great Barrier Reef Coast Marine Park and Great Barrier Reef Marine Park, and managed separately by the Queensland Government. The State and Commonwealth waters and islands of the region became a World Heritage Area in 1981 when they were inscribed on the World Heritage List (GBRMPA 1981).

In addition to the marine parks that control the spatial distribution of extractive activities a variety of other management tools are employed by the Queensland and Australian Governments to ensure the sustainable use of the GBRWHA. These tools include statutory Plans of Management, permits, Special Management Areas, agreements with Indigenous groups and industry-specific accreditation (*see* Day 2008). Queensland Government agencies are directly responsible for the management of activities in State waters, ports and the GBRWHA land catchment.

The Day-to-Day Management Program is a partnership between the Queensland and Australian Governments to coordinate the day-to-day activities and field operations of the GBRWHA. Field operations are performed by multiple Queensland and Australian Government agencies that include: the Queensland Department of Environment and Resource Management, Fisheries Queensland, Queensland Boat and Fisheries Patrol, and Queensland Water Police; and the Australian Coastwatch, Customs National Marine Unit and Federal Police. The Great Barrier Reef Marine Park Authority also participates in activities such as compliance, monitoring and the assessment of permits.

Between 1981 and the late 1990s approximately 15,800 km² (~ 4.5%) of the Great Barrier Reef Marine Park was zoned as 'no-take' areas (marine reserves); a further 450 km² was set aside for scientific research. A review of one section of the Great Barrier Reef Marine Park in the early 1990s found that the amount and distribution of 'no-take' areas were most likely inadequate to ensure protection of the entire range of

marine biodiversity in the GBRWHA (Fernandes et al. 2005; Day 2008). The Representative Area Program was subsequently initiated to re-zone the GBRWHA to improve biodiversity protection through a comprehensive and representative multiple-use zoning regime (Day et al. 2000). The objective of the Representative Area Program was to maintain biological diversity by optimising the design of a network of ‘no-take’ areas, covering the range of habitats and communities found within the GBRWHA (Day et al. 2000).

The Great Barrier Reef Marine Park Authority worked with an independent expert Scientific Steering Committee to establish 11 Biophysical Operational Principles to guide the agency in developing the ‘no-take’ network (Fernandes et al. 2009). These Principles were developed with regard to: (1) the biological objectives of the program; (2) available data on and knowledge of the reef ecosystem; (3) available data on the science of marine reserve design; and, (4) communications between experts in reef and non-reef ecosystems and reserve design (Fernandes et al. 2005). The 11 Biophysical Operational Principles, expert opinion, stakeholder involvement and analytical approaches such as the marine reserve design software MarXan (Ball and Possingham 2000) informed the Great Barrier Reef Marine Park Authority in the design of the new zoning plan.

The Australian Government’s *Great Barrier Reef Marine Park Zoning Plan* 2003 and Queensland’s *Great Barrier Reef Coast Marine Park Zoning Plan* 2004 are both based on the outcomes of the Representative Areas Program. The Plans provide a high level of protection for specific areas whilst also allowing a variety of activities to occur in certain zones. These activities include shipping, dredging, aquaculture, tourism, boating, diving, military training, commercial fishing and recreational fishing. Figure 2.2 provides a description of the activities that are allowed within the seven zones; and Figure 2.3 shows their spatial configuration. The Plans protect approximately 33% of the GBRWHA in ‘no-take’ and ‘no-go’ zones, and are a substantial improvement on the previous zoning arrangements that protected only 4.5% of the region.

ACTIVITIES GUIDE (see relevant <i>Zoning Plans and Regulations</i> for details)		General Use Zone	Habitat Protection Zone	Conservation Park Zone	Buffer Zone	Scientific Research Zone ²	Marine National Park Zone	Preservation Zone
Aquaculture	Permit	Permit	Permit ¹	×	×	×	×	×
Bait netting	✓	✓	✓	×	×	×	×	×
Boating, diving, photography	✓	✓	✓	✓	✓ ²	✓	×	×
Crabbing (trapping)	✓	✓	✓ ³	×	×	×	×	×
Harvest fishing for aquarium fish, coral and beachworm	Permit	Permit	Permit ¹	×	×	×	×	×
Harvest fishing for sea cucumber, trochus, tropical rock lobster	Permit	Permit	×	×	×	×	×	×
Limited collecting	✓ ⁴	✓ ⁴	✓ ⁴	×	×	×	×	×
Limited spearfishing (snorkel only)	✓	✓	✓ ¹	×	×	×	×	×
Line fishing	✓ ⁵	✓ ⁵	✓ ⁶	×	×	×	×	×
Netting (other than bait netting)	✓	✓	×	×	×	×	×	×
Research (other than limited impact research)	Permit	Permit	Permit	Permit	Permit	Permit	Permit	Permit
Shipping (other than in a designated shipping area)	✓	Permit	Permit	Permit	Permit	Permit	Permit	×
Tourism programme	Permit	Permit	Permit	Permit	Permit	Permit	Permit	×
Traditional use of marine resources	✓ ⁷	✓ ⁷	✓ ⁷	✓ ⁷	✓ ⁷	✓ ⁷	✓ ⁷	×
Trawling	✓	×	×	×	×	×	×	×
Trolling	✓ ⁵	✓ ⁵	✓ ⁵	✓ ^{5,8}	×	×	×	×

PLEASE NOTE: This guide provides an introduction to Zoning in the Great Barrier Reef Marine Park. Relevant Queensland Marine Park Zoning Plans or the Queensland Environmental Protection Agency should be consulted for confirmation of use or entry requirements.

1. Restrictions apply to aquaculture, spearfishing and harvest fishing for aquarium fish, beachworm and coral in the Conservation Park Zone.
2. Except for One Tree Island Reef (SR-23-2010) and Australian Institute of Marine Science (SR-19-2008) which are closed to public access and shown as orange, all other Scientific Research Zones are shown as green with an orange outline.
3. Limited to 4 catch devices (eg. crab pots, dillies and inverted dillies) per person.
4. By hand or hand-held implement and generally no more than 5 of a species.
5. Maximum of 3 lines/rods per person with a combined total of 6 hooks per person.
6. Limited to 1 line/rod per person and 1 hook per line. Only 1 dory detached from a commercial fishing vessel.
7. Apart from traditional use of marine resources in accordance with s.211 of the *Native Title Act 1993*, an accredited Traditional Use of Marine Resources Agreement or permit is required.
8. Pelagic species only. Seasonal Closures apply to some Buffer Zones.

Detailed information is contained in the Great Barrier Reef Marine Park Zoning Plan and Regulations.

- Permits are required for most other activities not listed above.
- Commonwealth owned islands in the Great Barrier Reef Marine Park are zoned "Commonwealth Islands Zone" - shown as cream.
- All Commonwealth Islands may not be shown.
- Special Management Areas may provide additional restrictions at some locations.
- The Zoning Plan does not affect the operation of s.211 of the *Native Title Act 1993*.

ACCESS TO ALL ZONES IS PERMITTED IN AN EMERGENCY.

Figure 2.2: Activities guide for zones within the GBRWHA marine parks. © Commonwealth of Australia (July 2004).

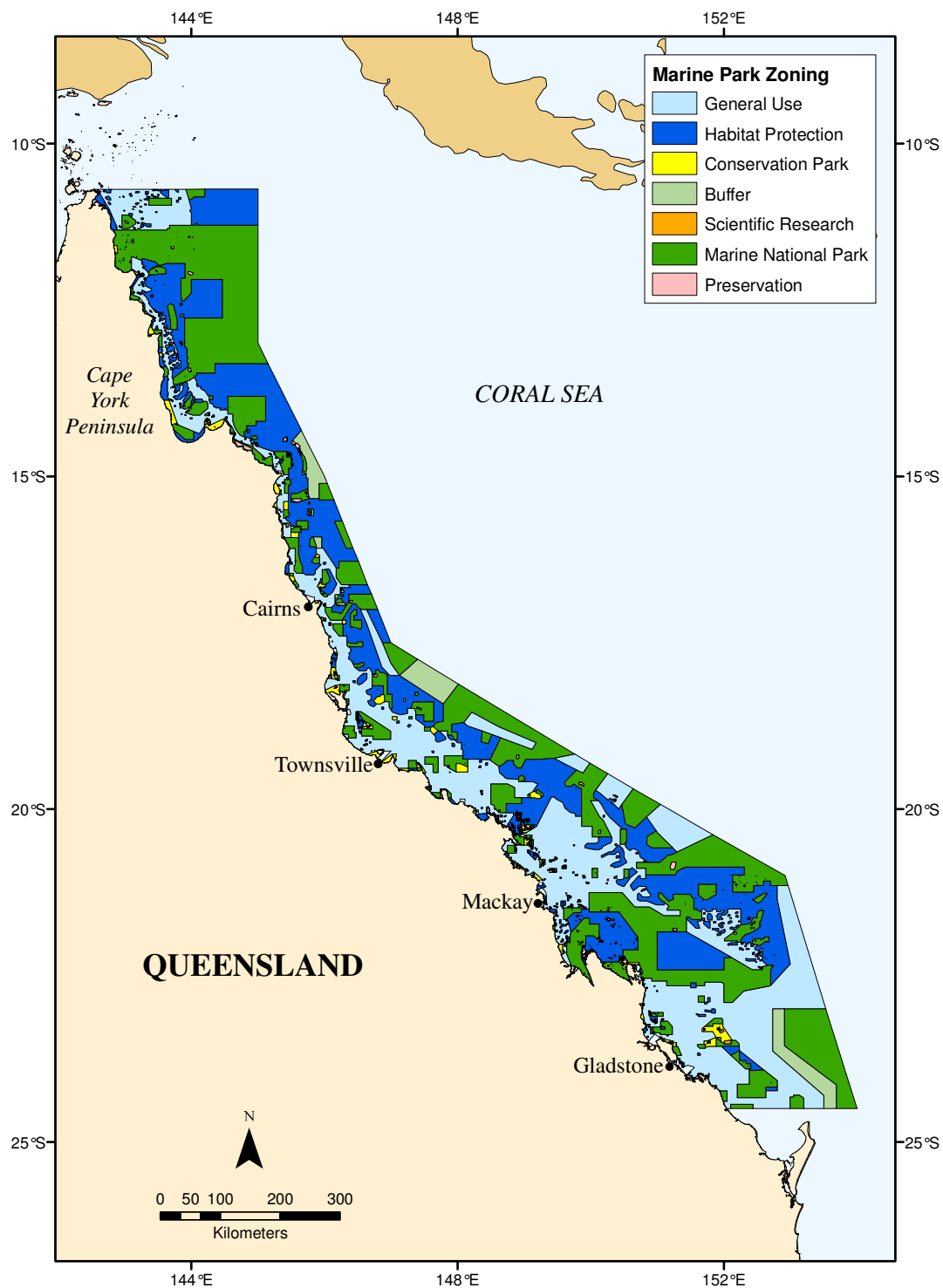


Figure 2.3: The Australian Government's Great Barrier Reef Marine Park Zoning Plan and Queensland's contiguous Great Barrier Reef Coast Marine Park Zoning Plan.

Seagrass

Seagrasses are specialised marine flowering plants that grow in the estuarine and coastal environments of most of the world's continents. There are relatively few species globally (about 60) and these are grouped into 13 Genera and 5 Families. Most species are entirely marine although some species (such as *Enhalus acoroides*) cannot reproduce unless emergent at low tide. Seagrasses survive in a range of conditions that can encompass fresh water, estuarine, marine or hypersaline habitats. Multiple parameters control the distribution of seagrasses, and include: (1) physical drivers that regulate the physiological activity of seagrass (i.e. salinity, wave energy, temperature, currents, depth, substrate and day length); (2) natural phenomena that limit the photosynthetic activity of seagrass (i.e. light, nutrients, epiphytes and disease); and, (3) anthropogenic activities and inputs that inhibit the access to available plant resources (i.e. nutrient and sediment loading) and/or have a physical impact on seagrass meadows (Short et al. 2001). Combinations of these parameters permit, encourage or eliminate seagrass from a specific area.

Seagrass habitats form some of the most productive ecosystems on earth (Waycott et al. 2009) and provide a variety of ecosystem services that had an estimated global value of over AU\$3.8 billion per annum in 1997 (Constanza et al. 1997). Seagrasses are breeding grounds and nurseries for crustacean, finfish and shellfish populations that form the basis of economically valuable subsistence and/or commercial fisheries in many parts of the world (Coles and Fortes 2001). Seagrasses provide food for green sea turtles, dugongs, fish species, and waterfowl; and seagrass habitats are the basis of a detrital food chain (Walker et al. 2001). Seagrass habitats stabilise the seafloor, reduce water currents and protect coastal areas from physical disturbance. In tropical Australia, sediment trapping within seagrass meadows is largely insignificant as intertidal meadows are predominantly ephemeral and comprised of small species of low biomass (Mellors et al. 2002). Tropical seagrass habitats are closely associated with coral reefs and mangroves; and strong linkages among these three habitats make any loss of seagrass a factor in the degradation of tropical coastal ecosystems.

Fifteen percent of seagrass species are considered threatened (Randall-Hughes et al. 2009) and seagrass habitats are reported to be declining worldwide (Waycott et al.

2009). Destruction of seagrass habitats has resulted from anthropogenic activities, including: dredging, coastal development, damage associated with over exploitation of coastal resources, recreational boating activities and nutrient and sediment loading from adjacent land catchments (Cambridge and McComb 1984; Short and Wyllie-Echeverria 1996; Coles et al. 2003; Orth et al. 2006; Waycott et al. 2009). While coastal seagrass habitats in the GBRWHA have remained relatively stable in distribution (Coles et al. 2007), increases in coastal human population density and resource developments in mining and agriculture inevitably lead to negative pressures on the adjacent marine environment.

As explained by Carruthers et al. (2002), seagrass habitats in the GBRWHA are characterised by low nutrient concentrations and high disturbance, and are spatially and temporally dynamic (ephemeral). There are fifteen species of seagrass in the GBRWHA and they occur in four major habitat types: river estuaries, coastal, reef and deepwater (Carruthers et al. 2002). It is estimated that more than 4,000 km² of coastal seagrass meadows are in GBRWHA waters shallower than 15 m and approximately 40,000 km² of the seafloor deeper than 15 m is expected to have some seagrass (Coles et al. 2007). Together, these meadows represent about 36% of the total recorded area of seagrass in Australia.

The boundaries of the GBRWHA rarely extend into river estuaries and the State and Commonwealth marine parks primarily restrict activities in coastal, reef and deepwater seagrass habitats. Seagrass is protected under Queensland law through provisions of the *Fisheries Act* 1994. The provisions state that destruction, damage or disturbance of seagrass without prior approval from Fisheries Queensland is prohibited. Seagrass is protected on all private and public lands, and protection applies to plants that are alive or dead. Penalties apply to any unauthorised disturbances that impact on seagrass, and can be as high as AU\$750,000. Queensland's declared Fish Habitat Area network protects seagrass from physical disturbance associated with coastal development. Declared Fish Habitat Areas are a multiple-use marine protected area (MPA) where activities such as legal fishing, scientific research and boating are allowed, and the development of infrastructure is restricted.

Dugongs

The dugong (*Dugong dugon*) is the only strictly marine herbivorous mammal, the only member of the family Dugongidae, and, together with the three species of manatee, one of four living species of the order Sirenia. Dugongs can live for about 70 years, and reach a maximum size of between 2 – 3 m. Female dugongs have a minimum pre-reproductive period of between 6 – 17 yr (Marsh et al. 1984; Boyd et al. 1999; Kwan 2002) and a mean calving interval of between 2.4 – 7 yr. The age at first breeding and the time interval between calves is likely to be dependent on the status of food supply (Boyd et al. 1999; Marsh and Kwan 2008).

Dugongs are bottom feeders, and seagrasses are the most important component of their diet. Dugongs graze predominantly on intertidal and subtidal tropical and subtropical seagrass meadows, but are also known to feed on macro-invertebrates and algae. They feed on nine of the ten seagrass genera and on most of the species of seagrass that occur within their range (Green and Short 2003). Dugongs typically feed on several species of seagrass in mixed species seagrass meadows (Johnstone and Hudson 1981). The selection of seagrass habitats by dugongs is influenced by multiple factors including fibre, nitrogen and starch content and biomass (Lanyon and Sanson 2006a and b).

Dugongs are found in the shallow, protected coastal waters of about 40 countries in the tropical and subtropical Indian and Pacific Oceans. At a global scale, their distribution is characterised by relict populations separated by large areas where dugongs are believed to be close to extinction or extinct locally (Marsh et al. 2002). A recent synthesis suggests that dugongs are declining or locally extinct in at least a third of its range; of unknown status in nearly half; and possibly stable in the remainder (Marsh 2008). Dugongs are listed under the: IUCN Red List of Threatened Species as vulnerable to extinction; Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora 1973; and Appendix II of the Convention for Migratory Species of Wild Animals 1979.

Based on the length of the coastline, around a quarter of global dugong habitats occur in northern Australia's waters between Moreton Bay in Queensland and Shark Bay in Western Australia (Figure 2.4), and the region is internationally recognised as

supporting the most globally significant remaining dugong populations (Marsh et al. 2002 and 2003). Aerial surveys that provide information on dugong distribution and abundance have been conducted over >120,000 km² of the coastal waters of northern Australia since the 1980s (Figure 2.4). The surveys use a standardised systematic transect technique that attempts to correct for perception and availability biases (Marsh and Sinclair 1989; Pollock et al. 2006). Population estimates from the >120,000 km² surveyed since 2005 (Figure 2.4) total approximately 68,700 dugongs. Results of the 20-year time series of aerial surveys suggest that dugong populations are stable in most of the dugongs range in Australia; and fluctuate in Torres Strait, Hervey and Moreton Bays (Figure 2.4). These fluctuations are likely due to dugongs moving between survey regions. In Australian waters dugongs exhibit high levels of genetic diversity and are partitioned into two distinct lineages. One lineage is restricted geographically to Queensland and the Northern Territory; and the other lineage occurs across the entire range of the dugong (Blair *in review*; Figure 2.4).

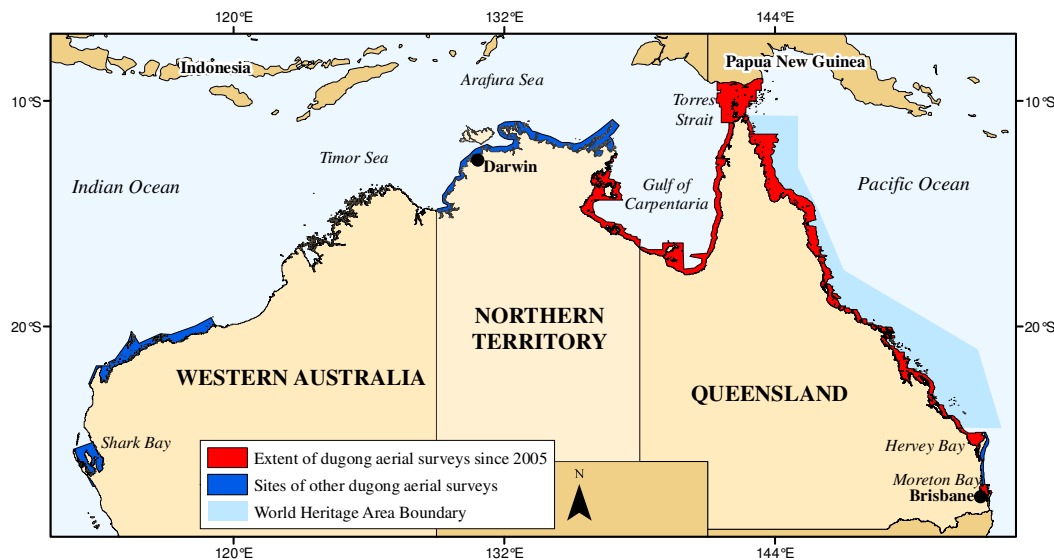


Figure 2.4: Dugongs are distributed in the coastal waters of northern Australia between Moreton Bay and Shark Bay. The extent of dugong aerial surveys is shown in red and blue.

Significant populations of dugongs are found in the waters of the GBRWHA (Marsh et al. 2002) and were an explicit reason for the region's World Heritage listing (GBRMPA 1981) due to their high biodiversity value, very high cultural value to Indigenous Australians and high intrinsic value to non-Indigenous Australians. Dugongs generally occur in the shallow, protected inshore waters of the GBRWHA; but are also known to exploit deep water seagrass habitats (Lee Long and Coles 1997; Sheppard et al 2006) and the estuarine creeks and streams that are adjacent to the GBRWHA (Lawler et al. 2002).

Dugongs undertake large scale movements (defined as moves > 15 km) between sites of significant seagrass habitat (Gales et al. 2004; Marsh et al. 2004; Marsh et al. 2005; Sheppard et al. 2006). In the GBRWHA, dugongs apparently make large scale movements in response to changes in the availability of forage as a result of cyclone and flood events which can destroy seagrass meadows (Preen and Marsh 1995). In latitudes higher than the GBRWHA such as Moreton Bay (Preen 1992) and Hervey Bay in southeast Queensland (Sheppard et al. 2006) and Shark Bay in Western Australia (Holley et al. 2006; Figure 2.4), dugongs exhibit seasonal movements in response to low water temperatures in winter like those of their relative, the Florida manatee (Deutsch et al. 2003). The minimum temperature thresholds triggering such movements are not reached in the GBRWHA and seasonal movements have not been recorded.

Dugong populations along the urban coast of the GBRWHA are only a small fraction of pre-European levels (Marsh et al. 2005; Daley et al. 2008) which is most likely a result of multiple factors including: the commercial dugong oil industry (1847 - 1967); Indigenous hunting; poaching; incidental drowning in commercial gill-nets and shark nets set for bathers protection; vessel strike; and habitat loss (Marsh et al. 1996 and 2005). The original zoning plans developed for the GBRWHA in the 1980s protected some important dugong habitats (seagrass meadows) in 'no-take' and 'no-go' zones. The Australian and Queensland Governments introduced emergency measures in August 1997 to further protect dugongs on the urban coast of the GBRWHA and adjacent Hervey Bay (Figure 2.5) after a serious decline in their population was documented (Marsh et al. 1996 and 2005). The emergency measures established 16 Dugong Protection Areas (DPAs), declared under Queensland legislation. DPAs provide varying levels of protection for dugongs through spatial and gear restrictions

and prohibitions on the use of some types of fishing nets (Marsh 2000). The potential for the DPAs to protect dugongs was questioned for several reasons (Marsh 2000), including the dugong's mobility; individual animals can move hundreds of kilometres in a few days (Sheppard et al. 2006). In 1999, similar measures were used to protect the dugong's seagrass habitats from trawling (Turner et al. 1999; Gray 2000). The 2004 changes to zoning arrangements in the GBRWHA further upgraded dugong protection in accordance with Biophysical Operating Principle 8 (Fernandes et al. 2005), by increasing the amount of area closed to commercial netting and trawling as demonstrated in detail in this thesis.

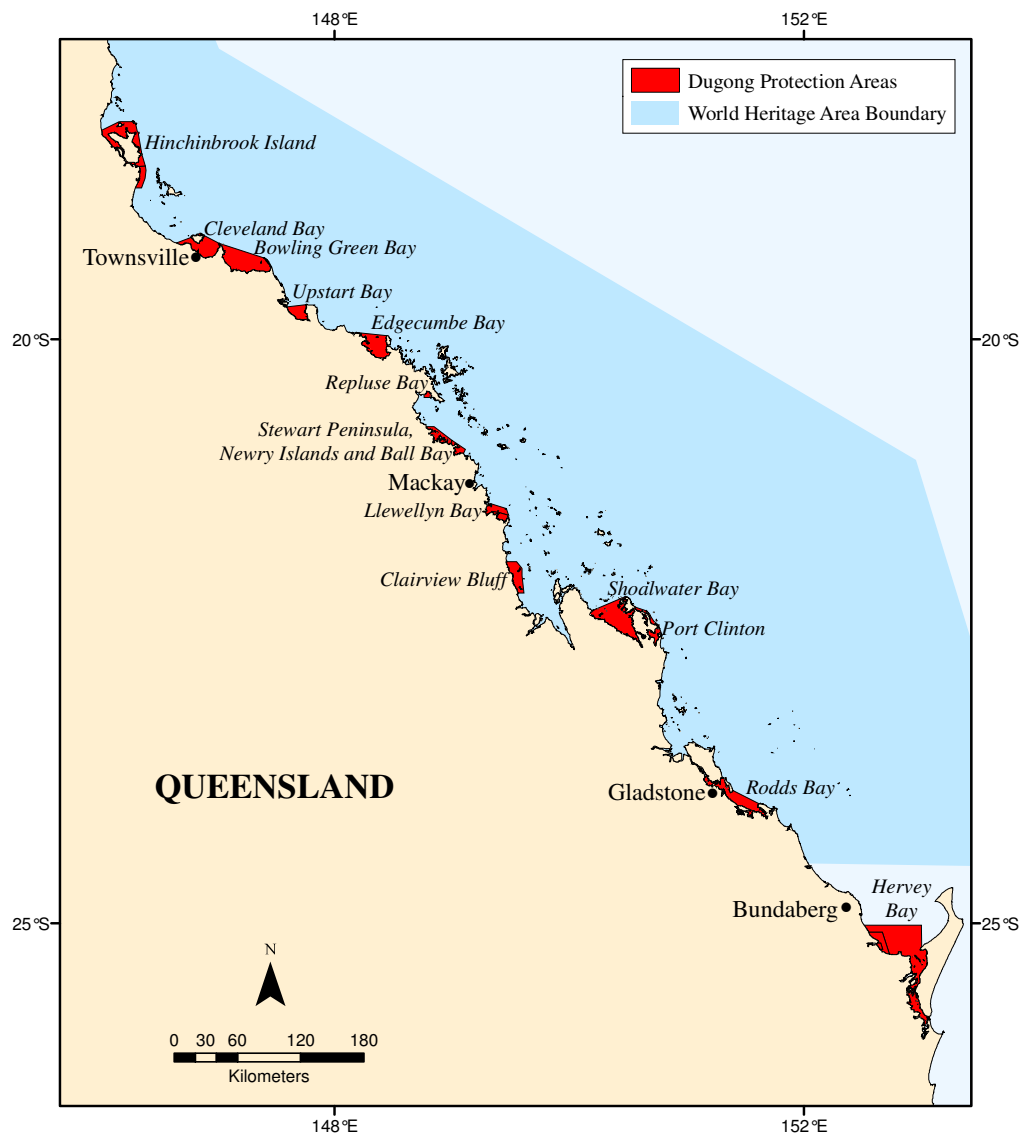


Figure 2.5: Location of Dugong Protection Areas in the southern GBRWHA and Hervey Bay, Queensland.

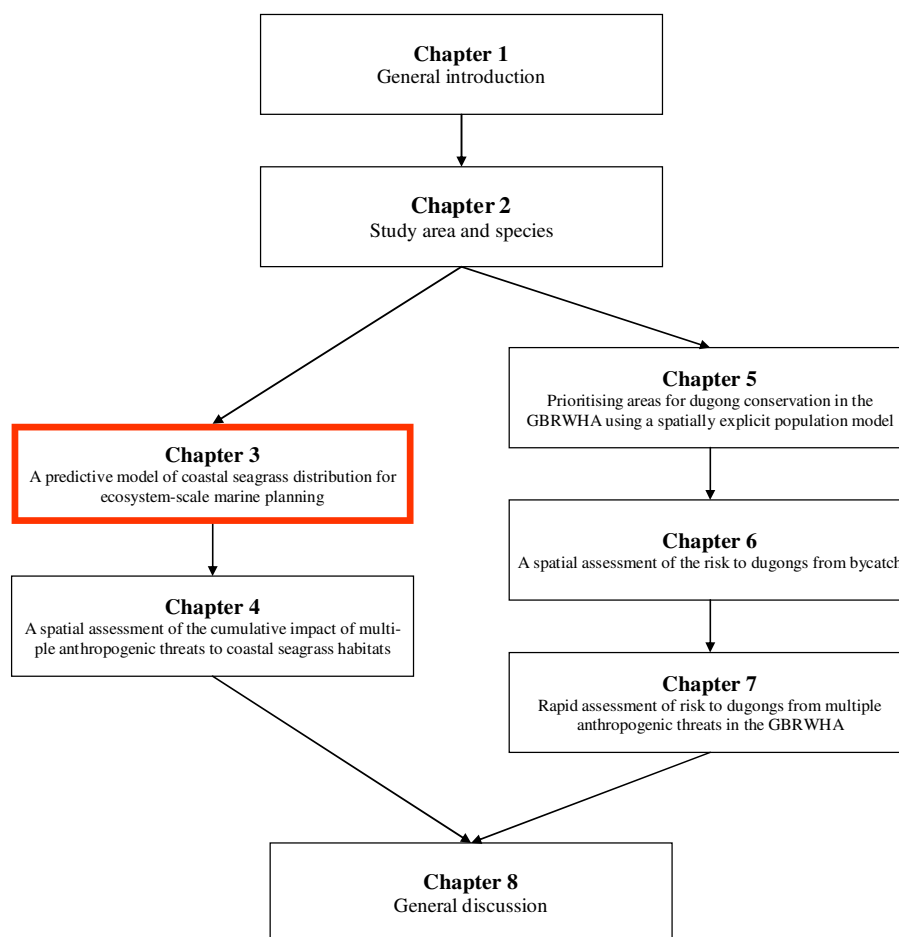
Chapter Summary

- The GBRWHA supports a variety of habitats and species, including 15 species of seagrass and the threatened dugong. The region is jointly managed by the Queensland and Australian Governments as a multiple-use MPA.
- The GBRWHA is functionally divisible into two distinct management regions based upon biophysical and demographic attributes: the developed urban coast and the remote Cape York region.
- Coastal seagrasses occur in the intertidal and subtidal habitats of the GBRWHA in waters to approximately -15 m below mean sea. Coastal seagrass habitats are characterised by low nutrient concentrations and high disturbance, and are spatially and temporally dynamic (ephemeral).
- Significant populations of dugongs are found in the coastal waters of the GBRWHA and were an explicit reason for the region's World Heritage listing due to their high biodiversity value, very high cultural value to Indigenous Australians and high intrinsic value to non-Indigenous Australians. Dugongs are the only strictly marine herbivorous mammal, and seagrasses are the most important component of their diet.

Chapter 3

A predictive model of coastal seagrass distribution for ecosystem-scale marine planning¹

In this chapter, I determine the presence and distribution of coastal seagrasses in the GBRWHA by generating a GIS-based habitat suitability model. The Bayesian belief network quantifies the relationship (dependencies) between seagrass habitats and eight environmental drivers. Outputs of the model include probabilistic GIS-surfaces of seagrass habitat suitability in two seasons and at a planning unit of cell size 2 km * 2 km.



¹ Grech, A. and Coles, R. *in press*. An ecosystem-scale predictive model of coastal seagrass distribution. *Aquatic Conservation*.

Introduction

As discussed in Chapter 1, safeguarding the delivery of marine ecosystem services requires the maintenance of ecological processes that underpin the functioning of marine ecosystems (Agardy, 1994; Daily, 1997; Roberts et al., 2003). The data necessary to inform management at these scales are far less organised and available for most marine environments than for terrestrial environments (Carr et al., 2003). Spatial information on the distribution of habitats and species is difficult to collect in marine environments at the scale of ecosystems as it is expensive and logistically difficult (Ban, 2009).

Data on the occurrence of species are commonly available in two forms: point localities and predicted distributions. Point locality data are more readily available, easy to use and have low rates of commission errors (Rondini et al. 2006). However, using point locality data to delineate the occurrences of species is challenging when: (1) data sets are incomplete; (2) species occur over broad spatial scales; and, (3) species respond both spatially and temporally to environmental change (i.e. ephemeral species).

Predictive habitat distribution models informed by point locality data can overcome some of these challenges (Rondini et al. 2006) by quantifying species-environment relationships and generating spatially explicit models of species distribution and habitat structure (Guisan and Zimmermann 2000). Modeling techniques have an advantage over a static mapping approach as they can account for habitat response to environmental change both spatially and temporally. Consequently, predictive habitat models are frequently used in conservation planning (Margules and Pressey 2000; Guisan and Thuiller 2005). Predictive habitat models are increasingly required to predict the distribution of species at ecosystem-scales because of the lack of resources available to collect complete field data sets at that scale.

Information on the distribution of coastal seagrass habitats in the Great Barrier Reef World Heritage Area (GBRWhA) of Queensland, Australia has been collected as point localities since 1984 by the Northern Fisheries Centre (Fisheries Queensland). Seagrasses are mapped using expensive field-based surveys described in detail by McKenzie et al. (2001). Remote sensing instruments are unable to reliably detect seagrasses in the GBRWhA because of the turbid coastal waters and the low biomass of

tropical seagrass species. Due to the large geographic extent of coastal waters within the GBRWHA (~ 22,600 km²) and the high monetary costs associated with field-based surveys, mapping of coastal seagrass habitats is generally restricted to: (1) regions where seagrass is known to occur; (2) sites that require habitat assessments because of existing or proposed infrastructure development (e.g. ports); and, (3) the ‘urban coast’ where human population densities are high and coastal areas are readily accessible (Figure 1.1). Although when compared with most of the world the seagrass information for the GBRWHA is data rich, point locality data sets are inadequate for informing the management of seagrasses at an ecosystem-scale. The field-based surveys and static mapping approach results in a patchwork distribution of data sets that do not cover the entire range of coastal seagrass habitats of the GBRWHA (i.e. are spatially and temporarily biased), and the data sets do not effectively represent spatio-temporal changes in seagrass distribution.

Fonseca et al. (2002), Kelly et al. (2001), Lathrop et al. (2001), and Holmes et al. (2007) quantify seagrass species-environment relationships at fine spatial scales and develop habitat suitability maps for relatively small geographic areas (< 1000 km²). Coles et al. (2009) model the distribution of deepwater seagrasses in waters greater than -15 m below mean sea level at a GBRWHA scale (approximately 348,000 km²). In this chapter, I used spatial information on the distribution of coastal seagrasses and predictor variables along with ecological theory and expert knowledge to inform the design of a Bayesian belief network, and to develop a predictive habitat model of seagrasses at the scale of the coastal GBRWHA (~ 22,600 km²). This approach builds on the existing seagrass point locality data sets to provide a representation of coastal seagrass distribution at a whole of the GBRWHA scale and to inform management at that scale.

Methods

As explained in Chapter 2, the climate of the GBRWHA is influenced by monsoonal wind and rainfall patterns. Strong south-easterly winds dominate during the dry season (April – October). Weaker variable winds are more common during the wet season (November – March). The GBRWHA is divisible into four distinct sections¹ based on

¹ The four sections referred to in this study are not the four geo-political management regions that are stipulated in the *Great Barrier Reef Marine Park Zoning Plan 1975*.

their biophysical attributes (Maxwell 1968): Northern GBR, Wet Tropics, Dry Tropics and Southern GBR (Figure 3.1). I modelled the distribution of intertidal and subtidal seagrass habitats in waters to approximately -15 m below mean sea level across all four sections, as shown in Figure 3.1. I did not model the distribution of seagrasses in deep water (waters deeper than -15 m bathymetry), on reefs or in estuaries and creeks that are adjacent to the GBRWHA (*see* Chapter 2 page 29).

There are multiple drivers of coastal seagrass distribution in the GBRWHA, including: temperature, salinity, bathymetry, substrate, day length, light, nutrients, water currents, relative wave exposure, and epiphytes and disease, tidal range, and terrestrial runoff from land catchments (Coles et al. 2007). Combinations of these parameters encourage or eliminate seagrass presence at varying spatial and temporal scales (Coles et al. 2007). These biophysical parameters are not independent. For example, the availability of nutrients is dependent on rainfall and the presence of a river mouth. The dependencies between drivers, and the degree of influence of various drivers on the distribution of coastal seagrass habitats, are poorly understood. Uncertainty in the degree of influence results from limited information on the relationships among drivers, and on the state of coastal seagrass distribution when the supply and strength of individual drivers and combinations of drivers change.

Bayesian belief networks are effective for modelling when some data are known and certain and other data are incomplete or uncertain to various degrees. Bayesian belief networks are probabilistic graphical models that represent variables and probabilistic independencies between variables (Ben-Gal 2007) and provide consistent semantics for representing causes and effects. Bayesian belief networks are used to make probabilistic inferences about multiple drivers (predictor variables) of species distribution that are characterised by complexity and uncertainty (McCarthy 2007), and have interactions that are not fully understood (Guisan and Zimmermann 2000). When there is information on the spatial distribution of a species and its predictor variables, a Bayesian belief network can learn conditional probabilities from real data. Conditional probabilities are the likelihood of a species or driver occurring based upon the presence or absence of another or multiple predictor variables.

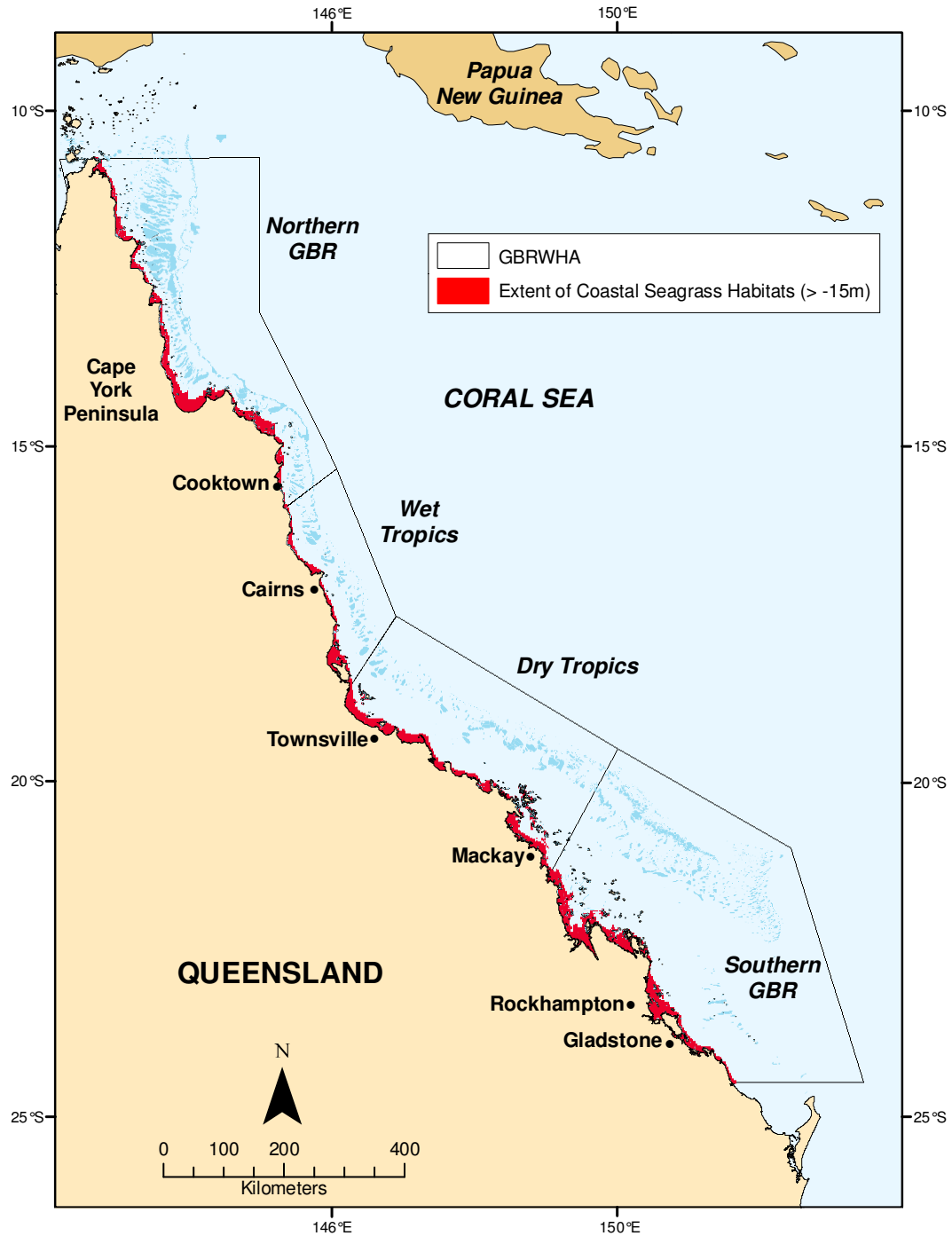


Figure 3.1: The GBRWHA is divisible into four distinct sections based on their biophysical attributes, as illustrated: Northern GBR, Wet Tropics, Dry Tropics and Southern GBR. The four sections referred to in this study are not the four geo-political management regions that are stipulated in the *Great Barrier Reef Marine Park Zoning Plan 1975*. The region shaded in red is the extent of coastal waters to approximately -15 m below mean sea level.

Data preparation

I sourced presence/absence information for the dependent variable seagrass from the Northern Fisheries Centre (Queensland Primary Industries and Fisheries) in Cairns, Australia. The total number of survey sites is 11,562 (674 in the Northern GBR; 1959 in the Wet Tropics; 4526 in the Dry Tropics; and 4403 in the Southern GBR). Seagrass was present in less than half of the surveyed sites (43.4%). All surveys were conducted between 1984 and 2001, and approx. 80% of sites were surveyed in November and December (late dry/early wet season). The spatial footprint of the individual survey sites was at least ~ 5 m * 1 m. All of the survey sites (11,562) were used to inform the Bayesian belief network.

The distribution and abundance of coastal seagrasses in the GBRWHA varies seasonally, and seagrass species assemblages change across the latitudinal gradient (McKenzie et al. 1998; Rasheed, 2000; Coles et al. 2002). I split the seagrass point locality data into wet (November – April) and dry (May – October) seasons and across four sections of the GBRWHA coast (Figure 3.1). Season and section were included as predictor variables in the Bayesian belief network (Figure 3.2). I tested the spatial dependency in the seagrass data set by means of the Moran's I statistic of the Spatial Analyst© extension of ArcGIS® 9.2 (Environmental Systems Research Institute 2006). The observations were interdependent and it was therefore unnecessary to account for spatial autocorrelation in the data.

I used a planning unit of 2 km * 2 km to model the Bayesian belief network predictions at the scale of the entire GBRWHA coast (~22,600 km²). This scale was chosen as it was the spatial scale of the majority of data sets that extend to the limits of the GBRWHA, and was fine enough to represent coastal seagrass habitat features without compromising computing power. To enhance the accuracy and predictive power of the model, the predictive variables were reduced (Guisan and Zimmermann 2000) by eliminating those variables that addressed one or more of the following criteria: (1) operated at a spatial scale finer than the output habitat suitability maps (e.g. factors that influenced the dispersal of seagrass propagules, epiphytes and disease); (2) were not considered to be a major driver of seagrass distribution in the GBRWHA (e.g. day

length and nutrients; Schaffelke et al. 2005; Collier and Waycott 2009); and (3) were dependent on other variables (e.g. light).

Light is a major driver of coastal seagrass distribution in the GBRWHA (Collier and Waycott, 2009), but there is no spatial information on light at the scale of the GBRWHA. The factors that determine light availability (bathymetry, flood plumes and wave exposure) were included as predictor variables in the model. Cyclone and storm activities were not included as predictor variables in the coastal seagrass model because: (1) all regions of the GBRWHA are likely to be affected by physical disturbance from cyclones and storms (Waycott et al. 2007); and (2) any potential regional differences would be accounted for in the final GIS-habitat suitability maps as separate models were developed for four sections of the GBRWHA.

The eight predictor variables of coastal seagrass distribution included in the Bayesian belief network were: season, section, bathymetry, substrate, sea surface temperature, tidal range, spatial extent of flood plumes and wave exposure. Table 3.1 provides a description of the data sets that delineate the spatial distribution of predictor variables, a justification for their use, and their reclassification method and discrete states. Figures showing the spatial distribution of the predictor variables are provided in Appendix A.

Relative wave exposure index

A relative wave exposure index is a model of the relative degree of force applied to a location due to the impact of wind-generated waves. I developed a model of relative wave exposure in the coastal GBRWHA to provide an index of the hydrodynamic forces that drive coastal seagrass distribution. I used a modified version of the equations developed by Keddy (1982) and Murphey and Fonseca (1995) to develop the index. Wind exposure information for 30 weather stations along the GBRWHA coast was sourced from the Australian Bureau of Meteorology. Stations collect data on wind speed and direction twice daily (0900 and 1500). I derived the model of relative wave exposure from wind station data collected by the Australian Bureau of Meteorology from May 2005 - April 2006. This time period did not feature abnormal wind conditions, and I therefore considered it to be an appropriate surrogate measurement of mean wind conditions in the wet and dry seasons.

Table 3.1: Description of the six predictor variables that were nodes in the Bayesian belief network. Figures showing the spatial distribution of predictor variables are provided in Appendix A.

<i>Predictor variable and node identifier</i>	<i>Justification for predictor variable</i>	<i>Source of data set</i>	<i>Data model and attributes</i>	<i>Reclassification method</i>	<i>Discrete states</i>
Bathymetry (<i>Bathymetry</i>)	Bathymetry controls the amount of light available for photosynthesis.	Great Barrier Reef Marine Park Authority (Lewis 2001)	Terrain model of the entire Great Barrier Reef, cell size 500 m * 500 m.	Equal interval	high (< -10 m) medium (-5 - -10 m) low (0 - -5 m)
Substrate (<i>SedBasin</i>)	A substrate feature that drives the distribution of coastal seagrass habitats is sedimentary basins; a low and usually sinking region filled with sediments from adjacent catchments that inhibit the colonisation and growth of seagrass habitats.	Geoscience Australia 2007	Vector data set of marine sediments delineated at the scale of Australia's Marine Jurisdiction.	Presence absence	sedimentary basin present sedimentary basin absent
Sea surface temperature (<i>SST</i>)	Sea surface temperature affects the community structure of seagrass habitats.	Australian Commonwealth Scientific and Research Organisation 2007	Mean sea surface temperature of the Australian region, cell size 2 km * 2 km.	Natural breaks ⁴	high (> 29.5 °C) medium (28.0 – 29.5) low (< 28.0 °C)
Tidal range (<i>Tides</i>)	Tidal range affects the availability of habitat for coastal seagrasses and their exposure.	Australian Maritime College (Hopley et al. 2007)	Indian Springs tidal range, cell size 2 km * 2 km.	Natural breaks	high (> 7 m) medium (3 – 7 m) low (< 3 m)
Spatial extent of flood plumes (<i>Rivers</i>)	Flood plumes are an important source of nutrients and sediments for coastal seagrass habitats.	Australian Centre for Tropical Freshwater Research (Devlin et al. 2001)	Vector data set of the spatial extent of flood plumes in the GBRWHA.	Presence absence	flood plumes present flood plumes absent
Relative wave exposure (<i>REI</i>)	Hydrodynamic forces are a physical driver of seagrass distribution.	James Cook University (Grech)	Relative wave exposure index of the coastal GBR, cell size 2 km * 2 km.	Natural breaks	high (> 500) medium (170 – 500) low (< 170)

⁴ The natural breaks function is the most appropriate tool to categorize the GIS-layers in the absence of biophysical data. Classes are based on natural groupings that are inherent in the data, and breaks points are identified by grouping similar values and maximizing the differences between classes. The GIS-layers are therefore classified into groups that represent relatively big jumps in the data values.

I calculated the relative wave exposure index for each weather station in ArcView® GIS 3.3 (Environmental Systems Research Institute 2002) using the following equation (Keddy 1982; Murphey and Fonseca 1995):

$$REI = \sum_{i=1}^{16} (V_i \times P_i \times EF_i)$$

Where REI = relative exposure index, i = i th compass heading (1 to 16 [N, NNE, NE, etc.]), in 22.5° increments, V = average monthly wind speed (m s^{-1}), P = percent (%) frequency wind occurs in the i th direction, and EF = effective fetch (m). I computed effective fetch by: (1) measuring fetch (in metres) along two lines radiating out from either side of the i th compass heading at increments of 11.25°, including the i th compass heading ($n = 5$); and, (2) summing the product of the fetch and multiplying it by the cosine of the angle of departure from the i th heading over each of the five lines and dividing it by the sum of the cosine of all the angles (Fonseca et al. 2002). When fetch was greater than 50 km, I assumed that fetch was unlimited in the i th direction (Puotinen 2005).

The output of the above calculations was a relative wave exposure index for both the wet (November – April) and dry (May – October) seasons with a cell size of 2 km * 2 km (Appendix A). I reclassified the relative exposure index for the wet and dry models using the natural breaks classification method to delineate three discrete states of exposure: high (> 500), medium (170 – 500) and low (< 170).

Model fitting and evaluation

After assembling the series of spatial indicators; seagrass ecological theory and expert scientific knowledge were used to underpin a Bayesian belief network that explored the extent that predictor variables determined seagrass distribution. The Bayesian belief network was assembled in SamIam© (Automated Reasoning Group, University of California 2004) by identifying structural dependency relationships between predictor variables, and between predictor variables and the presence of coastal seagrass habitats. For example, wave exposure and sea surface temperature vary seasonally. Season is then a parent node in the Bayesian belief network, and wave exposure and sea surface temperature child nodes. The EM learning tool of SamIam© was used to learn

conditional probabilities within the network via the expectation maximisation algorithm.

The strength of the relationship between variables was summarized in a conditional probability table. The conditional probability table specifies the conditional probability of the child nodes being in a particular state given the states of its parents and the likelihood of seagrass presence/absence given a particular state of all predictor variables. The results of the conditional probability table were explored via a sensitivity analysis that identified the network components that had the greatest influence on coastal seagrass habitat distribution. The sensitivity analysis was conducted by systematically varying the values of individual network components in SamIam© to determine how this variation affected the seagrass nodal variable. The values for the states in each predictor variable node were varied over their possible ranges, and all other nodes were held constant at their most likely value.

Spatial predictions and assessment of model applicability

The conditional probability table was exported to ArcGIS® 9.2 (Environmental Systems Research Institute 2006), and joined to spatial information on seagrass predictor variables (Table 3.1) to create GIS-based habitat suitability maps in the wet and dry seasons and across four sections of the GBR (scale 2 km * 2 km). The Bayesian belief network was tested for its predictive capacity using a re-substitution approach because independently collected data on seagrass distribution at the scale of the GBRWHA does not exist in a suitable form. A random sub-sample of observations constituting 75% of the seagrass point locality data were used to inform the Bayesian belief network. The GIS-based habitat suitability maps were then tested against the remaining 25% of seagrass point locality data.

Results

The Bayesian belief network (Figure 3.2) showed conditional dependencies between: (1) season and wave exposure and sea surface temperature; (2) section of the GBR and sea surface temperature, substrate, spatial extent of flood plumes, tidal range and bathymetry; and (3) seagrass presence and wave exposure, sea surface temperature, substrate, spatial extent of flood plumes, tidal range and bathymetry.

Table 3.2 shows the total number of point locality data where seagrass was present and absent within various levels of predicted likelihoods of seagrass presence from the model using a re-substitution approach. The predictive rate is the proportion of point locality data where seagrass was actually present within the various levels of predicted likelihoods. Seagrass planning units with predicted likelihood of seagrass presence > 0.6 were found to have a high predictive rate (and low error; Table 3.2). As the predicted likelihood of seagrass habitat presence decreased, so did the predictive rate of the model. For example, when seagrass habitat presence was predicted at between 0.0 and 0.1 the Bayesian belief network had a predictive rate of 6.7%; and when habitat presence was predicted at between 0.7 and 0.8, the Bayesian belief network had a predictive rate of 68% (Table 3.2). In all levels of predicted probability, the proportion of seagrass point locality data where seagrass was present closely matched the likelihood of seagrass presence predicted by the model.

The conditional probability table was explored using a sensitivity analysis that highlighted how much the mean value of the seagrass node was influenced by a single finding at each of the other nodes in the network (Figure 3.3). The range of variation in the seagrass nodal variable when values for the states in each predictor variable node were varied over their possible ranges was ~0.4. The Bayesian belief network predicted that the likelihood of coastal seagrasses being present at any given planning unit was first determined by the tidal range of that site (a low tidal range increased the likelihood of seagrass presence); followed by the relative exposure of that site to wave activity (a low wave exposure increased the likelihood of seagrass presence). At predicted probabilities of about 0.23 and higher, coastal seagrass presence was also determined by sea surface temperature, substrate, presence of rivers and bathymetry.

The GIS-based habitat suitability maps for coastal seagrasses in the wet and dry seasons and across four sections of the GBRWHA are shown in Figures 3.4 and 3.5. The probabilistic surfaces were converted into presence/absence data with probability thresholds set a priori at : > 0.0001; > 0.25; > 0.50; and > 0.75. Planning units with higher likelihood of seagrass presence were limited to regions of low-medium relative wave exposure such as sheltered bays that are north-facing (Figures 3.4 and 3.5). The exception to this pattern was the extensive north-facing bay of Broad Sound along the urban coast (Figure 3.5). Seagrasses do not grow in Broad Sound presumably because

of its high tidal range (~ 9 m). The Bayesian belief network predicted that the coastal waters of the Northern GBR had the greatest number of planning units with a high (i.e. > 50%) likelihood of seagrass presence (wet 42.9% of units; dry 56.7%) followed by the Dry Tropics (wet 37.1%; dry 40.8%), Southern GBR (wet 12.6%; dry 10.6%), and Wet Tropics (wet 7.4%; dry 13.3%) sections.

Table 3.2: The total number of point locality data (Fisheries Queensland) where seagrass was present and absent within various levels of predicted likelihood of seagrass presence from the model. The predictive rate is the proportion of point locality data where seagrass was present within the various levels of predicted likelihoods.

<i>Predicted likelihood of seagrass presence from the model</i>		<i>0 – 10</i>	<i>10 – 20</i>	<i>20 – 30</i>	<i>30 – 40</i>	<i>40 – 50</i>	<i>50 – 60</i>	<i>60 – 70</i>	<i>70 – 80</i>	<i>80 – 90</i>	<i>90 – 100</i>
Seagrass point locality data	Absent	112	72	230	247	867	240	28	4	1	0
	Present	20	12	37	186	729	359	59	13	6	6
<i>Predictive rate (%)</i>		15	14	14	43	46	60	68	77	86	100

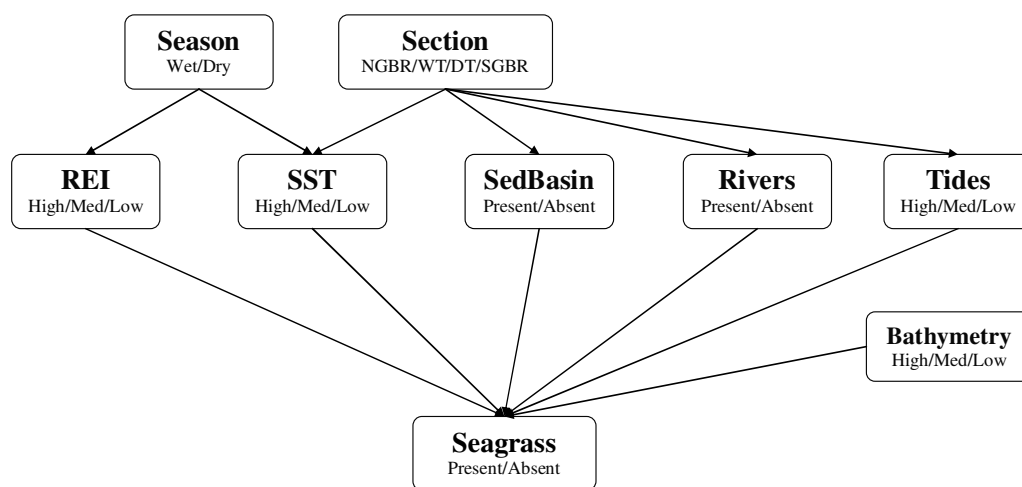


Figure 3.2: Bayesian belief network for coastal seagrass habitats (Seagrass: present and absent) in the GBRWHA. Predictor variable nodes and their discrete states included: season (Season: wet and dry); section (Section: Northern GBR, Wet Tropics, Dry Tropics, Southern GBR); relative wave exposure (REI: high, medium and low); sea surface temperature (SST: high, medium and low); substrate (SedBasin: present and absent); spatial extent of flood plumes (Rivers: present and absent); tidal range (Tides; high, medium and low); and bathymetry (Bathymetry: high, medium and low). See Table 3.1 for more information on the predictor variable nodes and their discrete states.

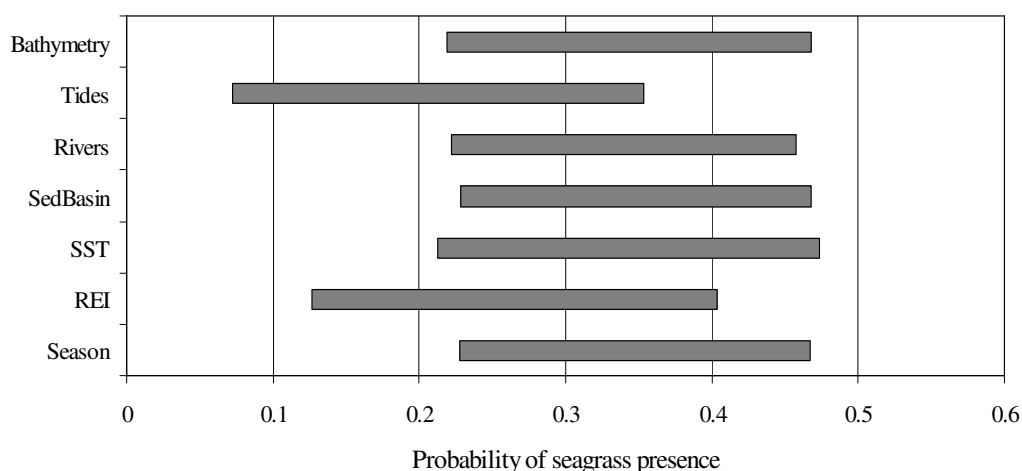


Figure 3.3: Sensitivity of coastal seagrass habitat presence to changes in individual nodes of the Bayesian belief network (Figure 3.2). Predictor variable nodes included: season (Season); section (Section); relative wave exposure (REI); sea surface temperature (SST); substrate (SedBasin); spatial extent of flood plumes (Rivers); tidal range (Tides); and bathymetry (Bathymetry). The bars represent the range of variation in the seagrass nodal variable when values for the states in each predictor variable node were varied over their possible ranges, and all other nodes were held constant at their most likely value. *See* Table 3.1 for more information on the predictor variable nodes and their discrete states.

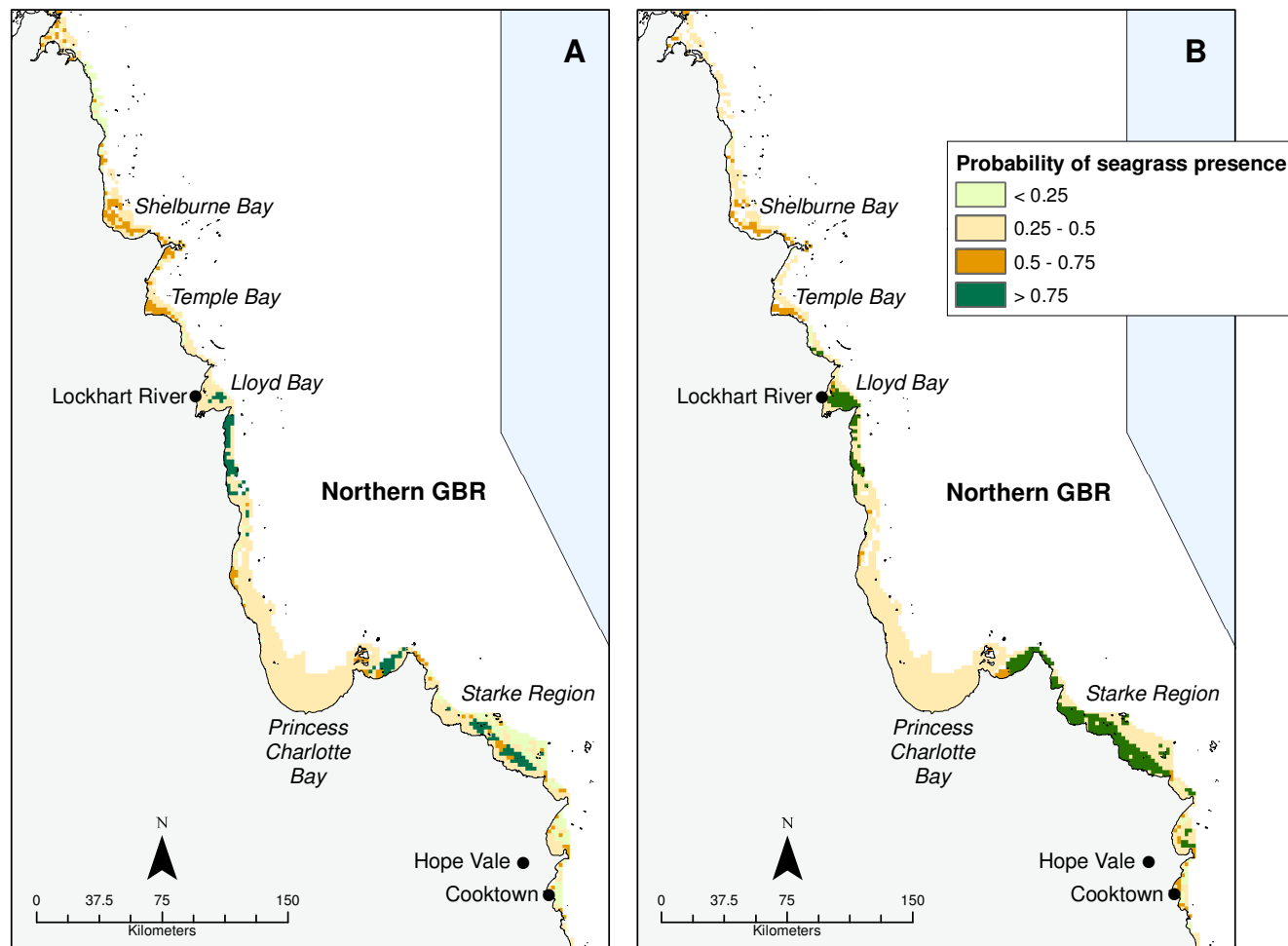


Figure 3.4: Habitat suitability maps of coastal seagrass distribution in the wet (A) and dry (B) seasons in the remote Cape York region.

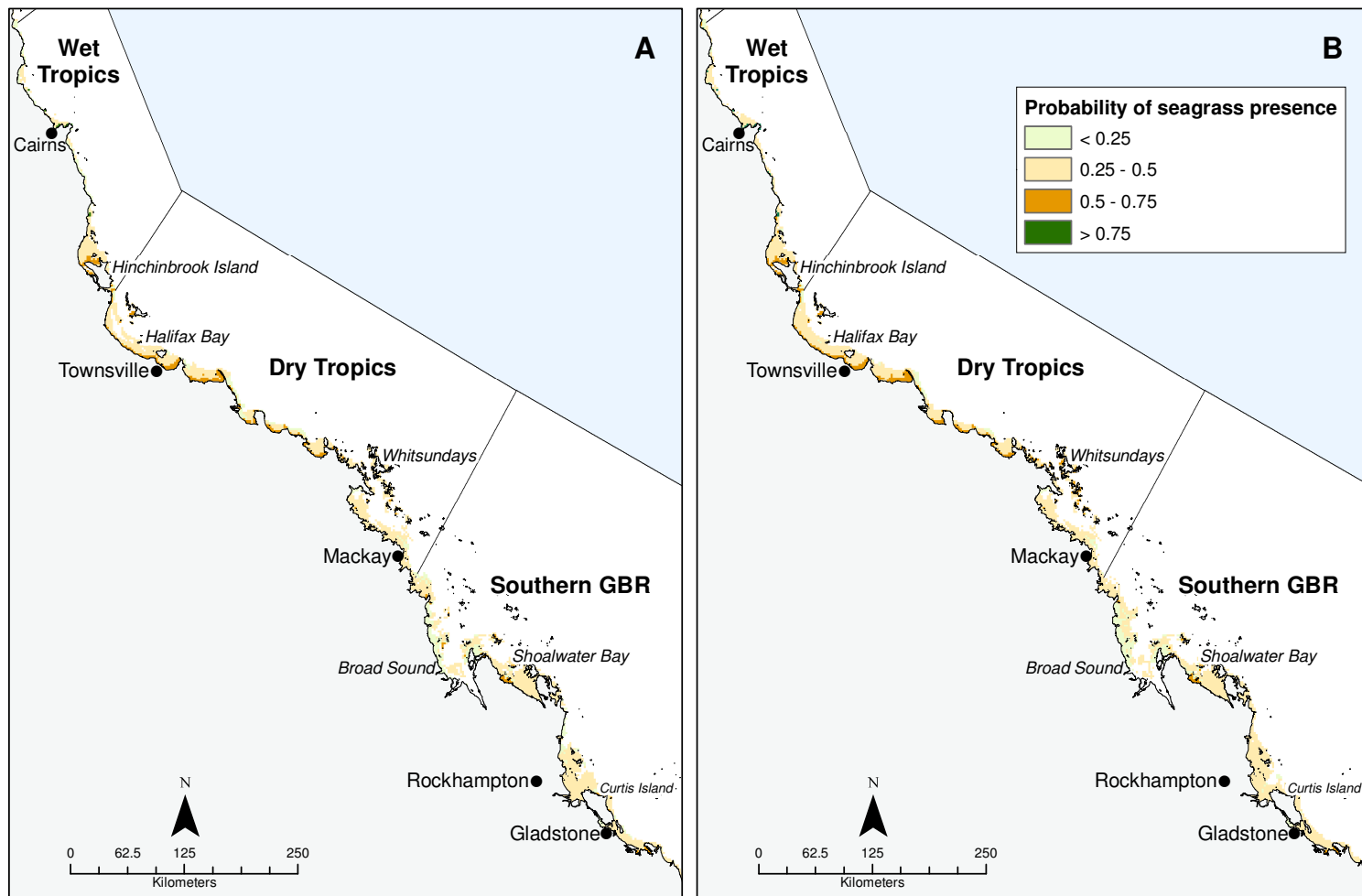


Figure 3.5: Habitat suitability maps of coastal seagrass distribution in the wet (A) and dry (B) seasons along the urban coast.

Discussion

There are few studies where the distribution of seagrass habitats over ecosystem-scales is modeled mathematically. Coles et al. (2009) have modeled seagrasses in GBRWHA waters deeper than 15 m bathymetry in the GBRWHA using boosted regression trees. Most seagrass distribution maps at ecosystem-scales are compiled from point location data or generated by interpretation of remote sensing data. Conceptual modeling (e.g. Carruthers et al. 2002) has been a common approach to understanding seagrass systems, but only helps to explain the role various drivers have in different locations. I used a Bayesian belief network to investigate the dependencies among coastal seagrass responses (presence/absence) and environmental drivers in the GBRWHA. This approach is appropriate because the area of interest was large, available data were a mixture of hard and soft 'evidence', and beliefs or conditional probabilities could be learnt from point locality data sets. The model allows predictive ability in areas where seagrass point locality data is readily available to be spread to areas where data is limited. The outputs of the modeling exercise provided probabilistic GIS-surfaces of habitat suitability for the entire ~ 2,300 km GBRWHA coast in both the wet and dry seasons.

The model predicts a significant ($p < 0.01$) difference between the mean likelihood of seagrass presence in the dry season (0.39; standard deviation 0.22) than the wet (monsoon) season (0.35; standard deviation 0.20). The dry season habitat suitability map (Figures 4 and 5) also had more planning units with likelihood of seagrass presence > 0.50 (796 km²) than the wet model (580 km²). This supports previous studies which found coastal seagrass abundance peaks in the dry season (McKenzie et al. 1998; Rasheed 2000; Coles et al. 2002). The Bayesian belief network predicted a substantial difference in seagrass distribution between the wet and dry seasons, and this should be incorporated in marine planning decisions.

I found that tidal range and relative wave exposure were the major limiting factors to seagrass presence and distribution. At the scale of the coastal GBRWHA, this result is explained by maximum tidal ranges in coastal waters that can range from 2.5 to 9 metres (Hopley et al. 2007); and the long, thin shape of the reef lagoon and direction of the prevailing winds that exacerbate the effect of wave exposure along the coastline.

Coles et al. (2009) also note the influence of the large tidal range on the distribution of seagrasses between 15 m and -60 m in the GBRWHA. Contemporary studies on the drivers of coastal seagrass distribution are mostly conducted at scales finer than ecosystems (e.g. Rasheed 2004); and Carruthers et al. (2002) considers coastal seagrass species in the GBRWHA to be limited by physical disturbance from storm and cyclone related waves and swell, associated sediment movement, and macro-grazers (e.g. marine turtles and dugongs). The Bayesian belief model I developed in Chapter 4 suggests that the importance of tides and relative wave exposure in driving seagrass presence and distribution has been underestimated by previous studies in the GBRWHA that are conducted at fine spatial scales.

There are some locations in the GBRWHA where the model may over-predict the distribution of seagrass e.g. the shallow waters of Halifax Bay (Figure 3.5) where the model predicted a likelihood of seagrass presence > 50%. Surveys conducted by the Northern Fisheries Centre in 1986 and 1987 found very little seagrass in this region. The survey team expected to find seagrass, and suggest that the absence of meadows is due to temporal variability resulting from changing environmental and/or anthropogenic conditions. Seagrass surveys of nearby Cleveland Bay (Figure 3.5) in 1987 and 1996 also found only a negligible amount of seagrass in the centre of the bay where the model predicts a likelihood of seagrass presence > 35%. However, a recent survey of Cleveland Bay found there are now extensive seagrass meadows, supporting the model's prediction. Actual seagrass distribution in Halifax Bay could similarly change and match the model predictions at some times even though seagrass was not found in this region during the 1986 and 1987 surveys.

Marine plants are protected in Queensland by the Fisheries Queensland under the Queensland Fisheries Act 1994. Fisheries Queensland has a no net loss policy target for marine plants defined as maintaining the distribution of seagrass to at least 90% of the 1990 distribution. It also advocates a policy principal of avoiding, minimizing or offsetting loss of seagrass or other marine plant communities in determining coastal management and development decisions (Couchman and Beumer 2007). Seagrass distribution and meadow location is key data in these processes and the model I developed in this study usefully provides the maximum likely extent of coastal seagrass distribution, and can be sliced for any given probability required.

However, the Bayesian belief network and habitat suitability maps predict the realised niche of coastal seagrasses and not the actual seagrass distribution (Guisan and Zimmermann, 2000). The model is therefore unable to account for or predict species distribution under changing environmental conditions (e.g. rising sea levels) in the same way as the point locality data it was based on cannot. The model will need to be updated in the future to account for alterations in environmental drivers that may result from changes in climate and biological and geological changes to remain useful through time. In the short-term, the habitat suitability maps can be used to inform marine planning and management in the GBRWHA as modeling the realised niche of a species from empirical field data sets is a valid and powerful approach when the model is required to have a high predictive precision at a broad spatial scale (Guisan and Thuiller, 2005). Modeling the realised niche of a species using Bayesian belief networks would also be appropriate at different spatial scales and for other species where the appropriate point locality and environmental data exist.

The approach I described in Chapter 4 is applicable to other areas where survey data is collected across ecosystem-scales. At that scale, investment in survey data involves strongly diminishing returns as the collection and interpretation of point locality data sets is time-consuming and expensive (Grantham et al., 2008). My study suggests an approach of targeting collection of point locality data sets to improve the performance of predictive habitat models and habitat suitability maps over ecosystem-scales would be a cost effective approach.

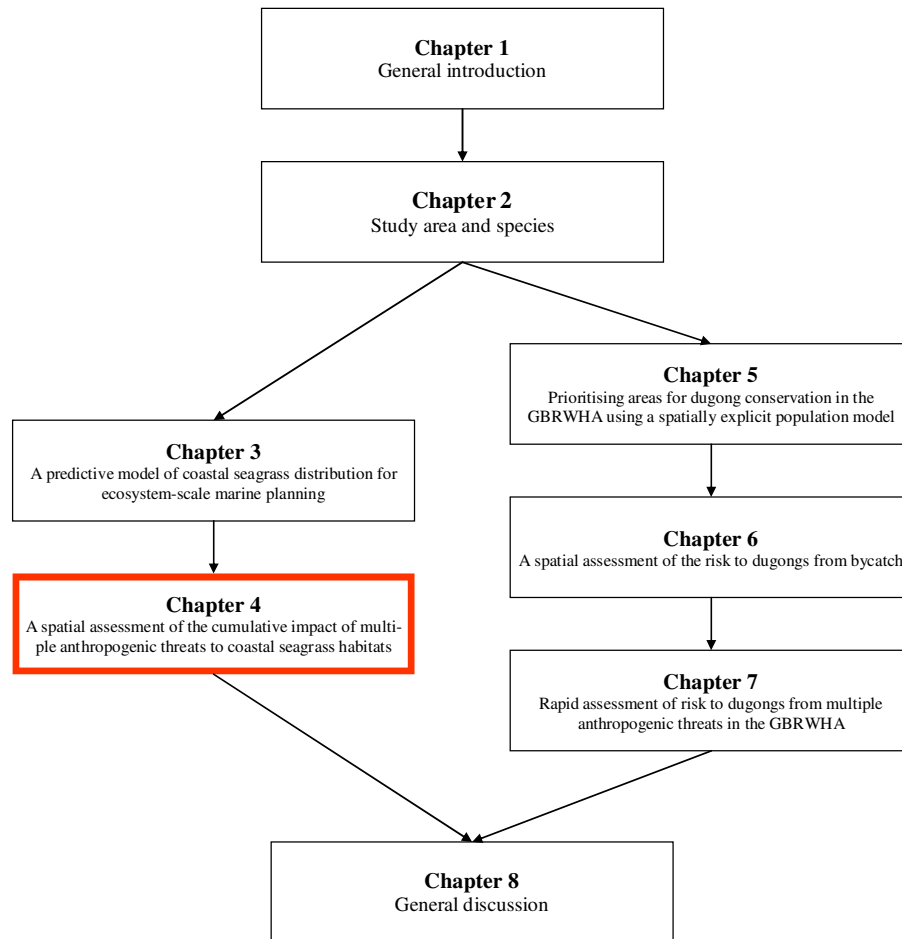
Chapter Summary

- Ecosystem-scale networks of MPAs are important management tools. Nonetheless, it is difficult to acquire information to inform their design and management because of the high cost associated with collecting data at that scale.
- To inform the planning and management of coastal seagrass habitats (approximately -15 m below mean sea level) at the scale of the GBRWHA (~22,600 km²), I determined the presence and distribution of seagrasses by generating a GIS-based habitat suitability model.
- A Bayesian belief network quantified the relationship (dependencies) between seagrass and eight environmental drivers: relative wave exposure, bathymetry, spatial extent of flood plumes, season, substrate, region, tidal range and sea surface temperature. The analysis showed that at the scale of the entire coastal GBRWHA the main drivers of seagrass presence were tidal range and relative wave exposure. Outputs of the model included probabilistic GIS-surfaces of seagrass habitat suitability in two seasons and at a planning unit of cell size 2 km * 2 km.
- The habitat suitability maps addressed the problems associated with delineating habitats at the scale appropriate for the design and management of ecosystem-scale networks of MPAs as they extended along the entire GBRWHA coast.

Chapter 4

A spatial assessment of the risk to coastal seagrass habitats to multiple anthropogenic threats in the GBRWHA¹

In this chapter, I use expert opinion and a risk assessment framework to delineate areas of low, medium and high relative impact to coastal seagrass habitats from multiple anthropogenic threats. I compare the distribution of threats with the probabilistic model of coastal seagrass distribution to estimate the risk of coastal seagrass habitats from the threats at the scale of the coastal GBRWHA. Outputs of the assessment are the identification of: (1) anthropogenic threats with the greatest relative impact on coastal seagrass habitats; and, (2) ‘hot spots’ that are a priority for conservation actions.



¹ **Grech, A.** and Coles, R. *in prep.* A spatial assessment of the risk to coastal seagrass habitats from multiple anthropogenic threats in the GBRWHA. Target journal *Aquatic Conservation*.

Introduction

As discussed in Chapter 1, coastal marine ecosystems are characterised by high productivity and biodiversity values (Gray 1997), and provide a variety of goods and services (Costanza et al. 1997; Worm et al. 2006). Coastal ecosystems are threatened by multiple anthropogenic activities (Jackson et al. 2001) and are some of the most at-risk marine ecosystems in the world (Halpern et al. 2008a). Within coastal ecosystems, seagrass communities are considered to be one of the most highly threatened habitats along with coral reefs, mangroves and salt marshes (Waycott et al. 2009).

In most parts of the world, some seagrass habitats have been lost (Short and Wyllie-Echeverria 1996; Larkum et al. 2006), sometimes from natural causes (den Hartog 1987) including storms (Poiner et al. 1989). However, destruction of seagrass habitats has commonly resulted from anthropogenic activities (Cambridge and McComb 1984; Coles et al. 2003; Waycott et al. 2009). Dredging, coastal development, damage associated with over exploitation of coastal resources, recreational boating activities and nutrient and sediment loading from adjacent land catchments have dramatically reduced seagrass distribution in many regions (Short and Wyllie-Echeverria 1996; Orth et al. 2006; Waycott et al. 2009).

Coles et al. (2007) found the overall area of coastal seagrass meadows in the GBRWHA to be relatively stable over the past 20 years. Despite this result, some tropical seagrasses tend to be ephemeral and respond both spatially and temporally to environmental change (*see* Chapter 3). This characteristic makes it difficult for monitoring programmes to assess the impact on seagrass habitats from anthropogenic threats at a GBRWHA scale as the new growth of seagrass in one area may hide the fact that seagrass has been irretrievably lost from another area. Furthermore, the ephemeral nature of some tropical seagrasses makes it difficult to determine if changes in seagrass distribution and abundance are a result of natural causes or in response to anthropogenic impacts.

Increases in coastal human population density and resource developments in mining and agriculture in the GBRWHA catchment (Figure 2.1) have lead to negative pressures on coastal seagrass habitats (Coles et al. 2007). These pressures can influence seagrass

habitats at fine spatial scales of hundreds of metres (e.g. marina developments), or at the broad spatial scale of 10,000s kilometres (e.g. agricultural chemicals from terrestrial runoff). Such threats can have an absolute impact by removing seagrass entirely from an area, or a more subtle impact by slowing growth or limiting plant reproduction.

Anthropogenic threats can be one-off events, intermittent, or occur repeatedly. Seagrass habitats in the GBRWHA are simultaneously subjected to multiple threats; it is not known whether the cumulative effect of these threats is additive, synergistic or antagonistic. Furthermore, the interrelationships between anthropogenic threats and natural changes in seagrass distribution are poorly understood (Duarte 2002). Because of this complexity, the impact of multiple anthropogenic threats on seagrass habitats in the GBRWHA is difficult to measure and predict; a similar problem exists in other seagrass ecosystems.

Managing anthropogenic threats along the entire GBRWHA coast (~ 2,300 km) is constrained by the costs associated with implementing conservation actions at that scale. Conservation actions that target all anthropogenic threats in all areas are not politically feasible and limited funding and resources restricts the number of sites that can be managed (Halpern et al. 2007). An alternative approach is to prioritise conservation actions by ranking the relative impact of anthropogenic threats and identifying the spatial location of vulnerable habitats (Crain et al. 2009). This approach allows for the identification of sites that if targeted would have the greatest conservation benefit to the ecosystem at the scale of the assessment. These areas are ‘hot spots’ for conservation actions because they have high biodiversity value, are threatened by multiple anthropogenic threats, or both.

The identification of ‘hot spots’ for conservation action requires spatial information on the distribution of species and the intensity and distribution of anthropogenic threats. Sanderson et al. (2002), Ban and Alder (2008), Halpern et al. (2008), Halpern et al. (2009) and Selkoe et al. (2009) combine ecosystem vulnerability assessments with spatial information on the distribution of anthropogenic threats to assess the cumulative impact of multiple threats across multiple ecosystems (ecozones). Halpern et al. (2007) and Selkoe et al. (2008) describe a systematic method for collecting expert opinion to inform a qualitative assessment of the relative impact of multiple anthropogenic threats on multiple ecosystems. The outputs of the assessments were subsequently combined

with spatial information on the intensity and distribution of anthropogenic threats to identify ‘hot spots’ for conservation action (Halpern et al. 2008a; Halpern et al. 2009; and Selkoe et al. 2009).

In this chapter, I used an approach analogous to Halpern et al. (2008a), Ban and Alder (2008), Halpern et al. (2009) and Selkoe et al. (2009) to assess the present risk of coastal seagrass habitats in the GBRWHA to multiple anthropogenic threats. Risk is determined by measuring two components: (1) the likelihood or probability of the hazard event occurring; and, (2) the consequences or the effects of an adverse event (Norton et al. 1996). I quantified the likelihood of a hazard event occurring by modelling the spatial distribution and intensity of multiple anthropogenic threats. The consequence of an adverse event is more difficult to quantify as the concept of consequence involves features of ecosystem services and resilience; variables that are species specific and location dependent. In this assessment, I quantified consequence using the probabilistic model of coastal seagrass distribution outlined in Chapter 3. I assumed that the greater the likelihood of seagrass presence, the greater the consequence if it is damaged or lost. The outputs of the risk assessment included the identification of seagrass ‘hot spots’ that are a priority for conservation actions at the scale of the entire GBRWHA coast.

Methods

I followed Suter (1993) and estimated the risk to coastal seagrass habitats from anthropogenic threats in the GBRWHA by: (1) identifying the hazards; (2) quantifying the exposure of coastal seagrass habitats to the hazards; and, (3) estimating the risk to coastal seagrass habitats.

Hazard identification

I identified from expert opinion and the literature, especially Coles et al. (2007), the anthropogenic hazards to coastal seagrass habitats in the GBRWHA as: (1) poor quality terrestrial runoff from agricultural, urban and industrial activities in adjacent land catchments (Orth et al. 2006); (2) commercial and recreational boats that can damage seagrass meadows (Orth et al. 2006); (3) habitat loss associated with urban and port infrastructure development and harbour dredging (McKenzie et al. 2000); (4) prawn

trawling which can damage bottom habitats; (5) commercial fishing other than trawling (e.g. netting); and, (6) commercial shipping accidents (e.g. oil spills). I did not include mining and oil drilling as these activities are banned in the Great Barrier Reef Marine Park (*see* Chapter 2 page 22).

Waycott et al. (2007) conducted an assessment of the vulnerability of seagrasses in the GBRWHA to anthropogenic climate change, and made predictions on how climate change related threats will impact seagrass under multiple climate change scenarios. Short and Neckles (1999) predicted the effects of anthropogenic climate change on seagrasses at a global scale. Both studies based their predictions on a comprehensive review of published research on individual seagrass species and habitat tolerances. They found no evidence that seagrasses are currently threatened by anthropogenic climate change in the GBRWHA. I did not identify anthropogenic climate change as a hazard as I assessed the present level of risk to coastal seagrass habitats rather than future risks.

Exposure quantification

I quantified the spatial distribution of the nine hazards to coastal seagrass habitats in the GBRWHA at a planning unit of cell size 2 km * 2 km (the same scale as the predictive seagrass model outlined in Chapter 3): agricultural runoff, recreational-vessel traffic, commercial-vessel traffic, urban/port infrastructure development, dredging, trawling, netting, shipping accidents, and urban/industrial runoff. Figures showing the spatial distribution of the nine hazards are provided in Appendix C.

Water pollution in the GBRWHA is predominantly caused by runoff from adjacent land catchments (*see* Chapter 2 page 20; Figure 2.1). Beef grazing and sugar cane production dominate agricultural land-use; industrial and urban developments are relatively localised (Brodie et al. 2001). Agricultural, industrial and urban areas pollute the terrestrial waters that run into the GBRWHA with sediment, nutrients (including nitrogen and phosphorus) and herbicides. I obtained information on the distribution of poor quality terrestrial runoff from agricultural activities in the GBRWHA from the *Reef Exposure Model* developed by the Australian Centre for Tropical Freshwater Research (Maughan et al. 2008). Maughan et al. (2008) use a series of environmental and spatial indicators and an inverse distance weighted interpolation to delineate sites

within the GBRWHA that are of low, medium, medium – high or high risk from poor quality terrestrial runoff from agricultural activities. I modelled the risk to seagrass planning units from runoff in urban and industrial areas from information on human populations within catchments and urban centres (Australian Bureau of Statistics 2009), and the distribution of urban and industrial land-use within catchments (Queensland Department of Environment and Resource Management 2002). I assumed that: (1) urban and industrial land-use as delineated by the Queensland Department of Environment and Resource Management in 2002 determined the spatial extent of influence of runoff; and, (2) human population size and land-use type determined the magnitude of impact of urban and industrial runoff. I classified seagrass planning units as no impact, low, medium, and high impact from urban and industrial runoff.

I derived a model of relative inshore (< 4 m long) and reef (> 4 m) recreational-vessel traffic from the number of recreational boats registered within regional centres bordering the GBRWHA (Queensland Transport 2004). Queensland Transport tracks the number and geographic location of registered vessels by the post code of owner's residences. Seventy-five percent of registered vessels are inshore boats that do not operate offshore (> 15 km from the coast) (S. Sutton, personal communication). Twenty-five percent are recreational reef vessels that can operate in all waters of the GBRWHA. Using information on the geographic location of each owner's residential mail code, I assigned registered vessels to their closest port or marina. I assumed that: (1) a vessel was used only in the region it was registered; and, (2) all vessels (reef and inshore) operated within 15 km from the coast (not including islands) and 25% of registered vessels also operated in waters outside that region. To each seagrass planning unit, I assigned the number of registered vessels of its closest regional center (of 19 available, mean Euclidean distance between centres = 90 km). On advice from an expert in the recreational use of the marine environment, I classified vessel numbers per planning unit as: low impact when the number of registered vessels < 300; medium impact, 301 to 1000; medium-high impact, 1001 - 3000; and high impact, > 3001 (S. Sutton, personal communication).

Commercial vessels are generally larger than recreational vessels, function for transport, fishing or tourism, and operate mostly within designated shipping lanes or areas open to commercial fishing and/or tourism. I could not model commercial-vessel traffic using

the same method as recreational-vessel traffic because many commercial vessels do not operate in the waters surrounding their port of registration. Instead, I delineated the impact to coastal seagrass habitats from commercial-vessel traffic by using information on shipping lanes and the number of commercial vessels that access individual ports. In consultation with staff from the State and Federal Governments who have expertise in commercial shipping activities and seagrasses, I classified seagrass planning units as: no impact when the number of commercial vessels that moved through the unit per annum = 0; medium impact, 1 – 40; medium-high impact, 41 – 300; and high impact, 301 – 950.

Coastal engineering projects that create urban and port infrastructure may physically remove seagrass habitats. I assumed that the impact on coastal seagrass habitats from infrastructure development is a product of three factors: (1) the population of urban centres; (2) the spatial extent of ports and their relative importance (determined by the number of vessels that used each port per annum); and, (3) whether the port was presently undergoing expansion or had a expansion proposed within the next two years. I assumed that the extent of influence of urban/port infrastructure developments was no more than two seagrass planning units from the GBRWHA coast (4 km). I acquired human population estimates from the Australian 2006 census (Australian Bureau of Statistics) to delineate areas of low – high impact from urban development. I assumed that the present population of urban centres are an indicator of the likelihood of coastal development, and the larger the population the higher the impact of development. I classified seagrass planning units within a 4 km radius from the coast as: no impact when the human population = 0; low impact, 1 – 750; medium impact, 751 – 3000; and high impact, > 3001. I assumed that planning units > 4 km from the coast were unlikely to be affected by urban development. I obtained information on the spatial extent of ports, the number of vessels that used each port per annum and present and/or proposed port infrastructure developments from the relevant port authorities. I classified seagrass planning units within the limits of ports as: (1) low impact if there were no present or proposed infrastructure developments and < 20 ships used the port per annum; (2) medium impact if there were present or proposed developments that were small in magnitude and 21 – 60 ships used the port per annum; and, (3) high impact if there were present or proposed developments that were large in magnitude (i.e. developments that result in substantial land reclamation) and > 61 ships used the port per annum. Planning

units that fell within Fish Habitat Areas were classified as no impact as the development of infrastructure is restricted at those sites (*see* Chapter 2 page 29).

Dredging (the collection and disposal of material to deepen or maintain waterways and to create harbors, channels, docks and berths) occurs in six of the ten ports in the GBRWHA. At the sites of dredging and disposal, seagrass habitats are removed and/or disturbed by suspended sediment and turbidity in the water and by increased sedimentation on the seabed. I obtained information on the location, frequency and spatial extent of influence of both construction and maintenance dredging activities from the relevant port authorities. I classified planning units within and around ports as: (1) no impact from dredging activities where no dredging occurred; (2) low impact when maintenance dredging was conducted only once per year; and, (3) high impact when maintenance dredging occurred more than once per year and/or when there were construction activities that included dredging.

Fisheries Queensland models trawl effort data in grid cells of approximately 1 km². Each grid cell represents the number of hours trawled each year and the amount of catch. I used this information to identify seagrass planning units where trawling activities have occurred (present or absent) in the recent past (2002 – 2005). Fisheries Queensland also monitors the catch of the commercial gill-netting industry through compulsory daily logbooks completed by fishers. This information is then aggregated by Fisheries Queensland into grids of a 6 nm resolution. I used this information to identify seagrass planning units where netting activities have occurred (present or absent) in the recent past (2004 – 2006).

Queensland Transport and the Great Barrier Reef Marine Park Authority conducted an assessment of oil spill risk for the coastal waters of the GBRWHA in 2000. They identified coastal areas that have a low, medium and high risk from serious marine oil spills from shipping. I used their assessment to delineate seagrass planning units of low medium and high risk of shipping accidents that result in oil spills. Oil spills from drilling and/or mining activities were not included in the assessment as these activities are banned in the GBRWHA (*see* Chapter 2 page 22).

I used the habitat suitability maps for coastal seagrasses outlined in Chapter 3 to quantify the exposure of seagrass habitats to their anthropogenic hazards. I assumed that consequence is a function of the likelihood of seagrass presence i.e. the greater the likelihood of seagrass presence, the greater the consequence if it is damaged or lost. Seagrass planning units with a likelihood of seagrass presence < 0.5 were classified as low conservation value, $0.5 - 0.75$ medium, and > 0.75 high conservation value (*see* Chapter 3; Figures 3.4 and 3.5).

Risk Estimation

Quantitative information and empirical data on the relative impact of each of the anthropogenic hazards on coastal seagrass habitats in the GBRWHA is incomplete or unavailable. In the light of this uncertainty, I evaluated the relative impact of hazards on coastal seagrass habitats using the method of Halpern et al. (2007) and Selkoe et al. (2008). Halpern et al. (2007) develop a systematic approach for collecting expert opinion on the relative impact of multiple anthropogenic hazards to marine ecosystems. The vulnerability of ecosystems to individual hazards is characterised by five attributes (termed ‘vulnerability factors’): (1) the average scale at which a hazard affects the ecosystem; (2) the frequency of a hazard event occurrence; (3) the hazard’s functional impact (i.e. number of species within a community or ecosystem that are impacted by a hazard event); (4) the resistance of the ecosystem to disturbance by a hazard; and, (5) the resilience (ie. recovery time) of the ecosystem following a disturbance. Halpern et al. (2007) and Selkoe et al. (2008) devise a ranking system to quantify the score of each vulnerability factor, as summarised in Table 4.1.

I created an online survey (Appendix D) to collect information on rankings and scores for the five vulnerability factors from seagrass experts. Thirty-two experts from academic institutions and government agencies were invited to participate in the online survey. The experts were selected because they had expertise in seagrass ecology and biology, marine and terrestrial management, water quality, and/or spatial information. Survey participants were provided with information on the aims and objectives of the study and a description of the five vulnerability factors and how to score them. In addition to providing individual scores for each of the five vulnerability factors, survey participants were asked to quantify the certainty of their estimates for each hazard

(Table 4.1). This project received approval from the James Cook University Human Research Ethics Review Committee (approval number H3510).

Table 4.1: Ranking system for the five vulnerability factors (Halpern et al. 2007; Selkoe et al. 2008).

<i>Score</i>	<i>Scale</i>	<i>Frequency</i>	<i>Functional impact</i>	<i>Resistance</i>	<i>Recovery time</i>	<i>Certainty</i>
0	No impact	Never occurs	No impact	Not applicable	No impact	Not at all certain
1	< 1 km ²	Rare	Species level	High resistance	< 1 year	Low certainty
2	1 – 10 km ²	Occasional	Single trophic level	Medium resistance	1 – 10 years	Moderate certainty
3	10 – 100 km ²	Annual or regular	Multiple trophic levels	Low resistance	10 – 100 years	High certainty
4	100 – 1,000 km ²	Persistent	Entire community		> 100 years	Very certain
5	1,000 – 10,000 km ²					
6	> 10,000 km ²					

I used a hierarchical cluster analysis in SPSS 17.0 (Polar Engineering and Consulting 2008) to detect outliers in the survey participants. I measured the standard error and coefficient of variation in scores across responses for each vulnerability factor of individual hazards. The value of the coefficient of variation indicated the dispersion (variation) in the scores provided by experts and was used to assess the degree of consensus among experts. This approach was chosen as the coefficient of variation allows for the comparison of the degree of variation for each vulnerability factor of individual hazards, even though their means are different from each other.

I rescaled from 0 – 4 the rank values for the vulnerability factors of scale and resistance (Table 4.1) so that all factors had the same range of values. I then calculated the relative vulnerability of coastal seagrass habitats to the nine anthropogenic hazards using the method of Selkoe et al. (2008). I assumed that each vulnerability factor had equal weighting, and combined the mean scores across responses for each of the five vulnerability factors of individual hazards into a single weighted-average vulnerability score. For example, the mean scores across responses for each vulnerability factor

(Table 4.1) for dredging were: 1.9 (scale); 2.8 (frequency); 3.5 (functional impact); 3.4 (resistance); and 2.0 (recovery time). The weighted-average vulnerability score for dredging is the mean of those five scores (2.7).

The hazards do not have a homogenous distribution (see *Exposure quantification* and Appendix C). By way of illustration, some ports are dredged more frequently than other ports, and some ports are not dredged at all. I used the weighted-average vulnerability scores and information on the intensity (e.g. low, medium, or high) and distribution (e.g. present or absent) of hazards to derive a score for each impact level within a hazard. For example, the dredging impact level scores relative to other hazards was: 0 when there was no impact from dredging; 1.35 (half of the weighted-average score of 2.7) when there was a medium impact; and 2.7 when there was a high impact.

I imported the weighted-average vulnerability scores for the impact level of each hazard into the descriptive GIS layers (Appendix C). No empirical information existed on the cumulative and interactive effect of multiple hazards on coastal seagrass habitats, so I assumed that the cumulative impact of hazards was additive. I intersected all of the GIS layers and calculated a cumulative score for each planning unit (cumulative scores range from 0 to 100). Planning units with a high cumulative score posed the greatest relative threat to coastal seagrass habitats in the GBRWHA. I identified clusters of planning units with low, medium and high cumulative scores using the Getis-Ord G_i^* statistic in ArcGIS 9.3. The G_i^* statistic identifies clusters of high values by comparing a cell's value with the value of its neighboring cells. For instance, if a planning unit's cumulative score is high and the values of all of its neighboring planning units are high, the G_i^* statistic classifies the region as a hot spot. The G_i^* statistic is a Z score; and indicates the statistical significance of observed spatial patterns. I used the value of the G_i^* statistic (Z score) to classify seagrass planning units as low, medium or high relative impact to coastal seagrass habitats. I evaluated, spatially, the relative risk of coastal seagrass habitats to their anthropogenic hazards by comparing the cumulative hazard coverage with the habitat suitability maps for coastal seagrasses outlined in Chapter 3.

Results

Survey results

I received responses from 14 (44%) of the 32 experts to whom I forwarded the online survey. Four responses were from the staff of academic institutions, two were from government research agencies and eight were from government management agencies. I did not test for respondent bias in the vulnerability scores due to gender or institutional affiliation because the sample size was low ($n = 14$). The hierarchical cluster analysis revealed: (1) two dominant clusters of almost equal size (Figure 4.1); and, (2) no outliers in the survey participants. Neither of the two clusters were dominated by survey participants from any particular gender or institution.

Table 4.2 shows the range, mean, standard error, standard deviation and coefficient of variation in scores across responses for each of the vulnerability factors of individual hazards. The coefficient of variation across responses for all vulnerability factors was low ($< 11.2\%$) for the hazards of agricultural runoff, urban/industrial runoff, and urban/port infrastructure development; indicating a strong consensus among experts on the vulnerability of seagrasses to these hazards. The coefficient of variation across responses of fishing (other than trawling) for four of its five vulnerability factors was $> 12.0\%$, which may be an indication of uncertainty among experts on the impact of fishing activities on coastal seagrass habitats.

Using the method of Selkoe et al. (2008), I combined all of the expert's individual scores for the five vulnerability factors into a single, weighted-average vulnerability score (Table 4.3). The hazard with the largest weighted-average vulnerability score was agricultural runoff (mean score 3.2 out of 4, coefficient of variation 8.5%), followed in decreasing order of magnitude by: urban and industrial runoff (2.9, 12.1%); urban and port infrastructure development (2.8, 14.2%); dredging (2.7, 12.6%); shipping accidents (2.5, 17.2%); trawling (2.6, 14.6%); recreational boat damage (1.9, 20.2%); commercial boat damage (1.9, 16.6%); and fishing (1.8, 18.4%). I used a two-way ANOVA to test for the potential effect of survey respondents on the weighted-average vulnerability scores, and found no significant effect of respondent ($F = 5.8$, $p = 0.315$). There was also no significant effect on the weighted-average vulnerability scores from hazards (F

= 39.1, $p = 0.123$), or from the interaction between respondents and hazards ($F = 1.721$, $p = 0.552$).

Survey participants were asked to provide scores ranging from zero (not at all certain) to four (very certain) to provide a qualitative assessment of the certainty of their estimates for the vulnerability of seagrasses to each hazard (Tables 4.1, 4.2 and 4.3). The hazard with the greatest mean certainty score was dredging (3.0); followed by urban and port infrastructure development (2.7); shipping accidents (2.6); trawling (2.4); agricultural runoff and urban and industrial runoff (2.3); recreational boat damage (2.2); commercial boat damage (2.1); and fishing (1.8). I used a two-way ANOVA to test for the potential effect of survey respondents on the mean certainty scores, and found no significant effect of respondents ($F = 1.108$, $p = 0.641$). I also found no significant effect in the interaction between respondents and hazards on the certainty score ($F = 0.232$, $p = 0.960$), and no significant effect of hazard on the certainty scores ($F = 0.887$, $p = 0.681$). All but one of the hazards received certainty scores > 2 , indicating moderate – high certainty among experts on the impact of the anthropogenic hazards on coastal seagrass habitats.

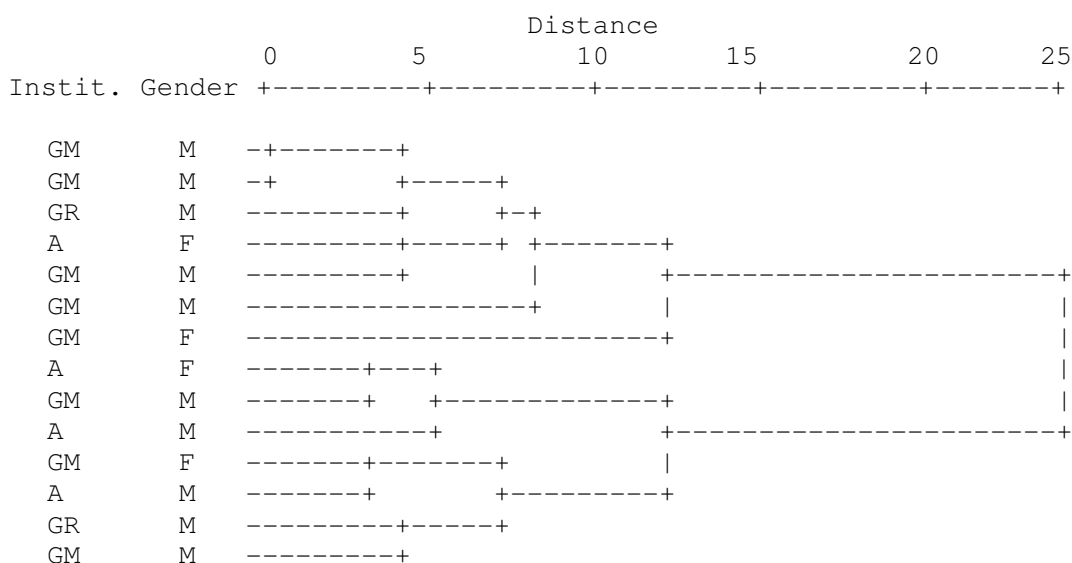


Figure 4.1: Dendrogram showing the two major clusters of survey respondents, their institutional affiliation and gender. GM = government management agency; GR = government research agency; A = academic institution; M = male; and F = female.

Table 4.2: Descriptive statistics of the scores across responses for each of the vulnerability factors of individual hazards.

Hazard	Vulnerability factor	Range	Mean	Standard error	Standard deviation	Coefficient of variation (%)
Agricultural runoff	Scale	2.7 - 4	3.1	0.1	0.4	3.5
	Frequency	2 - 4	3.1	0.1	0.5	4.5
	Functional Impact	3 - 4	3.9	0.1	0.4	2.5
	Resistance	1.4 - 4	3.0	0.2	0.8	7.1
	Recovery time	2 - 3	2.2	0.1	0.4	5.1
Boat damage (commercial)	Certainty	1 - 4	2.3	0.2	0.8	9.7
	Scale	0.7 - 2	1.1	0.1	0.4	10.3
	Frequency	1 - 4	2.5	0.3	1.2	12.4
	Functional Impact	1 - 4	2.3	0.4	1.3	15.5
	Resistance	1.4 - 4	2.4	0.3	1.1	12.0
Boat damage (recreational)	Recovery time	1 - 2	1.1	0.1	0.4	8.5
	Certainty	1 - 4	2.1	0.2	0.9	10.8
	Scale	0.7 - 3.4	1.0	0.2	0.7	20.4
	Frequency	1 - 4	2.9	0.3	1.1	9.8
	Functional Impact	1 - 4	2.1	0.4	1.4	16.8
Dredging	Resistance	1.4 - 4	2.4	0.3	1.1	12.0
	Recovery time	1 - 2	1.1	0.1	0.3	6.7
	Certainty	1 - 4	2.2	0.2	0.9	10.8
	Scale	1.4 - 2.7	1.9	0.1	0.5	6.7
	Frequency	2 - 4	2.7	0.2	0.6	6.0
Fishing (other than trawling)	Functional Impact	1 - 4	3.4	0.3	1.0	7.9
	Resistance	1.4 - 4	3.3	0.2	0.9	7.0
	Recovery time	1 - 3	1.9	0.1	0.5	6.6
	Certainty	1 - 4	3.0	0.3	1.0	9.2
	Scale	0.7 - 2	1.0	0.1	0.4	12.1
Shipping accidents (e.g. oil spills)	Frequency	1 - 4	2.6	0.3	1.2	12.7
	Functional Impact	1 - 4	2.4	0.3	1.1	12.0
	Resistance	1.4 - 4	1.8	0.2	0.8	12.5
	Recovery time	1 - 2	1.1	0.1	0.4	8.5
	Certainty	0 - 3	1.8	0.2	0.8	12.0
Trawling	Scale	1.4 - 3.4	2.1	0.2	0.6	7.4
	Frequency	1 - 4	1.4	0.2	0.9	15.9
	Functional Impact	1 - 4	3.6	0.2	0.9	6.4
	Resistance	2.7 - 4	3.4	0.2	0.7	5.3
	Recovery time	1 - 2	1.9	0.1	0.3	3.7
Urban/ industrial runoff	Certainty	1 - 4	2.6	0.2	0.8	8.5
	Scale	0 - 3.4	1.6	0.2	0.9	14.5
	Frequency	0 - 4	3.1	0.3	1.1	9.9
	Functional Impact	1 - 4	3.3	0.3	1.0	8.1
	Resistance	0 - 4	2.7	0.3	1.3	12.8
Urban/port infrastructure development	Recovery time	0 - 3	1.6	0.2	0.8	13.7
	Certainty	2 - 4	2.4	0.2	0.6	7.1
	Scale	1.4 - 3.4	2.1	0.1	0.4	5.6
	Frequency	2 - 4	3.3	0.2	0.9	7.4
	Functional Impact	2 - 4	3.9	0.1	0.5	3.7
	Resistance	2.7 - 4	3.0	0.2	0.6	5.5
	Recovery time	2 - 3	2.1	0.1	0.3	3.4
	Certainty	2 - 4	2.3	0.2	0.6	7.1
	Scale	0.7 - 2.7	1.7	0.2	0.6	8.9
	Frequency	1 - 4	2.4	0.3	1.0	11.2
	Functional Impact	1 - 4	3.8	0.2	0.8	5.7
	Resistance	2.7 - 4	3.5	0.2	0.7	5.0
	Recovery time	1 - 4	2.3	0.2	0.8	9.7
	Certainty	2 - 4	2.7	0.2	0.7	7.2

Table 4.3: Weighted-average vulnerability and certainty scores derived from expert opinion.

<i>Hazard</i>	<i>Mean Certainty (x/4)</i>	<i>Weighted-average vulnerability score (x/4)</i>	<i>Cumulative risk scores</i>		
			<i>Impact level</i>	<i>Score relative to other levels within hazard^a</i>	<i>Impact level score relative to other hazards^b</i>
Agricultural runoff	2.3	3.1	Low	0	0
			Medium	33	1.02
			Medium - high	67	2.08
			High	100	3.1
Urban/industrial runoff	2.3	2.9	No impact	0	0
			Low	33	0.96
			Medium	67	1.94
			High	100	2.9
Urban/port infrastructure development	2.7	2.8	No impact	0	0
			Low	33	0.92
			Medium	67	1.88
			High	100	2.8
Dredging	3.0	2.6	No impact	0	0
			Low	50	1.3
			High	100	2.6
Shipping accidents (eg. oil spills)	2.6	2.5	Low	0	0
			Medium	50	1.25
			High	100	2.5
Trawling	2.4	2.4	Absent	0	0
			Present	100	2.4
Boat damage (recreational)	2.2	1.9	Low	0	0
			Medium	33	0.63
			Medium - high	67	1.27
			High	100	1.9
Boat damage (commercial)	2.1	1.9	No impact	0	0
			Low	33	0.63
			Medium	67	1.27
			High	100	1.9
Fishing (other than trawling)	1.8	1.8	Absent	0	0
			Present	100	1.8

^aDerived by ranking the relative importance of the individual impact levels of anthropogenic hazards out of 100.

^cDerived by calculating the relative importance of impact levels of anthropogenic hazards on the basis of the weighted-average vulnerability scores.

Spatial analysis

The minimum cumulative score was 0 (i.e. no hazards present) and the maximum score was 78.7. Along the urban coast the range of cumulative scores was 0 – 78.7, and in the remote Cape York region the range of cumulative scores was 0 – 32.7. The Getis-Ord G_i^* statistic detected clusters of low (cumulative score < 21.3), medium (21.3 – 37.0) and high (> 37.0) relative impact to coastal seagrass habitats in the GBRWHA (Figures 4.2 and 4.3). Planning units of high relative impact to coastal seagrass habitats were within ports and adjacent to the populated urban centres of Cairns, Townsville, Mackay, Yeppoon and Gladstone (urban coast; Figure 4.3). Other regions not adjacent to urban centres with planning units of high relative impact to coastal seagrass habitats included Bowling Green Bay, Alva Beach, Edgecombe Bay, Whitsunday Passage, Repulse Bay and Keppel Bay (urban coast; Figure 4.3). Areas with low composite hazard scores included most of the coastal waters north of Cooktown (remote Cape York region; Figure 4.2), Halifax Bay and Shoalwater Bay (urban coast; Figure 4.3).

I compared the habitat suitability maps for coastal seagrasses (Figures 3.4 and 3.5) outlined in Chapter 3 with the cumulative hazard coverage (Figures 4.2 and 4.3), as summarised in Table 4.4. At the scale of the entire GBRWHA, approximately 88% and 89% of planning units of high conservation value to seagrasses during the dry and wet season are at low relative risk from anthropogenic hazards respectively (Table 4.4; Figures 4.4 and 4.5). As the conservation value (or likelihood of seagrass presence) decreases, so does the proportion of planning units at low risk from anthropogenic hazards (Table 4.4). There is a marked difference in the relative risk to planning units along the urban coast compared with the remote Cape York region. Almost all (> 95%) seagrass habitats in the remote Cape York region are at low risk from anthropogenic hazards. Along the urban coast, less than 34% of coastal seagrass habitats are at low risk (Table 4.4). I identified planning units of high and medium conservation value to coastal seagrass habitats that are at risk from multiple anthropogenic threats and are a priority for conservation actions ('hot spots') because the present level of impact is high and/or medium: Lloyd Bay (remote Cape York region; Figure 4.4); and the beaches north of Cairns, Trinity Inlet (adjacent to Cairns), the Cassowary Coast, Hinchinbrook region, Cleveland Bay, Bowling Green Bay, Alva Beach, Upstart Bay, Abbot Bay, Edgecombe Bay, Whitsunday Islands, and Rodds Bay (urban coast; Figure 4.5).

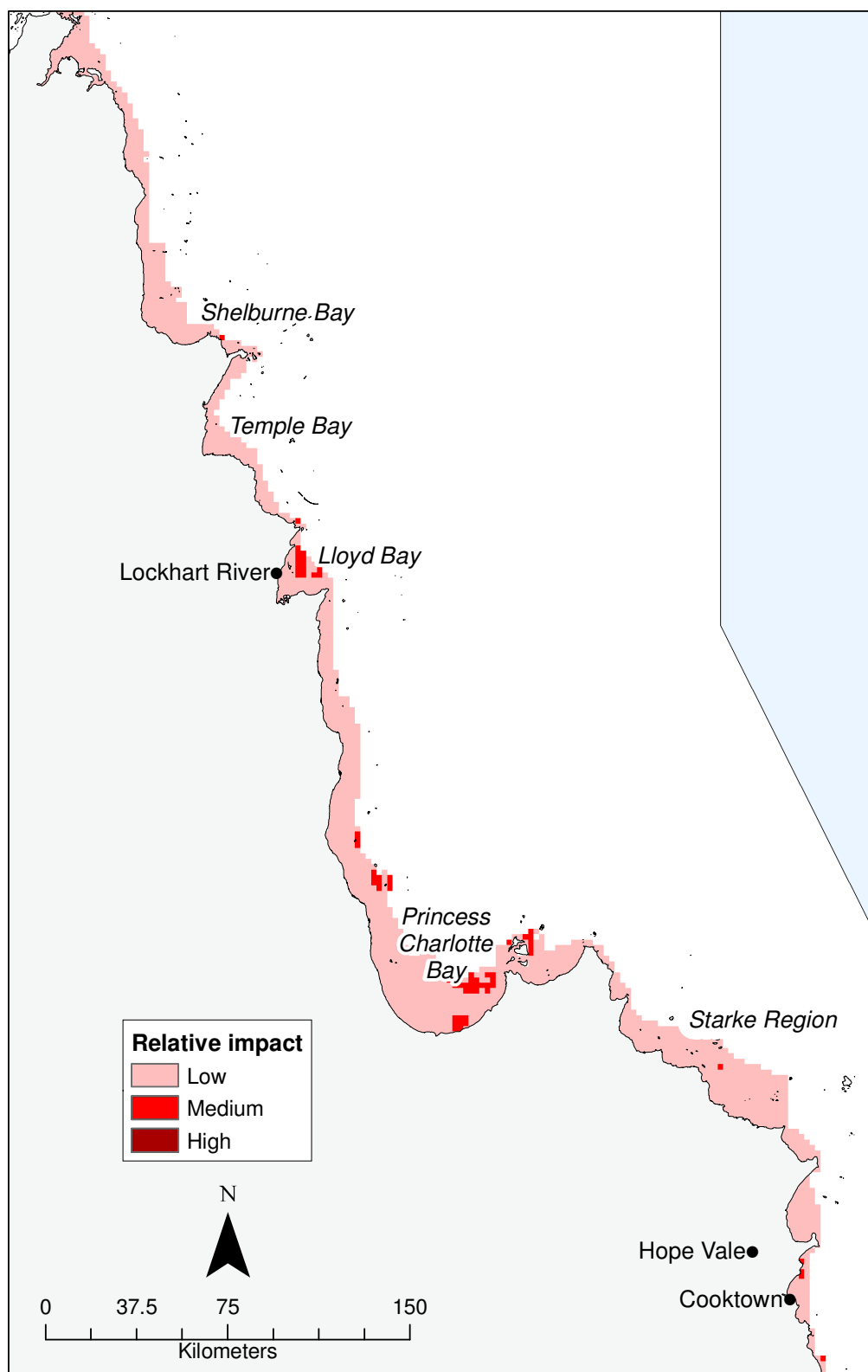


Figure 4.2: Clusters of planning units of low (cumulative score < 21.3), medium (21.3 – 37.0) and high relative impact (> 37.0) on coastal seagrass habitats in the remote Cape York region.

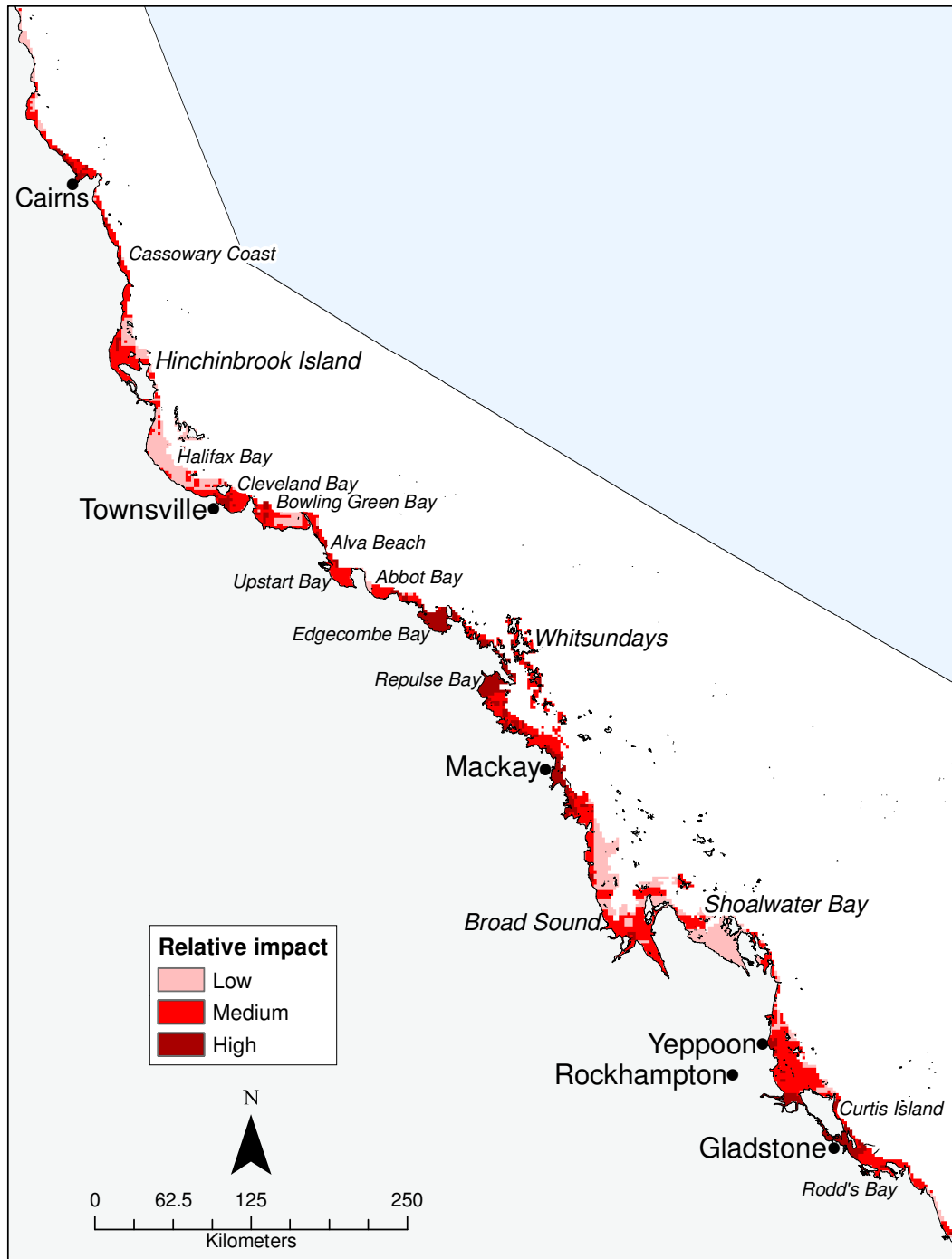


Figure 4.3: Clusters of planning units of low (cumulative score < 21.3), medium (21.3 – 37.0) and high relative impact (> 37.0) on coastal seagrass habitats along the urban coast.

Table 4.4: Percentage (%) of seagrass planning units of low (< 0.5), medium (0.5 – 0.75) and high (> 0.75) conservation value in the entire GBRWHA and the urban coast and remote Cape York regions with a low, medium and high risk from anthropogenic activities

	<i>Probability of seagrass occurrence</i>	<i>Total area (km²)</i>	<i>GBRWHA</i>			<i>Total area (km²)</i>	<i>Urban coast</i>			<i>Total area (km²)</i>	<i>Remote Cape York</i>		
			Low	Moderate	High		Low	Moderate	High		Low	Moderate	High
<i>Dry season</i>	< 0.5	16,580	50.6	36.8	12.6	12,136	34.3	48.4	17.3	4,444	95.0	5.0	0.0
	0.5 – 0.75	2,132	46.5	45.6	7.9	1,720	34.2	56.0	9.8	412	98.1	1.9	0.0
	> 0.75	1,336	88.0	9.3	2.7	164	19.5	58.5	22.0	1,172	97.6	2.4	0.0
<i>Wet season</i>	< 0.5	17,164	50.9	36.9	12.2	11,916	31.3	51.2	17.6	5,248	95.4	4.6	0.0
	0.5 – 0.75	2,276	53.4	39.2	7.4	1,580	33.2	56.2	10.6	696	99.4	0.6	0.0
	> 0.75	580	89.0	7.6	3.4	64	18.8	50.0	31.3	516	97.7	2.3	0.0

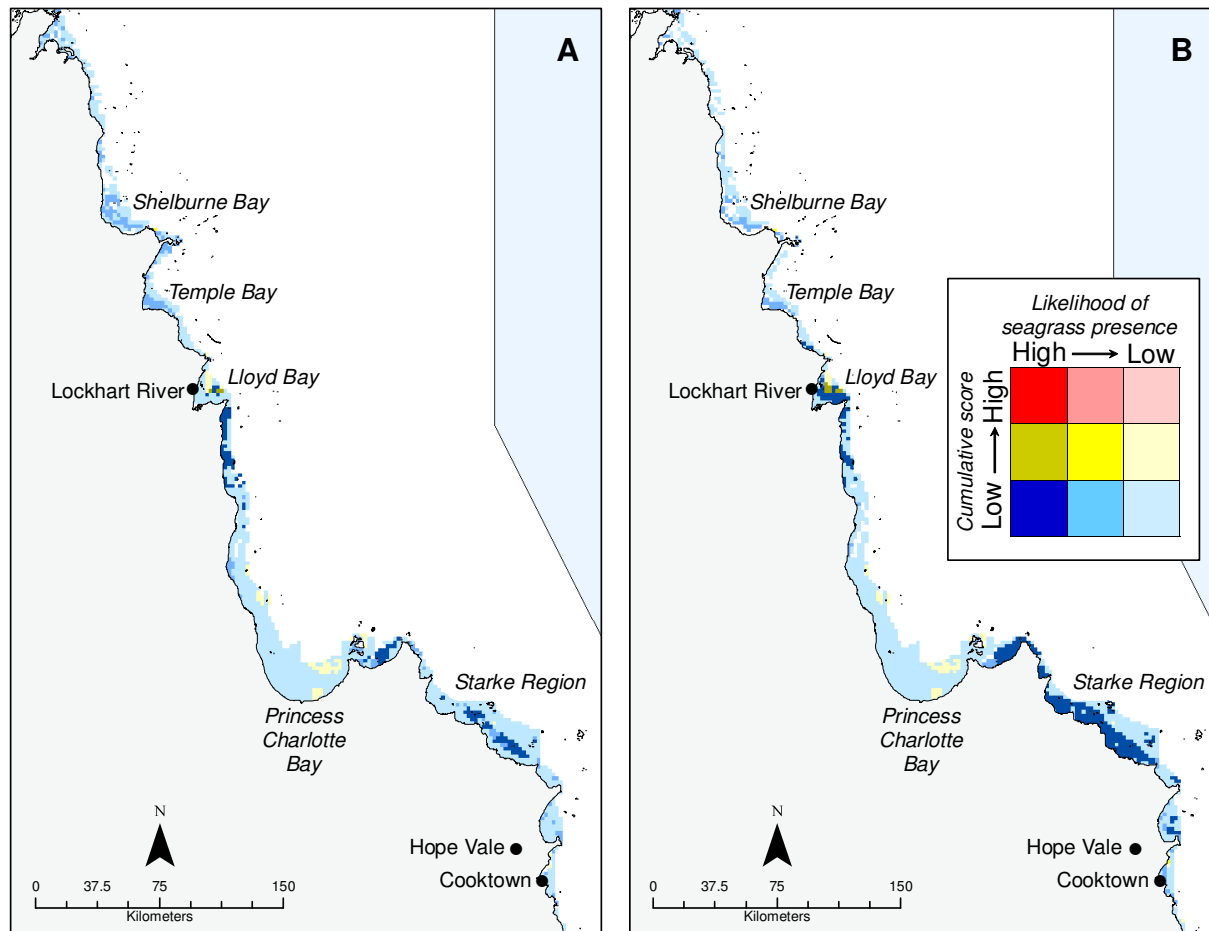


Figure 4.4: Planning units of high, medium and low conservation value to coastal seagrass habitats in the wet (A) and dry (B) seasons within areas of high, medium and low cumulative hazard scores in the remote Cape York region.

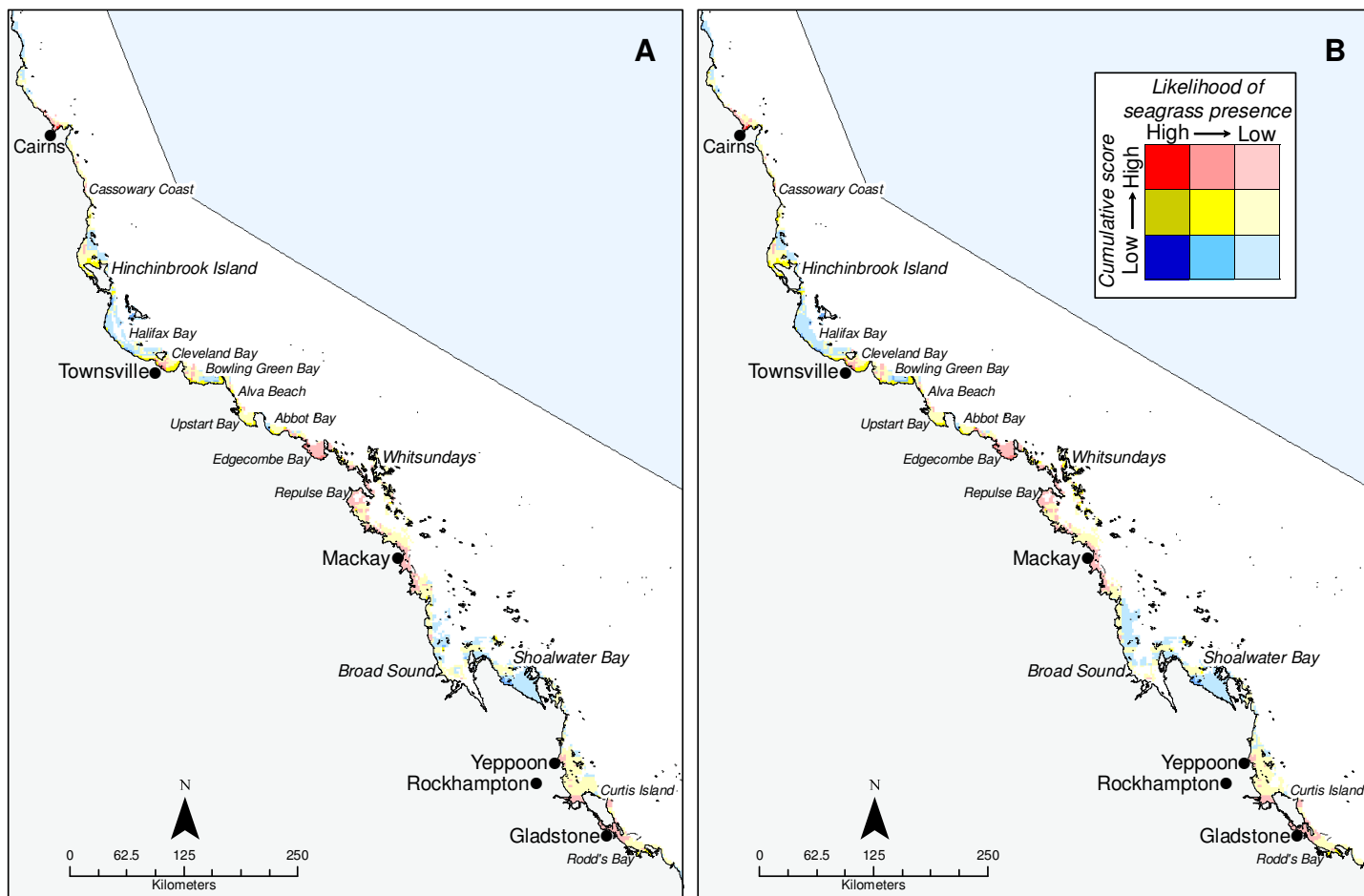


Figure 4.5: Planning units of high, medium and low conservation value to coastal seagrass habitats in the wet (A) and dry (B) seasons within areas of high, medium and low cumulative hazard scores along the urban coast.

Discussion

Quantitative information on the relative impact of anthropogenic threats on coastal seagrass habitats in the GBRWHA is incomplete or unavailable, and the cumulative impact of multiple threats is difficult to measure and predict. In this chapter, I overcame the difficulties associated with assessing the impact of multiple threats on coastal seagrass habitats by using expert opinion and a risk assessment framework. The outputs of the risk assessment were the identification of: (1) anthropogenic hazards with the greatest relative impact on coastal seagrass habitats; and, (2) 'hot spots' that are a priority for conservation actions at the scale of the coastal GBRWHA (22,600 km²).

The survey of experts identified agricultural runoff as the greatest hazard to coastal seagrass habitats relative to other threats, followed by urban and industrial runoff, urban and port infrastructure development, dredging, shipping accidents, trawling, recreational boat damage, commercial boat damage and fishing (Table 4.3). However, I found no significant differences in the weighted-average vulnerability scores provided by experts. This indicates that all hazards should be important to managers as they were considered by experts to have a similar impact on coastal seagrasses.

Certainty estimates provided by experts allowed a qualitative assessment of the depth of knowledge used to determine vulnerability scores (Tables 4.2 and 4.3). Experts reported the greatest certainty for the direct impacts of dredging, followed by urban and port infrastructure development and shipping accidents. Agricultural runoff and urban and industrial runoff had lower certainty scores (Table 4.3). Poor quality terrestrial runoff from adjacent land catchments and increased loading of sediment, contaminants and nutrients has demonstrable impacts on the health of coastal seagrass habitats worldwide (Orth et al. 2006). However, the link between habitat loss and poor water quality is likely to be more spatially and temporally variable and therefore less certain than those direct impacts such as dredging and coastal development that mechanically remove seagrass habitats (Duarte 2002). Estimates of certainty provided by the experts reflected the relative difference in knowledge on the threat to coastal seagrass habitats from direct and indirect impacts. Lack of certainty on the impact of poor water quality on coastal seagrass habitats (which was deemed by experts to be the greatest hazard relative to other threats) highlights the need for experimental work that assesses the responses of

seagrasses to changes in water quality. Research that investigates the seasonal factors that affect seagrass growth and reproduction and nutrient pulsing during high-rainfall events are also required. It should be noted that: (1) there was no significant difference in the certainty scores provided by experts; and (2) although experts are less certain about the relative impact of poor water quality on coastal seagrass habitats, there was strong consensus among experts on its relative impact on coastal seagrass habitats in the GBRWHA (Table 4.2).

I delineated areas of low, medium and high relative impact (Figures 4.2 and 4.3) to coastal seagrass habitats and compared the output with the seagrass habitat suitability maps outlined in Chapter 3 (Figures 3.4 and 3.5). I found that at the scale of the entire GBRWHA approximately 88% and 89% of planning units of medium and high conservation value to seagrass during the dry and wet seasons are at low relative risk from anthropogenic hazards respectively (Table 4.4; Figures 4.4 and 4.5). I calculated the relative risk of coastal seagrass habitats in the urban coast and remote Cape York region and found: a third (< 34%) of coastal seagrass habitats along the urban coast were at low risk from anthropogenic hazards; and almost all (> 95%) coastal seagrass habitats in the remote Cape York region were at low risk. The substantial difference in risk between the two regions is due to the small size of the land catchments in the remote Cape York region (*see* Chapter 2 page 20; Figure 2.1), and the relatively minor impact of industrial activities, mines, urban centres and agricultural activities in those catchments (*see* Chapter 2 page 23).

I identified planning units that are a priority for conservation action ('hot spots') because: (1) the cumulative hazard score was medium or high; and, (2) the seagrass conservation value of the planning units was medium or high. Along the remote coast of Cape York Peninsula, one 'hot spot' was identified in the deeper waters (> -5m bathymetry) of Lloyd Bay, adjacent to the Lockhart River community (Figure 4.4). Three hazards were present in this 'hot spot': trawling, recreational boating activities and fishing (other than trawling). Each of the three hazards had relatively lower weighted-average vulnerability scores (Table 4.3); but when combined their cumulative scores were classified as a medium impact to coastal seagrass habitats (Figure 4.2).

In the coastal waters along the urban coast, I identified twelve ‘hot spots’ for conservation action: the beaches north of Cairns, Trinity Inlet (adjacent to Cairns), the Cassowary Coast, Hinchinbrook region, Cleveland Bay, Bowling Green Bay, Alva Beach, Upstart Bay, Abbot Bay, Edgecombe Bay, Whitsunday Islands, and Rodds Bay (Figure 4.5). The ‘hot spots’ were limited to planning units in shallow waters (> -5 m bathymetry) as coastal seagrass habitats are more likely to be present at this depth along the urban coast than in deeper waters (*see* Chapter 3). The hazards that contributed to the medium and high cumulative scores for the ‘hot spots’ along the urban coast included a combination of three or more of the following hazards: agricultural runoff, urban and industrial runoff, urban and port infrastructure development, dredging, shipping accidents, recreational boating activities, and fishing (other than trawling). Hazards that did not contribute to the ‘hot spots’ included commercial boating activities and trawling. The ‘hot spots’ generally occurred within the limits of ports and/or were adjacent to urban/industrial centres e.g. Cairns, Cardwell, Townsville, and Bowen (Figure 4.5).

Limitations of the assessment

Halpern et al. (2008a) and Selkoe et al. (2009) identify the impacts of anthropogenic climate change as the greatest relative threat to marine ecosystems. I did not identify the impacts of anthropogenic climate change as hazards to coastal seagrass habitats in this study because there is no evidence that seagrasses are currently threatened by climate change in the GBRWHA (Short and Neckles 1999; Waycott et al. 2009). However, the Great Barrier Reef Outlook Report (2009) identified climate change as the greatest risk to the future health of ecosystems and species in the GBRWHA. The assessment outlined in this chapter will need to be updated in the future to account for climate change related impacts and other changes in the anthropogenic threats when more information is available.

I assumed that the cumulative effect of multiple anthropogenic threats on coastal seagrass habitats was additive because there is no evidence of synergistic or antagonistic relationships between two or more threats in the GBRWHA. Crain et al. (2008) review studies on the interactive and cumulative effects of multiple anthropogenic threats in marine systems and found: (1) the cumulative effect of two threats in individual studies

is additive (26% of individual studies), synergistic (36%) and antagonistic (38%); and, (2) the number of synergistic interactions doubles when three or more threats are present. The findings of Crain et al. (2008) have implications for this study's risk assessment because synergisms between hazards may amplify the cumulative risk to seagrass habitats when two or more hazards are present. The difference in relative risk between 'hot spots' where there are few hazards and 'hot spots' where there are many hazards may be larger than indicated by this assessment.

In Chapter 3, I identified potential sources of error and uncertainties in the predictive model of coastal seagrass distribution. A consequence of the error and uncertainties in that model is the incorrect prediction of the distribution of coastal seagrass habitats in the habitat suitability map; with consequential implications for the outputs of this risk assessment because the identification of 'hot spots' for conservation action were informed by the habitat suitability maps. Potential impacts of model uncertainty for the risk assessment included errors of commission (i.e. seagrass planning units that do not have a medium/high conservation value and are therefore not 'hot spots') and/or omission (i.e. planning units were not identified as a 'hot spot' because their conservation value was improperly identified as low). As new information becomes available, this assessment can easily be improved by updating the predictive model and geographic layers, and revaluating the risk assessment.

Comparison of results with a global assessment

Halpern et al. (2007) devise an approach to evaluate and rank the vulnerability of multiple marine ecosystems to anthropogenic threats at a global scale. I used their approach to evaluate the vulnerability of coastal seagrass habitats at the scale of the coastal GBRWHA (22,600 km²). Halpern et al. (2007) found that at a global scale the anthropogenic threats with the greatest relative impact on coastal seagrasses are: coastal development (e.g. land fill, land reclamation and dredging); increases in sediment input from activities such as logging, agriculture and urban development; sea level rise; direct human impacts (e.g. trampling, noise and light pollution); and coastal engineering (e.g. the construction of seawalls, jetties and piers). The anthropogenic activities identified by Halpern et al. (2007) are similar to those hazards identified as having the greatest relative impact on coastal seagrasses in this chapter (Table 4.3), even though my

assessment was developed specifically for tropical seagrass habitats and was conducted at a regional scale.

The differences in results in my assessment and Halpern et al. (2007) can be explained by the scale at which the assessments were conducted. Halpern et al. (2007) assesses the vulnerability of all seagrass species (i.e. tropical, subtropical and temperate species) while I assessed the vulnerability of tropical seagrasses only. There are large differences in the potential impact of some anthropogenic threats on tropical, subtropical and temperate seagrasses. For example, trawl fisheries are considered to be the greatest threat to temperate seagrass habitats in the Mediterranean Sea as these habitats are dominated by the slow-growing *Posidonia oceanica* which has a low resistance to trawl events and a long recovery time (Guillén et al. 1994). The relative impact of the trawl fishery on coastal seagrass habitats in the GBRWHA is much lower than in the Mediterranean Sea. The resistance and recovery time of tropical seagrass habitats exposed to trawling in the GBRWHA was considered by experts to be high as these habitats are dominated by fast-growing and ephemeral species.

Some of the anthropogenic activities listed as a threat to marine ecosystems by Halpern et al. (2007) were not identified by experts as a hazard to coastal seagrasses in the GBRWHA. At a global scale, the waters of northern Australia are considered one of the least impacted regions (Halpern et al. 2008), and anthropogenic activities that are threats to coastal seagrass habitats in most parts of the world do not occur in the GBRWHA. For example: oil drilling and mining are banned within the Great Barrier Reef Marine Park (*see* Chapter 2 page 22); harmful algae blooms are rare; the impact of aquaculture is minor (there is currently only one commercial aquaculture operation in the entire GBRWHA); and noise, light and thermal pollution are minimal (GBRMPA 2009). My assessment of the vulnerability of coastal seagrasses to anthropogenic threats takes into account these regional differences.

Implications for the planning and management of coastal seagrass habitats in the GBRWHA

The coastal waters (approximately -15 m below mean sea level) along the urban coast are approximately 15,700 km²; more than twice the size of the coastal waters off the

Cape York Peninsula (~ 6,900 km²). Although the extent of the coastal waters in the remote Cape York region are considerably less than along the urban coast, almost half of seagrass planning units with high and medium conservation value are found in the remote Cape York region (*see* Chapter 3 page 47). The virtual absence of medium and high risk areas to coastal seagrass habitats in the remote Cape York region has resulted in a substantial proportion of seagrass planning units being classified as low risk at the scale of the entire coastal GBRWHA (i.e. ~ 88%). In contrast, along the urban coast, I found that almost two thirds of seagrass habitats of high or medium conservation value are at high or medium risk from multiple anthropogenic activities (Table 4.4).

Coastal seagrasses in the GBRWHA are protected under Queensland and Australian laws. Queensland's *Fisheries Act* 1994 allows for the destruction, damage or disturbance of seagrass habitats when a permit has been assessed and issued by Queensland Primary Industries and Fisheries. The zero net-loss policy in Australia (Coles and Fortes 2001) requires the Queensland Government to consider any losses caused by direct human intervention when issuing a permit. These losses should be compensated by the creation or protection of a similar extent of seagrass habitat. Anthropogenic activities that are an indirect threat to seagrasses such as poor water quality from agricultural, urban and industrial runoff do not require a permit under the *Fisheries Act* 1994. In this chapter, I found that almost two thirds of habitats along the urban coast are in areas of high or medium risk from both direct and indirect threats. The zero net-loss policy has not succeeded in protecting seagrass habitats along the urban coast as it does not take into account the destruction, damage or disturbance caused to seagrass by indirect threats. A zero net-loss policy that considered both direct and indirect threats to coastal seagrass habitats would require the Queensland and Australian Governments to: (1) mitigate the potential loss of seagrass habitats along the urban coast by offsetting its loss with the improved protection of habitats in the remote Cape York region; or, (2) reduce the risk to seagrass habitats along the urban coast, especially in areas identified in this study as 'hot spots' for conservation action. Reducing the risk to coastal seagrass habitats in these 'hot spots' will require addressing all the hazards by: (1) improving the quality of terrestrial water that enters the GBRWHA; (2) mitigating the impacts of urban and port infrastructure development and dredging; and, (3) reducing the incidence of shipping accidents and recreational boat damage along the urban coast of the GBRWHA.

Direct impacts that physically remove or destroy seagrass habitats can generally be addressed with management tools such as legislation and sanctions; whilst indirect threats require a cross-jurisdictional approach (Coles and Fortes 2001). As a result, current management arrangements that control the risk to seagrass habitats in the GBRWHA are complex. McGrath (2003) identifies nearly 50 legislative instruments, International, Commonwealth and State that make up the Queensland environmental legal system all of which have potential to influence seagrass management. Like the anthropogenic threats they are designed to address, they also have a spatial dimension in application. Intense and small scale impacts such as dredging are well defined in legislation and have a high level of management intervention. Broad scale impacts that cross jurisdictions such as the impact of poor quality runoff from the adjacent land catchments on seagrass habitats are less intensely managed. The approach I have adopted in this study provides a step towards recognising and separating out those levels of risk and management intervention.

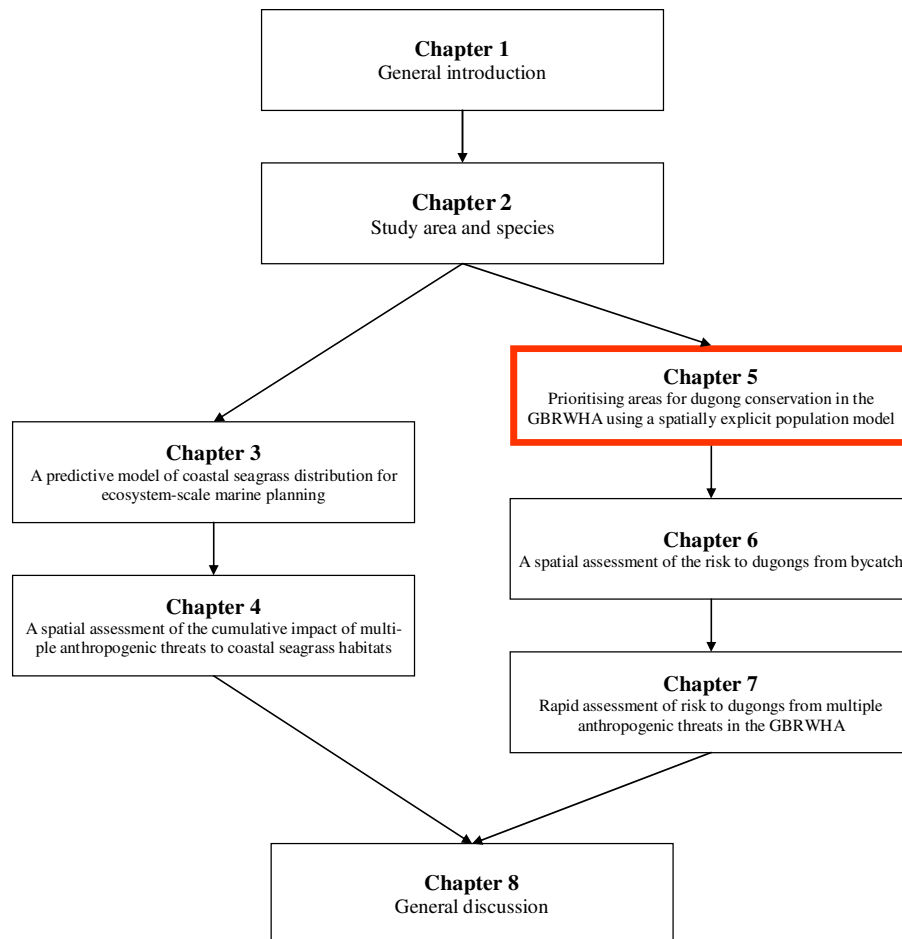
Chapter Summary

- Seagrasses in the coastal GBRWHA (22,600 km²) are exposed to multiple anthropogenic threats, but assessing the impact of one and/or more threats and the protection afforded to seagrass habitats is made difficult by the uncertainties in the data available, and the cost associated with collecting information at that scale.
- In this chapter, I used a spatial risk assessment approach to assess the risk of coastal seagrass habitats from multiple anthropogenic threats. In the face of the uncertainty of the impact of multiple threats on coastal seagrasses, I used a qualitative assessment informed by expert opinion to quantify the relative impact of multiple threats.
- I compared the habitat suitability maps for coastal seagrasses outlined in Chapter 3, with models of the distribution of threats to quantify the risk of seagrass habitats from their anthropogenic threats in the GBRWHA.
- I found a marked difference in the relative risk to planning units along the urban coast compared with the remote Cape York region. Almost all (> 95%) seagrass habitats in the remote Cape York region are at low risk from anthropogenic threats. Along the urban coast, less than 34% of seagrass habitats are at low risk.
- In the coastal waters of the GBRWHA I identified 13 ‘hot spots’ for conservation action: Lloyd Bay in the remote Cape York region; and the beaches north of Cairns, Trinity Inlet (adjacent to Cairns), the Cassowary Coast, Hinchinbrook region, Cleveland Bay, Bowling Green Bay, Alva Beach, Upstart Bay, Abbot Bay, Edgecombe Bay, Whitsunday Islands, and Rodds Bay along the urban coast.

Chapter 5

Prioritising areas for dugong conservation in the GBRWHA using a spatially-explicit population model¹

In this chapter, I outline the development of a spatially-explicit dugong population model that can assist managers in prioritising the administration of conservation resources at the GBRWHA scale. I use information collected from dugong aerial surveys in conjunction with geostatistical techniques to develop a model of dugong distribution and relative abundance. I classify each dugong planning unit as low, medium, or high conservation value on the basis of the relative density of dugongs estimated from the model and a frequency analyses.



¹ Grech, A. and Marsh, H. 2007. Prioritising areas for dugong conservation in a marine protected area using a spatially explicit population model. *Applied GIS* 3(2): 1 – 14.

Introduction

Aerial surveys conducted by Helene Marsh and her group at James Cook University have systematically monitored dugong (*Dugong dugon*) abundance and distribution in the Great Barrier Reef World Heritage Area (GBRWHA), Queensland since the mid 1980s (Marsh and Saalfeld 1989 and 1990; Marsh et al. 1993 and 1996; Marsh and Lawler 2001 and 2002). The surveys extend from the southern edge of the GBRWHA (24°30'S) into the remote Cape York region (11°32'S) (Figure 5.1) and cover ~73,000 km² (21%) of the GBRWHA; substantially more than the dugong's recorded range within the region.

Aerial surveys provide information on shifts in dugong habitat use at sub-regional (bay) scales including dugong responses to seagrass loss from extreme weather events (Preen and Marsh 1995; Gales et al. 2004; Marsh et al. 2004; Marsh et al. 2006). Aerial surveys have been less useful in detecting long-term trends in abundance and for setting sustainable catch quotas because of difficulties in: (1) separating changes in spatial distributions of animals from changes in population size; (2) estimating absolute population size in the absence of defined stock boundaries; and (3) stabilising the corrections for availability bias which varies during the day due to diurnal changes in dugong behavior (H. Marsh, personal communication).

Similar to the coastal seagrass habitats discussed in Chapters 3 and 4, effective management of dugongs along the entire coastline of the GBRWHA (~ 2,300 km) is constrained by the cost associated with implementing conservation action at that scale. In the absence of information on long-term trends in abundance and dugong responses to management interventions, spatially-explicit models of dugong distribution are required to inform the strategic deployment of conservation resources in the GBRWHA. Spatially explicit models of species distributions identify sites where species are most abundant over broad spatial scales, and where conservation actions should provide the greatest positive impact over their entire distributional range (Theobald 2003).

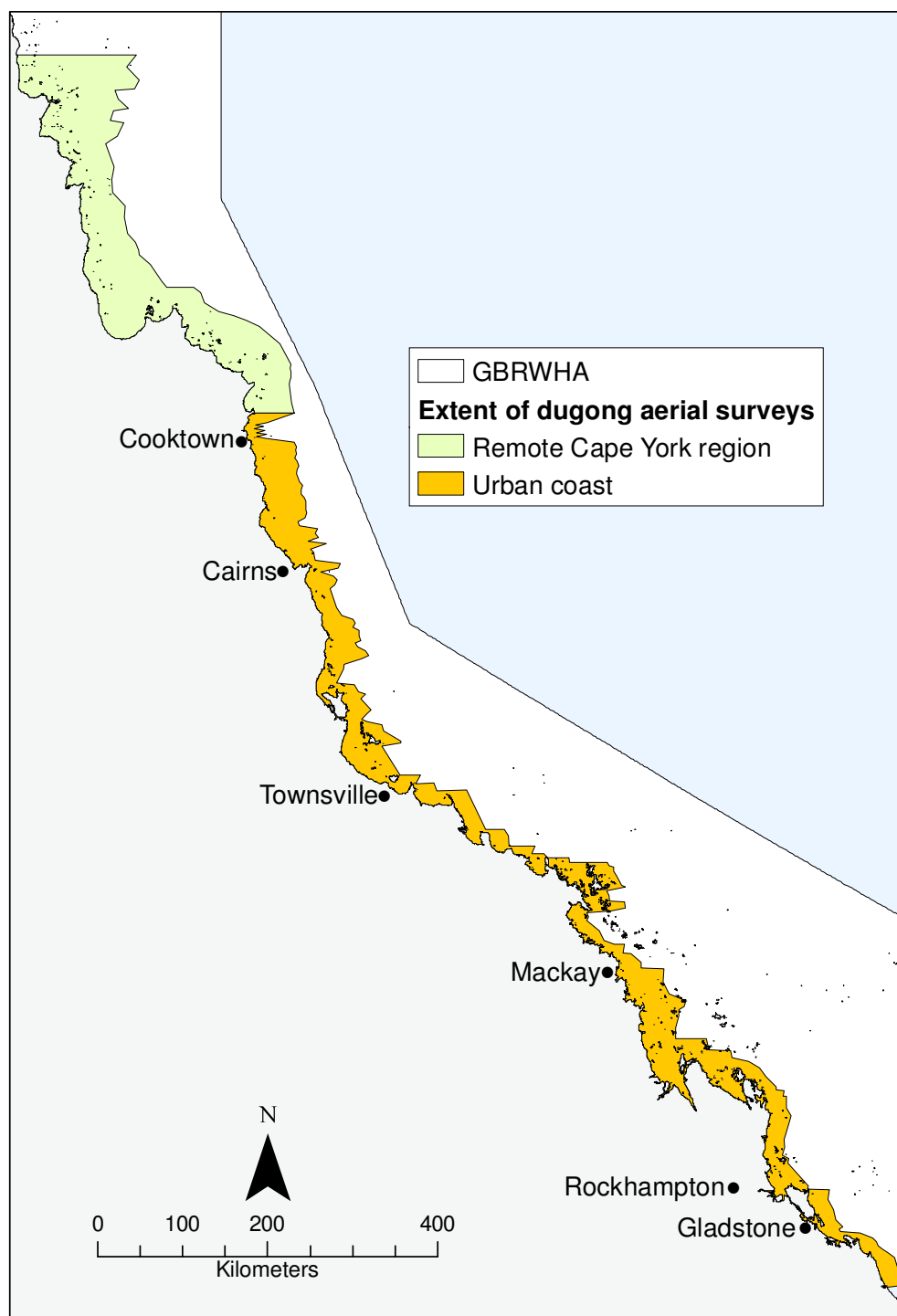


Figure 5.1: Extent of dugong aerial surveys ($\sim 73,000 \text{ km}^2$) along the $\sim 2,300 \text{ km}$ GBRWHA coastline.

As discussed in Chapter 2, dugongs are specialist feeders on most of the species of tropical and subtropical seagrasses within their range. Although dugong distribution is highly correlated with the distribution of intertidal and subtidal seagrass habitats, dugongs do not exploit all the coastal seagrass habitats that are in the GBRWHA. For example, the number of dugongs sighted in Trinity Inlet adjacent to the urban centre of Cairns is low, even though the region is known to support extensive seagrass habitats (Figure 3.5). A reason for the lack of correlation between dugong and seagrass distribution at some sites is that the selection of species and habitats by dugongs is influenced by multiple factors including fibre, starch and nitrogen content and biomass (Lanyon and Sanson 2006a and b; Sheppard et al. 2008; *see* Chapter 2 page 30). A predictive habitat distribution model for dugongs would require information on the distribution of: (1) individual seagrass species; (2) seagrass habitat community composition; and (3) the various factors that influence the choice of species or habitats by dugongs. As discussed in Chapter 3, this information is unavailable at the spatial scale of the entire coastal GBRWHA. The habitat suitability maps for coastal seagrasses that I outlined in Chapter 3 are unable to inform a predictive habitat model for dugongs in the GBRWHA as they: (1) do not predict the distribution or biomass of individual species of seagrass; and, (2) do not extend to the entire range of dugongs in the GBRWHA because dugongs are also found in offshore waters, especially in the remote Cape York region.

In this chapter, I used a different modelling approach than the one described in Chapter 3 because of the lack of information on the distribution of the dugongs preferred habitat. Instead of quantifying species-environment relationships, I developed a spatially explicit model of dugong distribution and relative abundance using: (1) information collected from the 20 year time-series of dugong aerial surveys; and (2) the geostatistical interpolation technique, universal kriging. I classified dugong planning units as low, medium, or high conservation value at the scale of the GBRWHA on the basis of the relative density of dugongs estimated from the model.

Methods

Aerial surveys are conducted using the strip transect method described by Marsh and Sinclair (1989). The survey region is divided into blocks containing systematic transects of varying length, which are typically perpendicular to the coast across the depth gradient. Tandem teams with two observers on each side of the aircraft independently record sightings of dugongs, including information on group size and calf numbers. Transects are 200 m wide at the water's surface on either side of the aircraft.

I derived the spatially-explicit dugong population model from information collected during six aerial surveys of the urban coast (1986, 1987, 1992, 1994, 1999, and 2005) and three surveys of the remote Cape York region (1990, 1995, and 2000). Similar to the seagrass surveys (*see* Chapter 3 page 31), the dugong aerial surveys are conducted in late spring or early summer when weather and sea states provide optimum survey conditions. Dugongs are unlikely to exhibit a seasonal component in their movements in the GBRWHA (*see* Chapter 2 page 32), and so conducting multiple surveys in the same season should not confound the relative estimates of dugong distribution and abundance.

I estimated dugong distribution and relative abundance at a planning unit of cell size 2 km * 2 km, the same scale as the predictive seagrass model I outlined in Chapter 3. This scale was chosen as: (1) it corresponds with the scale of the aerial survey data allowing the model to account for: (a) slight changes in altitude of the aircraft (which affects transect width at the surface); and, (b) the blind area under the aircraft; and, (2) it is the scale recommended for use by managers of wildlife under Criterion B of the *International Union for Conservation of Nature and Natural Resources Red List* (IUCN 2001).

The sampling intensity of each survey block was determined by calculating the proportion of area surveyed. There are some (relatively minor) differences in sampling intensity per block and area sampled between surveys. The sampling intensity within surveys varies between survey blocks depending on dugong abundance, and range from approximately 6% to 40%. In my analysis, four computations were required within each 2 km * 2 km planning unit to correct for these differences.

Firstly, each dugong observation (u) was multiplied by the reciprocal of the sampling intensity (SI) of its transect:

$$u_n \frac{1}{SI_n}$$

Secondly, the mean sampling intensity of all relevant survey transects was calculated:

$$\frac{\sum_{i=1}^n SI}{N}$$

where N is the number of surveys within the region and 1, 2,...n is an individual survey identifier. Each value from the first step was then divided by the mean sampling intensity of the relevant transect:

$$\sum_{i=1}^n u_i \frac{1}{SI_i} \bigg/ \frac{\sum_{i=1}^n SI}{N}$$

Lastly, the resulting value was divided by the number of surveys conducted on each transect to obtain a mean index of dugong abundance for each planning unit.

Geostatistical analysis

I conducted the geostatistical analyses in the following sequence.

(1) The spatial autocorrelation of the data was investigated by a variogram analysis using the Geostatistical Analyst extension of ArcGIS® 9.0 (Environmental Systems Research Institute 2004). Spatial autocorrelation detects the spatial dependence of the relationship between two samples as a function of their separation distance and estimates the strength of this dependency (Vasiliev 1996).

The variogram analysis used the following circular model to estimate semivariance (Johnston et al. 2003):

$$\gamma(h; \theta) = \begin{cases} \frac{2\theta_s}{\pi} \left[\frac{\|h\|}{\theta_r} \sqrt{1 - \left(\frac{\|h\|}{\theta_r}\right)^2} + \arcsin \frac{\|h\|}{\theta_r} \right] & \text{for } 0 \leq \|h\| \leq \theta_r \\ \theta_s & \text{for } 0 \leq \|h\| \leq \theta_r \end{cases}$$

where $\theta_s \geq 0$ is the partial sill parameter, $\theta_r \geq 0$ is the range parameter and h the separation distance (lag) in metres. Lag h is defined as a vector that separates any two locations and has both distance and directional attributes. The directional effect of h was considered by estimating semivariance at different directions of h (0, 45, 90 and 135 degrees).

(2) Universal kriging was chosen as the most appropriate interpolation technique as it is robust to common attributes of ecological data (McKenney 1998; Ver Hoef 1993). Universal kriging is a geostatistical estimation method that returns unbiased linear estimates of point values where trends in data vary and regression coefficients are unknown. The Spatial Analyst© extension of ArcGIS® 9.1 (Environmental Systems Research Institute 2005) was used to spatially interpolate the data to a 2 km * 2 km planning unit.

Independently collected data on dugong distribution and abundance at the scale of the GBRWHA does not exist so I used a re-substitution approach to validate the spatially-explicit population model. I removed a random sub-sample of observations constituting 30% of the total observations and then tested it against dugong distribution and relative abundance predicted from a krige using the remaining 70% of observations.

I used a frequency histogram to categorise the dugong density of each planning unit as low, medium or high conservation value. This approach makes the assumption that dugong density is a robust index of a region's conservation value for dugongs. This assumption is justified because: (1) specialised areas of high conservation value such as calving or mating areas and migratory corridors have not been identified; and, (2) density estimates are regarded as robust surrogates of habitat utilisation (Hooker and Gerber 2004). Nonetheless, the classification scheme may underestimate the past conservation value of some areas along the urban coast by using information collected on dugong distribution and abundance, only since 1985. I assumed that there has been no overall decline in the dugong's extent of occurrence in the GBRWHA since 1985; a conclusion supported by the aerial survey results (although the power of the surveys to detect trends is limited).

Results and Discussion

The parameters of the variogram analysis are shown in Table 5.1, and the semivariogram in Figure 5.2. The variogram analysis indicated that dugong distribution is spatially autocorrelated i.e. the spatial component inherent within the data explained the sampled variation and the data are therefore suitable for interpolation. The directional variograms showed an isotropic relationship for dugong spatial distribution (Table 5.1). Cross-validation of the final kriging (Figures 5.3 and 5.4) reported reliable predictions of the model, with a low standard error (0.046).

The index of average relative dugong density in the entire GBRWHA was estimated to be 0.077 dugongs/km²; relative density ranged from 0.0011 to 9.86 dugongs/km². Average dugong relative density in the remote Cape York region was seven times higher than along the urban coast (0.145 compared with 0.020 dugongs/km²). The highest relative density was also greater in the remote Cape York region with a range of 0.0036 to 9.86 dugongs/km² compared with the urban coast (0.0011 to 1.92 dugongs/km²). Planning units of highest relative density in the remote Cape York region are adjacent to Friendly Point and Port Stewart and between Lookout Point and Princess Charlotte Bay (Figure 5.3). Planning units of highest relative density along the urban coast are north of Hinchinbrook Island, and in Cleveland Bay, Shoalwater Bay region and Port Clinton (Figure 5.4). The planning units identified as high dugong density areas relative to other units were consistent with the regions identified in previous studies as important seagrass habitats for dugongs in the GBRWHA (*see* Marsh and Saalfeld 1989 and 1990; Marsh et al. 1993 and 1996; Marsh and Lawler 2001 and 2002).

Aerial surveys estimate absolute dugong abundance by correcting sightings for perception bias (animals that are available to, but missed by, observers) and availability bias (animals that are unavailable to observers because of water turbidity) *sensu* Marsh and Sinclair (1989) and Pollock et al. (2006). The corrections for these biases are applied at the spatial scale of entire surveys (thousands of square kilometres), an inappropriate spatial scale for this research. As a result, the model of dugong distribution and abundance (Figures 5.3 and 5.4) is based on relative rather than

absolute total density. Nonetheless, the relative densities among regions should be approximately comparable (H. Marsh, personal communication).

Table 5.1: Isotropic and directional (0° , 45° , 90° and 135°) variogram model parameters for dugongs in the GBRWHA. RMSE: root-mean-square error between observed and predicted semivariance; SE: standard error.

Direction	Isotropic	0°	45°	90°	135°
Model fitted	Circular	Circular	Circular	Circular	Circular
Nugget (c_0)	0.262	0.255	0.261	0.255	0.261
Sill	0.465	0.459	0.464	0.455	0.464
Range (a)	437.39	437.39	437.39	358.78	429.24
Mean	0.077	0.076	0.076	0.075	0.076
RMSE	0.872	0.872	0.878	0.872	0.878
SE	0.046	0.015	0.009	0.015	0.009

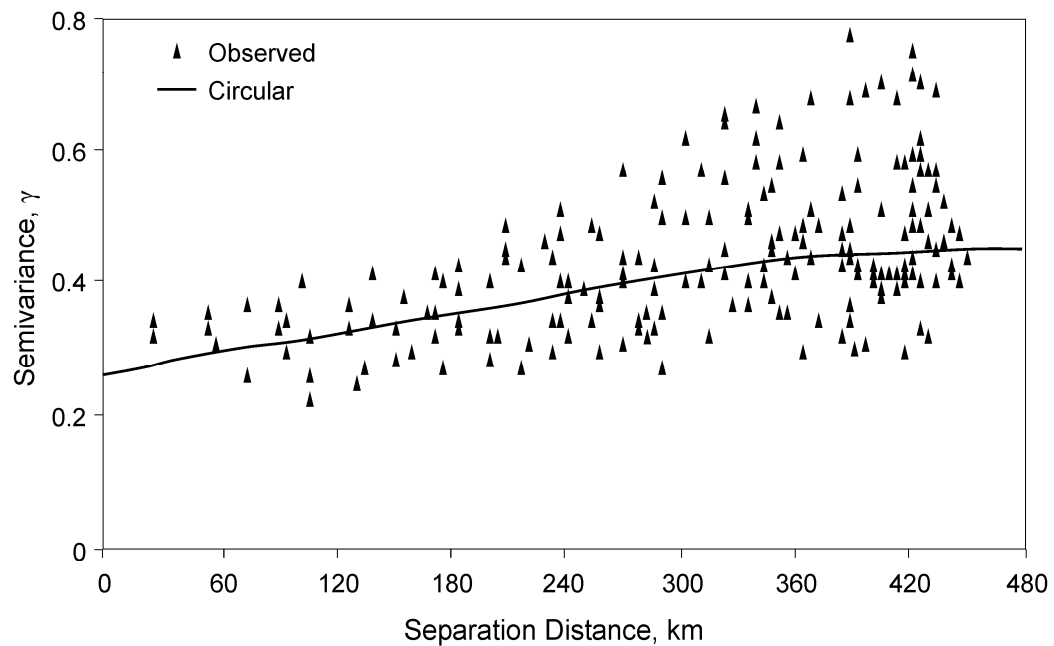


Figure 5.2: Dugong semivariogram for the observed aerial survey data and circular model. Distance is measured in kilometres.

The frequency histogram (Figure 5.5) revealed relative density thresholds within the dugong population for separation into areas of low, medium and high conservation value. Low conservation value areas had relative dugong densities of between 0.0015 - 0.25 dugongs/km²; medium conservation value 0.25 - 0.5 dugongs/km²; and high conservation value > 0.5 dugongs/km² (Figures 5.3 and 5.4).

The total area of dugong planning units in the GBRWHA predicted to be of high, medium and low conservation value were 2399 km², 2175 km², and 27,490 km², respectively (Table 5.2; Figures 5.3 and 5.4). The remote Cape York region had the greatest proportion of planning units of high (93%), medium (88%) and low (57%) conservation value for dugongs (Table 5.2; Figure 5.3). The predictions of the dugong distribution model are similar to the predictions of the habitat suitability maps for coastal seagrass I outlined in Chapter 3 as almost half of the seagrass planning units with high and medium conservation value are found in the remote Cape York region.

If additional critical areas for dugongs in the GBRWHA are identified (including breeding locations and migratory pathways), the approach I used to prioritise dugong conservation value would need to be modified. A suitable method would be that implemented in the initial stage of the Great Barrier Reef Marine Park re-zoning process; where special or unique places were identified and included within the network of 'no take' areas, regardless of the impact this approach had on the final model (Day et al. 2000; Fernandes et al. 2005). One of the strengths of the model is that it takes account of the large scale dugong movements that occur due to changes in seagrass habitats (Preen and Marsh 1995; Gales et al. 2004; Marsh et al. 2002, 2004 and 2005; Holly et al. 2006) because it is based on integrated data from six aerial surveys spanning 19 years along the urban coast and three surveys of the remote Cape York region conducted over a decade.

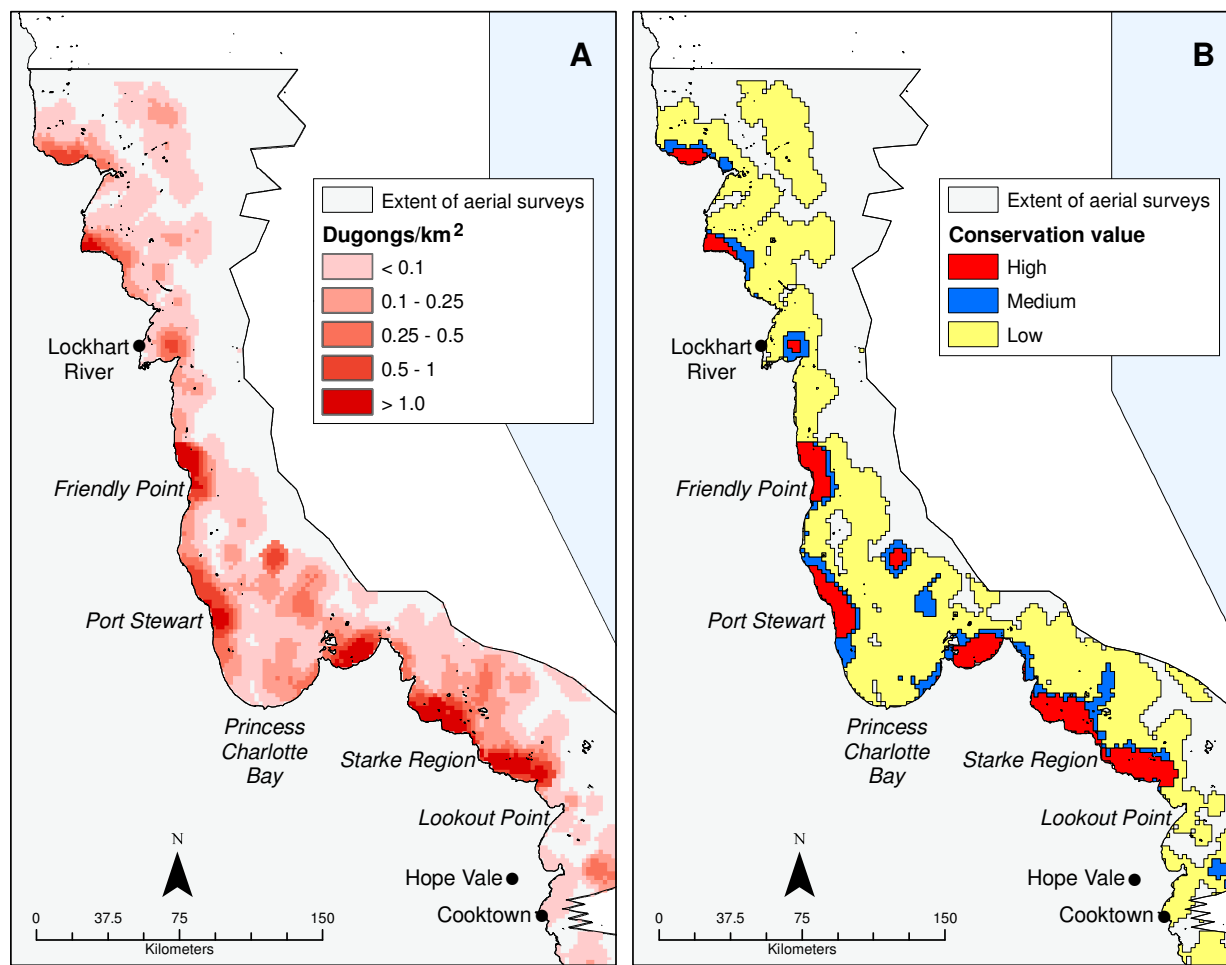


Figure 5.3: (A) Model of dugong distribution and relative abundance based on aerial survey data and a kriging interpolation; and (B) the corresponding levels of dugong conservation value in the remote Cape York region.

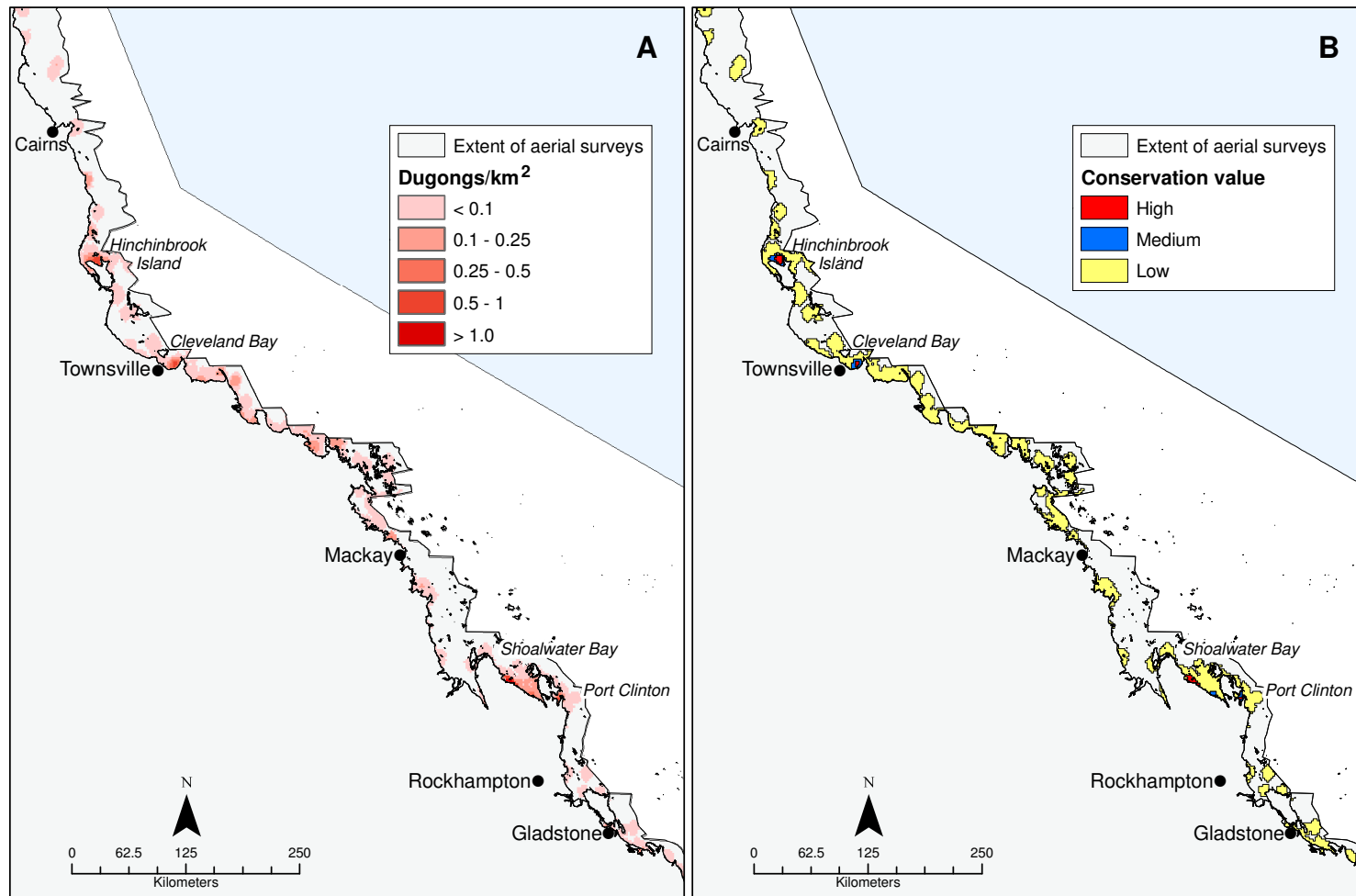


Figure 5.4: (A) Model of dugong distribution and relative abundance based on aerial survey data and a kriging interpolation; and (B) the corresponding levels of dugong conservation value in the along the urban coast.

Table 5.2: Total area (km²) of dugong planning units with high, medium and low conservation value in the entire GBRWHA, urban coast and remote Cape York region.

<i>Dugong Conservation Value</i>	<i>GBRWHA</i>	<i>Urban coast</i>	<i>Remote Cape York</i>
High	2,399	173	2,226
Medium	2,175	259	1,916
Low	27,490	11,771	15,719

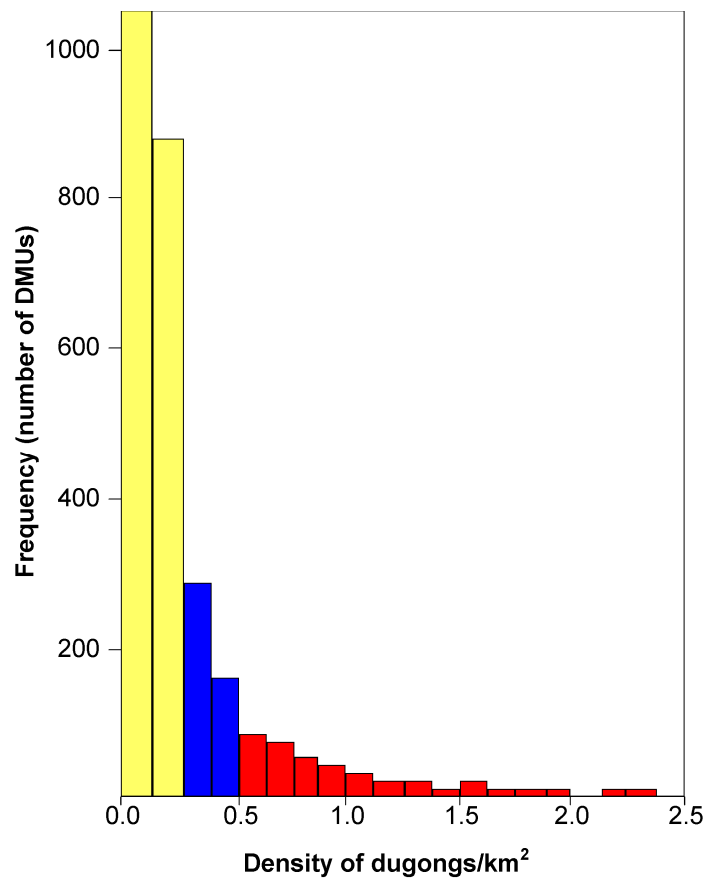


Figure 5.5: Frequency diagram of dugong relative density derived from a spatially explicit population model. Low conservation values have dugong densities between 0.0015 - 0.25 dugongs/km² (identified in yellow); medium conservation value 0.25 - 0.5 dugongs/km² (blue); and high conservation value > 0.5 dugongs/km² (red).

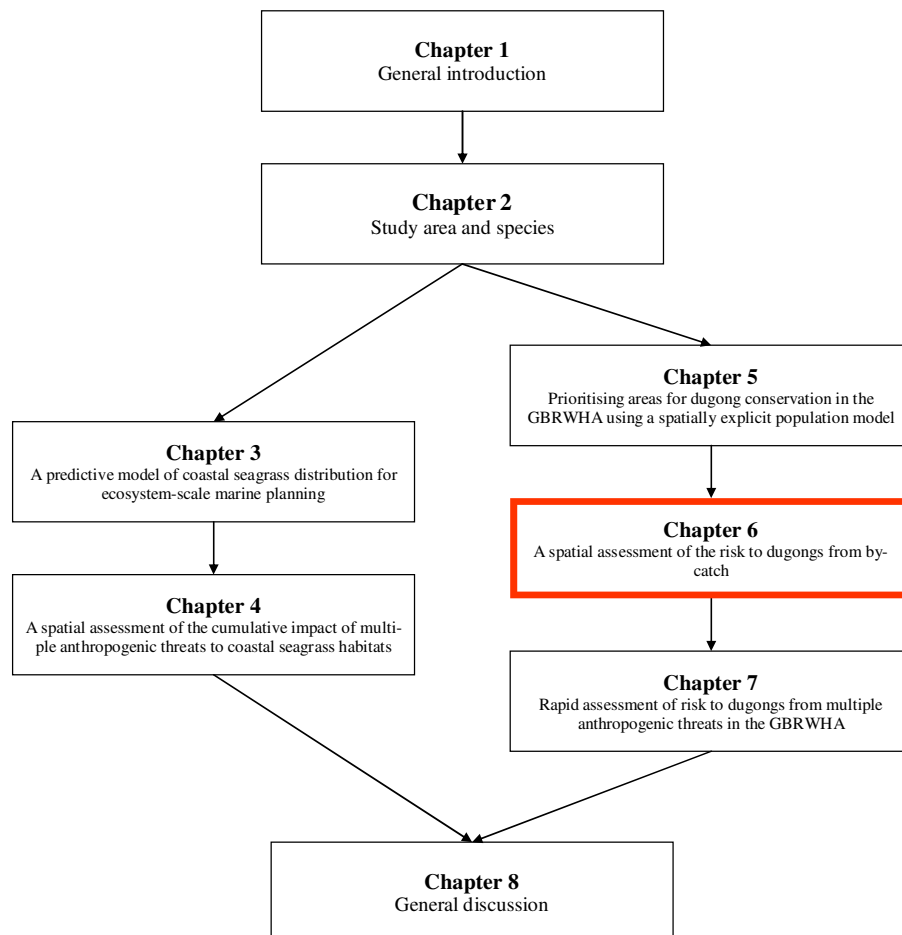
Chapter Summary

- In this chapter I outlined the development of a spatially-explicit model of dugong distribution and relative abundance that informs the strategic deployment of conservation resources along the ~2,300 km GBRWHA coastline.
- Information collected by dugong aerial surveys was used in conjunction with geostatistical techniques including universal kriging to develop a model of dugong distribution and relative abundance. I classified each dugong planning unit as low, medium, or high conservation value on the basis of the relative density of dugongs estimated from the model and a frequency analyses.
- Planning units of high conservation value in the remote Cape York region are adjacent to Friendly Point and Port Stewart and between Lookout Point and Princess Charlotte Bay. Along the urban coast, high conservation value planning units are north of Hinchinbrook Island, and in Cleveland Bay, Shoalwater Bay and Port Clinton.
- The spatially explicit model informs marine planning by detecting planning units where dugongs are most abundant at the scale of the GBRWHA, and where conservation actions should provide the greatest positive impact over their entire distributional range.

Chapter 6

A spatial assessment of the risk to dugongs from bycatch¹

In this chapter, I use a spatial risk assessment approach to evaluate the re-zoning of the Great Barrier Reef Marine Park in 2004 and associated industry restructuring for their ability to reduce the risk of dugong bycatch in gill-nets of the Queensland East Coast Inshore Fin Fish Fishery. I discuss how the approach is applicable to other situations where there is limited information on the location and intensity of bycatch, including remote regions and developing countries where resources are limited.



¹ Grech, A., Marsh, H. and Coles, R. 2008. A spatial assessment of the risk to a mobile marine mammal from bycatch. *Aquatic Conservation* **18**: 1127 – 1139.

Introduction

The single greatest threat to many stocks of marine mammals is incidental entanglement and mortality in fishing gear (bycatch) (Read and Rosenberg 2002). Many species of marine mega fauna are at risk of extinction from fisheries bycatch (Lewison et al. 2004). This threat has increased in frequency and intensity over time as a result of human population growth and the industrialisation of fisheries, and is expected to continue and rise (Read et al. 2006).

Several countries have developed comprehensive scientific and management programmes to evaluate interactions between marine mammals and fisheries (e.g. Read and Wade 2000). This approach is generally beyond the means of developing nations and is difficult to implement in small scale fisheries and in remote areas (Marsh et al. 2003). For example, the Potential Biological Removal technique, the approach required by the *US Marine Mammal Protection Act*, requires estimates of the absolute abundance and life history parameters of a marine mammal stock, and the bycatch of the fishery. Estimates of bycatch are usually collected through an observer program (Wade 1998), which can be impractical to implement in an artisanal fishery (Lewison et al. 2004). In addition, the Potential Biological Removal approach does not incorporate threats to marine mammal stocks other than anthropogenic-induced mortality. Such threats include habitat degradation, ecosystem changes, depletion of the prey base, predation or disease (*see* Chapter 4 and Taylor et al. 2007). Even the uncertainty analysis approach advocated by Lewison et al. (2004) may require more information than is available to inform effective solutions to the bycatch problem.

Several techniques have been developed and implemented to reduce the risk of fisheries bycatch to marine mammals including: (1) devices and gear changes that mitigate bycatch; and, (2) fisheries management policies such as time and area closures and fisheries moratoria (Lewison et al. 2004). As explained in Chapter 4, the concept of risk has two elements: (1) the likelihood of something happening; and, (2) the consequences if it happens (Norton et al. 1996). Technological solutions can be very effective at reducing either or both these elements of risk. For example, bycatch reduction devices (Hall et al. 2000) reduce the consequence of bycatch by allowing an animal which has been caught in fishing gear to escape. However, such solutions are generally difficult to

implement effectively for artisanal fisheries in developing countries and remote areas, largely because of cost (Marsh et al. 2003). Temporal closures restrict fishing for particular species for a period of time or season and are typically designed to reduce the consequences of bycatch when they are particularly serious e.g. when the bycatch species is breeding. Spatial (area) closures restrict the areas that can be fished and typically eliminate fishing from areas which consistently support high densities of the bycatch species. Area closures are not designed to eliminate the likelihood of individuals of mobile species being caught as bycatch; rather they reduce the risk to the bycatch population by eliminating the likelihood of bycatch to that proportion of the population that uses the closed area, either temporarily or permanently. When designed appropriately, closures can be very effective in reducing the bycatch of marine mammals (Murray et al. 2000).

Area closures have been used since the early 1980s in association with other fisheries management policies to reduce the risk of dugong (*Dugong dugon*) bycatch in commercial gill-nets in the Great Barrier Reef World Heritage Area (GBRWhA) and adjacent Hervey Bay (Figure 1.1). In 1998, management policies including 16 Dugong Protection Areas (DPAs) (Marsh 2000) were also introduced under fisheries legislation to further reduce incidental dugong mortality in gill-nets (*see* Chapter 2 page 33; Figure 2.5). Foreshore and offshore set or drift nets were prohibited in seven Zone A DPAs totaling 2,407 km². Less-restrictive modifications were introduced in eight Zone B DPAs totaling 2,243 km² (Marsh 2000).

As explained in Chapter 2, the GBRWhA was re-zoned in 2004 to increase the protection of marine biodiversity through a comprehensive and representative multiple-use zoning regime (Fernandes et al. 2005). Although biodiversity protection was the primary reason for this re-zoning, the Biophysical Operational Principles developed to guide the management agency in developing the network of 'no-take' areas included a commitment to ensure that 50% of 29 high priority dugong habitats were closed to all fishing activities, including gill and mesh nets used in the Queensland East Coast Inshore Fin Fish Fishery (Fernandes et al. 2005).

The Australian government provided a structural adjustment package to assist fishers adversely affected by the 2004 re-zoning. The package was also intended to: (1) prevent

displacement in fishing effort to other fishing grounds that would result in unsustainable ecological or economic impacts; and, (2) reduce overall effort in concordance with area closures (Marine Protected Area News 2006). Queensland Primary Industry and Fisheries also implemented a policy to control latent effort (allocated effort that is not currently deployed) in the East Coast Inshore Fin Fish Fishery to ensure its economic sustainability; resulting in a 40% reduction in the number of inshore net licenses in 2005.

As discussed in Chapter 1, Fernandes et al. (2005) and Dobbs et al. (2008) assess the degree to which new zoning in the GBRWHA achieved the goal of closing 50% of 29 high priority dugong habitats to fishing activities. Fernandes et al. (2005) found that this goal is achieved, whilst Dobbs et al. (2008) found that only 42% of the 29 sites are in 'no-take' areas. Both studies are limited in their ability to assess the degree to which new zoning in the GBRWHA protects dugongs as they do not use information that accurately delineates the dugong's spatial distribution (*see* Chapter 1 page 9). Furthermore, Fernandes et al. (2005) and Dobbs et al. (2008) do not provide explicit information on the risk to dugongs from bycatch in commercial gill-nets *per se* as all fishing activities (i.e. netting, trawling, crabbing, long-lining and recreational fishing) were included in their assessments.

The impact of new spatial closures and industry restructuring on the dugong population is difficult to quantify. Although the GBRWHA has won international recognition as one of the world's most advanced networks of marine protected areas (MPA), accurate information on dugong bycatch in commercial gill-nets is unavailable for several reasons: (1) the large geographic extent of the GBRWHA; (2) the remoteness of the region adjacent to the Cape York Peninsula where most dugongs occur; (3) the nature of the fishery which makes boat-based observer (surveillance) programmes logistically difficult and expensive, largely because of the small size of the fishing vessels; and, (4) the lack of observers who are independent of the fishing industry and appropriately trained (Lewison et al. 2004).

As explained in Chapter 4, uncertainty and incomplete information can be a major constraint to the decision making process (Bacic et al. 2006). Decision-support tools, such as spatial risk assessments in geographical information systems (GIS), can assist in

evaluating the risk to marine mega fauna from bycatch in an uncertain environment. This approach combines spatial data on the distribution of a species and fisheries (Dunning et al. 1995) to identify areas where management intervention is likely to be most effective (Theobald 2003; Andersen et al. 2004). GIS-based spatial risk assessments are particularly valuable in large geographic regions where information is limited as they can incorporate different kinds of quantitative and qualitative spatial data to support the estimation, evaluation and comparison of alternative management interventions.

In this chapter, I used a spatial risk assessment approach to rapidly evaluate the capacity of the new zoning and industry restructuring in the GBRWHA to minimise the risk of dugong bycatch in commercial gill-netting operations based on available data. I also discussed the applicability of this approach to other situations where there are uncertainties about the magnitude of the bycatch of marine mega fauna, including remote regions and developing countries where resources are limited.

Methods

The usual consequence of a dugong being caught in a commercial gill-net is that it drowns (Marsh et al. 2005). Dugongs of all ages and both sexes are caught, and the distributions of sizes, sexes, and estimated ages contains no major gaps (Marsh 1980), suggesting that the likelihood of a dugong drowning in a net is independent of the animal's reproductive value. Thus the likelihood of a dugong being caught in a net should be a robust surrogate of the risk of dugong bycatch. Given that there is no evidence of bycatch selectivity, the probability of a dugong being caught in a net should be a function of dugong density and fishing effort (Marsh 2000). If fishing effort is banned from an area and that ban is enforced, then the likelihood of dugong bycatch in that area should be reduced to zero, irrespective of whether individual animals use the area permanently or temporarily.

Exposure quantification and risk estimation

There are multiple marine legislative boundaries that currently control the distribution of the Queensland East Coast Inshore Fin Fish Fishery, as described in Table 6.1. Together, these arrangements regulate five levels of netting restrictions that are relevant

to dugongs in the GBRWHA. Levels 1 and 2 provide dugongs with an assumed nil risk of bycatch from netting activities as no netting is permitted or the permitted netting practices prevent dugong entanglement.

To derive a single coverage of the *current* spatial extent of bycatch risk, I used the intersect tool in ArcGIS® 9.1 (Environmental Systems Research Institute 2005) to combine GIS layers of multiple marine legislative boundaries that currently control the distribution of netting in the GBRWHA. I overlaid the resultant coverage with the spatial model of dugong conservation value outlined in Chapter 5 to provide an overall spatial matrix of the current likelihood of dugong bycatch. The corresponding likelihood of dugong bycatch under the pre-2004 zoning regime was derived by combining the multiple marine legislative boundaries controlling the distribution of commercial netting in the GBRWHA prior to re-zoning with the spatial model of dugong conservation value. I used the two matrices of the likelihood of dugong bycatch to estimate the proportion of high, medium and low conservation value dugong planning units where: (1) the risk of bycatch is currently assumed to be nil; and (2) the risk of bycatch was assumed to be nil prior to re-zoning.

I integrated commercial catch information into the risk assessment to provide an empirically derived limit on the spatial extent of commercial netting in the GBRWHA. Fisheries Queensland monitors catch of the netting industry through compulsory daily logbooks completed by fishers. The information collected in these logbooks includes: (1) the day's catch (weight and species); (2) locations fished; and, (3) the time spent fishing. This information is then aggregated into grids of resolution of 6 nautical miles. I used information extracted to 6 nautical miles grids for the time periods January – June 2004 (pre re-zoning) and January – June 2005 (post re-zoning) to quantify the change in risk to dugongs from bycatch as a result of netting area closures. These time periods were chosen because: (1) temporal closures occur based on the life history of the target fish species making it important to compare similar times of year; and, (2) industry restructuring by Fisheries Queensland did not affect the fishery until after January 2005.

Table 6.1. The five levels of restrictions on commercial gill-netting relevant to dugongs in the GBRWHA¹. The risk of bycatch of dugongs is low with Level 1 and 2 restrictions.

<i>Level 1 restrictions</i>	<i>Level 2 restrictions</i>	<i>Level 3 restrictions</i>	<i>Level 4 restrictions</i>	<i>Level 5 restrictions</i>
All netting prohibited or bait netting only permitted	Offshore set, foreshore set and drift nets prohibited. River set nets allowed with modification ^a	Offshore set nets, nets that are not fixed or hauled prohibited. Restrictions on set mesh nets on the foreshore	Restrictions on mesh netting	Netting permitted under regulations of the Queensland <i>Fisheries Act</i> 1994 and <i>Fisheries Regulations</i> 1995

¹The marine legislative boundaries that currently control the distribution of commercial gill-netting includes: Dugong Protection Areas (*Queensland Fisheries Regulations* 1995); *Great Barrier Reef Marine Park Zoning Plan* 2003; *Great Barrier Reef Coast Marine Park Zoning Plan* 2004; Princess Charlotte Bay Special Management Area; and the boundaries of ports as designated by the relevant port authorities. Prior to the 2004 re-zoning, netting was controlled by: Dugong Protection Areas; the former *Great Barrier Reef Marine Park Zoning Plan*; pre-2004 relevant State Marine Parks; developmental areas; and port authorities.

^aProhibited in Hinchinbrook and Shoalwater Bay Dugong Protection Areas along the urban coast

For three sites in the January – June 2005 period, catch information was aggregated by Fisheries Queensland to a 30 nautical mile scale because of miss-reporting of catch and effort information by fishers. My results possibly overestimated the spatial extent of netting in this period as an artefact of the large spatial extent of these grids. To eliminate this false positive, I removed sections of the 30 nautical mile grids of the January – June 2005 period that did not overlap with the 6 nautical mile fisheries grids in the January – June 2004 period where fishing was conducted. The associated error should be low as new area closures and industry restructuring reduced the spatial extent of commercial netting, total catch and effort across all other grids between 2004 and 2005 (Table 6.2).

Because of the relatively large spatial scale of 6 nautical mile fisheries catch grids, portions of some grids extend into areas where netting is not permitted. I removed those portions by erasing sections of grids that fell outside the region where netting is permitted before and after the 2004 re-zoning. I calculated the relative effort for each grid under the current and former zoning regime as the ratio of the number of days spent fishing in a grid / area of that grid where netting was permitted. The output coverages were used to estimate the proportion of areas of dugong planning units where the risk of bycatch was assumed to be nil under: (1) the current zoning (January – June 2005) and industry arrangements; and, (2) under former zoning and before industry restructuring (January – June 2004). Table 6.2 summarises and evaluates the assumptions of my analyses.

I converted the final coverages of relative effort to a raster grid of cell size 2 km * 2 km (the same scale as dugong planning units, *see* Chapter 5) to quantify change in effort between the two time periods. Minor spatial errors were introduced by changing the scale of fisheries grids. However, as almost exactly 24 cells of 2 km * 2 km resolution fall inside one 6 nautical mile grid and the output layer was not used for spatial analyses, this minor error should not affect the final result. I assigned each dugong planning unit the value of the fishing effort that it overlayed for: (1) January – June 2004; and, (2) January- June 2005. I exported this information to the statistical program GenStat 8th Edition (Lawes Agricultural Trust 2005). Frequency distributions of dugong conservation value and effort data did not approximate a Gaussian distribution and assumptions of parametric statistical tests were not met. I analysed this information instead with paired Mann-Whitney U tests with $\alpha = 0.05$.

Results

I estimated that under the zoning and management arrangements in operation since January 2005 the risk of dugong bycatch should be nil in approximately 67% (801 km²) of the dugong planning units of high conservation value identified in Chapter 5; a relative improvement of 56% (573 km²) from the previous zoning arrangements (Table 6.3). Currently, there is 'nil' risk of bycatch in all high conservation value planning units along the urban coast and in 36% (801 km²) of the corresponding units in the remote Cape York region (a relative increase of 0% and 68% respectively). A 'nil' risk of bycatch is present in half (1,089 km²) of the planning units of medium conservation value (5% along the urban coast and 49% in the remote Cape York region), a 20% (311 km²) relative increase from the previous zoning and management arrangements.

I found that only about 7% (172 km²) of the high conservation value and 11% (232 km²) of the medium conservation value dugong planning units where the risk of bycatch was nil were within the designated Dugong Protection Areas (DPAs) Zone A (foreshore and offshore set or drift nets are prohibited). DPAs now play a relatively minor role in the overall protection of dugongs in the GBRWHA despite their iconic status because most of the planning units of high conservation value to dugongs are in the remote Cape York region and all the designated DPAs are along the urban coast. Nonetheless, along the urban coast, all the dugong management units of high conservation value and 90% of the units of medium conservation value where current zoning provides an assumed nil risk of bycatch to dugongs were within DPAs Zone A. If the netting restrictions for DPAs Zone B were upgraded to the same level as Zone A, the increase in dugong protection would be minimal.

Table 6.2: Evaluation of the assumptions underpinning the analyses.

<i>Assumption</i>	<i>Justification</i>	<i>Risk of false assumption</i>
The spatially explicit model of dugong distribution and relative abundance outlined in Chapter 5 accounts for temporal changes in dugong habitat use since 1985.	By using a time series of data collected over 19 years, the spatially explicit model accounts for temporal changes in the use of various regions by dugongs including movements resulting from events such as seagrass dieback.	Low
Since 1985, there has been no decline in the extent of dugong occurrence, although there may be decline in area of occupancy.	There is no evidence that the anthropogenic activities that have caused dugong decline have removed populations at the spatial scale of the management arrangements. This assumption should not increase the uncertainty in my risk assessment for two reasons: (1) the present rather than the past risk of dugong bycatch is assessed; and (2) the spatial scale of dugong management in the GBRWHA is far broader than any reduction in the area used by dugongs within their range in the GBRWHA.	Low
The risk to dugongs is nil in zones where: (1) all netting is prohibited or bait netting only permitted; (2) offshore set, foreshore set and drift nets prohibited; and (3) river set nets allowed with modification.	The listed restrictions remove the netting practices that cause dugong entanglement in gill and mesh nets, and are effectively enforced along the urban coast.	Low
Netting was not conducted in sections of the 30 nautical mile grids of the January – June 2005 period that did not overlap with commercial netting activities conducted in the 6 nautical mile fisheries grids of the January – June 2004 period.	New area closures and industry restructuring decreased the extent of permitted commercial netting activities, total catch, and effort.	Low
The only influences on the distribution of catch between January – June 2004 and January – June 2005 are the area closures associated with re-zoning and industry restructuring. No external factor including environmental conditions affected the catch available to fishers. Anthropogenic factors, including the cost of fuel and the condition of vessels did not influence the distribution of commercial catch and effort.	There is little documented evidence that the catch available to fishers was affected by external factors during January-June 2004 and 2005. Information that is available cannot be quantified spatially across the entire GBRWHA.	Medium
No illegal netting occurred.	Information on the distribution of illegal net fisheries is unavailable at an appropriate scale.	Medium

When the spatial effort data were considered, the reduction in bycatch risk to dugongs from the new zoning and management arrangements in the GBRWHA is even more significant than suggested by the area closures alone. Between January – June 2005, commercial gill-netting did not occur in approximately 96% (2,303 km²) of the dugong planning units of high conservation value (Table 6.3), a relative increase of 22% (408 km²) from January – June 2004. Along the urban coast, no units of high conservation value were commercially netted between January – June 2005; the corresponding value in the remote Cape York region was 96% (2,136 km²). Between January – June 2005, netting was not conducted in 91% (1,979 km²) of the units of medium conservation value (95% on the urban coast, and 91% in the remote Cape York region); a 6% (109 km²) relative decrease from the corresponding period in 2004. The mean reduction in netting effort per dugong management unit was statistically significant in the following areas: (1) the entire GBRWHA; (2) urban coast; and, (3) remote Cape York region for dugong management units of high, medium and low conservation value (Table 6.4).

I identified four anomalous locations where commercial gill-netting is still permitted in dugong planning units of high and/or medium conservation value and where netting was conducted between January – June 2005. They included the regions surrounding: Bathurst Head, Friendly Point and Lookout Point in the remote Cape York region (Figure 6.1); and the regions between Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay, and Shoalwater Bay and Port Clinton along the urban coast (Figure 6.2).

Table 6.4: Results of the paired Mann-Whitney U tests comparing commercial netting effort for each planning unit in the January – June periods of 2004 and 2005 within areas of high, medium and low conservation value in the GBRWHA, urban coast and remote Cape York regions.

		<i>Dugong conservation value</i>		
		High	Medium	Low
GBRWHA	<i>n</i> ^a	100	68	1975
	<i>P</i> ^b	<0.001 ^c	0.001 ^c	<0.001 ^c
Urban coast	<i>n</i> ^a	-	3	1531
	<i>P</i> ^b	-	-	<0.001 ^c
Remote Cape York	<i>n</i> ^a	100	65	444
	<i>P</i> ^b	<0.001 ^c	0.001 ^c	<0.001 ^c

^aNumber of dugong management units for each test.

^bSignificance level observed from a paired Mann-Whitney U test.

^cHighest rank score observed in the January-June 2004 period.

Table 6.3: Percentage of dugong planning units of high, medium and low conservation value in: (1) the entire GBRWHA; (2) urban coast; and, (3) remote Cape York regions where: (1) the risk of dugong bycatch was assumed to be nil because commercial gill-netting is banned; and, (2) no gill-netting actually occurred during: (a) January-June 2005 (post re-zoning and industry restructuring) and (b) January-June 2004 (pre re-zoning and industry restructuring). The total area considered here is the known area used by dugongs in the GBRWHA as outlined in Chapter 5.

	<i>Dugong conservation value</i>	<i>GBRWHA</i>			<i>Urban coast</i>			<i>Remote Cape York</i>		
		<i>Area (km²)</i>	<i>Jan – Jun 2004</i>	<i>Jan – Jun 2005</i>	<i>Area (km²)</i>	<i>Jan – Jun 2004</i>	<i>Jan – Jun 2005</i>	<i>Area (km²)</i>	<i>Jan – Jun 2004</i>	<i>Jan – Jun 2005</i>
<i>Assumed nil risk of bycatch^a</i>	High	2399	43	67	173	100	100	2226	38	64
	Medium	2175	38	50	259	85	95	1916	30	44
	Low	27490	23	34	11771	22	38	15719	25	31
<i>No risk of bycatch^b</i>	High	2399	79	96	173	100	100	2226	78	96
	Medium	2175	86	91	259	96	95	1916	85	91
	Low	27490	72	80	11771	47	62	15719	89	93

^aRisk of bycatch is assumed nil as no netting is permitted or permitted netting practices do not cause dugong entanglement in gill and mesh nets.

^bThere was no risk of dugong bycatch as netting was not conducted.

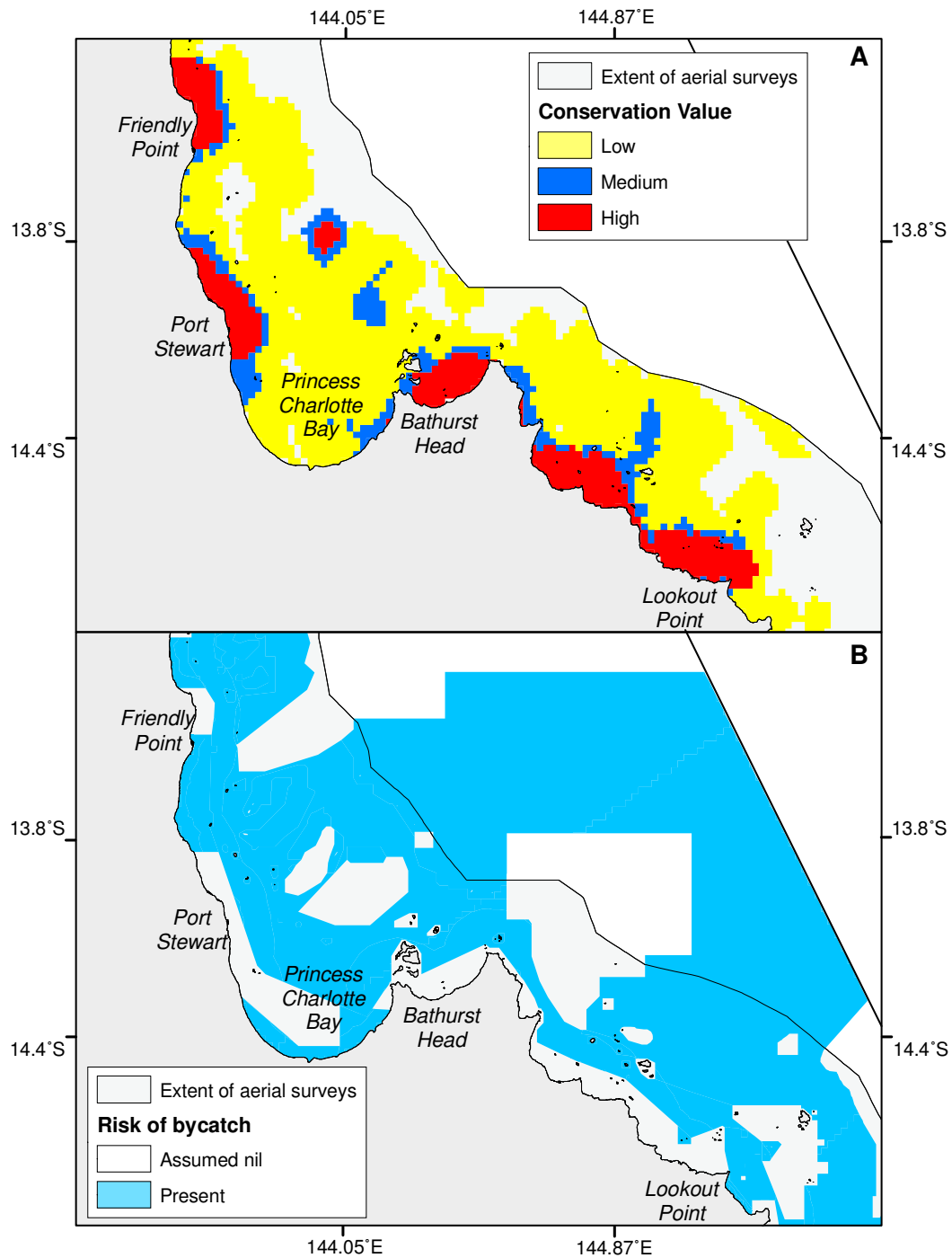


Figure 6.1: (A) Model of dugong planning units with high, medium and low dugong conservation value; and, (B) the current risk of bycatch from commercial netting derived from the five levels of netting restrictions relevant to dugongs between Lookout Point and Friendly Point in the remote Cape York region.

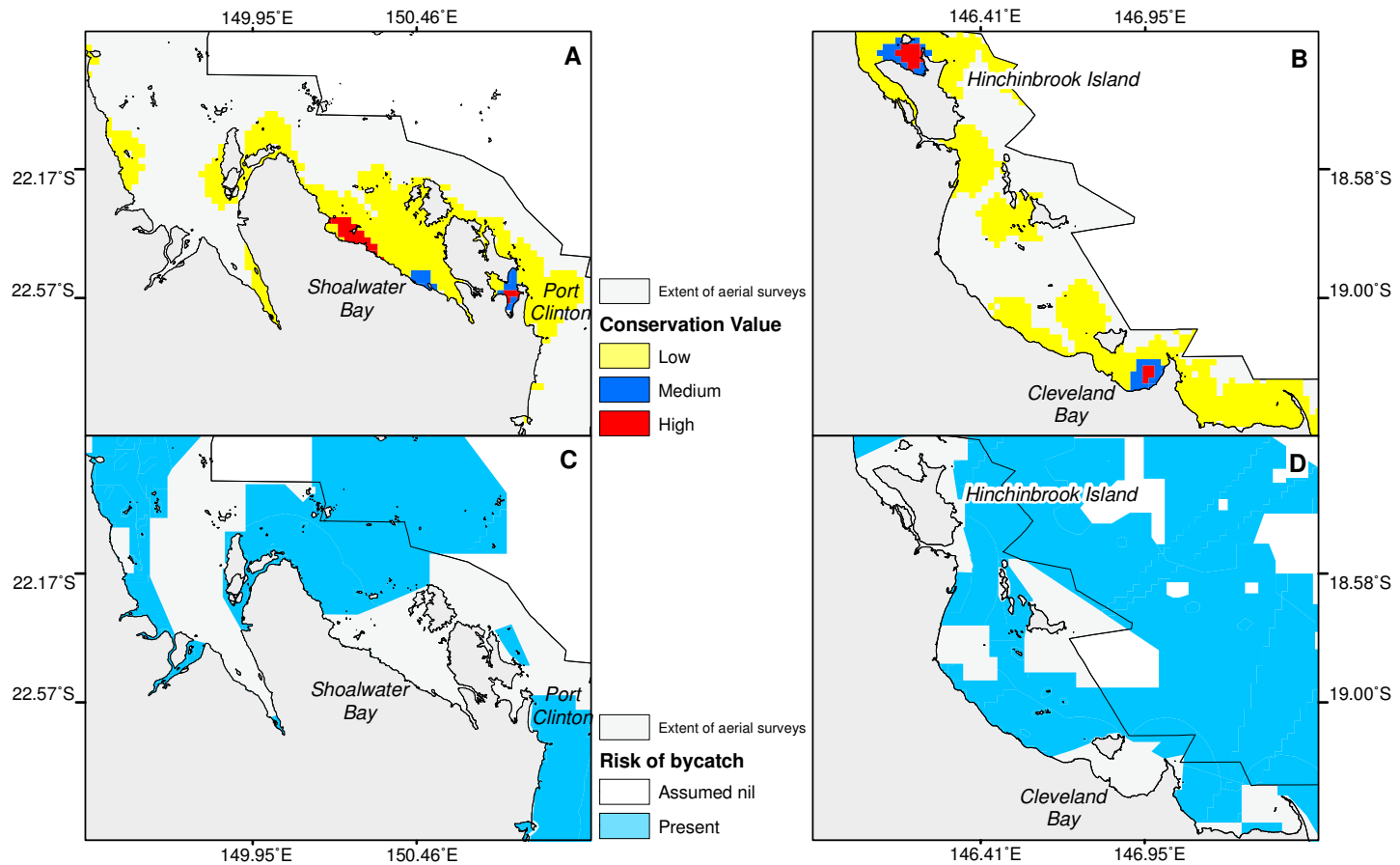


Figure 6.2: (A, B) Models of dugong planning units with high, medium and low dugong conservation value; and, (C, D) the current risk of bycatch from commercial gill-netting derived from the five levels of netting restrictions relevant to dugongs along the urban coast. A and C represent the Shoalwater Bay region, and B and D the region between Cleveland Bay and Hinchinbrook Island. Dugongs are not limited to the regions shown in A and B, but are also distributed at low density along the entire urban coast, in areas defined in Chapter 5 as low dugong conservation value areas.

Discussion

The re-zoning of the GBRWHA in 2004 was designed to improve biodiversity protection through a comprehensive, adequate and representative multiple-use zoning regime (Day et al., 2000). I found that the new zoning in the GBRWHA significantly reduced the risk of dugong mortality in commercial gill and mesh nets by reducing the area in which commercial gill-netting activities were permitted in dugong habitats of high, medium and low conservation value. The restructuring of the fishing industry further reduced the spatial distribution of gill-netting and overall fishing effort.

Accuracy of estimates

Catch information from commercial fisheries in Queensland is aggregated to 30 and 6 nautical mile grids, and the paths of individual boats are not recorded. It is likely that I have overestimated the distribution of netting activities by using 30 and 6 nautical mile grid data in my risk assessment. Consequently, my estimates of the proportion of dugong planning units where netting is conducted underestimates the protection afforded to dugongs in the GBRWHA. Finer scale information on the distribution of netting activities would reduce this error.

It is politically impossible to protect a species by restricting activities for the coastline of an entire region as large as the GBRWHA (~ 2,300 km). However, Roberts et al. (2001) stress the need to effectively manage those areas where the species are most vulnerable. I assumed that these areas are the sites of high dugong density, typically seagrass meadows e.g. Shoalwater Bay, Cleveland Bay and north of Hinchinbrook Island (Figure 6.2). Dugongs undertake macro (> 100 km) and meso-scale (15 - 100 km) movements between such meadows (Gales et al. 2004; Marsh et al. 2004 and 2005; Sheppard et al. 2006). Timed depth recorders show that dugongs track the bottom when undertaking macro-scale movements between meadows leaving them vulnerable to bottom set gill-nets (Sheppard et al. 2006). Data from some 70 satellite-tracked dugongs suggest that their movements are individualistic and not restricted to defined movement corridors (Sheppard et al. 2006). If movement corridors do exist, and those corridors are in areas where commercial gill-netting is permitted and conducted, my analysis would overestimate the protection afforded to dugongs by area closures and industry restructuring by classifying potential movement corridors as low conservation value to

dugongs. Fisheries Queensland attempted to address this problem in the 2007 review of the East Coast Inshore Fin Fish Fishery by requiring commercial fishers to be within 100 m of a net when they are in use (except in those circumstances where such a distance is considered impracticable). Stronger net attendance rules in areas where dugongs are moving between seagrass meadows reduces the consequence of dugong entanglement in a net by increasing the likelihood that it would be released alive.

I assumed that no illegal netting was conducted within the GBRWHA. Gribble and Robertson (1998) observed vessels of the Queensland east coast prawn trawl fleet consistently trawling in areas where trawling is not permitted in the remote Cape York region. Illegal trawl fishing is less likely to occur in populated inshore areas of the urban coast (Davis et al. 2004; Williamson et al. 2004), and is becoming less of an issue with the introduction in recent years of vessel monitoring systems. Lack of compliance in the remote Cape York region is a consequence of limited surveillance and enforcement activities in this region. The inability to quantify illegal netting means that I probably overestimated the protection afforded to dugongs by area closures in the GBRWHA by an unknown amount, especially in the remote Cape York region where dugongs are most abundant. Increasing surveillance and enforcement activities in the remote Cape York region will require a substantial monetary investment by the Queensland Government. An alternative approach that the Queensland and Australian Governments are currently considering is the introduction of more spatial closures in the region.

The overall effect of these sources of inaccuracy could be in either direction and is unknown.

Implications for dugongs in the GBRWHA

Although my analysis indicates that area closures and industry restructuring have reduced the proportion of the dugong's range at risk from bycatch in commercial gill-nets, further evaluation of the likely effectiveness of these management interventions will require accurate data on the actual levels of incidental mortality from netting in various areas of the GBRWHA. On the basis of Potential Biological Removal modelling (Wade 1998), Marsh et al. (2005) estimated that management should aim for an

anthropogenic mortality target of zero to maximise the likelihood of dugong populations recovering on the urban coast of the GBRWHA. Future dugong management strategies in the GBRWHA should consider the potential effects of continued commercial gill-netting along the urban coast, especially in the regions between Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay, and Shoalwater Bay and Port Clinton where netting activities are conducted in potential dugong corridors between high conservation value dugong habitats (Figure 6.2). Fisheries Queensland attempted to address this problem in the 2007 review of the East Coast Inshore Fin Fish Fishery by introducing a 500 m exclusion zone around headlands adjacent to DPAs. Further protection would be provided to dugongs by increasing attendance at net rules to reduce the consequence of dugong entanglement in a net by increasing the likelihood that it would be released alive. In addition, management authorities also need to consider strategies to minimise other sources of anthropogenic mortality in this region such as vessel strike and poor quality terrestrial runoff (Marsh et al. 2005).

Heinsohn et al. (2004) conducted a Population Viability Analysis of the remote Cape York region's dugong population where the Indigenous harvest of dugongs is the major source of mortality and concluded that this mortality was not sustainable. Indigenous hunting is a Native Title right (*see* Chapter 2 page 22) making it difficult for the regulatory agencies to limit catches; a situation that increases concern about gill-netting bycatch. I identified three locations in the remote Cape York region where netting is still permitted and conducted in dugong management units of high or medium conservation value as: the regions surrounding (1) Bathurst Head; (2) Friendly Point; and, (3) Lookout Point (Figure 6.1). Increased protection or modified fishing practices are being considered for these regions.

Advantage of embedding area closures in a network of MPAs

My analysis indicates that the dedicated dugong MPAs (DPAs) did not provide protection comparable to that of the ecosystem-scale network of MPAs implemented in the GBRWHA in 2004. Only 7% of the high conservation value dugong management units and 12% of the medium conservation value units that have a low risk of bycatch are within designated DPAs. Furthermore, increasing netting restrictions in DPAs Zoned B will have a minimal affect on dugong protection in the region. This

demonstrates the potential power of the over-arching design of an ecosystem-scale MPA to protect a mobile marine mammal relative to independent small areas dedicated to protecting a single species.

The multiple-use zoning regime in the GBRWHA was designed to increase the likelihood of sustainable fisheries and the preservation of cultural values in addition to protecting biodiversity features. It is impossible to restrict extractive activities throughout the entire GBRWHA. Nonetheless, by protecting sites of high conservation value, re-zoning decreased the risk to dugongs from bycatch in commercial fisheries. It would have been difficult to expand the DPAs and restructure the Queensland East Coast Inshore Fin Fish Fishery independently of the overall re-zoning of the entire GBRWHA. Instead, dugong protection benefited from the political will to protect the entire GBRWHA ecosystem because of its iconic value both to Australians and the rest of the world.

A further benefit to the dugong population from the re-zoning of the GBRWHA was the reduction in total area where commercial trawling is permitted (Coles et al. 2007). Dedicated dugong MPAs (DPAs) do not provide an equivalent degree of protection to dugongs because DPAs only control the distribution of commercial gill-netting and the type of gear within their boundaries. They are not 'no-take' zones as provided under the current zoning regime.

Generic implications for programmes to reduce marine mammal bycatch

The challenge of addressing the problem of bycatch of marine mammals in fisheries is complicated by the many uncertainties in existing data on marine mammal bycatch and natural systems. GIS-based decision support systems such as spatial risk assessments are a potentially important addition to the tools listed by Lewison et al. (2004) for monitoring and evaluating interactions between marine mammals and fisheries.

GIS-based decision support systems and spatial risk assessments are particularly valuable in large and remote coastal regions and in the coastal waters of developing countries where there is scant scientific information on the level of bycatch and other threats to the species of concern. GIS-based decision support systems can collate a

variety of information, collected by both quantitative and qualitative methods, at a scale relevant to communities and managers. In large and remote coastal regions and in the coastal waters of developing countries, GIS-based spatial risk assessments can provide a simple, clear way of creating, communicating and analysing data from a variety of sources. Community-based GIS mapping programmes (such as Public Participatory GIS programmes involving key informants) can provide valuable qualitative information on the distributions of species and artisanal fishing effort, which can be used in association with quantitative information collected by scientists and managers in GIS-based decision support systems. A benefit of using information derived from such programmes is that it provides remote communities and communities in developing countries with a means to participate actively in the decision making process by contributing to the data on which decisions are made, increasing the likelihood that appropriate responses to marine mammal bycatch problems will be developed (Craig et al. 2002).

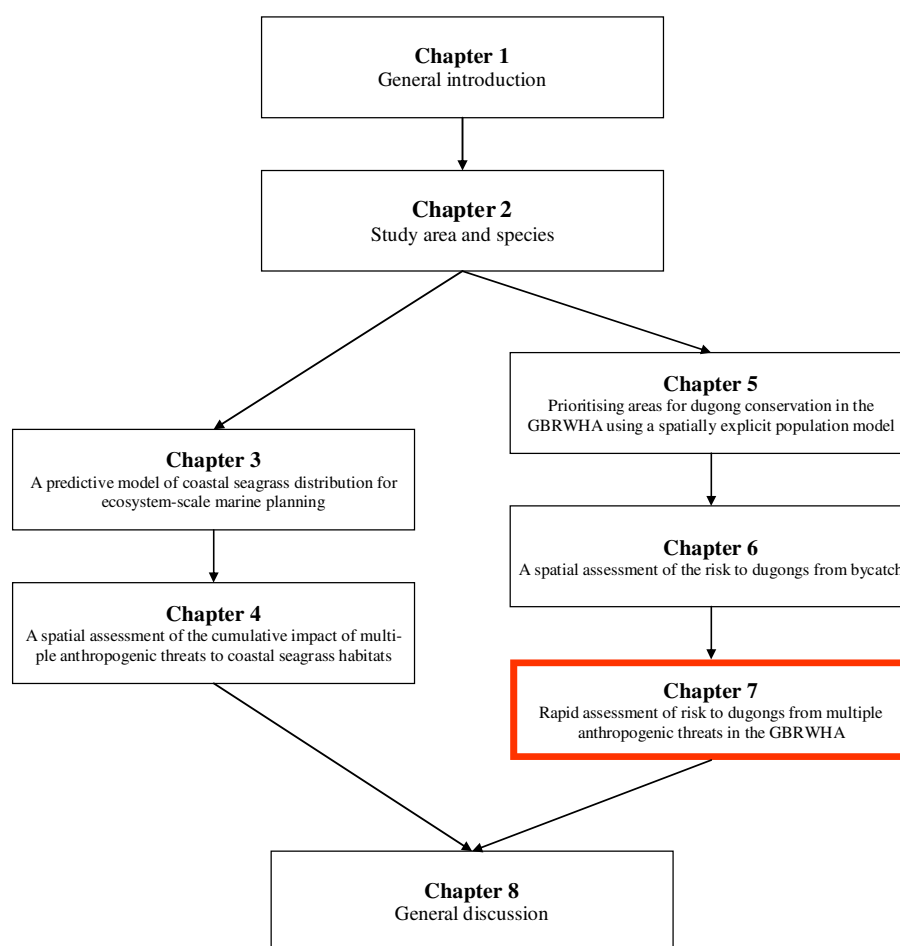
Chapter Summary

- Re-zoning of the GBRWHA in 2004 closed 33% of the region to extractive activities, including commercial gill-netting. The impact of re-zoning and the associated industry restructuring on dugongs is difficult to quantify. In the face of this uncertainty, I used a spatial risk assessment approach to evaluate the re-zoning and associated industry restructuring for their ability to reduce the risk of dugong bycatch in commercial gill-nets.
- I found that the new zoning arrangements appreciably reduced the risk of dugong bycatch by reducing the total area where commercial gill-netting is permitted. Netting is currently not permitted in 67% of dugong habitats of high conservation value, a 56% improvement over the former arrangements. Re-zoning and industry restructuring also contributed to a 22% decline in the spatial extent of conducted netting.
- I identified five ‘hot spots’ for conservation actions where commercial gill-netting is still permitted in dugong planning units of high and/or medium conservation value and where netting was conducted between January – June 2005. They included the regions surrounding: Bathurst Head, Friendly Point and Lookout Point in the remote Cape York region; and between Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay, and Shoalwater Bay and Port Clinton along the urban coast.
- A spatial risk assessment approach that evaluates the risk of mobile marine mammals from bycatch are applicable to other situations where there is limited information on the location and intensity of bycatch, including remote regions and developing countries where resources are limited.

Chapter 7

Rapid assessment of risk to dugongs from multiple anthropogenic threats in the GBRWHA¹

In this chapter, I use expert opinion and a spatial risk assessment approach to rapidly assess the risk to dugongs from multiple anthropogenic threats in the GBRWHA. Outputs of the assessment are the identification of anthropogenic hazards with the greatest relative impact on dugongs at the scale of the entire GBRWHA, and ‘hot spots’ that are a priority for conservation action because the risk to dugongs from multiple anthropogenic threats is high.



¹ Grech, A. and Marsh, H. 2008. Rapid assessment of risks to a mobile marine mammal in an ecosystem-scale marine protected area. *Conservation Biology* **22**(3): 711 – 720.

Introduction

As explained in Chapter 1, ecosystem-scale networks of marine protected areas (MPAs) are increasingly favoured over small, isolated reserves, as demonstrated by the recently declared Northwestern Hawaiian Island Marine National Monument and the proposed global network of high-seas marine reserves (Roberts et al. 2006). Such initiatives are potentially effective tools for conserving marine mammals (Hoyt 2005). However, the capacity of a specific MPA network to protect marine mammals is difficult to quantify in a timeframe appropriate to species conservation because the uncertainties associated with evaluating their effectiveness is generally high (*see* Chapter 6 page 103). It is impossible to detect even large changes in most populations of marine mammals with current levels of investment in surveys, survey technology, and survey design (Taylor et al. 2007). Evaluating the effectiveness of ecosystem approaches such as networks of MPAs is made even more difficult because data are generally lacking, and the way in which different components of the ecosystem are linked is poorly understood. At the broad spatial scale of networks of MPAs, information is inevitably scarce as a result of multiple factors including time, expertise, and cost constraints (Galloway et al. 2002).

Globally significant populations of the dugong (*Dugong dugon*) inhabit the coastal waters of the Great Barrier Reef World Heritage Area (GBRWHA) of Queensland, Australia. Dugongs have a high biodiversity value as the only herbivorous mammal that is strictly marine, very high cultural and nutritional value to Indigenous Australians, and are regarded as a flagship species by non-Indigenous Australians. All these factors make dugong management a high priority for managers of the GBRWHA. Dugongs are threatened by multiple anthropogenic activities, and regulatory decisions to manage such activities are typically made with incomplete scientific information (e.g. Fernandes et al. 2005). However, a good management decision should not require large numbers of precise estimates to trigger warranted management actions (Taylor et al. 2007). For example, the Queensland and Australian Governments are presently considering the introduction of additional commercial gill-netting spatial closures in important dugong habitats in the remote Cape York region of the GBRWHA without accurate information on the location and intensity of dugong bycatch (*see* Chapter 6 page 117). The consideration by the Queensland and Australian Governments to introduce additional spatial closures in the region was prompted by the outcomes of the spatial assessment of

risk to dugongs from bycatch outlined in Chapter 6. The approach outlined in Chapter 6 was able to locate sites for additional spatial closures in the data-inadequate remote Cape York region by quantifying the overlap between dugong habitats (*see* Chapter 5) and permitted and conducted commercial netting activities.

As explained in Chapter 1, assessing only those anthropogenic activities that are within the regulatory control of the GBRWHA's ecosystem-scale network of MPAs is insufficient to inform the planning and management of dugongs and their seagrass habitats. In addition to commercial and recreational fishing activities, dugongs are directly threatened by Indigenous hunting and vessel strike (Marsh et al. 2002; Heinsohn et al. 2004); seagrass habitats are threatened by several anthropogenic activities (*see* Chapter 4 page 59). Multiple Queensland and Australian government agencies manage the various anthropogenic activities that are known or thought to have impacts on dugongs and their seagrass habitats in the GBRWHA across jurisdictions (*see* Chapter 2 page 23), making it a challenge to quantify the extent to which the species is actually protected. Although the GBRWHA is one of the world's most well-studied and managed marine ecosystems, knowledge of the distribution and relative impact of the various threats is inadequate. For example, the current level of dugong mortality caused by any one of the major causes of human-induced mortality is not known (Marsh et al. 2002). This makes it difficult to quantify the impact of all the anthropogenic threats to the GBRWHA's dugong population.

In the face of this uncertainty, I applied the same approach used to assess the risk of coastal seagrasses from multiple threats in Chapter 4 and dugongs to bycatch in Chapter 6, to: (1) rapidly assess the level of risk to dugongs from multiple anthropogenic threats under the current zoning and management arrangements in the GBRWHA; and, (2) evaluate options to ameliorate that risk. My approach can be generalised to situations in which ecosystem-scale MPA networks are used to protect mobile marine mammals.

Methods

I used the same approach outlined in Chapter 4 to estimate the risk to dugongs from multiple anthropogenic threats in the GBRWHA: (1) identify the hazards; (2) quantify the exposure of dugongs to these hazards; and, (3) estimate the risk to dugongs (Sutur 1993).

Hazard identification

I identified the anthropogenic hazards to the dugong population in the GBRWHA using a Delphi technique (Veal 1992) at a meeting of experts in the region's commercial fisheries, Indigenous issues, species conservation, and dugong ecology and management. The experts identified the hazards as: (1) incidental mortality in gill and mesh nets (Marsh 2000) and in prawn (shrimp) trawlers (R. Coles, personal communication); (2) seagrass habitat loss associated with poor water quality terrestrial runoff from agricultural activities (McKenzie et al. 2000); (3) prawn trawling which can damage bottom habitats (Marsh et al. 2002); (4) vessel strikes (Hodgson 2004; Greenland and Limpus 2006); (5) displacement of dugongs from key habitats as a result of vessel traffic (Hodgson and Marsh 2007); and, (6) poaching and legal Indigenous hunting (Heinsohn et al. 2004). The dugong experts did not identify several of the hazards to dugong's seagrass habitats that were assessed in Chapter 4, including: urban and industrial runoff, coastal development, dredging and shipping accidents.

Exposure quantification

I quantified the spatial distribution of commercial gill-netting, trawling, and Indigenous hunting as outlined below. Descriptions of the layers that delineated vessel traffic (recreational and commercial) and the risk of poor water quality terrestrial runoff from agricultural activities are provided in Chapter 4, pages 60 - 63. Figures showing the spatial distribution of the five hazards are provided in Appendixes C and E.

Five levels of commercial gill-netting restrictions, ranging from no netting permitted to gear restrictions are relevant to dugongs in the GBRWHA (*see* Chapter 6 page 106; Tables 6.1 and 7.1). Trawling is delineated by the same marine political boundaries as netting, with the Queensland *East Coast Trawl Management Plan* 2000 providing additional spatial and temporal restrictions. Trawling is either prohibited or allowed

within zones of the Great Barrier Reef Marine Park. I assumed that the spatial extent of commercial netting and trawling was limited only by these regulations (i.e. I did not quantify the spatial extent of ‘actual’ netting or trawling activities, as outlined in Chapter 4 page 65 and Chapter 6 page 109). I did not include the ‘actual’ distribution of netting and trawling activities in this study because I am assessing the effectiveness of the current zoning arrangements in mitigating the hazards to dugongs (i.e. I did not attempt to quantify potential dugong mortality as outlined in Chapter 6).

I obtained spatial information on the distribution of Indigenous hunting of dugongs from the Great Barrier Reef Marine Park Authority. In 2004, the Great Barrier Reef Marine Park Authority collected information on dugong hunting through interviews conducted by marine parks staff (T. Stokes, personal communication). I used the qualitative (presence-absence) data to model the spatial extent of Indigenous hunting in various regions. On advice from marine parks staff, I assumed Indigenous hunting was not present more than approximately 5.4 km (3 nautical mile) from the coast or in waters > 5 m deep (C. Turner and T. Stokes, personal communication).

I intersected GIS layers of the spatial distribution of gill-netting, trawling, vessel traffic, poor quality terrestrial runoff, and Indigenous hunting to form a composite hazard coverage.

Risk estimation

Information on dugong mortality, trauma or stress from the five hazards across various areas of the GBRWHA was only available from the Queensland Marine Wildlife Stranding and Mortality Database (Greenland and Limpus 2006). This database grossly underestimates dugong mortality especially in the remote Cape York region because an unknown percentage of carcasses is not recovered or made available for necropsy, and Indigenous catches are not reported. In the light of this uncertainty and the other limitations in the data, I used a Delphi technique (Veal 1992) at a meeting of experts to rank the relative impact of the hazards to dugongs. The experts provided scores from 0 – 100 on the relative impact to dugongs and their seagrass habitats from each of the five anthropogenic hazards, and the relative importance of the components of each hazard (Table 7.1). The experts agreed that Indigenous hunting had the greatest relative impact

on the GBRWHA dugong population, followed in decreasing order of effect by commercial gill-netting, vessel traffic, terrestrial runoff, and trawling. The experts also agreed that where one of the factors was not present, or posed only a low impact to dugongs, a rating of 0 was appropriate for that planning unit.

Table 7.1: Individual and composite ratings of the relative impact of anthropogenic hazards to dugongs and their habitats developed by experts using a Delphi technique.

<i>Hazard</i>	<i>Composite rating of hazard relative to other impacts^a</i>	<i>Impact level within hazard</i>	<i>Rating of impact level within hazard^b</i>	<i>Composite rating of impact level^c</i>
Indigenous hunting	40	Not present	0	0
		Present	100	40
Gill-netting	32	NP/HR ^d	0	0
		Level 1 restrictions	20	6.4
		Level 2 restrictions	50	16
		Level 3 restrictions	80	25.6
		Limited restrictions	100	32
Trawling	5	Prohibited	0	0
		Not prohibited	100	5
Vessel traffic	11.5	Low	0	0
		Medium	40	4.6
		Medium - high	70	8.1
		High	100	11.5
Terrestrial runoff	11.5	Low	0	0
		Medium	40	4.6
		Medium - high	70	8.1
		High	100	11.5

^aDerived by ranking the relative importance of the effects of anthropogenic hazards out of 100.

^bDerived by ranking the relative importance of the individual impact levels of anthropogenic hazards out of 100.

^cDerived by calculating the relative importance of impact levels of anthropogenic hazards on the basis of the composite rating of activities relative to other impacts.

^dNot permitted or highly restricted.

I imported the ratings for each impact level within each hazard to the composite coverage of hazards generated above. I overlaid the composite hazard coverage with the spatial model of dugong conservation value outlined in Chapter 5 to provide an overall matrix of dugong protection. I conducted a sensitivity analysis of the different levels of the composite hazard coverage to define a cut-off score at which the cumulative impact of the five hazards could be considered to be of low risk to dugongs (Store and Kangas 2001). The sensitivity analysis was conducted by systematically varying the cut-off score to determine how this variation affected the estimated risk to dugongs. I found that using scores equal to or below 40 made a trivial difference to the overall results, and scores above 40 were swamped by the presence of Indigenous hunting. I assumed that dugong planning units have a high risk from anthropogenic activities when two or more hazards were present; and a low risk when one or no activity was present. Hazards were weighted on their relative impact on dugongs, and Indigenous hunting received the greatest relative impact with a score of 40, thus I used an overall impact index of $0 \leq 40$ to represent a low level of risk to dugongs and an index of > 40 to represent a high level of risk. Dugongs inhabit the coastal waters of the GBRWHA; therefore I have overestimated the actual protection afforded to the species because I modelled the composite hazard coverage over the extent of the entire GBRWHA rather than limiting my analysis to dugong's area of occupancy. Finally, I assessed the sensitivity of the matrix of dugong protection to changes in the presence and distribution of each hazard by sequentially removing the hazards from the composite hazard coverage and reanalysing the matrix.

Results

I found that under the current GBRWHA zoning and management arrangements, 96% of dugong planning units of high conservation value are at low risk from anthropogenic activities (100% on the urban coast and 96% in the remote Cape York region; Table 7.2). Ninety-three percent of units with medium conservation value are at low risk (95% along the urban coast and 93% in the remote Cape York region), as are 72% of units with low conservation value (42% along the urban coast and 94% in the remote Cape York region).

By sequentially removing threats from the composite hazard coverage, I assessed the sensitivity of the matrix of dugong protection to changes in the presence and distribution of each hazard (Table 7.3). All planning units with a high conservation value to dugongs along the urban coast were at low risk from anthropogenic impacts, so removing additional hazards had no effect on those units. Removing the risk of terrestrial runoff and vessel strike or commercial netting produced a low risk from anthropogenic activities for approximately all dugong planning units of medium conservation value, and nearly all units of low conservation value. Removing Indigenous hunting and trawling provided a negligible increase in the proportion of units of medium and low conservation value designated as low risk along the urban coast.

When I removed either Indigenous hunting or commercial gill-netting in dugong planning units of high, medium and low conservation value in the remote Cape York region, approximately 100% of the region was at low risk from the cumulative impact of multiple anthropogenic activities (Table 7.3). Removing the impacts of terrestrial runoff, vessel traffic, and commercial trawling in the remote Cape York region provided no increase in the proportion of planning units of high conservation value designated as low risk. Dugong planning units with medium conservation value exhibited a similar pattern.

I identified a ‘hot spot’ for conservation action between Port Stewart and Friendly Point in the remote Cape York region (Figure 7.1) where planning units had a high and/or medium conservation value and the current level of risk to dugongs was high. Along the urban coast, I identified planning units where dugongs moving between high and medium conservation value habitats were at risk from multiple anthropogenic threats as: between Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay; and Shoalwater Bay and Port Clinton (Figure 7.2).

Table 7.2: Area (km²) and percentage of dugong planning units with high, medium, and low conservation value in the entire GBRWHA and the urban coast and remote Cape York regions where the risk of all anthropogenic hazards^a is low under the current zoning and management arrangements.

<i>Conservation value^b</i>	<i>GBRWHA (%)</i>	<i>Urban coast (%)</i>	<i>Remote Cape York (%)</i>
High	2,303 (96)	173 (100)	2,137 (96)
Medium	2,088 (93)	246 (95)	1,782 (93)
Low	19,793 (72)	4,944 (42)	14,776 (94)

^aDefined as weighted composite impact index ≤ 40 .

^bAs defined in Chapter 5.

Table 7.3: Percentage of dugong planning units of high, medium, and low conservation value in the entire GBRWHA and the urban coast and remote Cape York regions with a low risk from anthropogenic activities under various hypothetical scenarios.

<i>Activities</i>	<i>GBRWHA</i>			<i>Urban coast</i>			<i>Remote Cape York</i>		
	<i>high</i>	<i>med</i>	<i>low</i>	<i>high</i>	<i>med</i>	<i>low</i>	<i>high</i>	<i>med</i>	<i>low</i>
Current zoning and management arrangements	96	93	72	100	95	42	96	93	94
No gill-netting	100	100	99	100	100	97	100	100	100
No trawling	96	93	75	100	95	45	96	93	97
No netting and trawling	100	100	99	100	100	97	100	100	100
No risk from terrestrial runoff	96	93	79	100	97	54	96	93	97
No vessel traffic	96	93	77	100	96	53	96	93	94
No risk from terrestrial runoff and vessel traffic	96	93	98	100	100	98	96	93	97
No Indigenous hunting	100	99	74	100	95	44	100	100	97

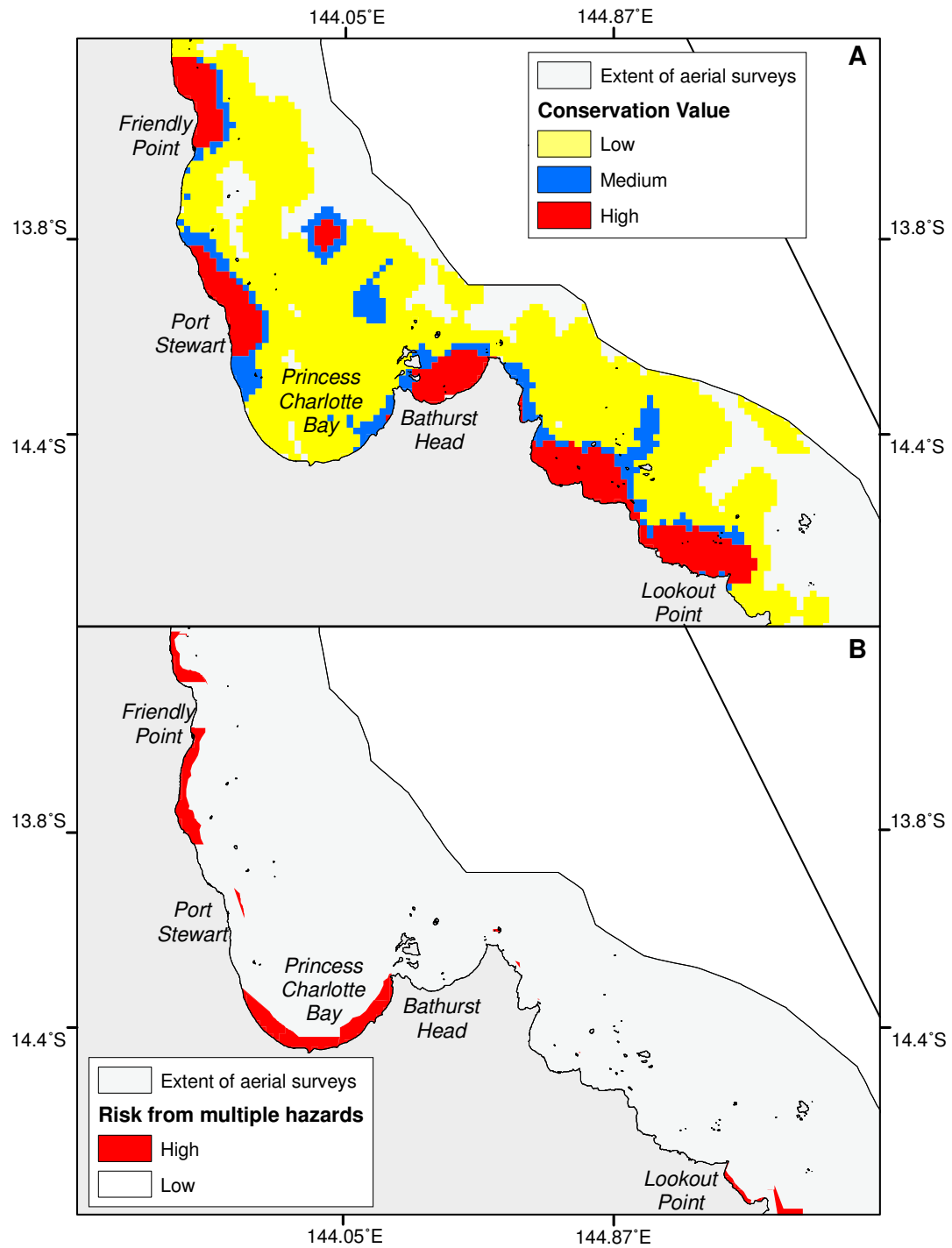


Figure 7.1: (A) Model of dugong planning units with high, medium and low dugong conservation value; and, (B) regions of high and low risk from all anthropogenic activities between Lookout Point and Friendly Point in the remote Cape York region.

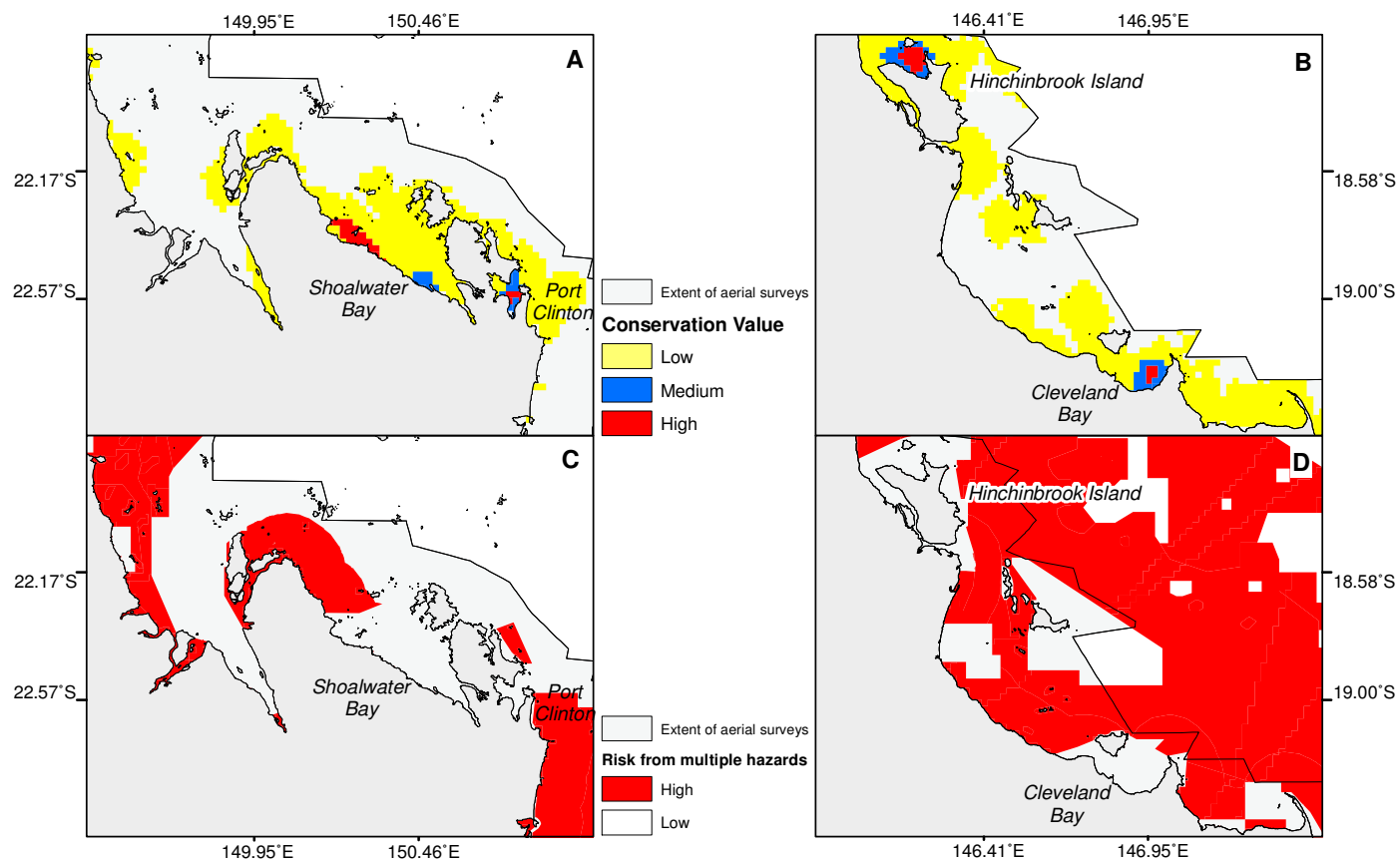


Figure 7.2: (A, B) Models of dugong planning units with high, medium and low dugong conservation value; and, (C, D) regions of high and low risk from all anthropogenic activities along the urban coast. A and C represent the Shoalwater Bay region, and B and D the region between Cleveland Bay and Hinchinbrook Island. Dugongs are not limited to the regions shown in A and B, but are also distributed at low density along the urban coast, in areas defined in Chapter 5 as low dugong conservation value areas.

Discussion

In this chapter, I found that the present ecosystem-scale network of MPAs and associated management arrangements resulted in dugongs being categorised as at low risk from multiple anthropogenic hazards for approximately 96% of planning units with high conservation value and 93% of planning units with medium conservation value along the ~2,300 km GBRWHA coastline. Outputs of the assessment were the identification of anthropogenic hazards with the greatest relative impact on dugongs at the scale of the entire GBRWHA, and ‘hot spots’ that are a priority for conservation action because the risk to dugongs from multiple anthropogenic threats is high.

By testing the sensitivity of the matrix of dugong protection, I identified those threats that if removed, would provide the greatest additional benefit to dugong conservation in the region, assuming the present patterns of use by people and dugongs (Table 7.3). In the remote Cape York region, I found that banning commercial gill-netting or Indigenous hunting would result in virtually all dugong planning units of high and medium conservation value being designated low risk (Table 7.3). Along the urban coast, mitigating the impacts of poor water quality terrestrial runoff from agricultural activities and vessel traffic or commercial gill-netting would result in all dugong planning units of medium conservation value, and nearly all units of low conservation value being categorised as low risk.

Accuracy of estimates

In this chapter, I identified the anthropogenic hazards to the dugong population in the GBRWHA using a Delphi technique (Veal 1992) at a meeting of experts. The meeting was conducted prior to the seagrass assessment outlined in Chapter 4, and several of the threats to coastal seagrass habitats that were identified by seagrass experts were not identified by dugong experts (i.e. urban and industrial runoff, urban and port infrastructure development, dredging and shipping accidents). The results of my analysis may have underestimated the risk to dugongs because all of the hazards to their seagrass habitats were not included in the assessment. However, the error would be low as seagrass planning units that had a high relative impact (*see* Chapter 4; Figures 4.1 and 4.2) did not overlap with dugong planning units of high or medium conservation value. The dugong experts rated the relative impact of anthropogenic threats on seagrass

habitats (i.e. trawling and terrestrial runoff) as low relative to direct threats such as Indigenous hunting and bycatch in gill-nets (Table 7.1). Although my assessment of risks to dugongs did not include all the hazards to coastal seagrass habitats, the location of 'hot spots' for dugong conservation action should be accurate as impacts on seagrass habitats are considered of less relative importance by dugong experts.

I found that banning commercial gill-netting or Indigenous hunting in the remote Cape York region would result in a low risk from the cumulative impact of multiple threats for almost all dugong planning units of high and medium conservation value. However, the Population Viability Analysis modelling of Heinsohn et al. (2004) suggests that Indigenous hunting is not sustainable in the region. If this is correct, I have overestimated the area in which dugongs have a low level of risk by not including areas in which Indigenous hunting is the only human impact. If I had classified areas where Indigenous hunting is the only human impact as 'high risk' then 61, 80 and 92 % of dugong planning units of high, medium and low conservation value respectively were at low risk from anthropogenic activities. Classifying Indigenous hunting as a high risk factor for dugongs in the remote Cape York region makes a considerable difference in the level of risk to dugongs in high and medium conservation value areas, and little difference in the level of risk to dugongs in low conservation value areas (Table 7.2).

I believe my original analysis is robust to this uncertainty because there has been no detected change in the population of dugongs in the remote Cape York region (Marsh et al. 2007). The failure of four large-scale aerial surveys spanning 20 years to detect any evidence of decline in the dugong populations despite the relatively high precision of the population estimates suggests that dugong aerial surveys may still underestimate population size, probably largely because the availability correction factor is underestimated (H. Marsh personal communication.). As a consequence, the Population Viability Analysis modelling of Heinsohn et al. (2004) may be inaccurate as it requires the estimates of the dugong population to be absolute. The Great Barrier Reef Marine Park Authority and the Queensland Department of Environment and Resource Management are working across jurisdictions with Indigenous Traditional Owner groups to develop management arrangements (e.g. Traditional Use of Marine Resource Agreements or Memoranda of Understanding) that inform and manage hunting and other issues of priority to Indigenous peoples in the GBRWHA (Havemann et al. 2005).

My assessment of the sensitivity of the matrix of dugong protection to removal of various threats (Table 7.3) does not mean that the consequential reduction in anthropogenic mortality would be the same for removal of each threat. My estimates of relative impact of threats based on expert opinion are tentative. More-robust comparisons will require: (1) accurate data on the actual levels of anthropogenic mortality from impacts in various areas of the GBRWHA; (2) additional information on the genetic structuring of the dugong stock; and, (3) information on the risks to dugongs moving between areas of occupancy in the GBRWHA. This information will take many years to obtain.

As explained in Chapter 6, dugongs undertake macroscale movements between bays and mesoscale movements within bays between sites of significant seagrass habitat (Gales et al. 2004; Marsh et al. 2004, 2005; Sheppard et al. 2006). These movements are generally restricted to within 20 km off the coast, but no discrete movement corridors have been identified (Sheppard et al. 2006). If defined corridors exist, the spatial model of dugong conservation value outlined in Chapter 5 will have to be revised because it classifies potential movement corridors as having low conservation value relative to other sites within the dugong's range in the GBRWHA on the basis of their low dugong density. There is currently limited protection for dugongs moving between sites of significant habitat along the urban coast (Figure 7.2); and I have identified these areas as potential sites for conservation action.

Lessons learned

Along the urban coast I found that banning commercial gill-netting or simultaneously mitigating the hazards of poor water quality terrestrial runoff and vessel traffic would provide the greatest improvement in protection for dugong planning units of medium and low conservation value. Removing trawling or Indigenous hunting would have a minimal impact on protection, largely because of the voluntary moratorium on hunting currently implemented by most Traditional Owner groups in the region. To maximise the likelihood of dugong populations recovering along the urban coast, management should aim for an anthropogenic mortality target of zero as advocated by Marsh et al. (2005) on the basis of Potential Biological Removal modelling (Wade 1998). This management approach necessitates: (1) asking Indigenous groups to continue their

moratorium on hunting; (2) banning commercial gill-netting along the urban coast of the GBRWHA; and, (3) addressing the hazards of vessel traffic and poor water quality terrestrial runoff.

The protection afforded by the ecosystem-scale network of MPAs in the GBRWHA is limited by their ability to mitigate all the factors that threaten the marine environment, including activities in the adjacent coastal catchments. Hoyt (2005) and Jameson et al. (2002) describe a basic principle that should be observed when designing MPAs: appreciate the links between marine, coastal, and terrestrial ecosystems. Thus, potential anthropogenic impacts from contaminated terrestrial runoff from agriculture, vessel traffic, Indigenous hunting, coastal development and pollution should also be recognised and managed when designing ecosystem-scale networks of MPAs. As for most broad-scale MPAs, effective management of all anthropogenic activities in the GBRWHA will require increased cross-jurisdictional collaboration.

A tool for rapidly assessing ecosystem-scale MPA networks

By using a spatial risk assessment approach, I was able to compare and rank threats in order to identify the most severe threats first, and to locate ‘hot spots’ that are a priority for dugong conservation action. I found that dugong planning units of high and medium conservation value in the coastal waters between Port Stewart and Friendly Point (Figure 7.1) in the remote Cape York region have a comparatively high risk from anthropogenic activities, largely because of the limited restrictions on commercial gill-netting. For MPAs to be effective in species conservation over large geographic regions, it is essential to manage effectively those areas where the species are most vulnerable (Roberts et al. 2001). It may be politically untenable to protect a species by restricting anthropogenic activities for an entire region’s coastline, but management plans can be successful by protecting sites where species are abundant. Furthermore, targeting management initiatives to ensure these areas are resilient to anthropogenic threats will further enhance species conservation goals.

Ecosystem-scale networks of MPAs are implemented over broad spatial scales and quantitative information on the distribution of species such as mobile marine mammals and their anthropogenic threats is inevitably scarce. Uncertainty and incomplete

information can be a major constraint to the decision-making process (Bacic et al. 2006). However spatial risk assessments are a tool that can rapidly evaluate the risk to mobile marine mammals from a variety of activities that adversely impact them in an uncertain environment. I believe that rapid spatial risk assessments are potentially valuable for managing species that range over large areas that are managed across jurisdictions, such as World Heritage Areas and other large-scale marine-planning initiatives. For example, a spatial risk assessment could be used to assess the status of risk for New Zealand's endangered Hector's dolphin (*Cephalorhynchus hectori*), an inshore species threatened by multiple factors, including incidental mortality in gill-nets (Slooten et al. 2000). This management tool could also be used to assess existing MPAs including the Galapagos Marine Reserve ($> 133,000 \text{ km}^2$). The Galapagos Marine Reserve, which is also a World Heritage site, supports other species of mobile marine wildlife and is threatened by commercial fishing activities and tourism.

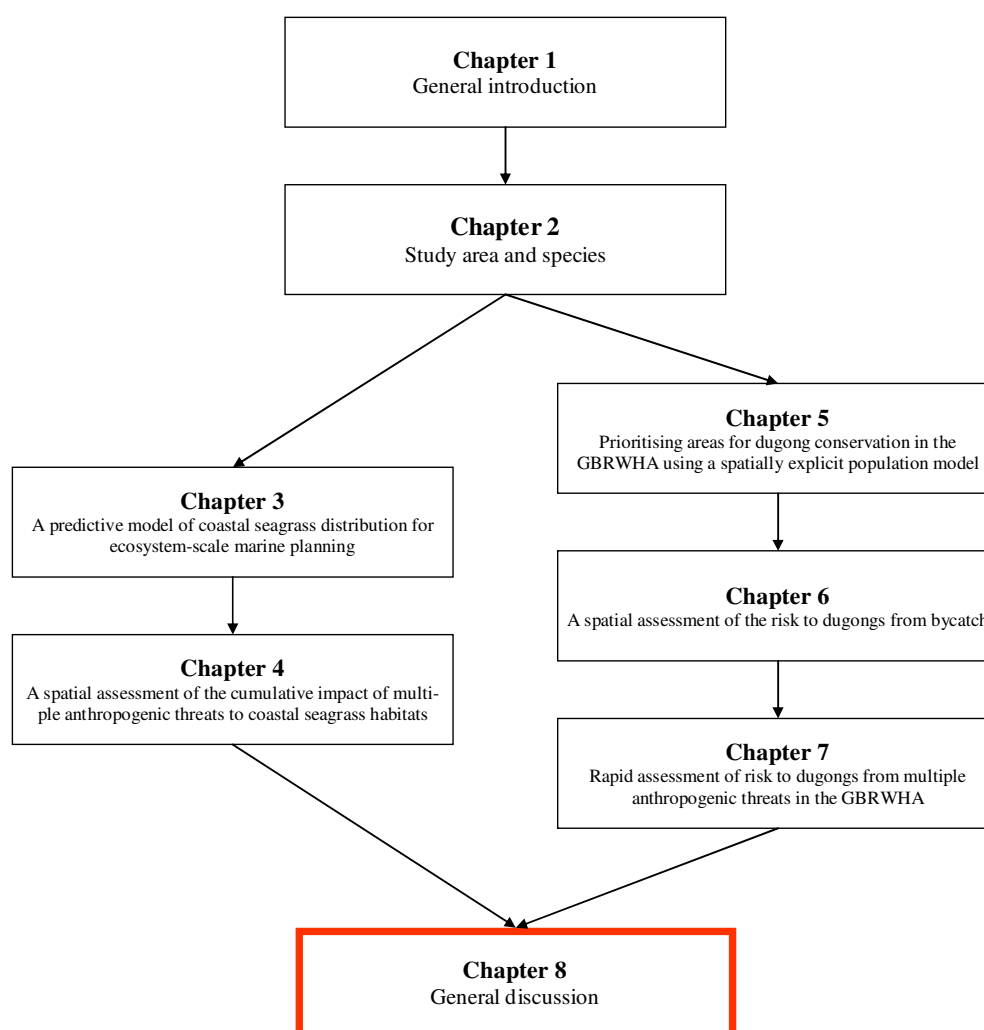
Chapter Summary

- Ecosystem-scale networks of MPAs are important management tools, but their effectiveness is difficult to quantify in a timeframe appropriate to species conservation because of the uncertainties in the data available.
- In this chapter, I outlined a rapid approach to assess the risk to dugongs in the GBRWHA and evaluate options to ameliorate that risk. I used expert opinion and a Delphi technique to identify and rank five anthropogenic activities with the potential to adversely impact dugongs and their seagrass habitats: commercial gill-netting, Indigenous hunting, trawling, vessel traffic, and poor quality terrestrial runoff. I quantified and compared the distribution of these factors with the spatially explicit model of dugong distribution and relative abundance outlined in Chapter 5.
- I found that under the current GBRWHA zoning and management arrangements, 96% of dugong planning units of high conservation value are at low risk from anthropogenic activities (100% along the urban coast and 96% in the remote Cape York region).
- I found that a decrease in risk would require commercial gill-netting or Indigenous hunting to be banned in remote areas, and the impacts of vessel traffic, terrestrial runoff and commercial gill-netting to be mitigated in urban areas.
- I identified dugong planning units of high and medium conservation value that are ‘hot spots’ for conservation action because the current level of risk to dugongs is high. These planning units included the coastal waters between Port Stewart and Friendly Point in the remote Cape York region; and between the high and medium conservation value planning units of Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay, and Shoalwater Bay and Port Clinton along the urban coast.

Chapter 8

General discussion¹

In this chapter, I provide a summary of the outcomes of this thesis and their implications for the planning and management of coastal seagrasses and dugongs at the scale of the GBRHWA. I discuss how spatial models and risk assessments are effective tools for informing a systematic approach to marine planning at ecosystem-scales. I conclude that future research should focus on understanding the constraints and opportunities to the implementation of conservation actions.



¹ **Grech, A.**, Marsh, H. and Coles, R. *in prep.* Constraints and opportunities for implementing conservation actions in a multiple-use marine protected area. Target journal *Conservation Biology*.

Informing the planning and management of coastal seagrass habitats and dugongs in the GBRWHA

The Great Barrier Reef World Heritage Area (GBRWHA) of Queensland, Australia is an ecosystem-scale, multiple-use marine protected area (MPA) jointly managed by the Queensland and Australian Governments (*see* Chapter 2). The region supports a variety of habitats and species including coastal seagrasses and globally significant populations of dugongs, a threatened species of conservation concern. Coastal seagrasses occur along the entire 2,300 km coastline of the GBRWHA and informing their management requires data at that scale. Individual dugongs can move hundreds of kilometres in a few days (Sheppard et al. 2006), and all dugongs in the GBRWHA are considered part of a single stock (McDonald 2006). Management activities need to be conducted and evaluated at the scale of the dugong's distributional range in the GBRWHA ($> 30,000 \text{ km}^2$). As discussed in Chapters 4 and 7, coastal seagrasses and dugongs seagrass habitats are threatened by poor quality terrestrial runoff from adjacent land catchments, and habitat modification as a result of land reclamation, dredging, trawling and infrastructure development (Coles et al. 2007). Dugongs are also directly threatened by vessel strike, bycatch in commercial gill-nets and Indigenous hunting (Marsh et al. 2002).

Assessing the current GBRWHA management regime with regard to its ability to protect seagrasses and dugongs from their anthropogenic threats is challenging due to the difficulties associated with data collection and monitoring at the scale of the coastal GBRWHA. Collecting data on the distribution of habitats and species at that scale is expensive and logistically difficult. Accurate and efficient monitoring programmes that assess the responses of habitats and species are generally unavailable at the scale of the coastal GBRWHA due to multiple factors, including time, expertise, and cost constraints. It is unlikely that an experimental approach can be used to assess the effectiveness of management actions because seagrasses and dugongs are threatened by multiple anthropogenic activities, and it is impossible to determine the impact of one threat in the presence of multiple threats at the scale of the coastal GBRWHA due to logistical, ethical and political difficulties. Evaluating the effectiveness of management actions in the GBRWHA is made even more complicated when those actions attempt to

mitigate more than one threat (e.g. Great Barrier Reef Marine Park Zoning Plan, *see* Chapter 2 page 25).

The goal of this thesis was to contribute to the scientific basis for optimising the conservation of coastal seagrass habitats and dugongs by informing their planning and management at the scale of the GBRWHA. To achieve this goal I: (1) developed spatially-explicit models of seagrasses and dugongs at the scale of the coastal GBRWHA; (2) determined the relative impact of multiple anthropogenic threats using qualitative assessments informed by expert opinion; and, (3) used a spatial risk assessment approach to assess the effectiveness of the current GBRWHA management regime, and inform the design of future conservation actions.

Thesis outcomes

Objective 1: Quantify the spatial distribution of coastal seagrass habitats and dugongs at the scale of the entire GBRWHA.

Some tropical seagrasses in the GBRWHA are ephemeral, and an object-orientated mapping approach does not effectively represent spatial-temporal changes in coastal seagrass habitat distribution. In Chapter 3, I used spatial information on the distribution of seagrasses and predictor variables along with ecological theory and expert knowledge to inform the design of a Bayesian belief network and to develop a predictive habitat model. The Bayesian belief network quantified the relationships (dependencies) between seagrass and eight environmental drivers: relative wave exposure, bathymetry, spatial extent of flood plumes, season, substrate, region, tidal range and sea surface temperature. The analysis showed that at the scale of the entire coastal GBRWHA, the main factors associated with seagrass presence were tidal range and relative wave exposure. The outputs were probabilistic geographical information systems (GIS)-surfaces of seagrass habitat suitability for the entire GBRWHA coast in both the wet and dry seasons at a planning unit of scale 2 km * 2 km.

Prioritising areas of dugong conservation value is important for administering management resources at a GBRWHA scale as dugongs occur in high densities in a few localised habitats over a very large area. In Chapter 5, I developed a spatially explicit dugong population model to inform the prioritisation of dugong conservation actions at

a GBRHWA scale. Point-locality data from the 20 year time-series of aerial surveys were corrected for differences in sampling intensity and area sampled between surveys prior to the development of the model. I interpolated the corrected data to the extent of the aerial surveys using the geostatistical estimation method universal kriging. The model estimated the relative density of dugongs across the GBRHWA at the scale of 2 km * 2 km dugong planning units; the same scale as the predictive seagrass model, and the spatial scale recommended for managers under Criterion B of the *International Union for Conservation of Nature and Natural Resources Red List* (IUCN 2001). I classified each dugong planning unit as of low, medium, or high conservation value on the basis of the relative density of dugongs estimated from the model and a frequency analyses. Dugong planning units with a high conservation value are a priority for management in the GBRHWA because those units consistently supported high densities of dugongs.

Both of the spatially explicit models affirmed the relative importance of coastal ecosystems in the remote Cape York region for seagrasses and dugongs; even though the coastline along the urban coast (~ 3100 km) is almost three times the size as the coastline in the remote Cape York region (~ 1,100 km). Almost half of the planning units with medium and high conservation value for seagrasses were located in the remote Cape York region (*see* Chapter 3 page 47). The remote Cape York region also featured the greatest proportion of planning units of high and medium conservation value for dugongs (*see* Chapter 5 page 95).

Objective 2: Estimate the risk of coastal seagrass habitats and dugongs from their anthropogenic threats.

In Chapter 4, I used expert knowledge to evaluate the relative impact of coastal seagrass habitats to their anthropogenic threats based on the method of Halpern et al. (2007) and Selkoe et al. (2008). The vulnerability scores derived from expert opinion and spatial information on the distribution of threats were used to delineate areas of low, medium and high relative impact to coastal seagrass habitats. I compared the output with the predictive model of coastal seagrass distribution outlined in Chapter 3 and found that whilst most planning units in the remote Cape York region were classified as low relative risk, almost two thirds of coastal seagrass habitats along the urban coast were at high or medium relative risk from multiple anthropogenic threats.

The effects of the 2004 re-zoning of the GBRWHA and the associated industry restructuring on dugongs is difficult to quantify as accurate information on dugong bycatch in commercial gill-nets is unavailable. In Chapter 6, I used a spatial risk-assessment approach to evaluate the 2004 re-zoning and associated industry restructuring for their ability to reduce the risk of dugong bycatch. I found that the new zoning arrangements in the GBRWHA appreciably reduced the risk of dugong bycatch by reducing the total area where commercial netting is permitted. Re-zoning and industry restructuring also contributed to a 22% decline in the spatial extent of conducted netting activities.

In addition to commercial gill-netting, dugongs are directly threatened by Indigenous hunting, trawling, and vessel strikes; their seagrass habitats are affected by anthropogenic impacts such as trawling, poor quality water from terrestrial runoff, dredging and coastal developments (*see* Chapter 4 page 59). In Chapter 7, I developed a rapid approach to assess the risk to dugongs from multiple anthropogenic threats in the GBRWHA. Expert opinion and a Delphi technique (Veal 1992) were used to identify and rank anthropogenic activities with the potential to adversely impact dugongs and their habitats. I then quantified and compared the distribution of these activities with the spatially explicit model of dugong distribution outlined in Chapter 5. I found that almost all dugong habitats of high (96 %) and medium (93 %) conservation value in the GBRWHA were at low relative risk from anthropogenic threats.

Objective 3: Inform the management of coastal seagrass habitats and dugongs at the scale of the entire GBRWHA.

It would be unreasonable to protect coastal seagrasses and dugongs by restricting anthropogenic activities along the entire GBRWHA coastline (~ 2,300 km). However, management plans can be successful by protecting sites where coastal seagrasses and dugongs are most vulnerable (Roberts et al. 2001). In Chapters 4, 6 and 7, I identified seagrass and dugong planning units that are a priority for conservation action relative to other planning units in the GBRWHA because the present level of risk to the species was estimated to be high (Table 8.1 and Figure 8.1).

In Chapter 4, I identified 13 ‘hot spots’ of medium and high conservation value to coastal seagrass habitats that are at risk from multiple anthropogenic activities (Table

8.1 and Figure 8.1). Reducing the risk to coastal seagrass habitats in these ‘hot spots’ will require: (1) improving the quality of terrestrial water that enters the GBRWHA; (2) mitigating the impacts of urban and port infrastructure development and dredging; and (3) addressing the hazards of shipping accidents and recreational boat damage.

In Chapter 6, I identified three locations in the remote Cape York region where commercial gill-netting is still permitted and conducted in dugong management units of medium and/or high conservation value (Table 8.1 and Figure 8.1). Dugongs are also threatened by bycatch along the urban coast when they move between Missionary Bay (north of Hinchinbrook Island) and Cleveland Bay, and Shoalwater Bay and Port Clinton. In Chapter 7, I assessed the risk to dugongs from multiple anthropogenic threats, and identified a further five ‘hot spots’ for conservation action where the current levels of risk to dugongs was high (Table 8.1 and Figure 8.1). Decreasing the risk to dugongs from multiple anthropogenic threats would require mitigating the impact of: commercial gill-netting or Indigenous hunting in the remote Cape York region; and vessel traffic, poor quality terrestrial runoff and commercial gill-netting along the urban coast.

In total, I identified 18 ‘hot spots’ for seagrass and dugong conservation action at the scale of the coastal GBRWHA. The majority of these ‘hot spots’ occurred along the urban coast of the GBRWHA (Table 8.1 and Figure 8.1).

Table 8.1: Priority sites for coastal seagrass and dugong conservation action identified in Chapters 4, 6 and 7.

Coastal seagrasses	Dugongs
<i>Remote Cape York region</i>	
Lloyd Bay	Friendly Point – Port Stewart Bathurst Head Lookout Point
<i>Urban coast</i>	
North of Cairns	Hinchinbrook Island – Cleveland Bay
Trinity Inlet	Shoalwater Bay – Port Clinton
Cassowary Coast	
Hinchinbrook region	
Cleveland Bay	
Bowling Green Bay	
Alva Beach	
Upstart Bay	
Abbot Bay	
Edgecombe Bay	
Whitsunday Islands	
Rodds Bay	

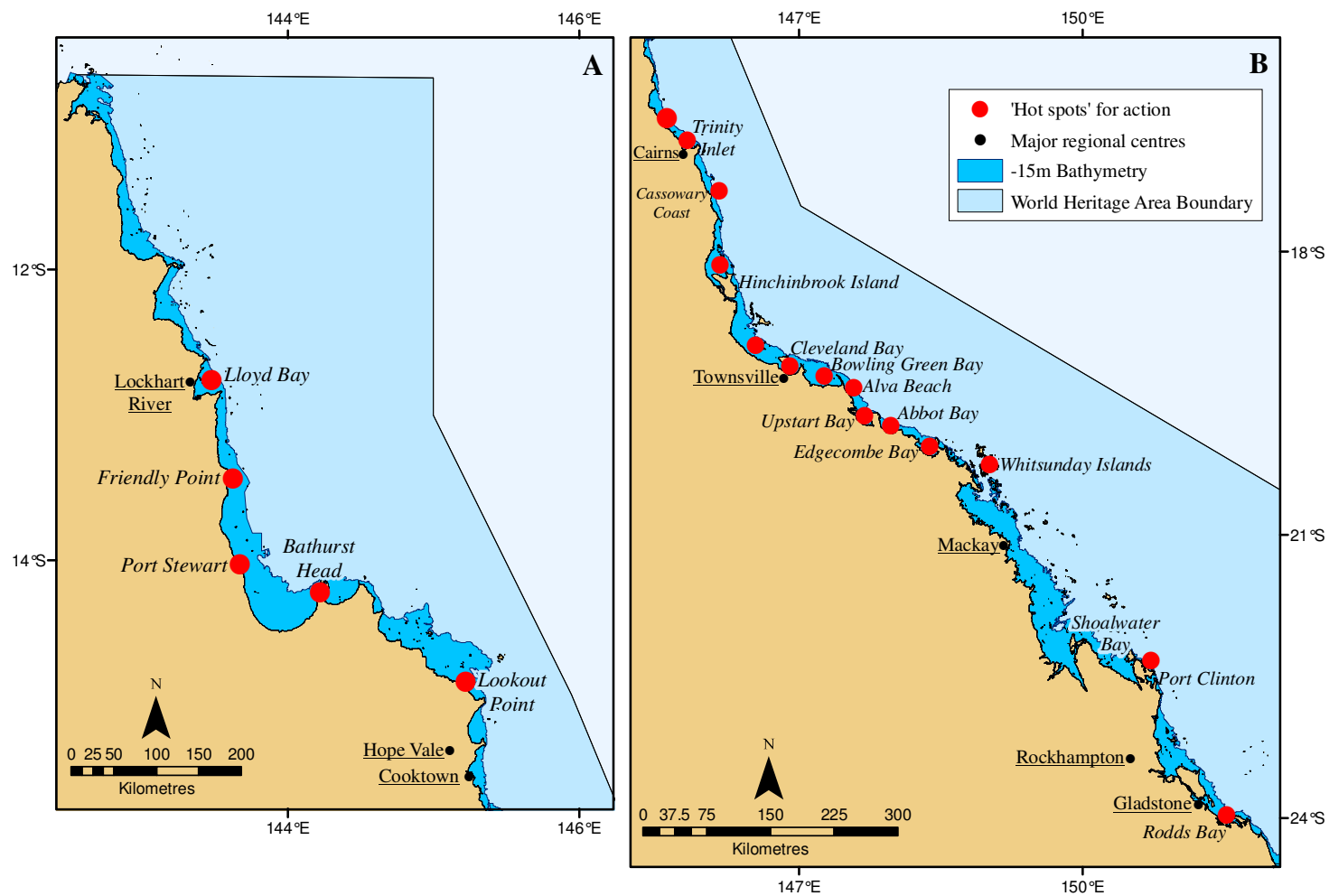


Figure 8.1: Priority sites ('hot spots') for coastal seagrass and dugong conservation action in the (A) remote Cape York region and (B) urban coast.

Evaluating the approach

Quantitative information and empirical data on the relative impact of anthropogenic threats on coastal seagrass habitats was incomplete or unavailable, and the cumulative impact of multiple threats was difficult to measure and predict. Similarly, information on dugong mortality, trauma or stress from anthropogenic impacts was unavailable; especially in the remote Cape York region where most dugongs occur. In the light of these uncertainties, I used expert knowledge to evaluate the vulnerability of coastal seagrass habitats to their hazards based on the method of Halpern et al. (2007) and Selkoe et al. (2008). The approach described by Halpern et al. (2007) and Selkoe et al. (2008) was specifically developed to assess the impact of threats to marine habitats such as seagrass, and not species. In Chapter 7, I used a different approach from Halpern et al. (2007) and Selkoe et al. (2008) to assess the risk of dugongs from multiple anthropogenic threats that accounted for the direct impact to dugongs from Indigenous hunting, vessel strike and bycatch in commercial gill-nets. I used a Delphi technique (Veal 1992) at a meeting of experts to: (1) identify the threats to dugongs; and, (2) rank and weight the relative impact of the threats.

As explained in Chapter 7, the meeting of dugong experts was conducted prior to the seagrass assessment, and several of the hazards to coastal seagrass habitats that were identified by seagrass experts were not identified by dugong experts (i.e. urban and industrial runoff, urban and port infrastructure development, dredging and shipping accidents). I may have underestimated the risk to dugongs in Chapter 7 because I did not include all of the threats to coastal seagrass habitats in the assessment. The error should be low as: (1) seagrass planning units that had a high relative impact did not overlap with dugong planning units of high or medium conservation value; and, (2) the dugong experts rated the composite impact of anthropogenic hazards on seagrass habitats (i.e. trawling and poor quality terrestrial runoff) as low relative to direct threats such as Indigenous hunting and bycatch in commercial gill-nets (Table 7.1). Although my assessment of risks to dugongs in Chapter 7 did not include all the hazards to their seagrass habitats, the location of ‘hot spots’ for dugong conservation action should be accurate as impacts on seagrass habitats are considered of lower relative importance by dugong experts than dugong mortality *per se*.

In Chapters 4 and 7, I did not identify the impacts of anthropogenic climate change as hazards to coastal seagrass and dugong habitats because there is no evidence that seagrasses are currently threatened by climate change in the GBRWHA (Short and Neckles 1999; Waycott et al. 2009). However, the Great Barrier Reef Outlook Report (2009) identified climate change as the greatest risk to the future health of ecosystems and species in the GBRWHA. Retaining the protection to sites that I identified as of a high value to seagrasses and dugongs is required to provide resilience to these species in the face of increasing sea temperatures, ocean acidification and rising sea levels. The seagrass and dugong risk assessments developed in Chapters 4 and 7 will also need to be updated in the future to account for changes in the global climate, and other changes in the anthropogenic threats I identified in this thesis.

I attempted to minimise the uncertainty in my analysis of the relative risk to coastal seagrasses and dugongs from multiple anthropogenic threats by basing my assumptions on quantitative and qualitative information made available through the literature and expert opinion. The models still contain uncertainties that are difficult to quantify due to the current lack of information on the characteristics and spatial distribution of factors that impact coastal seagrasses and dugongs in the GBRHWA and the future impacts of climate change. As new information becomes available, the assessment can easily be improved by reevaluating the assumptions, updating the geographic layers, and adjusting the expert weightings of hazards.

Management implications for coastal seagrasses and dugongs

I consider the assessments of Fernandes et al. (2005) and Dobbs et al. (2008) as inadequate because they did not use information that accurately delineates coastal seagrass and dugong distribution at the scale of the GBRWHA (*see* Chapter 1 page 9). As discussed in Chapter 1, the assessments of Fernandes et al. (2005) and Dobbs et al. (2008) are limited in their ability to inform the management of coastal seagrasses and dugongs in the GBRWHA as they only consider those anthropogenic activities that are within the regulatory control of ‘no-take’ areas. I found that the relative impact of trawling on dugongs and their seagrass habitats was low when compared to other anthropogenic threats such as Indigenous hunting, coastal development and poor water quality terrestrial runoff from adjacent land catchments. The approach that I used in this

thesis informs the management of coastal seagrasses and dugongs by assessing both the zoning arrangements and other management arrangements in the GBRWHA.

The information derived from this thesis is relevant to the Queensland and Australian Governments in relation to their conservation goals for coastal seagrasses and dugongs. The conservation goal for seagrasses advocated by the Queensland and Australian Governments is zero net-loss (Coles and Fortes 2001). Implementation of the zero net-loss policy would require the Queensland and Australian Governments to either: (1) reduce the risk to seagrass habitats along the urban coast, especially in areas identified in this thesis as 'hot spots' for conservation action; or (2) mitigate the potential loss of seagrass habitats along the urban coast by offsetting its loss with the improved protection of seagrass habitats in the remote Cape York region. Reducing the risk to seagrasses along the urban coast would require improving the quality of terrestrial water that enters the GBRWHA and reducing the impacts of urban and port infrastructure development, dredging, shipping accidents and recreational boat damage.

The Australian Government's conservation goal for dugongs is to '*facilitate the recovery of dugong populations such that they fulfill their ecological role within the GBR[WHA] ecosystem*' (GBRMPA 2007). I agree with Marsh et al. (2005) that management should aim for an anthropogenic mortality target of zero on the urban coast to maximise the likelihood of dugong populations recovering there. This management approach necessitates: (1) asking Indigenous groups to continue their moratorium on hunting; (2) banning commercial gill-netting in dugong habitats identified in this thesis as 'hot spots' for conservation action; (3) addressing the hazard of vessel strike; (4) improving the quality of terrestrial water that enters the GBRWHA; and, (5) reducing the risk to dugong's seagrass habitats along the urban coast, including the hazards resulting from urban and port infrastructure developments. In the remote Cape York region, recovery of the dugong population would require banning commercial gill-netting or Indigenous groups agreeing to a moratoria on hunting in dugong habitats that were identified as 'hot spots' for conservation action (Table 8.1 and Figure 8.1).

Along the urban coast, it will become increasingly difficult to implement conservation actions that decrease the risk to coastal seagrasses, dugongs and their habitats due to the political, social and economic costs associated with those actions. As explained in

Chapter 2, the projected population growth rate in the GBRWHA catchments is about 2% per annum (OESR 2008). A larger human population along the urban coast will inevitably lead to increased pressure on coastal ecosystems from recreational boating activities, urban and industrial development and poor water quality terrestrial runoff. The majority of Queensland's ports are located along the urban coast in the GBRWHA and 6,000 ships move through the region each year. The proposed expansion of several ports in the region, especially the ports of Gladstone and Townsville, will lead to an increase in risk to coastal seagrasses and dugong habitats from dredging, land reclamation, the development of port infrastructure and shipping accidents. The Queensland and Australian Governments are therefore more likely to achieve their goals of zero net-loss of seagrass and dugong recovery by: (1) offsetting the loss of coastal seagrasses along the urban coast with the improved protection of seagrass habitats in the remote Cape York region; and, (2) banning commercial gill-netting or reducing the impact of Indigenous hunting there.

The relative importance of the remote Cape York region for seagrasses and dugongs has significant policy consequences for the Queensland and Australian Governments. The coastal waters of the remote Cape York region support the majority of the GBRWHA's seagrass and dugong habitats (*see* Chapters 3 and 5), and the risk to seagrasses and dugongs from their anthropogenic threats in the region relative to the urban coast is small (*see* Chapters 4, 6 and 7). The political, social and economic costs associated with implementing conservation actions that protect all seagrass and dugong habitats in the remote Cape York region are minor compared to the amount required to achieve a similar outcome along the urban coast. However, it would not be appropriate for the Queensland and Australian Governments to limit their conservation resources and actions to the remote Cape York region. Lucas et al. (1997) state that the '*World Heritage Value of the Great Barrier Reef is a consequence of many attributes combining to produce a whole which cannot be reduced, without loss, to disconnected components*'. Maintaining the World Heritage Value of the region (especially for dugongs that were an explicit reason for the region's World Heritage listing [GBRMPA 1981]) would therefore require the Queensland and Australian Governments to improve the protection of seagrasses and dugongs along the urban coast.

Spatial models and risk assessments to inform marine planning at ecosystem-scales

As explained above, political, social and economic costs and limited conservation resources make it unreasonable to protect species by mitigating the impact of anthropogenic activities along the entire GBRWHA coastline. For conservation actions to be effective, it is essential to target resources to individual sites that can facilitate the achievement of conservation goals at that scale. There are a variety of approaches available to assist planners and managers in allocating conservation resources. Jameson et al. (2002) recommend a business plan approach that allocates conservation resources to sites where: (1) the number of uncontrollable threats is low; (2) the ability to manage these threats is high; and (3) community and institutional capacity is high. Vander Schaaf et al. (2006) present a similar approach that targets resources for conservation actions to sites where threats are lowest and successful implementation of actions are considered likely.

A shortcoming of both the above approaches is that they can result in the allocation of limited conservation resources to sites that are not a real priority as they are only marginally threatened (Pressey and Bottrill 2008). Margules and Pressey (2000), and Pressey and Taffs (2001) advocate the targeting of conservation resources and actions to sites that have a high irreplaceability and are highly threatened. The approach I used to identify ‘hot spots’ for conservation actions in this thesis is analogous to the framework for identifying conservation priorities originally described in Margules and Pressey (2000). I assumed irreplaceability was a function of the likelihood of seagrass habitat presence and dugong density, and I measured vulnerability by determining the relative impact of multiple threats and mapping their cumulative distribution. Spatial models and spatial risk assessments were used to identify sites that were the highest priority for conservation resources and actions as they were simultaneously: (1) important seagrass and dugong habitats at the scale of the coastal GBRWHA; and, (2) threatened by multiple anthropogenic activities. It is important to target conservation resources and actions to these sites because they: (1) are more likely to be lost; (2) have none or a small number of replacement areas; and, (3) their loss will have the most serious impact on the achievement of Queensland and Australia’s conservation goals for seagrasses and dugongs at the scale of the GBRWHA.

Informing a systematic approach to marine planning

Systematic conservation planning is the process of allocating limited conservation resources to minimise the loss of biodiversity, species and ecosystem services (Pressey and Bottrill 2009; *see* Chapter 1 page 5). This thesis informed a systematic planning approach for seagrasses and dugongs in the GBRWHA by: developing spatially-explicit models of species distribution at the scale of the coastal GBRWHA (*Stage 1*); identifying conservation goals for coastal seagrass and dugongs (*Stage 2*); and using a risk assessment framework to review the effectiveness of existing management arrangements (*Stage 3*) and identifying ‘hot spots’ for conservation actions (*Stage 4*).

The final two stages in a systematic approach to conservation planning are the implementation of conservation actions (*Stage 5*) and the maintenance of the required values of conservation areas (*Stage 6*). Margules and Pressey (2000) acknowledge that implementation of conservation actions and maintenance of conservation areas is complicated by the variety of social, economic, political and cultural interests within a region, and by the time and cost associated with applying conservation actions in some areas (Knight et al. 2006). Operational frameworks are being developed that explicitly incorporate socio-economic considerations from the outset (Cowling and Pressey 2003) by integrating social, economic, and political considerations with the technical aspects of analysing data on biodiversity. Pressey and Bottrill (2008 and 2009) build on earlier frameworks by including five new stages that precede the six technical stages of Margules and Pressey (2000). The five new stages include but are not limited to the: (1) collection of spatially-explicit, socio-economic data on threats, vulnerabilities and existing management arrangements; and, (2) assessment of the social, economic and political context for the planning process, including constraints and opportunities for implementing conservation actions.

I provided information that is relevant to two of the additional planning stages described by Pressey and Bottrill (2008 and 2009) by collecting, measuring and incorporating spatially-explicit data on threats, vulnerabilities and management arrangements at the scale of the coastal GBRWHA. Data on multiple threats and species vulnerabilities are rarely available (Wilson et al. 2005) especially at the broad spatial scales of ecosystems. The approach I outlined in this thesis overcame the difficulties associated with

measuring and incorporating threats and vulnerabilities into conservation planning at ecosystem-scales. I determined the relative impact of multiple anthropogenic threats on coastal seagrass habitats and dugongs in the GBRWHA by using qualitative assessments informed by expert opinion. A risk assessment framework was used to integrate: (1) expert opinion; (2) spatially explicit models of species distribution; and (3) qualitative and quantitative information on the distribution of multiple anthropogenic threats.

Future research

I identified sites that are the highest priority for seagrass and dugong conservation actions from information on the vulnerability of the species to anthropogenic activities and the distribution of threats. However, as pointed out by Pressey and Taffs (2001) and Pressey and Bottril (2009), it is insufficient to define conservation priorities only in terms of threats and vulnerabilities as conservation actions are impossible to implement unless constraints and opportunities are understood and accounted for in the planning process. Social, economic and cultural conditions in a region shape constraints and opportunities for conservation actions, and inevitably control the implementation of conservation plans. Implementation of conservation actions are politically constrained when those actions are anticipated to have a potentially negative impact on the social, cultural or economic wellbeing of communities and/or industry. Opportunities for conservation actions exist when the degree of impact on communities or industries is perceived to be low and when there is a political and/or organisational will and community interest (Green et al. 2009).

The social, economic and cultural constraints to and opportunities for increasing the proportion of ‘no-take’ areas in the GBRWHA from ~4 % – 33 % (*see* Chapter 2 page 25) were investigated by the Cooperative Research Centre for the Great Barrier Reef World Heritage Area (CRC Reef). CRC Reef commissioned a series of studies that evaluated the resilience of coastal communities and regions to changes in zoning and commercial fishing effort (Fenton and Marshall 2001; Fenton 2003). The resilience of communities and regions to changes in the spatial distribution of ‘no-take’ areas and reductions in commercial effort was found to be directly related to the mobility of fishing operations. Fishers with lower mobility operations, such as small scale inshore net fishers, have localised patterns of use and limited flexibility to alter their operations

and seek alternate fishing grounds in response to changes in zoning (BRS 2003). The artisanal nature of the inshore net fishery results in small operations and boats that restrict the amount of fuel that they can be carried and the distance that can be travelled. The trawl fishery is Queensland's largest commercial fishery, and trawl fishers have high mobility potential due to the scale that the fishery operates at (> 10,000 km) and the large size of the trawl boats, which allows fishers to exploit new trawl grounds on the introduction of area closures.

The Bureau of Rural Sciences (2003) and Marshall et al. (2007) found that resilience and mobility are typically associated with socio-demographic factors that include: age and family structure, income, housing type, employment, and education. Specifically, Marshall et al. (2007) found that coastal communities have limited resilience and mobility when there is a lack of employment opportunities within communities and when fishers have: (1) high levels of attachment to their occupation; (2) low transferable skills; and, (3) family members residing in the community. These conditions are more likely to occur in small regional centres along the urban coast of the GBRWHA. In large regional centres (i.e. human population > 50,000) fishers have greater mobility and resilience as: (1) they have more opportunities for employment outside of the fishing industry; and, (2) the fishing industry is dominated by trawl operations that have higher mobility potential.

The Bureau of Rural Sciences (2003) and Marshall et al. (2007) conclude that small regional centres along the GBRWHA coast were less resilient to changes in marine zoning; and the social and economic impacts of increasing the size and number of 'no-take' areas adjacent to these communities is greater than in the larger regional centres of Bundaberg, Cairns, Gladstone, Mackay and Townsville (Figures 1.1 and 8.1). During the re-zoning of the GBRWHA, implementation of new zoning arrangements was strongly constrained in small regional centres due to the lack of community support for 'no-take' areas (Olsson et al. 2008). In recognition of the economic costs associated with new zoning, especially to small regional centres, the Australian Government agreed to a structural adjustment package that provided compensation to commercial fishing industries that totaled AU\$211 million (DEWHA 2008).

As demonstrated by the re-zoning of the GBRWHA, the resilience and mobility of communities and industries can either constrain or provide opportunities for the successful implementation of conservation actions. Constraints and opportunities for conservation actions associated with resilience and mobility are not limited to commercial fishing activities. As explained in Chapter 2, Traditional Owners¹ can conduct Indigenous hunting of dugongs as Indigenous hunting rights have been affirmed by the Australian Government's *Native Title Act 1993*, subsequent judgments in the High Court of Australia and the Australian Government's *Environmental Protection and Biodiversity Conservation Act 1999* (see Havemann et al. 2005). Therefore, Indigenous hunting of dugongs cannot be banned in areas that I identified as a 'hot spots' for dugong conservation action in the remote Cape York region where Native Title is determined or likely to be determined. Opportunities for implementation of conservation actions exist when Traditional Owners have the capacity to alter the intensity of dugong take in their communities. The capacity to alter the intensity of take is related to cultural obligations and the availability for alternative sources of food, which is usually greater in urban areas than in the remote Cape York region.

In this thesis, I identified 'hot spots' for conservation actions where coastal seagrasses and dugongs are threatened by multiple anthropogenic activities (Table 8.1 and Figure 8.1), and proposed conservation actions that would mitigate the risk to the species at these sites. In proposing these actions I did not consider the resilience and mobility of users of the GBRWHA and its catchments and the implications it has on the planning and management of seagrasses and dugongs. Conservation actions that prohibit anthropogenic activities are unlikely to be implemented in 'hot spots' where a substantial proportion of the community is dependent upon those activities for commercial, recreational or cultural purposes and/or if those activities are commercially valuable to the State. In Table 8.2, I have provided a summary of the conditions that: (1) limit the mobility of activities or industries; (2) limit the resilience of communities to changes in activities or industries; and, (3) provide opportunities for conserving seagrasses and dugongs.

¹ Traditional Owners are Aboriginal and Torres Strait Islander people who are descendants of the tribe or ethnic group that occupied a particular region before European settlement.

In Boxes 8.1 and 8.2, I present case studies on the constraints to and opportunities for the implementation of conservation actions in two seagrass and dugong ‘hot spots’ identified in this thesis. I consider constraints and opportunities in the context of resilience and mobility of users of the GBRWHA and its catchments (*see* Table 8.2). I have provided updated recommendations for conservation actions by identifying those actions that are most likely to be implemented in the two ‘hot spots’. Future research on the planning and management of coastal seagrass habitats and dugongs should consider constraints and opportunities for implementation of conservation actions in the context of community resilience and mobility. I intend to develop these ideas further before preparing a manuscript on a systematic approach to evaluating the constraints to and opportunities for conservation actions, using seagrasses and dugongs in the GBRWHA as a case study.

Table 8.2: Conditions that generate varying levels of resilience and mobility for users of the GBRWHA and its catchments, categorised by their activity and/or industry.

Activity/Industry	Limits to mobility	Limits to resilience	Options for conserving seagrass and dugongs
Indigenous hunting	Native title removal rights; cultural obligations	Community size and isolation; limited employment opportunities; poverty; limited education; expense of store bought food	Poverty reduction (employment opportunities), habitat stewardship and reduction in size of dugong catch through Sea Country management agreements with government e.g. Traditional Use of Marine Resource Agreements (Havemann et al. 2005).
Urban and industrial development	Existing infrastructure	Alternative livelihood opportunities (inversely correlated with size of urban centre)	Infrastructure consolidation; some potential for restrictions on location and nature of new developments through municipal land use zoning and Environmental Impact Assessment process; establishment of buffer zones around urban centres through MPA zoning; declaration of Fish Habitat Areas that restrict infrastructure development; construction is prohibited in waters designated as a 'Remote Natural Area'.
Agriculture in land catchments	Land tenure is private and fixed	Alternative livelihood opportunities (inversely correlated with employability and attachment to occupation)	The State of Queensland and Commonwealth of Australia (2003) identified several catchments where communities are considered to have the capacity to change land management practices that potentially cause land-based pollution. These included the catchments adjacent to the seagrass 'hot spots' of Trinity Inlet and Whitsunday Islands.
Ports and marinas	Existing infrastructure		Restrictions on location and style of new developments through municipal land use zoning and Environmental Impact Assessment process; establishment of buffer zones around ports (e.g. eastern Cleveland Bay Box 8.2) through MPA zoning; recently signed Memorandum of Understanding between the Queensland Ports Association and the Great Barrier Reef Marine Park Authority will strategically improve coordination associated with port activity in the GBRWHA; construction is prohibited in waters designated as a 'Remote Natural Area'.

Activity/Industry	Limits to mobility	Limits to resilience	Options for conserving seagrass and dugongs
Recreational fishing and boating	‘Traditional’ access rights of local fishers	Communities dependent on tourism.	Controls on use through MPA re-zoning, vessel lanes and speed restrictions; establish buffer zones around existing marinas through MPA zoning; some potential to work with municipal authorities and EIA process about location of new marinas; motorised water sports are prohibited in waters designated as a ‘Remote Natural Area’.
Commercial shipping	Existing port infrastructure and access channels		Some potential to work with port authorities to adjust/ relocate shipping lanes and any associated dredging; potential for such adjustments decreases with proximity to port.
Gill-net fishery	Small scale fishery	Alternative livelihood opportunities (inversely correlated with size of urban centre and positively correlated with education levels and lifestyle preferences of fishers)	Controls on use through MPA re-zoning; net attendance rules; gear restrictions and modifications.
Trawl fishery		Alternative livelihood opportunities (positively correlated with education levels and lifestyle preferences of fishers)	Controls on use through MPA re-zoning, Bycatch Reduction Devices.
Military use	Existing land use zoning and infrastructure		Limited potential for spatial changes in existing use through negotiations with environment managing agency; some potential for restricting location and nature of new areas through municipal land use zoning and Environmental Impact Assessment process; also potential for the creation of exclusion zones around sites identified as important habitats for seagrasses and dugongs.

Box 8.1: Re-evaluating conservation actions between Port Stewart and Friendly Point

Location: Remote Cape York region.

Value: Majority of the coastline is of high dugong conservation value.

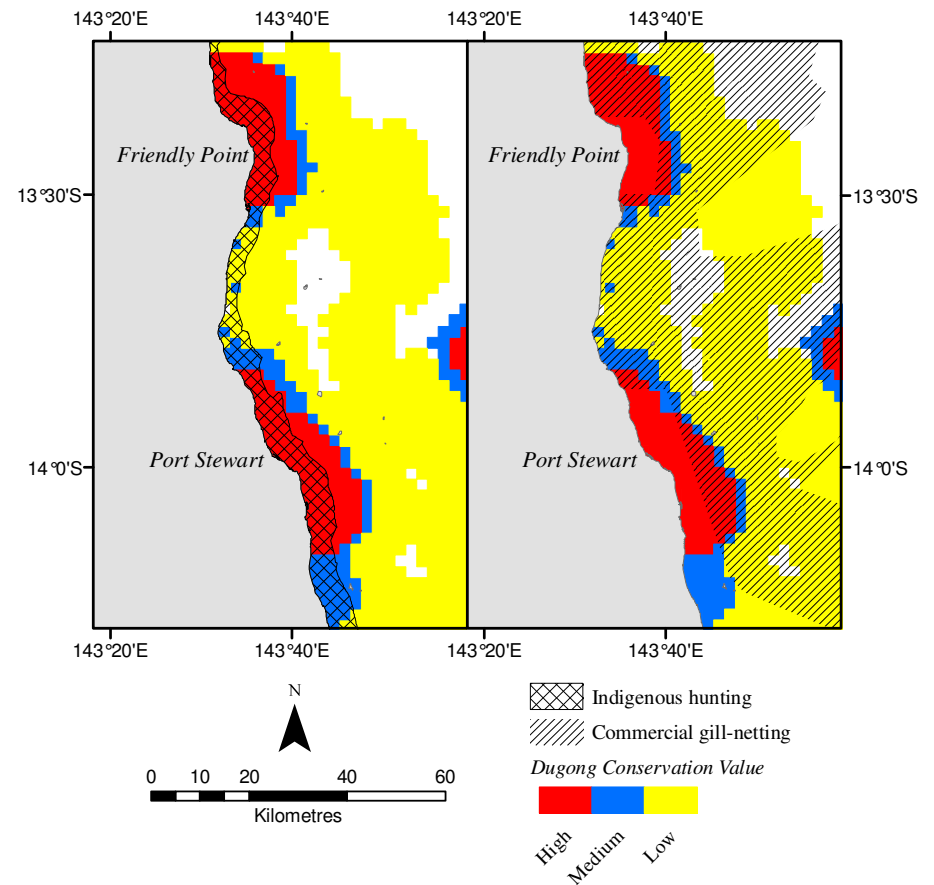
Threats: Indigenous hunting and commercial gill-netting.

Recommended conservation actions: Mitigation of the impacts of Indigenous hunting **or** commercial gill-netting.

Constraints: Banning Indigenous hunting is politically impossible as it would be interpreted as reducing the Native Title rights of Aboriginal Australians, and the small scale of the commercial gill-net fishery limits the mobility of fishers.

Opportunities: Fishers that operate in the coastal waters between Port Stewart and Friendly Point should be relatively resilient to changes in the fishery as they reside in the large regional centre of Cairns, which provides them with opportunities for alternative employment.

Updated recommendation in light of constraints and opportunities: Decreasing the risk to dugongs between Port Stewart and Friendly Point requires banning commercial gill-netting activities. It may also be possible for the relevant management agencies to develop management arrangements with Indigenous Traditional Owner groups (e.g. Traditional Use of Marine Resource Agreements or Memoranda of Understanding) to inform and manage hunting in the region.



Box 8.2: Re-evaluating conservation actions in Cleveland Bay

Location: Urban coast, adjacent to Townsville.

Value: The inshore waters of Cleveland Bay support inter-tidal and sub-tidal seagrass habitats. The south-east section of the bay is of high and medium dugong conservation value.

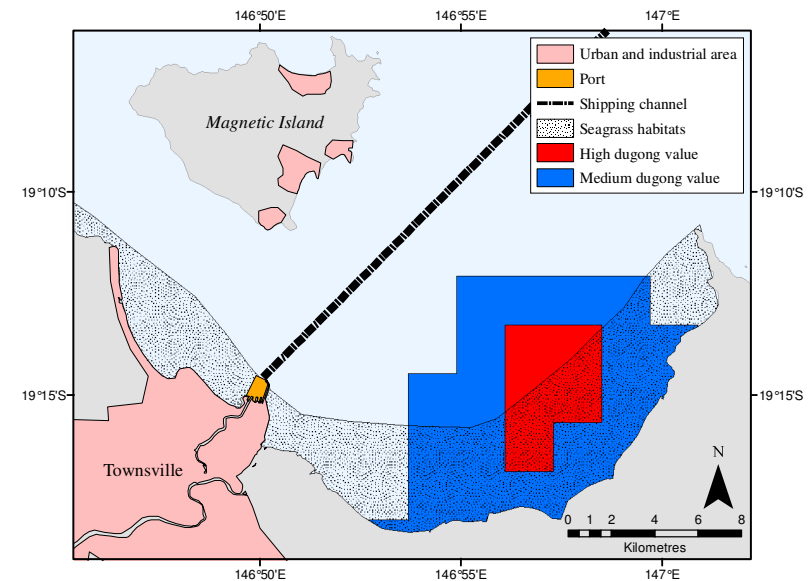
Threats: Urban and industrial development and runoff, port (dredging), shipping lane, recreational fishing and boating, commercial gill-netting

Recommended conservation action: Mitigation of all the identified threats.

Constraints: Mitigation of the impacts of current urban and industrial infrastructure, port and shipping lanes is unlikely.

Opportunities: Fishers that operate in Cleveland Bay are relatively resilient to changes in the fishery and have high mobility as they reside in the large regional centre of Townsville. Recreational fishing and boating activities can be managed through the implementation of speed restrictions and 'no-go' zones. The Cleveland Bay Fish Habitat Area restricts the development of infrastructure in most of the bay. Future urban, industrial and port developments in the south-east of Cleveland Bay that impact dugongs would require approval from the Australian Government under the *Environment Protection and Biodiversity Conservation Act 1997*. The *Great Barrier Reef Marine Park Zoning Plan* could be modified to include a buffer zone around the port and urban areas to remove commercial fishing activities.

Updated recommendation in light of constraints and opportunities: Decreasing the risk to coastal seagrasses, dugongs and their habitats in Cleveland Bay would require banning commercial fishing activities and restricting recreational boating activities. New urban, industrial and port developments should be limited to the already developed western section of the Bay. The Queensland and Australian Governments need to consider protecting the eastern section of Cleveland Bay, e.g. a declared Special Management Area, similar to Princess Charlotte Bay (see Chapter 6 page 110) would have the power to restrict boat activities and prevent the site from being developed in the future.



Applications of this research

Changes to the East Coast Inshore Fin Fish Fishery

From 2006 to 2008 the Queensland Government conducted a review of the management arrangements of the East Coast Inshore Fin Fish Fishery. A number of changes were proposed, including new and amended bag and size limits, new netting arrangements and improvements to the management of shark resources. Several changes to regulations controlling netting in Dugong Protection Areas (*see* Chapter 2) were also proposed, and included the extension of the Rodd's Bay Dugong Protection Area and increased netting restrictions around headlands. I was a member of Queensland's Primary Industries and Fisheries dugong working group that informed the Ministerial Advisory Committee on the changes to netting regulations in Dugong Protection Areas. I provided the Queensland Government and dugong working group with the model of dugong relative abundance and distribution and data on the distribution of netting activities to inform their review.

The proposed changes to the East Coast Inshore Fin Fish Fishery management arrangements did not address the risk of dugong bycatch in the coastal waters off the remote Cape York region. In partnership with our colleagues at James Cook University and the University of Queensland, Helene Marsh and I submitted a response to the proposed changes in netting regulations to both the Queensland and Australian Governments in February 2008. Based on the analysis conducted in Chapter 6 and published in *Aquatic Conservation: Marine and Freshwater Ecosystems*, we recommended to the Queensland and Australian Governments that netting should be banned in areas of high and medium conservation value to dugongs in the coastal waters of the remote Cape York region. Specifically, we identified three sites where commercial gill-netting is still permitted in dugong planning units of high or medium conservation value and where netting was conducted between January – June 2005 (*see* Chapter 6) We suggested to the Queensland and Australian Governments that removing commercial gill-netting at these sites would result in all dugong planning units of high and medium conservation value in the remote Cape York being designated as low risk from all anthropogenic activities (*see* Chapter 7).

The East Coast Inshore Fin Fish Fishery requires a Wildlife Trade Operation approval under Part 13 of the Australian Government's *Environment Protection and Biodiversity Conservation Act* 1999 to export its catch overseas. In February 2009 the Australian Government approved overseas export in the fishery for a further three years subject to 18 conditions and 14 recommendations. One of these conditions requires the Fisheries Queensland to examine and report on the significance of conservation benefits of additional spatial closures in waters north of Cooktown by December 1st 2009. I participated in the working group that reported on this requirement.

Spatially explicit dugong population models

Helene Marsh and I have received funding from the Queensland and Australian Governments to develop spatially explicit dugong population models for the 120,000 km² of Northern Territory and Queensland waters surveyed since 2005 (*see* Chapter 2 page 31; Figure 8.2). The models have been provided to the Australian Government to inform the development of Marine Bioregional Plans and a National Representative System of Marine Protected Areas in Commonwealth waters; and the Queensland and Northern Territory Governments to inform marine planning in their waters. The models have been used by the Australian Government to identify areas for further assessment as Marine Protected Areas (<http://155.187.2.69/coasts/mbp/publications/north-west/nw-and-north-factsheet.html>). The models have also been provided to the Torres Strait Regional Authority, Aboriginal communities in the GBRWHA and Gulf of Carpentaria and private Queensland organisations to provide a context for dugong management at local scales and to inform Environmental Impact Assessments.

Review of East Coast Otter Trawl Fishery

Similar to the net fishery, a Wildlife Trade Operation approval is required for the Queensland East Coast Otter Trawl Fishery to export its catch overseas. The Australian Government approved the export of catch to the fishery in 2004 on the condition that Fisheries Queensland initiate a review and provide a preliminary report on: (1) the adequacy of protection provided to species and benthic habitats in the fishery by the current system of closures within and outside the GBRWHA, and, (2) the potential benefit of additional closures outside the GBRWHA. In 2008 Rob Coles and I

developed the preliminary report using information on the distribution of dugongs and coastal seagrass habitats developed in this thesis and other datasets (Coles et al. 2008).

Concluding remarks

Informing marine planning and the management of species at ecosystem-scales is difficult because data are generally lacking at that scale. The approach I outlined in this thesis was able overcome the difficulties associated with informing the planning management of coastal seagrasses and dugongs in the GBRWHA by using spatial models and risk assessments in geographical information systems.

My approach has applications for systematic conservation planning at ecosystem-scales in other marine areas, including remote regions and developing countries where resources are limited. I was able to inform marine planning in a data-inadequate environment by combining qualitative assessments on the relative impact of multiple anthropogenic threats with spatial models of species and threat distributions. This approach allowed me to compare and rank the threats to identify the most severe risks, and to locate specific sites that require conservation actions.

Implementing conservation actions at the sites that I identified for management will provide the greatest positive result for seagrasses and dugongs at the scale of the GBRWHA. Future research should be directed at understanding the constraints and opportunities for management in the region to ensure that effective implementation of conservation actions can be achieved.

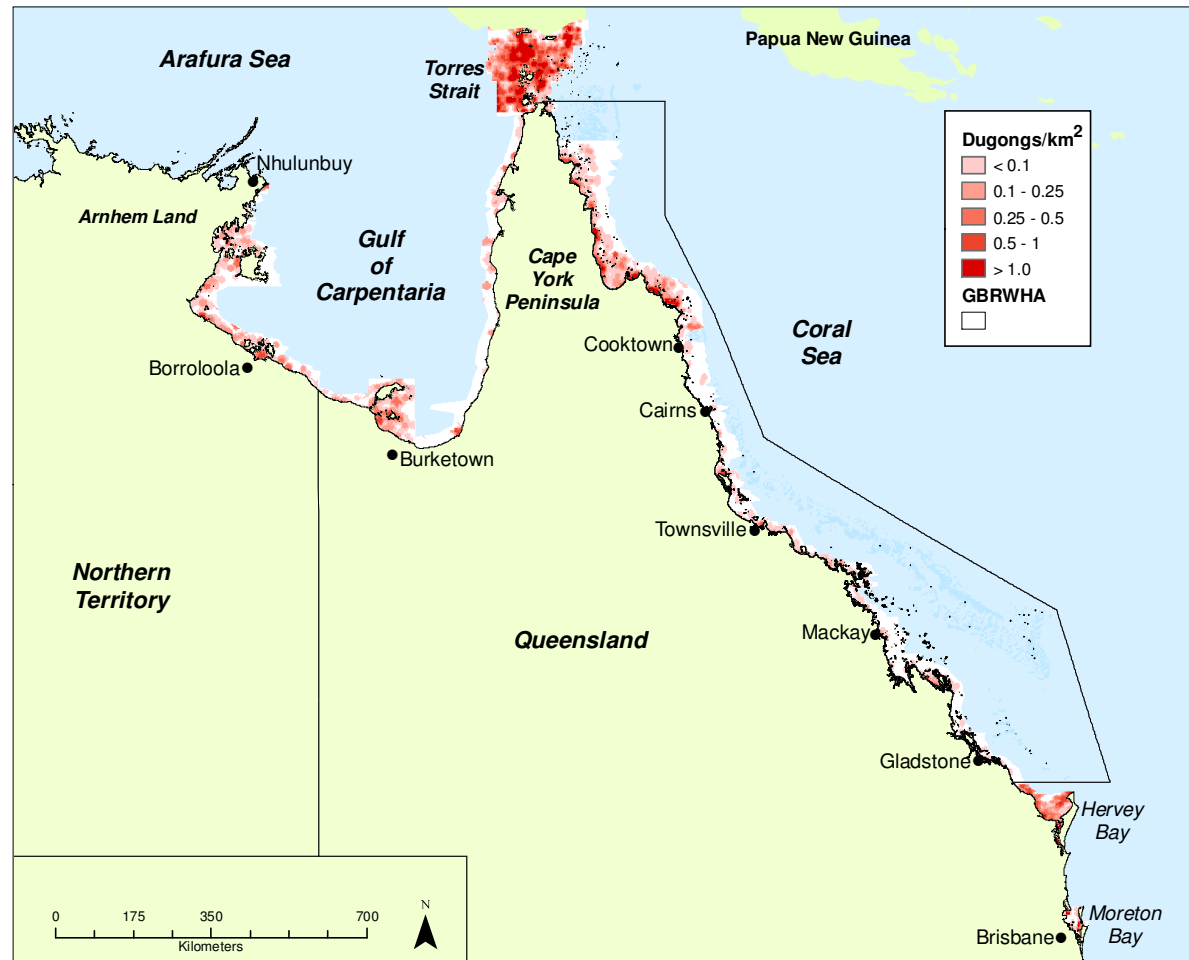


Figure 8.2: Models of dugong distribution and relative abundance for the region between Moreton Bay, Queensland and Nhulunbuy, Northern Territory.

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Appendix A

Chapter 3 supporting figures

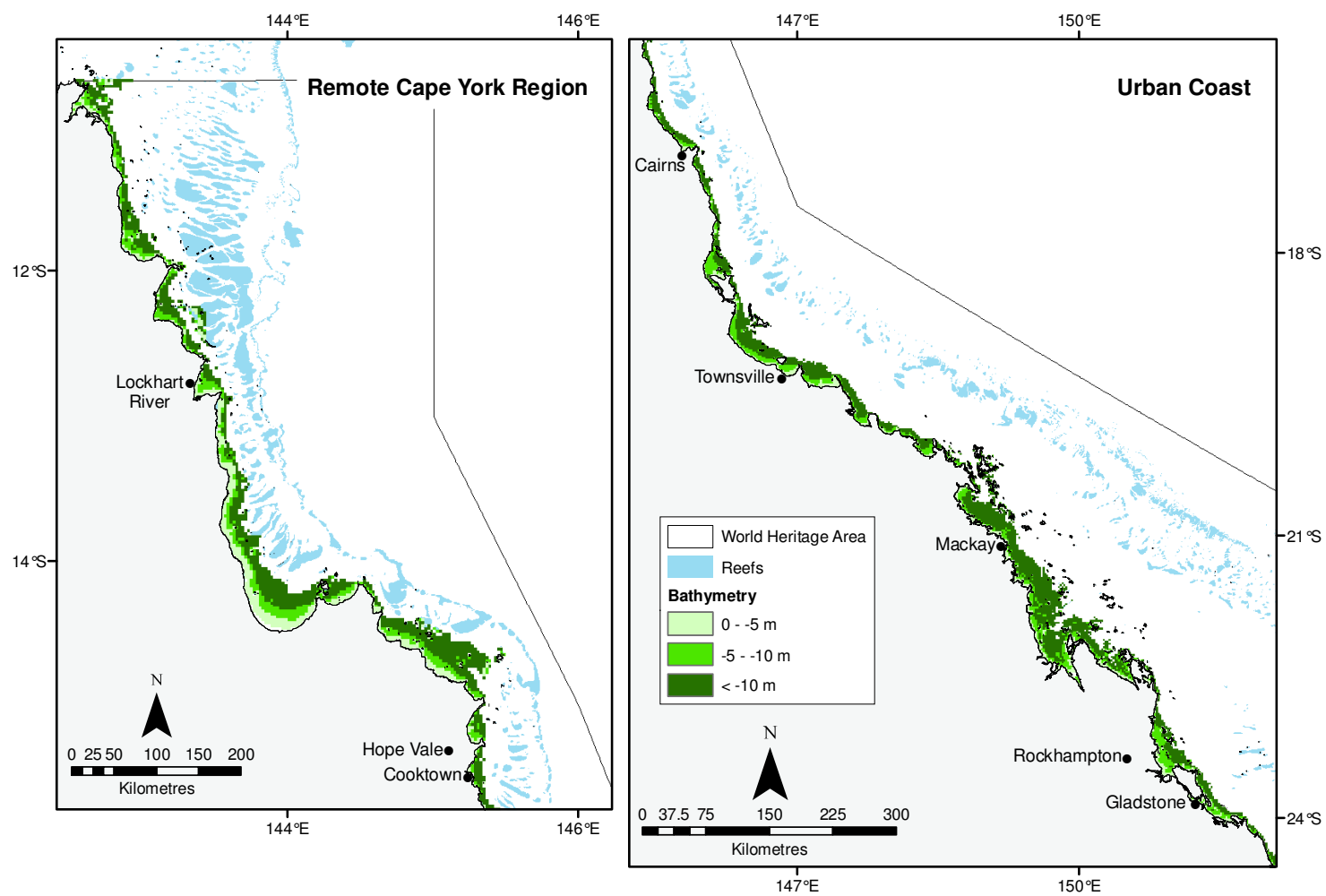


Figure A.1: Bathymetry (Great Barrier Reef Marine Park Authority; Lewis 2001)

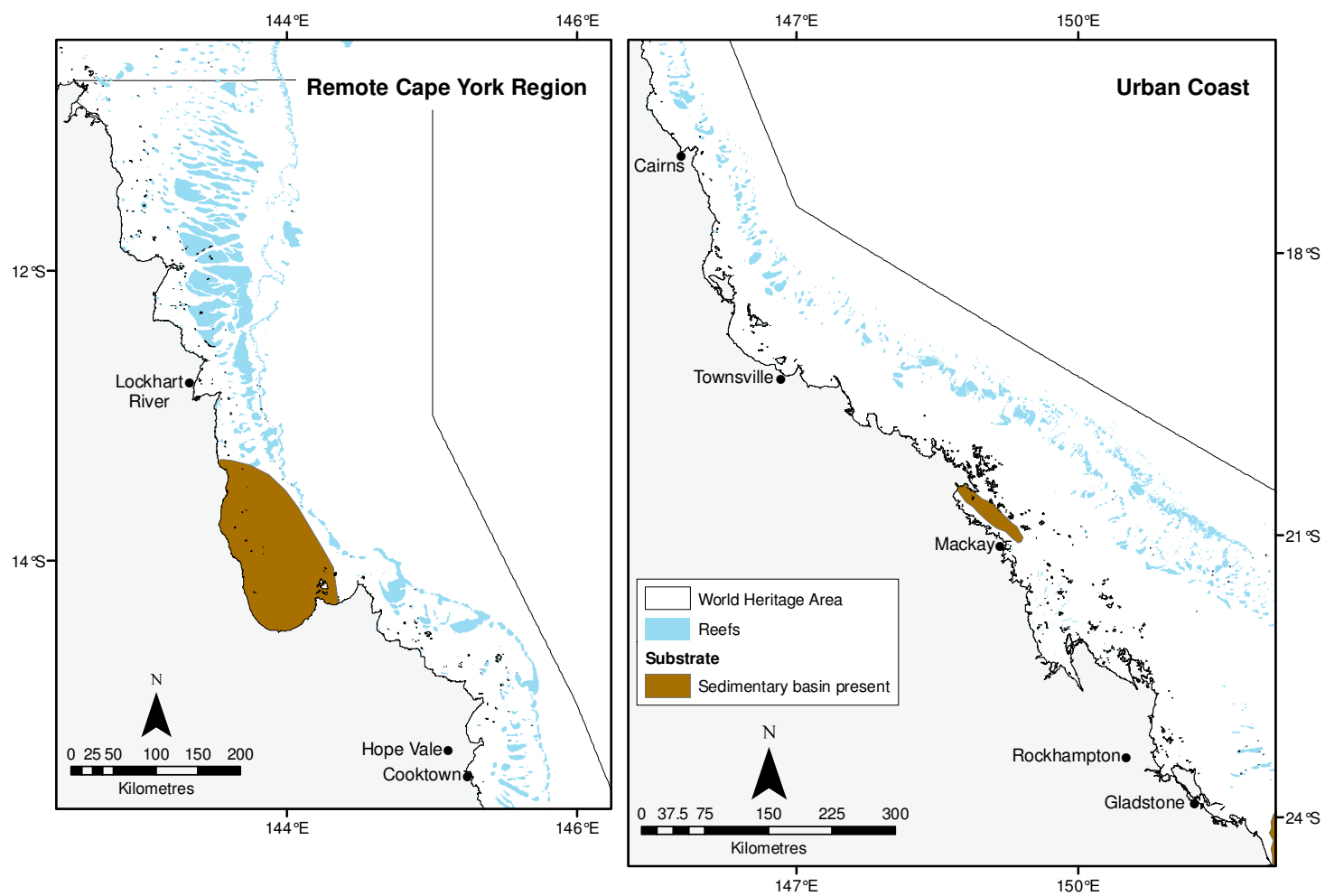


Figure A.2: Substrate (Geoscience Australia 2007)

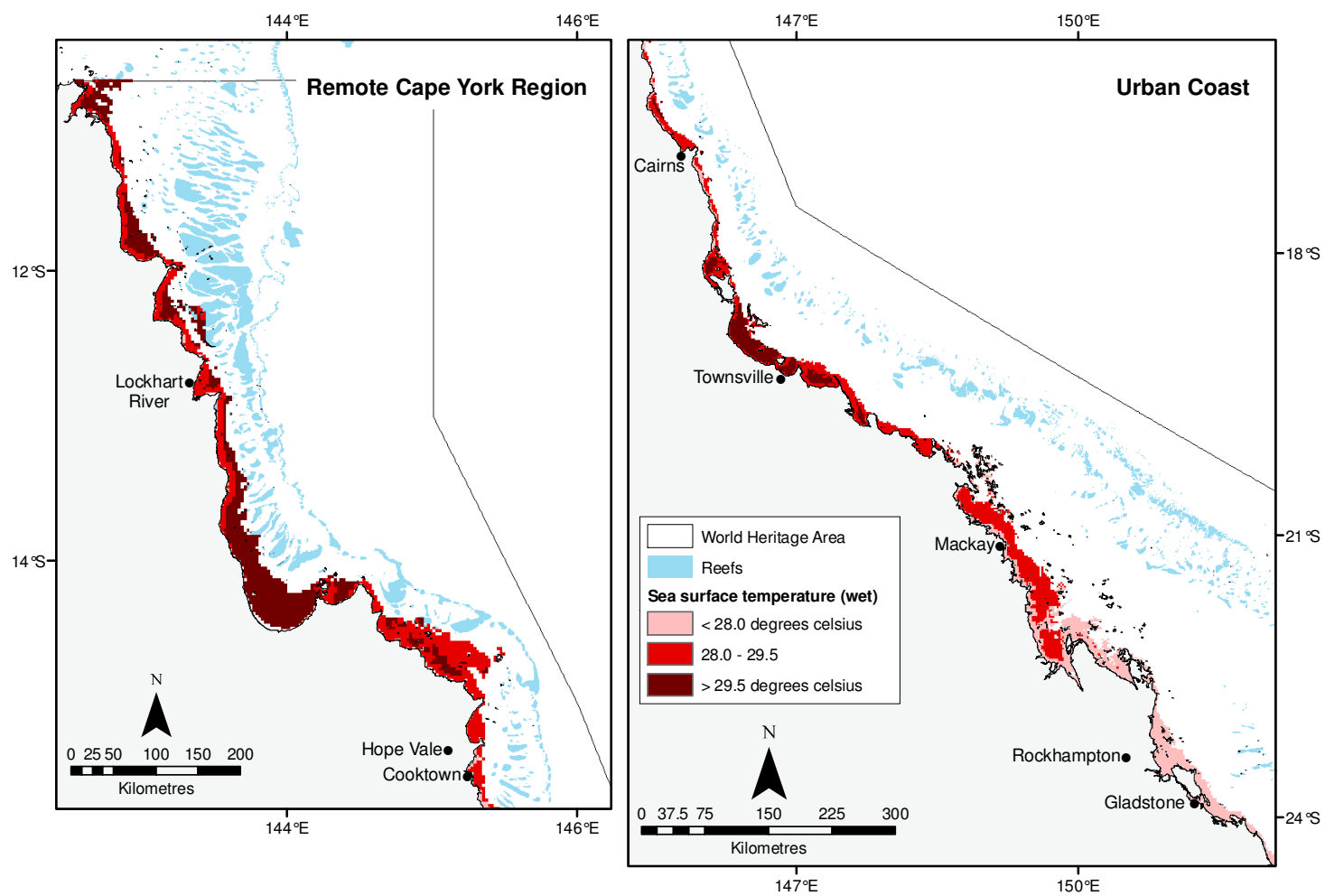


Figure A.3: Average sea surface temperature during the wet season (Australian Commonwealth Scientific and Research Organization 2007)

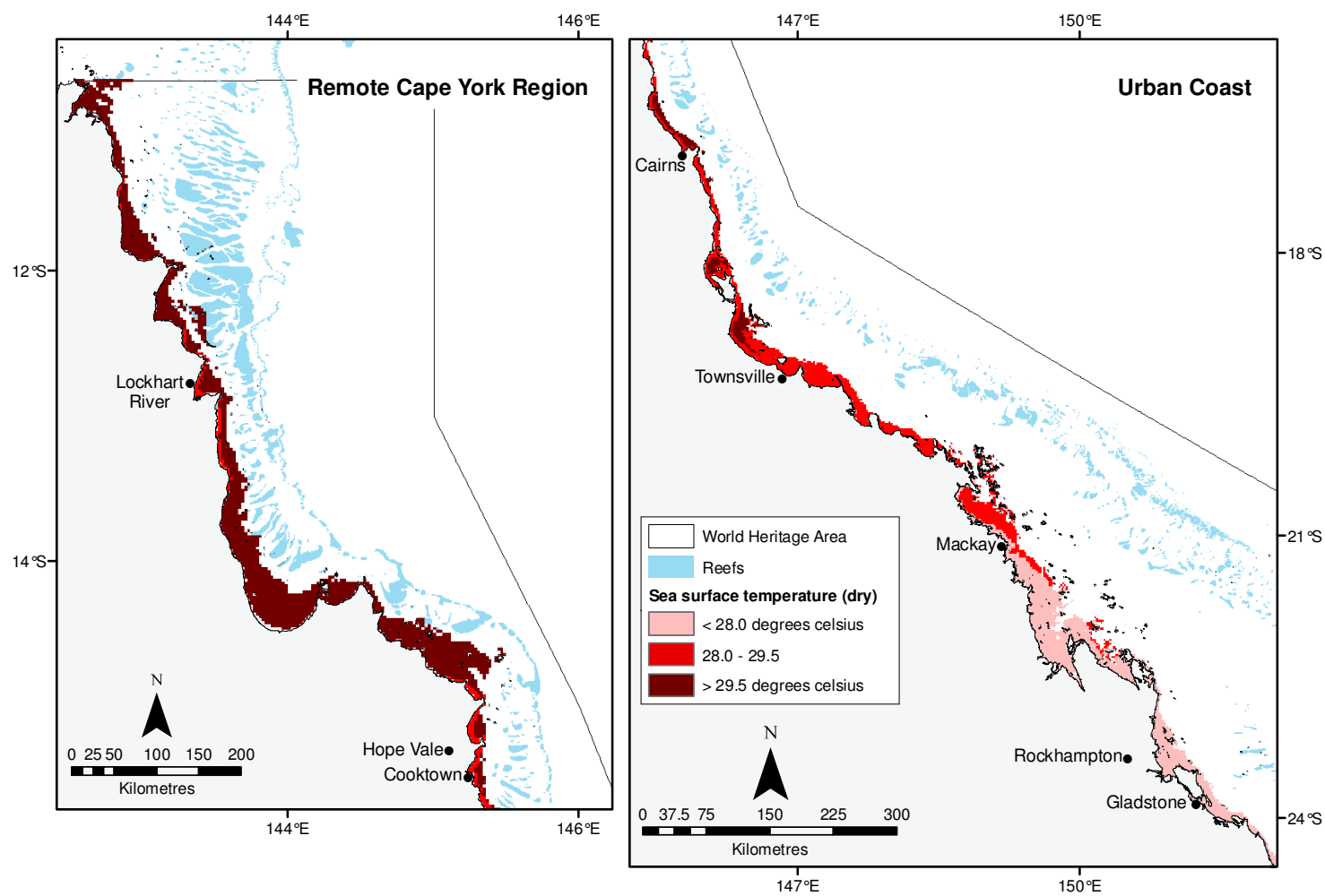


Figure A.4: Average sea surface temperature during the dry season (Australian Commonwealth Scientific and Research Organization 2007)

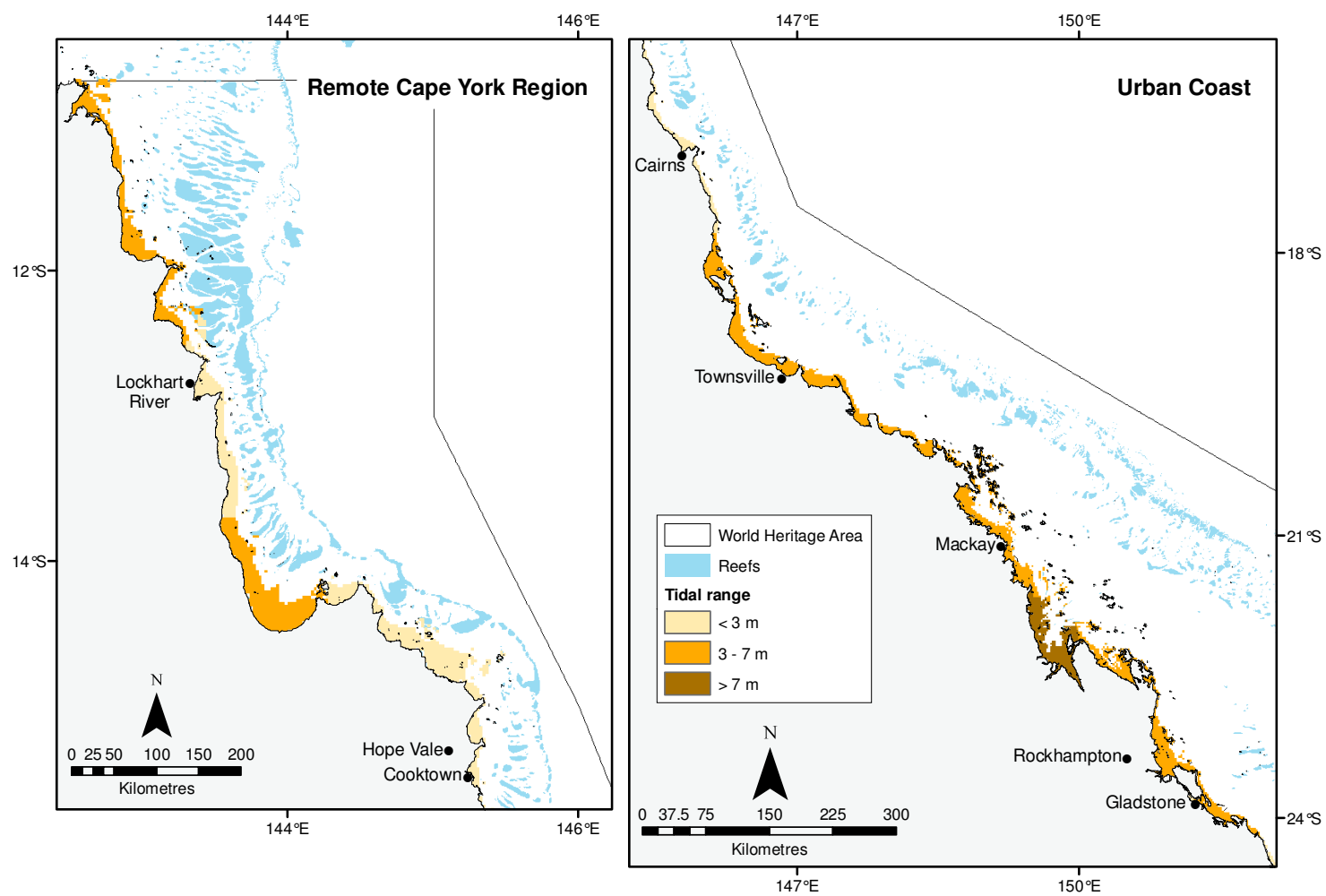


Figure A.5: Tidal range (Australian Maritime College; Hopley et al. 2007)

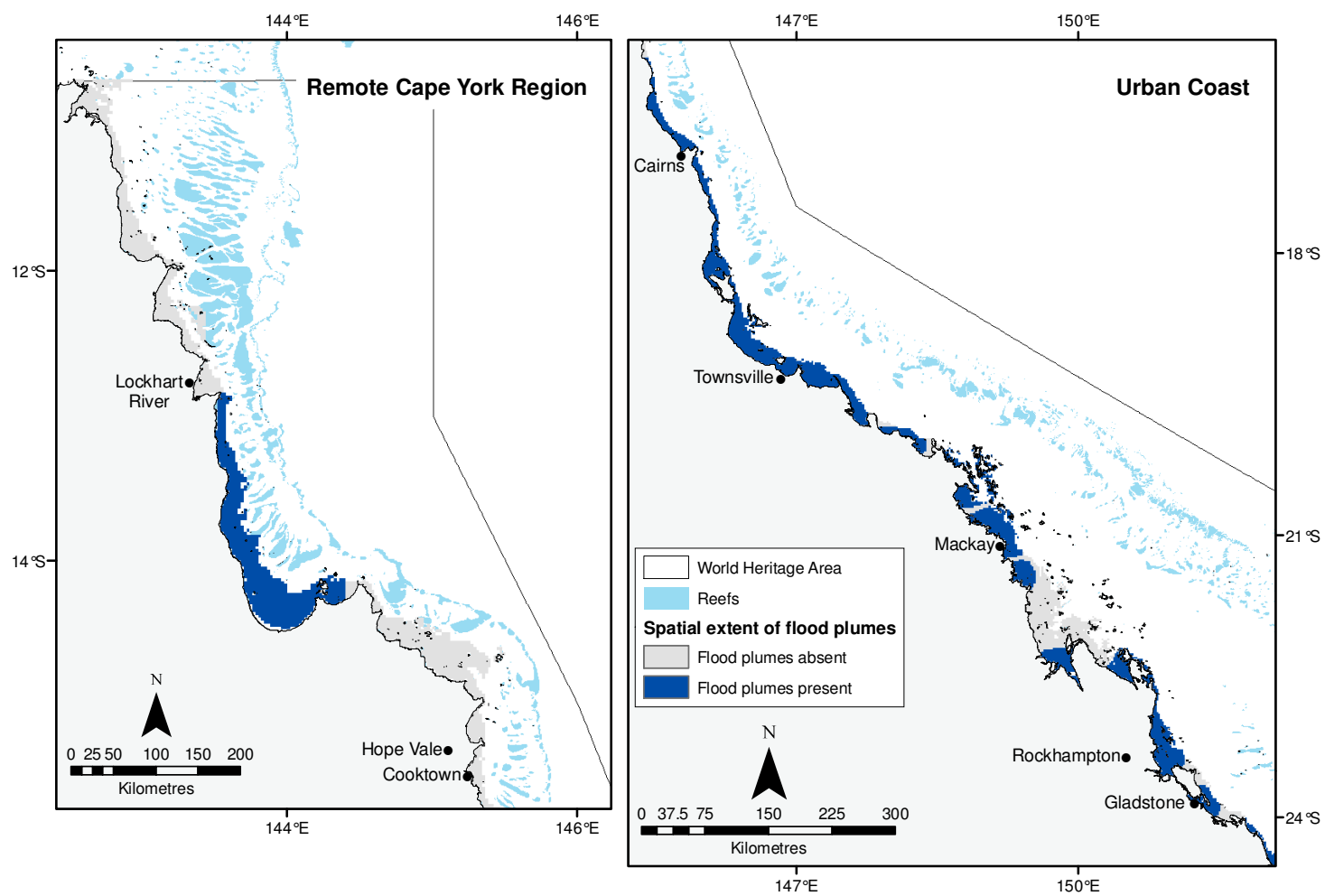


Figure A.6: Spatial extent of flood plumes (Australian Centre for Tropical Freshwater Research; Devlin et al. 2001)

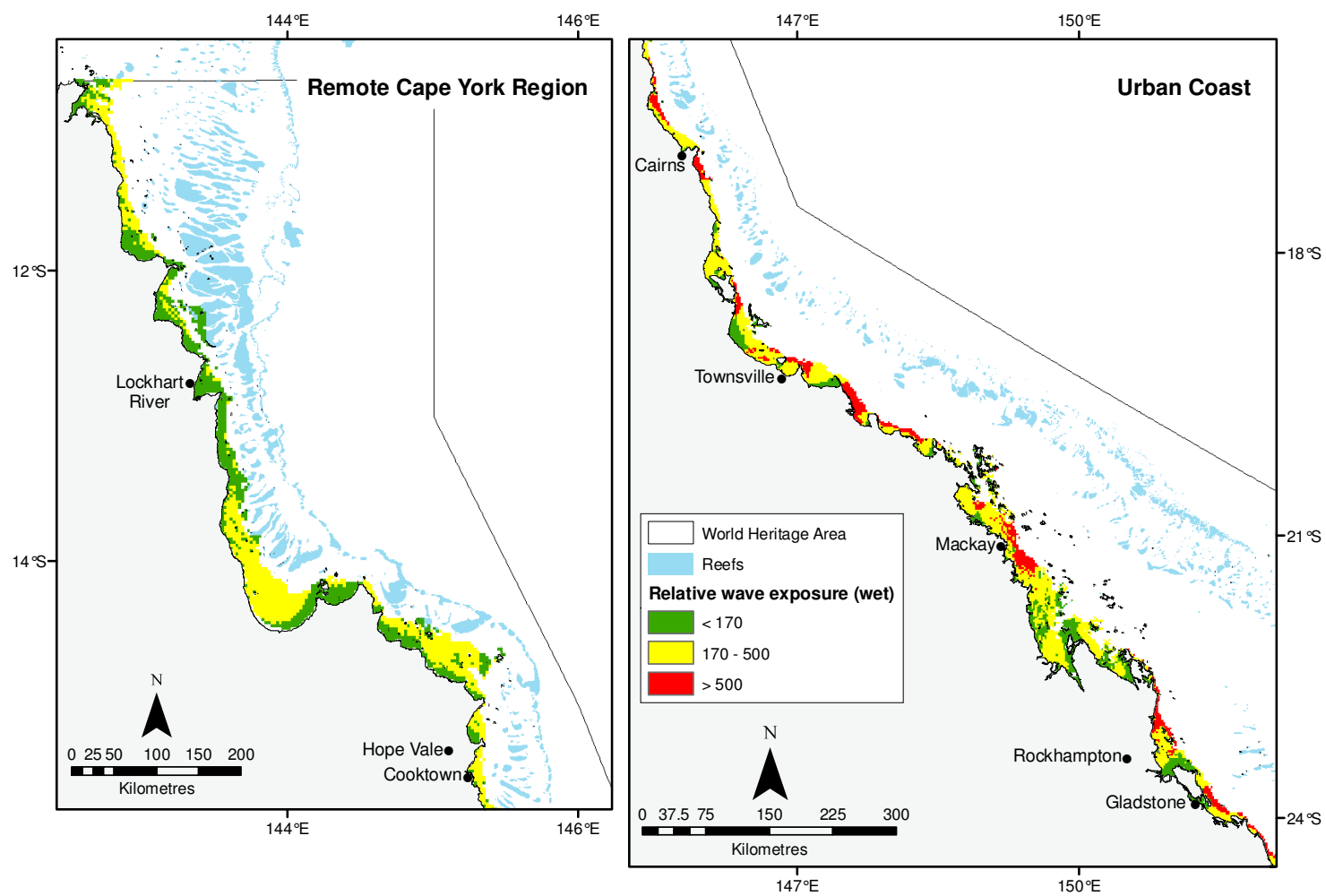


Figure A.7: Relative wave exposure index during the wet season

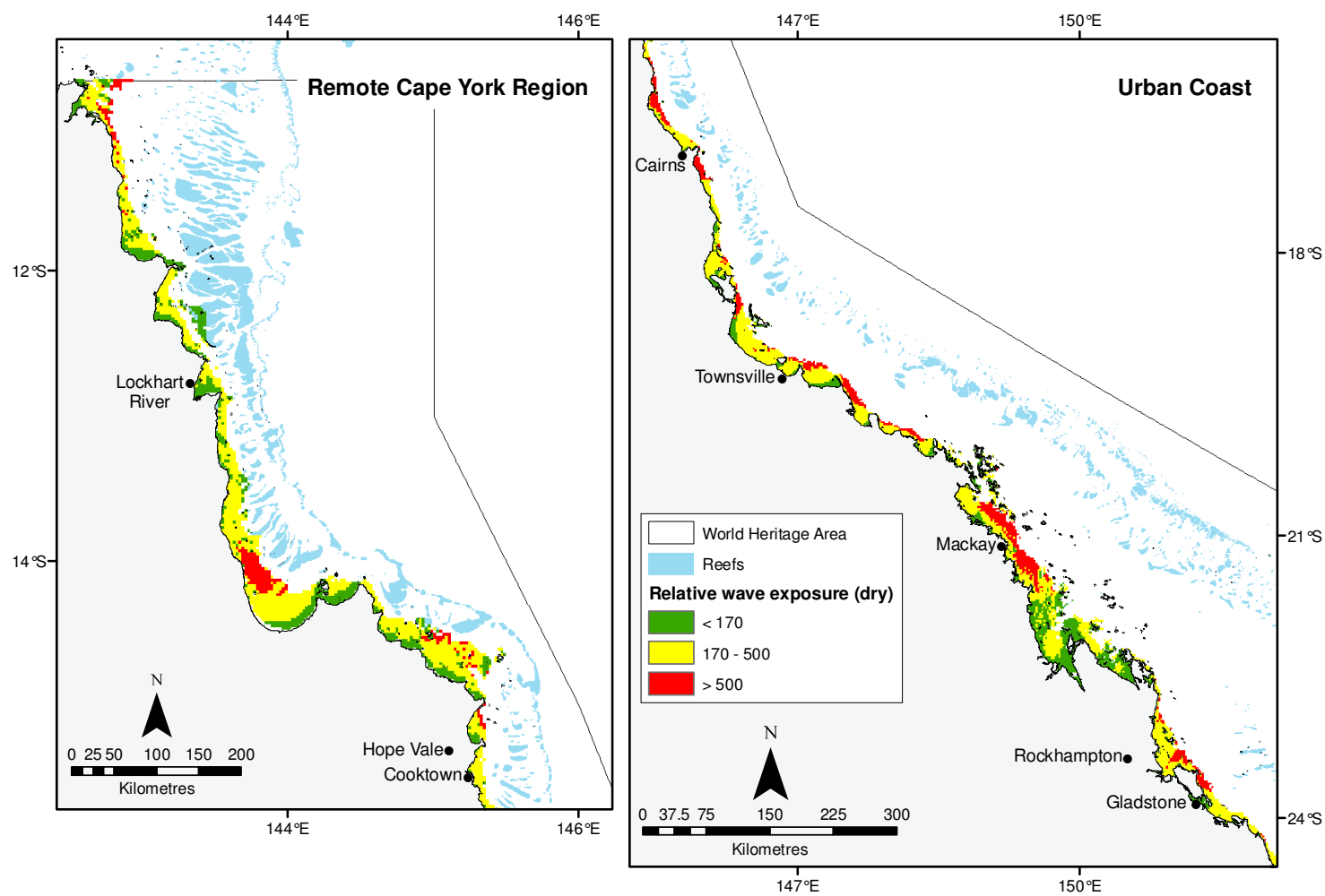


Figure A.8: Relative wave exposure index during the dry season

Appendix B

Conditional probability table of Bayesian belief network outlined
in Chapter 3

<i>Seagrass present</i>	<i>Seagrass absent</i>	<i>Bathymetry</i>	<i>REI</i>	<i>SST</i>	<i>Rivers</i>	<i>Tides</i>	<i>SedBasin</i>
0.000	1.000	low	low	low	present	high	absent
0.000	1.000	low	low	low	present	low	present
0.000	1.000	low	low	low	absent	low	absent
0.000	1.000	low	low	high	absent	medium	absent
0.000	1.000	medium	low	low	present	high	absent
0.000	1.000	medium	low	medium	absent	high	absent
0.000	1.000	medium	low	high	present	low	present
0.000	1.000	medium	medium	low	present	high	absent
0.000	1.000	medium	medium	low	present	medium	present
0.000	1.000	medium	medium	low	absent	low	absent
0.000	1.000	medium	medium	medium	present	high	absent
0.000	1.000	medium	medium	medium	present	medium	present
0.000	1.000	medium	medium	medium	absent	high	absent
0.000	1.000	medium	medium	high	present	low	present
0.000	1.000	medium	medium	high	absent	medium	absent
0.000	1.000	high	low	low	absent	low	present
0.000	1.000	high	low	medium	present	medium	absent
0.000	1.000	high	low	medium	absent	high	present
0.000	1.000	high	low	medium	absent	medium	present
0.000	1.000	high	low	high	present	high	present
0.000	1.000	high	low	high	present	medium	absent
0.000	1.000	high	low	high	present	low	absent
0.000	1.000	high	low	high	absent	high	present
0.000	1.000	high	medium	low	present	high	absent
0.000	1.000	high	medium	low	absent	medium	present
0.000	1.000	high	medium	medium	present	low	absent
0.000	1.000	high	medium	medium	absent	high	absent
0.000	1.000	high	medium	medium	absent	medium	absent
0.000	1.000	high	high	low	present	medium	absent
0.000	1.000	high	high	high	present	medium	absent
0.036	0.964	low	high	low	present	medium	absent
0.045	0.955	medium	high	low	present	low	absent
0.048	0.952	medium	medium	low	absent	high	absent
0.048	0.952	high	medium	medium	absent	low	absent
0.048	0.952	low	high	low	present	low	absent
0.059	0.941	high	medium	low	present	low	absent
0.071	0.929	medium	low	low	present	low	absent
0.074	0.926	medium	high	low	present	medium	absent
0.078	0.922	medium	low	low	absent	high	absent
0.083	0.917	medium	medium	low	present	low	absent
0.091	0.909	low	medium	medium	present	low	present
0.111	0.889	high	medium	low	absent	low	present
0.125	0.875	low	medium	medium	present	medium	present
0.148	0.852	medium	high	medium	present	medium	absent
0.158	0.842	low	medium	low	present	low	absent
0.163	0.837	low	medium	low	present	medium	present
0.167	0.833	high	medium	low	present	low	present
0.167	0.833	high	high	medium	present	medium	absent
0.182	0.818	medium	low	low	absent	medium	absent
0.200	0.800	low	medium	low	present	high	absent
0.222	0.778	low	low	low	absent	high	absent

<i>Seagrass present</i>	<i>Seagrass absent</i>	<i>Bathymetry</i>	<i>REI</i>	<i>SST</i>	<i>Rivers</i>	<i>Tides</i>	<i>SedBasin</i>
0.227	0.773	low	low	low	present	low	absent
0.240	0.760	medium	medium	medium	absent	medium	absent
0.250	0.750	low	medium	low	absent	high	absent
0.250	0.750	low	medium	medium	absent	medium	absent
0.262	0.738	medium	medium	low	absent	medium	absent
0.266	0.734	medium	medium	low	present	medium	absent
0.273	0.727	medium	low	high	present	medium	present
0.285	0.715	high	medium	low	present	medium	absent
0.286	0.714	medium	medium	medium	present	low	absent
0.298	0.702	medium	medium	high	present	medium	absent
0.300	0.700	high	low	medium	absent	low	present
0.327	0.673	low	medium	low	absent	medium	absent
0.333	0.667	low	low	medium	present	low	present
0.333	0.667	low	medium	low	absent	low	absent
0.333	0.667	high	low	low	present	high	present
0.333	0.667	low	medium	medium	absent	low	absent
0.333	0.667	high	low	high	present	low	present
0.333	0.667	high	medium	high	present	medium	absent
0.353	0.647	high	medium	medium	present	medium	absent
0.360	0.640	medium	low	high	present	medium	absent
0.368	0.632	medium	medium	medium	present	medium	absent
0.373	0.627	medium	low	medium	present	medium	absent
0.390	0.610	high	low	medium	present	low	present
0.407	0.593	low	low	high	present	medium	absent
0.420	0.580	low	medium	low	present	medium	absent
0.420	0.580	medium	low	low	present	medium	absent
0.429	0.571	medium	low	medium	absent	low	absent
0.438	0.563	medium	low	medium	present	low	absent
0.440	0.560	low	low	high	present	medium	present
0.444	0.556	low	medium	medium	present	low	absent
0.452	0.548	low	medium	high	present	medium	present
0.467	0.533	low	low	medium	absent	low	absent
0.474	0.526	low	low	low	present	medium	absent
0.524	0.476	medium	low	medium	absent	medium	absent
0.536	0.464	low	high	medium	present	medium	absent
0.549	0.451	low	medium	medium	present	medium	absent
0.556	0.444	low	low	medium	absent	high	absent
0.569	0.431	low	low	medium	present	medium	absent
0.598	0.402	low	low	low	absent	medium	absent
0.600	0.400	medium	low	high	absent	medium	absent
0.605	0.395	high	low	low	present	low	present
0.636	0.364	medium	medium	medium	absent	low	absent
0.667	0.333	low	low	high	present	low	present
0.667	0.333	medium	low	medium	present	medium	present
0.692	0.308	low	medium	high	present	medium	absent
0.714	0.286	low	low	medium	present	medium	present
0.784	0.216	low	low	medium	present	low	absent
0.833	0.167	medium	low	medium	present	low	present
0.857	0.143	low	low	high	absent	low	absent
0.889	0.111	medium	low	high	absent	low	absent
1.000	0.000	low	low	high	present	low	absent
1.000	0.000	medium	low	high	present	low	absent

<i>Seagrass present</i>	<i>Seagrass absent</i>	<i>Bathymetry</i>	<i>REI</i>	<i>SST</i>	<i>Rivers</i>	<i>Tides</i>	<i>SedBasin</i>
1.000	0.000	medium	medium	high	absent	low	absent
1.000	0.000	medium	high	high	present	low	present
1.000	0.000	high	low	low	absent	high	present
1.000	0.000	high	low	high	absent	low	present

Appendix C

Chapter 4 supporting figures

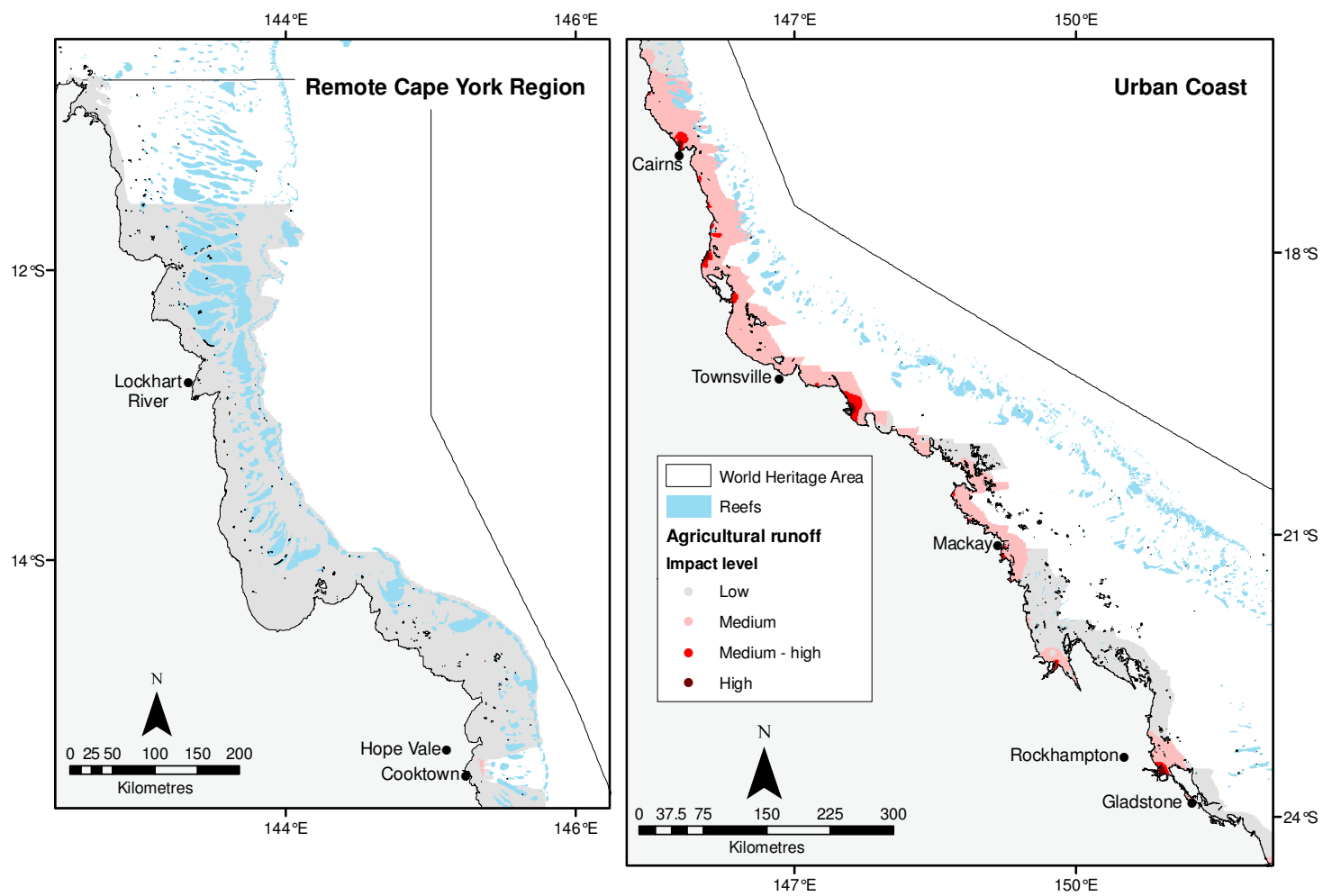


Figure C.1: Agricultural runoff (Australian Centre for Tropical Freshwater Research; Maughan et al. 2008)

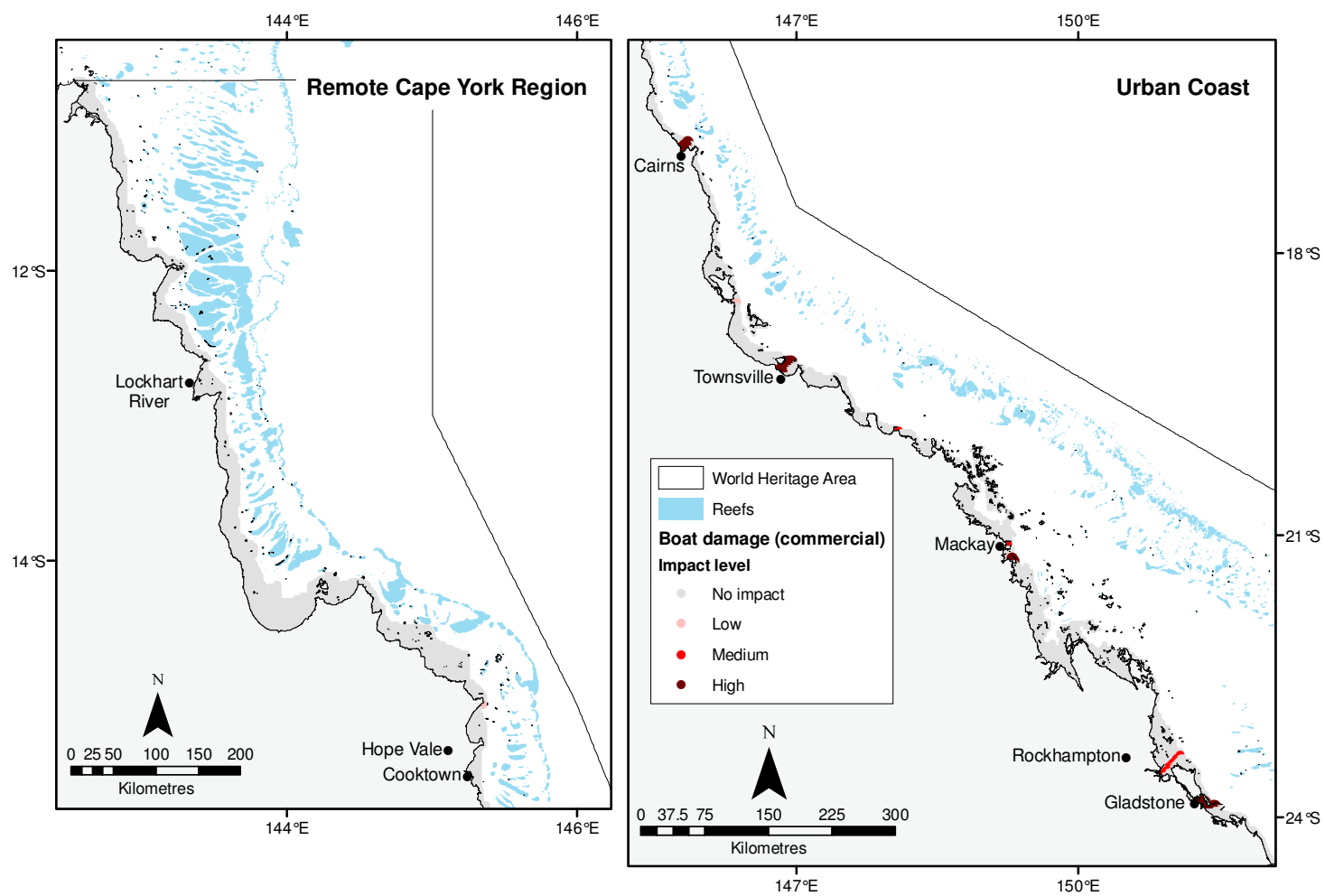


Figure C.2: Boat damage (commercial)

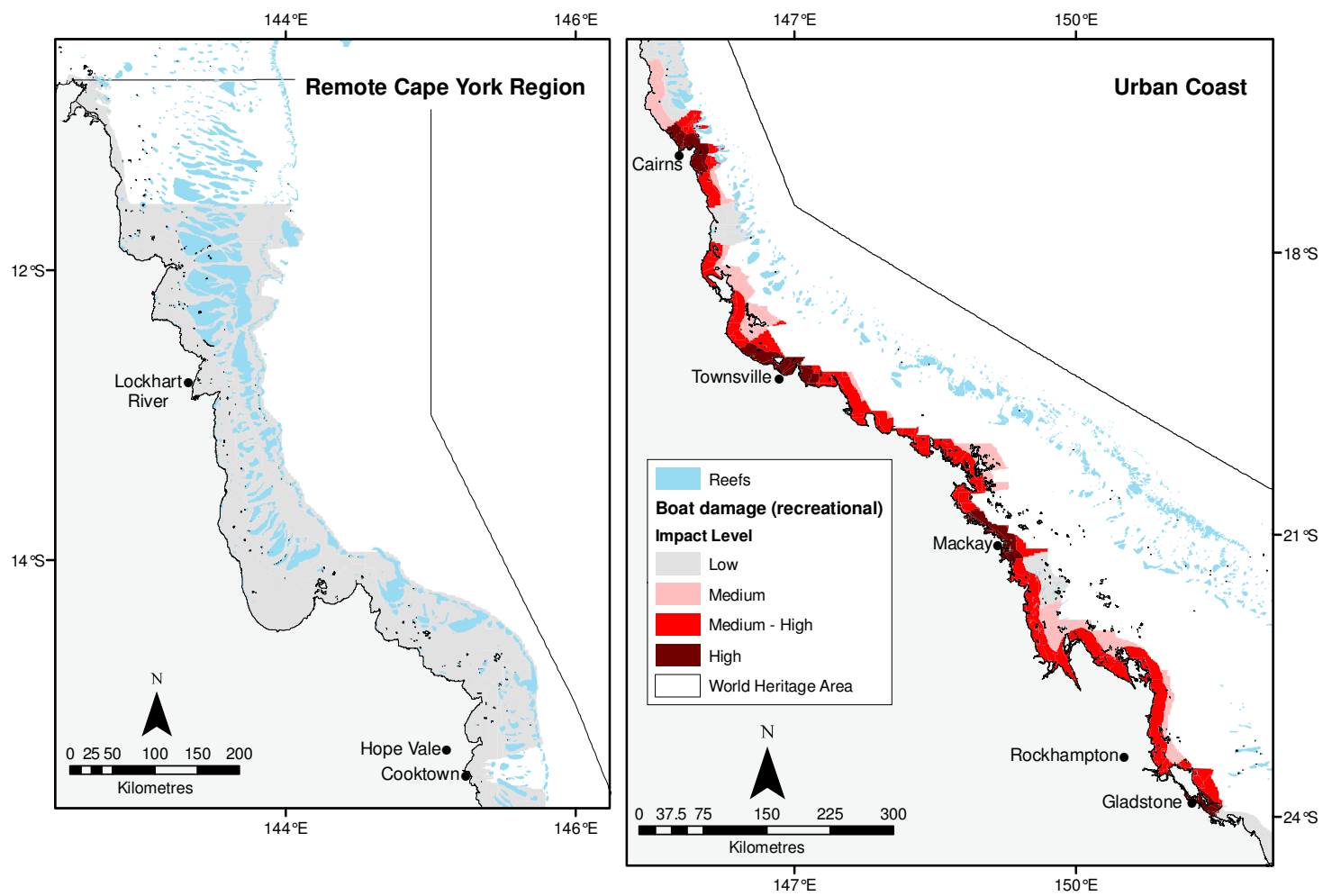


Figure C.3: Boat damage (recreational)

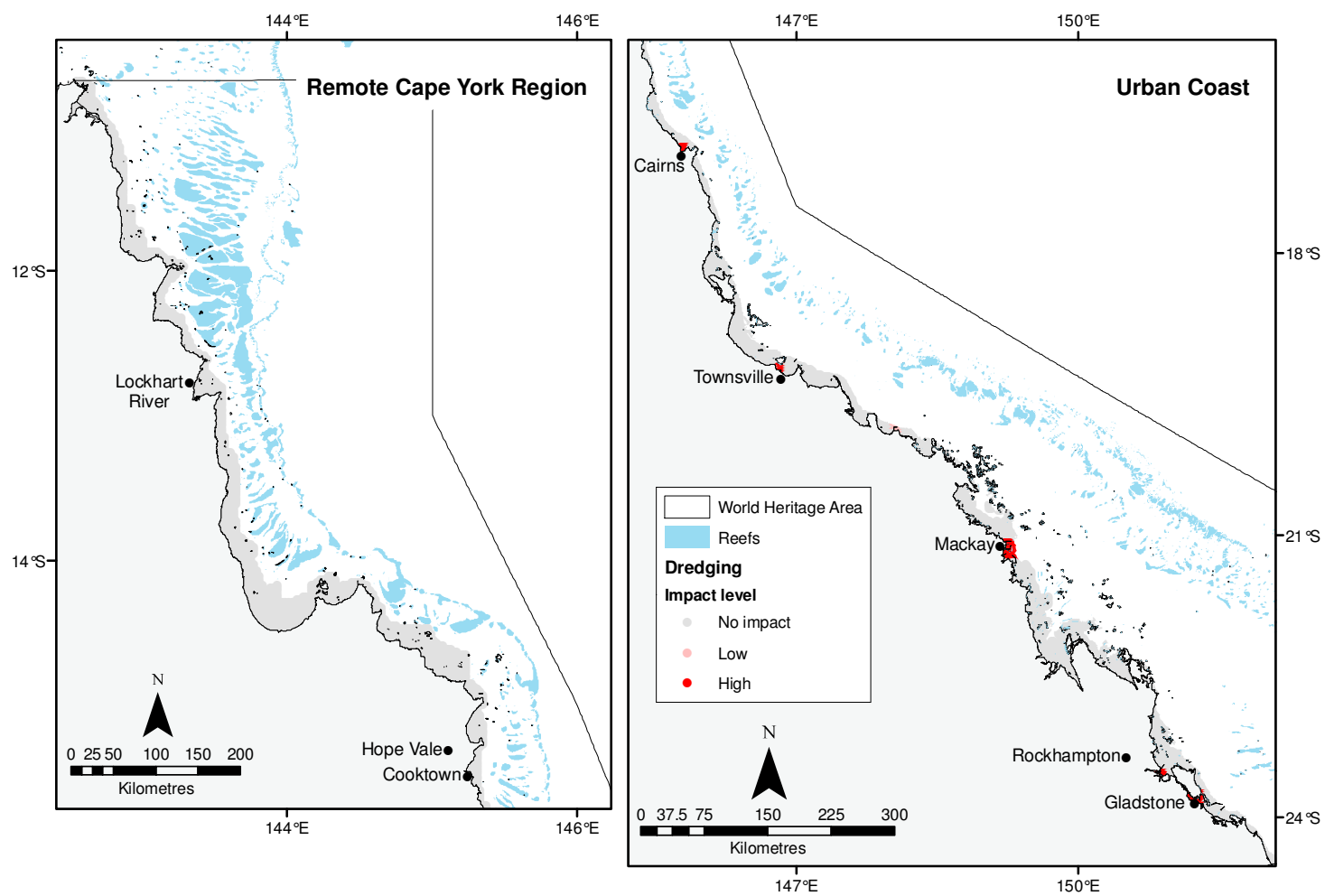


Figure C.4: Dredging

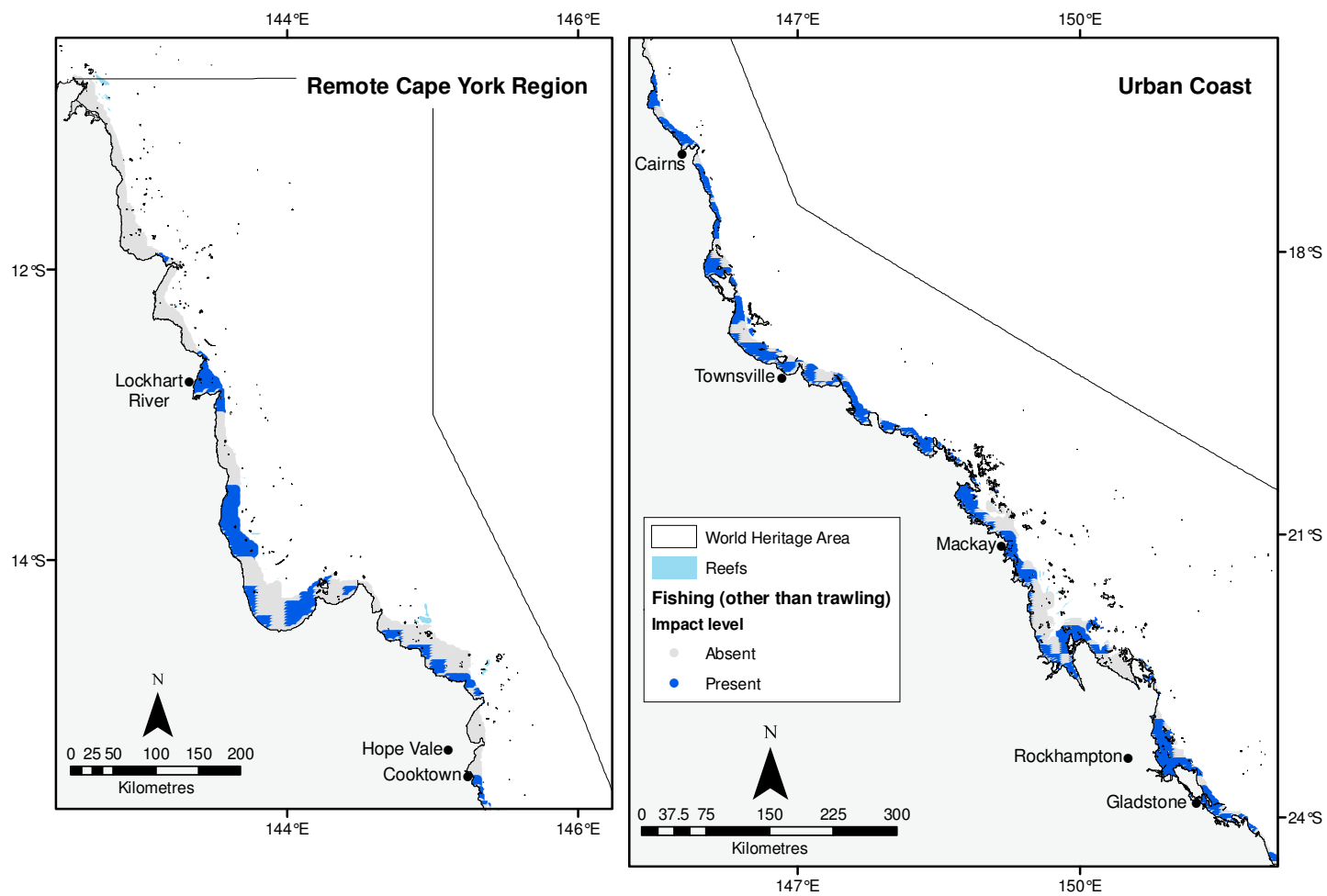


Figure C.5: Commercial fishing other than trawling (Queensland Primary Industries and Fisheries 2004 - 2005)

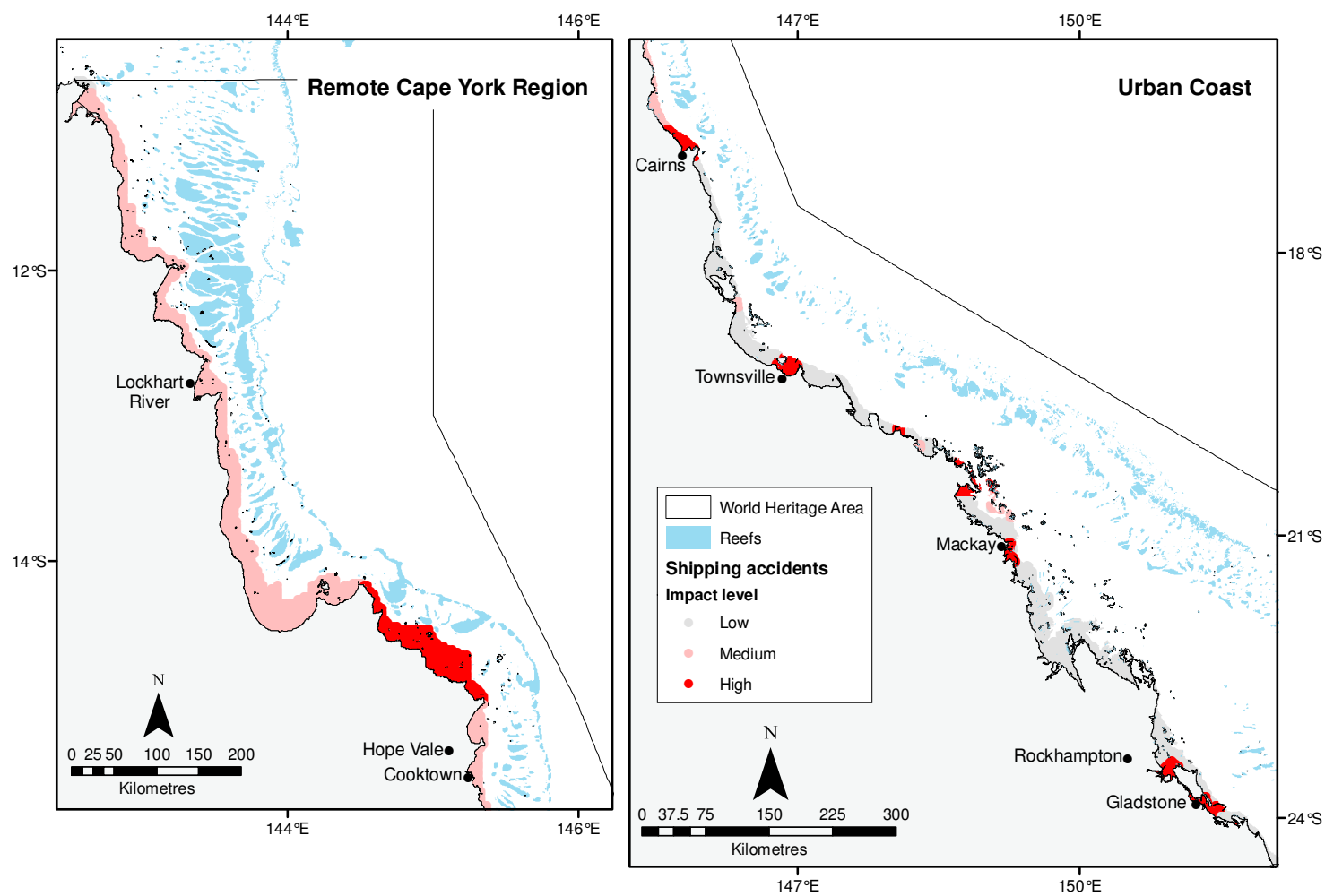


Figure C.6: Shipping accidents (Queensland Transport and the Great Barrier Reef Marine Park Authority 2000)

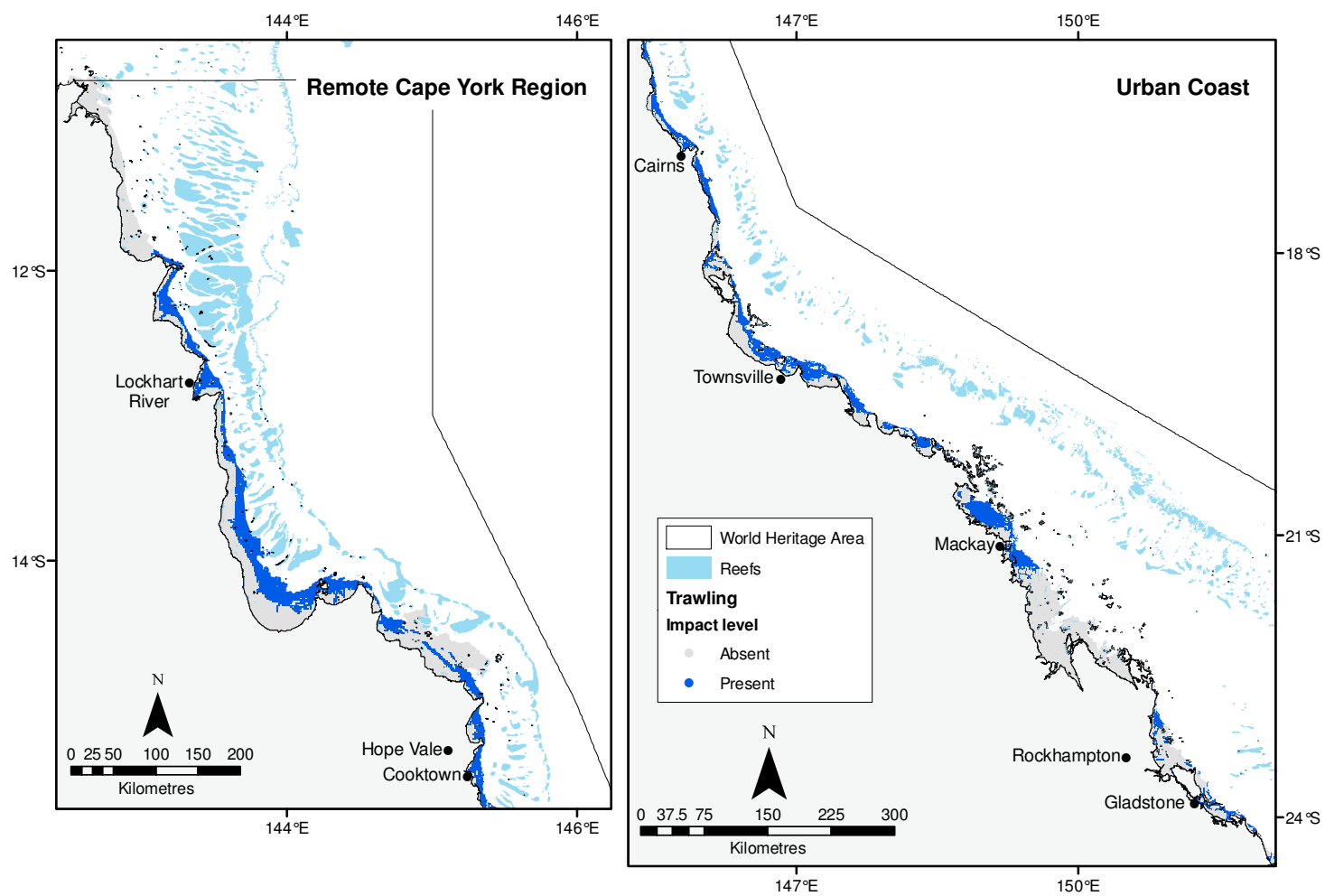


Figure C.7: Trawling (Queensland Primary Industries and Fisheries 2002 - 2005)

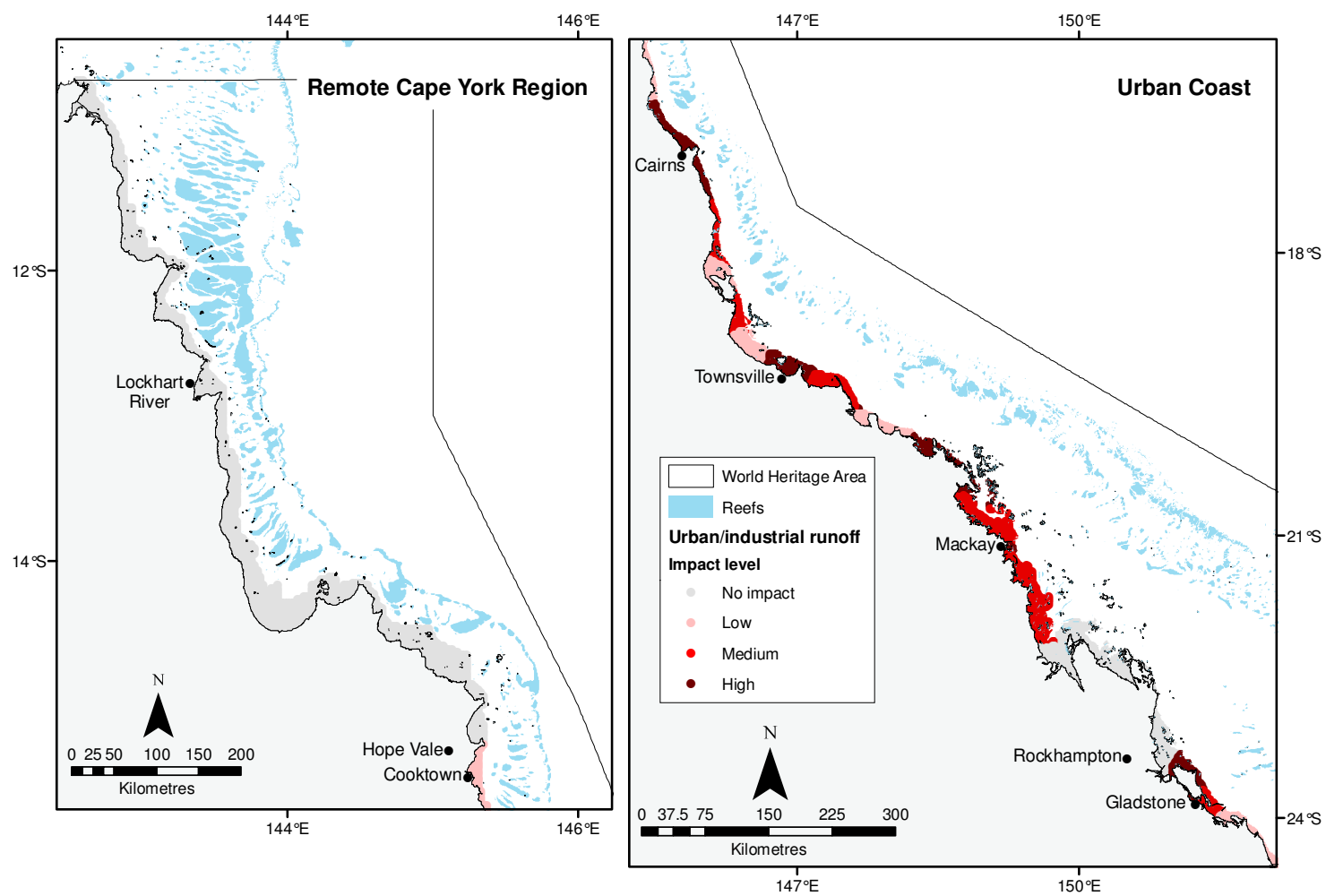


Figure C.8: Urban/industrial runoff

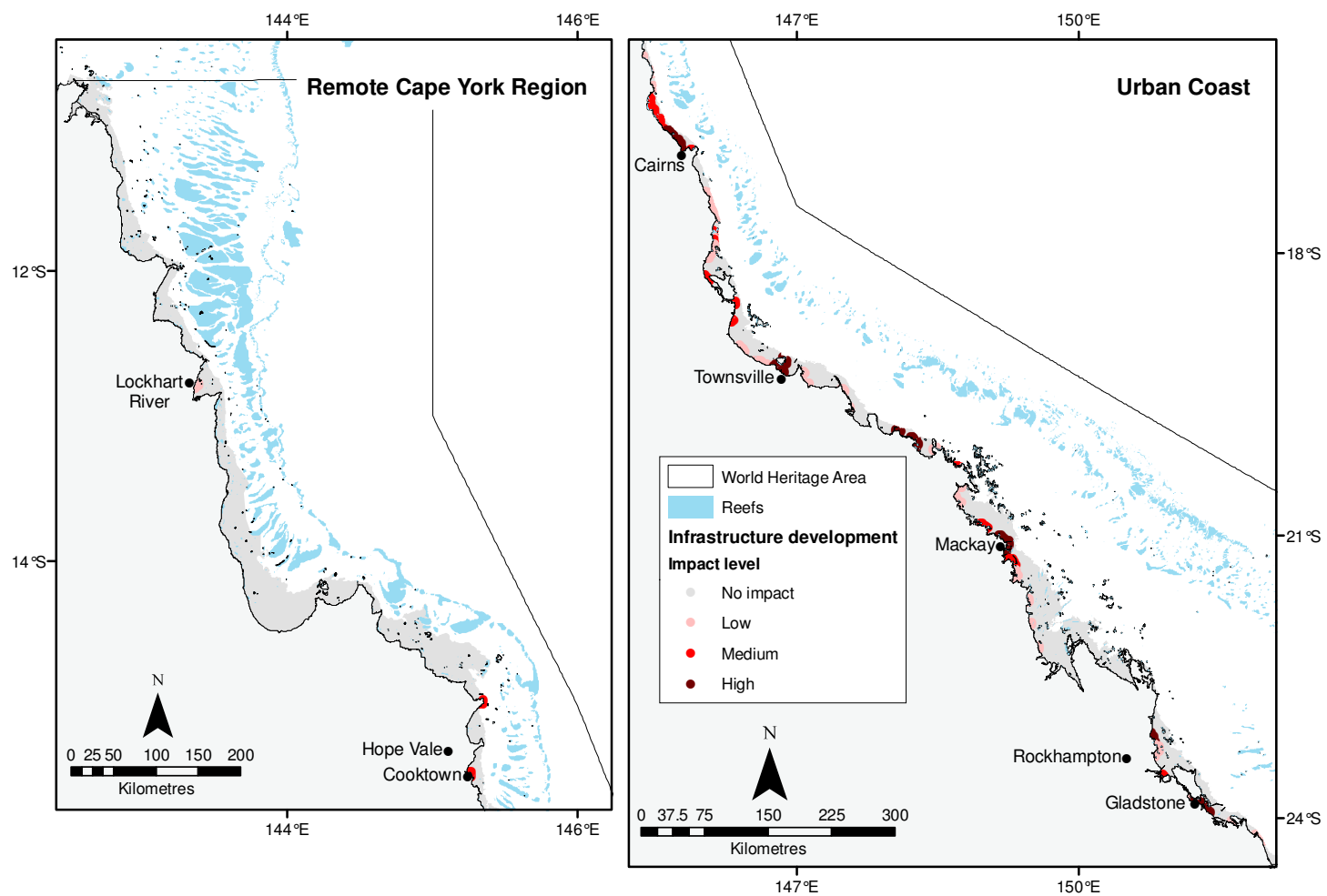


Figure C.9: Urban/port infrastructure development

Appendix D

Copy of the online survey used to collect information on rankings
for the five vulnerability factors from seagrass experts

Coastal Seagrasses Risk Assessment

1. Risk Assessment for Coastal Seagrass Habitats in the GBR - Consent Form

Thank you for participating in a survey on the risk to coastal seagrass habitats of the GBR.

We would like you to evaluate the vulnerability of coastal seagrass habitats of the GBR by considering the spatial scale, frequency, and functional impact of hazards to seagrass; the resistance of seagrass to disturbance by each hazard; the resilience (ie. recovery time) of seagrass following a disturbance; and the certainty of your estimates.

For this survey, we have defined coastal seagrass habitats as both sub-tidal and inter-tidal habitats in coastal waters > -15m (Carruthers et al. 2002).

Information concerning individuals is strictly confidential, and will not be published or released. Your participation is entirely voluntary.

This survey will take approximately 20 minutes of your time to complete.

Once you start the survey you cannot exit.

Information concerning individuals is strictly confidential, and will not be published or released. Your participation is entirely voluntary.

If you have any questions, please contact:

Alana Grech
School of Earth and Environmental Sciences
James Cook University
Townsville, Queensland 4810
Ph: +61 7 4781 5561
Email: alana.grech@jcu.edu.au

1. I have been informed about the nature of this survey, its confidentiality, and I agree to participate. I am aware that my participation is entirely voluntary.

If you do not wish to participate, please exit now by using the 'exit survey' button (top right-hand corner)

☐ Yes

2. Name of respondent

If you have any concerns regarding the ethical conduct of the study, please contact:
Tina Langford, Ethics Officer, Research Office, James Cook University, Townsville, Qld, 4811. Phone: 4781 4342, Tina.Langford@jcu.edu.au

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Coastal Seagrasses Risk Assessment

2. Before you start!

All questions require an answer and you must complete the survey now you have started.

As you fill out the survey we ask that you consider:

- (1) the CURRENT vulnerability of seagrass communities to anthropogenic threats
- (2) the AVERAGE manner in which a threat affects seagrass communities

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Coastal Seagrasses Risk Assessment

3. Vulnerability Measure: Scale

Scale is defined as the average scale at which a hazard event affects coastal seagrass habitats. Scale does not refer to the spatial scale at which the hazard exists (eg. a single pass of a demersal trawl may cover approximately 1–10 km², whereas demersal trawling occurs over the entire GBRWHA; the vulnerability measure Scale focuses on the first scale). Scale includes both direct and indirect impacts.

1. What is the average scale (in km²) at which the hazards affects coastal seagrass habitats?

	No threat	< 1	1 - 10	10 - 100	100 - 1,000	1,000 - 10,000	> 10,000
Agricultural runoff	☐	☐	☐	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐	☐	☐	☐
Dredging	☐	☐	☐	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐	☐	☐	☐
Trawling	☐	☐	☐	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

4. Vulnerability Measure: Frequency

Frequency describes how often discrete hazard events occur in coastal seagrass communities. For those hazards that occur as discrete events, frequency represents how often new events occur, not the duration of a single event.

1. How frequent do the hazard events occur in coastal seagrass habitats of the GBRWHA?

	Never occurs	Rare	Occasional	Annual or regular	Persistent
Agricultural runoff	☐	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐	☐
Dredging	☐	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐	☐
Trawling	☐	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

5. Vulnerability Measure: Functional Impact

Functional impact defines a hazards extent of influence within a coastal seagrass community. Some hazards may affect only a few species, whereas others affect the entire ecosystem. These differences are captured within the Functional impact vulnerability measure as it uses a ranking scheme ranging from the species to entire community level.

1. What is the functional impact of the hazards to coastal seagrass habitats of the GBRWHA?

	No impact	Species (single or multiple)	Single trophic level	> 1 trophic level	Entire community
Agricultural runoff	☐	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐	☐
Dredging	☐	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐	☐
Trawling	☐	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

6. Vulnerability Measure: Resistance

Resistance describes the average tendency of a species, trophic level, or community to resist changing its natural state in response to a hazard. The ranking scheme refers to the resistance of the ecosystem components that react to the hazard (ie. the functional level identified previously).

1. What is the resistance of coastal seagrass habitats to the hazards?

	No impact	High	Medium	Low
Agricultural runoff	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐
Dredging	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐
Trawling	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

7. Vulnerability Measure: Recovery Time

Recovery time is the average time required for the affected coastal seagrass community to return to its pre-hazard state. Because seagrass meadows are dynamic they need not (and are unlikely to) return to their exact pre-hazard condition to be deemed recovered. In this exercise, a recovered seagrass meadow is deemed within 20% of its starting point. For persistent hazards we consider recovery time following removal of the threat.

1. What is the recovery time (in years) of coastal seagrass habitats in the GBRWHA after a hazard event occurs?

	No impact	<1	1 - 10	10 - 100	>100
Agricultural runoff	☐	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐	☐
Dredging	☐	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐	☐
Trawling	☐	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

8. Vulnerability Measure: Certainty

Certainty is a qualitative measure of certainty that indicates the depth of knowledge used to determine vulnerability.

1. How certain are you in determining vulnerability to coastal seagrass habitats for the following list of anthropogenic hazards?

	Not at all certain	Low certainty	Moderately certain	High certainty	Very certain
Agricultural runoff	☐	☐	☐	☐	☐
Boat damage (commercial)	☐	☐	☐	☐	☐
Boat damage (recreational)	☐	☐	☐	☐	☐
Dredging	☐	☐	☐	☐	☐
Fishing (other than trawling)	☐	☐	☐	☐	☐
Shipping accidents (eg. oil spills)	☐	☐	☐	☐	☐
Trawling	☐	☐	☐	☐	☐
Urban/industrial runoff	☐	☐	☐	☐	☐
Urban/port infrastructure development	☐	☐	☐	☐	☐

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Coastal Seagrasses Risk Assessment

9. Thank you!

Thank you for participating in the survey.

We know how valuable your time is, and we appreciate the effort you have made.

We look forward to returning the results of the survey to you, and working with you again in the future.

Alana Grech and Rob Coles

1. Was this survey easy to understand and complete?

- ☐ Yes, all of the time
- ☐ Yes, sometimes
- ☐ No, not very often
- ☐ No, not at all

2. Any comments or suggestions?

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Done

Appendix E

Chapter 7 supporting figures

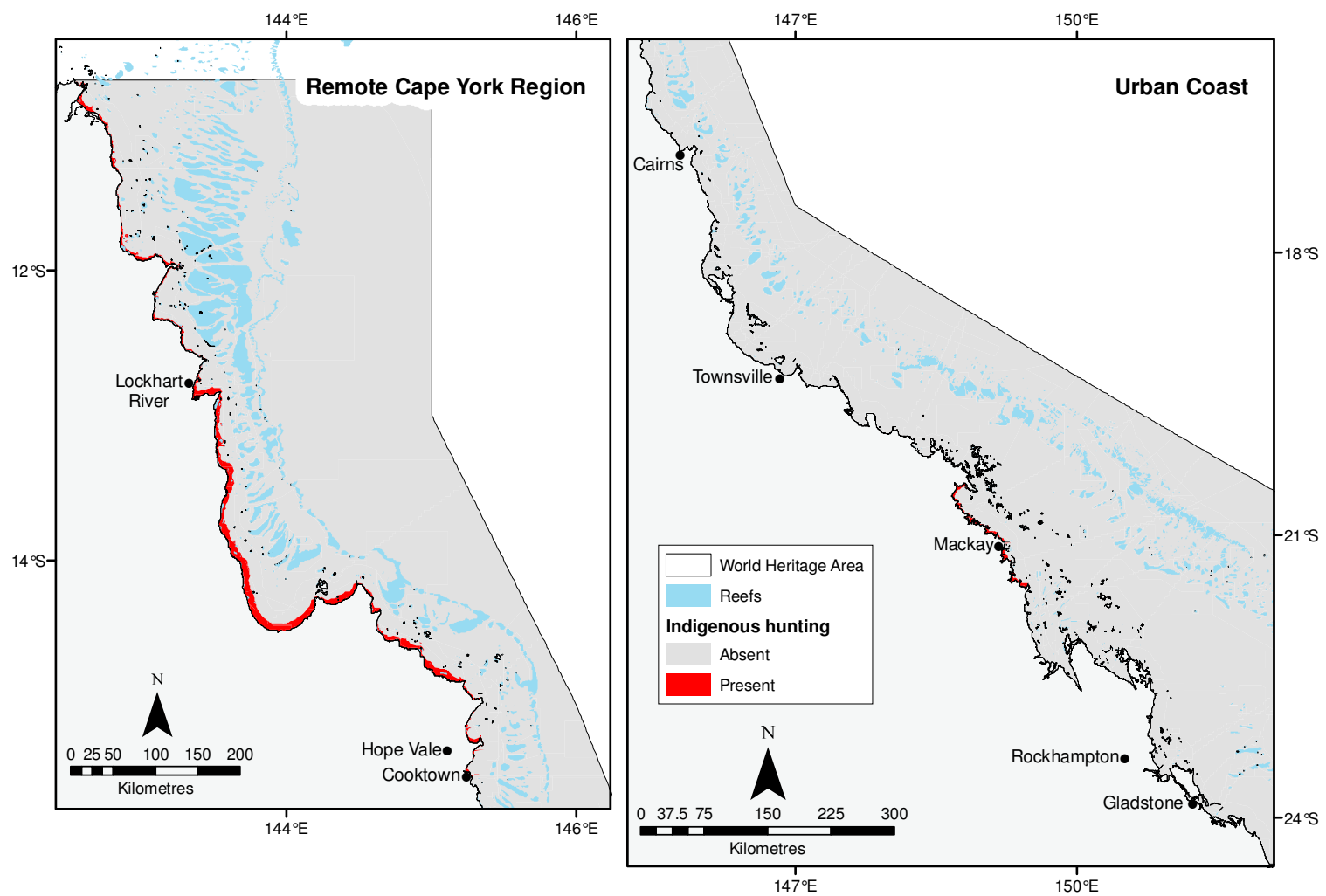


Figure E.1: Indigenous hunting

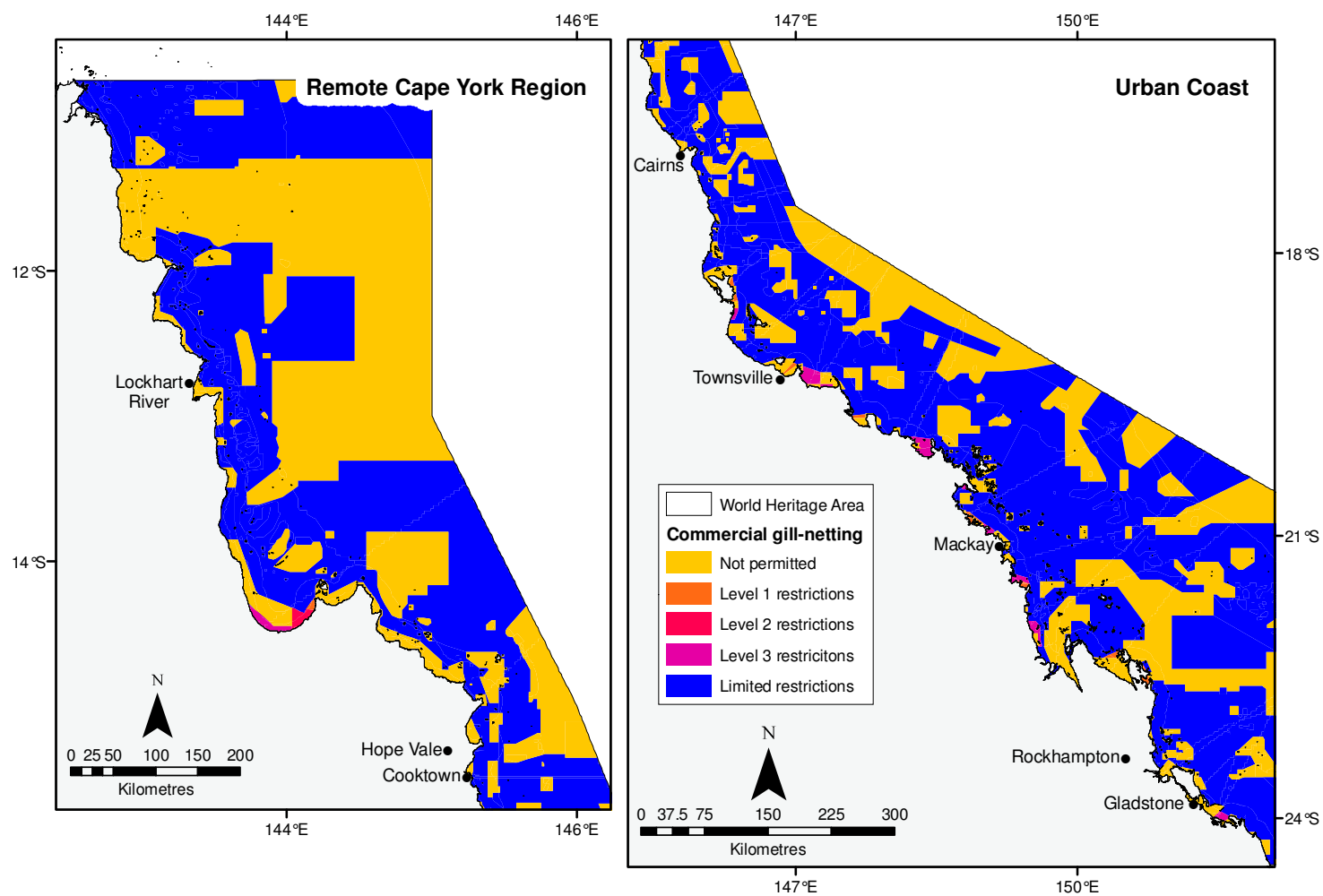


Figure E.2: Commercial gill-netting (Great Barrier Reef Marine Park Authority 2004; Queensland Primary Industries and Fisheries 2004)

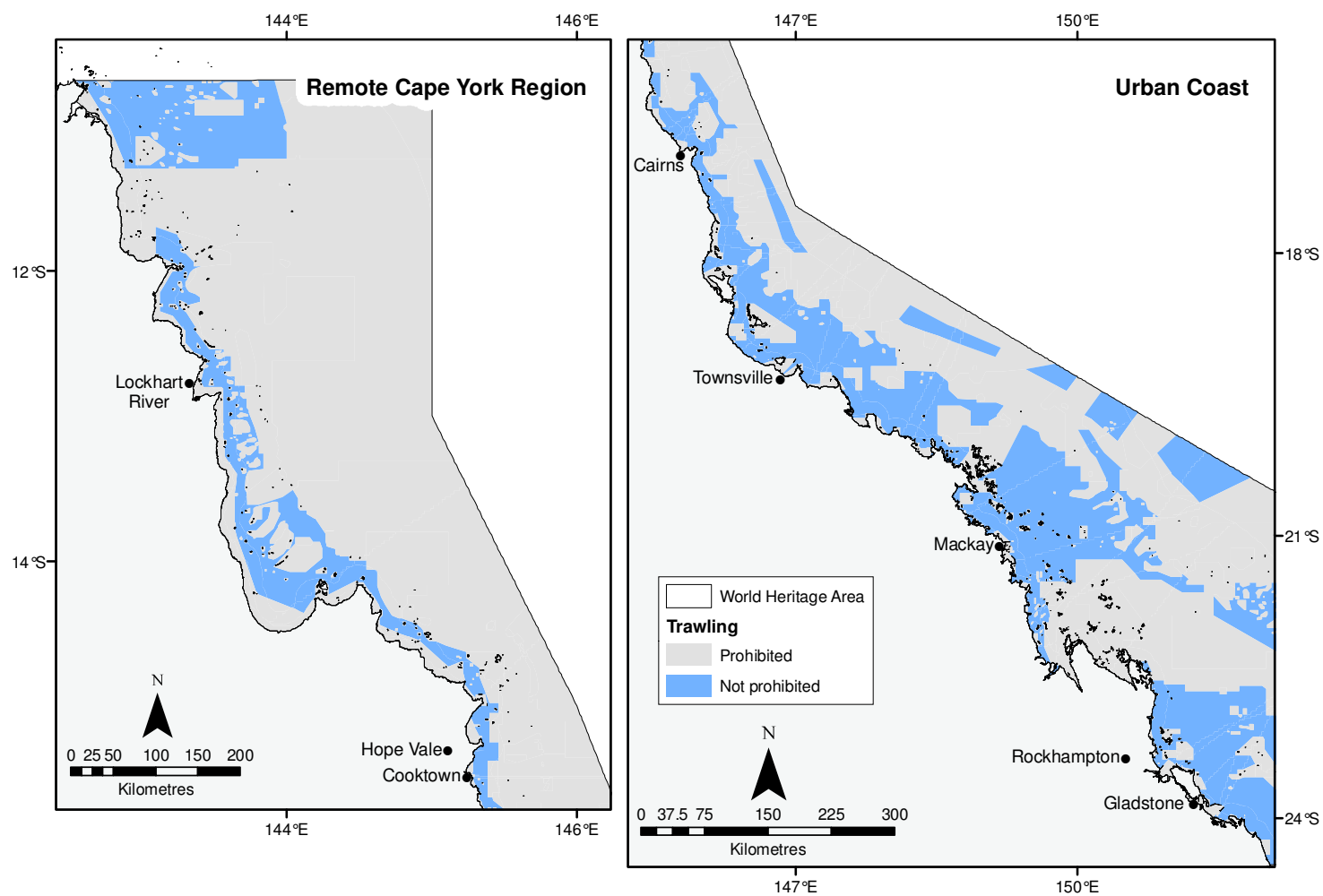


Figure E.3: Trawling (Great Barrier Reef Marine Park Authority 2004)