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Chapter 4

SEAGRASS MEADOWS OF THE WAKATOBI NATIONAL PARK

Richard K.F. Unsworth
Marine Ecology Group, Northern Fisheries Centre, Department of Employment, Economic Development and Innovation, Queensland Government, PO Box 5396, Cairns, QLD 4870, Australia

ABSTRACT

Seagrasses are abundant marine plants that form large areas of shallow water marine habitat throughout the Wakatobi National Park (WNP). Although there are only a small number of seagrass species in the region, the habitat they create contains a diverse and abundant fauna that has enormous ecological and economic importance to the marine park. The variability of these seagrass meadows and the extent to which different floral species develop has been found to have a large impact upon their value as a habitat for invertebrate and fish assemblages in the WNP. Seagrass meadows in the WNP support over 180 species of fish, many of which co-utilise coral reef and mangrove habitats on a daily basis. In fact, the interaction between these three habitats has a large influence on the abundance, diversity and trophic structure of seagrass fish assemblages in the WNP. Seagrass meadows in the WNP are heavily utilised for their abundant resources, particularly their fish assemblages and their large invertebrates exposed at low tide. Like other marine ecosystems worldwide, these resources are under increasing threat of overexploitation, particularly as a result of increasing local human population size resulting in a greater need for more efficient fishing methods. Seagrass meadows of the WNP require conservation and fisheries management, not just to protect biodiversity, but also for the sustainable development of protein supply within the local region.

INTRODUCTION TO SEAGRASS BEDS

Seagrasses are the only marine representatives of the Angiospermae and belong to the order Helobiae, in two families: Potamogetonaceae and Hydrocharitaceae (Den Hartog, 1970). These plants are rhizomatous (they have stems extending horizontally below the sediment surface) and are modular, composed of repeating units (ramets) that show clonal growth (Hemminga and Duarte, 2000). In contrast to other submerged marine vegetation (e.g. macro algae), seagrasses flower, develop fruit and produce seeds (Ackerman, 2006). They also have true roots and internal gaseous and nutrient transport systems (Hemminga and Duarte, 2000). Seagrass can be patchy, but more often forms large areas of vegetation, sometimes over 10,000km^2 in size (Hemminga and Duarte, 2000). Within many tropical coastal seas, seagrass meadows are a dominant feature of the inter-tidal and are often abundant in subtidal shallow waters.

With their extensive root-rhizome system, and well-developed canopy, seagrass meadows provide many important ecosystem services (Duffy *et al.*, 2005). These services have been highlighted for the economic value they provide, particularly the role they have in nutrient cycling (Costanza *et al.*, 1997). Seagrass meadows also provide other vital services including nursery refugia (Beck *et al.*, 2003), sea defence (Christiansen *et al.*, 1981), water treatment (Hemminga and Duarte, 2000), and are important in the cycling of important global atmospheric gases, particularly CO_2 (Duarte and Cebrian, 1996). Although seagrass meadows cover only 0.15% of the global oceans (Hemminga and Duarte, 2000), they represent 1.13% of the total marine primary production, and potentially act as a sink for CO_2 (Duarte and Cebrian, 1996). They are also thought to have an important, yet poorly understood, role in the cycling of dimethylsulfide (DMS) (Lopez and Duarte, 2004). Of vital importance to many human populations is the function of seagrass meadows as biologically important and economically valuable fishing grounds (Watson *et al.*, 1988; de la Torre-Castro and Rönnbäck, 2004). Seagrass meadows comprise a significant proportion of Indonesia's coastal marine habitats, and are therefore extremely important, with estimates suggesting they cover an area of at least 30,000 km^2 (Kuriandewa *et al.*, 2003).

Together with marine floral and faunal taxa, seagrasses have their centre of generic richness in the Indo-West Pacific (Fortes, 1991; Allen and Werner, 2002; Bell and Smith, 2004). This creates a series of inter-connected habitats and ecosystems that support many complex trophic interactions (Unsworth *et al.*, 2008). Within Indonesian seagrass meadows, 12 floral species have been identified, although individual meadows are commonly dominated by only three or four species (Kuriandewa *et al.*, 2003). The most abundant seagrass species in Indonesia are *Thalassia hemprichii* (Ehrenberg) and *Enhalus acoroides* (L.f.) Royle (Neinhuis *et al.*, 1989; Kiswara, 1996).

SPATIAL COVERAGE

Seagrass meadows are a major feature of the intertidal and subtidal coastal waters of the WNP, and are abundant in many areas, particularly the major islands, as well as within the atolls and around the outer islands of the marine park (Figure 12). To date no comprehensive survey of the spatial extent of seagrass meadows of the WNP has been conducted, and spatial information is based solely upon the author's personal observations, anecdotal information and basic observations by non-governmental organizations (Pedju *et al.*, 2004). The World Atlas of Seagrasses estimates the WNP to contain between 100-1000ha of seagrass (Kuriandewa *et al.*, 2003). Seagrasses make up a large proportion of marine habitats in the WNP (Pedju *et al.*, 2004), and therefore 100-1000ha is probably a gross underestimation.

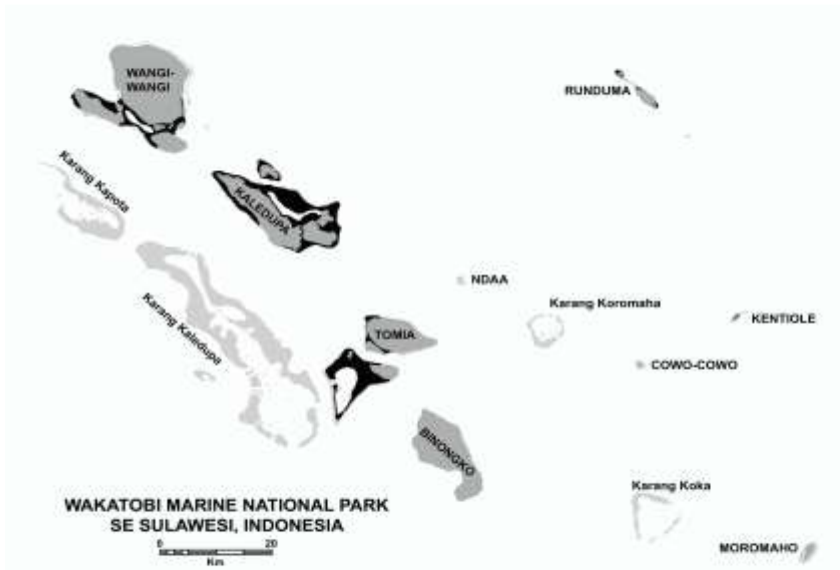

Figure 12. Distribution of seagrass (black areas denote seagrass) within the Wakatobi Marine National Park (adapted from Pedju *et al.*, 2004). Note minor coverage of seagrass on Kentiole and Runduma.

Understanding the spatial extent of seagrass meadows would be a first major step towards managing their associated environmental risk from natural and anthropogenic factors. Binongko Island is the only major land mass thought not to be surrounded by seagrass, but the presence of mangrove indicates that seagrass may be present, but only within mangrove channels. Mangrove channels in other parts of the national park commonly contain seagrass, particularly *Enhalus acoroides* (L.f.) Royle.

Seagrass meadows in the WNP are dominated by two floral species, *Thalassia hemprichii* and *Enhalus acoroides* (Unsworth *et al.*, 2007e). However, a further seven species (*Thallasodendron cilliatum*, *Halodule uninervis*, *Halophila spinulosa*, *Halophila ovalis Halophila decipiens*, *Cymodocea rotundata*, *Syringodium isoetifolium*) have been recorded throughout the park.

In common with research throughout Indonesia and the Indo-Pacific (Neinhuis *et al.*, 1989; Kiswara, 1996; De Troch *et al.*, 2001), *Halophila spp.* and *Halodule spp.* are found at the colonising edges of seagrass beds as the ruderal species, while the slower growing *Cymodocea sp*., *Thalassia hemprichii* and *Enhalus acoroides* exist as the climax species. Within subtidal seagrass beds, *Thallasodendron cilliatum* and *Syringodium isoetifolium* are also often part of the climax community composition in the WNP. Deep-water seagrass has also been observed by the author within the marine park, with *Halophila ovalis* and *Halophila spinulosa* commonly found within small patches in sandy deep-water reef areas (30-50m depth). Such deep-water seagrass meadows are poorly understood, however their distribution has been recorded in many locations such as the Caribbean and NE Australia (Carruthers *et al.*, 2002).

SEAGRASS COMMUNITY DYNAMICS AND HERBIVORY

Seagrass species within tropical intertidal areas of the Indo-Pacific exhibit vertical zonation down the shore (Neinhuis *et al.*, 1989; Björk *et al.*, 1999; De Troch *et al.*, 2001). Similar patterns have also been found within seagrass meadows of the WNP (Unsworth, 2007). Unlike temperate rocky shores, which are commonly dominated by macroalgae, the factors controlling distribution patterns have been poorly investigated. Information regarding the flora of intertidal habitats mostly arises from macroalgal research (Dring, 1986; Falkowski and Raven, 1997). For example, it is widely accepted that light is one of the main abiotic factors regulating macro-algal distribution within the intertidal zone (Gerard, 1988; Hanelt *et al.*, 1997), yet the influence of light is poorly understood in a seagrass context (Ralph *et al.*, 2007). Although tropical seagrasses often fill a similar ecological niche to macroalgae, seagrasses are higher plants and are therefore likely to respond differently to environmental regulatory processes. Within Indo-Pacific seagrass habitats, the environmental regulatory processes controlling photosynthetic productivity have received little attention (Björk *et al.*, 1999; Campbell *et al.*, 2006). Central to the continuing viability of seagrass meadows is primary production. It is important that researchers consider the primary factors controlling the distribution of seagrasses due to the intrinsic importance of seagrass primary productivity, particularly in supporting associated fauna.

By examining seagrass photosynthesis in unison with an assessment of the community characteristics and growth, Unsworth (2007) provided an important baseline of information on the influence of seagrass photoacclimation upon seagrass community dynamics. This research found that the mechanisms of photoacclimation in tropical Indo-Pacific seagrass species may confer some competitive advantage to individual species of seagrass, enabling them to successfully inhabit specific environmental conditions.

This study also demonstrated that *C. rotundata* and *H. ovalis* were the most abundant species within the upper-shore environment of the WNP, which is characterised by high light and temperature stress on a daily basis (Unsworth, 2007). Evidence was provided to suggest that the presence of *C. rotundata* was most likely the result of processes of photo-acclimation, conferring a competitive advantage over the other dominant large leaved species (*T. hemprichii*). Although the abundance of *H. ovalis* is highest at the limits of the upper shore, observed morphology and reduced temperature tolerance suggested that it may be physiologically more successful within deeper waters (Duarte *et al.*, 2000; Unsworth, 2007). In many environments *H. ovalis* can quickly become out-competed with respect to light by larger species due to its small size (Duarte *et al.*, 2000). Within the upper shore, a lack of large leaved seagrass species may reduce light competition releasing *H. ovalis* from competition (Unsworth, 2007).

Although *E. acoroides* is not abundant on the upper shore of the WNP, and has a deeper sub-tidal distribution, it remained photosynthetically successful throughout the middle of the day when photosynthetically active radiation (PAR) and hence light and heat stress is at its peak. Furthermore, its dark-adapted photosynthetic rate changed very little between different depth environments (Unsworth, 2007) suggesting an adaptation to high light and heat stress. Data from Zanzibar corroborated this resilience to stress and found *E. acoroides* to have the highest desiccation resistance of seven Indo-Pacific species (Björk *et al.*, 1999). This suggests that other factors known to influence seagrass distribution (e.g. sediment depth and nutrient availability), may be of greater importance than the light and temperature regime (Duarte *et al.*, 2000).

Tropical seagrass meadows are commonly dynamic, undergoing large temporal annual and seasonal changes (Mellors *et al.*, 1993; McKenzie, 1994). For example, in a Northern Australian meadow, seagrass biomass has been recorded to reduce to <10% of its 15 year average as a result of natural climate variability (Rasheed and Unsworth, 2009). Such changes have enormous implications to dependent fauna and hence key fisheries, as seagrass loss in many areas has resulted declining fisheries catch (McArthur and Boland, 2006). In the WNP, temporal seagrass dynamics remain poorly understood and require investigation.

Despite being a globally important source of primary production, the use of seagrass biomass by herbivores has previously been thought to be under-utilised (Nienhuis and Van Ierland, 1978; Thayer *et al.*, 1984; Nienhuis and Groenendijk, 1986). Research in the WNP has found that a seagrass meadow within an Indo-Pacific setting may represent a large food source for herbivorous fish, particularly scarids (Unsworth *et al.*, 2007d), which supports previous work in the Caribbean and Mediterranean seas (Kirsch *et al.*, 2002; Tomas *et al.*, 2005). In the WNP scarids not only took up carbon into the food chain through direct consumption, but also made an important functional contribution to the detrital food chain through the discarding of large amounts of seagrass material as a by-product of feeding. This seagrass biomass may then be exported from these habitats or decay *in situ* (Unsworth *et al.*, 2007d). Scarids, like many fish within seagrass beds of the WNP, are vital food sources to local human populations and are regularly caught by most types of fishing gear (Unsworth *et al.*, 2007d). If seagrass meadows are to remain productive sources of food and economic wealth, conservation and fisheries management should focus on considering the functional roles of particular faunal groups. The balance of herbivory relative to floral and algal growth has been disrupted by fishing activities within Caribbean coral reefs, often resulting in reef degradation (Mumby *et al.*, 2007; Mumby and Hastings, 2008). Although a rate of scarid herbivory was quantified within the WNP, the ecological functions of this process remain poorly understood, which is indicative of the poor general understanding of the functional ecology of seagrass meadows throughout the Indo-Pacific region. More detailed information is therefore vital for the future management and conservation of these systems.

INVERTEBRATE COMMUNITIES

Seagrass meadows within the WNP contain abundant and diverse invertebrate life. This is illustrated by the presence of an up-market dive tourism operator in the park advertising these seagrass meadows as locations to observe exotic creatures such as blue-ringed octopus (*Hapalochlaena lunulata* Quoy and Gaimard, 1832) and Razorfish (*Centriscinae spp.*). Unfortunately little of this invertebrate diversity has been documented by the scientific literature. The only detailed ecological study of seagrass invertebrates in the WNP have been conducted upon the caridean shrimp fauna. Seventeen species of shrimp have been recorded within seagrass beds of the WNP, with a mean density of 83 ± 8.6 shrimp m^{-3} (Unsworth *et al.*, 2007b). The shrimp assemblage was numerically dominated by *Chlorocurtis jactans*, which was also the most frequently sampled species, occurring in 98% of samples (Unsworth *et al.*, 2007b). At the family level, the fauna was dominated by the nocturnally active

Processidae, whilst the Hippolytidae were not as numerically abundant as they are in the Caribbean. Additional invertebrate studies have been conducted within the WNP, but these were carried out as fisheries surveys, and hence ignore species that are not economically important. The inter-tidal reef flat and seagrass meadows of the WNP were fished at low tide for a total of 43 invertebrate species, mostly comprising Holothurians, but also gastropods, sea urchins and bivalves (Robinson, 2002).

FISH ASSEMBLAGES

Seagrass meadows support abundant epiphytes and associated fauna, and consequently high fish and invertebrate species richness (Blaber and Blaber, 1980; Johnson and Johnstone, 1995; Nagelkerken *et al.*, 2000; Jackson *et al.*, 2001). The Indo-Pacific bioregion, particularly the central areas incorporating Indonesia, Papua New Guinea, Philippines and Malaysia contain the world's highest marine speciation (Briggs, 2005), including fish fauna (Allen and Werner, 2002). Fish assemblages associated with tropical seagrass meadows often contain more than 80 species within only a few hectares (Hemminga and Duarte, 2000; Gell and Whittington, 2002; Dorenbosch *et al.*, 2005). In Indonesia, a total of 360 species of fish were recorded in four separate studies of seagrass meadows conducted between 1977 and 1991 (Tomascik *et al.*, 1997). Other studies within the region have also revealed high species richness of fish. For example, Gell and Whittington (2002) found 249 species of fish from 62 families within the seagrass beds of Northern Mozambique. Despite the high number of seagrass fish species within Indonesia, fish assemblages are commonly dominated by Siganidae (Rabbitfishes), particularly *Siganus canalisulatus* (Kuriandewa *et al.*, 2003).

Seagrass meadows are utilised by fish species for a variety of their ecological functions. Fish use of Indonesian seagrass meadows has been classified into four principal species assemblages (Tomascik *et al.*, 1997) based on an adaptation of the scheme described by Bell and Pollard (1989). These comprise permanent seagrass residents (e.g. Chequered cardinal, *Apogon margaritophorus*), permanent residents that spawn outside the seagrass (e.g. *Halichoeres argus*), juvenile seagrass residents (e.g. *Siganus canalisulatus*), and transients or residents seeking food or shelter. Due to the interaction of seagrass fish assemblages with nearby habitats, they have also been classified based on their habitat usage (Unsworth *et al.*, 2008). Seventeen of the 40 most abundant seagrass fish species in the WNP were associated with seagrass meadows that were in close proximity to mangroves and distant from reefs, while 11 species were associated with seagrass beds close to both mangrove and reefs, and five species with seagrass close to reefs (Unsworth *et al.*, 2008).

As detailed in Table 6, a total of 188 species of fish have been positively identified in the seagrass beds of the WNP in studies utilising a range of techniques such as seine netting and underwater visual census (Unsworth *et al.*, 2007a; Unsworth *et al.*, 2007e; Unsworth *et al.*, 2008; Unsworth *et al.*, 2009b; Unsworth *et al.*, 2009c). This is a minimum estimate of the number of fish species residing within Wakatobi seagrass meadows, as at least a further 30 species have only been identified to genera, and observations have been geographically restricted to the Kaledupa area of the WNP. The coral reefs of the WNP, along with comparable areas such as Milne Bay in PNG and Komodo in Indonesia, are thought to contain the highest fish biodiversity in the world (Halford, 2003) Furthermore, seagrass fish sampling has been spatially limited in the WNP, hence this area may have higher seagrass fish biodiversity than previously thought. In addition, species of the families gobidae and blennidae are particularly difficult to identify and have been found in many locations throughout the Indo-Pacific to have high speciation. Many of the sampling techniques used in previous WNP fish censuses are likely to have also under-sampled these fish families. Species identification has also often not been possible, leading to a number of researchers suspecting that both these fish families contain numerous undescribed species within the WNP.

Table 6. Seagrass fish species list (188 species) for the Wakatobi Marine Park (orders are bold capitalised, families capitalised)

Order and family	Species name
ANGUILIFORMES	
MURAENIDAE	*Siderea picta* (Ahl,1789)
	Gymnothorax albimarginatus (Temminck & Schlegel, 1846)
	Gymnothorax fimbriatus (Bennett, 1832)
OPHICHTHIDAE	*Myrichthys colubrinus* (Boddaert, 1781)
ATHERINIFORMES	
ATHERINIDAE	*Atherinomorus lacunosus* (Forster, 1801)
BELONIDAE	*Tylosurus crocodilus* (Péron & Lesueur, 1821)
AULOPIFORMES	
SYNODONTIDAE	*Synodus binotatus* (Schultz, 1953)
BELONIFORMES	
HEMIRAMPHIDAE	*Hyporhamphus affinis* (Günther 1866
	Hyporhamphus dussumieri (Valenciennes, 1847)
	Zenarchopterus dispar (Valenciennes, 1847)
BERYCIFORMES	
HOLOCENTRIDAE	*Myrispristis pralinia* (Cuvier, 1829)
	Neoniphon argenteus (Valenciennes, 1831)
	Neoniphon sammara (Forsskål, 1775)
CLUPEIFORMES	
ENGRAULIDAE	*Stolephorus indicus* (van Hasselt, 1823)
GASTEROSTEIFORMES	
FISTULARIIDAE	*Fistularia commersonii* (Rüppell, 1838)
HETEROPTERA	
GERRIDAE	*Gerres acinaces* (Bleeker, 1854)
	Gerres filamentosus (Cuvier 1829)
	Gerres oyena (Forsskål, 1775)
	Gerres subfasciatus (Cuvier, 1830)
MYLIOBATIFORMES	
DASYATIDAE	*Taeniura lymma* (Forsskål, 1775)
PERCIFORMES	
ACANTHURIDAE	*Acanthurus fowleri* (de Beaufort, 1951)
	Acanthurus grammoptilus (Richardson, 1843)
	Acanthurus nigrofuscus (Forsskål, 1775)
	Naso unicornis (Forsskål, 1775)
	Naso vlamingii (Valenciennes, 1835)
APOGONIDAE	*Apogon ceramensis* (Bleeker, 1852)
	Apogon exostigima (Jordan & Starks, 1906)

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Table 6 (contd.)

Order and family	Species name
PERCIFORMES (contd.)	
APOGONIDAE (contd.)	*Apogon fuscus* (Quoy & Gaimard, 1825)
	Apogon hartzfeldii (Bleeker, 1852)
	Apogon hartzfeldii (Bleeker, 1852)
	Apogon hoevenii (Bleeker, 1854)
	Apogon melas (Bleeker, 1848)
	Apogon thermalis (Cuvier, 1829)
	Cheilodipterus artus (Smith, 1961)
	Cheilodipterus isostigmus (Schultz, 1940)
	Cheilodipterus macrodon (Lacepède, 1802)
	Cheilodipterus quinquelineatus (Cuvier, 1828)
	Fowleria punctulata (Rüppell, 1838)
	Pseudamia gelatinosa (Smith, 1955)
	Sphaeramia orbicularis (Cuvier, 1828)
BLENNIIDAE	*Blenniella paula* (Bryan & Herre, 1903).
CARANGIDAE	*Carangoides oblongus* (Cuvier, 1833)
	Carangoides orthogrammus (Jordan & Gilbert, 1882)
	Caranx ignobilis (Forsskål, 1775)
	Caranx melampygus (Cuvier, 1833)
	Trachinotus blochi (Lacepède, 1801)
CENTRISCIDAE	*Centriscus scutatus* (Linnaeus, 1758)
CENTROGENIIDAE	*Centrogenys vaigiensis* *(*Quoy & Gaimard, 1824)

Order and family	Species name
PERCIFORMES (contd.)	
CHAETODONTIDAE	*Chaetodon kleinii (Bloch, 1790)*
	Chaetodon lunula (Lacepède, 1802)
	Chaetodon ocellicaudus (Cuvier, 1831)
	Chaetodon vagabundus (Linnaeus, 1758)
	Coradion chrysozonus (Cuvier, 1831)
EPHIPPIDAE	*Platax teira* (Forsskål, 1775)
GOBIDAE	*Amblygobius phalaena* (Valenciennes, 1837).
	Amblygobius sphynx (Valenciennes, 1837)
	Cryptocentrus polyophthalmus (Bleeker, 1853)
	Yongeichthys nebulosus (Forsskål, 1775)
	Exyrias bellisimus (Smith, 1959)
	Exyrias ferrarisi (Murdy, 1985)
	Exyrias puntang *(*Bleeker, 1851)
	Istigobius ornatus (Rüppell, 1830)
HAEMULIDAE	*Plectorhinchus gibbosus* (Lacepède, 1802)
	Plectorhinchus lessonii (Cuvier, 1830)
	Plectorhinchus orientalis (Bloch, 1793)
	Plectorhinchus vittatus (Linnaeus, 1758)

Table 6 (contd.)

Order and family	Species name
PERCIFORMES (contd.)	
LABRIDAE	*Cheilinus chlorourus* (Bloch, 1791)
	Cheilinus trilobatus (Lacepède, 1801)
	Cheilio inermis (Forsskål, 1775)
	Choerodon anchorago (Bloch, 1791)
	Coris aurilineata (Randall & Kuiter, 1982)
	Coris gaimard (Quoy & Gaimard, 1824)
	Halichoeres argus (Bloch & Schneider, 1801)
	Halichoeres margaritaceus (Valenciennes, 1839)
	Halichoeres papilionaceus (Valenciennes, 1839)
	Halichoeres podostigma (Bleeker,1854)
	Halichoeres richmondi (Fowler & Bean, 1928)
	Halichoeres scapularis (Bennett, 1832)
	Halichoeres trimaculatus (Quoy & Gaimard, 1834)
	Hologymnosus doliatus (Lacepède, 1801)
	Labroides dimidiatus (Valenciennes, 1839)
	Stethojulis strigiventer (Bennett, 1833)
	Thalassoma hardwicke (Bennett, 1830)
LETHRINIDAE	*Lethrinus atkinsoni* (Seale, 1910)
	Lethrinus erythropterus (Valenciennes, 1830)
	Lethrinus harak (Forsskål, 1775)
	Lethrinus laticaudis (Alleyne & Macleay, 1877)

Order and family	Species name
PERCIFORMES (contd.)	
LETHRINIDAE (contd.)	*Lethrinus lentjan* (Lacepède, 1802)
	Lethrinus ornatus (Valenciennes, 1830)
	Lethrinus semicinctus (Valenciennes, 1830)
	Lethrinus variegatus (Valenciennes, 1830)
	Lethrinus xanthochilus (Klunzinger, 1870)
LUTJANIDAE	*Lutjanus decussatus* (Cuvier, 1828)
	Lutjanus ehrenbergii (Peters, 1869)
	Lutjanus fulvus (Forster, 1801)
	Lutjanus russelli (Bleeker, 1849)
	Lutjanus timorensis (Quoy & Gaimard, 1824)
MICRODESMIDAE	*Gunnelichthys monostigma* (Smith, 1958)
MUGILIDAE	*Liza vaigiensis* (Quoy & Gaimard, 1825)
	Neomyxus leuciscus (Günther, 1872)
MULLIDAE	*Mulloidichthys vanicolensis* (Valenciennes, 1831)
	Parupeneus barberinus (Lacepède, 1801)
	Parupeneus indicus (Shaw, 1803)
	Parupeneus macronemua (Lacepède, 1801)
	Parupeneus macronemua (Lacepède, 1801)
	Parupeneus multifasciatus (Quoy & Gaimard, 1824)
	Upeneus moluccensis (Bleeker, 1855)
	Upeneus sundaicus (Bleeker, 1855)

Table 6 (contd.)

Order and family	Species name
PERCIFORMES (contd.)	
MULLIDAE (contd.)	*Upeneus tragula* (Richardson, 1846)
NEMIPTERIDAE	*Pentapodus caninus* (Cuvier, 1830)
	Pentapodus trivittatus (Bloch, 1791)
	Scolopsis affinis (Peters, 1877)
	Scolopsis bilineatus (Bloch, 1793)
	Scolopsis ghanam (Forsskål, 1775)
	Scolopsis lineatus (Quoy & Gaimard, 1824)
	Scolopsis monogramma (Cuvier, 1830).
	Scolopsis temporalis (Cuvier, 1830).
	*Scolopsis trilineatus (*Kner, 1868)
PINGUIPEDIDAE	*Parapercis cylindrica* (Bloch, 1792)
	Parapercis millipunctata (Günther, 1860)
POMACENTRIDAE	*Abudefduf septemfasciatus* (Cuvier, 1830)
	Chrysiptera biocellata (Quoy & Gaimard, 1825).
	Chrysiptera cyanea (Quoy & Gaimard, 1825)
	Chrysiptera parasema (Fowler, 1918)
	Dascyllus aruanus (Linnaeus, 1758)
	Dascyllus reticulatus (Richardson, 1846)
	Dascyllus trimaculatus (Rüppell, 1829)
	Dischistodus chrysopoecilus (Schlegel & Müller, 1839)
	Dischistodus fasciatus (Cuvier, 1830)

Order and family	Species name
PERCIFORMES (contd.)	
POMACENTRIDAE (contd.)	*Dischistodus perspicillatus* (Cuvier, 1830)
	Dischistodus prosopotaenia (Bleeker, 1852)
	Dischistodus pseudochrysopoecilus (Allen & Robertson, 1974)
	Hemiglyphidodon plagiometopon (Bleeker, 1852)
	Plectroglyphidodon leucozonus (Bleeker, 1859)
	Pomacentrus burroughi (Fowler, 1918)
	Pomacentrus chrysurus (Cuvier, 1830)
	Pomacentrus lepidogenys (Fowler & Bean, 1928)
	Pomacentrus taeniometopon (Bleeker, 1852)
	Pomacentrus tripunctatus (Cuvier, 1830)
	*Stegastes nigricans (*Lacepède, 1802)
	Abudefduf sordidus (Forsskål, 1775)
SCARIDAE	*Calothomus spinidens* (Quoy & Gaimard, 1824)
	Leptoscarus vaigiensis (Quoy & Gaimard, 1824)
	Scarus globiceps (Valenciennes, 1840)
	Scarus schlegeli (Bleeker, 1861)
SERRANIDAE	*Ephinephelus ongus* (Bloch, 1790)
	Epinephelus merra (Bloch, 1793)

Table 6 (contd).

Order and family	Species name
PERCIFORMES (contd.)	
SIGANIDAE	*Siganus canaliculatus* (Park, 1797)
	Siganus doliatus (Guérin-Méneville, 1829-38)
	Siganus fuscescens (Houttuyn, 1782)
	Siganus guttatus (Bloch, 1787)
	Siganus punctatus (Schneider & Forster, 1801)
	Siganus virgatus (Valenciennes, 1835)
SPHYRAENIDAE	*Sphyraena barracuda* (Edwards, 1771)
	Sphyraena qenie (Klunzinger, 1870)
	Corythoichthys haematopterus (Bleeker, 1851)
	Corythoichthys intestinalis (Ramsay, 1881)
	Hippocampus taeniopterus (Bleeker, 1852)
	Syngnathoides biaculeatus (Bloch, 1785)
TERAPONTIDAE	*Mesopristes argenteus* (Cuvier, 1829)
	Terapon jarbua (Forsskål, 1775)
TOXOTIDAE	*Toxotes jaculatrix* (Pallas, 1767)
PLEURONECTIFORMES	
BOTHIDAE	*Bothus mancus* (Broussonet, 1782)
	Bothus pantherinus (Rüppell, 1830)
SOLEIDAE	*Pardachirus pavoninus* (Lacepède, 1802)
	Zebrias fasciatus (Basilewsky, 1855)
SCORPAENIFORMES	
TETRAROGIDAE	*Ablabys taenianotus* (Cuvier, 1829)
	Paracentropogon longispinus (Cuvier,1829)
	Richardsonichthys leucogaster (Richardson, 1848)
SILURIFORMES	
PLOTOSIDAE	*Paraplotosus albilabris* (Valenciennes, 1840)
	Plotosus lineatus (Valenciennes, 1840)
TETRAODONTIFORMES	
BALISTIDAE	*Rhinecanthus aculeatus* (Linnaeus, 1758)
	Balistapus undulatus (Park, 1797)
	Balistoides viridescens (Bloch & Schneider,1801)
DIODONTIDAE	*Diodon liturosus* (Shaw, 1804)
MONACANTHIDAE	*Acreichthys hajan* (Bleeker, 1852)
	Acreichthys tomentosus (Linnaeus, 1758)
	Aluterus scriptus (Osbeck, 1765)
	Monacanthus chinensis (Osbeck, 1765)
	Pseudomonacanthus macrurus (Bleeker, 1857)
	Pseudomonacanthus macrurus (Bleeker, 1857)
OSTRACIIDAE	*Lactoria cornuta* (Linnaeus, 1758)
TETRAODONTIDAE	*Arothron manilensis* (de Procé, 1822)
	Arothron reticularis (Bloch & Schneider,1801)
	Arothron stellatus (Bloch & Schneider, 1801)
	Canthigaster bennetti (Bleeker, 1854)
	Canthigaster compressa (Marion de Proce 1822)

Within the intertidal seagrass meadows of the WNP, *Atherinomorus lacunosus*, *Lethrinus harak*, and species of Apogonidae were the most common fish species reported (Unsworth *et al.*, 2007a; Unsworth *et al.*, 2007e). These assemblages have been recorded in the WNP to be significantly altered by the presence of nearby mangrove, resulting in *Siganus canaliculatus* and *Naso vlamingii* becoming additional representative species (Unsworth *et al.*, 2008). The trophic structure of the fish assemblage of WNP seagrass meadows was dominated by predatory fish and planktivores, with herbivores and omnivores contributing a minor proportion of the assemblage (Unsworth *et al.*, 2008). Unfortunately, no work has been conducted on the fish assemblages of the subtidal seagrass beds of the WNP. Given their extensive coverage, these are priority areas for further research.

ECOSYSTEM INTERACTIONS

The presence of a complex array of inter-linked coastal and marine habitats and ecosystems within the WNP indicates the need for the development of ecosystem-level integrated coastal zone management. For such management strategies to be effective it is important to understand the extent that individual habitats are inter-connected.

The influence of habitat connectivity on the fish assemblages of seagrass meadows of the WNP has been extensively investigated (Unsworth *et al.*, 2007a; Unsworth *et al.*, 2007e; Unsworth *et al.*, 2008; Unsworth *et al.*, 2009b; Unsworth *et al.*, 2009c). This research found connectivity to be a key structuring process of fish assemblages in seagrass meadows in the WNP. Fish assemblages in WNP meadows are structured on local, short-term temporal scales by environmental diel and tidal cycles (Unsworth *et al.*, 2007a; Unsworth *et al.*, 2007e). Seagrass fish assemblages were found to increase in abundance and diversity at night, suggesting either migration onto these habitats from nearby habitats such as reefs, mangroves or deep water and/or increased activity from resident seagrass fish. Research in the WNP has indicated that diel changes in seagrass fish assemblages are predominantly structured by food availability, although other factors such as increased night-time shelter provision are also thought to be important, albeit to a much lesser extent.

Research in the WNP has also confirmed that the diverse fish assemblages of seagrass habitats have complex behavioural patterns that include tidal migrations to nearby habitats (Unsworth *et al.*, 2007a). These tidal migrations are thought to be the result of feeding and predatory pressures, together with the requirement for acceptable and available habitat.

For the first time in an Indo-Pacific seagrass system, research in the WNP has found clear evidence that mangroves and coral reefs can have an important role in influencing fish assemblage structure of seagrass beds (Unsworth *et al.*, 2008). This was found to be on a very local small scale at the seagrass-mangrove edge, and at a wider ecosystem level (Unsworth *et al.*, 2008; Unsworth *et al.*, 2009b). A key ecosystem level control on fish assemblages was the presence of adjacent mangrove habitats that act as important feeding grounds for seagrass and reef fish (Unsworth *et al.*, 2008). It was proposed that mangroves also provide an important source of organic matter and nutrients to the seagrass food web, resulting in the stimulation of a diverse and abundant fish fauna. Seagrass beds and mangroves also in the WNP are important habitat for juvenile fish, and when in a three-way continuum with nearby reefs, provide a greater fish nursery function. Results of connectivity studies in the WNP support the need for ecosystem-level management of shallow water tropical habitats, but also suggest that management requires local-level knowledge of habitat interactions and water circulation to successfully enhance or conserve fish assemblages.

THREATS TO SEAGRASS MEADOWS

There is growing global evidence that seagrass ecosystems are experiencing an unprecedented level of damage, deterioration and overexploitation, mostly attributed to human activities (Orth *et al.*, 2006). Examination of global trends indicates that seagrass meadows are being lost at a rate similar to coral reefs and tropical rainforests (Waycott *et al.*, 2009). This degradation of seagrass beds has been associated with nutrient run-off, sedimentation, physical degradation and pesticide leaching. In many areas of the Indo-Pacific, particularly within small island communities such as the WNP, seagrass ecosystems are increasingly threatened by over-exploitation of their productive fish and invertebrate assemblages (Fortes, 1990). This exploitation is largely undocumented within the region but thought to be increasing in areas of high human population growth (Fortes, 1990), such as the WNP. Tropical marine research and conservation often focuses on the exploitation and decline of coral reefs (Pandolfi *et al.*, 2003). Many studies have documented the long-term decline of reef resources, whilst paying little attention to important nearby fishery and nursery grounds, such as seagrass beds and mangroves (McManus, 1997; Jennings *et al.*, 1999). As shallow water coastal habitats that are often intertidal, seagrass meadows and mangroves provide readily accessible and productive fishing grounds to human populations living within the coastal realm (Tomascik *et al.*, 1997; de la Torre-Castro and Rönnbäck, 2004). It is important to recognise that in many areas of the Indo-Pacific seagrass and mangrove habitats may be at greater threat from overexploitation than coral reefs. These shallow water habitats can often be fished in all weather conditions due to their coastal proximity and inter-tidal environment. In contrast, reefs are usually further from the land and less easily accessed during poor weather conditions (Fortes, 1990; May, 2003).

In areas of high biodiversity such as Sulawesi, fisheries management is urgently required where large areas of reefs and seagrass are increasingly threatened by over-fishing and destructive fishing practices (Hopley and Suharsono, 2000; Unsworth *et al.*, 2007c). This is particularly apparent within the Wakatobi where grouper populations have been observed to decline by approximately 50% over five years (Unsworth *et al.*, 2007c). In the WNP the local human populations are particularly dependant on marine resources, yet the local fishery is unmanaged and overexploited (Cullen, 2007; Cullen *et al.*, 2007) with fishers indicating a preference to utilise seagrass beds as fishing areas (Cullen *et al.*, 2006). One of the major impacts on seagrass meadows in the WNP is overexploitation, which is now detrimentally impacting fish and invertebrate stocks (Unsworth *et al.*, 2009a). The high exploitation rates are evident at low spring tide, when many people can be seen walking across exposed seagrass beds collecting a range of invertebrate species. This ‘gleaning’ activity mostly involves collection for subsistence purposes, but includes species for commercial export such as sea cucumber (beche-de-mer) and shells for the curio trade. In the Kaledupa area of the WNP, 25% of households said they gleaned regularly, with the average number of trips per week being 2.14 ± 0.26 for those households stating regular involvement (Cullen, 2007). Such exploitation at low tide is also common in other areas of the Indo-Pacific, A survey of edible molluscs harvested by local fishermen in Bais Bay in the southern Philippines included 27 species of bivalves and univalves amounting to 69 kg ha^{-1} of edible molluscs seagrass beds comprised of mono and multi specific stands (Alcala and Alcazar, 1994).

Seagrass meadows are also vitally important to the local people for fin fisheries. Common fishing methods include gill netting, seine netting and small trap fishing. In terms of their economic contribution, seagrass fish are far more economically valuable to the local population than the invertebrate communities despite their high subsistence importance (Unsworth *et al.,* 2009a). A mean value of US$77.9 ± 40.4 ha^{-1} was calculated for the fisheries standing stock of unmanaged seagrass in the WNP (Unsworth *et al.,* 2009a). This was found to increase by 66% in a protected area with the value becoming dominated by fish, indicating the extent to which WNP seagrass meadow fish assemblages are now over-exploited. What is of most urgent concern is the increasing use of high-yield fishing methods, particularly the use of fish fences which comprise a non-selective method utilising the tidal movement of fish to trap all individuals on the reef flat and seagrass beds as the tide ebbs (May, 2003). Fish fences, known locally as '*sero*', have increased twelve fold from 2002 to 2007 within the WNP and their catch per unit effort has steadily declined by over 60% (see Exton, this volume). Regardless of trophic level, most fish in the WNP are now harvested before they reach sexual maturity, with a large proportion of the catch coming directly from seagrass habitats (Unsworth, 2007). Fish families regularly targeted in seagrass meadows of the WNP for subsistence include Lethrinidae (Emperor), Scaridae (Parrotfish) and Siganidae (Rabbitfish). Harvested invertebrates include gastropods (Strombidae, Cymbiola, and the Cowries) and sea cucumbers (May, 2005). Many local fishermen of the WNP now believe that these once abundant resources are now in decline (May, 2003).

However the ecological consequences of sustained over-fishing in the seagrass meadows of the WNP are not presently clear. In other regions of the Indo-Pacific, a reduction in the abundance of important seagrass predators such as Triggerfish and Wrasse have been observed to have detrimental cascade effects on seagrass ecosystems, resulting in large increases in the populations of urchins such as *Tripneustes gratillus* and Diadema *sp.* (McClanahan *et al.*, 1996; Eklöf *et al.*, 2008). The *Tripneustes gratillus* fishery in the WNP has been found in recent years to be booming despite high extraction rates, indicating that the stock populations of these species may be increasing. Although this suggests the presence of a sustainable fishery; this could be to the detriment of ecosystem equilibrium. As urchins are grazers on seagrass, it remains to be seen whether the productivity of the seagrass habitat can sustain such a population without damage and whether other unseen effects of ecosystem alteration are yet to emerge (Eklöf *et al.*, 2008). Seagrass overgrazing episodes in many areas of the world, for example, have been attributed to sea urchin population explosions (Rose, 1999; Eklöf *et al.*, 2008).

Marine pollution is not currently thought to be a significant risk to marine biodiversity or habitat coverage in the WNP, although it is an issue which requires consideration for the long term protection of seagrass systems. In areas surrounding the growing population of small towns such as Wanci, Ambeua, Sama Bahari and Mantigola, intertidal seagrasses are likely to bear the brunt of high nutrient loading. Such elevated nutrients may have a positive effect on seagrasses by providing elevated phosphate to nutrient limited carbonate reef seagrasses (Udy *et al.*, 1999), but also have the potential to result in eutrophic conditions that could cause algal overgrowth (Taylor *et al.*, 1995). Such overgrowth can currently be observed at the edges of these small towns. Personal observations in these areas indicate that seagrass is becoming overgrown with filamentous algae and mud-flats are increasingly covered with thick algal mats in some small areas. Of particular concern within the WNP is the widespread dumping of refuse into the sea. This waste is routinely washed up on beaches and can be seen floating within the water column of seagrass beds, constituting a real threat to large marine organisms such as the abundant cetaceans of the WNP that are capable of ingesting this waste.

Cultivation of marine algae, principally Gracilaria spp. and Euchema spp., is of increasing economic importance to the WNP (Cullen, 2007). These species are cultivated throughout the WNP on floating strings in shallow subtidal seagrass meadows, being harvested, dried and exported for the extraction of various gelatinous products such as carrageenan. These products have diverse applications in the global food, cosmetics and pharmaceutical industries (Cullen, 2007). Recent evidence from Tanzania suggests that such seaweed cultivation can result in a loss of up to 40% of seagrass biomass, probably due to light reduction (Eklöf *et al.*, 2006). This presents a dilemma as the cultivation of marine algae presents a significant alternative income to fishers and is a favoured management policy in order to reduce pressure on marine habitats. It is important that conservation management and the future expansion of seaweed cultivation considers the potential impacts that aquaculture may have upon marine ecosystems.

FUTURE CLIMATE CHANGE SCENARIOS

Global climate is now undergoing a period of sustained warming that is predicted to continue at a rapid rate for the next 100 years, resulting in increases of between 0.6°C and 6.4°C (IPCC, 2001; 2007). Such warming and atmospheric build up of CO_2 and other greenhouse gases threatens the health and productivity of global seagrass ecosystems (Short and Neckles, 1999; Duarte, 2002; Waycott *et al.*, 2007). As sensitive 'canaries' of environmental change in marginal coastal habitats, seagrass meadows are likely to be at the forefront of changing climate conditions. Although increasing temperature and sea-level are the widely reported potential impacts of climate change, increasing storm prevalence, ocean acidification and altered UV are also key factors that may influence seagrass meadows in the future (Björk *et al.*, 2008).

Short-term temperature increases have been found to cause thermal stress in a number of seagrass species common within the WNP (*Halophila ovalis* and *Syringodium isoetifolium*) at ecologically relevant temperatures of up to 40°C (Campbell *et al.*, 2006). Further elevated temperatures quickly caused the death of the majority of other seagrass species that are abundant within the WNP (*Thalassia hemprichii, Cymodocea rotundata* and *halodule uninervis*) (Campbell *et al.*, 2006). These laboratory findings support evidence from the field that indicate extreme temperatures to negatively impact upon seagrass biomass (Thorhaug, 1978; Stapel, 1997; Rasheed and Unsworth, 2009), and indicates that seagrass species of the WNP may be at risk from future climate scenarios that cause elevated seawater temperature.

Whilst the impact that ocean acidification may have upon seagrass is difficult to predict, a number of researchers have utilised small scale field and laboratory experiments to suggest that decreasing pH may benefit seagrass growth and production by releasing these C_3 plants (low-light adapted plants that evolved in the Mesozoic and Paleozoic eras) from carbon limitation (Zimmerman *et al.*, 1997; Palacios and Zimmerman, 2007; Hall-Spencer *et al.*, 2008). Other investigations have also found that dense seagrass meadows may make the water column more alkaline over a daily photosynthetic cycle (Invers *et al.*, 1997). As with the majority of seagrass research, experiments have investigated mostly temperate and sub-tropical seagrasses, and the impacts of decreasing pH on the species that are commonly found in the WNP are as yet unknown.

Extreme weather events that are commonly a force of natural disturbance within tropical marine systems are considered likely to increase in frequency and duration within the tropics. Such events could potentially pose a future threat to the marine environment of the WNP. Floods and cyclones have been commonly documented to detrimentally impact upon seagrasses in Northern Australia, and in some cases recovery has taken up to three years (Preen *et al.*, 1995). If such events do become more frequent then the natural cycle of seagrass loss and recovery may become weakened as seagrass meadows become less resilient to repeated stress.

Although it is unlikely that seagrass will be altered in the short to medium term directly by climate change, anthropogenic impacts can often act cumulatively. By providing good resource and ecological management for these important ecosystems, seagrass meadows are more likely to be in a healthy and resilient state, and hence more likely to survive the impacts of some future climate impacts (Björk *et al.*, 2008).

CONSERVATION MANAGEMENT

Recent research within the WNP has provided compelling evidence for the connectivity of fish assemblages between seagrass, reef and mangrove (see Unsworth and Salinas, this volume), the importance of processes such as herbivory, and some of the photo-physiological controlling factors of seagrass distribution. These are all important findings, but further research is necessary to fully understand ecosystems that are highly biodiverse and contain complex trophic inter-relationships with mangroves and coral reefs. There is a particular need to develop a greater understanding of the diversity and ecological role of the invertebrate communities within the WNP seagrass meadows, as these represent some of the species at the forefront of overexploitation. We also have a poor understanding of the temporal and spatial variability within all the faunal assemblages of seagrasses of the region, the regularity of fish movements and migration, and the food web supporting economically important fauna.

Conserving marine ecosystems, particularly seagrass meadows, not only has biological and economic implications, it is also important from a cultural perspective. The WNP contains many thousands of people who are culturally distinct from the Indonesian culture. The *Bajo Laut* (often referred to as sea gypsy people) live in stilted villages that are usually sited on shallow intertidal or subtidal seagrass beds. In the case of the Wakatobi, the Bajo have large villages at Mola, Mantigola, Sama Bahari and LaHoa, all of which are in areas of extensive seagrass. Their very existence and culture is intricately intertwined with seagrass beds as the latter represents an abundant source of protein from fish and invertebrates that is collected by everyone in the community. Their exists a complex yet poorly understood interaction between population growth, economic development and the demands of modern society that has the potential to result in degraded and overexploited seagrass meadows that may ultimately be of detriment to their unique way of life.

Seagrass communities are important for conservation not only due to their role in supporting biodiversity, but also because they are habitats for species on the IUCN Red List such as dugongs (*Dugong dugon*) and green turtles (*Chelonia mydas*). In tropical Australian seagrass communities, these large macro grazers are found to be important components in community ecological structuring (Carruthers *et al.*, 2002). Whilst no data from the WNP exists to explain the abundance of either of these key species, dugongs are thought to only remain as isolated families (*pers obs*) (Figure 13), and green turtles are present only on outer islands where they are known to nest. Future management should consider these important species, despite the limited information available about their populations.

Future research is therefore required to fully understand the ecology of these important ecosystems, particularly in light of evidence underlining their overexploited state, their importance for harbouring endangered and biodiverse faunal species, their role as juvenile fish nurseries, and the dependence of the local people upon these ecosystems for their daily protein requirements. Park managers, fishermen and other local stakeholders need to heed the warnings of what are clearly declining natural resources and take the precautionary principle, embodied in Principle 15 of the Rio Declaration (Earth Summit 1992): “Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation”.

Figure 13. Dugong calf rescued from fishermen in Sama Bahari in September 2003. This specimen had been caught as by-catch within a gill net in sub-tidal seagrass between Lahoa and Buranga.

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