Water quality in the Great Barrier Reef region: responses of mangrove, seagrass and macroalgal communities

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Abstract

Marine plants colonise several interconnected ecosystems in the Great Barrier Reef region including tidal wetlands, seagrass meadows and coral reefs. Water quality in some coastal areas is declining from human activities. Losses of mangrove and other tidal wetland communities are mostly the result of reclamation for coastal development of estuaries, e.g. for residential use, port infrastructure or marina development, and result in river bank destabilisation, deterioration of water clarity and loss of key coastal marine habitat. Coastal seagrass meadows are characterized by small ephemeral species. They are disturbed by increased turbidity after extreme flood events, but generally recover. There is no evidence of an overall seagrass decline or expansion. High nutrient and substrate availability and low grazing pressure on nearshore reefs have lead to changed benthic communities with high macroalgal abundance. Conservation and management of GBR macrophytes and their ecosystems is hampered by scarce ecological knowledge across macrophyte community types.
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1. Introduction

The Great Barrier Reef World Heritage Area (GBRWHA) contains a large number of interconnected ecosystems colonised by marine plants including coral reef systems, extensive seagrass meadows and estuarine systems with mangrove forests. These systems are also closely linked with adjacent coastal river catchments in a catchment-to-reef continuum. Globally, the rate of degradation of coral reefs, seagrass beds, mangrove forests and other tropical marine ecosystems is increasing as a consequence of direct human pressures, including input of pollutants such as sediments, nutrient and toxic compounds. Additionally, indirect pressures on coral reefs are global climate change (coral bleaching), crown-of-thorns starfish outbreaks and coral diseases (Wilkinson, 2002). The Great Barrier Reef (GBR) is considered to be in a 'near-pristine state over large areas', however, water quality in some coastal areas appears to be declining from effects of human activities (Australian State of the Environment Committee, 2001; Brodie, 1997).

A large body of evidence now shows that water quality and ecological integrity of some of the coastal area of GBRWHA are affected by material originating from a range of human activities on the catchment, such as primary industries (agriculture, aquaculture), urban and industrial development (reviewed in Haynes, 2001; Williams, 2002; Baker et al., 2003; Furnas, 2003). In summary, delivery of sediments and nutrients to rivers discharging into GBR waters has increased by at least four times compared to estimates from before 1850;

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chlorophyll *a* levels in coastal waters are higher than off-shore, especially in the more developed central and southern sections of the GBR; and herbicides (principally diuron) have been found in coastal and intertidal mangrove sediments and seagrasses adjacent to catchments with high agricultural use, at levels shown to adversely affect seagrass productivity and implicated in local dieback of mangroves. GBR nearshore coral reefs adjacent to catchments with intense agricultural activity, i.e. the Wet Tropics region, have higher water column suspended solids and nutrients, and cover and richness of hard and soft corals were lower and cover of macroalgae was higher, compared to reefs adjacent to less developed catchments (Fabricius and De'ath, in press; Fabricius et al., in press). In addition, coastal development has greatly altered shoreline and catchment landscapes along the Queensland coast, affecting the natural estuarine functioning. The most notable changes are significant losses and declining ecosystem health of many coastal wetlands including mangroves (e.g., Duke, 1997; Russell and Hales, 1994; Russell et al., 1996).

Increased inputs of nutrients can cause significant changes in tropical marine ecosystems, principally by increasing primary production. Some coral reefs overseas are subject to chronic increased availability of dissolved nutrients (Szmant, 2002), while nearshore reefs in the GBR region are mainly exposed to periodically increased nutrient and sediment loads from runoff during the summer monsoons. Nutrient and particulate matter levels in the GBR lagoon under non-flood conditions are generally low, although higher values are typically found close to the coast (Furnas et al., 1995; Furnas, 2003). In contrast, levels are between 10 and 400 times higher in flood plumes, which are the main delivery mechanism for nutrients (in dissolved and particulate form) and suspended sediments to GBR coastal waters (Devlin et al., 2001; Devlin and Brodie, in press). The dissolved nutrients in river plumes are rapidly taken up by phytoplankton and are generally not in contact with benthic macrophytes for extended periods of time. The resulting phytoplankton blooms are consumed rapidly by zooplankton, resulting in nutrients being recycled several times and transported in particulate organic form further away from the coast where they may settle on macrophyte and coral reef communities (Furnas, 2003; Furnas et al., in press). In nearshore waters fine particles of suspended sediment combine with suspended aggregates of organic matter called 'marine snow' (detritus particles, fecal material, mucus secreted by phytoplankton and bacteria) to form heavier flocs that can settle on benthic communities (Fabricius and Wolanski, 2000; Wolanski et al., 2003). This synergy of nutrients, fine sediments and associated toxicants is believed to be the greatest threat of runoff to GBR ecosystems (Fabricius and Wolanski, 2000; Wolanski et al., 2003; Fabricius et al., 2003; Fabricius and De'ath, in press).

Metals and organochlorine contaminants (e.g. herbicide and pesticide residues) are generally found in low concentration in GBR waters and sediments, except for sites adjacent to ports and harbours, urban centres and areas of intense agricultural activity (Haynes and Johnson, 2000). Herbicides are transported to aquatic environments either dissolved in the water or bound to sediment during rain events. They affect marine plants by interfering with the normal functioning of the photosynthetic apparatus (Percival and Baker, 1991). Heavy metals inhibit pathways in photosystem II or interfere with pigment synthesis, however, permanent damage may not occur (Prasad and Strzalka, 1999). The operation of shipping within the GBR also poses the risk of significant oil spills, likely to have serious impacts on intertidal and coastal marine plant habitats, and of release of pollutants from antifouling paints, for example after ship groundings.

Here, we review responses of tropical marine plants in the Great Barrier Reef region to changes in water quality.

2. Marine plants in the Great Barrier Reef region

Marine plants are widely distributed in the GBR region, colonising different habitats. While mangroves form a unique and dominant ecosystem predominantly bordering coastal margins of the GBR region, seagrasses and macroalgae occur in a variety of ecosystems across the GBR shelf (Fig. 1).

Mangroves are inter-tidal marine plants, mostly trees, and thrive in saline conditions and daily inundation between mean sea level and highest astronomical tides. Mangroves are not a monophyletic taxonomic unit. Fewer than 22 plant families have developed specialised morphological and physiological characteristics that characterize mangrove plants, such as buttress trunks and roots providing support in soft sediments and physiological adaptations for excluding and expelling salt. Globally, there are just 70 'true' mangroves species (Tomlinson, 1986; Duke et al., 1998). In the GBR region there are around 2070 km^2 of mangroves with 37 species (Duke, 1997; Duke et al., 1998). The benefits of mangroves are based on their primary and secondary production, as well as their standing woody biomass and forested ecosystem structure (Duke et al., 1998). These benefits include (adapted from Tomlinson, 1986):

- visual amenity and shoreline beautification;
- shoreline protection, based on mangrove tree and root structure, which reduces erosion and provides stand protection from waves and water movement;
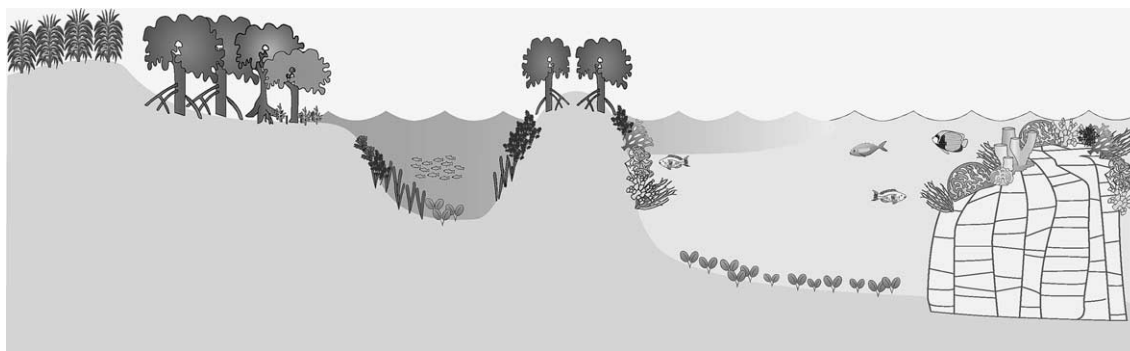- nutrient uptake, fixation, trapping and turnover;

Fig. 1. Conceptual model of macrophyte habitats in the Great Barrier Reef region from the coast across the continental shelf.

- carbon sink and sequestration;
- secondary production, via grazing and decomposition of mangrove biomass and associated microbial and faunal production;
- sediment trapping, based on mangroves being a depositional site for both water and airborne sediments, which in turn reduces turbidity of coastal waters;
- meso-climate; forests might moderate evapo-transpiration to create a specialised niche climate;
- habitat for specialised fauna, nursery habitat, protection and food resources for mature fauna such as migratory birds and fish;
- forest products (e.g. timber, firewood- although not widely used in Australia); and
- fishery products, including both estuarine and coastal fish, crustaceans and molluscs.

Mangroves are highly valued for some of these benefits, such as their importance for fish biomass and diversity (e.g. Mumby et al., 2004). However, mangrove areas have been steadily removed from more populated estuaries in the GBR region over the last 150 years. In the Mackay region an average 5 ha of around 620 ha of mangroves per year was removed between 1945 and 1998 (Duke and Wolanski, 2001) indicating that tidal wetlands were valued more for their conversion to other land-uses than for their collective benefits.

Similar to mangroves, seagrass is a functional but not a taxonomic category. Present seagrass diversity arose from at least three separate lineages (Waycott and Les, 1996; Les et al., 1997), having evolved in adaptation to the marine environment (Walker et al., 1999). Approximately 68 species of seagrass are found globally (Les et al., 1997). Fifteen species are recorded from Queensland waters (Lee Long et al., 2000). An estimated 40,000 km^2 of lagoon, inter-reef and reef-top areas within the Great Barrier Reef World Heritage Area comprise seagrass habitat (Coles et al., 2003). Seagrasses in the GBR region are important as they support fisheries productivity, and are a food source for dugongs and turtles. Their presence allows for the recruitment of many other plants and animals, e.g. living epiphytically or within the meadow. Seagrasses, especially structurally large species, also affect coastal and reefal water quality by absorbing nutrients and trapping sediments. GBR seagrasses are presumably low light-adapted (Pollard and Greenway, 1993), as they have recruited, grown and evolved under disturbance regimes of pulsed increases in turbidity and nutrients, as well as reduced salinity nearshore and physical disturbances that are seasonal and episodic in their extent and amplitude (Birch and Birch, 1984; Devlin et al., 2001).

The GBR region macroalgal flora is diverse with an estimated 400–500 species. However, detailed floristic studies are limited to only a few locations and often only cover intertidal algae (McCook and Price, 1997). The Rhodophyta (red algae) in the ubiquitous turf algae assemblages, which mainly colonise dead coral substratum, are well-described (Price and Scott, 1992). A catalogue of the Phaeophyceae (brown algae) from Queensland has recently been compiled from herbarium records and published material (Phillips and Price, 1997). Apart from algal turfs, prominent macroalgal assemblages in the GBR region include the pavement of crustose coralline algae on windward sides of coral reefs, the epiphytes on seagrass and mangrove roots, and large fleshy macroalgae on nearshore reef flats, often dominated by brown algae and gelatinous red algae. Macroalgae have fundamental ecosystem functions in the GBR region; including provision of habitat, food source and recruitment substratum for benthic fauna and consolidation of reef frameworks.

3. Responses of GBR marine plants to changes in water quality

The three marine plant community types are affected differently by changes in water quality. Due to their proximity to land-based sources of pollution estuarine and coastal mangroves are likely to be the most exposed community type. Seagrass and macroalgal communities occur across the continental shelf and hence across a diminishing gradient of exposure to land-based

pollutants. Knowledge of water quality responses is, however, available for only a few community types, e.g. for estuarine mangroves, coastal seagrass (with limited information for reef-associated seagrass) and reef-associated macroalgae. Unfortunately, this limits inferences about responses across community types.

3.1. Mangrove responses

In addition to mangroves being affected by changed water quality, which is discussed below, the disturbance/destruction of mangrove wetlands may in turn have consequences for coastal water quality, possibly extending offshore. Mangroves have the capacity to hold and bind sediment (Furukawa et al., 1997; Wolanski et al., 1997), dissolved nutrients and carbon (Alongi and McKinnon, in press), although they are also a major source of organic material through export of mangrove litter (Alongi and McKinnon, in press). The sediment and nutrient trapping capacity allows coastal areas to be relatively free of turbid waters and suitable for coral reef and seagrass meadow growth (Morell and Corredor, 1993; Wolanski et al., 1997).

Nutrient loads are considered to be generally high in coastal and estuarine regions where most mangroves occur. Loads are often excessive in areas associated with increased human activities, e.g. from sewage inputs and fertiliser run-off from upstream croplands (Baker et al., 2003). Mangrove responses to nutrients are complex, with examples of both enhanced growth and associated dieback. While excess nutrients alone have not been directly linked with mangrove dieback and damage, indirect links do exist. Excess nutrients have been associated with dieback of *Avicennia marina* in South Australia. The dieback began six years after establishment of a sewage outfall; 250 ha of mangroves died since 1956 and dieback is ongoing (EPA, 1998). The dieback was attributed to smothering by algae of the genus *Ulva*, assumed to retard growth of mangrove seedlings, and to smother and kill aerial roots of established mangrove trees. Algal blooms were also proposed as the cause of a more recent mangrove dieback in Moreton Bay, Queensland (Laegdsgaard and Morton, 1998). We include 'nutrient excess' as an indirect and unintended, less obviously human-related cause, as one of 12 likely drivers of change influencing mangrove ecosystems (Table 1).

As with the nuisance algae, nutrients enhance growth of mangrove plants. This has been demonstrated in nutrient enrichment experiments, which also showed that nitrogen and phosphorus were growth-limiting differently at lower and higher intertidal positions (Boto and Wellington, 1984). Nutrients derived from sewage discharge can be beneficial for growth and productivity of mangroves (Clough et al., 1983). However, under high nutrient demand other chemicals, such as herbi-

cides, may be taken up at greater rates along with extra nutrients (Duke et al., 2003a). This synergistic effect of increased nutrients has resulted in the increased phytotoxicity of specific herbicides (Hatzios and Penner, 1982). High nutrient levels may also alter faunal communities which might affect the vulnerable trophic links between mangrove trees and fauna (Robertson et al., 1992).

Increased sediment loads in runoff from catchments affect mangrove distributions within estuaries as well as water quality (Duke et al., 2003b). In recent decades, there have been unprecedented gains in mangrove areas at the mouths of at least four GBR river estuaries, Trinity Inlet (Duke and Wolanski, 2001), Johnstone River (Russell and Hales, 1994), Pioneer River (Duke and Wolanski, 2001) and Fitzroy River (Duke et al., 2003b). Although these rivers occupy a broad range of climatic and geographic conditions, they each have characteristic and significant new mangrove stands. At the mouth of the Fitzroy River, the area of mangroves had been relatively constant for a century but increased rapidly after the 1970s. The increases were correlated with concurrent human activities including increased clearing of vegetation in the catchment, which increased sediment loads in runoff, and the construction of a major river barrage, which reduced river flows and flushing.

Agricultural chemicals in runoff appear to affect mangrove health in the GBR region. In the Mackay region, the unusual species-specific dieback of *Avicennia marina*, first observed in the mid 1990s, affected by 2000 > 30 km^2 of mangroves in at least five adjacent estuaries. Agricultural chemicals applied on adjacent catchments are reported to be transported downstream into estuarine and nearshore water and sediments (White et al., 2002; McMahon et al., in press). Strong correlative and causative evidence implicates that herbicides used in sugar cane production, particularly diuron, have seriously affected a non-target mangrove species in downstream estuarine habitats (Duke and Bell, in press; Duke et al., 2003a). In 2000 and 2002, diuron and other herbicides were present in mangrove sediments (up to 8 µg/kg for diuron) and porewater (up to 14 ng/L for diuron), and in stream/drain waters flowing into mangrove areas (up to 1.2 µg/L for diuron). Monitoring by the local Healthy Waterways program reported the first high flow event of the 2001–2002 wet season alone brought 470 kg of diuron, at up to 8.5 µg/L, into the Pioneer River estuary (White et al., 2002). Key indicators of mangrove plant health were significantly correlated with diuron concentrations in sediments at the base of trees, planthouse trials demonstrated that salt-excreting mangrove plants such as *A. marina* were more affected by herbicides than salt-excluding species, and the effects were species-specific (Bell and Duke, in press). To date, there had been no substantive evidence for a primary causative agent other than diuron. Kirkwood

Table 1
Types and drivers of change/impacts on Australian mangrove and salt marsh tidal wetlands

Type of change	Driver of change	Reference(s)
A. Direct–intended and obviously human related		
(1) Reclamation loss. Replacement with structures/sites—ports, industrial, urban, canals.	Port, industry and urban development.	Duke (1997) and Duke et al. (2003b)
(2) Direct damage. Dieback/damage/loss caused by cutting, root exposure, sediment disturbance, root burial, ponded pastures and agricultural encroachment.	Access to, construction of retaining walls for ponded pastures and tide blocking drains.	Duke et al. (2003b)
B. Direct–Unintended and obviously human related		
(3) Restricted tidal exchange. Dieback/ damage associated with construction and development projects often resulting in impoundment inundation of breathing roots.	Constructions, like roads and sea walls, altering water flow and tidal exchange.	Olsen (1983) and Gordon (1987)
(4) Spill damage. Dieback/damage following incidents/ accidents involving spills of toxic chemicals, which smother breathing surfaces.	Spillage of toxic chemicals, oil spills.	Duke and Burns (2003) and Duke et al. (2000)
C. Indirect–unintended and less obviously human related		
(5) Depositional gains and losses, e.g. at estuary mouths and areas behind groins and training walls, dieback/damage associated with sediment burial.	Catchment vegetation clearing, soil disturbance, and construction of river/shoreline training walls.	Duke (1997), Duke and Wolanski (2001) and Duke et al. (2003b)
(6) Nutrient excess. Dieback/damage associated with excess algal growth on breathing roots.	Inputs of fertiliser and sewage.	Dennison and Abal (1999), Laegdsgaard and Morton (1998), and EPA (1998)
(7) Species-specific effects. Dieback/damage of species sensitive to toxic chemicals.	Inputs of toxic chemicals, e.g. in catchment runoff.	Duke et al. (2003a) and Duke and Bell (in press)
D. Not obviously human related		
(8) Wrack accumulation. Dieback/damage associated with build-up of beach wrack on breathing roots and localised impoundment.	Post-storm and algal bloom debris accumulation, possibly associated with poor water quality.	Duke and Pedersen (personal observation) and Duke et al. (2003b)
(9) Herbivore/insect attack. Dieback/damage associated with excessive herbivore/insect attacks on foliage or tree stems.	Effects on herbivore/insect, possibly associated with stressed habitat.	Robertson and Duke (1987) and Duke (2002)
(10) Storm damage. Dieback/damage associated with severe storm activity and incidents.	Severe storms, cyclonic winds, strong wave activity, high stream flows, lightning.	Duke, 2001, Houston (1999) and Duke et al. (2003b)
(11) Ecotone shift. Dieback/damage associated with climate change—shifts within the tidal zone.	Climate (rainfall) change affected by local and/or global factors.	Fosberg (1961) and Duke et al. (2003b)
(12) Zonal shift. Dieback/damage associated with sea level change in the entire tidal wetland (mangrove/saltmarsh) zone.	Sea level change affected by local and/or global factors.	Duke et al. (2003b)

and Dowling (2002) acknowledged the severe and extensive dieback, but failed to present any data or observations to justify their assertion that sediment smothering, not diuron, was the most likely cause of the dieback.

Based on observations of mangrove condition and sediment herbicide levels in the Mackay region and the Johnstone and Daintree rivers (Duke et al., 2003a), a threshold concentration of 2 µg diuron/kg sediment was determined, above which *A. marina* dieback might be expected. In these estuaries, it was notable that *A. marina* was either absent, unhealthy, or dead, where diuron concentrations in sediments were >2 µg/kg, corresponding with concentrations in concurrently collected water samples of >4 ng/L, (Duke et al., 2003a). Beyond the coastal margin, seagrass and coral growth might also be affected by diuron at concentrations detected in sediments and waters along the GBR coastline (Haynes et al., 2000a). Corals show signs of photosynthetic stress at diuron concentrations of ⩾1 µg/L (Jones et al., 2003), coralline algae ⩾1 µg/L (Harrington et al., in press), and seagrass ⩾0.1 µg/L (Haynes et al., 2000a).

In contrast to most terrestrial plants mangroves are able to accumulate heavy metals before exhibiting signs of stress (Yim and Tam, 1999). In metal-polluted environments, plants may display visible toxicity symptoms such as leaf discolouration, chlorosis and necrosis, or may not have visible symptoms at all (Bargagli, 1998). Arsenic (As), cadmium (Cd), copper (Cu), mercury (Hg) and zinc (Zn) are potential toxins and can enter the environment as a consequence of agricultural activity (Haynes and Johnson, 2000) such as the use of inorganic phosphate fertilisers (Bargagli, 1998). While some are essential elements (Zn, Cu, Ni and Cr), others are not required by plants and are toxic at high concentrations (Pb and As) (Raven et al., 1992). The presence of detectable concentrations of Cd and Hg in recently deposited marine sediments strongly suggests that agricultural soil is getting to the near-shore environment (Cavanagh et al., 1999). However, concentrations detected in field surveys (Mackey and Hodgkinson, 1995; Duke et al., 2003a) are well below those known to affect mangrove plants in planthouse experiments (MacFarlane and Burchett, 2001).

Oil spills are considered a major threat to mangroves (Duke and Burns, 2003). The first casualty when oil enters a mangrove forest is benthic fauna (e.g. crabs, burrowing lobsters and worms), which die on the day of oiling (Duke et al., 2000). By contrast, mature trees take longer to die, possibly taking a year in some cases depending on seasonal fluctuations in growth, and disrupted linkages with crabs recycling litter below-ground. Mangrove infauna densities are significantly lower in oiled sediments indicating impaired functioning of mangrove ecosystems in chronically oiled locations of major shipping ports, harbours and marinas. Furthermore

locations in SE Queensland with high petroleum hydrocarbon concentrations in sediments were also correlated with high frequencies of a lethal recessive gene in mangrove trees greatly reducing photosynthetic productivity of otherwise viable progeny (Duke and Watkinson, 2002).

3.2. Seagrass responses

Most knowledge of seagrass ecology is from studies on structurally large species of the North-West Atlantic, Mediterranean Sea and Caribbean, which form perennial meadows of high biomass (Duarte, 1999). Although structurally large seagrasses inhabit the GBR region, small species (e.g. *Halodule* and *Halophila*) comprise the majority of the coastal nearshore seagrass meadows (Lee Long et al., 1993). Responses to water quality demonstrated for structurally large seagrasses might not be the same for small seagrasses forming ephemeral, low biomass meadows (Mellors et al., 2002). Recovery from disturbance also differs between structurally large and small seagrasses (Gordon et al., 1994; Longstaff and Dennison, 1999).

Globally, loss of seagrass, at the meadow level, has been linked to declining water quality (McComb et al., 1981; Dennison et al., 1993). The most common cause of seagrass loss has been the reduction of light availability due to chronic increases in dissolved nutrients leading to proliferation of algae reducing the amount of light reaching the seagrass, e.g. phytoplankton, macroalgae or algal epiphytes on seagrass leaves and stems (Cambridge and McComb, 1984; Short and Wyllie-Echeverria, 1996) or chronic and pulsed increases in suspended sediments and particles leading to increased turbidity (Walker and McComb, 1992; Preen et al., 1995; Longstaff and Dennison, 1999). In addition, changes of sediment characteristics may also play a critical role in seagrasses loss (Hemminga and Duarte, 2000).

Abal and Dennison (1996) predicted that detectable impacts on seagrass meadows may occur if higher sediment and associated nutrients were transported into the nearshore areas of the GBR region. To date, no major decline in seagrass abundance in the GBR region has been caused by increased nutrient availability, though localised declines have occurred in the Whitsunday and Hervey Bay areas. In both cases light deprivation was implicated, by (i) algal overgrowth caused by nutrient enrichment from sewage effluent (Campbell and McKenzie, 2001) and (ii) smothering by settling particles and high suspended particle load from flood plumes (Preen et al., 1995; Longstaff and Dennison, 1999). In contrast, the expansion of seagrass meadows around Green Island off Cairns since the 1970's is attributed to an increase in nutrients (Udy et al., 1999), and a recent study indicated that the nitrogen from terrestrial

fertilizer runoff was assimilated by these seagrasses (McKenzie et al., 2004). GBR seagrasses usually recover from transient impacts, with recovery of subtidal and intertidal meadows after flood-related loss observed within two years (Preen et al., 1995; McKenzie et al., 2000; Campbell and McKenzie, 2004). In contrast, meadows around Green Island showed a shift in species composition in the recovering community after experimental disturbance, possibly facilitated by high nutrient availability at this site (Rasheed, 2004).

Generally, seagrasses are nutrient-limited (Duarte, 1999) and thus increases in nutrient availability promote seagrass growth. The role of nutrients in controlling seagrass growth has been evaluated through in situ nutrient enhancement experiments (Dennison et al., 1987; Erftemeijer et al., 1994; Udy and Dennison, 1997; Udy et al., 1999). Such experiments indicate that Australian seagrasses are nitrogen- rather than phosphorus-limited (Udy and Dennison, 1996, 1997; Udy et al., 1999; Walker et al., 1999). A nutrient addition experiment with *Halophila ovalis* in the central GBR supported this for the growing season of this species when increased nitrogen demand is expected (Mellors, 2003). Contrastingly, *H. ovalis* biomass did not respond to nutrient addition during the slow growing season but tissue nutrient levels increased, indicating luxury uptake (Mellors, 2003). Tissue nutrients in *H. ovalis* in the central GBR have markedly increased over a 25-year period, corresponding with increased fertiliser usage in the adjacent Burdekin River catchment (Mellors et al., in press). Hence, *H. ovalis* may be a potential bio-indicator of nutrients in GBR waters. The consequences for seagrass health of higher tissue nutrients are unknown. Factors other than nutrient availability may limit growth, such as temperature and light availability, the latter also negatively affected by land runoff.

Global critical values for porewater and plant tissue nutrients have been devised, below which nutrient limitation of growth is assumed. These are $100 \mu M$ and $1 \mu M$ for porewater ammonium (Dennison et al., 1987) and phosphate (Erftemeijer et al., 1994), respectively, and 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte, 1990). Porewater concentrations from the GBR region are on average lower than the global critical values, while plant tissue nutrient concentrations are higher (Mellors et al., in press). We suggest that the global tissue nutrient values, derived mainly from temperate and structurally large species, are not applicable to GBR species.

Herbicide exposure was implicated in the slow recovery of seagrass meadows in Hervey Bay lost due to the impact of flood plumes. Seagrass photosynthesis is rapidly inhibited after water column exposure of $0.1 \mu g/L$ to $100 \mu g/L$ of the herbicides diuron, atrazine and simazine (Haynes et al., 2000b; Macinnis-Ng and Ralph, 2003a). Seagrasses showed varying abilities to recover after exposure, depending on the species and exposure levels (Haynes et al., 2000b). In general, plants in situ recovered more rapidly than those exposed in the laboratory, indicating that seagrasses in natural environments are not as strongly affected by herbicide exposure as those in laboratory conditions (Macinnis-Ng and Ralph, 2003a). Seagrasses from a variety of field locations responded and recovered at different rates suggesting that there may be other factors modulating the effects of herbicides.

The Australian low risk trigger value (water column) for diuron is $0.2 \mu g/L$, for protection of 95% of species in a slight to moderately disturbed aquatic environment (ANZECC, 2000). In Hervey Bay this limit was not exceeded (McMahon et al., in press). There is no Australian guideline for pesticides in sediments. Highest concentrations of herbicides in GBR seagrass sediments are $1.7 \mu g/kg$ for diuron and $0.3 \mu g/kg$ for atrazine (Haynes et al., 2000a). In comparison, levels of diuron in Hervey Bay sediments were slightly lower, while atrazine levels were higher, however, seagrass stress was not detected (McMahon et al., in press). The herbicide Irgarol 1501, used in antifouling paints, was found in *Halodule* spp. tissue in the GBR and *Zostera marina* at south Queensland sites at concentrations potentially inhibiting photosynthesis, with very high values associated with localised areas of high boat use (Scarlett et al., 1999a,b). There is no information about the effects of chronic exposure of tropical seagrasses to low herbicide levels. Chronic levels as well as higher exposure levels during river flood events may reduce growth and reproductive effort, important processes in the recovery of seagrass meadows after disturbance by turbidity and freshwater runoff.

Seagrasses accumulate metals from the environment by rapid uptake through the leaves under both chronic and pulsed exposure (Schroeder and Thorhaug, 1980). The stress response of seagrass in laboratory studies was both species- and contaminant-specific (Ralph and Burchett, 1998; Prange and Dennison, 2000). Photosynthesis of *H. ovalis* was most affected by copper and zinc and least by cadmium and lead (Ralph and Burchett, 1998). In situ experiments with *Zostera capricorni* also showed copper having the greatest effect, followed by zinc, while cadmium and lead exposure had no effect (Macinnis-Ng and Ralph, 2002). Samples exposed to zinc recovered to pre-exposure levels but those exposed to copper did not. Overseas studies indicate a strong relationship between metal concentrations in seagrass tissue and availability in sediment and water column (Lyngby and Brix, 1984). Australian studies have also shown a relationship between distance from metal point sources and tissue concentrations (Ward, 1987). Recent Australian in situ experiments indicate some degree of metal tolerance in a field population of *Zostera capricorni* (Macinnis-Ng and Ralph, 2004). Seagrasses from

an estuary with little coastal development were most sensitive to metals, while seagrasses from more developed estuaries were more tolerant; sensitive seagrass had metal levels similar to tolerant seagrass and background metal levels in sediments were similar between sites. Responses of seagrasses to metal contamination are likely to differ between different species, populations and locations.

The greatest impact of petrochemical pollution is the smothering of intertidal seagrasses by the washed up oil at low tides (Howard et al., 1989). The *Exxon Valdez* oil spill in 1989 caused high mortality of seagrasses below oiled beaches (Juday and Foster, 1990). Long-term effects of this oil spill were decreased densities of shoots and flowering shoots (Dean et al., 1998). Associated with petrochemical spills is the use of dispersants during clean-up operations. Dispersants consist of a surfactant in a carrier or solvent (Macinnis-Ng and Ralph, 2003b). Solvents allow the toxic surfactant to penetrate the waxy protective coating of the seagrass blade, thereby affecting cellular membranes and chloroplasts (Howard et al., 1989). Structurally large seagrasses were stressed more and showed no recovery after experimental exposure to a combination of oil and dispersant than to oil alone (Hatcher and Larkum, 1982). *Halophila ovalis* was more tolerant and showed little difference between exposure to oil, oil and dispersant and dispersant alone (Ralph and Burchett, 1998). In field experiments with *Zostera capricorni*, crude oil heavily impacted experimental plots, while exposure to dispersant alone had no effect (Macinnis-Ng and Ralph, 2003b). In this study, the combined treatment of oil and dispersant was less toxic than the exposure oil only and dispersant only. Comparisons between studies are problematic as different species and different types of oils and dispersants were used. Factors such as contact time, droplet size (Howard et al., 1989) and the presence of bacterial infauna (Atlas, 1995) affect the toxicity of oil emulsions, whereas warmer temperatures enhance emulsifying and ultraviolet radiation enhances toxicity due to photomodification (Thorhaug, 1992; Ren et al., 1994).

3.3. Macroalgal responses

Responses of GBR macroalgae to water quality, in particular nutrients, have been studied since the 1970s. Early nutrient addition experiments in the southern GBR showed increased net community production but usually no increase in macroalgal biomass (Kinsey and Davies, 1979). Hatcher and Larkum (1983) found that some algal communities were nitrogen-limited but grazing controlled their biomass, whereas responses at other sites were species-specific and controlled by a variety of abiotic and biotic factors. Turf algae communities at One Tree Reef did not respond to nutrient enrichment during the ENCORE experiment (Larkum and Koop,

1997). This was attributed to sufficient nitrogen being cycled within the turfs through nitrogen fixation of associated cyanobacteria and trapping of organic matter. Other macroalgae showed no response to the nutrient enrichment in the ENCORE experiment, or only responses of a few parameters such as nutrient uptake rates (Koop et al., 2001). Some of the unexpected results of that study may be explained by nutrient levels at One Tree reef being naturally higher than at other GBR reefs (Hatcher and Hatcher, 1981; Larkum and Steven, 1994), which may satisfy the nutrient demand of most macroalgal species.

Foliose macroalgae are prevalent on many GBR nearshore reefs (Morrissey, 1980; Price, 1989; McCook et al., 1997; Schaffelke and Klumpp, 1997) and species of the genus *Sargassum* form dense, highly productive stands on reef flats (Schaffelke and Klumpp, 1997). These locations have high (albeit episodic) nutrient availability and turbidity, mainly due to flood and resuspension events (Furnas et al., 1995; Devlin et al., 2001), and grazing pressure is low (Williams, 1991). Although rates of productivity and growth are high, several nearshore macroalgal species are nitrogen- or phosphorus-limited (Schaffelke and Klumpp, 1998a,b; Schaffelke, 1999a). During transient pulses of dissolved nutrients, simulating flood events, these species efficiently take up nutrients. Nutrients surplus to immediate metabolic demand are stored in the tissue. After pulses, these nutrient stores sustain increased growth over several weeks.

Sargassum spp. also utilise alternative nutrient sources to sustain high productivity. Particulate organic nutrients that settles on coral reef benthos are taken up by *Sargassum* spp., possibly remineralised by an epiphytic microbial loop (Schaffelke, 1999b). Potential negative effects of organic particles settling on thalli (e.g. shading, anoxia) were outweighed by nutrient uptake from these particles, which increased growth. Particle- or detritus-based foodwebs on coral reefs have only recently gained attention (Johnson et al., 1995; Arias-Gonzalez et al., 1997) and may be important at locations with low herbivore abundance, such as disturbed reefs with low live coral cover. Several nearshore species (*Chnoospora implexa*, *Hormophysa cuneiformis*, *Hydroclathrus clathratus*, *Padina tenuis*, and *Sargassum* spp.) have high alkaline phosphatase activity (APA), an enzyme that releases phosphate from organic phosphorus (Schaffelke, 2001). These species are able to use the organic phosphorus pool in addition to dissolved phosphate. High APA may also compensate for a relative phosphorus limitation in locations subject to nitrogen inputs, for example some GBR nearshore reefs. Macroalgae from coral reefs in Florida and the Caribbean with high nitrogen and low phosphorus availability have even higher APA values (Lapointe et al., 1992).

Several GBR macroalgal species are likely to benefit from increased nutrient availability, for example in loca-

tions exposed to land runoff, due to their high nutrient demand and the ability to use a variety of nutrient forms. Species that are nutrient- sufficient in an oligotrophic environment would not benefit from a higher nutrient availability (Delgado and Lapointe, 1994; Schaffelke, 1999a). Nutrient-limitation is not apparent in the slow-growing reef algae *Chlorodesmis fastigiata* and *Turbinaria ornata* (Schaffelke, 1999a), which are common species on reefs across the GBR shelf (McCook et al., 1997; Schaffelke, personal observation).

GBR nearshore reefs exposed to high levels of sedimentation are unsuitable habitats for crustose coralline algae (Fabricius and De'ath, 2001). This is likely to have important implications for reef-consolidation and recruitment of species that require coralline algae for settlement. Photosynthetic yield of crustose coralline algae is significantly reduced when exposed simultaneously to sedimentation and the herbicide diuron (Harrington et al., in press). In addition, high phosphate levels (from sewage discharge) reduce growth of coralline algae (Björk et al., 1995).

The rapid responses of macroalgae to nutrient pulses have been used in the development of marine bioindicators for nutrient availability. For example, levels of chlorophyll *a*, tissue nitrogen and the amino acid citrulline in *Gracilaria edulis* respond rapidly to increased nutrients and may thus be useful indicators (Costanzo et al., 2000). Nitrogen stable isotope ratios in *Catenella nipae* have been used to track sewage plumes (Udy, unpublished data).

A baseline study of levels of zinc, copper, nickel, lead and mercury in macroalgal species found low levels of contamination (Denton and Burdon-Jones, 1986). *Chlorodesmis fastigiata* was suggested as a suitable organism for monitoring metal levels in macroalgae due to its wide distribution. To our knowledge there is no published information about levels of organochlorine contaminants in GBR macroalgae. Most information on levels and effects of pollutants is from temperate macroalgae (e.g. Lobban and Harrison, 1994; Eklund and Kautsky, 2003).

4. From species responses to ecosystem change

As outlined above, marine plants in the GBR region are often nutrient-limited, indicated by higher nutrient availability promoting growth of mangroves, seagrasses and certain species of macroalgae. However, interactions between higher nutrient availability and exposure to other pollutants, and between water quality parameters and other disturbances, are largely unknown. At the ecosystem level the response of marine plants to changed water quality is further confounded by other biological and abiotic factors that influence the health and production of marine plants. Examples for these are availabil-

ity/loss of substratum/habitat, physical disturbance such as removal and burial, changes in light climate, species-specific differences in responses to water quality changes, competition and grazing. Some of these factors, in turn, can also be affected by changes in water quality.

GBR mangroves are at present predominantly affected by physical disturbance, and in some cases affected by deteriorating water quality. The current condition of mangrove and tidal wetlands may be described and quantified as the extent and rates of change from the past. The responses of mangrove forests may also be used as sensitive and reliable indicators of natural (Duke et al., 1998) as well as human-related change. Twelve types of change have been identified representing most driving factors in Australia (Table 1, more detail in Duke et al., 2003b), which are grouped in four general categories of change:

A. Direct–intended, obviously human related;
B. Direct–un-intended, obviously human related;
C. Indirect–un-intended, less obviously human related; and
D. Not obviously human related.

A summary of 16 reported incidents impacting mangroves in the GBR region indicated that 50% are related to impacts on water quality (oil spills, herbicide exposure, sediment deposition), 31% directly to reclamation and direct damage through planned and permitted activities, and a total of 81% were the result of human activities (Table 2) and might be considered preventable.

Studies on responses of seagrasses to pollutants indicate that species traits as well local site history determine the response. The geographic setting of a location determines its sediment regime, while the frequency of disturbance determines meadow structure. Factors affecting the sediment regime at each location are: distance from major rivers, wind regime, frequency and magnitude of rainfall events, resuspension and other factors that affect light availability, sediment mineralogy and grain size. These factors determine sediment ion exchange, pH, redox potential, salinity, organic content and presence of bacteria, which all influence the nutrient/pesticide/ heavy metal regime. Tolerances to environmental and anthropogenic perturbations are different for different species and populations (Waycott, 1998), possibly leading to a loss of genetic diversity due to anthropogenic impacts (Alberte et al., 1994).

Compared to mangroves and seagrasses, we know more about ecosystem factors controlling macroalgal biomass on coral reefs. The mosaic of benthic macroalgal assemblages is dynamically determined by the balance of algal production (enhanced by higher nutrient availability for a number of macroalgal species) and grazing pressure, which is sporadically disrupted by

Table 2
Reported instances of disturbance of GBR mangrove ecosystems, ultimately affecting coastal water quality in the GBR region. See Table 1 for details of types of change

Location	Incident	Type of change	Reference(s)
Cape Flattery	Dieback of ~11 ha in 1992	Spill damage	Duke and Burns (2003)
Yorkeys Knob	Dumping of oily waste in 1994	Spill damage	Burns and Codi (1998)
Trinity Inlet	Loss of mangrove area overall, localised increase at mouth, 1945–1995	Reclamation and depositional gains	Duke and Wolanski (2001)
Trinity Inlet	Impoundment of large area for conversion to agricultural lands	Reclamation and direct damage	Duke and Wolanski (2001)
Johnstone River	Increase in mangrove area at river mouth	Depositional gain	Russell and Hales (1994)
Johnstone River	Herbicides in mangrove sediments	Species-specific effect (absence of *Avicennia*)	Duke et al. (2003a)
Hinchinbrook Channel	Loss of foreshore mangroves at Oyster Point	Reclamation and direct damage	Duke (1994)
Hinchinbrook Channel	Increase in mangrove area with corresponding loss of salt marsh, 1941–1991	Ecotone shift—possible increased rainfall	Ebert in Duke (1997)
Pioneer River	Loss of mangrove area, 1945–1995	Reclamation	Duke and Wolanski (2001)
Mackay region	Widespread dieback of *Avicennia marina*. Herbicides from runoff in sediments, esp. diuron	Species-specific effect	Duke et al. (2003a) and Duke and Bell (in press)
Fitzroy River	Extensive defoliation of *Excoecaria* trees in 1995.	Insect herbivory	McKillup and McKillup (1997)
Fitzroy River	Increase in mangrove 'islands' at river mouth from 1890 to 2001	Depositional gain	Duke et al. (2003b)
Fitzroy Estuary and Port Curtis	Loss of tidal wetlands for port and industrial development	Reclamation	Duke et al. (2003b)
Port Curtis	Dieback of *Avicennia marina* in the early 1970s	Species-specific effect (unknown cause)	Saenger in Duke et al. (2003a)
Port Curtis	Dieback of *Rhizophora*, *Ceriops* and *Aegiceras* in 1992	Hail storm damage	Houston (1999)
Port Curtis	Massive foliage loss in *Rhizophora stylosa* canopies, 1995–1998	Insect herbivory	Duke (2002)

Table 3
Summary of reported effects of water pollutants and physical disturbance on macrophytes in the Great Barrier Reef region (see text for further details and references)

	Macrophyte community type						
	Mangroves		Seagrasses		Macroalgae		
Nutrient availability	$\uparrow\downarrow$	Correlation of tissue nutrient content and availability Increased growth Increased epiphyte cover on roots Threshold effect?	$\uparrow\downarrow$	Correlation of tissue nutrient content and availability Increased growth Increased epiphyte cover on leaves Threshold effect?	$\uparrow\downarrow$	Correlation of tissue nutrient content and availability Increased growth Increased epiphyte cover Threshold effect?	
Sediments/suspended solids	$\downarrow$	Burial, smothering	$\downarrow$	Burial, smothering Decreased light availability	$\uparrow\downarrow$	Burial, smothering Decreased light availability Potential nutrient source	
Herbicides	$\downarrow$	Local mortality of sensitive species	$\downarrow$	Limited data Inhibition of photosynthesis	$\downarrow$	Limited data Inhibition of photosynthesis, synergism with sediment	
Heavy metals	$\downarrow$	Limited data on tissue metal content Species and site-specific tolerance?	$\downarrow$	Limited data on tissue metal content Inhibition of photosynthesis in exposure experiments Species and site-specific tolerance?	$\downarrow$	Limited data Low tissue levels	
Oil spills	$\downarrow$	Mortality/damage by smothering of breathing roots	$\downarrow$	Mortality by smothering Inhibition of photosynthesis by toxic effects of oil and dispersants	?	No data	
Reclamation/alteration of habitat	$\downarrow$	Mortality/damage by removal and coastal development	$\downarrow$	Mortality/damage by removal Slow recolonisation (years) after transient disturbance	$\downarrow$	Mortality/damage by removal Quick recolonisation (months) after transient disturbance	

Arrows indicate positive $\uparrow$ or negative $\downarrow$ effects on macrophyte health and production, black for strong, grey for weaker effects.

disturbances that make available new substratum for algal colonisation. Availability of new substratum for macroalgal settlement is generally due to a reduction in coral cover resulting from physical disturbance, coral bleaching, coral disease and crown-of-thorns starfish predation (Szmant, 2002). Increased substratum availability can also result from high inputs of terrestrial sediments and organic matter leading to increased coral mortality (Rogers, 1990; Stafford-Smith and Ormond, 1992), reduced removal of macroalgae by herbivorous fishes (Purcell, 2000), and reduced recruitment and growth of coralline algae which are an important substratum for recruitment of corals (Fabricius and De'ath, 2001). Previously, macroalgal blooms have been attributed to either 'bottom-up' (nutrient enrichment) or 'top-down' (grazer) control (e.g. Larned and Stimson, 1996; Lapointe, 1997; Lapointe et al., 1997; Hughes, 1994). However, anthropogenic nutrient-enrichment often coincides with low grazing pressure (Littler and Littler, 1984; Delgado and Lapointe, 1994; Lapointe et al., 1997), and it is most likely that top-down and bottom-up forces simultaneously control macroalgal biomass (Lapointe, 1999) and that abundant macroalgae are a consequence and not the cause of coral mortality (McCook et al., 2001; Szmant, 2002).

Few studies have tested the combined effects of nutrients and herbivory. Production and tissue nitrogen levels of turf algae increased for a short time after a cyclone, however, no increase in standing biomass was observed due to increased consumption by fish grazers (Russ and McCook, 1999). Overgrowth of corals by *Lobophora variegata* was enhanced by nutrient enrichment under grazer exclusion (Jompa and McCook, 2002). Herbivory affected growth and density of recruits of *L. variegata* and *Sargassum fissifolium*, but nutrient addition only had a minor effect on the former species (Diaz-Pulido and McCook, 2003). Biomass of macroalgae on a Hawaiian fringing reef increased either due to addition of nutrients or exclusion of grazers; macroalgal biomass was greatest with both nutrient addition and grazer exclusion (Smith et al., 2001). A high cover of filamentous macroalgae, in particular *Enteromorpha prolifera* and *Hincksia mitchelliae,* developed rapidly on settlement plates enrichment with phosphorus and within cages that allowed access of small herbivorous fish compared to unfertilised controls (McClanahan et al., 2002). While nutrient addition increases macroalgal growth grazing counteracts this by removing biomass, however, only up to a certain threshold where the resident grazer community is not able to consume any more biomass (Williams et al., 2001). In this situation selective grazing of more palatable or more nutrient-rich species can lead to changes in the composition of the macroalgal community and the establishment of communities with high abundance of species avoided by grazers (Stimson et al., 2001; Lapointe et al., 2004a). For instance, selective grazing and availability of new substratum may have caused development of a community dominated by the unpalatable *Asparagopsis taxiformis* after a ship grounding on the GBR (Hatcher, 1984).

A common theme in overseas case studies of increased macroalgal abundance is that an initial disturbance decreased live coral cover leading to the establishment of a high cover of macroalgae, which is sustained by a combination of high nutrient availability and low grazing pressure (e.g. Lapointe, 1999; Nyström et al., 2000; Stimson et al., 2001; Chazottes et al., 2002), or by high nutrient availability alone (e.g. Lapointe et al., 2004b, and references therein). Water column nutrients in these cases were usually at the high end of the range in GBR waters (e.g. Haynes, 2001). Low coral and high macroalgal cover is also found on some GBR nearshore reefs (Fabricius and De'ath, in press; Fabricius et al., in press), and is most likely sustained by decreased water quality and insufficient grazing pressure.

5. Conclusion

Knowledge of water quality responses is available for only a few macrophyte community types, e.g. for estuarine mangroves, coastal seagrass (with limited information for reef-associated seagrass) and reef-associated macroalgae. Even though this limits conclusions about responses across community types, there are clear indications that declining water quality negatively affects GBR macrophytes (summarised in Table 3). Pollutants such as herbicides, metals and petrochemicals clearly affect seagrass and mangrove health. In contrast, consequences of higher nutrient availability at the ecosystem level are less understood, and are load-, species-, season- and location-dependent. In some cases, high nutrient availability has lead to enhanced growth of valued species such as seagrass and mangroves, which is generally perceived as being positive. In contrast, increased growth of macroalgae in coral reef systems or as epiphytes on seagrass and mangroves is regarded as problematic. High nutrient availability, in conjunction with substrate availability (low coral cover) and insufficient grazing pressure, has lead on some GBR nearshore reef to altered benthic communities with high macroalgal cover. Removal of macroalgae or increase of grazing pressure may facilitate coral recovery. The latter factor, however, is assumed to be at normal rates in the GBR, and without also addressing deteriorated water quality such an intervention could only be an interim measure. The majority of impacts on mangroves are clearly human-related, by physical disturbance and/or water and sediment pollution, and hence might be managed by regulating human activities. Local declines of GBR seagrass have been attributed to light limitation caused

by flood events, which are natural but likely to be exacerbated by higher sediment and nutrient loads exported to the coast. Loss or disturbance of habitat-building macrophytes such as mangroves and seagrasses has serious down-stream effects for coastal water quality due to their capacity to assimilate nutrients and to consolidate sediments.

There is still much to learn about downstream effects of pollutants on marine plants, and how we might quantify and monitor these effects. Pollutant levels in coastal waters are extremely variable, depending on e.g. input loads, distance to waterways, hydrodynamics and catchment characteristics such as soil type, slope, vegetation cover, and rainfall. In contrast, concentrations in sediments appear less variable, especially in estuarine tidal and sub-tidal sediments where e.g. herbicides may adhere to sediments largely composed of fine-grained silts and clays with high levels of organic carbon and minimal porosity. It is currently not possible to evaluate the effects of long-term, chronic, low-level exposure of anthropogenic pollutants on marine plants, compared to short term pulsed, high-level exposure, e.g. during flood events. There is also limited knowledge of synergistic effects between higher nutrient availability and exposure to other pollutants, and between water quality parameters and other disturbances or factors that influence health and production of marine plants. However, these synergies are assumed to be most important for explaining observed changes in macrophyte communities. The lack of regionally relevant knowledge of macrophyte community ecology and of adequate assessments of change over the past century continues to hamper conservation and management of GBR macrophytes and their ecosystems.

Acknowledgments

References

Abal, E.G., Dennison, W.C., 1996. Seagrass depth range and water quality in southern Moreton Bay, Queensland, Australia. Marine and Freshwater Research 47 (6), 763–771.

Alberte, R.S., Suba, G.K., Procacini, G., Zimmerman, R.C., Fain, S.R., 1994. Assessment of genetic diversity of seagrass populations using DNA fingerprinting: implications for population stability and management. Proceedings of the National Academy of Science 91, 1049–1053.

Alongi, D.M., McKinnon, A.D., in press. The cycling and fate of terrestrially-derived sediments and nutrients in the coastal zone of the Great Barrier Reef shelf. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

ANZECC, 2000. Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council, Canberra.

Arias-Gonzalez, J.E., Delesalle, B., Salvat, B., Galzin, R., 1997. Trophic functioning of the Tiahura reef sector, Moorea Island, French Polynesia. Coral Reefs 16, 231–246.

Atlas, R.M., 1995. Petroleum biodegradation and oil spill bioremediation. Marine Pollution Bulletin 31, 178–182.

Australian State of the Environment Committee, 2001. Australia State of the Environment 2001. Independent Report to the Commonwealth Minister for the Environment and Heritage. CSIRO Publishing on behalf of the Department of the Environment and Heritage, Canberra.

Baker, J., Furnas, M., Johnson, A., Moss, A., Pearson, R., Rayment, G., Reichelt, R., Roth, C., Shaw, R., 2003. A report on the study of land-sourced pollutants and their impacts on water quality in and adjacent to the Great Barrier Reef. Report to the Intergovernmental Steering Committee Great Barrier Reef Water Quality Action Plan. Department of Primary Industries, Brisbane. <http://www.deh.gov.au/coasts/pollution/reef/science/index.html> (accessed 08.09.2004).

Bargagli, R., 1998. Trace Elements in Terrestrial Plants: An Ecophysiological Approach to Biomonitoring and Biorecovery. Springer, Berlin.

Bell, A.M., Duke, N.C., in press. Effects of Photosystem II-inhibiting herbicides on mangroves—preliminary toxicology trials. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Birch, W.R., Birch, M., 1984. Succession and pattern of tropical seagrasses in Cockle Bay, Queensland, Australia: a decade of observations. Aquatic Botany 19, 343–367.

Björk, M., Mohammed, S.M., Björklund, M., Semesi, A., 1995. Coralline algae, important coral-reef builders threatened by pollution. Ambio 24, 502–505.

Boto, K.G., Wellington, J.T., 1984. Soil characteristics and nutrient status in a northern Australian mangrove forest. Estuaries 7, 61–69.

Brodie, J., 1997. Nutrients in the Great Barrier Reef region. In: Cosser, P.R. (Ed.), Nutrients in Marine and Estuarine Environments. State of the Environment Technical Paper Series (Estuaries and the Sea). Department of the Environment, Canberra, pp. 9–27.

Burns, K.A., Codi, S., 1998. Contrasting impacts of localised versus catastrophic oil spills in mangrove sediments. Mangroves and Saltmarshes 2, 63–74.

Cambridge, M.L., McComb, A.J., 1984. The loss of seagrass from Cockburn Sound, Western Australia I: the time course and magnitude of seagrass decline in relation to industrial development. Aquatic Botany 20, 229–243.

Campbell, S.J., McKenzie, L.J., 2001. Community based monitoring of intertidal seagrass meadows in Hervey Bay and Whitsunday 1998–2001. DPI Information Series Q101090. Department of Primary Industries, Cairns.

Campbell, S.J., McKenzie, L.J., 2004. Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. Estuarine, Coastal and Shelf Science 60, 477–490.

Cavanagh, J.E., Burns, K.A., Brunskill, G.J., Coventry, R.J., 1999. Organochloride pesticide residues in soils and sediments of the Herbert and Burdekin river regions, North Queensland—implications for contamination of the Great Barrier Reef. Marine Pollution Bulletin 39, 367–375.

Chazottes, V., Le Campion-Alsumard, T., Peyrot-Clausade, M., Cuet, P., 2002. The effects of eutrophication-related alterations to coral reef communities on agents and rates of bioerosion (Reunion Island, Indian Ocean). Coral Reefs 21, 375–390.

Clough, B.F., Boto, K.G., Attiwill, P.M., 1983. Mangroves and sewage: a re-evaluation. In: Teas, H.J. (Ed.), Biology and Ecology of Mangroves. Junk Publishers, The Hague, pp. 151–162.

Coles, R.G., McKenzie, L., Campbell, S., 2003. The seagrasses of eastern Australia. In: Green, E.P., Short, F.T. (Eds.), World Atlas of Seagrasses. UNEP, WCMC, pp. 131–147.

Costanzo, S.D., O'Donohue, M.J., Dennison, W.C., 2000. *Gracilaria edulis* (Rhodophyta) as a biological indicator of pulsed nutrients in oligotrophic waters. Journal of Phycology 36, 680–685.

Dean, T.A., Stekoll, M.S., Jewett, S.C., Smith, R.O., Hose, J.E., 1998. Eelgrass (*Zostera marina* L.) in Prince William/sound, Alaska: effects of the Exxon Valdez oil spill. Marine Pollution Bulletin 36, 201–210.

Delgado, O., Lapointe, B.E., 1994. Nutrient-limited productivity of calcareous versus fleshy macroalgae in a eutrophic, carbonate-rich tropical marine environment. Coral Reefs 13, 151–159.

Dennison, W.C., Abal, E.G., 1999. Moreton Bay Study. A scientific basis for the Healthy Waterways Campaign. South East Queensland Regional Water Quality Management Strategy, Brisbane.

Dennison, W.C., Aller, R.C., Alberte, R.S., 1987. Sediment ammonium availability and eelgrass (*Zostera marina*) growth. Marine Biology 94, 469–477.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, J.C., Carter, V., Koller, S., Bergstrom, P.W., Batuik, R.A., 1993. Assessing water quality with submersed aquatic vegetation: habitat requirements as barometers of Chesapeake Bay Health. Bioscience 43, 86–94.

Denton, G.R.W., Burdon-Jones, C., 1986. Trace metals in algae from the Great Barrier Reef. Marine Pollution Bulletin 17, 98–107.

Devlin, M., Brodie, J., in press. Flood processing of river material in Great Barrier Reef waters. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Devlin, M., Waterhouse, J., Taylor, J., Brodie, J., 2001. Flood plumes in the Great Barrier Reef: spatial and temporal patterns in composition and distribution. Great Barrier Reef Marine Park Authority Research Publication No. 68. Great Barrier Reef Marine Park Authority, Townsville.

Diaz-Pulido, G., McCook, L.J., 2003. Relative roles of herbivory and nutrients in the recruitment of coral-reef seaweeds. Ecology 84, 2026–2033.

Duarte, C.M., 1990. Seagrass nutrient content. Marine Ecological Progress Series 67, 201–207.

Duarte, C.M., 1999. Seagrass ecology at the turn of the millennium: challenges for the new century. Aquatic Botany 65, 7–20.

Duke, N.C., 1994. The impact of clearing mangroves on the World Heritage Area and the status and possible rehabilitation of mangroves at Oyster Point. World Heritage and Biodiversity Branch, Australian Department of Environment, Sport and Territories, Canberra.

Duke, N.C., 1997. Mangroves in the Great Barrier Reef World Heritage Area: current status, long-term trends, management implications and research. In: Wachenfeld, D., Oliver, J., Davis, K. (Eds.), State of the Great Barrier Reef World Heritage Area Workshop. Great Barrier Reef Marine Park Authority, Townsville, pp. 288–299.

Duke, N.C., 2001. Gap creation and regenerative processes driving diversity and structure of mangrove ecosystems. Wetlands Ecology and Management 9, 257–269.

Duke, N.C., 2002. Sustained high levels of foliar herbivory of the mangrove *Rhizophora stylosa* by a moth larva *Doratifera stenosa* (Limacodidae) in north-eastern Australia. Wetlands Ecology and Management 10, 403–419.

Duke, N.C., Bell, A.M., in press. Herbicides implicated as the cause of serious mangrove dieback in the Mackay region, NE Australia—serious implications for marine plant habitats of the GBR World Heritage Area. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Duke, N.C., Burns, K.A., 2003. Fate and effects of oil and dispersed oil on mangrove ecosystems in Australia. In environmental implications of offshore oil and gas development in Australia: further research. A compilation of three scientific marine studies. Australian Petroleum Production and Exploration Association (APPEA), Canberra, pp. 232–363. Available from: <http://www.marine.uq.edu.au/marbot/publications/paperschapters2003.htm>.

Duke, N.C., Watkinson, A.J., 2002. Genetic abnormalities in *Avicennia marina* and longer term deterioration of mangroves growing in oil polluted sediments. Marine Pollution Bulletin 44, 1269–1276.

Duke, N.C., Wolanski, E., 2001. Muddy coastal waters and depleted mangrove coastlines—depleted seagrass and coral reefs. In: Wolanski, E. (Ed.), Oceanographic Processes of Coral Reefs. Physical and Biology Links in the Great Barrier Reef. CRC Press LLC, Washington, DC, pp. 77–91.

Duke, N.C., Ball, M.C., Ellison, J.C., 1998. Factors influencing biodiversity and distributional gradients in mangroves. Global Ecology and Biogeography Letters 7, 27–47.

Duke, N.C., Burns, K.A., Swannell, R.P.J., Dalhaus, O., Rupp, R.J., 2000. Dispersant use and a bioremediation strategy as alternate means of reducing the impact of large oil spills on mangrove biota in Australia: the Gladstone field trials. Marine Pollution Bulletin 41, 403–412.

Duke, N.C., Bell, A.M., Pedersen, D.K., Roelfsema, C.M., Godson, L.M., Zahmel, K.N., MacKenzie, J., Bengston-Nash, S., 2003a. Mackay mangrove dieback. Investigations in 2002 with recommendations for further research, monitoring and management. Final Report to Queensland Fisheries Service, Northern Region (DPI) and the Community of Mackay Region. Centre for Marine Studies, The University of Queensland: Brisbane. Available from <http://www.marine.uq.edu.au/marbot/significantfindings/mangrovedieback.htm>.

Duke, N.C., Lawn, T., Roelfsema, C.M., Phinn, S., Zahmel, K.N., Pedersen, D.K., Harris, C., Steggles, N., Tack, C., 2003b. Assessing historical change in coastal environments. Port Curtis, Fitzroy River Estuary and Moreton Bay regions. Final Report to the CRC for Coastal Zone Estuary and Waterway Management. Historical Coastlines Project, Marine Botany Group, Centre for Marine Studies, The University of Queensland, Brisbane. Available from: <http://www.coastal.crc.org.au/Publications/HistoricalCoastlines.html>.

Eklund, B.T., Kautsky, L., 2003. Review on toxicity testing with marine macroalgae and the need for method standardization-exemplified with copper and phenol. Marine Pollution Bulletin 46, 171–181.

EPA (Environment Protection Authority), 1998. State of the environment report for south Australia 1998—summary report. Prepared in cooperation with the Department for Environment, Heritage and Aboriginal Affairs. Environment Reporting Unit, Adelaide.

Erftemeijer, P.L.A., Stapel, J., Smeckens, M.J.E., Drossaert, W.M.E., 1994. The limited effect of in situ phosphorus and nitrogen additions to seagrass beds on carbonate and terrigenous sediments in South Sulawesi, Indonesia. Journal of Experimental Marine Biology and Ecology 182, 123–140.

Fabricius, K., De'ath, G., 2001. Environmental factors associated with the spatial distribution of crustose coralline algae on the Great Barrier Reef. Coral Reefs 19, 303–309.

Fabricius, K.E., De'ath, G., in press. A framework to identify ecological change and its causes: a case study on the effects of terrestrial run-off on coral reefs. Ecological Applications.

Fabricius, K., Wolanski, E., 2000. Rapid smothering of coral reef organisms by muddy marine snow. Estuarine, Coastal and Shelf Science 50, 115–120.

Fabricius, K.E., Wild, C., Wolanski, E., Abele, D., 2003. Effects of transparent exopolymer particles and muddy terrigenous sediments

on the survival of hard coral recruits. Estuarine, Coastal and Shelf Science 57, 613–621.

Fabricius, K.F., De'ath, G., McCook, L.J., Turak, E., Williams, D.M., in press. Coral, algae and fish communities on nearshore reefs of the Great Barrier Reef: ecological gradients along water quality gradients. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Fosberg, F.R., 1961. Vegetation-free zone on dry mangrove coastline. United States Geological Survey Professional Papers 424(D), 216–218.

Furnas, M., 2003. Catchments and Corals: Terrestrial Runoff to the Great Barrier Reef. Australian Institute of Marine Science, Townsville. 334 pp.

Furnas, M.J., Mitchell, A.W., Skuza, M., 1995. Nitrogen and phosphorus budgets for the central Great Barrier Reef shelf. Great Barrier Reef Marine Park Authority Research Publication No. 36. Great Barrier Reef Marine Park Authority, Townsville, 194 pp.

Furnas, M., Mitchell, A., Skuza, M., Brodie, J.E., in press. In the other 90 percent: phytoplankton responses to enhanced nutrient availability in the GBR lagoon. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Furukawa, K., Wolanski, E., Muller, H., 1997. Currents and sediment transport in mangrove forests. Estuarine, Coastal and Shelf Science 44, 301–310.

Gordon, D.M., 1987. Disturbance to mangroves in tropical-arid Western Australia: hypersalinity and restricted tidal exchange as factors to mortality. Journal of Arid Environments 15, 117–145.

Gordon, D.M., Grey, K.A., Chase, S.C., Simpson, C.J., 1994. Changes to the structure and productivity of *Posidonia sinuosa* meadow during and after imposed shading. Aquatic Botany 47, 265–275.

Harrington, L., Fabricius, K.F., Negri, A., in press. Synergistic effects of diuron and sedimentation on the photophysiology and survival of crustose coralline algae. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Hatcher, B.G., 1984. A maritime accident provides evidence for alternate stable states in benthic communities on coral reefs. Coral Reefs 3, 199–204.

Hatcher, A.I., Hatcher, B.G., 1981. Seasonal and spatial variation in dissolved inorganic nitrogen in One Tree Island lagoon. In: Proceedings of the 4th International Coral Reef Symposium, vol. 1. pp. 419–424.

Hatcher, A.I., Larkum, A.W.D., 1982. The effects of short term exposure to Bass Strait Crude Oil and Corexit 8667 on benthic community metabolism in *Posidonia australis* Hook.f. dominated microcosms. Aquatic Botany 12, 219–227.

Hatcher, B.G., Larkum, A.W.D., 1983. An experimental analysis of factors controlling the standing crop of the epilithic algal community on a coral reef. Journal of Experimental Marine Biology and Ecology 69, 61–84.

Hatzios, K.K., Penner, D., 1982. Metabolism of Herbicides in Higher Plants. Burgess Publishing Company, Minnesota.

Haynes, 2001. Great Barrier Reef Catchment Water Quality Current Issues. Great Barrier Reef Marine Park Authority, Townsville, 90 pp.

Haynes, D., Johnson, J.E., 2000. Organochlorine, heavy metal and polyaromatic hydrocarbon pollutant concentrations in the Great Barrier Reef (Australia) environment: a review. Marine Pollution Bulletin 41, 267–278.

Haynes, D., Ralph, P., Prange, J., Dennison, B., 2000a. The impact of the herbicide diuron on photosynthesis in three species of tropical seagrass. Marine Pollution Bulletin 41, 288–293.

Haynes, D., Müller, J., Carter, S., 2000b. Pesticide and herbicide residues in sediments and seagrass from the Great Barrier Reef World Heritage Area and Queensland coast. Marine Pollution Bulletin 41, 279–287.

Hemminga, M.A., Duarte, C.M., 2000. Seagrass Ecology. Cambridge University Press, Cambridge, 298 pp.

Houston, W.A., 1999. Severe hail damage to mangroves at Port Curtis, Australia. Mangroves and Saltmarshes 3, 29–40.

Howard, S., Baker, J.M., Hiscock, K., 1989. The effects of oil and dispersants on seagrasses in Milford Haven. In: Dicks, B. (Ed.), Ecological Impacts of the Oil Industry. John Wiley and Sons Ltd., Chichester, pp. 61–98.

Hughes, T.P., 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean Coral Reef. Science 265, 1547–1551.

Johnson, C.R., Klumpp, D.W., Field, J.G., Bradbury, R., 1995. Carbon flux on coral reefs: effects of large shifts ion community structure. Marine Ecology Progress Series 126, 123–143.

Jompa, J., McCook, L.J., 2002. The effects of nutrients and herbivory on competition between a hard coral (*Porites cylindrica*) and a brown alga (*Lobophora variegata*). Limnology and Oceanography 47, 527–534.

Jones, R.J., Müller, J., Haynes, D., Schreiber, U., 2003. Effects of herbicides diuron and atrazine on corals of the Great Barrier Reef, Australia. Marine Ecology Progress Series 251, 153–167.

Juday, G.P., Foster, N.R., 1990. A preliminary look at effects of the Exxon Valdez oil spill on Green Island Research natural area. Agroborealis 22, 10–17.

Kinsey, D.W., Davies, P.J., 1979. Effects of elevated nitrogen and phosphorus on coral reef growth. Limnology and Oceanography 24, 935–940.

Kirkwood, A., Dowling, R., 2002. Investigation of mangrove dieback, Pioneer River estuary, Mackay. Queensland Herbarium, Environmental Protection Agency, Brisbane.

Koop, K., Booth, D., Broadbent, A., Brodie, J., Bucher, D., Capone, D., Coll, J., Dennison, W., Erdmann, M., Harrison, P., Hoegh-Guldberg, O., Hutchings, P., Jones, G.B., Larkum, A.W.D., O'Neil, J., Steven, A., Tentori, E., Ward, S., Williamson, J., Yellowlees, D., 2001. ENCORE: the effect of nutrient enrichment on coral reefs. Synthesis of results and conclusions. Marine Pollution Bulletin 42, 91–120.

Laegdsgaard, P., Morton, R., 1998. Assessment of health, viability and sustainability of the mangrove community at Luggage Point. Report by WBM Oceanics Australia to the Brisbane City Council, Brisbane.

Lapointe, B.E., 1997. Nutrient thresholds for bottom-up control of macroalgal blooms on coral reefs in Jamaica and southeast Florida. Limnology and Oceanography 42, 1119–1131.

Lapointe, B.E., 1999. Simultaneous top-down and bottom-up forces control macroalgal blooms on coral reefs (Reply to comment by Hughes et al.). Limnology and Oceanography 44, 1586–1592.

Lapointe, B.E., Littler, M.M., Littler, D.S., 1992. Nutrient availability to marine macroalgae in siliciclastic versus carbonate-rich coastal waters. Estuaries 15, 75–82.

Lapointe, B.E., Littler, M.M., Littler, D.S., 1997. Macroalgal overgrowth of fringing coral reefs at Discovery Bay, Jamaica: bottom-up versus top-down control. In: Proceedings of the 8th International Coral Reef Symposium, vol. 1. pp. 927–932.

Lapointe, B.E., Barile, P.J., Yentsch, C.S., Littler, M.M., Littler, D.S., Kakuk, B., 2004a. The relative importance of nutrient enrichment and herbivory on macroalgal communities near Norman's Pond Cay, Exumas Cays, Bahamas: a "natural" enrichment experiment. Journal of Experimental Marine Biology and Ecology 298, 275–301.

Lapointe, B.E., Barile, P.J., Matzie, W.R., 2004b. Anthropogenic nutrient enrichment of seagrass and coral reef communities in the Lower Florida Keys: discrimination of local versus regional

nitrogen sources. Journal of Experimental Marine Biology and Ecology 308, 23–58.

Larkum, A.W.D., Koop, K., 1997. ENCORE, algal productivity and possible paradigm shifts. In: Proceedings of the 8th International Coral Reef Symposium, vol. 1. pp. 881–884.

Larkum, A.W.D., Steven, A.D.L., 1994. ENCORE: The effect of nutrient enrichment on coral reefs. 1. Experimental design and research programme. Marine Pollution Bulletin 29, 112–120.

Larned, S.T., Stimson, J., 1996. Nitrogen-limited growth in the coral reef chlorophyte *Dictyosphaeria cavernosa*, and the effect of exposure to sediment-derived nitrogen on growth. Marine Ecology Progress Series 145, 95–108.

Lee Long, W.J., Mellors, J.E., Coles, R.G., 1993. Seagrasses between Cape York and Hervey Bay, Queensland, Australia. Australian Journal of Marine and Freshwater Research 4, 19–33.

Lee Long, W.J., Coles, R.G., McKenzie, L.J., 2000. Issues for seagrass conservation management in Queensland. Pacific Conservation Biology 5, 321–328.

Les, D.H., Cleland, M.A., Waycott, M., 1997. Phylogenetic studies in Alismatidae II—evolution of marine angiosperms (seagrasses) and hydrophily. Systematic Botany 22, 443–463.

Littler, M.M., Littler, D.S., 1984. Models of tropical reef biogenesis: the contribution of algae. In: Round, F.E., Chapman, D.J. (Eds.), Progress in Phycological Research, vol. 3. Biopress, Bristol, pp. 30–55.

Lobban, C.S., Harrison, P.J., 1994. Seaweed Ecology and Physiology. Cambridge University Press, Cambridge.

Longstaff, B.J., Dennison, W.C., 1999. Seagrass survival during pulsed turbidity events: the effects of light deprivation on the seagrasses *Halodule pinifolia* and *Halophila ovalis*. Aquatic Botany 65, 105–121.

Lyngby, J.E., Brix, H., 1984. The uptake of heavy metals in eelgrass *Zostera marina* and their effect on growth. Ecological Bulletin 36, 81–89.

MacFarlane, G.R., Burchett, M.D., 2001. Photosynthetic pigments and peroxidase activity as indicators of heavy metal stress in the grey mangrove, *Avicennia marina* (Forsk.) Vierh. Marine Pollution Bulletin 42, 233–240.

Macinnis-Ng, C.M.O., Ralph, P.J., 2002. Towards a more ecologically relevant assessment of the impact of heavy metals on the seagrass, *Zostera capricorni*. Marine Pollution Bulletin 45, 100–106.

Macinnis-Ng, C.M.O., Ralph, P.J., 2003a. Short-term response and recovery of *Zostera capricorni* photosynthesis after herbicide exposure. Aquatic Botany 76, 1–15.

Macinnis-Ng, C.M.O., Ralph, P.J., 2003b. In situ impact of petrochemicals on the photosynthesis of the seagrass *Zostera capricorni*. Marine Pollution Bulletin 46, 1395–1407.

Macinnis-Ng, C.M.O., Ralph, P.J., 2004. Variations in sensitivity to copper and zinc among three isolated populations of the seagrass, *Zostera capricorni*. Journal of Experimental Marine Biology and Ecology 302, 63–83.

Mackey, A.P., Hodgkinson, M.C., 1995. Concentrations and spatial distribution of trace metals on mangrove sediments from the Brisbane River, Australia. Environmental Pollution 90, 181–186.

McClanahan, T.R., Cokos, B.A., Sala, E., 2002. Algal growth and species composition under experimental control of herbivory, phosphorus and coral abundance in Glovers Reef, Belize. Marine Pollution Bulletin 44, 441–451.

McComb, A.J., Atkins, R.P., Birch, P.B., Gordon, D.M., Lukateclich, R.J., 1981. Eutrophication in the Peel-Harvey estuarine system, Western Australia. In: Nielsen, B.J., Cronin, L.E. (Eds.), Estuaries and Nutrients. Humana Press, Clifton, NJ, pp. 323–342.

McCook, L.J., Price, I.R., 1997. The state of the algae of the Great Barrier Reef: what do we know? In: Wachenfeld, D., Oliver, J., Davis, K. (Eds.), State of the Great Barrier Reef World Heritage Area Workshop. Great Barrier Reef Marine Park Authority, Townsville, pp. 194–204.

McCook, L.J., Price, I.R., Klumpp, D.W., 1997. Macroalgae on the GBR: causes or consequences, indicators or models of reef degradation? In: Proceedings of the 8th International Coral Reef Symposium, vol. 2. pp. 1851–1856.

McCook, L.J., Jompa, J., Diaz-Pulido, G., 2001. Competition between corals and algae on coral reefs: a review of evidence and mechanisms. Coral Reefs 19, 400–417.

McKenzie, L.J., Roder, C.A., Roelofs, A.J., Lee Long, W.J., 2000. Post-flood monitoring of seagrasses in Hervey Bay and the Great Sandy Strait, 1999: implications for dugong, turtle and fisheries management. Department of Primary Industries Information Series QI00059, DPI, Northern Fisheries Centre, Cairns. 46 p.

McKenzie, L.J., Lee Long, W.J., Yoshida, R., 2004. Green Island seagrass monitoring and dynamics. In: Haynes, D., Schaffelke, B. (Eds.), Catchment to Reef: Water Quality issues in the Great Barrier Reef Region, 9–11 March, 2004, Townsville Conference Abstracts. CRC Reef Research Centre Technical Report No. 53, p. 47.

McKillup, S.C., McKillup, R.V., 1997. An outbreak of the moth *Achaea serva* (Fabr.) on the mangrove *Excoecaria agallocha* (L.). Pan-Pacific Entomologist 73, 184–185.

McMahon, K.M., Bengston-Nash, S., Eaglesham, G., Müller, J.F., Duke, N.C., Winderlich, S., in press. Herbicide contamination and the potential impact to seagrass meadows in south-east Queensland, Australia. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment to Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Mellors, J., 2003. Sediment and nutrient dynamics in coastal intertidal seagrass of north eastern tropical Australia. PhD thesis. James Cook University, Townsville. pp. 284.

Mellors, J., Marsh, M., Carruthers, T., Waycott, M., 2002. Testing the trapping paradigm of seagrass: do seagrasses influence nutrient status and sediment structure in tropical intertidal environments? Bulletin of Marine Science 71, 1215–1226.

Mellors, J., Waycott, M., Marsh, H., in press. Variation in biogeochemical parameters across intertidal seagrass meadows in the central Great Barrier Reef Region. In: Hutchings, P.A., Haynes, D. (Eds.), Proceedings of Catchment of Reef: Water Quality Issues in the Great Barrier Reef Region Conference. Marine Pollution Bulletin.

Morell, J.M., Corredor, J.E., 1993. Sediment nitrogen trapping in a mangrove lagoon. Estuarine, Coastal and Shelf Science 37, 203–212.

Morrissey, J., 1980. Community structure and zonation of macroalgae and hermatypic corals on a fringing reef flat of Magnetic Island (Queensland, Australia). Aquatic Botany 8, 91–139.

Mumby, P.J., Edwards, A.J., Arias-Gonzalez, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H., Wabnitz, C.C.C., Llewellyn, G., 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427, 533–536.

Nyström, M., Folke, F., Moberg, F., 2000. Coral reef disturbance and resilience in a human-dominated environment. Trends in Ecology and Evolution 15, 413–417.

Olsen, H.F., 1983. Biological resources of Trinity Inlet and Bay Queensland. Queensland Department of Primary Industries, Brisbane.

Percival, M.P., Baker, N.R., 1991. Herbicides and photosynthesis. In: Baker, N.R., Percival, M.P. (Eds.), Herbicides. Elsevier Science Publishers, Amsterdam, pp. 1–26.

Phillips, J.A., Price, I.R., 1997. A catalogue of phaeophyta (brown algae) from Queensland, Australia. Australian Systematic Botany 10, 683–721.

Pollard, P., Greenway, M., 1993. Photosynthetic characteristics of seagrasses (*Cymodocea serrulata*, *Thalassia hemprichii* and *Zostera capricorni*) in a low light environment, with a comparison of leaf

marking and lacunal-gas measurements of productivity. Australian Journal of Marine and Freshwater Research 44, 127–140.

Prange, J.A., Dennison, W.C., 2000. Physiological responses of five seagrass species to trace metals. Marine Pollution Bulletin 41, 327–336.

Prasad, M., Strzalka, K., 1999. Impact of heavy metals on photysnthesis. In: Prasad, M.V.N., Hagemeyer, J. (Eds.), Heavy Metal Stress in Plants. Springer, Berlin, pp. 117–138.

Preen, A.R., Lee Long, W.J., Coles, R.G., 1995. Flood and cyclone related loss, and partial recovery, of more than 10000 km^2 of seagrass in Hervey Bay, Queensland, Australia. Aquatic Botany 53, 3–17.

Price, I.R., 1989. Seaweed phenology in a tropical Australian locality (Townsville, North Queensland). Botanica Marina 32, 399–406.

Price, I.R., Scott, F.J., 1992. The turf algal flora of the Great Barrier Reef. Part I. Rhodophyta. James Cook University, Townsville.

Purcell, S.W., 2000. Association of epilithic algae with sediment distribution on a windward reef in the Northern Great Barrier Reef. Bulletin of Marine Science 66, 199–214.

Ralph, P.J., Burchett, M.D., 1998. Photosynthetic responses of *Halophila ovalis* to heavy metal stress. Environmental Pollution 103, 91–101.

Rasheed, M.A., 2004. Recovery and succession in a multi-species tropical seagrass meadow following experimental disturbance: the role of sexual and asexual reproduction. Journal of Experimental Marine Biology and Ecology 310, 13–45.

Raven, P.H., Evert, R.F., Eichhorn, S.E., 1992. Biology of Plants, fifth ed. Worth Publishers, New York.

Ren, L., Huang, X.D., McConkey, B.J., Dixon, D.G., Greenberg, B.M., 1994. Photoinduced toxicity of three polycyclic aromatic hydrocarbons (Fluoranthene, Pyrnene and Naphthalene) to the duckweed *Lemna gibba* L. Ecotoxicology and Environmental Safety 28, 160–171.

Robertson, A.I., Duke, N.C., 1987. Insect herbivory on mangrove leaves in North Queensland. Australian Journal of Ecology 12, 1–7.

Robertson, A.I., Alongi, D.M., Boto, K.G., 1992. Food chains and carbon fluxes. In: Robertson, A.I., Alongi, D.M. (Eds.), Tropical Mangrove Ecosystems. American Geophysical Union, Washington, DC, pp. 293–326.

Rogers, C.S., 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62, 185–202.

Russ, G.R., McCook, L.J., 1999. Potential effects of a cyclone on benthic algal production and yield to grazers on coral reefs across the Central Great Barrier Reef. Journal of Experimental Marine Biology and Ecology 235, 237–254.

Russell, D.J., Hales, P.W., 1994. Stream habitat and fisheries resources of the Johnstone River catchment. Northern Fisheries Centre, Queensland Department of Primary Industries, Brisbane.

Russell, D.J., Hales, P.W., Helmke, S.A., 1996. Fish resources and stream habitat of the Moresby River Catchment. Northern Fisheries Centre, Queensland Department of Primary Industries, Brisbane.

Scarlett, A., Donkin, P., Fileman, T.W., Evans, S.V., Donkin, M.E., 1999a. Risk posed by the antifouling agent Irgarol 1051 to the seagrass, *Zostera marina*. Aquatic Toxicology 45, 159–170.

Scarlett, A., Donkin, P., Fileman, T.W., Morris, R.J., 1999b. Occurrence of the Antifouling Herbicide, Irgarol 1051, within coastal-water seagrasses from Queensland, Australia. Marine Pollution Bulletin 38, 687–691.

Schaffelke, B., 1999a. Short-term nutrient pulses as tools to assess responses of coral reef macroalgae to enhanced nutrient availability. Marine Ecology Progress Series 182, 305–310.

Schaffelke, B., 1999b. Particulate nutrients as a novel nutrient source for tropical *Sargassum* species. Journal of Phycology 35, 1150–1157.

Schaffelke, B., 2001. Surface alkaline phosphatase activities of macroalgae on coral reefs of the central Great Barrier Reef Australia. Coral Reefs 19, 310–317.

Schaffelke, B., Klumpp, D.W., 1997. Biomass and productivity of tropical macroalgae on thre nearshore fringing reefs in the central Great Barrier Reef, Australia. Botanica Marina 40, 373–383.

Schaffelke, B., Klumpp, D.W., 1998a. Nutrient-limited growth of the coral reef macroalga *Sargassum baccularia* and experimental growth enhancement by nutrient addition in continuous flow culture. Marine Ecology Progress Series 164, 199–211.

Schaffelke, B., Klumpp, D.W., 1998b. Short-term nutrient pulses enhance growth and photosynthesis of the coral reef macroalga *Sargassum baccularia*. Marine Ecology Progress Series 170, 95–105.

Schroeder, P.B., Thorhaug, A., 1980. Trace metal cycling in tropical and sub-tropical estuaries dominated by the seagrass *Thalassia testudium*. American Journal of Botany 67, 1075–1088.

Short, F.T., Wyllie-Echeverria, S., 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation 23, 17–27.

Smith, J.E., Smith, C.M., Hunter, C.L., 2001. An experimental analysis of the effects of herbivory and nutrient enrichment on benthic community dynamics on a Hawaiian reef. Coral Reefs 19, 332–342.

Stafford-Smith, M.G., Ormond, R.F.G., 1992. Sediment-rejection mechanisms of 42 species of Australian scleractinian corals. Australian Journal of Marine and Freshwater Research 43, 683–705.

Stimson, J., Larned, S.T., Conklin, E., 2001. Effects of herbivory, nutrient levels, and introduced algae on the distribution and abundance of the invasive macroalga *Dictyosphaeria cavernosa* in Kaneohe Bay, Hawaii. Coral Reefs 19, 343–357.

Szmant, A.M., 2002. Nutrient enrichment on coral reefs: is it a major cause of coral reef decline? Estuaries 25, 743–766.

Thorhaug, A., 1992. Oil spills in the tropics and subtropics. In: Connell, D.W., Hawker, D. (Eds.), Pollution in Tropical Aquatic Systems. CRC Press Inc, London, pp. 99–127.

Tomlinson, P.B., 1986. The Botany of Mangroves. Cambridge, Cambridge University Press, 413 pp.

Udy, J.W., Dennison, W.C., 1996. Estimating nutrient availability in seagrass sediments In: Kuo, J., Phillips, R.C., Walker, D.I., Kirkman H. (Eds.), Seagrass Biology: Proceedings of an International Workshop Rottnest Island, Western Australia, 25–29 January 1996. pp. 163–172.

Udy, J.W., Dennison, W.C., 1997. Growth and physiological responses of three seagrass species to elevated nutrients in Moreton Bay, Australia. Journal of Experimental Marine Biology and Ecology 217, 253–257.

Udy, J.W., Dennison, W.C., Lee Long, W.J., McKenzie, L.J., 1999. Responses of seagrasses to nutrients in the Great Barrier Reef, Australia. Marine Ecology Progress Series 185, 257–271.

Walker, D.I., McComb, A.J., 1992. Seagrass degradation in Australian coastal waters. Marine Pollution Bulletin 25, 191–195.

Walker, D.I., Dennison, W.C., Edgar, G., 1999. Status of seagrass research and knowledge. In: Butler, A., Jernakoff, P. (Eds.), Seagrass in Australia; Strategic review and development of an R&D plan. FRDC, p. 1999.

Ward, T.J., 1987. Temporal variation of metals in the seagrass *Posidonia australis* and its potential as a sentinel accumulator near a lead smelter. Marine Biology 95, 315–321.

Waycott, M., 1998. Genetic variation, its assessment and implications to the conservation of seagrasses. Molecular Biology 7, 793–800.

Waycott, M., Les, D.H., 1996. An integrated approach to the evolutionary study of seagrasses. In: Kuo, J, Phillips, R.C., Walker, D.I., Kirkman, H. (Eds.), Seagrass Biology: Proceedings of an International Workshop, Rottnest Island, Western Australia, 25–29 January 1996. The University of Western Australia, Perth. pp. 71–78.

White, I., Brodie, J., Mitchell, C., 2002. Pioneer River catchment event based water quality sampling. Report by the Healthy Waterways Program to Mackay Whitsunday Regional Strategy Group and Department of Natural Resources and Mines (NRM), Mackay. 32 p.

Wilkinson, C., 2002. Status of the Coral Reefs of the World. Global Coral Reef Monitoring Network Australian Institute of Marine Science, Townsville, 378 pp. Available from: <http://www.gcrmn. org>.

Williams, D.McB., 1991. Patterns and processes in the distribution of coral reef fishes. In: Sale, P.F. (Ed.), The Ecology of Reef Fishes on Coral Reefs. Academic Press, San Diego, pp. 437–474.

Williams, D.McB., 2002. Review of the impacts of terrestrial runoff on the Great Barrier Reef World Heritage Area. CRC Reef Research Centre, Townsville.

Williams, I.D., Polunin, V.C., Hendrick, V.J., 2001. Limits to grazing by herbivorous fishes and the impact of low coral cover on macroalgal abundance on a coral reef in Belize. Marine Ecology Progress Series 222, 187–196.

Wolanski, E., Spagnol, S., Lim, E.G., 1997. The importance of mangrove flocs in sheltering seagrass in turbid coastal waters. Mangroves and Salt Marshes 1, 187–191.

Wolanski, E., Richmond, R., McCook, L., Sweatman, H., 2003. Mud, marine snow and coral reefs. American Scientist 91, 44–51.

Yim, M.W., Tam, N.F.Y., 1999. Effects of wastewater-borne heavy metals on mangrove plants and soil microbial activities. Marine Pollution Bulletin 39, 179–186.