Appendix J Factors Affecting Distributions of Habitat

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Appendix J. Factors Affecting Distribution of Habitat

J.1 Mangroves

The factors which determine the distribution of mangroves and saltmarshes in Queensland include temperature; sediment type; salinity of the soil water; drainage/aeration; degree and frequency of tidal inundation; exposure to currents; and exposure to freshwater.

Mangroves communities grow on a diverse range of sediments from rocky outcrops, to coarse sand, to fine silts and muds. However, they develop best in sheltered depositional environments on fine silts and clays (Hutchings and Saenger 1987). Drainage and aeration depend on sediment characteristics, frequency and period of fresh and saltwater inundation and elevation. Mangrove species differ in their ability to withstand poorly drained or aerated soils. Hutchings and Saenger (1987) produced a tentative grouping of mangroves based on the soil water content in which they grow. The height of some mangroves e.g. *Avicennia marina* appears to depend on drainage properties of the soil, with the tallest trees growing in well drained banks close to streams. Saltmarshes cannot remain vigorous on waterlogged anaerobic soils, and this maybe a major factor limiting their seaward distribution (Hutchings and Saenger 1987).

Salinity of the interstitial water is an important factor regulating, growth, height, survival and zonation of mangroves and saltmarsh plants (Hutchings and Saenger 1987). Salinity of the soil water is dependent on the salinity of the ocean or estuarine water, the period and frequency of inundation, volume and frequency of freshwater inputs, evaporation due to high temperature or wind, soil type and plant cover. In general, saltmarsh species are more tolerant of high salinities than mangroves. Of the mangroves, *Avicennia marina* grows over the largest salinity range. *Aegiceras corniculatum* will also grow over a broad range, although less than *Avicennia marina*. *Rhizophora stylosa, Bruguiera gymnorrhiza* and *Ceriops tagal* are recorded as growing at salinities 3 to 4 times the concentration of seawater (Hutchings and Saenger 1987).

Rural, urban and industrial development all have an affect on one or more of the environmental characteristics discussed above. For example, development adjacent to the landward fringe of mangroves may alter the soil water characteristics within the mangrove stand, and impact on the growth, survival and species composition of a stand.

J.2 Seagrasses

Water clarity, salinity, temperature, currents, exposure, sediment characteristics and nutrients in the water column and in the sediment, are probably the most important factors limiting the distribution of seagrasses.

Like all plants, seagrasses require light to photosynthesise. Light reaching a seagrass bed is a combination of the light intensity at the surface, the depth at which the seagrass is growing, the turbidity of the water, and the presence or absence of epiphytes on the seagrass. The turbidity of water above a seagrass bed can be increased either directly, by adding or resuspending fine sediment in the water column, or indirectly through enhanced nutrient levels which increase phytoplankton density (Shepherd *et al*. 1989). Thus water turbidity can increase due to, for example: floods, dredging, sludge discharge, upstream fertilisation of terrestrial land, and increasing sediment loads resulting from land clearance or fires (Abal and Dennison 1996).

The increase in turbidity associated with dredging and dredge spoil disposal can significantly decrease the penetration of light through the water column. Light availability, or specifically the duration of light intensity exceeding the photosynthetic light saturation point controls the depth distribution of seagrasses (Dennison and Alberte 1985; Dennison 1987; Abal & Dennison 1996). The depth range of seagrasses is a function of water clarity. For example, on average 30% of surface light; a light attenuation co-efficient of less than 1.4m-1 and total suspended solids of less than 10 mg L-1 are required for the survival of *Zostera capricorni* (Longstaff *et al*. 1998; Abal & Dennison 1996). *Halophila ovalis* another common species in the area, has a particularly low tolerance to light deprivation caused by pulsed turbidity such as floods and dredging (Longstaff & Dennison 1998).

Availability of light also affects productivity of seagrasses, Grice et al. (1996) found that seagrass exposed to higher light intensity were more productive than those in less intense light. Therefore dredging may result in lowered productivity of seagrasses, at least temporarily. Light has also been shown to control (under particular circumstances) the population dynamics of macroalgae (Lukatelich and McComb 1986a; cited in Lavery and McComb 1991).

Seagrasses can absorb nutrients through both the leaves and roots. Moderate amounts of additional nutrients in either the water column or in the sediment can increase seagrass growth (McRoy and Helfferich 1980). However, increased nutrient loads may lead to an increase in phytoplankton densities, and consequently a reduction in water clarity and seagrass depth distribution (Dennison *et al*. 1993).

Increased nutrient levels may also lead to an increase in macroalgal growth at the expense of seagrass. Macroalgae are more efficient at absorbing nutrients from the water column than seagrasses (Wheeler and Weidner 1983; Zimmerman and Kremer 1986). Consequently, benthic macroalgae may overgrow and displace seagrass, whilst drift and epiphytic algae may physically shade seagrass and reduce its growth and distribution (Twilley, *et al*. 1985; Silberstein *et al*. 1986; Maier and Pregnall 1990; Tomasko and Lapointe 1991). Epiphytic algae may also reduce diffusive exchange of dissolved nutrients and gases at leaf surfaces (Twilley, *et al*. 1985; Neckles *et al*. 1993).

Changes to sediment characteristics may also lead to changes in seagrass distribution and species composition. *Zostera capricorni* is most commonly found on reduced sandy or muddy sediment (den Hartog 1970). Changes to current regimes are also likely to effect distribution and species composition. Sub-tropical seagrasses, in general, grow in areas of low current, usually embayments or estuaries, or on sheltered coastlines, with each species having different capabilities of withstanding different current regimes. Changes to sediment and current regimes may also result in changes to the length of exposure of seagrass beds. With increasing exposure, desiccation and temperature are likely to increase. Consequently increased exposure during summer may result in a decrease in distribution and density, and will alter community composition.

J.3 Reefal Communities Including Corals

The distribution of rocky reef flora and fauna is determined principally by exposure to wave action, and water quality (in particular turbidity).

J.3.1 Turbidity and Sedimentation

As many of the fauna are filter feeders, high suspended solids levels may significantly effect feeding efficiency; contaminants may also be rapidly taken up through feeding. As for the floral communities of other habitats, suspended solids may also effect the depth to which macroalgae may grow, and community structure.

Hard coral communities fare best in areas characterised by clear, warm waters that are well mixed and free of sediment (Hubbard 1988). Clear water promotes the photosynthetic activity of zooxanthellae hosted by most shallow water corals. Active water movement circulates nutrients, removes wastes, maintains high oxygen levels, and discourages predation by herbivores and coralivores.

Diversity in coral communities may be controlled by physical factors, in particular salinity, turbidity, sedimentation and the availability of suitable substrate (Lovell 1989). Biotic factors including competition with soft corals and macroalgae may also be significant (Johannes *et al*. 1983; Smith & Simpson 1991). Macroalgae is likely to both compete for space (particularly in respect of substrate suitable for recruitment), and to physically inhibit coral growth through abrasion, shading and sediment retention.

Whilst various models for predicting critical levels of turbidity and nutrient enrichment have been proposed (Bell 1992; Lapointe 1997; Te 1997), various studies have shown that different species of corals have very different tolerances to nutrients. Concentrations of dissolved inorganic nutrients are poor indicators of reef status, and the concept of a simple threshold concentration that indicates eutrophication has little validity (McCook 1999)

Coral communities of the Whitsunday coast are influenced at a broad-scale by the discharges of the Proserpine and O’Connell Rivers. Whilst larval recruitment is near random, environmental factors determine which species survive and grow (Fisk and Harriot 1990; van Woesik 1999). These communities chronically experience sediment deposition rates considerably in excess of rates reported to be catastrophic for coral communities in other parts of the world. Despite this these communities continue to flourish and are healthy (e.g. Marshall and Orr, 1931; Rogers, 1990).

Severe, acute impacts (as may be associated with flood events) are likely to strongly influence the overall structure of coral communities, in particular the population age structure and survival of species over the short term. In the short term, such events are also likely to influence the variety of species found at any one location through affecting the availability of suitable substrate and eliminating less tolerant species. The chronic effects of elevated turbidities and sediment deposition, characteristic of an estuarine environment, broadly restrict diversity to those species able to withstand such conditions. Subtle changes in the character of the environment are likely to determine the character of the coral communities over the longer term (decades), and certainly determine the ability of a location to support populations of coral species in a self sustaining manner.

Increased sedimentation and turbidity in tropical coastal waters, related to changed land use, are among the most serious environmental problems facing coral reefs worldwide (van Woesik et al 1999). Negative impacts of sedimentation on the structure and function of coral communities have been shown through a variety of studies (Squires 1962; Loya 1976; Cortes and Risk 1985; van Katwijk et al. 1993; Te 1997). Human activity can also increase nutrient levels and decrease the abundance of grazing animals (Littler et al. 1992).

Increased turbidity in the water column leads to a decrease in photosynthesis and consequently decreased coral productivity. Effects range from mild coral stress to subtle changes in reef community structure, to outright coral mortality and ecological collapse of the reef (Raaymakers and Oliver, 1993). Continual resuspension and transport of sediments can cause reef degradation years after the delivery of the sediments ceases.

The impacts of increases in sediment deposition on coral communities can include reduced algal and coral diversity and reductions in epifaunal densities (Hatcher et al., 1989). The varied biota found associated with coral communities, living or feeding in the crevices and crannies within and around corals are likely to suffer as these spaces are filled by deposited sediments (Johannes, 1975).

Sedimentation is a major controlling factor in the distribution of reef organisms and in overall reef development (Hubbard, 1986; Macintyre, 1988, cited in Rogers, 1990). Reefs are generally better developed, are more diverse, and with greater coral cover and rates of coral growth the lower the sediment load is in overlying waters (Rogers, 1990).

Increased turbidity and sediment deposition can also impact the planktonic food supply of corals (this is unlikely to be significant for hermatypic corals); reduce available sites for larval settlement (Hubbard, 1988; Rogers, 1979; Johannes, 1975); stimulate energy-consuming sediment rejection behaviour and can abrade the coral.

J.3.2 Nutrient Levels

Elevated nutrient levels can also negatively impact coral communities. There is concern on the Great Barrier Reef in particular, that abundant macroalgae on inshore fringing reefs is a result of degradation due to anthropogenic increases in terrestrial inputs of sediments and nutrients (McCook and Price, cited in McCook 1999). Algal dominated reefs usually have lower fish stocks, less tourism appeal and coral biodiversity than coral dominated reefs (McCook 1999). Whilst increased nutrient loads and associated macroalgal blooms pervasively and fundamentally alter estuarine ecosystems (Valiela et al. 1997), the effects of increased nutrient loadings on coral reefs seem equivocal and discordant, and may be confounded by many indirect effects (van Woesik et al 1999).

The response of the corals themselves to increased nutrients is dependent on light and temperature (D'Elia 1977 cited in van Woesik et al 1999). Recent research indicates that increases in nutrients can have sub-lethal impacts on hard corals. In particular elevated nitrogen levels can stunt coral growth and decrease larval settlement (Koop et al 2001). In areas of high nutrient enrichment, corals may be replaced by macroalgae (Lapointe 1997), particularly if this is accompanied by a significant reduction in herbivores (e.g. Hughes et al. 1999; McCook 1999). By reducing growth and larval settlement, elevated nutrients may effectively prevent the recovery of corals that have suffered some form of acute stress (e.g. a bleaching event, flood or cyclone damage).

J.4 Soft Sediment Communities

Soft sediment benthic communities typically have marked fluctuations in both numbers of animals and number of species. Non-biological factors such as temperature and salinity cannot account for these distributions alone: changes between years appear to be more significant than changes between seasons (DEC 1989).

The fauna associated with soft sediment habitats is typically determined by the character of the sediment: its grain size and stability; and with the presence or absence of seagrass. Grain size influences the ability of organisms to burrow, and the stability of ‘permanent’ burrows. Unstable sediments (that is those that are frequently reworked by wave or current action) support less diverse benthic communities than those that are relatively stable. Resuspension of fine sediments can interfere with the feeding and respiration of benthic fauna. Contaminants within the sediments may also adversely affect benthic fauna, particularly those that ingest the sediment in feeding. Vegetated soft sediments support a greater diversity and abundance of organisms than unvegetated sediments.

J.5 References

Abal, E. G. and Dennison, W. C. 1996, ‘Seagrass depth range and water quality in Southern Moreton Bay, Queensland, Australia’, *Marine and Freshwater Research*. Vol. 47 (6).

Ferrell, D. J. & Bell, J. D. 1992, ‘Differences among assemblages of fish associated with *Zostera capricorni* and bare sand over a large spatial scale’, *Marine Ecology Progress Series*. 72:15-24.

den Hartog, C. 1970, *The Seagrasses of the World*, Verhandelingen der Koninklijke Nederlandse Akademie van Wetenschappen Afd. Natuurkunde Tweede Reeks, Deel 59 (1): 1 - 275.

Dennison, W. C. 1987, ‘Effects of light on seagrass photosynthesis, growth and depth distribution’, *Aquatic Botany*, 27: 15 - 26.

Dennison, W. C. & Alberte, R. S. 1985, ‘Role of daily light period in the depth distribution of *Zostera marina* (eelgrass)’, *Marine Ecology Progress Series*, 25:51-61.

Dennison, W. C., Orth, R. J., Moore, K. A., Stevenson, J. C., Carter, V., Kollar, S., Bergstrom P. W. & Batiuk, R. A. 1993, ‘Assessing water quality with submerged aquatic vegetation’, *Bioscience*, 43(2):86-94.

Hubbard, D. K. 1988, ‘Controls of modern and fossil reef development: common ground

Fisk, D. A. & Harriott, V. J. 1990, ‘Spatial and temporal variation in coral recruitment on the Great Barrier Reef: Implications for dispersal hypothesis’, *Marine Biology*, 107:485-90.

Grice, A. M, Loneragan, N. R. & Dennsion, W. C. 1996, ‚Light intensity and the interactions between physiology, morphology and stable isotope ratios in five species of seagrass’, *Journal of Experimental Biology and Ecology*, 195:91-110.

for biological and geological research’, *Proceedings of the Sixth International Coral Reef Symposium*, Australia 1:243-52.

Hughes, T. & Szmant, A. M. 1999, (comment) 'Algal Blooms on coral reefs: What are the causes?', *Limnology and Oceanography*, 44(6):1583-6.

Hutchings, P. & Saenger, P. 1987, *Ecology of mangroves*, Incitec Gypsum Stockpile, 1996, University of Queensland Press, St Lucia, pp. 387.

Johannes, R. E. 1975, ‘Pollution and degradation of coral reef communities’, *Tropical Marine Pollution,* eds E. J. Ferguson-Wood & R. E. Johannes, Elsevier Scientific Publishing, New York, pp. 13-51.

Johannes, R. E., Wiebe, W. J., Crossland, C. J., Rimmer, D. W. & Smith, S. V. 1983, ‘Latitudinal limits of coral reef growth’, *Marine Ecology Progress Series*, 11:105-11.

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 Enrishment on Coral Reefs. Synthesis of Results and Conclusions’, *Marine Pollution Bulletin*, 42(2): 91-120.

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(5, Part 2):1119-31.

Lavery, P. S. & McComb, A. J. 1991, ‘Macroalgal - sediment nutrient interactions and their importance to macroalgal nutrition in a eutrophic estuary’, *Estuarine and Coastal Shelf Science*, 32:281-95.

Longstaff, B. J., Dennison, W. C., Prange, J. A., Loneragan, N. & Drew, E. A. 1998, *Task SLR: Seagrass/Light Relationships*, Final report Submitted to the Brisbane River and Moreton Bay Wastewater Management Study.

Lovell, E. R. 1989, ‘Coral assemblages of Moreton Bay, Queensland, Australia, before and after a major flood’, *Memoirs of the Queensland Museum,* 27(2):535-50

Maier, C. M. & Pregnall, A. M. 1990, ‘Increased macrophyte nitrate reductase activity as a consequence of groundwater input of nitrate through sandy beaches’, *Marine Biology,* 107:263-71.

Marshall, S. & Orr, A. 1931, ‘Sedimentation on Low Isles Reef and it's relation to coral growth’, *Scientific Report of the Great Barrier Reef Expedition,* 1:94-133.

McCook, L. J. 1999, ‘Macroalgae, nutrients and phase shifts on coral reefs: Scientific issues and management consequences for the Great Barrier Reef’, *Coral Reefs,* 18:357-67.

McRoy, C. P, & Helfferich, C. 1980, ‘Applied aspects of seagrasses’, *Handbook of seagrass biology: An ecosystem perspective,* eds R. C. Phillips C. P. McRoy, Garland STPM Press, York and London.

Neckles, H. A., Wetzel, R. L. & Orth, R. J. 1993, ‘Relative effects of nutrient enrichment and grazing on epiphyte - macrophyte (*Zostera marina L.*) dynamics’, *Oecologia*, 93:285-95.

Rogers, C. S. 1979, ‘The effects of shading on coral reef structure and function’, *Journal of Experimental Marine Biology and Ecology*, 41(3): 269-88.

Rogers, C. S. 1990, ‘Responses of coral reefs and reef organisms to sedimentation’, *Marine Ecology Progress Series*, 62:185-202.

Shepherd, S., McComb, A., Bulthuis, D., and others, 1989, ‘Decline of Seagrasses’, Larkum, A.W.D., McComb, A.J., Shepherd, S.A., *Biology of Seagrasses*. A treatise on the biology of seagrasses with special reference to the Australian region, pp 346-393.

Silberstein, K., Chiffings, A. W. & McComb, A. J. 1986, ‘The loss of seagrass in Cockburn Sound, Western Australia, III The effect of epiphytes on productivity of *Posidonia australis* Hook. f.’, *Aquatic Botany*, 24:355-71.

Smith, S. D. A., and Simpson, R. D. 1991. ‘Nearshore corals of the Coffs Harbour region, mid north coast, New South Wales’ *Wetlands*, 11(1): 1-9.

Te, F. T. 1997, ‘Turbidity and its effect on corals: A model using the extinction coefficient (K) of photosynthetic active radiance (PAR)’, *Proceedings of the 8th International Coral Reef Symposium*, 2:1899-904.

Tomasko, D. A. & Lapointe, B. E. 1991, Productivity and biomass of *Thalassia testudinum* as related to water column nutrient availability: field observations and experimental studies, *Marine Ecology Progress series*, 75:9-17.

Twilley, R. R., Kemp, W. M., Staver, K. W., Stevenson, J. C. & Boynton, W. R. 1985, ‘Nutrient enrichment of estuarine submerged vascular plant communities, 1. Algal growth and effects on production of plants and associated communities’, *Marine Ecology Progress Series*, 23:179-91.

Valiela, I., McClelland, J., Hauzwell, J., Behr, P. J., Hersh, D. & Foreman, K. 1977, ‘Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem consequences’, *Limnology and Oceanography*, 42(5, Part 2):1105-18.

van Woesik, R., Tomascik, T. & Blake, S. 1999, ‘Coral assemblages and physico-chemical characteristics of the Whitsunday Islands: Evidence of recent community changes’, *Mar. Freshwater Res.*, 50:427-40.

Wheeler, W. N. & Weidner, M. 1983, 'Effects of external inorganic nitrogen concentration on metabolism, growth and activities of key carbon and nitrogen assimilatory enzymes of *Laminaria saccharina* (Phaeophyceae) in culture', *Journal of Phycology*, 19:92-6.

Zimmerman, R. C. & Kremer, J. N. 1986, 'In situ growth and chemical composition of the giant kelp, *Macrocystis pyrifera* – responses to temporal changes in ambient nutrient availability', *Marine Ecology Progress Series*, 27:277-85.

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