Report on Status and Trends of Water Quality and Ecosystem Health in the Great Barrier Reef World Heritage Area

September 2005

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Executive Summary

Based on a review of the existing scientific data, the Australian and Queensland Governments have agreed that there is an overwhelming case for halting and reversing the decline of water quality in the waterways entering the Great Barrier Reef. The value of the Great Barrier Reef and the sustainable development of its catchment are of sufficient importance that early action is justified to address what was seen as deleterious ecosystem changes. The primary framework through which this is to be addressed is outlined in the Reef Water Quality Protection Plan (Reef Plan), released by the Queensland and Australian Governments in October 2003. The goal of the Reef Plan is to "halt and reverse the decline in water quality entering the Reef within 10 years". Most of the Reef Plan is focused on land-based actions to improve land use practices within the Great Barrier Reef Catchment and thereby reduce the amount of nutrients and sediment entering river systems flowing into the Great Barrier Reef.

A key component of the Reef Plan is the implementation of the Great Barrier Reef Marine Park Authority's (GBRMPA) long-term water quality and ecosystem health monitoring program for the Great Barrier Reef lagoon, the Reef Plan Marine Monitoring Program (Reef Plan MMP). The primary objective of the program is to assess the long-term effectiveness of the Reef Plan in reducing nutrient and sediment loads entering the Great Barrier Reef lagoon and the responses of key biological communities in the Great Barrier Reef lagoon. The Reef Plan MMP has five basic components: river mouth water quality monitoring, inshore marine water quality monitoring, marine biological monitoring, pollutant bioaccumulation monitoring and monitoring of social and environmental indicators. This report does not incorporate information on the latter component.

The Reef Plan MMP will build upon outcomes from a number of existing research and monitoring projects within the Great Barrier Reef, which are summarised in this report. The reported information is from existing large-scale and long-term projects that describe the status and, in most cases, temporal trends of five key parts of the Great Barrier Reef ecosystem and its catchment:

1. Water quality in river water entering the Great Barrier Reef;
2. Water quality in the Great Barrier Reef lagoon;
3. The status of coral communities on inshore reefs;
4. Status and trends of intertidal seagrass communities; and
5. Bioaccumulation of pollutants by an iconic estuarine species.

A number of the summarised projects have been operational for time frames long enough (> 10 years) that allow baselines of natural levels of ecosystem variability to be resolved and, in some cases, changes in ecosystem status can be detected. This report summarises the current status and, where available, trends of water quality in river runoff and coastal waters and of important inshore coral reef and seagrass ecosystems affected by this runoff. The intent is to lay a foundation for measuring progress towards achieving the goals of the Reef Plan. The information summarised provides a baseline from which future changes in water quality can be detected and tracked. The datasets also provide necessary information for future improvements of sampling and monitoring programs to achieve the goals of the Reef Plan.

Water quality in rivers entering the Great Barrier Reef

Runoff from the catchments adjacent to the Great Barrier Reef is the major source of nutrients, sediments and anthropogenic pollutants entering the Great Barrier Reef lagoon. Exports of these materials from the 424,000 km^2 Great Barrier Reef Catchment are driven by episodic seasonal rainfall events which produce short-lived wet-season floods. While the

Great Barrier Reef has always received freshwater runoff, modern (post-1850) land use has substantively changed the quantities of sediment, nutrients and other pollutant materials in the runoff.

Most sediment and nutrients carried by rivers are derived from diffuse sources in agricultural lands. The pattern of land use varies widely across Great Barrier Reef Catchments. There are also noticeable differences between the water quality in, and exports from, wet- catchment rivers (Cape York to the Herbert River and the Pioneer River; catchments with an average annual rainfall generally > 1,500 mm) and dry-catchment rivers (catchments south of the Herbert River with an average annual rainfall generally < 1,500 mm). In most cases, higher sediment and nutrient concentrations are found in rivers draining dry catchments.

At the present time it is estimated that, on average, terrestrial runoff annually carries 11-14 million tonnes of fine sediment (largely silt and clays: particles < 63 μm carried as suspended load); 40,000-64,000 tonnes of nitrogen and 7,000-14,000 tonnes of phosphorus into the Great Barrier Reef. The ranges reflect different approaches applied to estimate average annual exports. On a year-to-year basis, runoff inputs can vary 3-fold from the average due to inter-annual climate-driven variations in the amount and source of freshwater runoff. A variety of evidence indicates that current average annual inputs of sediments and nutrients are 2- to 4 times greater than average annual inputs prior to European settlement of the catchment and the introduction of modern agricultural development. The primary reasons for the increases in sediment and nutrient exports are enhanced soil erosion and losses of agricultural fertilisers. There is no evidence that these factors are decreasing.

Much of the nitrogen (40-80%) and most of the phosphorus (70-80%) carried by rivers are transported in particulate form, attached to fine eroded soil particles (largely clays). Loads of sediment, nitrogen and phosphorus carried by the large rivers of the Dry Tropics (Burdekin and Fitzroy) during flood events are 2- to 4 times those carried in Wet Tropics rivers. However, the Wet Tropics rivers have higher losses on a per-area basis due to their steeper topographies and higher rain-driven erosion rates.

River sampling programs in a number of catchments (for example Mulgrave, Tully and Herbert) show that nutrient concentrations, particularly of soluble forms (e.g. nitrate), increase as river waters cross floodplains with extensive agricultural development.

A number of long term datasets are available for individual catchments to demonstrate changes in sediment and nutrient exports. For example, significant increases (4-6% p.a.) in particulate nitrogen and phosphorus concentrations were observed in the lower Tully River (Wet Tropics) over a 10-year period (1990-2000). These increases coincide with an intensification of agricultural land use in the catchment. Similar changes over time were not detected in the Burdekin River (Dry Tropics). This is likely to be due to several factors including high natural variability in water flow and sediment and nutrient loads, less sampling effort, and patterns of land use have not materially changed over the last two decades.

Herbicide and pesticide use is now an integral part of modern agricultural and land use practices. Unlike nutrients, which occur and vary naturally, their presence can be regarded as direct evidence for pollution. Initial monitoring of some rivers and estuaries of the Great Barrier Reef Catchment reveal low levels of herbicides (several nanograms per litre) in two Wet Tropics Rivers (Russell-Mulgrave, Johnstone) with active cropping on the adjacent floodplain. Higher levels of herbicides at biologically significant levels (several micrograms per litre) were measured in floodwaters of the Johnstone and Pioneer Rivers, indicating that under certain conditions, significant amounts of these agricultural chemicals can be flushed into regional river systems and subsequently the Great Barrier Reef.

The highly variable flow regimes of the monsoonal rivers of the Great Barrier Reef Catchment pose significant challenges for the monitoring of exports to the Great Barrier Reef and tracking progress in implementation of the Reef Plan. Discharges, sediment loads and nutrient concentrations vary significantly on an episodic, seasonal and inter-annual basis. Statistical analyses pf existing datasets indicate that monitoring time frames of 10 years, or longer, are necessary to reliably establish net changes in water quality parameters. High quality datasets of 2 to 13 years duration are now available for a number of rivers draining into the Great Barrier Reef World Heritage Area (GBRWHA). These datasets will provide the basis for baselines against which the Reef Plan can be evaluated.

Volume-normalised export loads are the most important performance measure of catchment status in terms of water quality in the Great Barrier Reef lagoon. However, high quality estimates of exports are only available for a small number of rivers and years. Effective sampling regimes to measure loads require high intensity sampling of short-lived flood or high-flow events, as is being implemented in the current Reef Plan MMP. Due to the high natural climate variability of the region, export-monitoring programs should be sustained over periods of 10 years or longer to produce comparable results.

Water quality in the Great Barrier Reef lagoon

Widespread and extensive water quality sampling has been carried out by the Australian Institute of Marine Science (AIMS) throughout the Great Barrier Reef lagoon since the mid-1970's as part of monitoring and biological oceanographic studies within the Great Barrier Reef region. Valuable datasets, produced using consistent methods, are available since 1980. As part of this large effort, repeated sampling along a transect of coastal stations between Cape Tribulation and Cairns since 1989. Broad-scale surface chlorophyll *a* monitoring program has also been carried out through most of the Great Barrier Reef since late-1992.

Both broad-scale and time series data sets of water quality parameters in the Great Barrier Reef lagoon indicate that nutrient, suspended particulate matter and chlorophyll *a* concentrations in Great Barrier Reef waters are generally low. This is in large part due to the low-nutrient levels of source waters for the Great Barrier Reef (the Coral Sea) and rapid biological uptake of nutrients by plankton communities in the Great Barrier Reef lagoon. However, the data show that there are persistent spatial and seasonal variations in average nutrient concentrations and other water quality parameters that reflect broad-scale and seasonal degrees of nutrient input to the Great Barrier Reef lagoon and regional-scale dispersion (mixing) rates. Efforts to measure the degree and direction of change in water quality within the Great Barrier Reef lagoon need to account for this inherent large-scale variability.

While ambient nutrient, chlorophyll *a* and suspended sediment levels are generally low, high concentrations of nutrients occur episodically in plumes of flooding rivers and over regional domains disturbed by the passage of tropical cyclones. Flood plumes are normally of short duration and may occur between one to several times per year on a regular basis in the Wet Tropics rivers, but less frequently (every several years to several decades) in the Dry Tropics rivers bordering the southern regions of the Great Barrier Reef. These high nutrient loads associated with flood plumes or cyclonic disturbances lead to the formation of short-lived plankton blooms which convert nutrients in runoff to organic matter. The contrasting behaviour of particulate nutrients and dissolved nutrients in flood plumes shows that nutrients discharged from rivers in dissolved form are transported great distances, and therefore have the ability to influence biological activity on much of the inner-shelf of the Great Barrier Reef. Nutrients discharged in a particulate form are trapped near the coast and probably do not have a major influence on, for example, most of the inner-shelf coral reefs. These results have important implications with respect to the degree of exposure of inner-shelf ecosystems to river sourced nutrients and suspended particulate matter. As different forms of nutrients are exported from different land uses on the catchment the results can also help decide on priorities for management to reduce export from specific land uses. In general it is very clear that the primary area where flood plumes are common is the inner shelf and that ecosystems in this area are at most risk from pollutants contained in river discharge. The spatial distribution of recurrent river plumes has been used, together with other criteria, for estimating risk to regional ecosystems from exposure to terrestrial runoff.

A gradual increase in suspended sediment, dissolved organic nitrogen and dissolved organic phosphorus concentrations has been observed over 15 years in coastal waters near Cairns. Dissolved organic nitrogen is the most abundant form of fixed nitrogen in Great Barrier Reef waters. Over the same time period, concentrations of dissolved inorganic and particulate nitrogen and phosphorus, and chlorophyll *a* (a measure of plankton biomass and proxy for nutrient availability) varied over a range of time scales, but showed no net change overall.

Regional-scale chlorophyll *a* monitoring in coastal and lagoon waters from 1992 shows that average concentrations within cross-shelf transects increase from north to south. Persistent cross-shelf gradients in chlorophyll *a* concentration are found in the central and southern regions of the Great Barrier Reef, reflecting enhanced nutrient availability at the coast from terrestrial runoff and recurrent resuspension of inshore sediments. To date however, no significant net changes in chlorophyll *a* concentration have been observed at the regional scale.

Large organic particles, one of the products of nutrient inputs, also exhibit higher concentrations in inshore waters. There is experimental evidence that mixtures of organic matter and fine terrestrial sediments are detrimental to hard corals and small sessile reef organisms. Little is known about the distribution, abundance and composition of this material in Great Barrier Reef waters. It is likely that particulate organic matter in Great Barrier Reef waters is a major driver of water quality effects on reef and reef organisms.

Statistical analysis of existing water quality data sets indicate that appropriate monitoring over time-frames of at least 10 years are necessary to resolve significant net changes in the naturally variable coastal waters of the Great Barrier Reef. There are now large-scale data sets available (AIMS Regional Water Quality Sampling; GBRMPA Long-term Chlorophyll Monitoring), to define regional baseline levels of important water quality variables such as nutrients and chlorophyll *a*. Two time-series of sampling are currently operational (AIMS Cairns Coastal Transect [1989-present], GBRMPA Long-term Chlorophyll Monitoring [1992-present]) with the capacity to detect long-term changes in lagoonal water quality if they are continued. These programs need to be expanded to better cover coastal waters bordering the Wet Tropics and southern regions of the Great Barrier Reef, which is part of the current Reef Plan MMP.

At present, long-term data sets are not available for agricultural chemicals or other man-made pollutants in Great Barrier Reef lagoon waters. Low levels (to several nanograms per litre) of herbicides have recently been detected in the water column at six inshore reef sites bordering the Wet Tropics. While water column herbicide concentrations measured to date are below experimentally determined effect levels, their presence does indicate that man-made pollutants (and other materials) in runoff are reaching inshore ecosystems of the Great Barrier Reef. The herbicide diuron has been measured in coastal and intertidal sediments at a number of locations along the Great Barrier Reef coast. The highest levels in these samples (to 10 micrograms per kg of sediment) were measured at sites adjacent to rivers draining catchment with extensive sugarcane cultivation. There is no information about chronic effects of herbicide exposure on Reef ecosystems or effects of higher levels that are expected to occur during flood events.

Aspects of coral communities in the inshore Great Barrier Reef

Of the 2900 reefs within the GBRWHA, approximately 750 are located in the inshore zone which is most affected by terrestrial runoff and resulting alterations of water quality. Of these, ca. 450 reefs are considered to be at a high risk of being exposed to terrestrial runoff. Very few of these reefs have had detailed or long-term biological surveys, and fewer still time series of observations to identify and track community changes.

Long-term regional-scale surveys of hard corals, soft corals, algae, fish and bio-eroders clearly show that reef community composition and the presence of individual species or groups vary along gradients of influence by terrestrial runoff or water quality. Inshore reef communities that are exposed to terrestrial influence are dominated by macro-algae, whereas reefs that are further from shore and exposed to flood plumes less frequently are dominated by hard corals. These gradients are observed within regional areas (Wet Tropics, Whitsunday Islands) and between regions with differing levels of influence from agricultural runoff (Princess Charlotte Bay, Wet Tropics). Bio-eroding organisms show clear cross-shelf differences with higher abundances of macro-borers at inshore sites, causing erosion of the reef framework. The small number of historical surveys, however, precludes determination of whether the spatial extent of terrestrial runoff influence is stable or changing.

Long term time series (late 1980's to the present) of changes in coral communities are only available for 12 inshore reefs. Coral cover has declined in the region between Cairns and Townsville. The declines coincide with outbreaks of Crown-of-thorns starfish and major coral bleaching events in the region. However, this region, being in the Wet Tropics, is also assessed to be at moderate to high risk of exposure to runoff. Over the same period, coral cover was stable in the Cooktown-Lizard and Whitsunday Islands sectors; regions not affected to the same degree by major disturbances and generally assessed to be at moderate risk of exposure to runoff. Studies of reefs between Cape Tribulation and the Frankland Islands, which are assessed at high risk from exposure to runoff, had very high coral cover in the 1980s and early 1990s, which was reduced during the 1990s by a number of disturbances including floods, cyclones, bleaching and Crown-of-thorns starfish (COTS). The majority of sites now show substantial recovery. The potential effects of exposure to runoff are likely to be confounded with effects of coral bleaching and other catastrophic events.

A detailed survey of coral communities on 27 inshore reefs in 2003-2004 established regional baselines of spatial variability in inshore reef communities and a basis for measuring future changes in coastal coral reef communities and their responses to water quality. Approximately half of the surveyed reefs are in areas considered to be at moderate to high risk to runoff. These reefs had highly variable levels of coral cover and species mixtures within a few kilometres of each other. A number of high-cover reefs supported both fast and slow growing species, suggesting that these coral communities have persisted for decades. Within a particular region, inshore reefs may be more variable in terms of coral cover and community types than mid-shelf and outer shelf reefs. This suggests that environmental gradients, e.g. turbidity and salinity, may be steeper in inshore regions or that disturbances or conditions for recovery are more localised. At most sites in the northern and central regions of the Great Barrier Reef, the surveys showed reasonable levels of recruitment for the majority of coral genera. This could indicate that inshore reefs in these locations are now in the process of recovering from major disturbances such as bleaching, floods, and in some cases, Crown-of-thorns starfish outbreaks. However, neither regional patterns of coral cover nor the number of recruit densities indicated a clear relation to the estimated risk of exposure to runoff; a result requiring further exploration.

Currently available time series measurements of coral recruitment are limited to a few reefs and indicate large spatial and year-to-year variations in recruitment. There is also only limited information about the proportion of recruits surviving to adulthood and the factors influencing this process.

In general, a lack of information about the characteristics of inshore reefs of the Great Barrier Reef prior to European colonisation, the great variation among closely located inshore reefs today and a lack of information about realistic rates of recovery from disturbances make it difficult to assess the health of inshore reef systems with any certainty. The lack of concurrent surveys of environmental factors also makes it difficult to interpret available data on coral reef status with regard to changes caused by differences in water quality. Assessment of ecosystem health of inshore reefs under the current Reef Plan MMP will measure coral cover, community structure, size-frequencies of coral colonies and coral recruitment rates, to provide a measure of the balance between disturbance and recovery processes. Integrating these coral data with knowledge of environmental factors such as temperature and water quality parameters and the disturbance history at a local scale will be critical to the interpretation of coral community changes over the next decade that will constitute an important indicator of the effectiveness of the Reef Plan.

Status and trends of intertidal seagrass communities

Seagrasses are a highly productive habitat and valuable nursery ground for many marine species in the Great Barrier Reef. There are nearly 6,000 km^2 of seagrasses in waters shallower than 15 metres, relatively close to the coast, and in locations that can potentially be influenced by adjacent land use practices. Data from a supervised community-based seagrass monitoring program (Seagrass-Watch: 1994-present) has been collected at a number of sites along the Queensland coast. This monitoring indicates that most intertidal seagrass meadows have been relatively stable over the last decade. Seagrass meadows in the Hervey Bay region suffered extensive losses of area and cover in the early 1990's due to storm damage and impacts of flooding. These meadows are now in a recovery state. Meadows in some other regions (e.g. Whitsunday), however, show net declines over the monitoring period.

The greatest potential for loss of seagrasses is associated with downstream effects of land use and from global influences such as climate change and the predicted increase in storm frequency. However, these influences are interlinked in complex ways and further work is required to understand and quantify these links. Current knowledge about causal relationships of water quality parameters and Reef seagrass health is scarce. Longer time series of seagrass health as well as measurements of environmental parameters are required to detect changes outside natural variability and to reveal controlling factors of seagrass cover and distribution.

Bioaccumulation of pollutants in estuarine species

The mud crab *Scylla serrata* is an iconic species with high fisheries value in the GBRWHA. It has been used as a bio-indicator species for trace metal and pesticide accumulation in biota from a limited number of Queensland rivers and estuaries. Higher trace metal levels were measured in *S. serrata* hepatopancreas tissues collected at an industrialised location (Port Curtis - Gladstone) compared to an agriculture-dominated site (Burdekin River). Pesticide analysis of *S. serrata* from Port Curtis and several other sites revealed historical contamination by DDTs and dieldrin.

The current data, whilst limited to the southern regions of the Reef, indicate that *S. serrata* should be a useful species for bioaccumulation monitoring of rivers adjacent to the Reef. The work proposed under the Reef Plan MMP will provide an important benchmark for future comparisons of pollutant levels along the spatial extent of the Reef, and to complement other measures of pollutant levels, such as passive samplers for pesticides and herbicides.

Conclusions

There is now abundant evidence, primarily from locations outside of Australia, that the overall health of coral reef and seagrass ecosystems are affected by the quality of water in which they live. Poor water quality leads to the loss or displacement of dominant or desirable species, reductions in coral or seagrass cover, loss of ecosystem amenity value, and in extreme cases, the destruction of the physical structure of the ecosystem.

Not surprisingly, the best documented cases and clearest relationships between poor water quality and the health of coral reefs or seagrasses have involved large inputs of sediment or nutrients to relatively small areas (e.g. Kaneohe Bay, Hawaii; Chesapeake Bay; Discovery Bay, Jamaica; Hervey Bay). Fortunately, these extreme situations have not appeared within the GBRWHA. However, the same processes and pressures that have caused large changes elsewhere are evident in the Great Barrier Reef and its Catchment. In almost every case study, major ecosystem degradation has begun with small, almost imperceptible changes, essentially indistinguishable from the 'normal' range of environmental variability.

However, there are some situations relating to declining water quality which are now clear in the Great Barrier Reef. A variety of evidence now clearly indicates that exports of sediment and nutrients from the Great Barrier Reef Catchment have increased substantially; at least 2 to 4 fold over the last 150 years. This agrees with global data of increases of nutrient exports with intensifying landuse. Increased nutrient availability in Reef waters has been demonstrated in a long-term time series off the Cairns coast, an indication that materials carried in land runoff are measurably affecting Great Barrier Reef waters. There is well-documented evidence within the Reef that benthic communities on inshore coral reefs vary along measured or presumed gradients of water quality or terrestrial influence. Observed changes include variations in the cover, composition and relative abundance of macroalgae, hard corals and soft corals, the recruitment of young hard corals and the abundance of coral bio-eroders. However, on a reef-wide scale the temporal and spatial dynamics of important ecosystems such as coral reefs and seagrass meadows are only beginning to be understood.

In order to assess the effects of land runoff on the Great Barrier Reef lagoon water quality and ecosystems with certainty, it is necessary to understand the assimilative capacity of the Great Barrier Reef lagoon and the critical thresholds of the ecosystem to ecological harm from the cumulative inputs of nutrients, sediments and pollutants. While dilution, sedimentation and biological uptake and transformation effectively remove nutrients and sediments from the water column, these materials stay in the system and are likely to slowly accumulate.

The highly variable (episodic, seasonal, inter-annual) monsoonal climate of the Great Barrier Reef region, and the episodic nature of disturbance processes operating within the Great Barrier Reef (cyclones, COTS, bleaching) produce high natural variability of river discharge, riverine sediment and nutrient loads, lagoonal water quality, biological community structure and community distributions particularly between wet and dry catchments. The variable influences of nutrient speciation, particularly nitrogen, require further consideration. Any successful monitoring program established to track progress toward achieving the goals of the Reef Plan needs to take a long-term perspective, be fit for purpose, and based on the existing reef-scale and long-term water quality and biological monitoring programs which have sufficient duration (5 to 15 years) and sampling density to provide useful baseline and trend/variability data.

In conclusion, much of the technical knowledge relating to water quality and ecosystem health in the Great Barrier Reef to date has been based on relatively small scale programs; the current research direction reflects a strong progression to extending this knowledge to a broader whole-of-system approach which is the foundation for the development of the Reef Plan. The Reef Plan MMP will assist to overcome this limitation by including inshore reef surveys and water quality monitoring at the same locations. In addition, passive samplers will be deployed within the corresponding regions to measure pesticide loads; sediment sampling will be undertaken to qualify these measurements. The Seagrass-Watch program will continue across the regions at sites that are complementary to river mouth and marine water quality monitoring programs. New and more extensive statistical analyses will be undertaken for the datasets. This improved capacity to integrate the subprograms in combination with ongoing research of the environmental implications of water quality on Great Barrier Reef ecosystems, will assist to improve our understanding of water quality influences and hence, changes in the Great Barrier Reef which may be attributable to implementation of the Reef Plan.

1. Introduction

The Great Barrier Reef is the largest contiguous coral reef ecosystem in the world. It encompasses over 250,000 km^2 on the continental shelf bordering North East Australia and Papua New Guinea. The full Great Barrier Reef ecosystem contains over 3,000 individual coral reefs. Most of this ecosystem has a high degree of protection within the largely co-located Great Barrier Reef World Heritage Area (GBRWHA) and Great Barrier Reef Marine Park (GBRMP). Approximately 225,000 km^2 of shallow shelf and reef habitat lie within the bounds of these protected and managed areas, which contain close to 2,900 reefs. The Great Barrier Reef ecosystem is one of the world's richest areas of biological diversity. In addition to coral reefs, the GBRWHA and adjoining coastline contains a diversity of hard and soft bottom marine habitats, including extensive seagrass and algal beds, mangrove forests and islands.

In the late 1990s the Great Barrier Reef was considered one of the least-disturbed coral reef systems in the world (SOEAC, 1996; Wachenfeld *et al*., 1998). However, the *Australia State of the Environment 2001* report noted that even though the Great Barrier Reef is generally in a "near-pristine state over large areas", "runoff of freshwater carrying nutrients, sediments and pollutants is affecting the coastal margins of the Great Barrier Reef region" and "water quality in parts of the coastal margin is likely to be in slow decline from cumulative effects of human activities" (ASOEC, 2001; Brodie, 1997).

In many parts of the world, increasing human populations and intensifying land use have altered the nature of freshwater runoff to coastal waters. Increasing loads of sediment, nutrients and other pollutants carried in freshwater runoff can influence coastal water quality, and thereby affect the health of coastal ecosystems, including coral reefs (Wilkinson, 2004). There is now abundant evidence from overseas examples that corals and coral reef ecosystems are influenced by water quality in a variety of ways (reviewed in Fabricius, 2005). Prominent examples of reef degradation include Hawaii (Kaneohe Bay, Pearl Harbor; Smith *et al*., 1981; Coles, 1999), Indonesia (Jakarta Bay; Tomascik *et al*., 1997), Hong Kong (Morton, 1994), Barbados (Tomascik and Sanders, 1985, 1987 and others). In all of the above cases, the adverse effects of poor water quality due to nutrient, sediment or organic loading are caused by conditions far more extreme than encountered in the Great Barrier Reef and within much smaller areas. There is, however, no doubt about the cause. A salient lesson from these systems is that even when the cause of reef degradation is clearly identified and an obvious solution applied, recovery can take many decades and is readily derailed by further smaller disturbances (Hunter and Evans, 1995).

Despite its large size and general remoteness from significant population centres, the Great Barrier Reef is not protected from problems related to changing terrestrial runoff. More than fifteen years of marine and land-based research on the Great Barrier Reef and the adjacent catchment have shown that the coastal area of the GBRWHA is influenced by material originating from a range of human activities on the land and in the water (reviewed in Haynes, 2001; Williams, 2002; Great Barrier Reef Protection Interdepartmental Committee Science Panel, 2003; Furnas, 2003; Haynes and Morris, 2003; see also Brodie *et al*., 2004; Fabricius, 2005; Fabricius *et al*., 2005; Schaffelke *et al*., 2005). In summary:

- The delivery of sediments and nutrients to rivers discharging into Great Barrier Reef waters has increased two- to six-fold over estimated inputs prior to European settlement of Queensland (ca. 1850) and amounts of nutrients in seasonal river flood plumes, which are the primary mechanism transporting land runoff into the Great Barrier Reef, are now at a level that may cause harm to the Great Barrier Reef ecosystems.

- Agricultural herbicide residues are found in coastal sediments and river flood waters, especially adjacent to catchments with high agricultural land use.
- A number of Great Barrier Reef inshore reefs exhibit characteristics consistent with impacts caused by enhanced nutrient availability or sedimentation.
- Overseas studies unequivocally demonstrate harmful effects of excess nutrients and sedimentation to reef systems and indicate that by the time widespread effects are obvious, systems can be almost irreparably damaged.

Human land-use, primarily for agriculture, delivers most of the enhanced loads of sediment, nutrients and chemical pollutants reaching the Great Barrier Reef. The estimated increases of sediment and nutrient runoff are due to widespread vegetation clearing within the Great Barrier Reef Catchment, and extensive rangeland cattle grazing and cropping including grains and cotton inland, and sugarcane and horticulture along the coast. Furnas, (2003) provides a comprehensive summary of current catchment status and land use changes since European settlement. These activities have led to increased soil erosion in the Great Barrier Reef Catchment and with it, enhanced losses of nutrients, both from natural soils and as losses of applied fertilisers. Application of fertiliser nitrogen and phosphorus has increased steadily over the last 50 years (Pulsford, 1993). Current applications to farmlands in the Great Barrier Reef Catchment are estimated to be on the order of 100,000 tonnes of nitrogen and 20,000 tonnes of phosphorus per annum (Furnas, 2003). Current land-uses differ between individual Great Barrier Reef Catchments (Furnas, 2003; Brodie *et al*., 2004). Each catchment contributes differently to sediment, nutrient and pollutant loads (GBRMPA, 2001).

Greiner *et al*., (2003) assessed the risk of Great Barrier Reef Catchments to the downstream Great Barrier Reef waters using a variety of criteria, including estimates of discharge of sediments and nutrients to the Great Barrier Reef, potential impacts of this discharge on adjacent ecosystems, and socio-economic criteria. Two catchments in particular, the Burdekin and Fitzroy, rated 'high' against all four aspects of risk in this assessment. This assessent used as a basis for the Catchment Risk Profiles included in the Reef Plan. There are several limitations to this approach such as the accuracy of inputs including the estimates of sediment load, catchment aquatic system condition, area of seagrass, and assumptions relating to capacity to effect change and risk to marine industries.

An earlier model estimated exposure of Great Barrier Reef inner-shelf reefs to terrestrial runoff using ratings of volume and frequency of discharge from major rivers, the predominant distribution of river plumes in Great Barrier Reef waters, loads of riverine pollutants, and distance of reefs to river mouths (Devlin *et al*., 2003; Figure 1.1). Coastal and island areas at high risk of exposure to terrestrial runoff were identified adjacent to the Wet Tropics region, from Tully to north of Cairns, and in the Whitsunday region. This model has a number of limitations, for example, it only includes run-off transported by major rivers, excluding locally important smaller waterways and coastal transport and recycling processes; it assumes a linear reduction of pollutant concentration with distance to river mouth; and it only assessed coral reefs as exposed ecosystems. This model is currently the only available marine exposure risk analysis and is a useful representation of the spatial extent of the coastal areas that are likely to be regularly exposed to land runoff. However, the model did not consider the consequences of this exposure, for example to coral reefs, which should be part of a complete risk assessment.

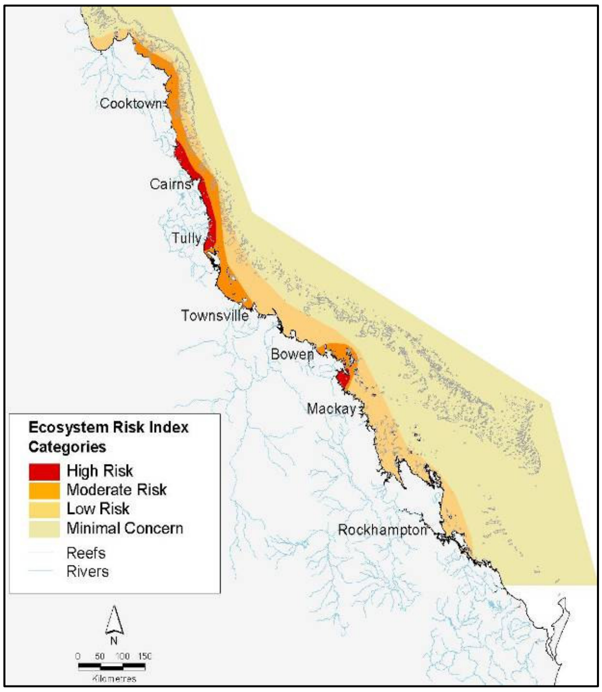

Figure 1.1 Reef risk assessment: estimated areas and degrees of ecological risk from terrestrial runoff. Source: Devlin et al., (2003).

Earlier hydrodynamic modelling studies have likewise identified that reefs off the Wet Tropics are regularly exposed to land runoff (King *et al*., 2002; Wolanski *et al*., 2003). Collectively, these risk assessments or projections were used in the process of developing the Reef Water Quality Protection Plan (Reef Plan) (see below).

Effects of terrestrial runoff on coral reef ecosystems of the Great Barrier Reef have been noted from the earliest days of scientific research on the reef (Rainford, 1925; Orr, 1933; Fairbridge and Teichert, 1948). There is a general appreciation that runoff from the catchments adjoining the reef influences waters and ecosystems within the Great Barrier Reef region. However, significant research to quantify the amount of sediment, nutrients and other materials carried in runoff from the catchment, and to identify and understand its influence on ecosystems within the Great Barrier Reef has only been undertaken within the last 20 years. As a result, significant management and policy action to address runoff-related issues is also relatively recent.

Important recent publications, policy and review steps towards the development of the Reef Plan are:

- The first general scientific conference dealing with water quality issues in the Great Barrier Reef region was held in Townsville (Yellowlees, 1991).
- The Cooperative Research Centre for Sustainable Development of the Great Barrier Reef was established and included water quality research (1992).
- The Great Barrier Reef Marine Park Authority (GBRMPA) identified the need for integrated catchment management in the Great Barrier Reef region to address water quality deterioration in its 25-year Strategic Plan for the GBRWHA (GBRMPA, 1994).
- Catchment-reef and water quality issues were the centre of or figure prominently in two large regional scientific conferences on the Great Barrier Reef (Hunter *et al*., 1997; Crossland, 1997)
- Key water quality issues related to management of the GBRWHA were reviewed and defined (Haynes, 2001).

- GBRMPA (2001) prepared the Great Barrier Reef Catchment Water Quality Action Plan, identifying targets for reducing sediment and nutrient runoff to the Great Barrier Reef.
- A wide-ranging scientific review of the Water Quality Action Plan, and of runoff effects on the Great Barrier Reef was commissioned by the Queensland government (Great Barrier Reef Protection Interdepartmental Committee Science Panel, 2003).
- The Reef and Rainforest Cooperative Research Centre's receive supplemental funding to establish a joint Catchment-to-Reef research program (2003).
- The Great Barrier Reef Catchments are a thematic and regional focus under the CSIRO's "Water for a Healthy Country" National Research Flagships Flagship Program to provide research support for the Reef Plan.
- Improved estimates of current and pre-settlement terrestrial sediment and nutrient inputs to the Great Barrier Reef are published (NLWRA, 2002; Furnas, 2003; Brodie *et al*., 2004).
- A wide range of recent research on catchment to reef and water quality issues are presented at a scientific meeting and published (Marine Pollution Bulletin Volume 51 - Hutchings *et al*., 2005b).

To specifically address the issues associated with degrading water quality from diffuse land-based sources, the Queensland and Australian Governments developed the Reef Plan in October 2003. The objectives of the Reef Plan are to reduce the load of pollutants from diffuse sources in the water entering the Great Barrier Reef; and rehabilitate and conserve areas of the Great Barrier Reef Catchment that have a role in removing water borne pollutants, e.g. wetlands and estuaries. These objectives are addressed through nine strategies, including self-management approaches; education and extension; economic incentives; better planning for natural resource and land use management; regulatory frameworks; research and information sharing; partnerships; priorities and targets; and monitoring and evaluation. The Reef Plan does not, however, include specific catchment water quality or export targets, e.g. as set in GBRMPA, (2001).

A key component of the Reef Plan is the implementation of a long-term water quality and ecosystem monitoring program in the Great Barrier Reef lagoon (Strategy I.4). The Reef Plan Marine Monitoring Program (Reef Plan MMP) was established by the GBRMPA to track trends in the amount of sediment, nutrients (nitrogen, phosphorus) and other pollutants in key rivers discharging into the Great Barrier Reef, the status of water quality within Great Barrier Reef lagoon waters, and the condition of key biological communities (coastal coral reefs, seagrasses) that are most likely to be influenced by water quality. The Queensland Government is developing a complementary water quality monitoring program for the Great Barrier Reef Catchment.

The Reef Plan MMP will build upon a number of previous and current long-term research and monitoring projects within the GBRWHA that directly support the objectives of the Reef Plan. These projects are summarised in this report and will provide essential baseline information on the status of water quality in reef and regional river waters, and upon the status of regional ecosystems. A number of these projects have been operational for time frames long enough (> 10 years) that baselines natural levels of ecosystem variability can now be resolved and changes in ecosystem status can be detected (De'ath, 2005). The data sets produced by these projects will also provide information useful for designing improved sampling and monitoring programs to achieve the stated goals of the Reef Plan.

The objective of this report is to summarise results from a number of individual environmental sampling, research and monitoring programs which are most relevant to the objectives of the marine monitoring component of the Reef Plan. This report describes the current status of, and recent (5-15 years) changes in: water quality in and sediment/nutrient delivery from a number of rivers flowing into the GBRWHA, the nutrient status of Great Barrier Reef coastal and lagoon waters, and the status of representative inshore coral reefs and intertidal seagrass beds. This information provides a baseline from which future changes in water quality and the status of key communities can be detected and tracked. It is important to note that these sampling programs were initiated and have been continued for a variety of scientific objectives, mostly un-related to the monitoring objectives of the Reef Plan. Consequently, the reported programs have varied sampling designs and analytical approaches which may also differ from those of the purpose-designed Reef Plan MMP. An integrative data analysis across programs is not considered to be within the scope of this report. However, it is anticipated that the design of the current and future Reef Plan MMP will enable integration of the datasets quantify levels of certainty in the understanding of causal links to ecosystem health. Although the focus of the report is to present data to be used as a baseline for the Reef Plan MMP, an overview of recent findings related to water quality and ecosystem health is provided in recognition of the importance of linking these findings to ecosystem health.

In the following, definitions of, and additional information about, important terms used in this report are provided.

Water quality *vs.* pollution - Throughout this report, the term 'water quality' is used to refer to the overall condition of water in rivers or the Great Barrier Reef lagoon. In the strict sense, 'water quality' is an arbitrary construct that defines the suitability of water from a source for a particular purpose. For example, a definition of water quality for an industrial use would likely be related to the concentrations of pollutants that affect the product or manufacturing process. In the case of human drinking water, water quality is largely related to concentrations of toxic materials (e.g. industrial chemicals or wastes) and the presence of pathogens (e.g. coliform bacteria).

In the case of aquatic ecosystems such as rivers or the Great Barrier Reef lagoon, water quality refers to the envelope of water characteristics (temperature, salinity, dissolved oxygen concentration, nutrient concentrations, turbidity, particle and toxin loads) which sustainably support a defined biological community, e.g. hard corals which build coral reefs. Characteristics or concentrations of materials which influence the ecosystem (e.g. turbidity, nutrients, suspended sediment) can naturally vary over considerable ranges, though mostly stay within much narrower bounds, the 'normal' water quality envelope. A persistent change in the normal water quality envelope would be expected to eventually change the biological communities living in (e.g. coral reefs) or influenced by the water.

Pollution, in the strict sense, is the human or human-abetted introduction of conditions or materials to an ecosystem that is ultimately detrimental to the resilience of the 'normal' and desired state of the ecosystem. In the case of manufactured materials such as pesticides or refined hydrocarbons, the identification of pollutants is straightforward. In other cases, 'polluting' materials such as nutrients may naturally occur in the environment. The question is then one of degree. Because nutrients occur naturally and can naturally vary over wide ranges, it is often difficult to identify when enhanced inputs of nutrients and organic matter begin to cause undesired changes and become pollution ('eutrophication'), especially when inputs are small but chronic.

Ecosystem health - A central goal of many, if not most, environmental management programs, and of associated monitoring programs to assess their effectiveness is to preserve, measure and track the state of 'ecosystem health'. As with water quality, a definition of ‘ecosystem health’ is arbitrary, and dependent upon the criteria used. There is no clear, unambiguous and universally agreed definition of what a healthy ecosystem is. However, almost all parties implicitly presume that a ‘healthy state’ is close to what an ecosystem would be in most of the time before it was influenced by human activities.

In the case of coral reefs, where coral growth and structural accretion are continually opposed by physical disturbance (e.g. cyclones) and bio-erosion, a central measure of health would be the ongoing replacement, growth and structural dominance of reef-building corals. Reefs without sufficient levels of coral cover and colony growth over time periods of decades to centuries cease to exist as coral reefs.

In the case of seagrass communities a clear and central measure of health based on the current understanding, is the ability of a regional or local community to sustain its areal extent and to maintain a stable population structure or stable progressions of community structure over extended time periods.

A variety of approaches have been taken to ‘measure’ ecosystem health. Some measures (e.g. coral recruitment) are forward looking, some (e.g. coral colony colour) determine a present status, and others (e.g. coral or seagrass cover) are integrative and backward looking. Individual measures are insufficient to describe ‘health’. Rather, an ensemble approach must be taken where a mix of forward, current and integrative measures are considered in unison.

Secular change – Reference to secular change is the continuous or near-continuous upward or downward movement (linear or non-linear) in the average value of a parameter or variable over a prolonged period of time. Over shorter periods, this overall trend may be obscured by variability on a range of time scales. This is distinct from temporal changes that are for example seasonal, cyclical or sporadic.

2. Water quality in rivers entering the Great Barrier Reef lagoon

Miles Furnas, Australian Institute of Marine Science

Introduction

For hydrological purposes, the many river and stream catchments discharging directly or indirectly into the GBRWHA have been divided into thirty-eight (38) drainage basins (Figure 2.1). Of these, thirty-five (35) are located on the mainland. Thirty-one (31) have at least one stream or river that is gauged for measurements of stream flow. Collectively, the mainland drainage basins have an area close to 424,000 km^2 (Table 2.1). Individual drainage basins range in size from 466 km^2 (Mossman River) to 142,000 km^2 (Fitzroy River). The three island drainage basins are small (total area 1,040 km^2) and have no significant permanent streams. The two largest drainage basins (Burdekin River - 130,000 km^2 and Fitzroy River - 142,000 km^2) account for 64 percent of the total Great Barrier Reef Catchment area and 23 % of total freshwater runoff (Table 2.1).

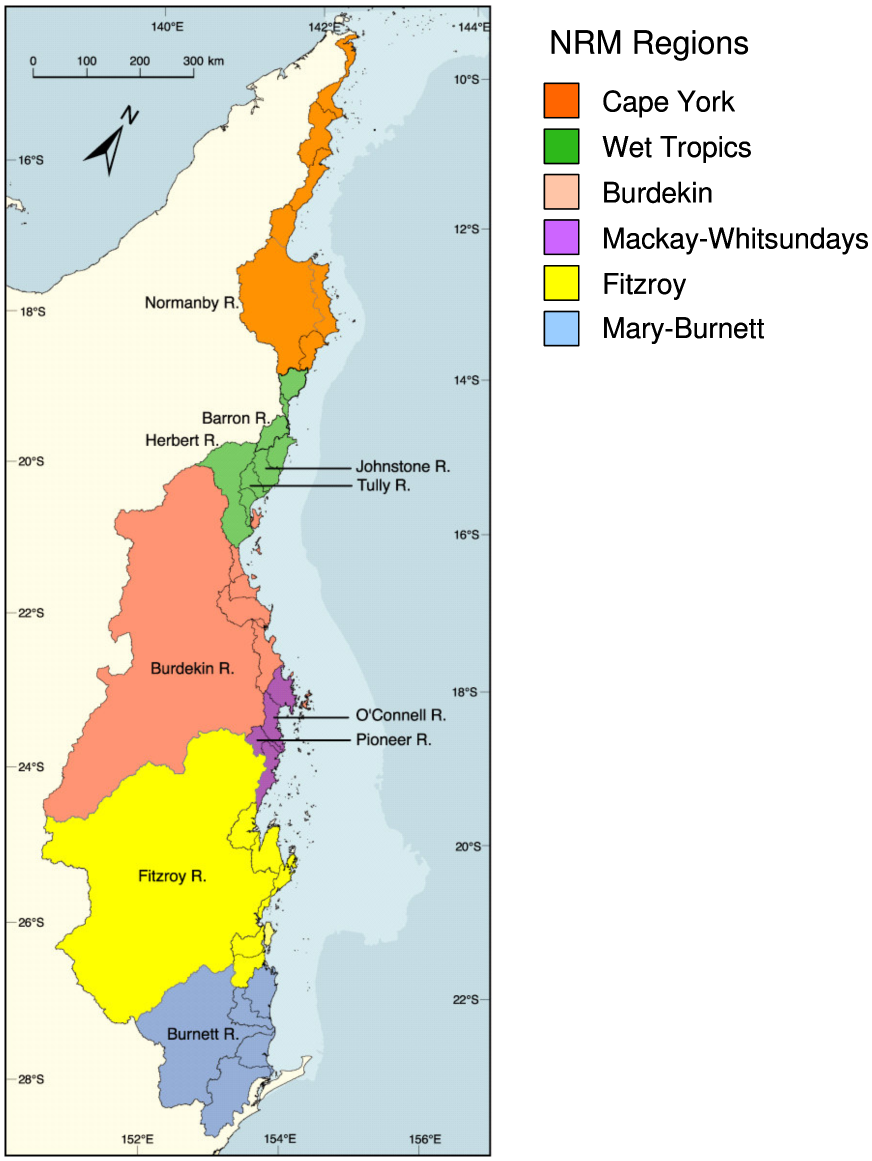

Figure 2.1 Drainage basins discharging into the GBRWHA and associated NRM regions. River names indicate catchments selected for current Reef Plan MMP river mouth monitoring. Monitoring information summarised in this report is from rivers in the Cape York, Wet

Tropics, Burdekin and Fitzroy NRM regions (see Table 2.1 for further details on individual rivers and drainage basins).

*Table 2.1 Estimated average annual exports of nitrogen, phosphorus and sediment from wet and dry drainage basins into the GBRWHA. Reef Plan MMP priority rivers are shown in bold. From Furnas, (2003). Asterisks indicate basins that have river monitoring information summarised in this report. **: monitoring undertaken in Barratta Creek in the Haughton basin.*

	Regional NRM Region Basin Name	Area (km^2)	Runoff-volume (km^3)	Wet/ Dry (km^3)	Fine Sediment Export (Kt)	Total Nitrogen Exports (t)	Total Phosphorus Exports (t)
	Cape York						
	Jacky-Jacky Creek	2,963	1.56	W	60	620	70
	Olive-Pascoe Rivers	4,179	3.71	W	150	1470	160
	Lockhart River	2,883	1.94	W	80	770	80
	Stewart River	2,743	1.21	W	50	480	50
*	**Normanby River**	**24,408**	**4.95**	**W**	**500**	**1960**	**210**
	Jeannie River	3,637	1.54	W	60	610	70
	Endeavour River	2,104	1.82	W	70	720	80
	Wet Tropics						
	Daintree River	2,192	1.26	W	50	500	50
	Mossman River	466	0.59	W	20	230	30
*	**Barron River**	**2,902**	**0.81**	**W**	**30**	**320**	**30**
*	Mulgrave-Russell Rivers	1,983	3.64	W	140	1440	150
*	**Johnstone River**	**2,325**	**4.67**	**W**	**180**	**1850**	**200**
*	**Tully River**	**1,683**	**3.29**	**W**	**130**	**1300**	**140**
*	Murray River	1,107	1.06	W	40	420	40
*	**Herbert River**	**9,843**	**4.01**	**W**	**540**	**1,590**	**170**
	Burdekin						
	Black River	1,057	0.38	D	140	320	60
	Ross River	1,707	0.49	D	180	410	80
**	Haughton River	4,044	0.74	D	270	620	120
*	**Burdekin River**	**130,126**	**10.29**	**D**	**3,770**	**8,630**	**1,700**
	Don River	3,695	0.75	D	280	630	120
	Mackay Whitsunday						
	Proserpine River	2,535	1.08	D	400	910	180
	O'Connell River	**2,387**	**1.54**	**D**	**560**	**1,290**	**250**
	Pioneer River	**1,570**	**1.19**	**W**	**50**	**470**	**50**
	Plane Creek	2,539	1.49	D	550	1250	250
	Styx River	3,012	1.58	D	580	1330	260
	Shoalwater River	3,605	1.83	D	670	1530	300
	Fitzroy						
	Waterpark Creek	1,835	1.11	D	400	930	180
*	**Fitzroy River**	**142,537**	**6.08**	**D**	**2,230**	**5,100**	**1,000**
	Calliope River	2,236	0.30	D	110	250	50
	Boyne River	2,590	0.29	D	110	240	50
	Baffle Creek	3,996	0.78	D	290	650	130
	Burnett - Mary						
	Kolan River	2,901	0.41	D	150	340	70
	Burnett River	**33,248**	**1.15**	**D**	**420**	**970**	**190**
	Burrum River	3,358	0.55	D	200	460	90
	Mary River	9,440	2.72	D	1,000	2,280	450
	Total	423,835	70.82		14,460	42,890	7,110

There are noticeable differences between the water quality in, and exports from, wet-catchment rivers (Cape York to the Herbert River and the Pioneer River; catchments with an average annual rainfall generally > 1,500 mm) and dry-catchment rivers (catchments south of the Herbert River with an average annual rainfall generally < 1,500 mm) (refer also Table 2.1). In most cases, higher sediment and nutrient concentrations are found in rivers draining dry catchments.

Land-based management activities under the Reef Plan, including river mouth monitoring, are carried out in six Natural Resource Management (NRM) regions associated with the Great Barrier Reef Catchment (Figure 2.1). With the exception of the Cape York NRM region, the boundaries of these regions coincide with those of the Great Barrier Reef Catchment or of individual drainage basins. Table 2.1 summarises the areas of, and estimated average annual freshwater runoff from, the individual mainland drainage basins in the Great Barrier Reef Catchment and their distribution amongst the NRM regions.

From the standpoint of the status of water quality in the Great Barrier Reef, the quality of water in rivers *per se* is of less importance than the amount of materials delivered by those rivers to the Great Barrier Reef lagoon (export = $\sum$ concentration x discharge). During dry-season, low- or no-flow periods, only small amounts of sediment, nutrients or other pollutants are delivered to the Great Barrier Reef lagoon, regardless of ambient concentrations in river waters. Conversely, the large water volumes in summer wet-season floods carry most of the annual river inputs, regardless of ambient concentration in river waters. In order to obtain reliable estimates of sediment and nutrient exports carried by the seasonally variable flows from catchments of the Great Barrier Reef region, it is essential to use sampling methods and schedules that focus on these high-flow events. To make useful and accurate estimates of material exports to the Great Barrier Reef lagoon, appropriately dense time series of nutrient and sediment concentrations, in conjunction with continuous or daily records of freshwater discharge are needed. In some rivers, the first flow event of a wet season (first flush) may carry disproportionately high concentrations of nutrients or other pollutants. The impact of these first flush events on coastal waters or the wider lagoon ecosystem, however, is still related to the volume of water in the event (small event = small effect, large event = large effect).

The objective of this section is to present a summary of the status of nutrient and suspended sediment concentrations in rivers flowing into the GBRWHA, and of estimates of freshwater, fine sediment and nutrient exports from Great Barrier Reef Catchments to the Great Barrier Reef lagoon relevant to the Reef Plan MMP. These summaries are largely based upon the extensive set of nutrient and suspended sediment concentration data collected by the Australian Institute of Marine Science (AIMS) between 1987 and 2000, focusing on Wet Tropics and some Dry Tropics rivers (Appendix 1). Data summaries and temporal trend analyses are presented for two rivers intensively sampled over the past decade, the Tully and Burdekin Rivers. Results of statistical analyses of the datasets undertaken by De'ath (2005) are included in this section with additional detail in Appendix 3. While these rivers cannot be considered with certainty to be representative of wet and dry river catchments, the Tully and Burdekin Rivers have been sampled for the longest period (1986/87-2000) and have the highest likelihood of showing long-term trends characteristic of these catchments. Important environmental factors that drive or influence runoff and material exports are also noted or presented. Extensive review of all river water quality data that may have been collected by other agencies in the Great Barrier Reef Catchment is not within the scope of this report.

The data are presented as concentrations (nutrient species or suspended sediment), or summaries of concentrations. Concentration values are ultimately needed to calculate estimated annual export loads delivered to the Great Barrier Reef lagoon. Concentration data summaries are given as both means (the easiest statistic to calculate), and median values because concentration values of most nutrient species exhibit log-normal distributions. Median values down-weight the less numerous high values and give a better indication of the central tendency for the distribution.

2.1 Suspended sediment and nutrient concentrations in rivers discharging to the Great Barrier Reef

Freshwater runoff is currently the largest identified external source of nutrients (e.g. nitrogen, phosphorus, silicate, iron) entering the GBRWHA and affecting water quality within the Great Barrier Reef ecosystem (Furnas, 2003). These nutrients enter in a variety of dissolved and particulate forms, and are subsequently taken up, transformed and stored by various physical, biological, chemical and geological processes. A substantial portion (50-80% depending on the specific nutrient and the catchment) of the nutrients in runoff is in particulate form, attached to fine sediment particles (largely silt and clays: particles < 63 µm carried as suspended load) eroded from catchment soils.

A number of water sampling programs have taken place in rivers and streams of the Great Barrier Reef Catchment. Most of these programs have been localised and of short duration, focusing on the ambient state of water quality in rivers and streams of the Great Barrier Reef Catchment, rather than the amount of sediment, nutrients and other materials discharged into the Great Barrier Reef lagoon. As a result, they are difficult to integrate into a larger picture of material exports to the GBRWHA. A few, however, have covered larger domains, or have run for a number of years (e.g. Anon, 1999; Furnas, 2003; Loxton and Gittins, 2004; Cox *et al*., 2005). For example, The Queensland Environmental Protection Agency has conducted an ambient water quality monitoring program in major Queensland waterways since 1992. Monitoring sites were located on major rivers in catchments flowing to the east coast, and were largely concentrated in estuarine and coastal waters in southern areas and freshwater sites in the Wet Tropics. The main aim of the program was to assess water quality condition and trend in major rivers and estuaries. Condition and trends in water quality were not consistent across the whole GBR catchment. However, there are some general conclusions that can be drawn. Sites in poor condition were generally impacted by point source discharges, and those in moderate condition were often located in agricultural areas. Sites in good condition were usually located in national parks or relatively unpopulated areas. Decreasing trends in nutrients were usually related to improvements in point source discharges, while increases in nutrients are suggestive of impacts from changing or intensifying land use. However, a detailed assessment of the trends in relation to changing land use has not been carried out (Cox *et al*., 2005).

Figure 2.2 shows the temporal pattern of river water quality sampling carried out by AIMS between 1987 and the present, encompassing a total of 16 rivers (see Appendix 1 for details of sampling periods and number of sampling occasions, Appendix 2 for a methods overview). It is the only sampling program in the Great Barrier Reef Catchment which included directed sampling to estimate loads exported from rivers. Manual nutrient sampling was terminated in 2000; instrumental sampling of suspended sediment loads has continued in 7 rivers. This latter data set is currently being analysed.

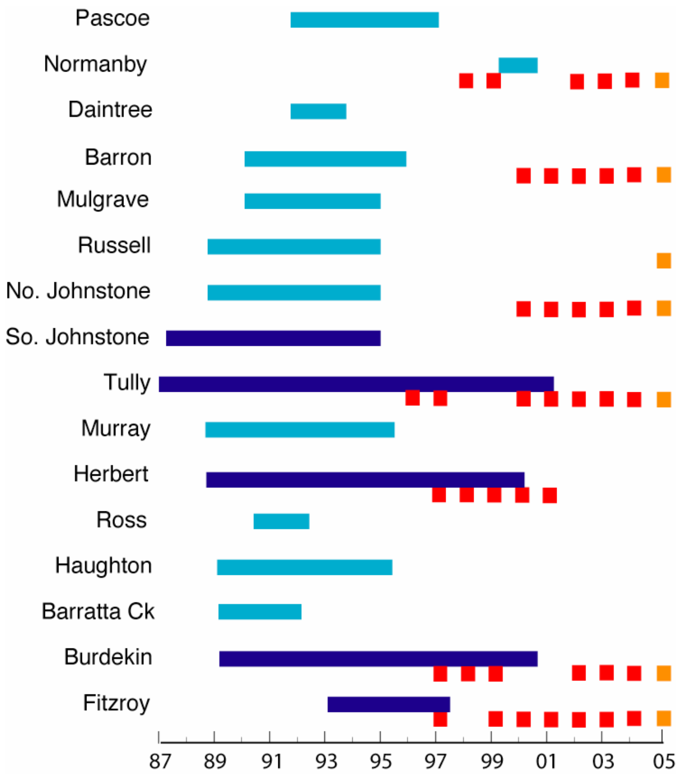

Figure 2.2 Timelines for sediment and nutrient sampling in rivers draining to the Great Barrier Reef by AIMS. Light blue bars indicate timeframes of low-intensity sampling for ambient water quality conditions. Dark blue bars indicate time frames for high-intensity wet-season sampling to estimate export loads. Short red bars indicate approximate wet season deployment times for autonomous turbidity loggers to estimate fine sediment loads. Yellow bars indicate rivers with deployed loggers for the 2004-2005 wet season.

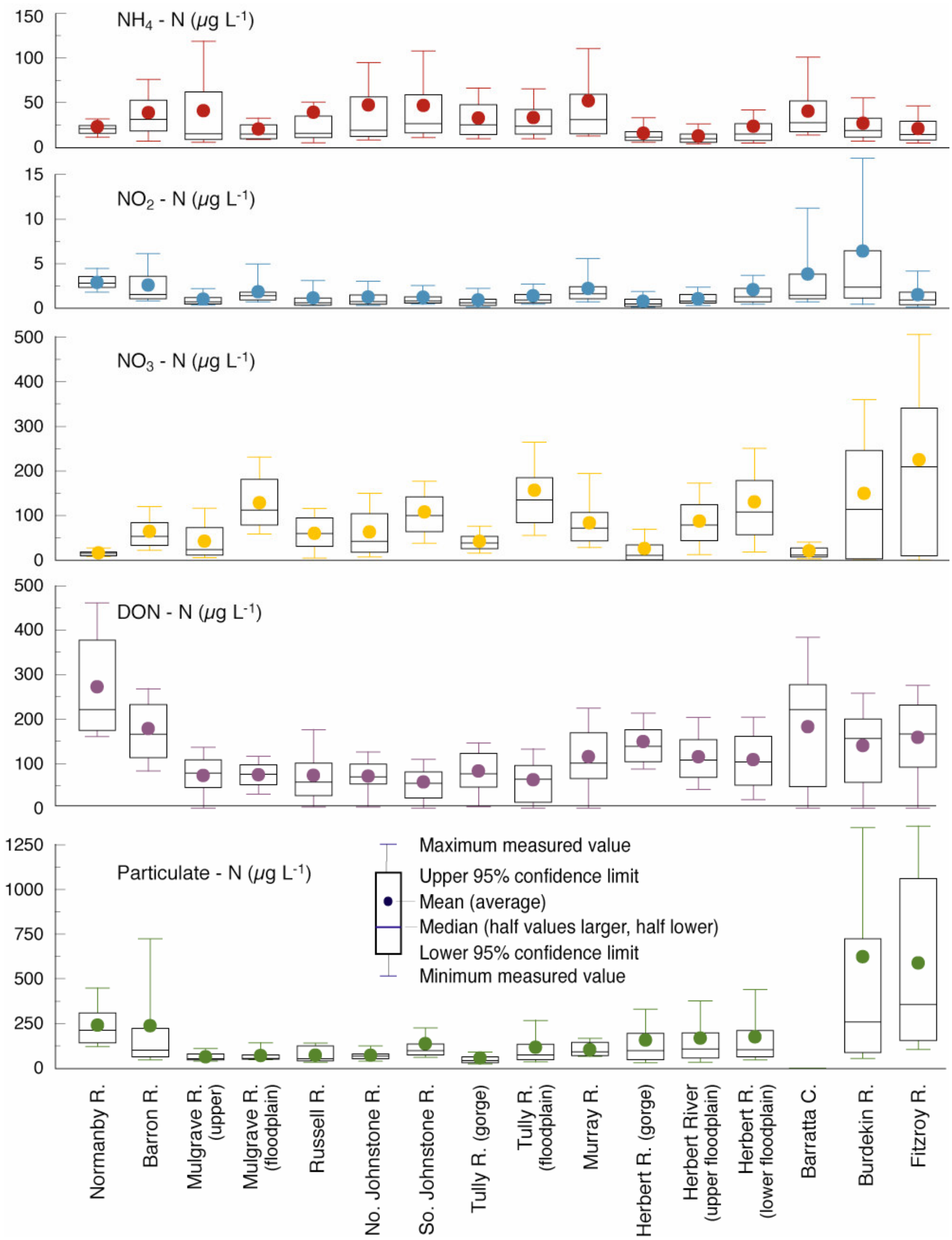

Figure 2.3 Concentrations and variability of dissolved and particulate nitrogen in rivers draining into the GBRWHA. Data summarized includes both wet- and dry-season sampling. A summary of data collected by AIMS between 1987 and 2000. From Furnas, (2003). See Appendix 1 for sample numbers.

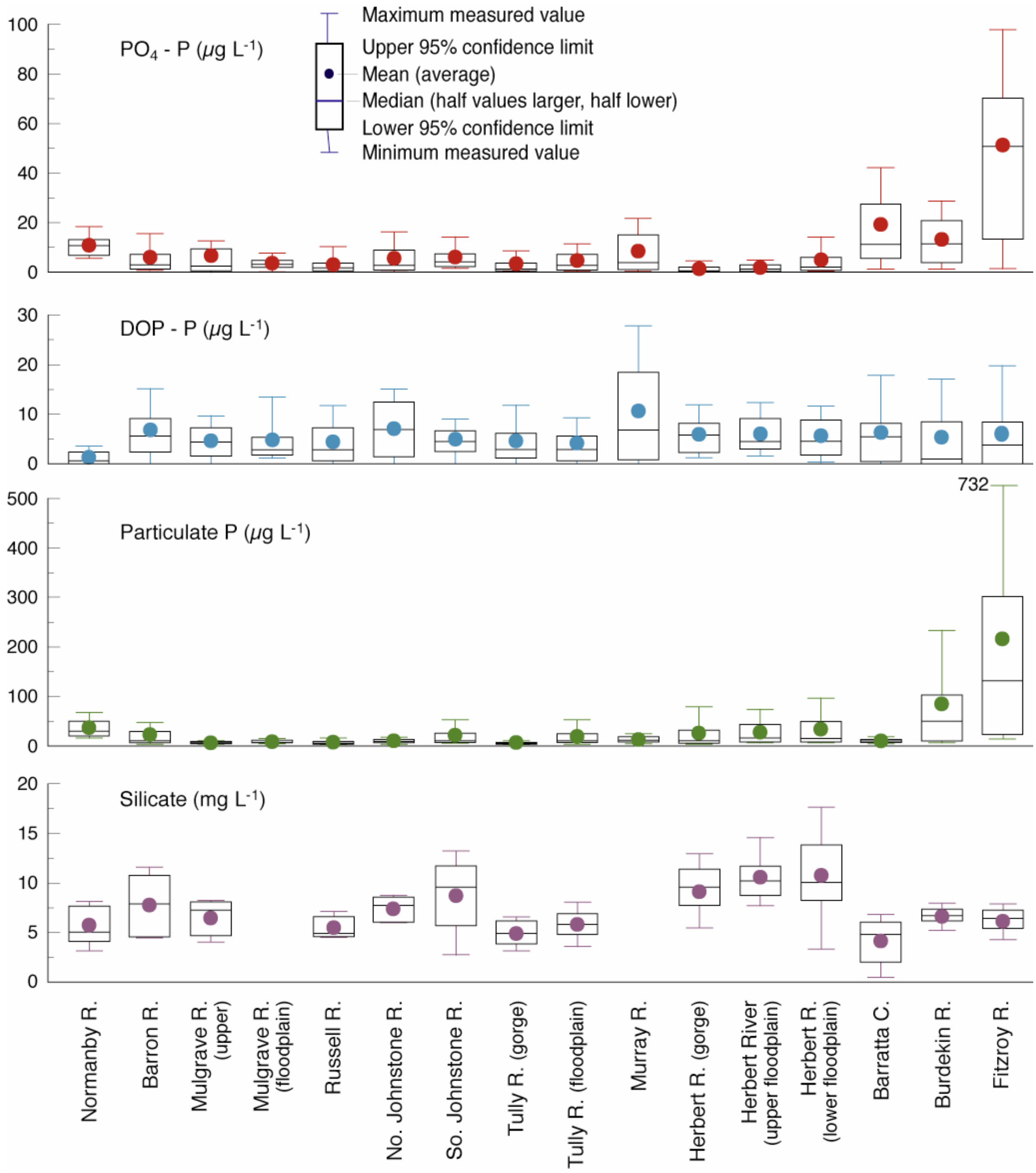

Figure 2.4 Concentrations and variability of dissolved and particulate phosphorus and silicic acid concentration in rivers draining into the GBRWHA. Data summarised includes both wet- and dry-season sampling. A summary of data collected by AIMS between 1987 and 2000. From Furnas, (2003). See Appendix 1 for sample numbers.

Particulate forms of nitrogen (PN) were generally the most abundant type of nitrogen in river waters, either as organic particles (mostly detritus) or bound to suspended sediment particles. The highest PN concentrations were consistently measured in the more turbid Burdekin and Fitzroy Rivers. PN concentrations in both wet- and dry catchment river waters are strongly correlated with suspended sediment concentrations (Furnas, 2003); this is particularly evident in the Tully and Burdekin River datasets (Appendix 3 Figure 1). In the dry-catchment rivers, the nitrogen content of suspended particulate matter closely matches the average nitrogen content of catchment soils. Particulate matter in wet catchment rivers was slightly nitrogen-enriched relative to catchment soils (Furnas, 2003).

The relative importance of nitrate (NO_3^-) and dissolved organic nitrogen (DON) varied between rivers. Elevated concentrations of DON were consistently measured in the dry catchment rivers (Normanby, Burdekin, Fitzroy). In contrast, nitrate was the predominant form of dissolved nitrogen in the lower reaches of several Wet Tropics rivers. Ammonium (NH_4^+) concentrations also tend to be higher and more variable in Wet Tropics rivers.

Nitrite was a relatively minor contributor to dissolved nitrogen pools, though elevated concentrations are indicative of enhanced microbial transformations of nitrogen between ammonium and nitrate. Higher concentrations of all nitrogen species were generally measured in the Tully and Burdekin Rivers during the summer wet season (Appendix 3 Figures 2 and 3) and during periods of high river flow (Appendix 3 Figures 4 and 5). Under some extreme flood conditions, however, dilution of dissolved nitrogen concentrations was observed. More discussion of temporal trends in these datasets is in Section 2.4.

Phosphate (PO_4^{3-}) concentrations in the dry catchment rivers are generally higher than those measured in wet catchment rivers (Figure 2.4; Appendix 3 Figure 6), though the difference is not great. Phosphate concentrations in river water are largely controlled by chemical solubility interactions between dissolved and particle-bound phosphorus, leading to relative stability in phosphate levels. There is no clear difference between dissolved organic phosphorus (DOP) concentrations in wet and dry catchment rivers (Figure 2.4, Appendix 3 Figure 6).

Most (70-80%) of the phosphorus in Great Barrier Reef Catchment rivers (wet and dry) is part of, or attached to, fine particulate matter (Figure 2.5), predominantly suspended soil particles. The highest particulate phosphorus (PP) concentrations measured are in the turbid waters of the Burdekin and Fitzroy Rivers (Figure 2.4). The lowest PP concentrations are found in the smaller Wet Tropics rivers even though a number of these rivers have soils derived from phosphorus-rich basalt. The mean phosphorus content of suspended particulate matter in both wet- and dry-catchment rivers closely matches the average phosphorus content of the catchment soils (Furnas, 2003).

Silicic acid concentrations were much higher than those of nitrogen and phosphorus (Figure 2.4), but showed no clear pattern with regard to wet- or dry- catchments or upstream-downstream location.

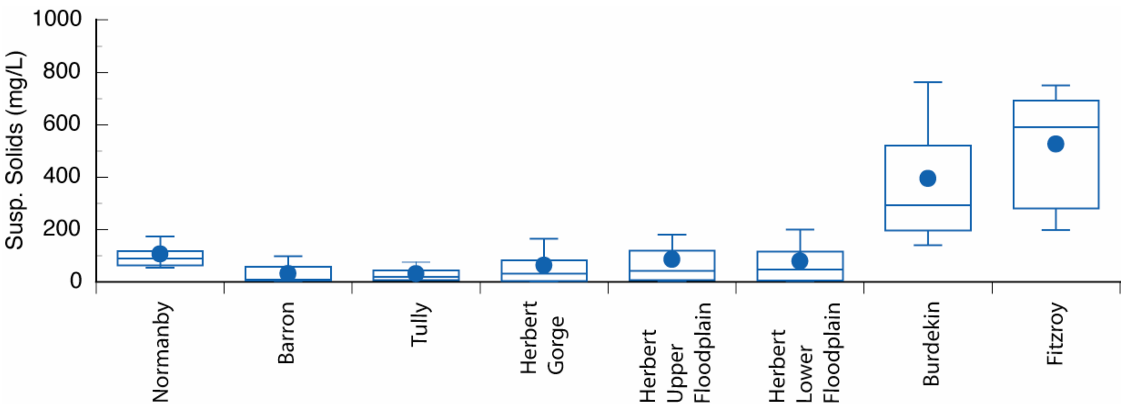

Figure 2.5 Concentrations and variability of suspended sediment concentration in rivers draining into the GBRWHA. Bar characteristics are as shown in Figures 2.3 and 2.4. The data summarised are from manual sampling programs and from rivers where a sufficient number of samples were taken to incorporate a representative mix of wet and dry season conditions. Data summarised includes both wet- and dry-season sampling. The data were collected by AIMS between 1987 and 2000. See Appendix 1 for sample numbers.

When they were flowing, suspended sediment concentrations in the two largest Dry Tropics rivers, Burdekin and Fitzroy, were considerably higher than those recorded in rivers of the Wet Tropics (Figure 2.5). These dry-catchment rivers are generally turbid when flowing due to the presence of fine clay particles. Ranges of suspended sediment concentrations in intermediate sized, largely-dry catchments, Normanby and Herbert, were intermediate between those of the Wet Tropics and the large dry catchments. Regardless of catchment size and type, suspended sediment concentrations in all rivers vary over large ranges, depending upon river flow rates and change rapidly with flow rates, sometimes within hours. The typical pattern is an abrupt rise in suspended sediment concentration as river levels and flow increase, followed by a more gradual, exponential decline over time. Peak suspended sediment concentrations during floods in the large dry catchment rivers range between 1 and 5 g per litre (tonnes per megalitre). In contrast, peak sediment loads in the wet catchment rivers are generally less than 1.5 g per litre. These high concentrations only last for a short time. As noted above, concentrations of particulate nitrogen and phosphorus are positively related to suspended sediment loads (Furnas, 2003).

Overall, volume-weighted sediment concentrations in the Burdekin and Fitzroy Rivers (average = 366 mg per litre) are nearly ten times greater than the in the well-sampled Tully River (average = 39 mg per litre) (Furnas, 2003). Volume-weighted sediment concentrations in the Normanby and Herbert Rivers with less developed or mixed catchments were between these concentrations (ca. 100 mg per litre) (Furnas, 2003). Total nitrogen and phosphorus concentrations in river waters from the dry Burdekin and Fitzroy River catchments are higher than total nitrogen and phosphorus concentrations in waters coming from Wet Tropics catchments. Volume-weighted total nitrogen and phosphorus concentrations in the large dry rivers averaged 840 µg nitrogen and 165 µg phosphorus per litre, respectively. In contrast, volume weighted total nitrogen and phosphorus concentrations in Wet Tropics rivers averaged 400 µg nitrogen and 42 µg phosphorus per litre. On a catchment area-specific basis, however, nitrogen and phosphorus exports from the Wet Tropics rivers are greater, reflecting the steeper topography and higher rainfall (erosion) rates in these catchments.

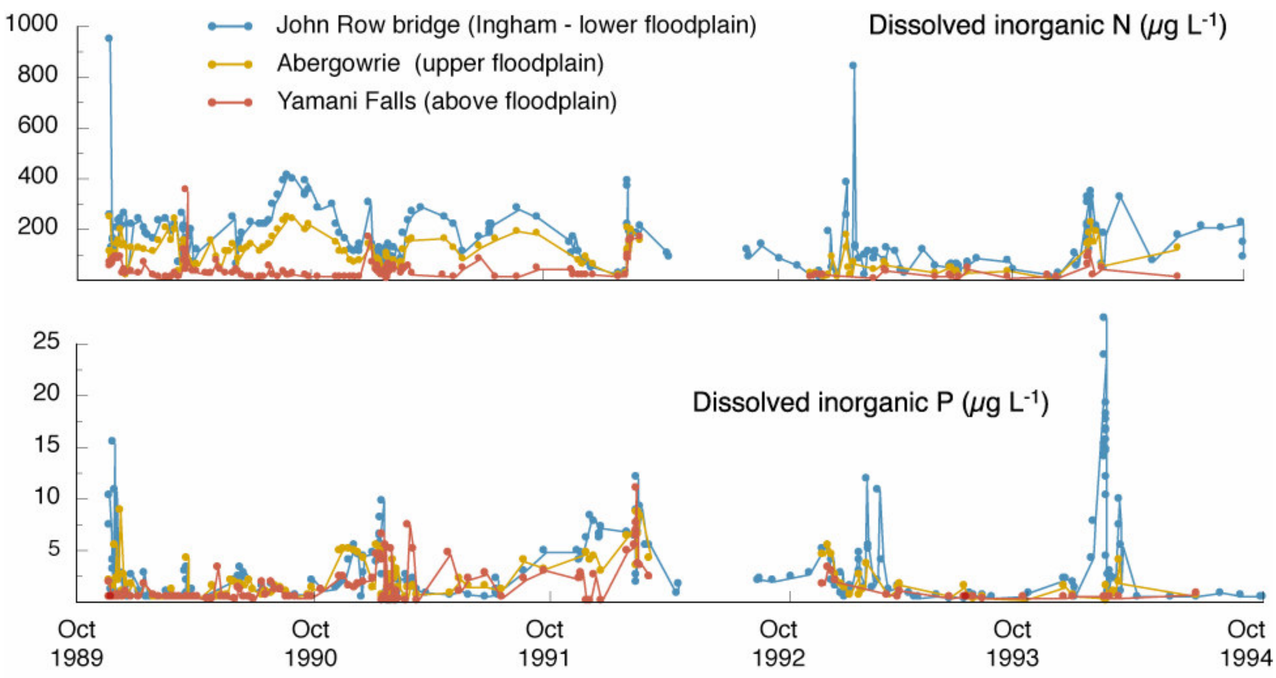

Figure 2.6 A comparison between concentrations of dissolved inorganic nitrogen (chiefly nitrate) and phosphorus sampled contemporaneously at three longitudinal sites on the lower Herbert River. From Furnas, (2003).

Where water samples were collected contemporaneously at two or more sites along rivers in the Great Barrier Reef Catchment, higher concentrations are often measured at flood plain sampling sites (Furnas, 2003; Loxton and Gittins, 2004; Cox *et al*., 2005). The degree of difference depends upon the river and season, the nutrient species in question and the nature of land use in the catchment. This downstream increase is particularly evident in Wet Tropics (e.g. Mulgrave, Tully, Herbert Rivers), where nitrate concentrations generally increase downstream as river waters flow through the floodplain (Figure 2.6). As it is unlikely that significant nitrogen fixation takes place on the floodplain, this downstream increase is most likely due to movement of fertiliser-nitrogen from soils and aquifers into the river. In contrast to inorganic nitrogen, there is no discernable upstream-downstream variation in any of the phosphorus species sampled. This is because phosphate concentrations are largely controlled by solubility interactions (or adsorption/desorption reactions) with sediment particles and sediment concentrations did not show strong longitudinal variations.

2.2 Water, sediment and nutrient exports to the Great Barrier Reef

Water is the principal driver of sediment and nutrient exports from the land to the Great Barrier Reef, from rainfall that erodes catchment soils, to the river plumes which finally carry materials into the Great Barrier Reef. Within the Great Barrier Reef Catchment, average annual rainfall ranges from more than 3,000 mm at sites in the Wet Tropics, to 700 mm along the western margin of the Fitzroy and Burnett River drainage basins. Each year on average, close to 380 km^3 of rainwater falls onto the Great Barrier Reef Catchment (1 km^3 = 1 billion m^3 = 1 million megalitres). Of this, an average of 71 km^3 runs off into the Great Barrier Reef (Table 2.1). On a year-to-year basis, however, runoff volume can vary nearly 3-fold from the mean value (27 km^3 in 1987 to ca. 180 km^3 in 1974) depending on the amount and location of rainfall into the Great Barrier Reef Catchment.

Estimates of current terrestrial sediment and nutrient inputs to the Great Barrier Reef have been made using two approaches:

1) intensive sampling of sediment and nutrient concentrations in representative types of rivers which are then integrated to produce volume-normalized discharge-export relationships that can be used to estimate exports from annual discharge data for un-sampled rivers (Furnas, 2003); and
2) spatial modelling of sediment (SEDNET) and nutrient (SEDNET/ANNEX) loss from landscapes and river systems based on regional rainfall, topography, soil type, vegetation cover and land use (NLWRA, 2002; Brodie *et al*., 2004).

At the scale of the whole Great Barrier Reef Catchment, these two approaches produce similar estimates of average annual sediment export to the Great Barrier Reef: 14 million tonnes per year (Furnas, 2003); 12 million tonnes per year (NLWRA, 2002); 16 million tonnes per year (Brodie *et al*., 2004), and yield similar latitudinal trends of sediment export on a catchment-by-catchment basis (Furnas, 2003). Based on inter-annual variations in rainfall, estimated annual sediment inputs to the lagoon can vary between 3 and 60 million tonnes per year (Furnas, 2003).

Estimates of nutrient (nitrogen and phosphorus) exports by the two approaches differ to a somewhat greater degree. Current terrestrial nitrogen and phosphorus inputs derived from direct sampling and extrapolation of derived discharge-export relationships are estimated to be close to 43,000 and 7,000 tonnes per year, respectively (Furnas, 2003). Parallel estimates from spatial modelling of runoff and erosion (SEDNET/ANNEX) are 63,000 tonnes of nitrogen and 11,000 tonnes of phosphorus per annum (Brodie *et al*., 2004). Discrepancies between the two estimates of nutrient export can be attributed to the relative applicability of discharge-export relationships derived for a small number of rivers to other catchments, and the still-considerable uncertainties in area-specific nutrient losses from landscapes or land uses within individual catchments. There is considerable management and research interest in improving both discharge-export relationships for individual catchments, and the quantitative understanding of soil and nutrient loss rates from different landscape and land use situations. The ca. 2-fold difference between average annual export estimates for the two approaches is small relative to the order-of-magnitude variation in annual river inputs of nitrogen (11,000-126,000 tonnes) and phosphorus (1,400 - 22,000 tonnes) due to inter-annual variations in freshwater runoff (Furnas, 2003).

Figure 2.7 shows estimates of current freshwater, suspended sediment, nitrogen and phosphorus exports to the Great Barrier Reef from terrestrial sources derived from river sampling and discharge-export relationships (Furnas, 2003), and the relative contribution of dissolved and PN and PP to total nutrient exports. Most nitrogen and phosphorus (70-80%) are exported from dry catchments in particulate form. Significant exports of dissolved nitrogen (up to 60%) (but not phosphorus) occur from the smaller high-rainfall catchments of the Wet Tropics. Up to half of the dissolved nitrogen exported from some Wet Tropics catchments may be in highly labile inorganic forms (nitrate and ammonia). Asterisks indicate the catchments selected for continued river mouth monitoring as part of the Reef Plan MMP. The largest inputs of freshwater, sediment and nutrients are derived from the Burdekin and Fitzroy Rivers.

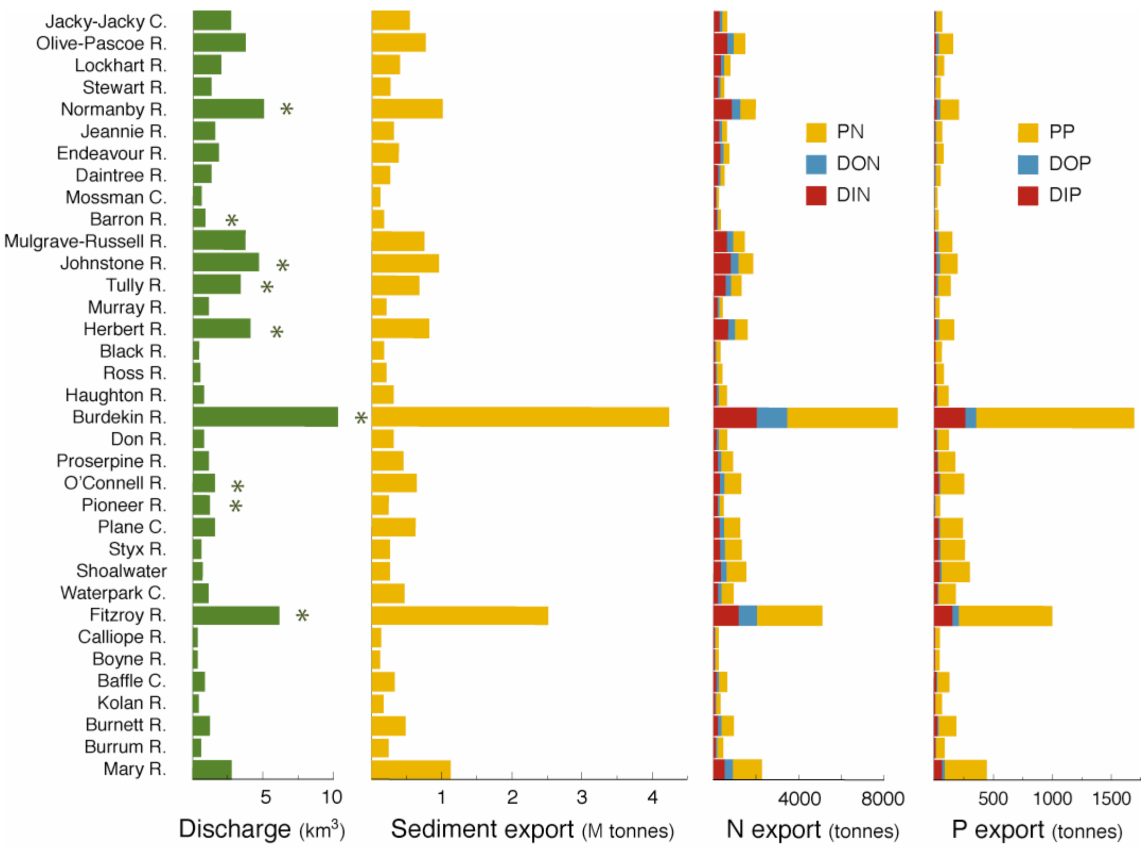

*Figure 2.7 Estimated freshwater discharges, fine sediment exports and exports of nitrogen and phosphorus from drainage basins flowing into the GBRWHA. * - basins selected for river mouth monitoring in the Reef Plan MMP. Modified from Furnas, 2003).*

2.3 Runoff, sediment and nutrient inputs by Regional NRM region

As the amount of sediment and nutrients delivered to the Great Barrier Reef in runoff is correlated with the volume of runoff (Furnas, 2003), annual terrestrial inputs of sediment and nutrients can also be expected to vary to a similar degree. Rainfall and runoff are highly seasonal, with most rainfall occurring during the summer wet season (December-April). Within each wet season, most freshwater discharge occurs during floods or high flow events associated with tropical cyclones and summer monsoonal rainfall. These events are usually brief, lasting for periods which range from several days in small Wet Tropics catchments to several weeks in the much larger Burdekin and Fitzroy River catchments. Year to year variations in discharge (and hence material exports) vary significantly between catchments. In Wet Tropics rivers (e.g. Tully River), inter-annual variations in discharge are relatively small (< 10-fold), while in the Dry Tropics of the central and southern Great Barrier Reef (e.g. Burdekin River), annual discharges can vary by 2 orders of magnitude. With a greater frequency of rainfall in the Wet Tropics catchments, these rivers may have from one to several flood events in a wet season. In contrast, rivers in the Dry Tropics typically have only a single significant flood event in a wet season, and during drought periods, may not flow at all. The proportion of rainfall into individual catchments which ultimately runs off into the Great Barrier Reef varies considerably. Average annual runoff from the Burdekin and Fitzroy River catchments, for example, is ≤ 10% of the average annual rainfall into those catchments. Conversely, average annual runoff from a number of Wet Tropics drainage basins (e.g. Mossman, Russell-Mulgrave, Johnstone, Tully) exceeds 50% of the average annual rainwater input.

Cape York

There are seven catchments within the Cape York NRM region (Table 2.1) with a total area of 42,918 km^2 (10% of the Great Barrier Reef Catchment).

Average annual runoff from these catchments (16.7 km^3) accounts for 24% of total freshwater runoff to the Great Barrier Reef. Suspended sediment and nutrient load sampling to date in this region has been limited to the Normanby River catchment. Average annual exports of fine sediment (960,000 tonnes), nitrogen (6,600 tonnes) and phosphorus (700 tonnes) from rivers of this region account for 6.7, 15 and 10% of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

Human population numbers within the Cape York region are very low and settlements are widely dispersed. While cattle grazing takes place throughout the region, cattle densities are much lower than in catchments bordering the central and southern Great Barrier Reef. As a result, nutrient and sediment levels in rivers on Cape York Peninsula are likely to be most similar to those prevailing in Great Barrier Reef Catchment rivers of comparable character prior to European settlement and agricultural development.

Wet Tropics

There are eight catchments within the Wet Tropics (Far North Queensland) NRM region (Table 2.1) with an area of 22,501 km^2 (5.3 % of the Great Barrier Reef Catchment). Average annual runoff from these catchments (19.3 km^3) accounts for 27% of total freshwater runoff to the Great Barrier Reef. Significant suspended sediment and nutrient sampling has been has been carried out in the Barron, Russell-Mulgrave, Johnstone, Tully and Herbert River catchments. Average annual exports of fine sediment (1.14 million tonnes), nitrogen (7,650 tonnes) and phosphorus (800 tonnes) to the Great Barrier Reef from rivers of this region account for 8, 18 and 11% of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

Sugarcane is the major agricultural crop in the Wet Tropics. A substantial fraction of the alluvial coastal plain has been cleared and planted with sugarcane and other tropical crops (e.g. bananas). These crops receive large inputs of N and P from agricultural fertiliser applications. Because of high levels of rainfall in most areas of the Wet Tropics region, there are high levels of vegetation cover which reduces erosion and sediment loss. Sediment losses are greatest in agricultural or newly cleared areas where there is bare ground.

Burdekin

There are five catchments within the Burdekin NRM region (Table 2.1) with an aggregate area of 140,629 km^2 (33 % of the Great Barrier Reef Catchment). Average annual freshwater runoff from these catchments (12.7 km^3) accounts for 18 % of total freshwater runoff to the Great Barrier Reef. Suspended sediment and nutrient sampling has only been carried out in the Burdekin River. Average annual exports of fine sediment (4.6 million tonnes), nitrogen (10,600 tonnes) and phosphorus (2,000 tonnes) from rivers of this region account for 32, 25 and 29 % of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

Cattle grazing is the primary land use in this region. Fluctuations in climate and cattle numbers greatly affect the state and nature of vegetation cover in these dry catchments, and therefore, the susceptibility of soils to erosion. There is extensive irrigated planting of sugarcane on the floodplains of the Burdekin and Haughton Rivers.

Mackay-Whitsunday

There are six catchments within the Mackay-Whitsunday NRM region (Table 2.1) with an area of 15,647 km^2 (3.7 % of the Great Barrier Reef Catchment). Average annual runoff from these catchments (8.7 km^3) accounts for 12 % of total freshwater runoff to the Great Barrier Reef. No substantial suspended sediment and nutrient sampling to measure export loads has been carried out in any of the rivers in this NRM region. Estimated annual exports of fine sediment (2.8 million tonnes), nitrogen (6,800 tonnes) and phosphorus (1,300 tonnes) from rivers of this region account for 19, 16 and 18 % of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

Fitzroy

There are five catchments within the Fitzroy NRM region (Table 2.1) with a total area of 153,194 km^2 (36.1% of the Great Barrier Reef Catchment). Average annual freshwater runoff from these catchments (8.6 km^3) accounts for 12 % of total freshwater runoff to the Great Barrier Reef. Suspended sediment and nutrient sampling has been limited to the Fitzroy River catchment to date. Average annual exports of fine sediment (3.1 million tonnes), nitrogen (7,200 tonnes) and phosphorus (1,400 tonnes) from rivers of this region account for 22, 17 and 20 % of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

As with the Burdekin NRM region, cattle grazing is the principal land use in this region. There has been extensive clearing of native vegetation in these catchments. Extensive irrigated plantings of cotton and grains occur in the upper portions of the Fitzroy River catchment.

Burnett-Mary

There are four catchments within the Burnett-Mary NRM region (Table 2.1) with an aggregate area of 48,946 km^2 (11.5 % of the Great Barrier Reef Catchment). Average annual runoff from these catchments (4.8 km^3) accounts for 7 % of total freshwater runoff to the Great Barrier Reef. No sediment or nutrient load sampling has been carried out in catchments of this region. Estimated annual exports of fine sediment (1.8 million tonnes), nitrogen (4,600 tonnes) and phosphorus (800 tonnes) from rivers of this region account for 12, 9 and 11% of total estimated river inputs, respectively, to the Great Barrier Reef (Figure 2.7).

2.4 Changes in river water quality and nutrient exports over time

Over the last 150 years, substantial changes to land cover and land use have occurred in the Great Barrier Reef Catchment (Furnas, 2003). These changes influence the amount of sediment and nutrients in runoff. Based on water quality sampling in pristine, or little disturbed streams (Furnas, 2003), and spatial modelling of runoff using pre-1850 estimates of vegetation cover (NLWRA, 2002; Brodie *et al*., 2004), it is estimated that average pre-1850 river exports of sediment, nitrogen and phosphorus to the Great Barrier Reef were on the order of 1-4 million tonnes, 20,000 tonnes and 2,400 tonnes, respectively. Based on conservative assumptions about likely pre-1850 nutrient loads in regional rivers derived from sampling and likely pre-European levels of vegetation cover, it can be estimated that sediment loads to the Great Barrier Reef have increased at least 3-fold, nitrogen loads have increased at least 2-fold and phosphorus loads have increased at least 3-fold. For example, nutrient exports from pasture land were 3 to 6 fold higher than to forested land (summarised in Brodie and Mitchell, 2005). These levels of increase are similar to global estimates of nitrogen and phosphorus exports comparing pristine and developed catchments (ibid.).

The changes to land use which affect runoff include: the widespread clearing and thinning of native forests and woodland vegetation for agriculture (ca. 40% of the Great Barrier Reef Catchment south of Daintree); the extensive grazing of cattle on natural and developed pastures throughout the catchment which reduces average grass cover; the development of cropping industries in both inland (e.g. grains, cotton) and coastal (sugarcane, horticulture) regions; and the extensive use of chemical fertilisers and other agricultural chemicals, particularly in the sugarcane industry. Over the last 50 years, usage of nitrogen and phosphorus in fertilisers has increased to an estimated application of 100,000 and 20,000 tonnes of nitrogen and phosphorus respectively in the Great Barrier Reef Catchment.

Because of the strong seasonality and event-based nature of rainfall and runoff in Great Barrier Reef Catchments, nutrient and suspended sediment concentrations in all rivers can be highly variable, making it difficult to detect subtle long-term trends. At present, high-density and long-term (> 10 years) time series are available for only two rivers: the Tully (wet catchment) and the Burdekin (dry catchment). Shorter time series (2-5 years) are available for a number of other rivers (Figure 2.2) as discussed in Section 2.1.

Recent changes in nitrogen and phosphorus concentrations have been detected in the Tully River, which flows year-round with common flood events (Furnas, 2003; De'ath, 2005). Visual inspection of the data indicates a progressive increase in baseflow (dry season) concentrations of nitrate, particulate nitrogen and phosphate in lower Tully River waters from ca. 1990 to 2000, when sampling ended (Figure 2.8). More detailed statistical analyses of the data (De'ath, 2005), accounting for seasonal and flow-related effects on nutrient levels, have confirmed a linear upward trend for particulate nitrogen and particulate phosphorus, while significant non-linear trends were detected for DON, total dissolved nitrogen and phosphorus, silicate and suspended sediment (Figure 2.9 and Appendix 3 Table 1). There is weak evidence for an increase in nitrate (P= 0.1104, Appendix 3 Table 1), because this parameter is strongly correlated with river flow, which also increased over the sampling period (Appendix 3 Figures 4 and 7). Sediment-associated particulate nitrogen and phosphorus has been increasing on the order of 4 and 6 % per year since 1990 (De'ath 2005). The beginning of this upward trend (ca. 1990) coincides with the start of significant land-use change in the Tully River catchment as wet tropical pastures (grass) were converted to more intensively cultivated and fertilised sugarcane and banana paddocks (Mitchell *et al.*, 2001).

No long-term linear trends in nutrient or suspended sediment loads can be detected in the Burdekin River data set (Appendix 3 Table 2 and Figure 8). This is likely to be due to the high natural level of flow and concentration variability in the data, but also to irregular sampling effort (Appendix 3 Figure 9).

Analyses of the Tully and Burdekin datasets showed that a 50% increase in concentrations of water quality parameters can be detected over a monitoring period of 10 years, assuming linear change over time (De'ath, 2005). The magnitude of variability shown in the longer data sets, especially the multi-year fluctuations, indicates that 10 or more years of high quality data will be required to reliably discern secular trends due to changing land use. For highly variable dry-catchment rivers (e.g. Burdekin, Fitzroy), even longer time frames may be required. This is supported by the statistical analyses undertaken by De'ath (2005).

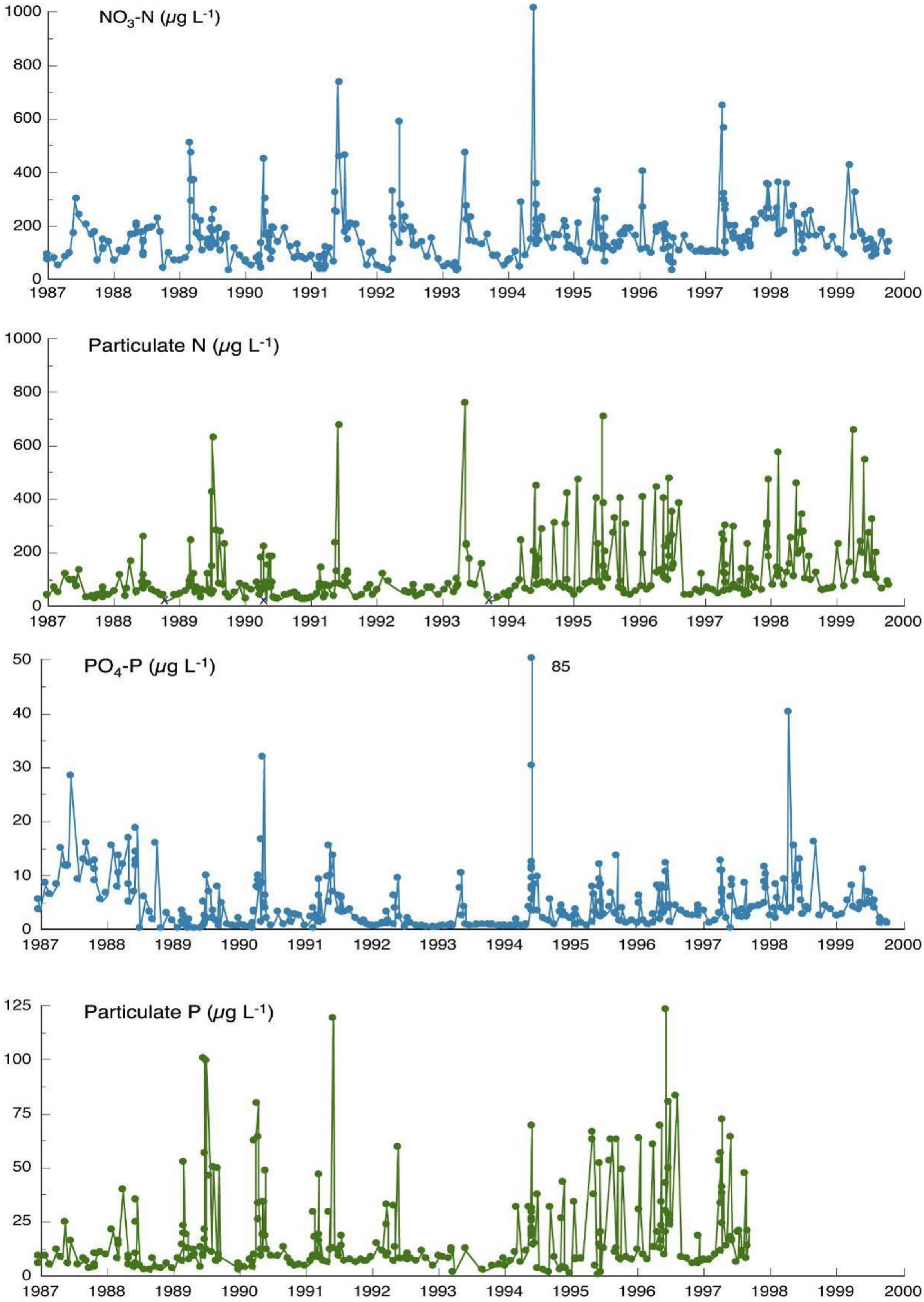

Figure 2.8 Time series of nitrate, particulate nitrogen, phosphate and particulate phosphorus measured in the lower Tully River between 1987 and 2000.

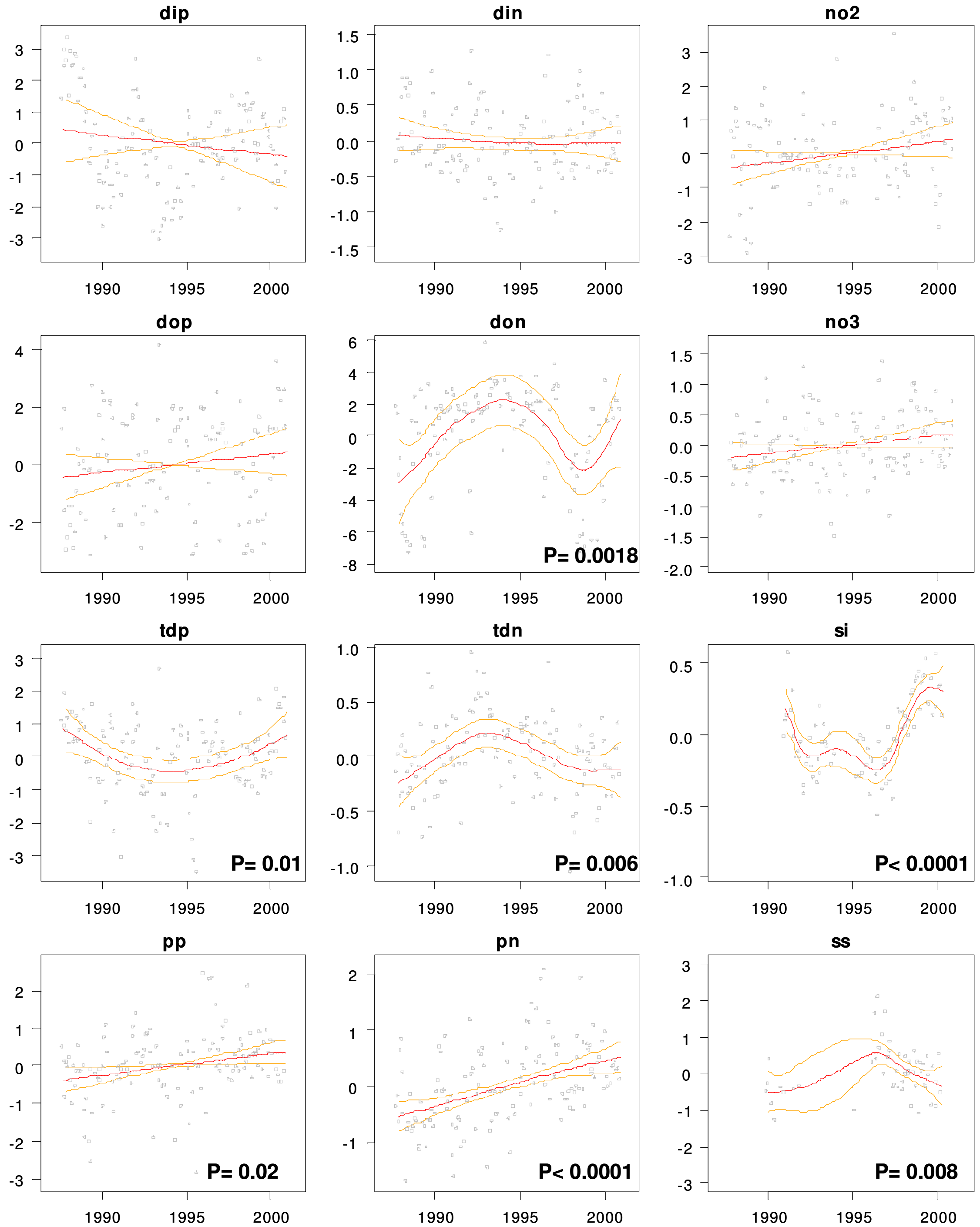

Figure 2.9 Estimation of temporal trends in water quality parameters for the Tully River over the period 1987-2000. Trends are estimated using smooth terms in a generalised linear model. The trends are adjusted for river flow effects and autocorrelation between successive months. Significant trends are indicated by the P value given inside figure panels. Data are log (base 2)-transformed. Units are µM for nutrient species [DIP= PO_4^{3-}, DIN, NO_2^-, DOP, DON, NO_3^-, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. Figure from De'ath (2005).

2.5 Pesticides and herbicides in Great Barrier Reef Catchment rivers

Herbicides and pesticides are widely used as part of modern agricultural and land use practices. These chemicals are used for a variety of purposes in the Great Barrier Reef Catchment. Although a detailed inventory of pesticide and herbicide use in all land use sectors has not been published, the sugar industry is likely the major user of herbicides and pesticides in the catchment (Hamilton and Haydon, 1996). A significant amount of chemicals are also used in the cotton industry.

To date, no long-term sampling programs or wide-ranging surveys of herbicide and pesticide levels in rivers of the Great Barrier Reef Catchment have been carried out. There have been a number of short-term studies carried out in several rivers. The first widespread survey detected the herbicides diuron and atrazine in intertidal sediments along the coast, ranging from 0.2 - to 10 µg kg^{-1} dry sediment, but generally at low levels of <1 µg kg^{-1} (Haynes *et al*. 2000a. The highest concentrations were measured in sediments collected near the mouths of rivers draining catchments with extensive sugarcane plantings. More recently, relatively high levels of herbicides have been measured in estuarine sediments of the Pioneer River (to 8 µg kg^{-1} dry sediment) and the Johnstone River (to 5 µg kg^{-1} dry sediment), while levels in the Daintree River were lower (to 1.1 µg kg^{-1} dry sediment) (Müller, pers. comm.).

First results from field studies using time integrated sampling techniques for polar organic pollutants ("passive samplers") showed detectable levels of three herbicides in both river and inshore waters of the Wet Tropics (Shaw and Müller, (in press); refer Section 3 of this report: Figure 3.13). Interestingly, river water concentrations of herbicides were higher before than after the wet season, while inshore waters showed higher concentrations during the wet season (see also Section 3 of this report). Concentrations in river water ranged from 0.2 ng L^{-1} atrazine to 2.5 ng L^{-1} simazine, with diuron concentrations around 1 ng L^{-1}.

Herbicides used in sugar cane production, particularly diuron, appear to have seriously affected mangroves in the Mackay region (Duke and Bell, 2005; Duke *et al*., 2003). An unusual species-specific dieback of *Avicennia marina* has been observed in the Mackay region since the mid 1990s (Duke and Bell, 2005). 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^{-1} for diuron), and in stream/drain waters flowing into mangrove areas (up to 1.2 µg L^{-1} for diuron) (Duke and Bell 2005). The highest herbicide concentrations measured to date were found in the Pioneer River (8.5 µg L^{-1} of diuron in a weir) following a flow event (Mitchell *et al*., 2005). Concurrently measured concentrations of the herbicide atrazine were 1.3 µg L^{-1}. Monitoring by the local Healthy Waterways program reported the first high flow event of the 2001 –2002 wet season transported 470 kg of diuron into the Pioneer River estuary, however, the high levels of diuron and other herbicides did not persist due to flushing and natural degradation processes (White *et al*., 2002).

Elevated diuron concentrations of 2.3 µg L^{-1} were also measured in spot samples taken during flood events in the Johnstone River. Sugarcane is farmed intensively in the lower parts of both the Pioneer and Johnstone River catchments. Maximum herbicide concentration (predominantly simazine) in the Mary River during a moderate river flow event was 4.3 µg L^{-1} (McMahon *et al*. 2005). The above values show that while normally low, herbicide concentrations in regional rivers can episodically reach levels of concern.

2.6 Ecological Implications

To affect water and habitat quality in the Great Barrier Reef, the state of water quality in rivers *per se* is of lesser importance than the quantity of materials that rivers deliver to the

Great Barrier Reef. Water quality within a particular river may be quite poor, but if there is little or no discharge, then there will be little or no effect on the Great Barrier Reef. It is the amount of sediment, nutrients and other materials in the water during peak discharge periods that matters most to the ecosystems of the GBRWHA.

Further discussion of the known implications of increased sediment, nutrient and pesticide loads to the Great Barrier Reef ecosystems are discussed in Section 7.

Conclusions

Runoff from the catchments adjacent to the Great Barrier Reef is the major source of nutrients, sediments and anthropogenic pollutants entering the Great Barrier Reef lagoon. Exports of these materials from the 424,000 km^2 Great Barrier Reef Catchment are driven by episodic seasonal rainfall events which produce short-lived wet-season floods. While the Great Barrier Reef has always received freshwater runoff, modern (post-1850) land use has substantively changed the quantities of sediment, nutrients and other pollutant materials in the runoff.

Most sediment and nutrients carried by rivers are derived from diffuse sources in agricultural lands. The pattern of land use varies widely across Great Barrier Reef Catchments. There are also noticeable differences between the water quality in, and exports from, wet- catchment rivers (Cape York to the Herbert River and the Pioneer River; catchments with an average annual rainfall generally > 1,500 mm) and dry-catchment rivers (catchments south of the Herbert River with an average annual rainfall generally < 1,500 mm). In most cases, higher sediment and nutrient concentrations are found in rivers draining dry catchments.

At the present time it is estimated that, on average, terrestrial runoff annually carries 11-14 million tonnes of fine sediment (largely silt and clays: particles < 63 µm carried as suspended load); 40,000-64,000 tonnes of nitrogen and 7,000-14,000 tonnes of phosphorus into the Great Barrier Reef. The ranges reflect different approaches applied to estimate average annual exports. On a year-to-year basis, runoff inputs can vary 3-fold from the average due to inter-annual climate-driven variations in the amount and source of freshwater runoff. A variety of evidence indicates that current average annual inputs of sediments and nutrients are 2- to 4 times greater than average annual inputs prior to European settlement of the catchment and the introduction of modern agricultural development. The primary reasons for the increases in sediment and nutrient exports are enhanced soil erosion and losses of agricultural fertilisers. There is no evidence that these factors are decreasing.

Much of the nitrogen (40-80%) and most of the phosphorus (70-80%) carried by rivers are transported in particulate form, attached to fine eroded soil particles (largely clays). Loads of sediment, nitrogen and phosphorus carried by the large rivers of the Dry Tropics (Burdekin and Fitzroy) during flood events are 2- to 4 times those carried in Wet Tropics rivers. However, the Wet Tropics rivers have higher losses on a per-area basis due to their steeper topographies and higher rain-driven erosion rates.

River sampling programs in a number of catchments (for example Mulgrave, Tully and Herbert) show that nutrient concentrations, particularly of soluble forms (e.g. nitrate), increase as river waters cross floodplains with extensive agricultural development.

A number of long term datasets are available for individual catchments to demonstrate changes in sediment and nutrient exports. For example, significant increases (4-6% p.a.) in particulate nitrogen and phosphorus concentrations were observed in the lower Tully River

(Wet Tropics) over a 10-year period (1990-2000). These increases coincide with an intensification of agricultural land use in the catchment. Similar changes over time were not detected in the Burdekin River (Dry Tropics). This is likely to be due to several factors including high natural variability in water flow and sediment and nutrient loads, less sampling effort, and patterns of land use have not materially changed over the last two decades.

Pesticides have been used within the Great Barrier Reef Catchment for many years in association with the development and expansion of agricultural and urban activities. Initial monitoring of some rivers and estuaries of the Great Barrier Reef Catchment reveal low levels of herbicides (several nanograms per litre) in two Wet Tropics Rivers (Russell-Mulgrave, Johnstone) with active cropping on the adjacent floodplain. Higher levels of herbicides at biologically significant levels (several micrograms per litre) were measured in floodwaters of the Johnstone and Pioneer Rivers, indicating that under certain conditions, significant amounts of these agricultural chemicals can be flushed into regional river systems and subsequently the Great Barrier Reef. Herbicides are considered to be a major factor in causing the currently observed mangrove dieback in estuaries of the Mackay region.

The highly variable flow regimes of the monsoonal rivers of the Great Barrier Reef Catchment pose significant challenges for the monitoring of exports to the Great Barrier Reef and tracking progress in implementation of the Reef Plan. Discharges, sediment loads and nutrient concentrations vary significantly on an episodic, seasonal and inter-annual basis. Statistical analyses pf existing datasets indicate that monitoring time frames of 10 years, or longer, are necessary to reliably establish net changes in water quality parameters. High quality datasets of 2 to 13 years duration are now available for a number of rivers draining into the GBRWHA. These datasets will provide the basis for baselines against which the Reef Plan can be evaluated.

Volume-normalised export loads are the most important performance measure of catchment status in terms of water quality in the Great Barrier Reef lagoon. However, high quality estimates of exports are only available for a small number of rivers and years. Effective sampling regimes to measure loads require high intensity sampling of short-lived flood or high-flow events, as is being implemented in the current Reef Plan MMP. Due to the high natural climate variability of the region, export-monitoring programs should be sustained over periods of 10 years or longer to produce comparable results.

3. Water quality in the Great Barrier Reef Lagoon

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Introduction

The Great Barrier Reef is the most conspicuous geological and biological feature of the north-eastern Australian continental shelf. The biological productivity of the Great Barrier Reef is supported by nutrients (e.g. nitrogen, phosphorus, silicate, iron), which are supplied by a number of processes and sources (Furnas, 1997; 2003). These include upwelling of nutrient-enriched subsurface water from the Coral Sea, rainwater, fixation of gaseous nitrogen by cyanobacteria and freshwater runoff from the adjacent catchment. Freshwater runoff is the largest of these nutrient sources (see Section 2). Direct point source inputs such as sewage discharges, are a very small fraction of total inputs. Most of the inorganic nutrients used by Great Barrier Reef marine plants and bacteria come from recycling of nutrients already within the Great Barrier Reef ecosystem (Furnas *et al*., 2005).

Within the boundaries of the GBRWHA, the continental shelf (depth < 80 m) has an area close to 225,000 km^2. Coral reefs cover approximately 10% of this shelf area. The shallow waters of the continental shelf are separated from the deeper low-nutrient waters of the Coral Sea by the matrix of reefs on the outer half of the shelf and along the shelfbreak, forming the Great Barrier Reef lagoon. The Great Barrier Reef lagoon has a volume of approximately 7,600 km^3 with an average depth of 36 m. Along the coast, approximately 16% of total shelf area is less than 20 m deep. This coastal band contains less than 4% of shelf volume. While this zone is small relative to the full Great Barrier Reef lagoon in terms of area and water volume, it is the primary recipient of freshwater runoff from the land and is most susceptible to resuspension of sediments by recurrent winds and wave action.

The large size and structural complexity of the Great Barrier Reef results in persistent regional water quality differences. This is because different regions are exposed to differing degrees of along-shelf mixing or lateral exchanges with the adjacent Coral Sea, and receive differing amounts of nutrient input from rainfall, upwelling and runoff. Because of the relatively small size of catchments bordering the far northern Great Barrier Reef, the low levels of agricultural or grazing activity in these catchments and low human populations, the waters of the far-northern Great Barrier Reef are likely closest to a pristine state.

The effects of terrestrial runoff and wind-driven sediment resuspension on water quality will be greatest in shallow waters near the coast as these areas are the first recipients of runoff, and most terrestrial sediment is deposited and retained near the coast. The major nutrient input processes are known to vary seasonally due to the monsoonal climate of the region, with most rainfall on land and sea falling during the summer (December - April). Large changes in lagoonal water characteristics, nutrient concentrations and biological productivity may occur for short periods of time following natural disturbances such as tropical cyclones or monsoonal flooding. Efforts to detect and monitor net changes in water quality over decadal time frames due to human activities, and to understand the role of water quality in reef health, must take into account these natural regional, seasonal and event-related variations.

Over the last 15 years, there have been a number of assertions in scientific and popular literature that Great Barrier Reef lagoon has become nutrient-enriched (eutrophied) or more sediment laden. With the exception of a small number of long-term datasets, these have largely been based upon comparisons between recent water quality programs and the very limited amount of data available from studies conducted more than 25 years ago. While such studies are instructive, the extent to which they can be used for direct comparison is limited. Sampling and analytical methods have changed greatly over the last few decades and care must be taken to avoid misinterpretations of small differences. Estimates of change derived from differences between the results of short-term (less than 1 year) programs undertaken at different times may also be confounded by natural variations in environmental conditions over daily to multi-year time frames.

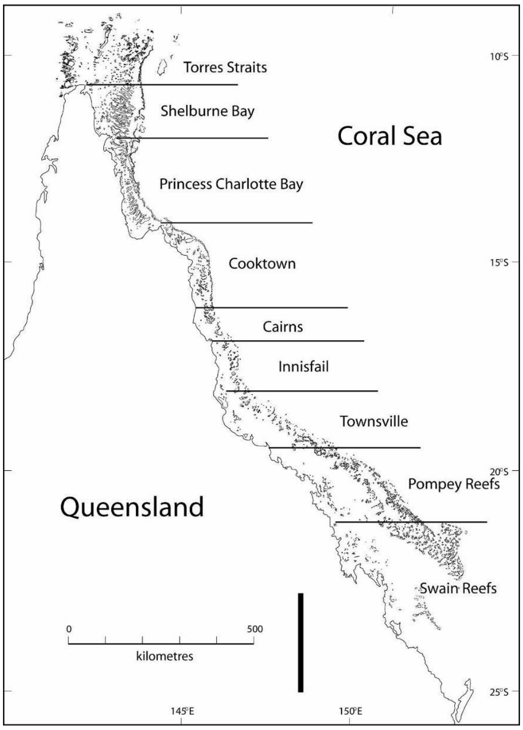
This section presents results from three long-term (10 to 15 year) and ongoing monitoring programs that resolve both long- and short-term levels of variability in water quality parameters. These programs have been carried out by AIMS using a consistent set of internationally recognised sampling and analytical methods (see Appendix 2 for an overview of methods used).

The spatial and temporal variability of water quality parameters (nutrients, suspended particulate matter, plant pigments) in the Great Barrier Reef lagoon and the first results of the analyses of long-term temporal trends are described. In total, data from over 3,000 sampling events are held at AIMS (Figure 3.2; see Appendix 4 for details on sampling periods and sample numbers). Detailed statistical analyses of this dataset are not yet available. Spatial variability of water quality parameters is presented as long-term medians.

3.1 Regional variations in water quality and nutrient concentration

For regional-scale data presentation, the Great Barrier Reef is divided into nine latitudinal bands (Figure 3.1) and two cross-shelf zones. The latitudinal bands are based on the large-scale structure of the Great Barrier Reef related to reef distribution, natural barriers to north-south water movement and historical sampling effort over the last 25 years. In the cross-shelf direction, the shelf can be divided into a shallow inner-shelf (inshore) zone less than 25 km wide and less than 20 m deep where wind-driven sediment resuspension takes place on a regular basis, and a deeper outer-shelf (offshore) zone. Wind stress and the Coriolis force also tend to hold river plumes within this narrow coastal band.

Figure 3.1 The Great Barrier Reef. For analysis and discussion of regional variations in water



quality, the Great Barrier Reef has been divided into nine latitudinal bands and two cross-shelf zones. The approximate width of the 25 km inshore zone is shown by the dark vertical bar.

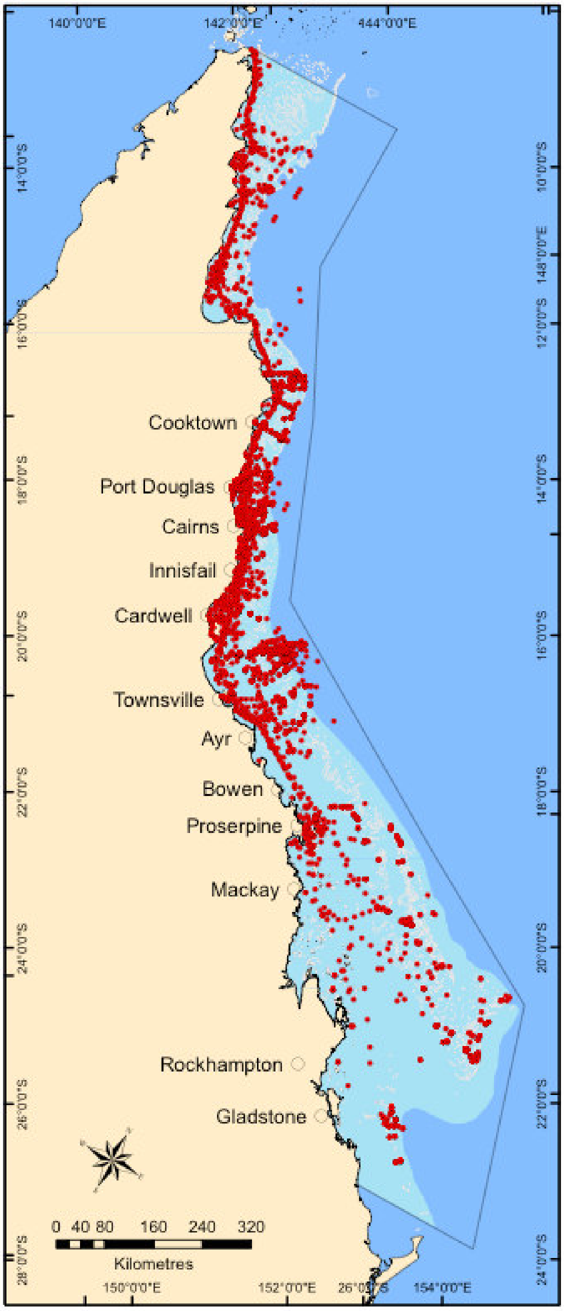


Figure 3.2 Locations of stations (n = 3,000+) within the GBRWHA where water quality data has been collected by AIMS between 1980 and the present. Note that many sites in the region between Townsville and Cooktown have been repeatedly sampled over the years.

The bottom three charts in Figure 3.3 illustrate the relative abundances of bio-available inorganic nitrogen (DIN = ammonium + nitrate + nitrite), DON and (largely organic) PN in Great Barrier Reef lagoon waters. There is no consistent latitudinal, seasonal or cross-shelf distribution pattern.

In most locations, water column concentrations of bio-available inorganic nitrogen are very low (< 1 µg N /L for individual forms, refer to Figure 3.3). This is due to the rapid uptake of these forms of nitrogen by phytoplankton, bacteria and macrophytes (Furnas *et al.*, 2005).

Direct measurements of inorganic nitrogen uptake indicate that ammonium and nitrate in Great Barrier Reef lagoon waters are turned over on time scales ranging from less than one hour to a few days (Furnas *et al*., 2005). Slightly higher nitrate concentrations are often found in deeper outer shelf regions between Innisfail and the Pompey Reefs (Mackay) where nutrient-enriched water from deeper areas of the Coral Sea episodically intrudes along the sea floor ('upwelling'). Elevated ammonium concentrations in inshore waters usually reflect nutrients released during episodic resuspension of coastal sediments and, occasionally, nutrient input from freshwater plumes from rivers.

DON is the largest pool of water column nitrogen. DON is a very heterogeneous mixture of various organic forms which differ greatly in their concentration and bio-availability to algae and bacteria. Some forms such as urea or amino acids are quickly utilized and are normally present at very low concentrations (ng/L). Most DON is part of larger molecules that are more resistant to biological degradation and utilisation. PN forms a significant proportion of total water column nitrogen (ca. 15-25%). This category includes living plankton and non-living organic matter (detritus). Throughout the Great Barrier Reef, a large proportion of the PN is associated with detritus. Measurements of vertical particle and organic matter fluxes with sediment traps (Furnas *et al*., 1995; Furnas, unpublished) indicate that much of the particulate organic matter in the water column of the Great Barrier Reef lagoon is resuspended from the bottom by wave action. This organic matter is available as food to a range of filter-feeding organisms in the plankton, on the bottom and on reefs.

Dissolved and particulate phosphorus concentrations in Great Barrier Reef waters are also very low (Figure 3.4), again largely due to high levels of demand by phytoplankton, bacteria and algae. Dissolved inorganic phosphorus concentrations (DIP) are generally less than 1 µg P/L. Slightly elevated DIP concentrations were measured in the Torres Strait and Shelburne Bay sectors and in the Innisfail and Townsville sectors where upwelling is seasonally observed. As with nitrogen, DOP is often the most abundant form. The highest DOP concentrations appear to occur during the dry season, particularly in inshore waters. The very high PP concentrations in inshore waters off Townsville are likely to be due to samples collected shortly after a flood event.

Silicic acid (Si) concentrations exhibit both a latitudinal and seasonal pattern (Figure 3.4). The highest Si concentrations are measured in inshore waters of the central Great Barrier Reef (Cairns - Townsville). This section receives runoff water from the Burdekin River, the largest river flowing into the Great Barrier Reef, and a number of rivers draining catchments of the Wet Tropics. The consistently higher concentrations in inshore waters reflect the runoff source of most of the Si (oceanic Si levels are very low). While silicic acid is required by some types of phytoplankton (diatoms), the dominant phytoplankton in Great Barrier Reef waters (cyanobacteria) have no requirement for Si. As a result, it is unlikely that Si concentrations in Great Barrier Reef waters are limiting to phytoplankton growth rates and biomass accumulation.

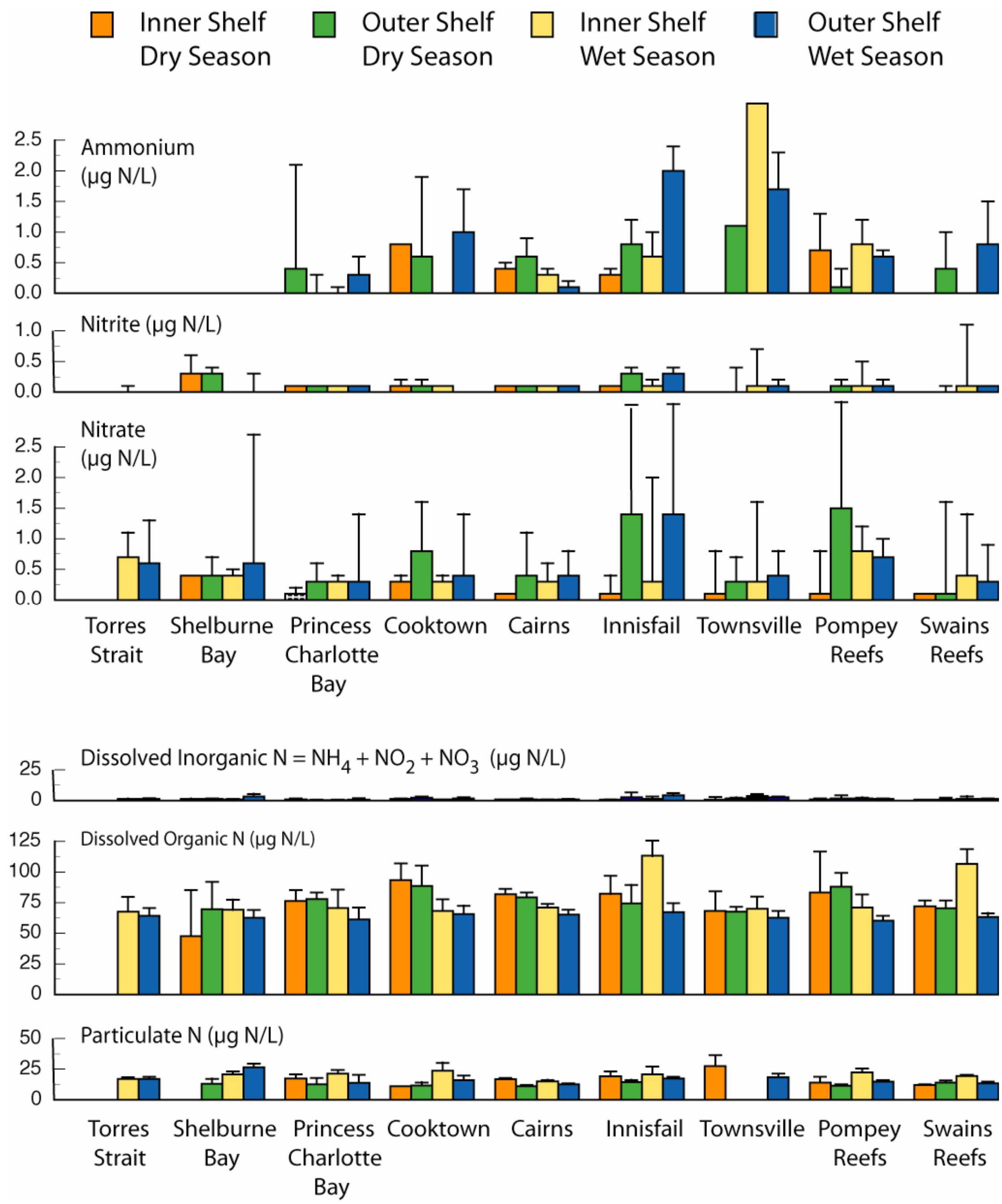


Figure 3.3 Latitudinal, cross-shelf and seasonal median concentrations of fixed nitrogen species in Great Barrier Reef lagoon waters sampled between 1980 and 2005. Error bars indicate the upper 95% confidence interval for the average concentration. The lower three histograms indicate the relative concentrations of total dissolved inorganic, dissolved organic and particulate forms of nitrogen. Modified from Furnas et al., (2005). See Appendix 4 for sample numbers.

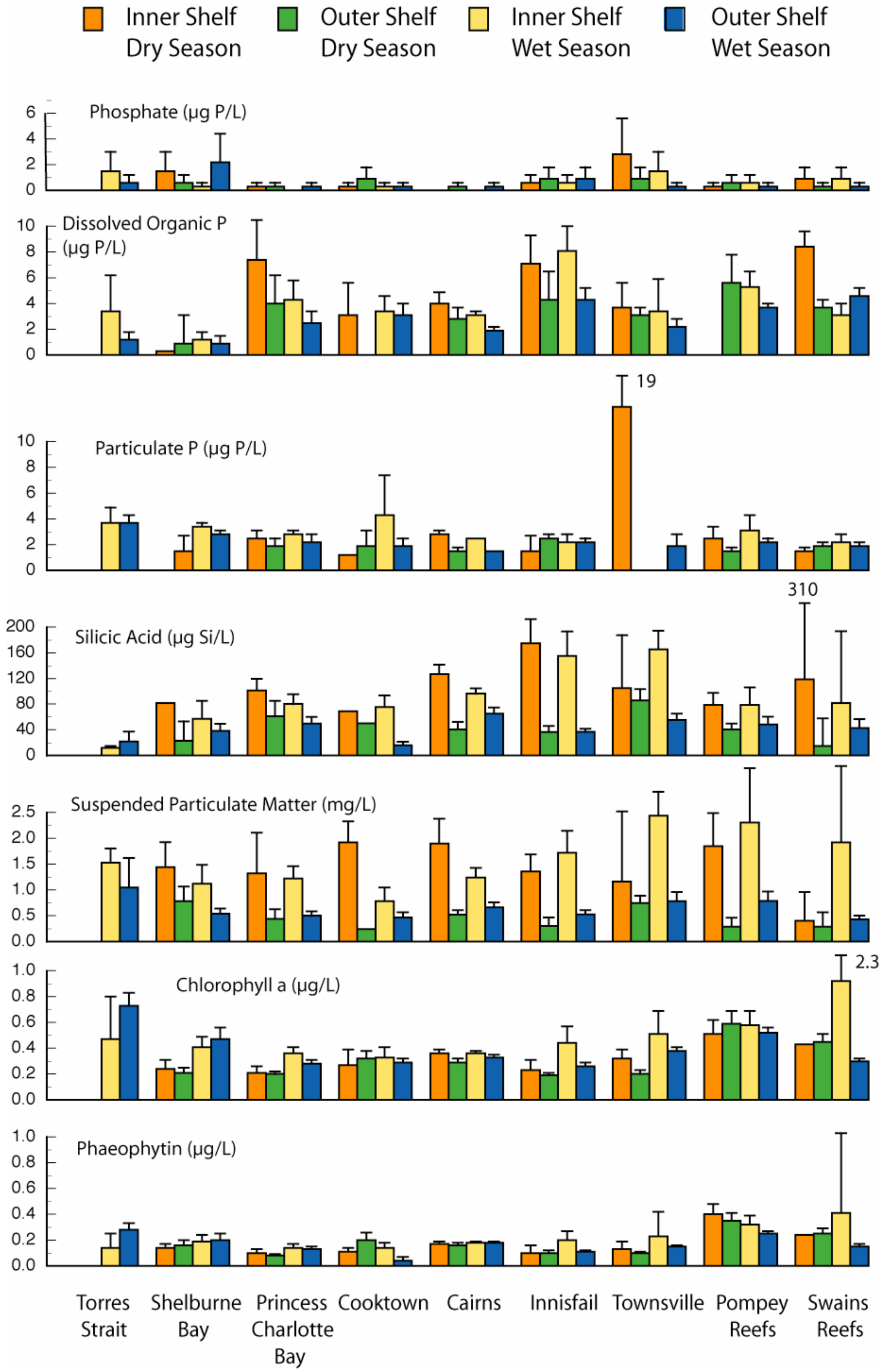


Figure 3.4 Latitudinal, cross-shelf and seasonal median concentrations of phosphorus species, silicic acid, suspended particulate matter, chlorophyll a *and phaeophytin in Great Barrier Reef lagoon water sampled between 1980 and 2005s. Error bars indicate the upper 95% confidence interval for the average concentration. Modified from Furnas et al., (2005). See Appendix 4 for sample numbers.*

Not surprisingly, the highest concentrations of suspended particulate matter are found in shallow (generally < 20 m) inshore waters (Figure 3.4). At most latitudes, offshore concentrations are considerably lower. During the wet season, the highest suspended sediment concentrations are in the Torres Strait sector, where tidal currents are often intense, and in coastal waters between the Innisfail and Mackay sectors. The highest concentrations are typically measured in bays near coastal rivers where fine sediments can accumulate.

At the broad regional scale, chlorophyll *a* concentrations exhibit a bimodal pattern, with higher concentrations at the far-northern and far-southern ends of the Great Barrier Reef (Figure 3.4). Median concentrations of phaeophytin generally track those of chlorophyll *a*, but are lower (Figure 3.4). Phytoplankton in warm tropical waters have very high growth rates and therefore a very high demand for nutrients, so regional distributions of chlorophyll *a* are used as a proxy indicator of nutrient (chiefly nitrogen) availability. The source of the nitrogen to support the higher northern chlorophyll *a* concentrations is unresolved. The low silicic acid levels in the Torres Strait sector indicate that river runoff, for example, from the Fly River in Papua New Guinea, is not a significant nutrient source. The most likely alternative is local nitrogen fixation by pelagic (*Trichodesmium*) and reef-associated cyanobacteria in the shallow waters with abundant coral reefs. Higher chlorophyll *a* concentrations in southern Great Barrier Reef waters, particularly around the Pompey Reefs sector appear to be due to upwelling onto the shelf, very active mixing by tidal currents and relatively long water residence times. The generally lower chlorophyll *a* concentrations in the central Great Barrier Reef reflect active onshore flow or mixing of low-chlorophyll *a* Coral Sea surface waters into the central Great Barrier Reef (Brinkman *et al*., 2001). Averaged at the broad regional scale, seasonal and cross-shelf differences in median chlorophyll *a* concentrations are relatively small.

While waters of the Great Barrier Reef lagoon are generally characterised by relatively low dissolved and particulate nutrient concentrations and small regional differences, considerably higher concentrations of most nutrient species may occur for short periods within regional areas following major disturbance events such as tropical cyclones or major floods in rivers (Brodie and Furnas, 1996, Devlin *et al*., 2001; Devlin and Brodie, 2005). River flood plumes from these events occur most frequently in coastal waters bordering the Wet Tropics (Figure 1.1). Catchments in this region receive flood-producing rainfalls nearly every year, and sometimes several times per year. Rivers of the Dry Tropics produce floods much less frequently, in the order of one every several years from the Burdekin River to every few decades from the Fitzroy River.

Once out of the rivers, flood plumes are generally restricted to coastal and inner-shelf waters. However, major flood plumes from the Burdekin and Fitzroy River catchments can influence large areas of the shelf due to the size of these catchments (for example, Figure 3.5). In some cases, plumes from the Burdekin River can be tracked more than 400 km to the north, near Cairns (Wolanski and van Senden, 1983). Large changes in nutrient concentrations and water quality can also occur over large areas (1,000's of km^2) following major mixing events associated with tropical cyclones (Furnas, 1989; Chongprasith, 1992). These perturbations of nutrient levels are due to the release of high nutrient porewaters from shelf sediments and the mineralisation of sediment organic matter in the water column. Nutrient concentrations have been recorded up to 10-100 times above ambient levels during these events (Devlin *et al*., 2001).

Nutrient behaviour from terrestrial discharges in the Great Barrier Reef lagoon has recently been reviewed in Devlin and Brodie, (2005). Most suspended particulate matter deposits from the plume close to the river mouth, often within a few kilometres of the mouth. Thus most of the particulate nutrient material will also be lost from the water column in this zone and not transported any great distances in the plume. In contrast, in the absence of biological uptake there is almost no loss of dissolved nutrients, except by dilution, in the plumes until salinities rise to above 25ppm which is generally 50–200km from the river mouth. The main reason for lack of biological uptake and phytoplankton growth appears to be the elevated turbidity in the early stages of the plume and the consequent light limitation. However, once particulate matter in the plume has settled, allowing light to penetrate into the water column, biological uptake of nutrients particularly of nitrogen, is rapid with short-lived blooms of phytoplankton developing within days (Furnas, 1989; McKinnon and Thorrold, 1993). After a bloom, the nutrients incorporated into plankton biomass are then distributed throughout planktonic and benthic food webs (Furnas, 2003).

The implications of the contrasting behaviour of particulate nutrients and dissolved nutrients are that nutrients discharged from rivers in dissolved form are transported great distances in the plume. They thus have the ability to influence biological activity on much of the inner-shelf of the Great Barrier Reef. Nutrients discharged in a particulate form are trapped near the coast and probably do not have a major influence on, for example, most of the inner-shelf coral reefs. These results have important implications with respect to the degree of exposure of inner-shelf ecosystems to river sourced nutrients and suspended particulate matter. As different forms of nutrients are exported from different land uses on the catchment the results can also help decide on priorities for management to reduce export from specific land uses. In general it is very clear that the primary area where flood plumes are common is the inner shelf and that ecosystems in this area are at most risk from pollutants contained in river discharge (Brodie *et al*., 2001; Brodie, 2002; Furnas, 2003; Fabricius and De'ath, 2004).

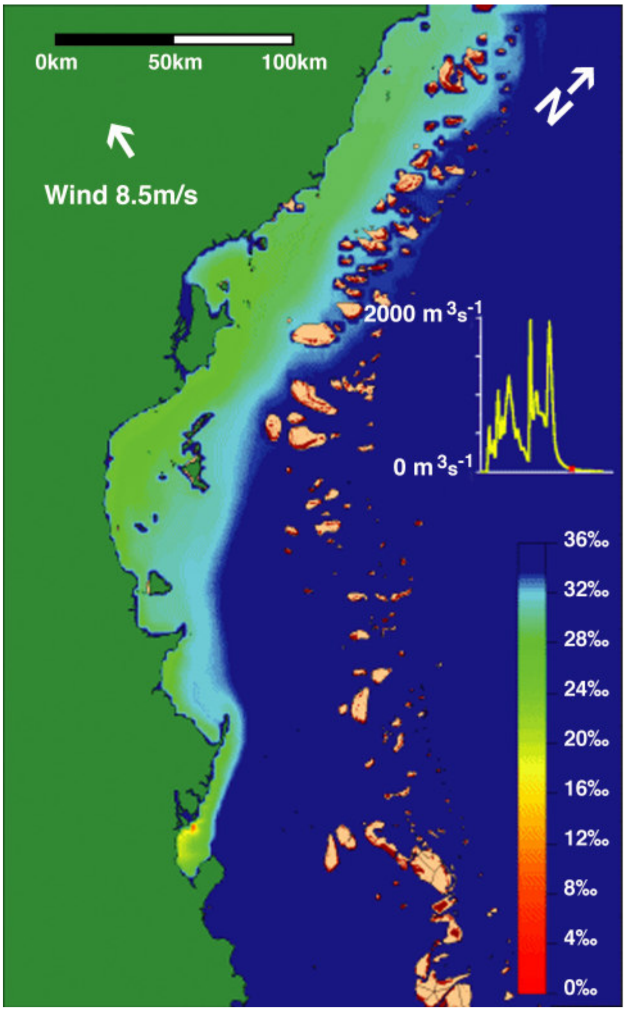


Figure 3.5 Modelled extent of the Burdekin River plume (January 1991) following Cyclone Joy. The hydrodynamic model used to simulate the plume was calibrated against field data (Wolanski and van Senden, 1983). From King et al., (2002).

3.2 Temporal changes of water quality in Great Barrier Reef coastal waters

The longest and most detailed time series of water quality data for the Great Barrier Reef lagoon, and most appropriate dataset to detect secular changes, was collected by AIMS in coastal waters off Douglas Shire (Cape Tribulation) and Cairns (Cape Grafton) from 1989 to the present (January 2005). Sampling sites are represented in Figure 3.6. Nutrients and other water quality parameters were sampled approximately twice-yearly (wet and dry season) at eleven sites along the coast and near Green Island. For this report, the dataset has been divided into three groups of sampling sites (stations). Four stations were located in coastal waters between Cape Tribulation and Port Douglas. Five stations were located in coastal waters off Cairns and one is in waters of a more oceanic character near Green Island.

Despite differing oceanographic settings in the three sampling zones (shallow coastal to open lagoonal) and the often visually different characteristics of the water (turbid brown inshore waters, clear blue mid-lagoon waters); salinities, temperatures (not shown) and concentrations of dissolved nutrients are very similar (Figures 3.7 and 3.8). This indicates that biological demand for nutrients by phytoplankton and bacteria is likely to be similar in all areas and generally exceeds inputs and recycling rates. As a result, concentrations of bio-available nitrogen (ammonium, nitrate, nitrite) and phosphorus (PO_4^{3-}) are all very low (generally < 1 µg N or P per litre).

Concentrations of suspended particulate matter, PN and PP are consistently higher in the shallow waters off Cairns where fine, readily resuspended sediments accumulate in Trinity Bay, located in the sheltered lee of Cape Grafton (Figure 3.8). Fewer fine sediments accumulate along the more exposed coastline bordering the Douglas Shire. Because of the similar, low nutrient levels in the three areas, phytoplankton biomass levels (as chlorophyll *a*) are also very similar (Figure 3.8).

Over the 15-years of monitoring in this area, gradual net increases in concentrations of suspended particulate matter, DON and DOP have occurred. These gradual changes are more easily observed when the data is transformed and replotted on a logarithmic (base 2) scale (Figure 3.9). While measured concentrations varied on a visit-by-visit and sometimes a yearly basis, there are statistically significant increases in all three variables (Appendix 3, Table 3, Figure 10). The DON and DOP concentrations increased linearly by approximately 50% and 400% respectively (Figure 3.9), however, using a generalised linear model with smooth trends it is evident that increases occurred during the later years of the surveys, after a relatively constant period (De'ath 2005). These increases are likely to reflect the biological conversion of readily bio-available inorganic nitrogen and phosphorus to less bio-available organic forms which can more readily accumulate in the water column. Suspended solids increased linearly by 100%. Reasons for these increases are not resolved at this time; they may reflect either increased resuspension activity (e.g. weather-related) or an increase in inputs of fine suspended or suspendable matter to the region.

Systematic change is evident for concentrations of other water quality parameters (nitrate, nitrite, PN, DIP, chlorophyll *a*; Appendix 3, Table 3, Figure 10). However changes were extremely small and their ecological relevance is unclear, or expressed as non-linear fluctuations of several years duration (see Figures 3.7 and 3.8), the reason for which is currently unknown. Precision estimates from this dataset indicate that an increase of 140% (median value for all measured parameters) can be detected, assuming linear change (De'ath, 2005).

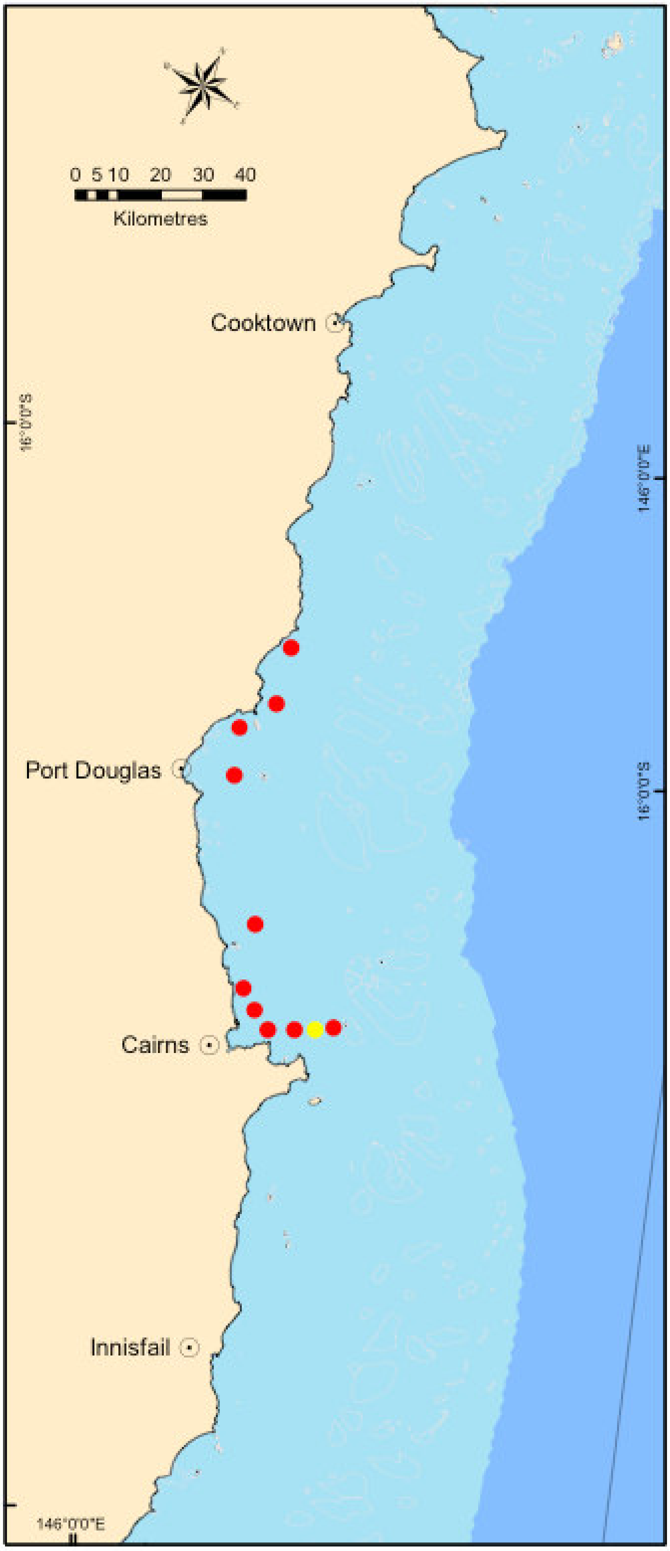


Figure 3.6 Locations of coastal stations in the Cairns region that have been repeatedly sampled by AIMS since 1989. The station marked by a yellow dot is not included in the analysis below as its characteristics vary between those of coastal stations and those near Green Island.

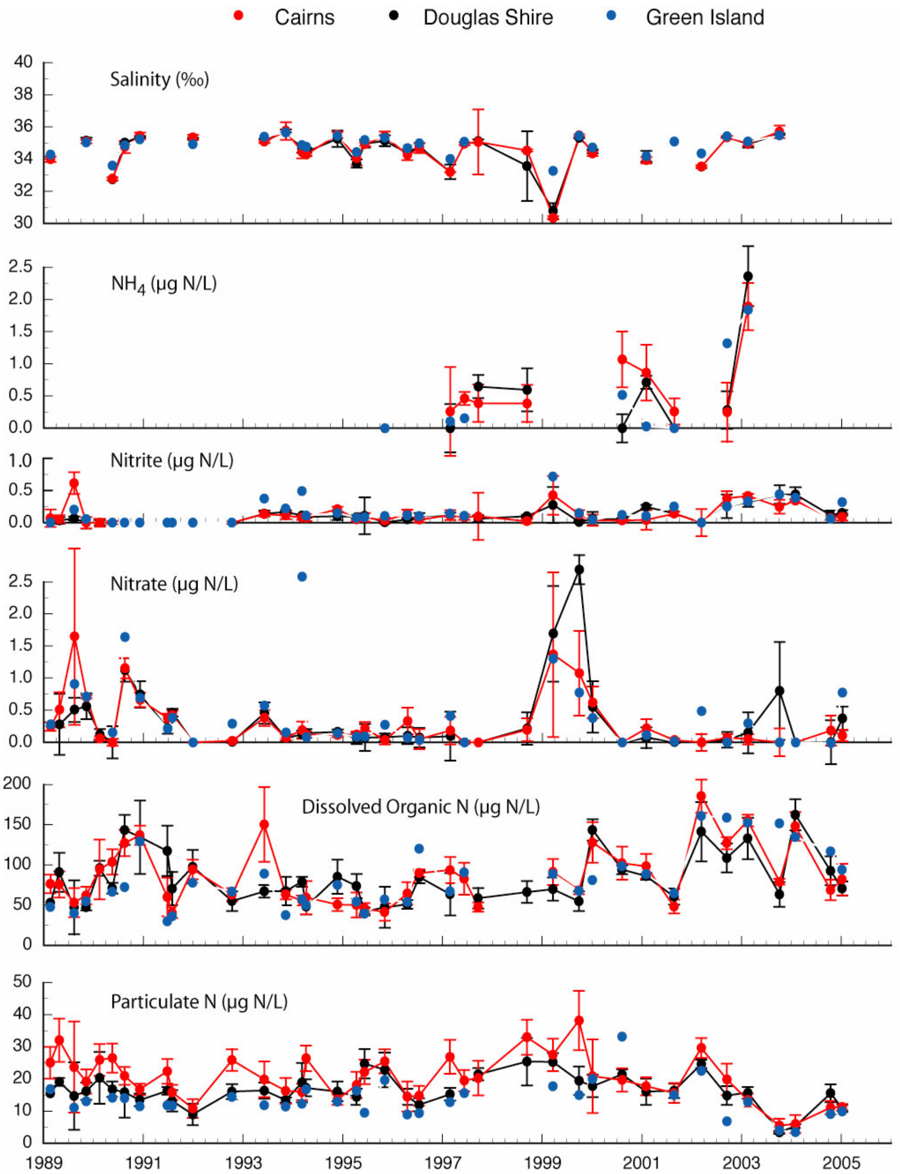


Figure 3.7 Median values of depth-weighted average water column salinities and concentrations of fixed nitrogen species in coastal and lagoonal waters off Douglas Shire (4 sites), Cairns (5 sites) and Green Island (1 site) between April 1989 and January 2005. Error bars for the pooled stations off Douglas Shire and Cairns show the 95% confidence range for these sites.

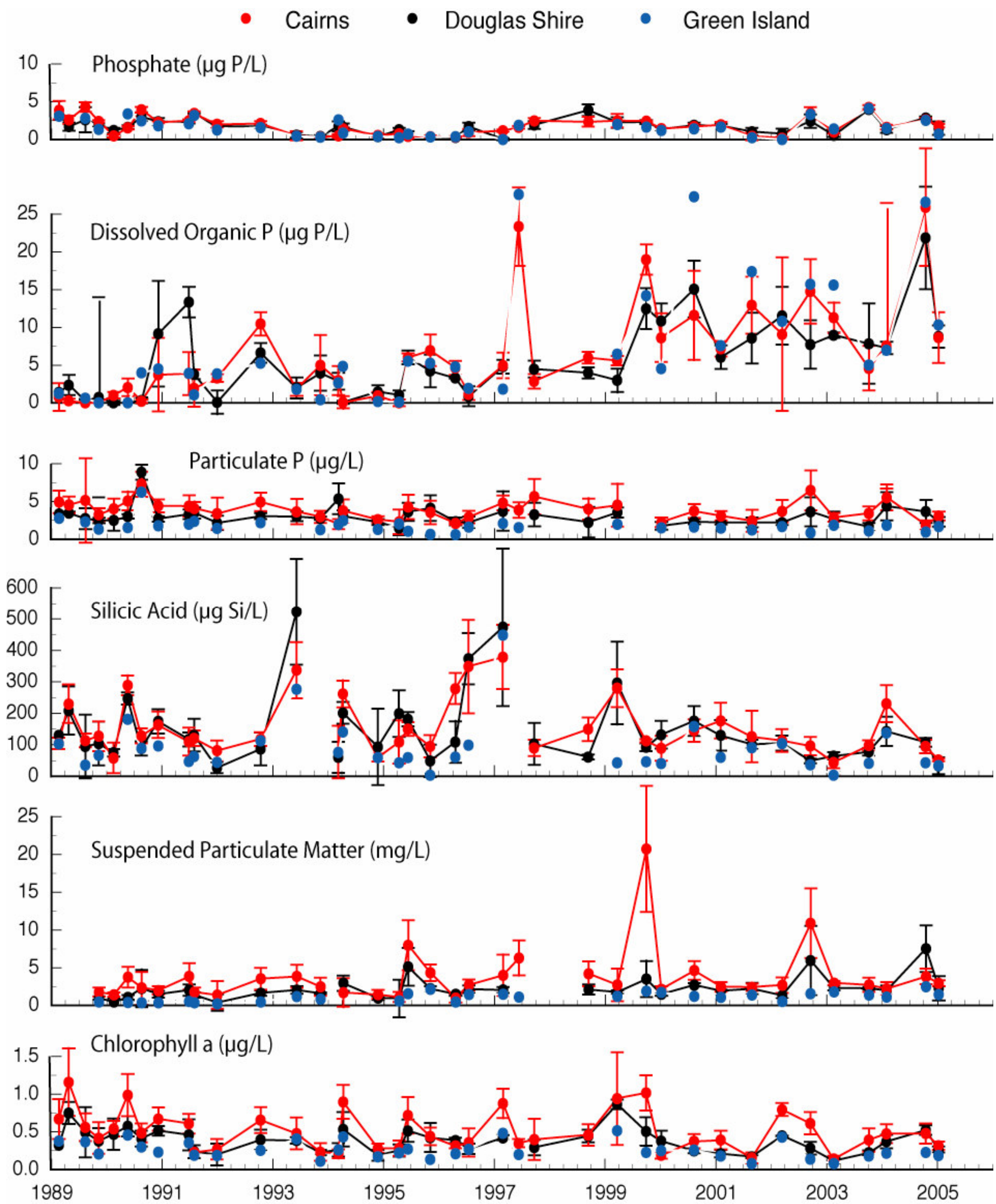


Figure 3.8 Median values of depth-weighted average water column concentrations of phosphorus species, silicic acid, suspended particulate matter and chlorophyll a *in coastal and lagoonal waters off Douglas Shire (4 sites), Cairns (5 sites) and Green Island (1 site) between April 1989 and January 2005. Error bars for the pooled stations off Douglas Shire and Cairns show the 95% confidence range for these sites.*

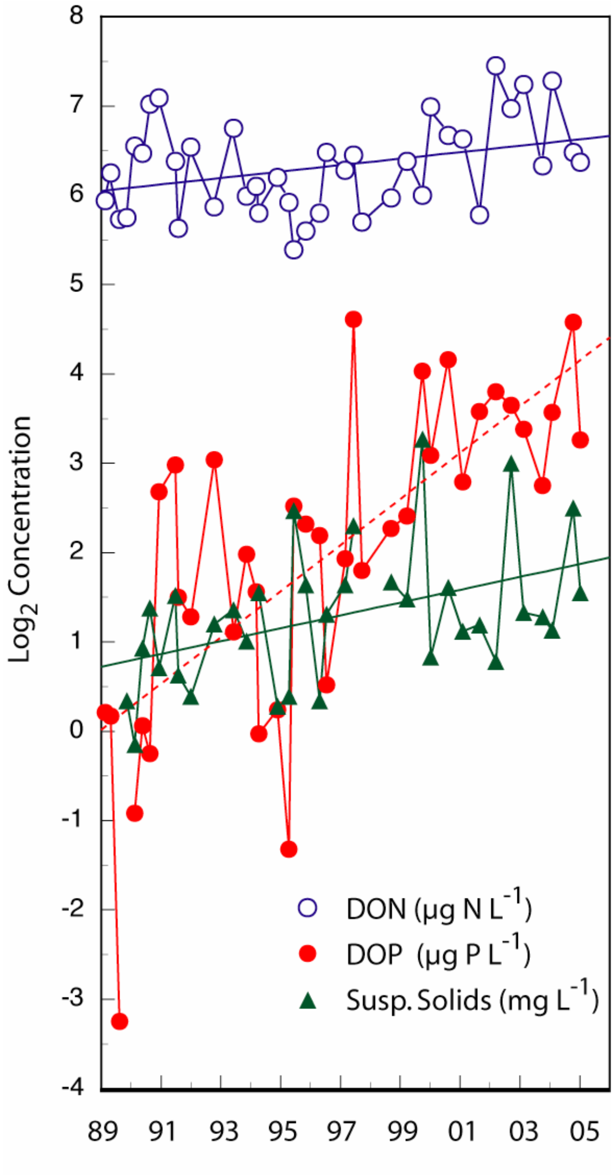


Figure 3.9 Decadal changes in median concentrations of dissolved organic nitrogen (DON), dissolved organic phosphorus (DOP) and suspended particulate matter in coastal and lagoon waters off Cairns and Douglas Shire. The data are plotted on a base-2 log scale where the interval between scale values represents a doubling or halving of concentration. Each data point is the median of all depth-weighted average water column concentrations (4 to 11 sites) measured on a particular sampling trip (n = 39).

3.3 Regional and temporal variations in surface chlorophyll *a*

Bio-available forms of nutrients are rapidly taken up by phytoplankton and bacteria in Great Barrier Reef waters (Furnas *et al*., 2005), therefore, dissolved inorganic nutrients are not the most sensitive indicator of nutrient availability. In addition, the low concentrations of nutrients which prevail are difficult to sample and measure without careful handling and highly sensitive analytical methods. Because the nutrients are rapidly assimilated into phytoplankton biomass and closely recycled, plant pigments such as chlorophyll *a* are therefore more useful proxy indicators of the quantity of nutrients which are circulating within an ecosystem.

The GBRMPA and AIMS have maintained a surface chlorophyll *a* monitoring program throughout the Great Barrier Reef since late 1992 (Brodie *et al*., 2005). Within particular regions, samples are collected contemporaneously at a number (2-9) of inner- and mid- to outer-shelf sites, generally on a monthly basis, but at greater intervals in some cases. Sampling sites are represented in Figure 3.10, and a short method summary is in Appendix 2. This dataset allows for the detection of both regional differences in nutrient availability (manifested as phytoplankton biomass) and temporal trends, and extends water quality data sets the discussed in Section 3.1.

Several features are evident in this dataset (Figure 3.11), which has been assessed by Brodie *et al*., 2005 and statistically by De'ath (2005). There is a general southward increase in mean chlorophyll *a* concentration (Figure 3.11; Appendix 3 Figure 11). There are no significant cross-shelf gradients in chlorophyll *a* concentrations in the three northernmost sectors (Far Northern, Lizard Island, Cooktown) (Appendix 3 Figure 12). The lack of a cross-shelf gradient in these sectors indicates the smaller magnitude of terrestrial nutrient inputs to this sector and perhaps, a greater degree of cross-shelf mixing. Significant differences between chlorophyll *a* levels near the coast and in the lagoon are evident in the sectors from Port Douglas southward (Figure 3.11; Brodie *et al*., 2005, Appendix 3 Figure 12). Inshore chlorophyll *a* levels were 1.4 to 3.4 times higher than offshore levels, depending on the region (De'ath 2005). The higher chlorophyll *a* concentrations at the inshore sites are likely to be due to enhanced nutrient availability from terrestrial runoff and recurrent resuspension of shallow inshore sediments.

Where there has been sufficient sampling to quantify levels of temporal variability, mean chlorophyll *a* concentrations exhibit significant seasonal fluctuations (Appendix 3 Figure 13) and fluctuations over longer periods of one to several years (Appendix 3 Figure 14). Examples include the generally higher chlorophyll *a* levels recorded in the Cairns, Townsville and Keppel Bay-Capricorn Bunker sectors between 1998 and 2000, and the low chlorophyll *a* concentrations recorded in most sectors during 2002. The causes of these longer fluctuations are still unresolved.

Higher levels of variability in data from the southernmost sectors also reflect the more frequent appearance of *Trichodesmium* in samples. *Trichodesmium* is a common, chlorophyll *a*-containing pelagic cyanobacterium that forms very large (0.5-1.0 µm) colonies that usually drift at or near the surface.

Overall, no long-term net (constantly increasing or decreasing) changes in mean chlorophyll *a* concentration have been observed in any sector or cross-shelf band. The temporal pattern of long-period temporal fluctuations is only apparent in data sets of 5 to 10 years, or longer duration (Appendix 3 Figure 14). De'ath (2005) concludes that the long-term chlorophyll *a* dataset provides a sufficiently precise estimate of long-term change of this parameter; an increase of approximately 32% can be detected, assuming linear change, over the current monitoring period.

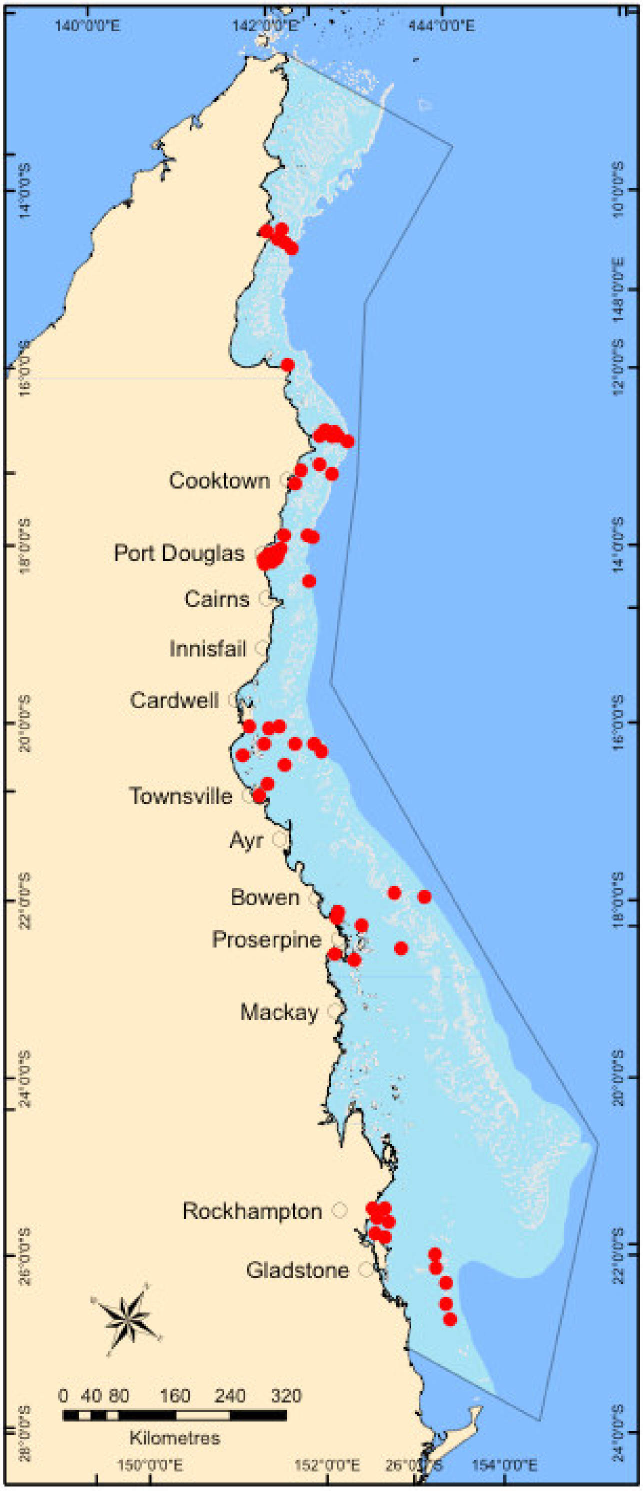


Figure 3.10 Locations of all stations sampled between 1992 and the present as part of the Long-term Chlorophyll Monitoring Program.

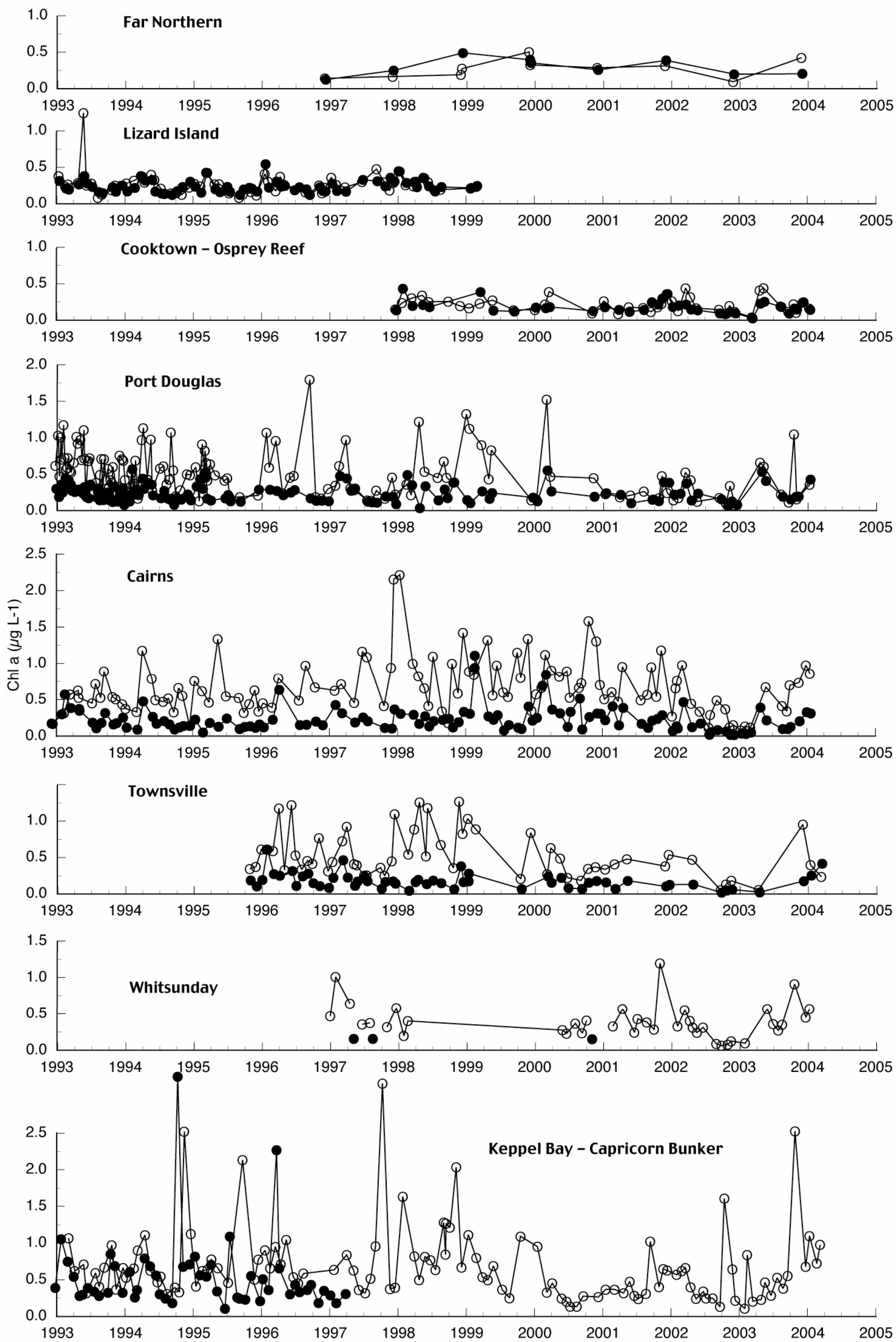


Figure 3.11 Time series of surface chlorophyll a *concentrations in eight latitudinal sectors. Results are divided into groups of stations located within 25 km of the coast (inshore - white symbols) and stations greater than 25 km from the coast (offshore - black symbols). Plotted values are mean concentrations for between 2 and 9 sampling stations within each region and cross-shelf combination.*

3.4 Water quality and organic matter in Great Barrier Reef waters

Particulate organic matter is an important contributor to the overall nutrient availability, especially in coastal regions. Compared to the other nutrients (nitrogen, phosphorus, silica), relatively little is known regarding concentrations and the nature of organic matter in Great Barrier Reef waters. Daily organic production in Great Barrier Reef lagoon waters is on the order of 0.5 - 1.0 tonnes of carbon (C) per km^2 (Furnas *et al*., 1989). Most of the organic matter suspended in Great Barrier Reef lagoon waters, however, is dissolved organic matter and non-living particulate organic matter (detritus), which is largely resuspended from the bottom and is recycled on a daily basis (Furnas *et al*., 1995). Marine plankton organisms exude dissolved mucopolysaccharides that may become particulate; such particles are known as transparent exopolymer particles (TEP) (Fabricius *et al*. ,2003). TEP forms sticky aggregates with other suspended particulate matter and is often referred to as marine snow (ibid.). During calm periods these aggregates are deposited on benthic organisms and the surrounding sea floor (Wolanski *et al*., 1998).

Regional surveys of large organic particles exhibit higher concentrations near the coast (Figure 3.12). This indicates that coastal reefs and ecosystems are exposed to higher levels of particulate organic matter than those offshore. Nutrient input processes that increase organic production in coastal waters will therefore also increase levels of organic matter affecting coastal reef systems.

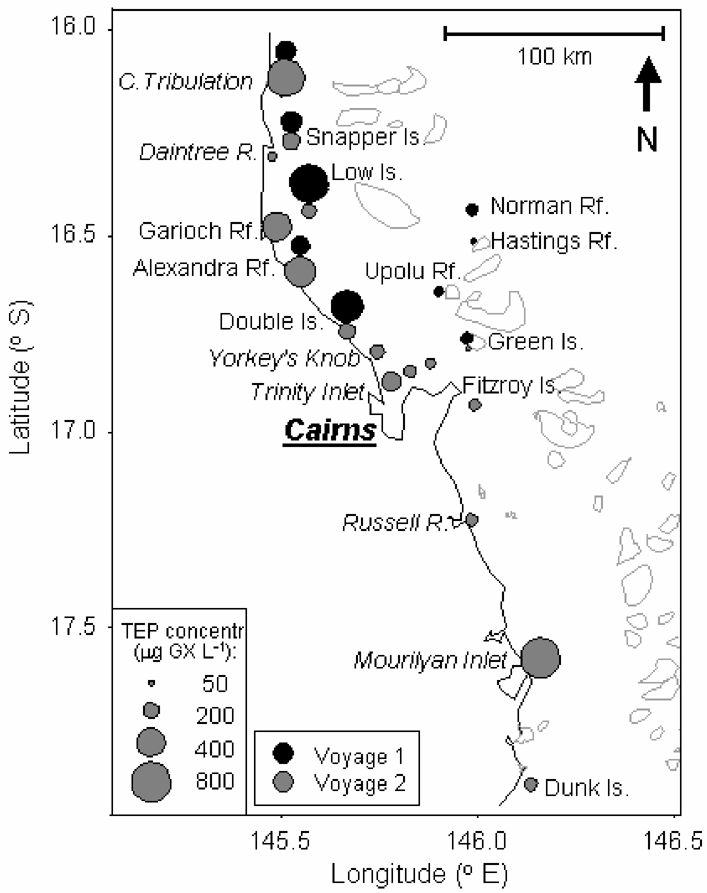


Figure 3.12 Cross-shelf and temporal variations in the concentration of large organic particles (TEP = transparent exopolymer particles) in Great Barrier Reef lagoon waters off the Wet Tropics. From Fabricius et al., (2003).

Dissolved inorganic nutrients can affect coral health by reducing calcification and fertilisation rates. In marine waters, dissolved inorganic nutrients are rapidly taken up by plankton and are usually only present in low concentrations (Szmant, 2002; Fabricius, 2005). However, increased nutrient availability leads to increased and often rapid, production of organic matter and changes of the biological communities which use or are otherwise affected by this organic matter. Organic matter is an important food resource for corals and other reef animals (Anthony, 2000; Anthony and Fabricius, 2000) and a nutrient source for some macroalgae (Schaffelke, 1999). Higher availability of organic matter enhances feeding rates and growth in some corals, especially under high light conditions, at very high levels of organic matter feeding of corals may be saturated and corals are affected by light reduction due to the high concentration of organic particles (reviewed in Fabricius, 2005). However, while some corals benefit from organic matter, heterotrophic filter feeders and possibly some macroalgae will benefit more than corals. This may lead to a shift in competitive advantage from corals that can grow at extremely low food concentrations to simpler, more heterotrophic communities. In combination with fine terrigenous sediment, organic matter can smother small corals and other reef organisms (Fabricius and Wolanski, 2000; Fabricius *et al*., 2003).

3.5 Pesticides and Herbicides in Great Barrier Reef waters and sediments

Herbicide and pesticide use is now an integral part of modern agricultural and land use practices. Unlike nutrients, which occur and vary naturally, their presence can be regarded as direct evidence for pollution, and have been detected in the Great Barrier Reef environment since the 1970s (Olafson, 1978). The largest amounts of herbicides are used by the sugar industry (e.g. 1996 estimates were 107 tonnes atrazine, 32 tonnes of diuron, 2 tonnes simazine per annum; Hamilton and Haydon, 1996); substantial amounts are also used in cultivating cotton (Fitzroy River catchment) and horticultural crops. Inevitably, some of these chemicals will be transported to the marine environment by runoff (see also Section 2).

Some studies have employed the analysis of biota or sediment as potential integrators or accumulators of pollutants in the system (von Westernhagen and Klumpp, 1995; Russell and Hales, 1993; Smith *et al*., 1985; Haynes *et al*., 2000a; Müller *et al*., 2000; Bengtson-Nash *et al*., 2005). Many of these studies provide further evidence that land based pollutants are entering waters of the Great Barrier Reef but suggest that the levels of pollutants are low, particularly in the offshore environment. However, the potential for these pollutants to affect environmental health on the Great Barrier Reef has been recognised (e.g. Brodie *et al*., 2001; Haynes, 2001; Bengtson-Nash *et al*., 2005) and due to the sensitive nature and high conservation value of the Great Barrier Reef, there is concern for potential consequences of the introduction of even low levels of these pollutants (refer to Section 7).

To date, there are no long-term or wide-ranging surveys of pesticides and herbicides in Great Barrier Reef waters, sediment or biota. A number of small-scale and one-off surveys have been carried out (a recent example is given in Figure 3.13). In general, measured concentrations of both pesticides (e.g. organochlorines and their degradation products; Olafson, 1978) and herbicides (e.g. Haynes *et al*., 2000a; Shaw and Müller, in press) are very low both in reef waters and sediments. A number of herbicides (atrazine, diuron, simazine) have been measured (0-7 ng L^{-1} range) in waters near coastal reefs in the Wet Tropics (Shaw and Müller, in press). Sediment herbicide concentrations in inshore sediments were generally < 0.1 µg kg^{-1} dry sediment, and pesticide concentrations < 0.05 µg kg^{-1} (Haynes *et al*., 2000a). While these concentrations are not high, they occur in Great Barrier Reef lagoon waters that have been subject to substantial levels of dilution. Higher concentrations may occur closer to sources.

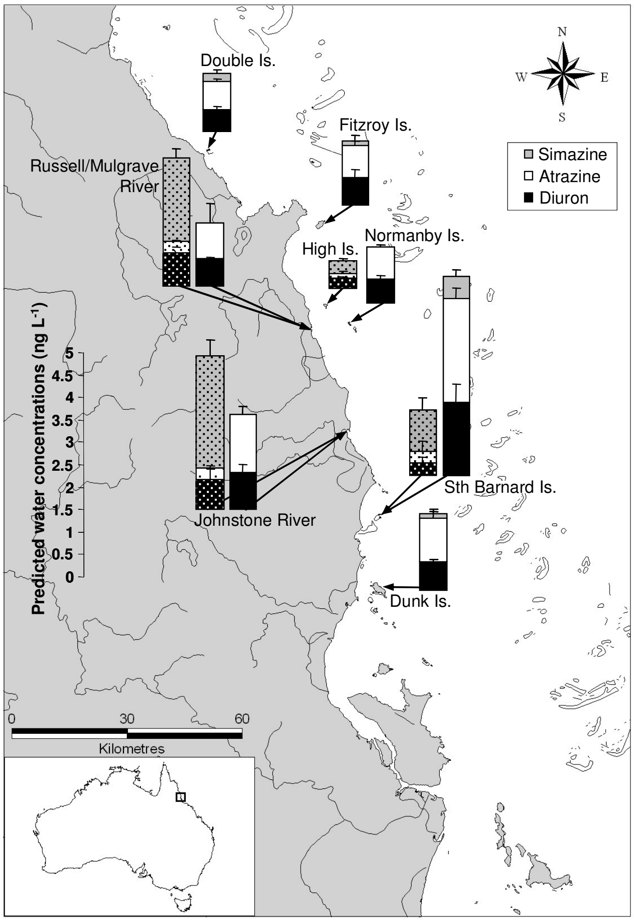


Figure 3.13 Measured concentrations of herbicides in two Wet Tropics rivers and coastal reef sites bordering the Wet Tropics. From Shaw and Müller, (in press).

The absence of any time-series data on pesticide or herbicide concentrations in coastal waters preclude conclusions as to whether the observed concentrations represent ephemeral events, or indicate persistent levels. There is very little information available on the sources and fates of pesticides and herbicides in Great Barrier Reef waters, or their effects on a wider range of marine organisms.

3.6 Ecological Implications

There is now abundant evidence, primarily from locations outside of Australia, that the overall health of coral reef and seagrass ecosystems are affected by the quality of water in which they live. Poor water quality is a driver in particular ecosystems or communities for ecological changes leading to the loss or displacement of dominant or desirable species, reductions in coral or seagrass cover, loss of ecosystem amenity value, and in extreme cases, the destruction of the physical structure of the ecosystem. The potential impacts of declining water quality on ecosystems in the Great Barrier Reef lagoon have been synthesised and reviewed in recent years (Hutchings and Haynes, 2000; Haynes, 2001; Williams, 2001; Baker, 2003; Furnas, 2003; see also Brodie *et al.*, 2004; Brodie *et al.*, 2005; Fabricius, 2005; Fabricius *et al.*, 2005; Schaffelke *et al.*, 2005).

Long-term effects of increased nutrients on some inner shelf coral reefs of the Great Barrier Reef are now evident. For example, in the Whitsundays, a nutrient/suspended sediment gradient from the Proserpine River has been correlated with reduction in coral cover, species richness and abundance combined with increased coral recruit mortality (van Woesik *et al*., 1999). Synergistic effects of nutrients and sediment (Fabricius and Wolanski, 2000) in association with the acute effects of cyclones, bleaching and COTS (Fabricius and De'ath, 2004, Fabricius *et al*., 2005) are the cause of the widespread reef degradation in inner shelf areas of the central Great Barrier Reef. At Green Island off Cairns the large expansion in the area of seagrass meadows on reefal areas normally without seagrass has been shown to be a result of increased nutrient supply from mainland river discharge (Udy *et al*., 1999). Knowledge of the transport of land-derived materials on the Great Barrier Reef shelf and hence the exposure of Great Barrier Reef ecosystems to this material allows us to better understand the changes which are occurring in these ecosystems. Further discussion of the potential impacts of changes in water quality on ecosystem health are discussed in Section 7.

Conclusions

Widespread and extensive water quality sampling has been carried out by AIMS throughout the Great Barrier Reef lagoon since the mid-1970's as part of monitoring and biological oceanographic studies within the Great Barrier Reef region. These programs have established the range of typical concentrations of nutrients, chlorophyll *a* or other water quality parameters and the occurrence of persistent regional and seasonal variations. These variations are determined by the large-scale structure of the Great Barrier Reef, regional-scale oceanographic processes and the seasonal nature of rainfall and runoff in northeast Australia. These variations must be considered when attempting to detect temporal trends in water quality at specific locations.

Both broad-scale and time series data sets of water quality parameters in the Great Barrier Reef lagoon indicate that nutrient, suspended particulate matter and chlorophyll *a* concentrations in Great Barrier Reef waters are generally low. This is in large part due to the low-nutrient levels of source waters for the Great Barrier Reef (the Coral Sea) and rapid biological uptake of nutrients by plankton communities in the Great Barrier Reef lagoon. However, the data show that there are persistent spatial and seasonal variations in average nutrient concentrations and other water quality parameters that reflect broad-scale and seasonal degrees of nutrient input to the Great Barrier Reef lagoon and regional-scale dispersion (mixing) rates. Efforts to measure the degree and direction of change in water quality within the Great Barrier Reef lagoon need to account for this inherent large-scale variability.

High concentrations of nutrients occur episodically in plumes of flooding rivers and over regional domains disturbed by the passage of tropical cyclones. Flood plumes are normally of short duration and may occur between one to several times per year on a regular basis in the Wet Tropics rivers, but less frequently (every several years to several decades) in the Dry Tropics rivers bordering the southern regions of the Great Barrier Reef. The high nutrient loads associated with flood plumes or cyclonic disturbances may lead to the formation of short-lived plankton blooms which convert nutrients in runoff to organic matter. The contrasting behaviour of particulate nutrients and dissolved nutrients in the plume shows that nutrients discharged from rivers in dissolved form are transported great distances, and therefore have the ability to influence biological activity on much of the inner-shelf of the Great Barrier Reef. Nutrients discharged in a particulate form are trapped near the coast and probably do not have a major influence on, for example, most of the inner-shelf coral reefs. These results have important implications with respect to the degree of exposure of inner-shelf ecosystems to river sourced nutrients and suspended particulate matter. As different forms of nutrients are exported from different land uses on the catchment the results can also help decide on priorities for management to reduce export from specific land uses. In general it is very clear that the primary area where flood plumes are common is the inner shelf and that ecosystems in this area are at most risk from pollutants contained in river discharge. The spatial distribution of recurrent river plumes has been used, together with other criteria, for estimating risk to regional ecosystems from exposure to terrestrial runoff.

A gradual increase in suspended sediment, dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) concentrations has been observed over 15 years in coastal waters near Cairns. DON is the most abundant form of fixed nitrogen in Great Barrier Reef waters. Over the same time period, concentrations of dissolved inorganic and particulate nitrogen and phosphorus, and chlorophyll *a* (a measure of plankton biomass and proxy for nutrient availability) varied over a range of time scales, but showed no net change overall.

Regional-scale chlorophyll monitoring in coastal and lagoon waters from 1992 shows that average concentrations within cross-shelf transects increase from north to south. Persistent cross-shelf gradients in chlorophyll *a* concentration are found in the central and southern regions of the Great Barrier Reef, reflecting enhanced nutrient availability at the coast from terrestrial runoff and recurrent resuspension of inshore sediments. To date however, no significant net changes in chlorophyll concentration have been observed at the regional scale.

Large organic particles, one of the product's of nutrient inputs, also exhibit higher concentrations in inshore waters. There is experimental evidence that mixtures of organic matter and fine terrestrial sediments are detrimental to hard corals and small sessile reef organisms. Little is known about the distribution, abundance and composition of this material in Great Barrier Reef waters. It is likely that particulate organic matter in Great Barrier Reef waters is a major driver of water quality effects on reef and reef organisms.

Statistical analysis of existing water quality data sets indicate that appropriate monitoring over time-frames of at least 10 years are necessary to resolve significant net changes in the naturally variable coastal waters of the Great Barrier Reef. There are now large-scale data sets available (AIMS Regional Water Quality Sampling; GBRMPA Long-term Chlorophyll Monitoring), to define regional baseline levels of important water quality variables such as nutrients and chlorophyll *a*. Two time-series of sampling are currently operational (AIMS Cairns Coastal Transect [1989-present], GBRMPA Long-term Chlorophyll Monitoring [1992-present]) with the capacity to detect long-term changes in lagoonal water quality if they are continued. These programs need to be expanded to better cover coastal waters bordering the Wet Tropics and southern regions of the Great Barrier Reef, which is part of the current Reef Plan MMP.

At present, long-term data sets are not available for agricultural chemicals or other man-made pollutants in Great Barrier Reef lagoon waters. Low levels (to several nanograms per litre) of herbicides have recently been detected in the water column at six inshore reef sites bordering the Wet Tropics. While water column herbicide concentrations measured to date are below experimentally determined effect levels, their presence does indicate that man-made pollutants (and other materials) in runoff are reaching inshore ecosystems of the Great Barrier Reef. The herbicide diuron has been measured in coastal and intertidal sediments at a number of locations along the Great Barrier Reef coast. The highest levels in these samples (to 10 micrograms per kg of sediment) were measured at sites adjacent to rivers draining catchment with extensive sugarcane cultivation. There is no information about chronic effects of herbicide exposure on Great Barrier Reef ecosystems or effects of higher levels that are expected to occur during flood events.

4. Aspects of coral communities on inshore reefs of the Great Barrier Reef

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Introduction

The coral reefs of the GBRWHA have immense aesthetic and great economic importance, supporting valuable tourisms and fisheries industries. Coral reefs are affected by a multitude of disturbances, including tropical cyclones, coral predation (e.g. by COTS, mainly affecting mid-shelf reefs), disease and high water temperatures causing mortality through bleaching. At present, 20% of the world's coral reefs are considered to be functionally destroyed by various impacts and may not recover (Wilkinson, 2004). In comparison, the Great Barrier Reef is considered to be still relatively healthy over most of its area (ibid.; Pandolfi *et al*., 2003). Second only to global climate change (e.g. Hughes *et al*., 2003), impacts caused by altered terrestrial runoff have been identified as a major threat to coral reefs from human activities (Fabricius, 2005). On the Great Barrier Reef, the potential impacts of terrestrial runoff are most likely to affect inshore reefs, particularly in the Whitsundays and Wet Tropics coasts (Devlin *et al*., (2003), Figure 1.1).

Long-term effects of increasing nutrients on some inner shelf coral reefs of the Great Barrier Reef are now evident (Devlin and Brodie, 2005). In the Whitsundays, a nutrient/suspended sediment gradient from the Proserpine River has been correlated with reduction in coral cover, species richness and abundance combined with increased coral recruit mortality (van Woesik *et al*., 1999). Synergistic effects of nutrients and sediment (Fabricius and Wolanski, 2000) in association with the acute effects of cyclones, bleaching and COTS (Fabricius and De'ath, 2004, Fabricius *et al*., 2005) are the cause of the widespread reef degradation in inner shelf areas of the central Great Barrier Reef. A major body of recent coral reef research funded through the CRC Reef Research Centre (CRC Reef) has focused on the ways in which runoff can affect coral reefs, using field and laboratory experiments and surveys on gradients in water quality. This is discussed further in Section 7 of this report.

The Reef Plan states that "*Coral reefs at a number of inshore locations along the coast (i.e. up to 20 km from shore) have been disturbed and have remained in a disturbed state*". The Great Barrier Reef Protection Interdepartmental Committee Science Panel (2003, Section 5.2.1 c) noted that "*Some areas of the coastal Great Barrier Reef, most affected by river run-off, appear to be degraded and/or slow to recover from natural events, such as cyclones. In this regard, we note the experiences documented overseas that the first major signs (that is, hard proof of adverse impact) appear when the coral reef system fails to recover from other disturbance (including natural events such as cyclonic level events)*". Because of the scarcity of long-term and large-scale information about inshore ecosystem health, the Reef Plan prescribes a precautionary and cooperative approach with the goal to: "*Halting and reversing the decline in water quality entering the Reef within 10 years*". The Reef Plan MMP will include assessments of the numbers of coral recruits and coral colony size distributions, which will provide evidence of the extent of recovery or non-recovery on inshore reefs.

This section reports on the patterns of coral cover and coral colony size found in a recent (2003-2004) extensive survey of inshore reefs of the central and southern Great Barrier Reef. The two longest data sets concerning coral cover on inshore reefs of the Great Barrier Reef are summarised, broader changes in coral cover across the Great Barrier Reef over 11 years are discussed and the limited data on coral recruitment rates from the Great Barrier Reef is briefly reviewed.

4.1 Variability in benthic cover on inshore reefs of the Great Barrier Reef based on surveys in 2003-2004

As part of a project funded by the CRC Reef, sites on 27 reefs between Cape Tribulation in the north (Wet Tropics) and the Fitzroy River in the south (Fitzroy Basin) were surveyed in 2003-2004 to describe the coral communities in terms of coral cover, community composition and size-frequency of colonies. These data are the first integrated survey of inshore reefs along much of the Great Barrier Reef coast and provide a baseline for future surveys. While the survey reefs were chosen based on a plan for an integrated water quality monitoring program (Brodie, 2002), the surveys themselves were aimed at assessing the general status of inshore reefs as indicated by coral cover, community composition and colony size distributions. Concurrent environmental variables such as water quality parameters were not assessed. All survey reefs were in areas of the Great Barrier Reef that are considered to be at some risk (low, moderate and high) of exposure to terrestrial runoff (Devlin *et al*. 2003; see also Figure 1.1 and limitations of this risk assessment noted in Section 1). Nine survey reefs were in areas designated as being at high risk. A full report to the CRC Reef is in preparation and is expected to be available by late 2005.

Data and methods

The assessment of near-shore coral communities aimed to survey reefs over a large area of the Great Barrier Reef coast. The number of inshore reefs in each of the latitudinal sectors of the Great Barrier Reef is limited; the choice of survey locations considered past survey history and the extent of reef development while including sites representing a range of orientations and distances to shore. The list of locations (e.g. sets of samples from zones (fronts or backs) within reefs that were surveyed) is presented in Table 4.1 and shown in Figure 4.1.

Where reef development permitted, two replicate sites were surveyed at each location. Each site consists of five transects, each 20m long and separated by 5m, starting at a haphazardly chosen point on the reef edge. Where reef development permitted, sets of transects were laid at two depths, 2m and 5m below Lowest Astronomical Tide (LAT), though the reef gave way to sand at less than 5m below LAT in several locations.

Hard corals and coralline algae are the major structural organisms of coral reefs and in conditions where recruitment is not limiting, coral cover indicates time since the last major disturbance. Total coral cover has obvious limitations as an indicator of reef condition, since communities can have similar total cover of corals but quite a different range of species may be present. Total coral cover does not correspond well to gradients in water quality (Fabricius and De'ath, 2004). For this reason, in these surveys and in the LTMP, corals are identified to the finest taxonomic level possible, usually at least to genus level. Cover of benthic organisms was estimated from a series of five video point intercept transects (20m long with 5m between consecutive transects at each depth and location) that were haphazardly placed along the depth contours. Organisms beneath the points were identified to highest possible taxonomic resolution governed by image quality.

For a reef community to be self-sustaining, it must include juveniles and sub-adults of each species as well as adult colonies, indicating significant recruitment within the average life-spans of coral colonies. For this reason, surveys paid particular attention to size frequencies of coral colonies. All hard and soft coral colonies falling wholly or partially within a 34cm wide (data slate width) belt consisting of the first 10m section of each transect were identified to genus and then classified into one of six size-categories, based on their longest linear dimension. Data from the five 10m sections at each site and depth were combined. The coral size categories used were <5cm, 5cm to <10cm, 10cm to <20cm, 20cm to < 50cm, 50cm to < 100cm and $\geq$100cm. Recruit densities were adjusted for "Available substrate" that is, areas of transects that were already occupied by living hard or soft corals or covered by sand or silt were considered unsuitable for recruits and excluded from calculations.

Linear mixed-effects models were used to assess differences in coral cover, recruitment and demography between zones and depths among the sampled sectors of the Great Barrier Reef. Reef locations were classified based on multivariate similarity in coral community type using hierarchical cluster analysis.

Table 4.1. Summary of inshore reef sites surveyed in 2003-2004.

Sector	Regional NRM area	Catchment	Reef Name	zone	risk[1]
Cairns	Wet Tropics	Baron Daintree	Cape Tribulation Nth	front	3
			Cape Tribulation Sth	front	3
			Double Island	front	4
			Snapper Island	back	4
				front	4
			Wentworth Reef	front	4
Innisfail		Tully Johnstone Russell Mulgrave	Dunk Island	back	4
				front	4
			Frankland Island	back	4
			High Island	back	4
				front	4
			King Reef	front	4
Townsville	Burdekin	Burdekin	Geoffrey Bay	front	3
			Middle Reef	front	3
			Nelly Bay	front	3
			Pandora Reef	back	3
Whitsunday	Mackay / Whitsunday	Proserpine	Black Currant Island	front	3
			Calf Island	front	3
			Daydream Island	back	3
			Manta Ray Island	front	3
			Pine Island	back	3
			Shute and Tancred Islands	front	3
Pompey		Pioneer	Keswick Island A	front	2
			Keswick Island C	back	2
				front	2
			St. Bees Island	back	2
Capricorn Bunkers	Fitzroy	Fitzroy	Humpy and Halfway Islands	back	2
				front	2
			Middle Island	back	2
			Nth Keppel Island	front	2
			Peak Island	front	2
			Pelican Island	back	2

[1]Risk refers to risk of exposure to contaminated runoff as assessed by Devlin et al., (2003). Category 1 = minimal risk, category 4 = high risk. Risk levels were based on indices of river pollution, likelihood of exposure to flood plumes and distance from river mouth, see Devlin et al., (2003) for details

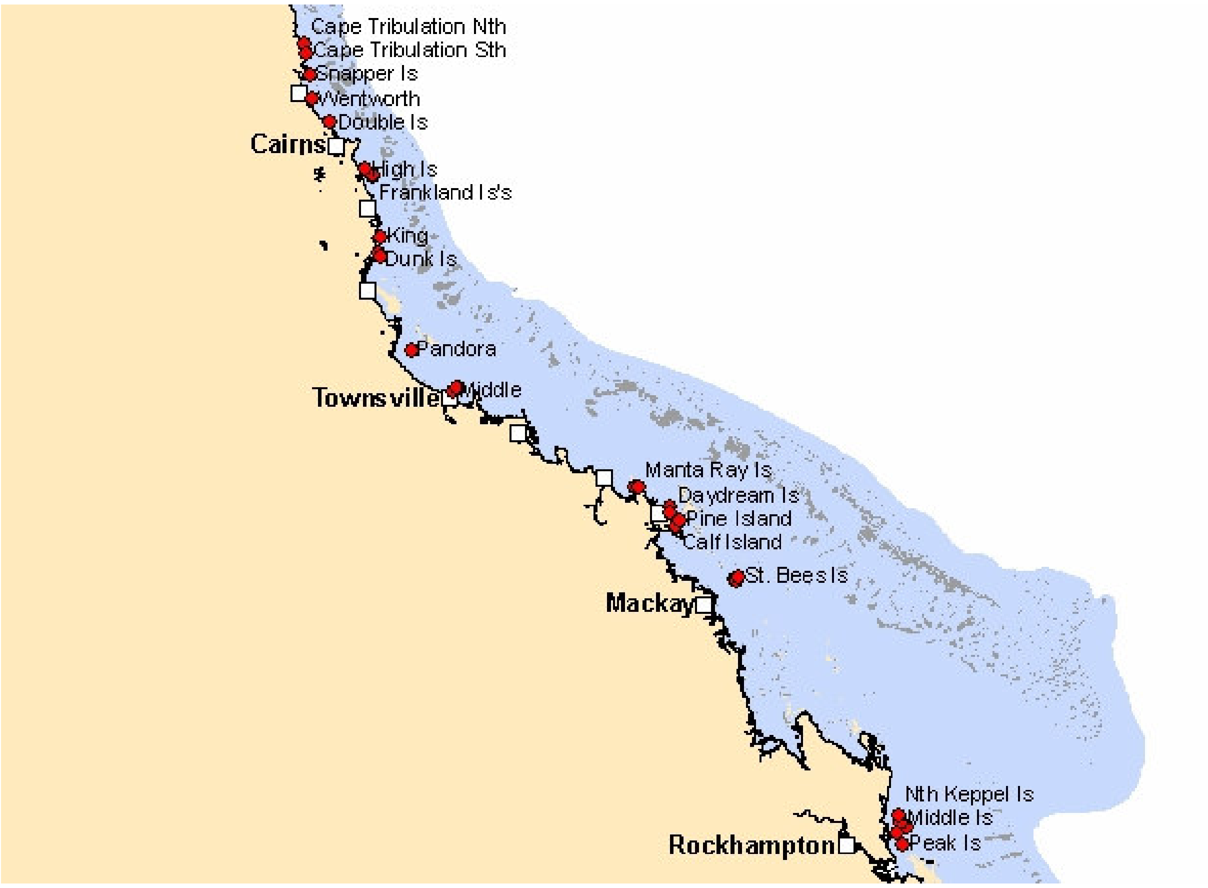


Figure 4.1 Map of sites surveyed in 2003-04. Note that not all sites are labelled but the full list is in Table 4.1.

Summary findings

Variation in hard coral cover on inshore reefs of the Great Barrier Reef

Average coral cover in 2003-2004 on the windward sides of inshore reefs (28%) was similar to values on north-east faces of reefs across the Great Barrier Reef (~30%, Sweatman *et al*., 2003). There were no consistent differences in average coral cover among sectors along the coast (Figure 4.2). Coral cover was highly variable among reefs within sectors. An extreme example occurred on fronts of reefs in the Capricorn-Bunker sector (Fitzroy Basin) where coral cover at Halfway and Humpy Islands averaged 72% compared with only 25% at Peak Island. Similarly, in the Cairns sector (Wet Tropics), cover was 52% at Double Island compared with 8% at Wentworth Reef. On average, coral cover did not change substantially with zone or depth (Figure 4.2). There was no clear correspondence to estimated risk of exposure to floods, which is estimated to be highest along the Wet Tropics coast and the Whitsundays mainland coast (Figure 1.1).

This high local variability in coral cover may reflect sharply uneven environmental conditions within sectors and/or localised disturbances. It implies that knowledge of disturbance history on a fine scale will be important for interpreting changes in reef communities. It is important to consider this in future decisions about the sampling frequency on inshore reefs under the Reef Plan MMP.

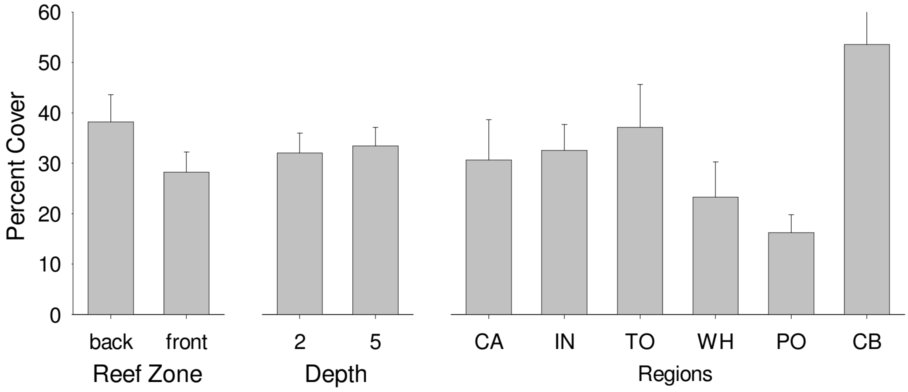


Figure 4.2 Hard coral cover in each reef zone, at each survey depth (2 and 5m) averaged over all surveyed reefs, and averaged by sectors. Cover was estimated from video transects. Error bars indicate standard errors.

Variation in taxonomic composition of coral communities on inshore reefs of the Great Barrier Reef

Reef locations were classified using hard coral demographic data (species/genus and colony size). The locations supported a wide range of community types. The coral communities lie on a continuum of combinations of genera rather than having distinct taxonomic compositions. However, a cluster analysis identified seven distinct community types (Figure 4.3) that were consistent with observations of community similarity from the field.

Three of these community types occurred in restricted geographic areas:

- Group 7, the most distinct group, included all sites at the more offshore reefs in the Keppel Islands (Halfway and Humpy Island, Middle Island, and North Keppel Island) (Fitzroy Basin). These sites generally had high coral cover and communities included a high proportion of branching *Acropora* spp.
- Group 5 included reefs that were dominated by the genus *Porites*. These were High Island and Frankland Islands in the Innisfail sector (Wet Tropics) and the shallow front reef of Snapper Island in the Cairns sector (Wet Tropics).
- Group 1 included communities with high representation of *Turbinaria*, *Montipora* and *Cyphastrea.* These were restricted to some front reef sites near Townsville (Burdekin) and a few sites from further north (Wet Tropics).

Each of the other community types identified by the cluster analysis was distributed across a number of sectors and occurred on both fronts and backs of reefs and in shallow and deep locations. Note that Group 6 includes sites that did not fall into other groups, i.e. there are no typical coral genera for this group (i.e. with high indicator value; Dufrêne and Legendre, 1997). Within Group 6 there was a split between the locations at Peak Island and Pelican Island (5m) (Fitzroy Basin), and the rest. The locations at Peak and Pelican Islands had higher cover (average 27%) and differed from all other locations because of the presence of large colonies of the genus *Psammocora*. The larger sub-group of 15 locations was made up of reefs with low coral cover (average 9%). There was no clear correspondence with estimated risk of exposure to flood plumes (Figure 1.1, Table 4.1). While Group 5 contained only survey locations from the Wet Tropics, an area considered at high risk of exposure to river pollution, some survey locations from the Wet Tropics fell into each of the other cluster groups except Group 7.

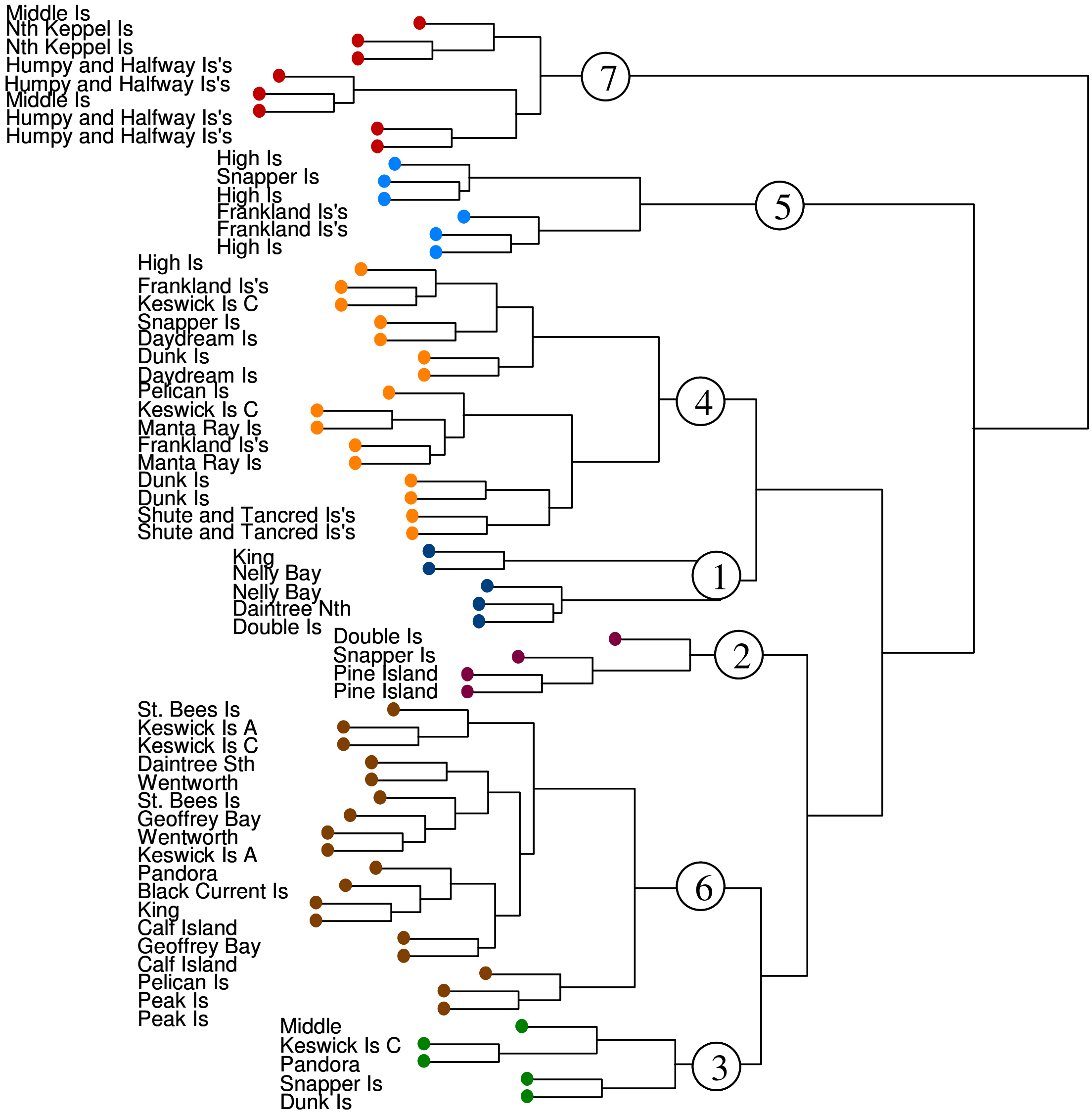


Figure 4.3 Cluster analysis of community types based on square root transformed cover estimates derived from demographic data. Demographic data were used because taxonomic resolution is more reliable than identifications from underwater video transects and estimates are not affected by the extensive macro algae on some reefs.

Densities of recruits on inshore reefs of the Great Barrier Reef

The significance of the presence of recruits was discussed in the introduction above. Coral colonies in the two smallest size-classes, <5cm and 5-10cm, were considered to be recruits. On front reef sites, both size classes of recruits occurred at higher densities at 5m depth than at 2m (Figure 4.4). Density of 5-10cm size class recruits also varied between sectors: reefs in the Pompey sector (Mackay-Whitsunday) and the Capricorn-Bunker sector (Fitzroy Basin) had low densities, especially compared with the high densities on reefs in the Innisfail sector (Wet Tropics) with an average of 7.1 colonies per square metre of available substrate (Figure 4.4).

The density of both small (<5cm) and large (5-10cm) recruits on back reefs showed no regional differences though there was considerable variation in density of recruits between reefs within sectors that would mask any differences between sectors. De'ath (2005) found the same lack of pattern in distribution of colonies in the smallest size class using these data, though the analysis was not corrected for available substrate.

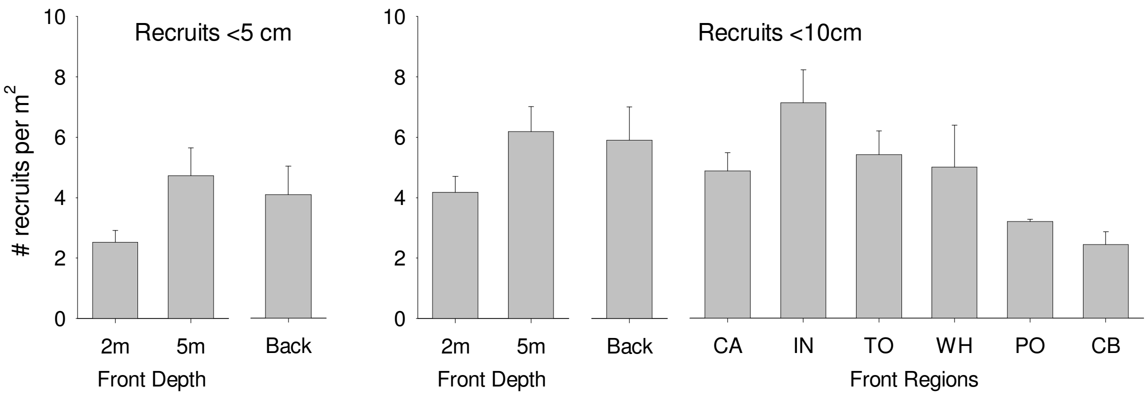


Figure 4.4 Spatial patterns in densities of recruits of hard corals on inshore reefs of the Great Barrier Reef. Estimates of recruitment are standardized to density of recruit-sized colonies (<5cm and 5-10cm diameter) per square metre of available substrate (see methods). Error bars indicate standard errors.

Density of recruits showed no clear correspondence with the estimated risk of exposure to runoff (Figure 1.1, Table 4.1). Very low density of recruits found on front reefs in the Capricorn-Bunker sector represent two quite different scenarios. Peak Island is close to the mouth of the Fitzroy River and is set in very turbid water; here high turbidity is a plausible limiting factor for coral recruitment. Other reefs in the sector are set in far clearer waters and have communities dominated by branching *Acropora* species and very high cover. In these sites recruitment may be limited by space or recruit density may have been underestimated because recruits rapidly grow out of the smaller size categories. Neither of these explanations applies to reefs in the Pompey sector (Mackay-Whitsunday) where space for recruitment was not limited, suggesting either limited larval supply or some undetermined environmental factor was controlling coral recruitment on these reefs.

Densities of recruits were consistently lower on shallow front reef sites compared with either adjacent deeper sites or back reef sites in general. The factors limiting recruitment in this zone need investigation.

Size distribution of hard coral colonies on inshore reefs of the Great Barrier Reef

Discontinuous size distributions of coral colonies at individual reefs can reflect past community dynamics (i.e. disturbance or differences in recruitment success). It is frequently suggested that low levels of stress from runoff, while not lethal to large, established colonies, may reduce recruitment of some species and so lead to a shift in community composition (Cortes and Risk, 1985; McCook *et al.*, 2001). If such stress were present at a reef, there would be substantial numbers of large colonies but few recruits of the same taxa.

Comparison of the number of large colonies of each genus with the number of recruits of that genus at each location showed little evidence for shifting community types through large adults not being replaced (Figure 4.5). Eighty-four percent of genera that were represented by large colonies at any location were also represented by recruit-sized individuals. This indicates that there is potential for the large colonies to be replaced in due course should they be killed. Reefs where recruitment of the genera represented by large individuals was limited, and thus chances of replacement of a similar community following a severe disturbance was reduced, included Peak and Pelican Island adjacent to the Fitzroy River mouth, Keswick Island back-reef in the channel between Keswick and St. Bees Is land (Mackay-Whitsunday), Middle Reef off Townsville (Burdekin) and coastal fringing reefs north of Cape Tribulation (Wet Tropics).

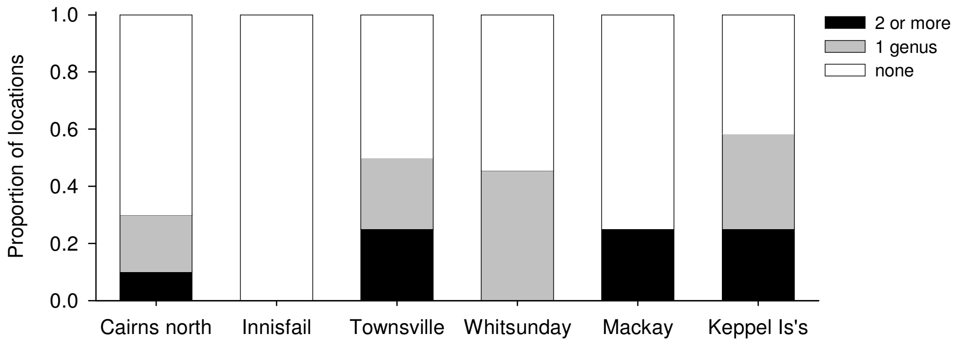
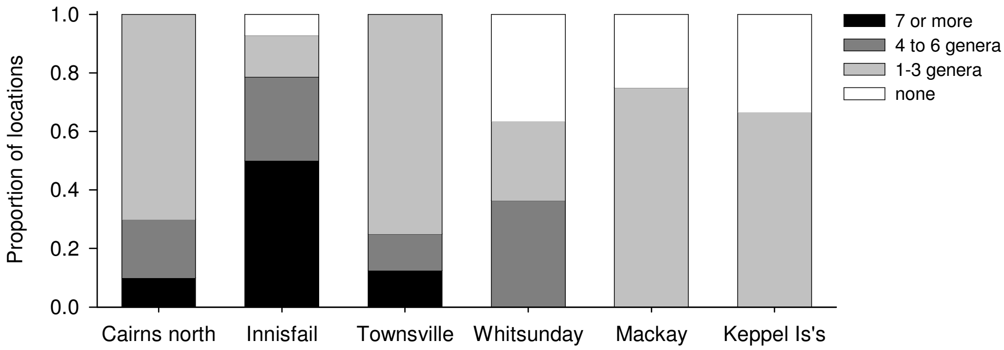


Figure 4.5 Proportion of locations (combination of reef, zone, depth) within each sector for which there was disparity between the number of large individuals and the number of recruits. For each location the number of recruits of each genus was compared with the number of large colonies. The number of genera that had at least 3 times the number of large individuals compared to recruits was determined. A genus was included at a location if at least three individuals of that genus were classified as 'large'. The large size class was based the size-distribution of individuals for a given genus recorded in all 63 locations surveyed in 2004.

The opposite situation also occurred: on some reefs there were coral communities with high numbers of recruits compared with numbers of large individuals of some genera. This probably represents recovery from a recent severe disturbance that reduced the number and diversity of large colonies. Comparing the number of genera with size distributions that were severely skewed toward recruit-sized individuals (Figure 4.6) shows that a high proportion of locations in the Innisfail sector (Wet Tropics) had many skewed populations. Inshore reefs in this region have a high risk of exposure to runoff (Figure 1.1). These reefs were affected by COTS and bleaching in the late 1990's (Ayling and Ayling, 2004). These recent surveys suggest that substantial recovery is now in progress.

Neither regional patterns in coral cover nor the numbers of recruits showed any clear relation to the estimated risk of exposure to runoff. Recruits were recorded in most sites suggesting that conditions on inshore reefs in most regions were not such as to cause recruitment failure and consequent shifts in the community taxonomic composition. However, there is no way to know if the observed recruitment rates are adequate to allow community persistence. Recruitment to the adult population depends on settlement and on subsequent survival, which also varies at a local scale (Smith *et al.*, 2005) for three reefs in the



Wet Tropics.

Figure 4.6 Proportion of locations within each sector for which specified number of genera showed distributions skewed toward recruit-sized individuals. A genus was included at a location if there were at least five recruit-sized individuals (<10cm diameter) and no individuals in larger size classes.

4.2 Trends in benthic cover on selected inshore reefs surveyed by the AIMS LTMP between 1992 and 2004

Data and statistical methods

Hard coral cover was estimated from video transects at 6-9m depth generally on north-east faces of three inshore reefs in each of four sectors of the Great Barrier Reef (Figure 4.8) over the period 1994-2004 (see details of AIMS LTMP survey methods in Sweatman *et al*., 2003). Data were averaged over transects within each of three sites per reef. Linear mixed-effects models were used to assess trends over time for the 12 individual inshore reefs in four sectors sampled by the LTMP. This analysis is basically the same as that presented in De'ath (2005), but includes recent data from surveys in 2004. Based on estimates of precision from the first 10 years of data, De'ath (2005) showed that considering the 10 year trend in percent hard coral cover at a reef, in the best case, the estimate of detectable difference over 10 years corresponded to a 29% increase or a 22% decrease in percent cover (i.e. given 20% cover initially, the detectable increase over 10 years would be 5.8% [to 25.8% cover] and the detectable decrease would be 4.4% [to 15.6% cover]). The worst-case estimate of detectable difference (i.e. not assuming linear change) over 10 years would correspond to a 43% increase or a 30% decrease in percent cover (i.e. given 20% cover, the detectable increase over 10 years would be 8.6% and the detectable decrease 6.0%).

Summary findings

Long-term trends on inshore reefs in the Cooktown-Lizard Island sector (Cape York) and the Whitsunday sector (Mackay-Whitsunday) indicate relatively stable or slightly increasing coral cover (Figure 4.7). Conversely, coral cover declined on most inshore reefs in the Cairns sector (Wet Tropics) and the Townsville sector (Burdekin) (Figure 4.7).

In the Cairns sector, AIMS LTMP recorded extensive bleaching and a few COTS at Fitzroy Island in 1998, outbreak levels of COTS at Low Isles in 1998-99. Coral cover at Green Island has not recovered from an outbreak of COTS in the early 1980s (Sweatman, 2003). Low numbers of COTS were seen consistently at Green Is between 1994 and 1999. No adult COTS have been seen at Green Island since 1999 but there has been little net increases in coral cover (Sweatman *et al*., 2003). Beside the COTS outbreak, Low Isles reef was also directly in the path of Cyclone Rona in 1997 (Cheal *et al*., 2002). In the Townsville sector, coral cover at Havannah Island declined steeply after extensive bleaching in 1998 and has not recovered substantially since. Substantial proportions of both hard and soft corals at Middle Reef bleached in early 1998, though there was little resultant mortality. Since 1999, coral cover has decreased through the loss of Poritidae and Acroporidae, but no simple cause is evident. Coral cover at Pandora Reef was very high (>55%)in the 1997, but in 1998 there was extensive bleaching and inundation by a flood plume following Cyclone Sid. Together these caused substantial decline in coral cover, mainly through loss of *Acropora* spp. (Sweatman *et al*., 2003). There have been no recent recruitment surveys of these reefs, so the extent of any recovery is unknown.

Inshore reefs in the Cooktown- Lizard Island sector (Cape York) did not suffer bleaching in 1998. No COTS or bleaching has been recorded at Decapolis Reef, COTS were present 1995 to 1999 at Linnett Reef and reached 'Incipient Outbreak' levels, but coral cover has generally increased to be more than 50% in 2003. COTS were present at Martin Reef between 1996 and

2001 and in outbreak densities 1998-1999, but this did not cause substantial loss of coral cover. Inshore reefs in the Whitsunday sector (Mackay-Whitsunday) did not suffer bleaching in 1998 though there was some bleaching in the sector in 2002 (Sweatman *et al.*, 2003). Only Langford Reef and Bird Island has had low densities of COTS in recent years.

These regional trends are generally compatible with the estimates of risk exposure to runoff produced by Devlin *et al.*, (2003) (see Figure 1.1): inshore reefs in the Cairns sector have the highest risk of exposure, whereas reefs in the other regions are at moderate risk. The difference between the Cairns and Townsville sectors, where coral cover on inshore reefs has declined, and the other sectors is that reefs in the Cairns and Townsville sectors suffered significant impact from bleaching in 1998 (Berkelmans and Oliver 1999). The potential effects of exposure to runoff are confounded with effects of coral bleaching.

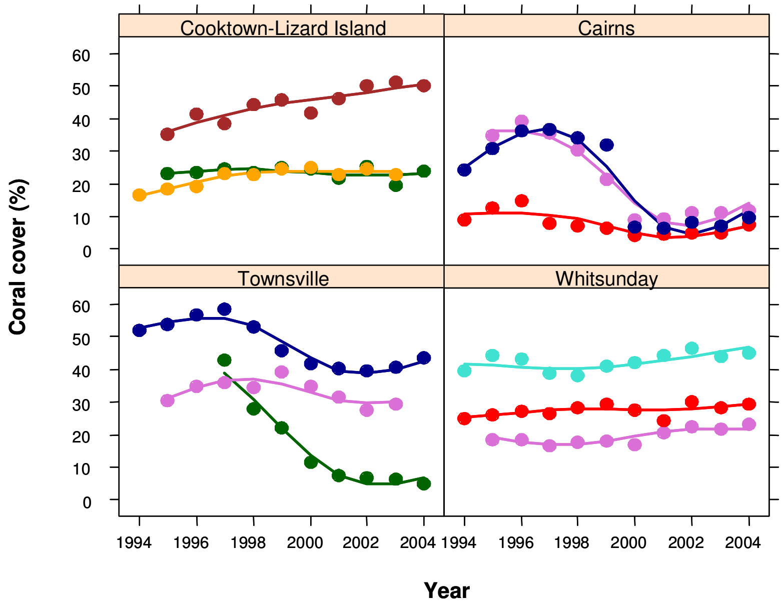


Figure 4.7 Estimated trends in hard coral cover from video transects for 12 inshore reefs within 4 sectors over 11 annual surveys between 1994 and 2004. Each line corresponds to one of the three inshore reefs within each sector. Cooktown – Lizard Island sector: brown = Linnett Reef, yellow = Decapolis Reef, Green = Martin Reef; Cairns sector: purple = Low Isles, blue = Fitzroy Is, red = Green Island; Townsville sector: blue = Pandora Reef, green = Havannah Island, purple = Middle Reef; Whitsunday sector: green = Hayman Island, purple = Langford and Bird Island, red = Border Island.

4.3 Trends in benthic cover on other inshore reefs of the Great Barrier Reef

There are a small number of longer term studies of inshore reefs. These studies highlight the dynamic nature of these reefs, showing large declines in coral cover between visits that coincide with specific disturbance events. The most comprehensive and longest running data sets are those collected by Sea Research from 1985 on coastal fringing reefs near Cape Tribulation and then expanded in 1994 to include reefs around Snapper Island. Reefs in the Frankland Islands were added in 1995 (Ayling and Ayling, 2004). Reefs at the Frankland Islands and Snapper Island are estimated to be at high risk of exposure to runoff; this risk is assessed as moderate at Cape Tribulation (Figure 1.1).

Cape Tribulation

There are data on coral cover and species-composition of the communities in 12 sites in 3 "locations" near Cape Tribulation that have been collected frequently since 1985. Reefs in all sites had more than 50% coral cover when surveys began in 1985, though a cyclone in 1986 caused a drop in cover and there was a bleaching event in 1987. By 1988 coral cover had recovered to 1985 levels through growth of the common *Acropora* spp. The reefs were not surveyed again until 1994 when coral cover was more than 50% in all sites. Coral cover has remained very high (>>50%) in two of the locations in spite of some bleaching in 1998 and 2002. Location 1 suffered a COTS outbreak in early 2000, which reduced coral cover by half. With further bleaching in 2002 the reefs in Location 1 have not recovered.

Snapper Island

The reefs at Snapper Island were first all surveyed in 1994 when coral cover was about 90%. A massive flood killed most of the coral on the south side to a depth of about 3m and coral cover dropped to about 10%. The effect on reefs on the north side was much less severe. In 1998 the northern reefs were affected by bleaching (many of the more susceptible corals on the southern reefs had been killed in the flood, so the effect was less intense on the south side). In 1999 all sites were damaged by waves from Cyclone Rona. Since then coral cover has increased on both sides of the island to be about 30% on southern reefs and about 40% on reefs on the north side.

The Frankland Islands

The first surveys of the reefs of the Frankland Islands in 1995 recorded an average coral cover of nearly 80%, though the reefs on the two sides had different coral communities. More than half the coral was lost as a result of bleaching in 1998. In 1999, there were some COTS in some eastern sites. Over 3 years, average coral cover dropped from more than 60% to less than 20% because of these disturbances. Since 2000 the coral cover has been increasing slowly in both the eastern (currently ~30%) and western sites (currently ~40%).

These reefs are considered to be at risk from exposure to runoff. However, they have had very high coral cover (and considerable diversity of corals Ayling and Ayling, 2004) in the past 20 years, well after most of the changes in land use in adjacent catchments had occurred. A series of identifiable disturbances has reduced the coral cover in the late 1990s, but there are clear signs of recovery in the great majority of sites.

4.4 Comparison of trends in benthic cover on inshore reefs with those seen in the other regions of the Great Barrier Reef in AIMS' LTMP surveys between 1992 and 2004

Data and statistical methods

Hard coral cover was estimated from video transects at 6-9m depth generally on north-east faces of three reefs in each of the inshore, mid-shelf and outer-shelf regions in 4 sectors of the Great Barrier Reef over the period 1994-2004 (Figure 4.8, see details of LTMP survey methods in Sweatman *et al.*, 2003). To estimate temporal trends in each region (i.e. at each shelf position in each sector) data averaged over all sites on each reef were used. Linear mixed effects models were used to estimate smooth trends over time and differences in these trends between sectors.

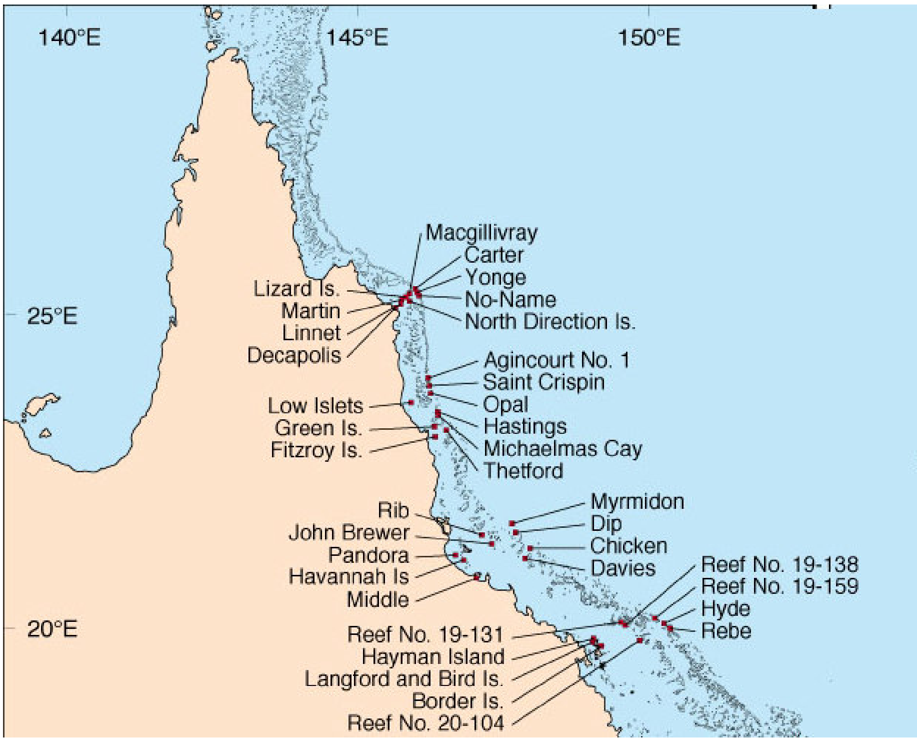


Figure 4.8 Survey reefs in (from North to South) the Cooktown-Lizard Island, Cairns, Townsville and Whitsunday sectors of the AIMS LTMP. Three inshore reefs, 3 mid-shelf reefs and 3 outer shelf reefs are surveyed in each sector.

Summary findings

A comparison of temporal trends in coral cover among regions shows strong regionalisation of impacts and disturbance history (Figure 4.9). This analysis combines the trajectories for coral cover on individual reefs (such as those for inshore reefs which were presented in the previous section) into regional means. Coral cover on inshore reefs declined in the Cairns sector (Wet Tropics) and the Townsville sector (Burdekin) by 15-20% over 10 years, but there was no evidence of net change in the Cooktown-Lizard Island sector (Cape York) or Whitsunday sector. As described above, declines in inshore reefs in the Cairns and Townsville sectors can be linked tentatively to temperature anomalies in these regions in 1998 leading to bleaching (Berkelmans and Oliver, 1999). Temperature anomalies were less extreme in other regions.

Mid-shelf reefs showed significant declines in the Cairns sector (~5%), Townsville sector (~30%) and Whitsunday sector (~15%). There has been subsequent increase (~10%) in the Whitsunday sector. There was no evidence of change in the Cooktown-Lizard Island sector. Declines in coral cover on mid-shelf reefs in the Wet Tropics and Dry Tropics regions are largely due to COTS: all three mid-shelf reefs in the Cairns sector- and Rib Reef and John Brewer Reef in the Townsville region have had 'Active' or 'Incipient' outbreaks of COTS within the past 5 years (Miller, 2002; AIMS LTMP website). The third mid-shelf survey reef in the Townsville sector, Davies Reef, had outbreaks until 1992 and had close to incipient levels again in 2004. The temporary decline in the Whitsunday mid-shelf reefs is probably the result of swells generated by Cyclone Justin in 1997 (Sweatman *et al.*, 2003).

Outer-shelf reefs showed a significant initial increase in coral cover Cooktown-Lizard Island sector (~40%) from a low level measured at the beginning of monitoring in 1994. There was a significant decline in coral cover in the Townsville sector by ~8%. There was no evidence of change on mid-shelf reefs in the Cairns and Whitsunday sectors. Recovery on the outer-shelf reefs in the Cooktown-Lizard Island sector occurred after unknown disturbances and the impact of Cyclone Ivor. Recent declines on outer-shelf reefs in the Townsville sector are due to bleaching in 2002 (particularly at Myrmidon Reef) while Chicken Reef has had an 'Active' COTS outbreak since 2003 COTS (AIMS Monitoring website).

In terms of risk of impacts from runoff, particularly from flood plumes, inshore reefs all along the Great Barrier Reef coast are estimated to be at moderate or high risk (Figure 1.1). Generally, mid-shelf and outer shelf reefs are not threatened by runoff (Williams *et al*., 2002). An exception is the mid-shelf region in the Cairns sector; Devlin *et al*., (2003) suggest that the northwards wind-driven flow of flood plumes from Wet Tropics rivers is deflected out across the shelf by Cape Grafton. This means that mid-shelf and even outer shelf reefs may be exposed to flood plumes relatively frequently. Whether these floods affect coral recruitment and early survival is unknown. Brodie *et al*., (2005) suggest that plumes enrich the lagoon waters. If this coincides with COTS spawning, it could lead to enhanced survival of larvae and generate primary outbreaks of COTS. This suggestion must be treated with caution, first because it is based on observations of enhanced larval survival under highly artificial conditions, and second because the evidence that primary COTS outbreaks have occurred in the Cairns sector is very weak.

The taxonomic composition of coral communities varies substantially across the continental shelf (Done, 1982) and this affects the susceptibility to impacts and the rates of recovery. For instance, the rapid increase in coral cover on outer-shelf reefs in the Cooktown-Lizard Island sector was possible because of the dominance of fast growing tabulate *Acropora* spp. on these reefs. The causes of disturbance also vary in their importance across the continental shelf: crown-of-thorns starfish outbreaks are most common on mid-shelf reefs while inshore reefs are exposed to flood plumes much more regularly (Devlin *et al*., 2003). Coral cover on outer shelf reefs in the Cooktown – Lizard Is sector was damaged by Cyclone Ivor early in 1990. Even communities of the fastest-growing corals in areas remote from human influence took 8 years to recover (Figure 4.9). Estimates of realistic recovery rates for inshore reef communities that include slow-growing species and may show erratic recruitment rates (Smith *et al*., 2005) will be important for the assessing the resilience of inshore reefs.

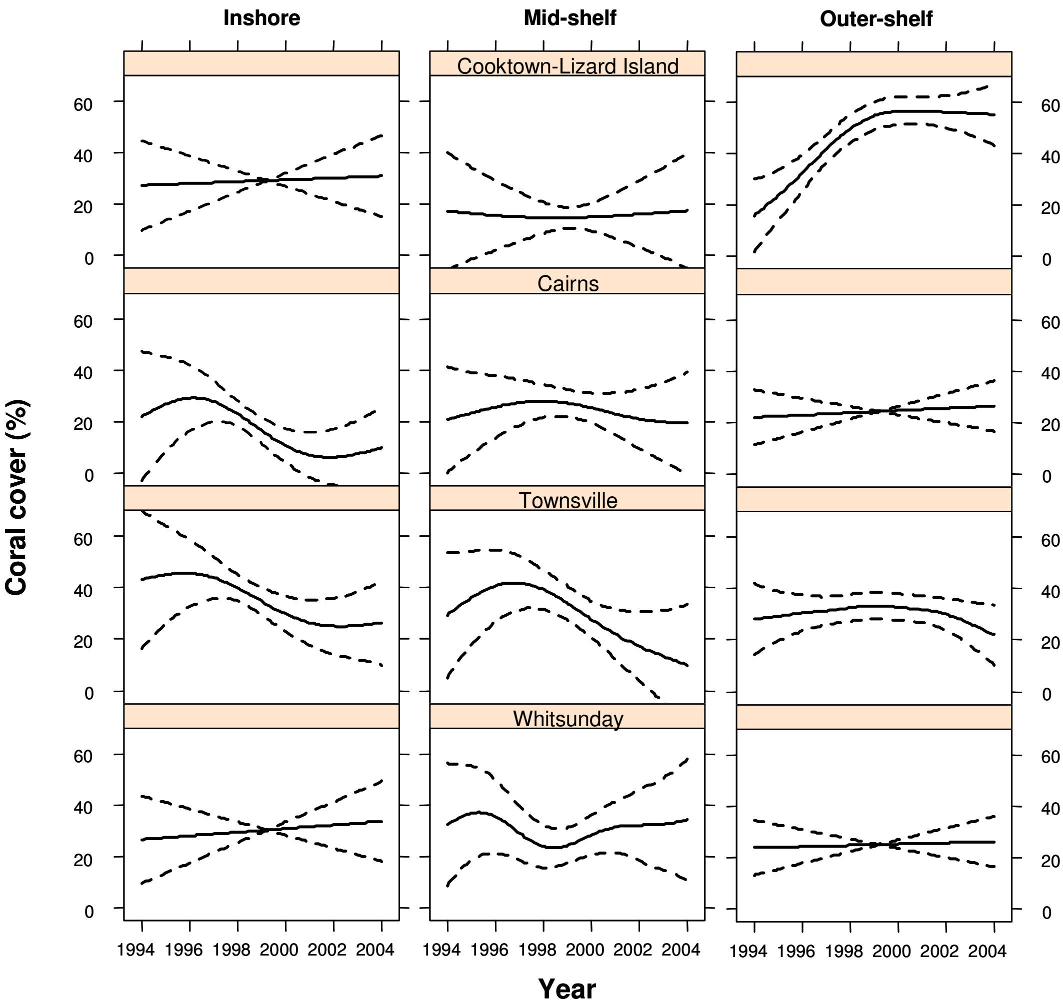


Figure 4.9 Estimated average trends in hard coral cover from video transects for all reefs within each of 4 latitudinal sectors in which the LTMP surveys inshore reefs between 1994 and 2004. Data from 11 annual surveys for the inshore, mid-shelf and outer-shelf regions. The solid lines correspond to the average trend within each sector and the dashed lines to approximate 95% confidence intervals based on variation between reefs. This analysis is similar to that reported in De'ath, (2005) but with data from the 2004 survey year included.

4.5 Rates of coral settlement at selected inshore reefs

This section summarises results from two independent studies of coral settlement and early survival.

Recruitment of Acropora corals on reefs of the Innisfail sector

Data and statistical methods

The rates of sexual recruitment of *Acropora* on reefs of the Frankland Islands, Fitzroy Island and High Island in the Innisfail sector (Wet Tropics) were quantified using terracotta settlement plates (110 x 110 x 10 mm) that were attached to the reef two weeks before the predicted mass spawning in 1999 and 2000 (Smith *et al.*, 2005). Three sets of five plates (n = 15) were attached approximately 50 m apart, at each of the four sites at each group of island reefs. The plates were collected after 8 weeks, bleached, and the number of recruits was counted under a dissecting microscope. Counts of *Acropora* recruits from three inshore islands were analysed using a generalised linear mixed model with an exponential link function and Poisson error variance. This allowed estimation of differences in the number of recruits among islands and between years on a proportional scale.

Summary findings

The number of recruits differed among islands and were different between survey years (Figure 4.10). The number of *Acropora* recruits was proportionally lower in 2000 than in 1999 by a factor of ~5 at Fitzroy Island, a factor of ~3 at Frankland Island and a factor of ~1.8 at High Island. The high temporal variability in rates of recruitment at these inshore reefs indicates that recruitment data from multiple years will be necessary to understand the processes of coral replenishment at these locations. These reefs are in a location that is at high risk of exposure to runoff and High Island, which is closest to the mouth of the Russell-Mulgrave River had lower rates of recruitment in these two years. These lower settlement rates could cause reefs at High Island to recover from disturbances relatively slowly compared with reefs at Fitzroy Island. However, adult populations are a product of recruitment rates and rates of subsequent mortality; further research needs to assess post-settlement mortality at each of these island locations. Note that these data refer to *Acropora* spp. recruits only, while the coral communities on the reefs of the Innisfail sector are dominated by *Porites* spp., and surveys in 2004 recorded the highest densities of small juvenile corals of all species on the reefs of the Innisfail sector; many of these recruits would have settled after 2000 (see Figure 4.3).

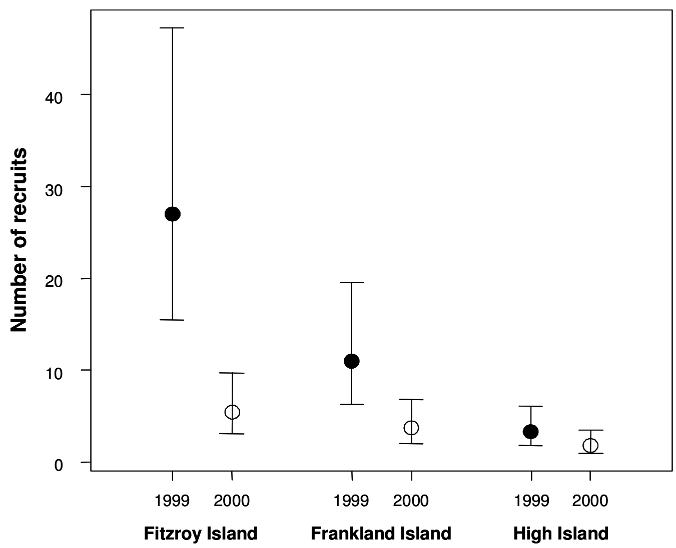


Figure 4.10 Mean number of Acropora spp. recruits per terracotta settlement plate on three inshore island reefs in 1999 and 2000 estimated from a generalised linear mixed model. Error bars represent 95% confidence intervals. Modified from Smith et al., (2005).

Recruitment of corals on mid-shelf and outer shelf reefs

Data and statistical methods

Coral recruits were recorded over seven years on three reefs: Lizard Island Reef (mid-shelf), Yonge Reef (outer shelf) in the Cooktown- Lizard Island sector (Cape York) and John Brewer Reef (mid-shelf) in the Townsville sector (Burdekin) of the Great Barrier Reef (E. Turak, unpublished data; De'ath, 2005). The three reefs were surveyed for annual coral recruitment on terracotta settlement plates from 1994 to 2000. Shallow and deep sites were surveyed on each reef. A total of 15 sites were used, each with 20 plates. Data were analysed using linear mixed effects models. Reefs were chosen as fixed so conclusions pertain to only these three reefs.

Summary findings

The numbers of coral spat settling per plate varied greatly, with standard deviations typically greater than the mean (Figure 4.11). Coral recruitment on John Brewer Reef declined over the seven years of surveys. This reduction occurred mainly in the last two years. Coral recruitment was relatively stable at Lizard Island and Yonge Reefs. A large recruitment event occurred at a shallow site on Yonge Reef in 1996. The mean numbers of recruits per tile were of the same order of magnitude as on the inshore reefs of the Wet Tropics (Figure 4.10). Other studies of coral settlement on the Great Barrier Reef have also found recruitment to be very variable both spatially and between years (Fisk and Harriott, 1990; Dunstan and Johnston, 1998) though comparisons between studies are complicated by different tile sizes (and hence edge effects) and different exposure times.

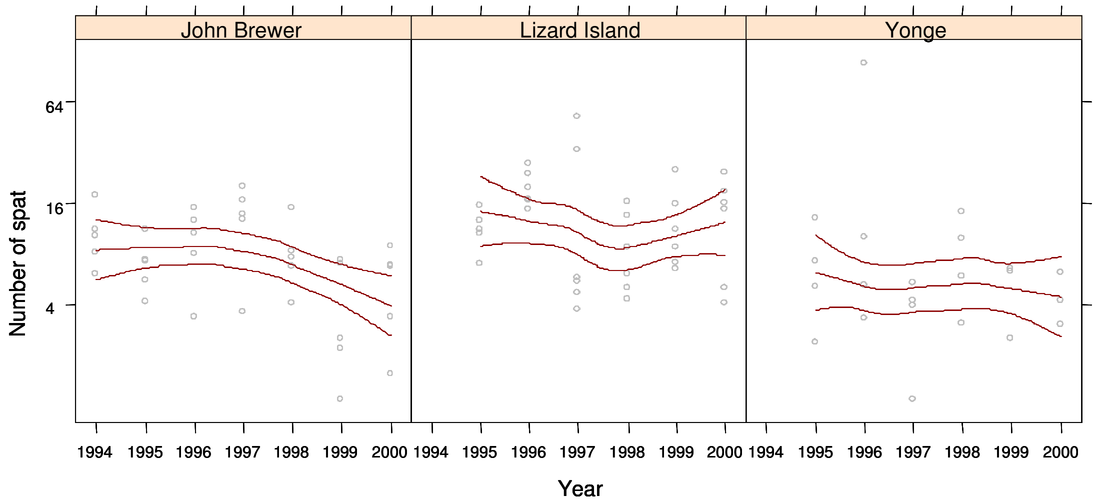


Figure 4.11 Modelled temporal trends (± 2SE) of recruitment numbers (spat per terracotta plate) on three reefs of the Great Barrier Reef: John Brewer Reef, Lizard Island Reef and Yonge Reef over seven years. The lines within each panel show the trends over time for each reef. Depth differences were negligible. John Brewer Reef showed weak evidence of decline in recruitment over time, whereas recruitment was more constant Lizard Island and Yonge Reefs. Note the log2 scale on the Y-axis. Modified from De'ath, (2005).

4.6 Water quality influences on inshore coral reefs

A major body of recent coral reef research funded through the CRC Reef Research Centre (CRC Reef) has focused on the ways in which runoff can affect coral reefs, using field and laboratory experiments and surveys on gradients in water quality. The numerous ways in which the various components of runoff can potentially affect coral reef communities have been reviewed by Fabricius, (2005).

The full consequences of changes in land use that have already occurred may not yet have flowed through to communities on inshore reefs of the Great Barrier Reef. Coral reproduction and recruitment is sensitive to a number of components of runoff. Laboratory and field studies show that elevated concentrations of nutrients and high levels of suspended sediment and turbidity can affect one or more of gametogenesis, fertilisation, planulation, egg size, and embryonic development in some coral species (reviewed by Fabricius, 2005). High levels of sedimentation can affect larval settlement or net recruitment of corals. Similar levels of these factors may have only sub-lethal effects on established adult colonies. Corals are potentially long-lived organisms, so reefs may maintain high coral cover under conditions of declining water quality, but they may be relic communities made up of adult colonies that became established under more favourable conditions. Further discussion of the potential impacts of changes in water quality to coral reefs is included in Section 7.

Conclusions

There are few long term studies of the coral communities on inshore reefs of the Great Barrier Reef but two major studies, by AIMS LTMP and by Sea Research, imply that the there were several major disturbances that affected inshore reefs in several areas during the 1990s. The most significant was the coral bleaching in 1998. Surveys in 2004 found there were inshore reefs with high coral cover (>30%) in most sectors. There were also reefs with low cover in most sectors. However, neither differences in coral cover nor difference in the taxonomic composition of the communities indicated a clear relationship to the estimated risk of exposure to runoff (Devlin *et al*., 2003).

Inshore reefs would have shown cycles of disturbance and recovery even in pre-European times. A major concern is that runoff from modified catchments now impedes recovery, particularly by causing reduction or failure of coral recruitment. The 2003-2004 surveys recorded the sizes of colonies that were present on inshore reefs. In most locations the majority of genera present as large colonies were also represented by recruit-sized individuals. However, there were relatively few small colonies at 5 out of the 33 survey locations: Peak and Pelican Islands (Fitzroy Basin), the back reef at Keswick Island (Mackay-Whitsunday), Middle Reef near Townsville (Burdekin) and coastal fringing reefs to the north of Cape Tribulation (Wet Tropics). In contrast, a high proportion of reefs in the Innisfail sector (Wet Tropics) had many genera that were only present as small individuals, probably indicating recovery from severe disturbance events in the past. While this shows that recruitment failure is not general, there is little information on rates of survival of small colonies. If mortality rates of small colonies are high, recruitment will still be inadequate to maintain the existing reef communities. Patterns in recruit density and recruitment in relation to existing community composition were not simply related to estimates of risk of exposure to runoff.

Long-term monitoring by the AIMS LTMP provides information on population trends of key groups of coral reef organisms, including corals, algae, reef fishes and crown-of-thorns starfish. Annual surveys of reefs in inshore, mid-shelf and outer shelf "regions" in six latitude- bands (sectors)on the Great Barrier Reef over 11 years show that reefs within regions have synchronised cycles of disturbance and recovery, but these cycles are not generally synchronised among regions. Coral cover on inshore reefs in the Cooktown-Lizard Is sector (Cape York) and the Whitsunday sector (Mackay-Whitsunday) has been stable or increased slightly over the 11 years. Coral cover declined on most inshore reefs in the Cairns (Wet Tropics) and Townsville (Burdekin) sectors, coinciding with a major bleaching event in 1998 and COTS outbreaks. However, these inshore regions are also assessed to be at moderate to high risk of exposure to land runoff. Coral cover also declined on mid-shelf reefs in the Cairns, Townsville and Whitsunday sectors and on outer-shelf reefs in the Townsville sector over the same period. Coral cover increased on the outer-shelf reefs in the Cooktown-Lizard Is sector from a low level at the beginning of surveys. There was no evidence of substantial change on the mid-shelf reefs Cooktown-Lizard Is sector, nor on outer-shelf reefs in the Cairns and Whitsunday sectors.

Surveys found little evidence that taxonomic composition of benthic communities on inshore reefs were changing, as might happen if relic communities were dying out. In most cases coral taxa that were present on a reef as adults were also present as small colonies. There were a number of reefs where recruits of many taxa were few. As with coral cover there was no simple relationship between recruit densities and estimated risk of exposure to runoff. The limited information on coral settlement shows that it varies from year to year and place to place. Settlement has occurred in many inshore locations in recent years, which shows that water quality is not generally so poor as to cause recruitment failure. There is little information on rates of post-settlement survival to adulthood, making it impossible to say whether the observed rates of settlement are likely to sustain the communities.

When the reefs of Great Barrier Reef in general are grouped by latitude and position on the continental shelf into 'regions', the corals on the reefs within regions tend to show synchronous patterns of decline and recovery after disturbance. These cycles are not synchronised among regions, reflecting the scale of disturbances. Within a region, inshore reefs seem to be more variable in terms of coral cover and community types than is the case for mid-shelf and outer shelf reefs. Inshore reefs with quite different levels of coral cover can occur only a few km apart. It could be that impacts are more localised or there is much greater variation in recovery rates on inshore reefs. Environmental gradients in turbidity, salinity, exposure to floods and many other stresses may be steeper in inshore areas.

Assessment of ecosystem health will require measurements of such environmental factors and knowledge of the taxonomic composition and ecology of the coral communities that can be expected to develop in such conditions. Since these do not yet exist it is hard to give definitive assessments of health of inshore ecosystems. Much of the circumstantial evidence for impacts of runoff on inshore reefs is based on the apparent slow recovery from disturbances. Many inshore reefs on the Wet Tropics coast were extensively damaged by bleaching in 1998. Reporting in 2003 on information presumably gathered a year or two beforehand (Baker, 2003) considered that recovery was slow or non-existent. In 2004, six years after the bleaching event, the highest densities of coral recruits in any survey locations were to be found on Wet Tropics reefs. An assessment of what are realistic rates of recovery from various types and intensities of impact is clearly critical.

To address this, the assessment of inshore reefs ecosystems under the current Reef Plan MMP will measure coral cover, community structure and community demographics, including recruitment rates, to assess the balance between disturbance and recovery processes. Integrating these coral data with knowledge of environmental factors such as temperature and water quality parameters, which are measured at the same sites, and the disturbance history at a local scale will be critical to the interpretation of coral community changes over the next decade.

5. Status and trends of intertidal seagrass communities in GBRWHA as determined by the Seagrass-Watch Monitoring Program

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Introduction

The bays, estuaries, lagoons and reef platforms of the GBRWHA provide habitat for 15 seagrass species. Seagrasses are important components of the Great Barrier Reef ecosystem and have the ability to act as a bio-sink for nutrients, sometimes containing high levels of tissue nitrogen and phosphorous. They also provide food and shelter for many organisms, and are a nursery grounds for commercially important prawn and fish species. Twenty prawn species and 134 fish species were found in the seagrass meadows from Cairns Harbour alone.

The first broad baseline of inshore seagrass resources in Queensland was conducted in the mid-80's. From the early/mid-90's, mapping has been repeated at greater/finer details at many locations related particularly to issues of dugong, fisheries and port management. These repeated maps found that the seagrass distribution was similar to previous maps. The extent/distribution of seagrass communities at a Queensland wide scale has remained relatively stable over the past 5-10 years (no net loss or gain). In the early nineties when assessments were few, there were significant losses recorded in the south related to flooding in Hervey Bay, and anecdotal reports of losses around the Cairns and Townsville regions. However, these meadows recovered within approximately 3 years. Recovery of seagrass meadows has also occurred in some areas of northern and southern Queensland that previously were damaged by flood waters in 1999-2000 (Campbell and McKenzie, 2004).

The best estimates of the area of seagrass meadows along the east coast are 5,668 km^2 of intertidal and shallow subtidal habitat (down to 15m water depth) (Hyland *et al*., 1989, Ayling *et al*., 1997; Lee Long *et al*., 1997; Lee Long *et al*., 1998; McKenzie *et al*., 1997; Coles *et al*., 2001a-e; McKenzie *et al*., 1998; McKenzie *et al*., 2000; Queensland Department of Primary Industries and Fisheries (DPI&F) unpublished data). Shallow and intertidal seagrass meadows are influenced by coastal topography and shelter with most seagrass meadows found in north facing bays and estuaries. The area of seagrass meadows in reef lagoon waters of the GBRWHA deeper than 15 m is likely to be as high as 40,000 km^2 (Coles *et al*., 2000). In these deeper waters there is a noticeable change in seagrass distribution and abundance from north to south. Seagrass meadows are sparse north of Princess Charlotte Bay and south of Mackay in the area where tidal velocities are high. The highest seagrass densities occur between Princess Charlotte Bay and Cairns and south of 23°S.

The greatest potential for loss of seagrasses is associated with downstream effects of land use and from global influences such as climate change and the related increase in storm frequency. In relation to water quality, the most common cause of seagrass loss is the reduction of light availability due to chronic increases in dissolved nutrients which leads to proliferation of algae thereby reducing the amount of light reaching the seagrass (e.g. phytoplankton, macroalgae or algal epiphytes on seagrass leaves and stems), or chronic and pulsed increases in suspended sediments and particles leading to increased turbidity (Schaffelke *et al.*, 2005). In addition, changes of sediment characteristics may also play a critical role in seagrasses loss (refer also to Section 7). There is 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. These influences are interlinked in complex ways and it is expected that the Reef Plan MMP will support the process of understanding and quantifying these links.

This section presents the status and trends of intertidal seagrass at sites in six GBRWHA coastal regions and in the Hervey Bay region, based on data from the Seagrass-Watch program. The information includes a qualitative assessment of the status of seagrass meadows in each region using the ratings Good, Fair, Poor, or undetermined condition if insufficient data available; the criteria is in Table 5.1. A rating system was developed to express the overall "state" of seagrasses resources in each locality and region. The rating system was based on an assessment of the status (poor, fair, good) of each parameter (e.g. trend in seagrass cover) at each site. Overall rating for each locality were based on an overview of all data from each site.

There are other long-term seagrass monitoring programs that provide information on specific sites within the GBRWHA that are not part of the present program. These are generally more targeted projects, such as monitoring ports, or fisheries protected areas (e.g. Cairns Harbour and Trinity Inlet [Rasheed *et al.*, 2004]; Port of Mourilyan [Thomas and Rasheed, 2004]). More information is available on these projects through the DPI&F website "Fishweb" (www.dpi.qld.gov.au/fishweb) or the CRC Reef (www.reef.crc.org.au).

5.1 The Seagrass-Watch program

The Seagrass-Watch monitoring program was established in 1998 as an initiative of the DPI&F. This program monitors the seasonal dynamics of seagrass meadows, the relationships between seagrass condition and climate change and the loss and recovery of seagrass meadows and provides an early warning of change of the intertidal seagrasses of the GBRWHA. It involves local community groups assisting in mapping and monitoring seagrass habitats vital for fisheries, turtles and dugongs. Local community volunteers are trained by DPI&F in the application of methods for scientifically rigorous assessment of seagrass resources. The sampling design and the parameters were developed in collaboration with the community and research scientists. Seagrass-Watch monitoring currently occurs in 11 regions across Queensland (Figure 5.1), 6 of which are within the GBRWHA. Seagrass-Watch is an ongoing program and current updates and information are available on www.seagrasswatch.org. DPI&F are charged under legislation with protection of seagrass fisheries habitat and the Seagrass-Watch program aims to satisfy this by ensuring no further losses of seagrass occur. No further loss of seagrass and the fisheries habitat function resulting from reduced water quality would be a measure of success of the Reef Plan.

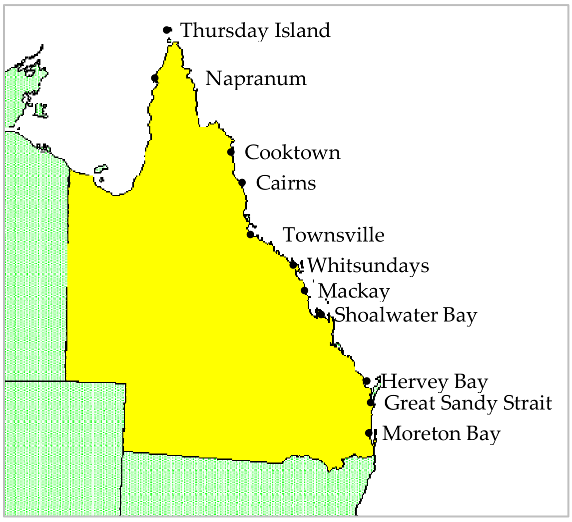

Figure 5.1 Location of Seagrass-Watch monitoring regions in Queensland.

Monitoring sites are selected in consultation with community volunteers, management agencies, local government, and seagrass researchers. Seagrass-Watch monitoring is coupled where possible with existing environmental monitoring programs (e.g. seagrass depth range, water quality and beach profile) to increase the ability to identify causes for change. The monitoring is conducted using a nested design at three scales: transect (metres), sites (kilometres) and locations (10s kilometres). Monitoring sites are established in areas of *a)* relatively high usage, *b)* where usage may be high in the near future and *c)* in comparable 'control' sites where current and predicted usage is low and likely to remain low. Generally, three sites are established at each location. At each site, three parallel 50 m transects (each 25 m apart) are established; the middle transect is permanently marked. The seagrass habitats along each transect are assessed by visual observation. At each transect, eleven quadrats are sampled (1 quadrat every 5 m) for seagrass cover and species composition, every three or six months, depending on site access and availability of volunteers (Figure. 5.2). Quadrats are photographed to ensure standardisation/calibration of observers and to provide a permanent record.

Figure 5.2 Seagrass-Watch volunteers applying seagrass community monitoring methods using transect tape and quadrats.

The visual observation data recorded for each quadrat includes: an estimation of the seagrass biomass for each species present, an estimation of the algae biomass, height of the seagrass canopy (dominant species), a sediment descriptor and counts of any animals seen in the quadrat. The comments column can also be used to alert the program if any changes are occurring outside the quadrat location. A qualitative ranking scale is given in plain English (Poor, Fair etc.) as feed back to the community groups (see Table 5.1). The response of seagrasses to changed water quality is not included because it is complex and can only be assessed on a site by site basis. Sediment loads in the water reducing available light would most likely result in loss of biomass. Eutrophication, on the other hand, may initially cause a growth spurt as the plants take up increased nutrients followed by an increase in epiphytes and eventual damage to the seagrass meadow. The Seagrass–Watch program was designed to provide an "early warning system" and any change would be followed up by more detailed studies and assessment.

There are many limitations to a community based intertidal program. For example, shallow subtidal communities may be the first to be impacted by reduced water clarity and this may not be immediately observed by focussing on the intertidal. However, these subtidal environments are logistically difficult and expensive to monitor on a regular and statewide basis.

De'ath, (2005) has undertaken exploratory analyses of the currently available Seagrass-Watch dataset to estimate expected performance of this monitoring program. The analyses included data from 2000–2004 at 63 sites in 29 locations in 6 regions (Cooktown, Cairns, Townsville, Whitsundays, Hervey Bay, Great Sandy Strait). Results indicate that the Seagrass-Watch monitoring methods are appropriate to detect change of intertidal seagrass communities on various scales. For the Reef Plan MMP two additional locations will be added to increase the spatial coverage of the monitoring.

5.2 Regional status and trends of intertidal seagrass communities

This section provides an overview of the results of the Seagrass-Watch program in each region of the GBRWHA. In general, there is little known about long-term natural cycles in the abundance and distribution of seagrasses in the Great Barrier Reef. From the results of the monitoring programs to date, no large scale losses of seagrass have occurred within the GBRWHA within the last 10 years. However, the Seagrass-Watch program has detected considerable variability in abundance at local, site or location scales (De'ath, 2005) with significant declines in some locations, e.g. Cid Harbour in the Whitsundays. This requires closer investigation but is likely to be an exception rather than the rule. At most sites there is likely to be local variability and a longer time series will be necessary to identify any trends.

Table 5.1. Criteria for qualitative rating Seagrass-Watch monitoring locations.

Category and weighting	*Poor state Declining*	*Fair state Variable/Stable*	*Good state Stable/Improving*
Trend in seagrass cover and canopy height	Trend in seagrass abundance has severely declined successively between months (after >12 months monitoring abundance is significantly lower from time 0) and greater than 75% loss with little sign of recovery.	Trend in seagrass abundance is variable and declined successively between months (after >12 months monitoring abundance is significantly lower from time 0) and less than 50% loss.	Typical seasonal trend in seagrass abundance (after >12 months monitoring abundance is not significantly different from time 0) with less than 10% loss.
Species composition	Dominant species has severely declined successively between months (after >12 months abundance is significantly lower from time 0) and greater than 75% loss with little sign of recovery, excluding seagrass successional change.	Dominant species has declined successively between months (after >12 months abundance is significantly lower from time 0) and less than 50% loss.	Dominant species has changed less than 10% in species composition (after >12 months composition is not significantly different from time 0).
Algae & Epiphytes	Algal blooms persistent over the year, changes in species composition and high abundance (>30% cover).	Algal blooms less persistent (eg 1 bloom per year). Epiphyte cover stable.	Epiphyte cover less than 50% cover and algal cover minimal (< 10%).
Dugong and turtle grazing	Dugong and turtle grazing not evident.	Dugong and turtle grazing occasional.	Dugong and turtle grazing frequent.
Associated fauna	Macro-invertebrate numbers reduced by more than 50% over 12 month period.	Macro-invertebrate numbers remained stable over a 12 month monitoring period.	Macro-invertebrate numbers and diversity increased over a 12 month monitoring period.
Physical disturbance	Sediment composition changed. Sediment movement causing disturbance.	Sediment composition fairly stable with occasional disturbance.	Stable sediments. Minimal sediment movement.

Cooktown region

The Cooktown region is part of the wet tropical coast of north-eastern Queensland. It includes a major catchment and river system (Endeavor and Annan Rivers). Fishing, shipping, and land based activities such as mining and sewerage effluent released into the Endeavour river all have the potential to impact seagrass meadows in the region, either directly (through anchoring) or indirectly (through changes in water quality).

Intertidal seagrass meadows in the region are situated along inshore sand and mud banks and mostly consist of *Halodule uninervis* and *Halophila* spp. dominated meadows. Shallow subtidal coastal meadows consist of *Halodule uninervis* and *Halophila* spp. communities mostly found along the sheltered coasts and bays. *Cymodocea* spp., *Thalassia hemprichii* *Syringodium isoetifolium* and a suite of *Halophila* species dominate intertidal reef platform habitats of this region.

One Seagrass-Watch site has been monitored in the Cooktown region since 2003. Seagrass meadows in the Cooktown region are rated as in **good** condition (as per Table 5.1), and results of monitoring indicate that seagrasses appear relatively healthy.

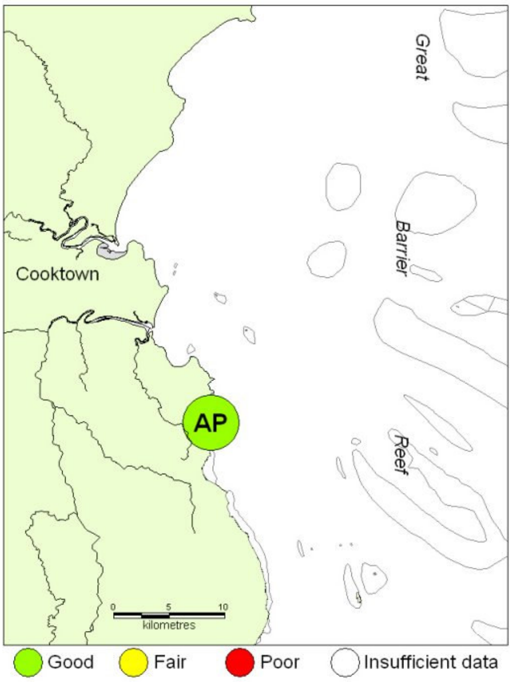

Figure 5.3 Status of seagrass meadows at the Seagrass-Watch monitoring location in the Cooktown region in February 2005. AP= Archer Point.

Archer Point

Location: On fringing reef flat in protected section of bay, fringed by mangroves, approximately 15km south of Cooktown.
Potential pressures: Land runoff.
Comments: Popular recreational fishing area and turtle feeding grounds.
Site summary: The site is dominated by *Halodule uninervis* and *Halophila ovalis* and seagrass cover is between 20 % in winter and 35 % in spring. Monitoring was established only in late 2003, and to date there are insufficient data to describe long-term trends.

Cairns region

The Cairns region is on the wet tropical coast of northeastern Queensland. It includes several major catchments and river systems. Agricultural (e.g. grazing, forestry and cropping) and urban developments are the primary land uses in coastal catchments and contribute high loads of nutrients and sediments to inshore coastal waters.

Intertidal seagrass meadows in the region are situated along inshore sand and mud banks and mostly consist of *Halodule uninervis* and *Halophila* spp. dominated meadows. In Cairns

Harbour however, meadows are dominated by *Zostera capricorni*. Shallow subtidal coastal meadows consist of *Halodule uninervis* and *Halophila* communities mostly found along the sheltered coasts and harbours. *Cymodocea* spp., *Thalassia hemprichii, Syringodium isoetifolium* and a suite of *Halophila* species dominate intertidal reef platform habitats of this region.

Seagrass meadows throughout the region are characterised by low nutrient concentrations and are primarily nitrogen limited. Nitrogen inputs to marine waters can therefore directly promote seagrass growth. The indirect consequences of high sediment and nutrient runoff can on the other hand detrimentally influence seagrass distribution and seasonality in the region. From December to March (wet season) low seagrass abundance often coincides with high rainfall and high loads of sediments and nutrients to inshore marine waters.

Sediments and nutrients decrease light availability for seagrass by reducing light penetration in the water column and promoting the growth of algae that can shade seagrass blades. Warm water temperatures during summer also promote growth of algae and can lower productivity of seagrass meadows. Coupled with physical disturbance from storm activity, seagrass growth and abundance may therefore be severely depleted over the wet season. Seagrass meadows are generally most abundant in October-November following the dry season that has more favourable light and temperature conditions.

Three Seagrass-Watch locations are monitored in the Cairns region since 2000, 2001 and 2002, respectively. Seagrass meadows in the Cairns region are rated as in **good** condition (as per Table 5.1), and results of monitoring indicate that seagrasses appear relatively healthy.

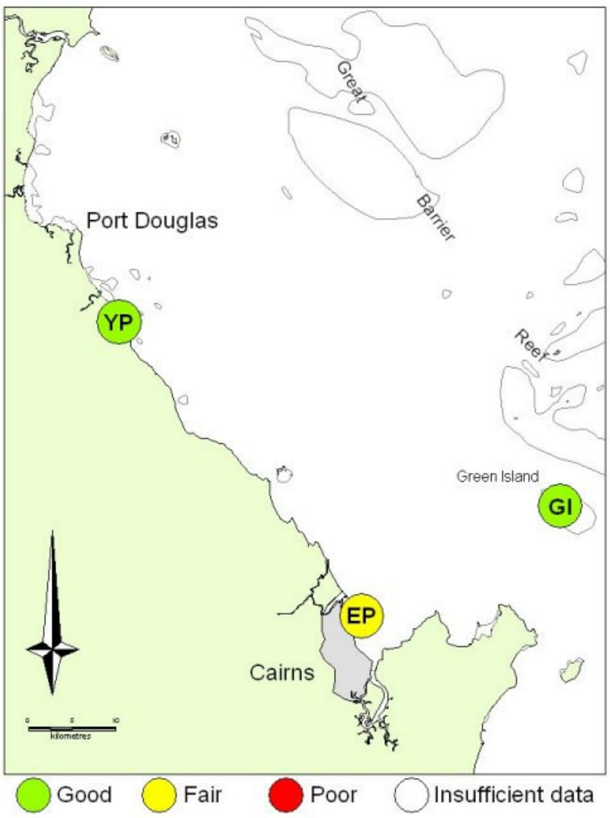

Figure 5.4 Status of seagrass meadows in the Cairns region in February 2005. YP= Yule Point, EP= ElliePoint, GI= Green Island.

Yule Point

Location: Coastal intertidal sand banks, protected by an extensive fringing reef.
Potential pressures: Storm water and land runoff, boat traffic.
Comments: Popular dugong feeding grounds.
Site summary: Two sites have been monitored since 2000. Sites are dominated by *Halodule uninervis* and *Halophila ovalis*. Associated macroalgae generally increase in spring/early summer, but are not at levels of concern. Epiphyte cover is high at times, and seems to be correlated with nutrient inputs from land runoff (i.e. high epiphytes after heavy rains). Finer sediments (e.g., fine sand) increased over 2004, but there were no obvious changes in seagrass species composition. Overall, changes at the sites appear to have changed relatively little since 1967, when den Hartog (1970) described the species present and sediment condition.

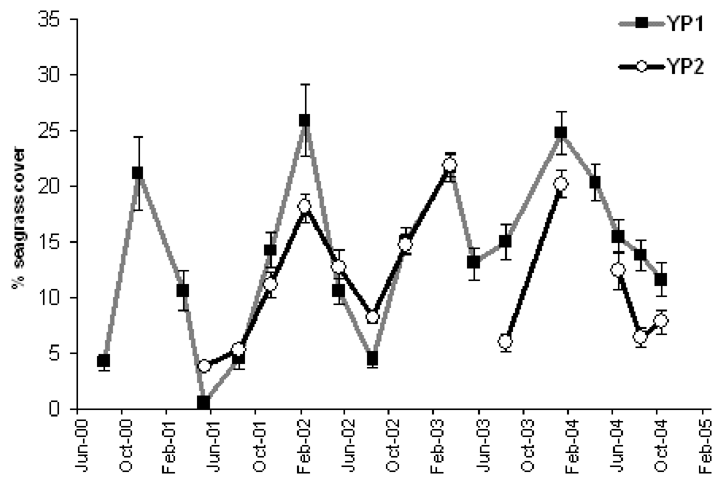

Figure 5.5 Seagrass cover (%) at the Yule Point sampling sites in the Cairns region.

Ellie Point

Location: On the north of Cairns Harbour, adjacent to the mouth of the Barron River.
Potential pressures: Land runoff and boat traffic.
Comments: Popular recreational fishing area and turtle feeding grounds.
Site summary: The site is dominated by *Zostera capricorni* with *Halodule uninervis* and *Halophila ovalis* and seagrass distribution and cover has recovered from substantial losses in December 2001 (Campbell *et al.,* 2002). Monitoring design at this location is not the standard 50x50m site, but rather at the meadow scale. Seagrasses at Ellie Point appear relatively healthy.

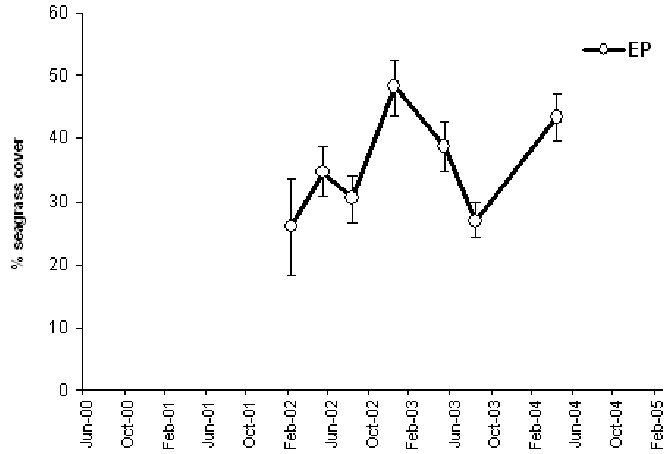

Figure 5.6 Seagrass cover (%) at the Ellie Point sampling site in the Cairns region.

Green Island

Location: Reef-platform on Great Barrier Reef mid shelf reef approximately 27 km north east of Cairns.
Potential pressures: Elevated nutrients and land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: The distribution of seagrass around Green Island has changed substantially in the last 50 years, possibly because of poor water quality (Udy *et al* 1999). The most dramatic change however, has been the seagrass species composition, with the species *Syringodium isoetifolium* now dominating most of the lagoon meadows. How these changes in the seagrass composition and abundance on Green Island will affect the sea turtle, dugong and fisheries is unknown.

The Seagrass-Watch site is dominated by *Cymodocea rotundata, Thalassia hemprichii, Halodule uninervis* and *Halophila ovalis.* Associated macroalgae generally increases in spring/early summer, although these are within levels of natural change.

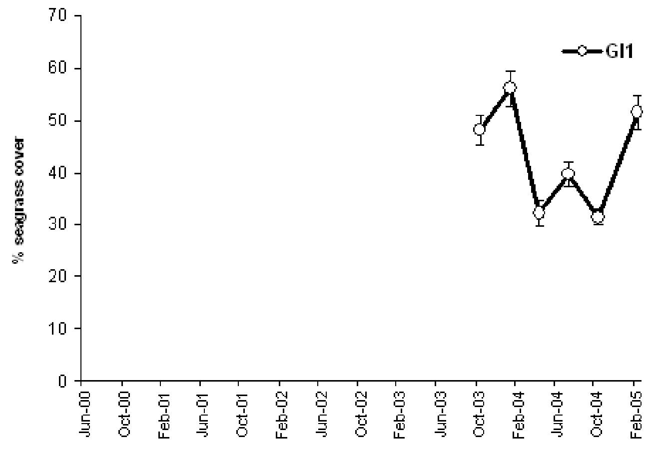

Figure 5.7 Seagrass cover (%) at the Green Island sampling site in the Cairns region.

Townsville region

The Townsville region is on the dry tropical coast of north-eastern Queensland. Intertidal and shallow subtidal seagrasses predominate and tend to form multi-specific meadows that are arranged in mono-specific bands across a depth gradient along Cape Cleveland, the

Strand, Cape Pallarenda, and around Magnetic Island. Area of seagrass in the region is estimated at 130 km^2, of mostly moderate (11-49%) and light (1-10%) cover.

The main seagrass species in shallow waters near Townsville are *Halophila ovalis, Halodule uninervis*, *Zostera capricorni*, and *Cymodocea serrulata*. *Halophila spinulosa* that has been washed up from deeper waters can sometimes be found. In April 2000, intertidal seagrass meadows in the Townsville region were decimated by a cyclone but they are now recovering.

The distribution of seagrasses along this coastline is predominately influenced by seasonal (April-November) south-easterly trade winds. Most sites show a fairly typical season pattern of seagrass abundance (higher in late spring-summer than winter). Seagrass meadows generally establish in places that offer protection from these winds, such as the large north opening bays and the lee ward sides of continental islands. The combination of seasonal terrestrial run-off, frequent cyclones, strong south-easterly trade winds and large tidal runs (in the south) creates significant coastal turbidity. Consequently seagrasses that inhabit this area are subjected to low light regimes, and high influxes of freshwater and sediment. To survive this regime seagrasses need to exhibit high vegetative growth rates and prolific seed banks. This has probably led to the predominance of opportunistic species, such as *Halodule* and *Halophila* within this region.

Three Seagrass-Watch locations have been monitored in the Townsville region since 2001 (Shelley Beach & Sandfly Creek) and 2002 (Bushland Beach), respectively. Seagrass meadows in the Townsville region are rated as in a fairly-good condition (as per Table 5.1) (Figure 5.8).

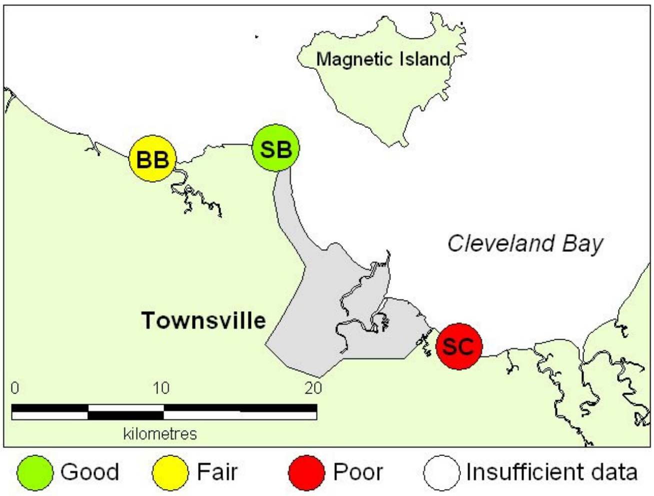

Figure 5.8 Status of seagrass meadows in the Townsville region in February 2005. BB= Bushland Beach, SB= Shelley Beach, SC= Sandfly Creek.

Bushland Beach

Location: South-western shores of Halifax Bay.
Potential pressures: Coastal development, land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: The site is dominated by *Halodule uninervis* with some *Halophila ovalis.* While cover of associated macroalgae was generally low, epiphyte cover was high in late 2004 (up to 85%).

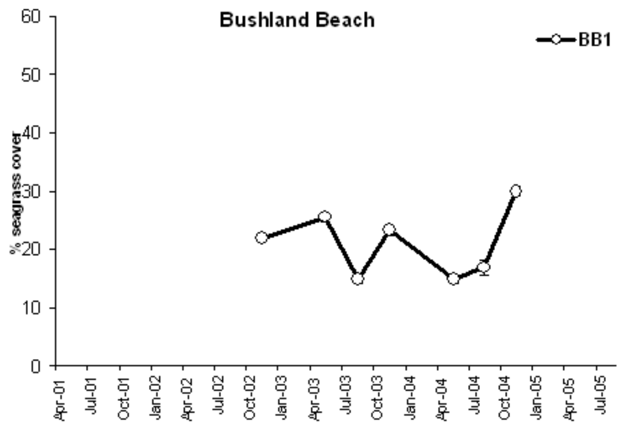

Figure 5.9 Seagrass cover (%) at the Bushland Beach sampling site in the Townsville region.

Shelley Beach

Location: Cape Pallarenda.
Potential pressures: Coastal development, land runoff.
Site summary: Sites were dominated by *Halodule uninervis* with *Halophila ovalis.* Seagrass abundance was noticeably lower in summer 2002, possibly caused by high water temperatures and other climatic factors. Associated macroalgae were generally higher in summer, and were more prevalent at SB1 than at SB2. Sediment at site SB1 became muddier over the last 18 months to February 2005, while no change was noted at SB2. SB2 has exceptionally abundant *Halodule uninervis* seeds, although this has been declining over the last 3 years. The seed bank has increased slightly at SB1.

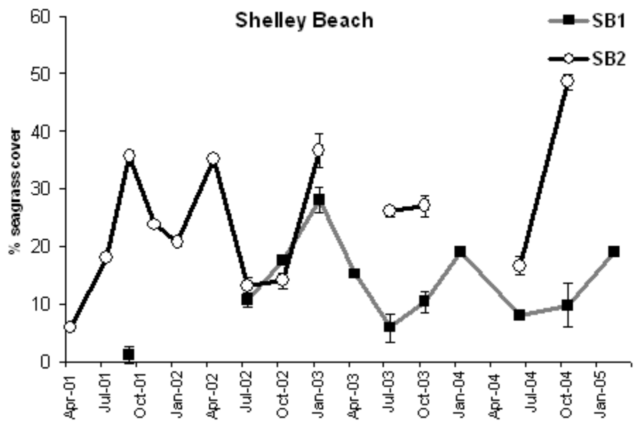

Figure 5.10 Seagrass cover (%) at the Bushland Beach sampling site in the Townsville region.

Sandfly Creek

Location: Southern shore of Cleveland Bay.
Potential pressures: Sewage treatment outfall, land runoff, coastal development.
Comments: Fishing grounds, Dugong and Turtle feeding grounds.
Site summary: The site is dominated by *Zostera capricornii* with *Halophila ovalis*, *Halodule uninervis* and *H. pinifolia*. There are currently insufficient data to describe long-term trends. Seagrass abundance at SC2 has substantially decreased since mid-2002, while SC1 has had low seagrass abundance. Associated macroalgae and epiphyte cover was lower than expected, but highly variable. The seagrass and mangrove aerial roots were covered in

filamentous algae in July 2004, which seemed to be widespread phenomena at that time of year. Sediments appeared less muddy than previously.

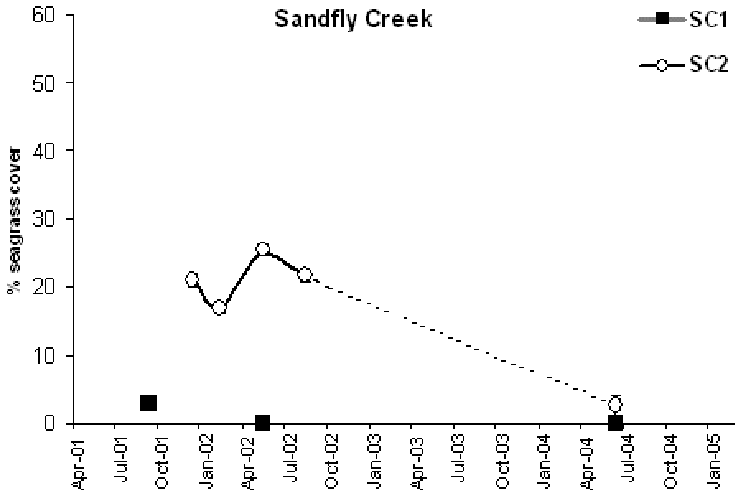

Figure 5.11 Seagrass cover (%) at the Sandfly Creek sampling site in the Townsville region.

Whitsundays region

The Whitsunday region on the central east Queensland coast, extends from Gloucester Island in the north to Midge Point in the south and includes a convoluted coastline and several large continental islands.

The Whitsunday region has extensive seagrass meadows occurring both on intertidal mudflats and in inshore and offshore subtidal regions. The region contains 5,554 hectares of seagrass from Midge Point in the south to Hydeaway Bay in the north. The tidal range is up to 4.1m. Meadows generally show a typical season pattern of seagrass abundance (higher in late spring-summer than during winter).

The catchment encompasses extensive urbanised residential areas fringing the coastline. Treated sewage effluent from the townships of Airlie Beach and Cannonvale is discharged into Pioneer Bay. Other townships in the region are unsewered and rely of septic tanks. Agricultural land use in the catchment includes cane production and lowland grazing. Cape Conway National Park is in the centre of the region. Two major rivers enter Repulse Bay: the Proserpine and the O'Connell Rivers.

Eight Seagrass-Watch locations are have been monitored in the Whitsundays region since 1999. Seagrass meadows in the Whitsunday region are rated as in a Fair condition (as per Table 5.1), and results of monitoring indicate that seagrasses appear relatively healthy, although poor health from 3 locations is of some concern (Figure 5.12)

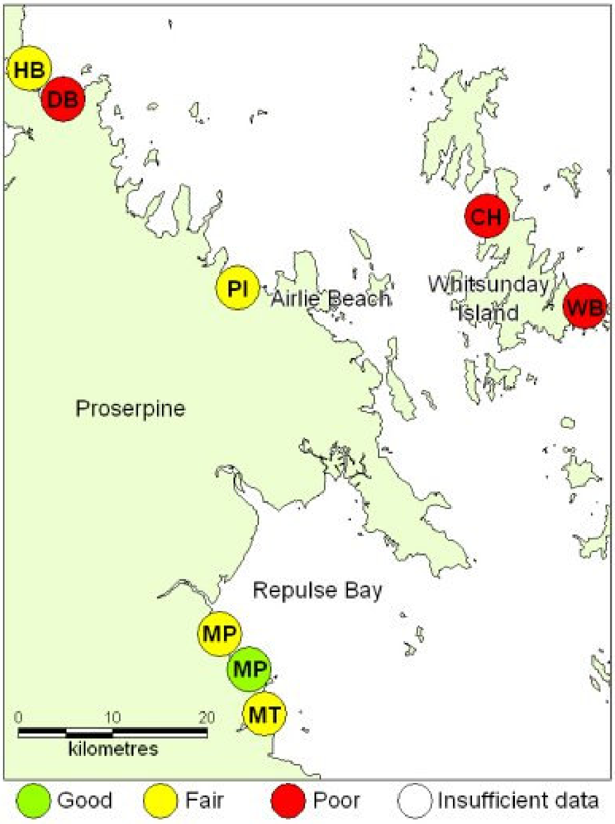

Figure 5.12 Status of seagrass meadows in the Whitsundays region in February 2005. HB= Hydeaway Bay, DB= Dingo Beach, PI=Pigeon Island, CH= Cid Harbour, WB= Whitehaven Beach, MP= Midge Point, MT= Midgeton.

Cid Harbour

Location: Subtidal banks on the north-western coast of Whitsunday Island.
Potential pressures: Increasing vessel use, anchor impacts.
Comments: Important foraging habitat for green sea turtles and dugongs.
Site summary: Cid Harbour meadows consist mainly of *Halodule uninervis* (wide leaf form) in association with *Cymodocea serrulata, Halophila spinulosa* and *H. ovalis*. Low proportions of *H. ovalis* and negligible *Syringodium isoetifolium* indicate low disturbance within meadows. The meadow adjacent to Cid Harbour is one of the largest in the region (~1433 ha), however, seagrass abundance has steadily declined over the past 4 years. Epiphyte and associated macroalgae cover was generally higher in spring, reflecting a seasonal response to increasing light and temperature. Sediments were composed of fine mud, sand and shell with a high organic component. Disturbance from boat anchors at Cid Harbour sites was minimal. Turtle and dugong grazing was common from September to February and dugongs were commonly observed at this location.

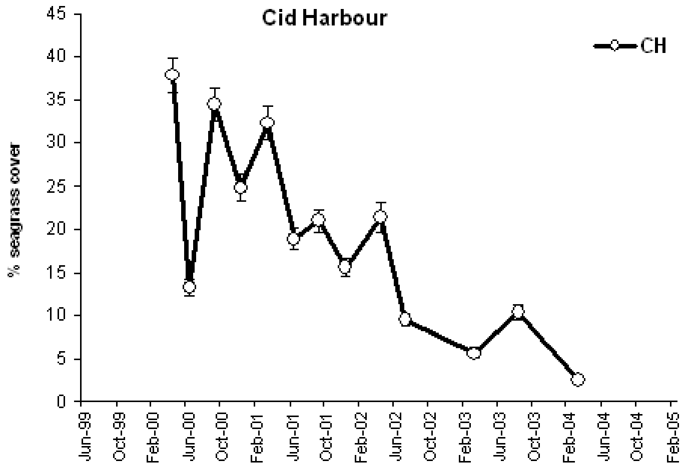

Figure 5.13 Seagrass cover (%) at the Cid Harbour sampling site in the Whitsundays region.

Dingo Beach

Location: Intertidal sand flats in north facing mainland bay, in the north of the region.
Potential pressures: Coastal development, sewage and groundwater inputs, siltation due to clearing and erosion.
Comments: Turtle and fish feeding grounds.
Site summary: Dingo Beach meadows cover approximately 55ha, which are predominately *Halodule uninervis* with *Halophila ovalis* and some *Thalassia hemprichii, Syringodium isoetifolium* and *Cymodocea serrulata*. The species composition has remained relatively stable over the monitoring period, indicative of natural and/or anthropogenic disturbance. Maximum epiphyte cover (>35-75%) occurred in summer and autumn, and minima in winter. Macroalgal cover remained below 20% with no seasonal pattern. High epiphyte cover in spring-summer may have been caused by high water temperatures and light availability, and possibly by nutrient enrichment after high summer rainfall. Sediments were fine to medium sands, exposed to wave action, and generally had a low seagrass abundance (<20% cover). There are no major rivers flowing into this coastal section and impacts from catchment inputs and urban and agricultural development are likely to be low. The seagrass in the region is a significant food source and habitat for green sea turtle and dugong.

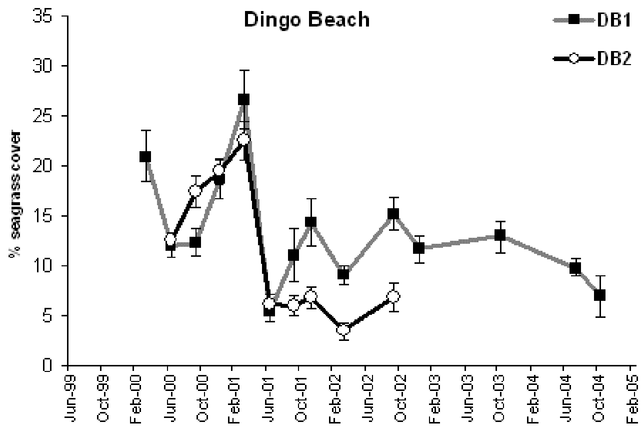

Figure 5.14 Seagrass cover (%) at the Dingo Beach sampling sites in the Whitsundays region.

Hydeaway Bay

Location: Large fringing reef-flat in north facing mainland bay, in the north of the region.

Potential pressures: see Dingo Beach above.
Comments: Turtle and fish feeding grounds.
Site summary: Meadows cover approximately 157ha and are predominately mixed meadows of *Halodule uninervis, Halophila ovalis, Cymodocea rotundata* and *Thalassia hemprichii.* The seagrass species composition has remained relatively stable over the monitoring period, indicative of natural and/or anthropogenic disturbance. Maximum epiphyte cover (>35-75%) occurred in summer and autumn, and minima in winter. Macroalgal cover remained below 20% with no seasonal pattern. High epiphyte cover in spring-summer may have been caused by high water temperatures and light availability, and possibly by nutrient enrichment after high summer rainfall. Sediments were fine to medium sands, exposed to wave action, and generally had a low seagrass abundance (<20% cover). Sediments had low organic matter content and seagrasses compete for space with soft and hard corals and macroalgae. There are no major rivers flowing into this coastal section and impacts from catchment inputs and urban and agricultural development are likely to be low. The seagrass in the region is a significant food source and habitat for green sea turtle and dugong.

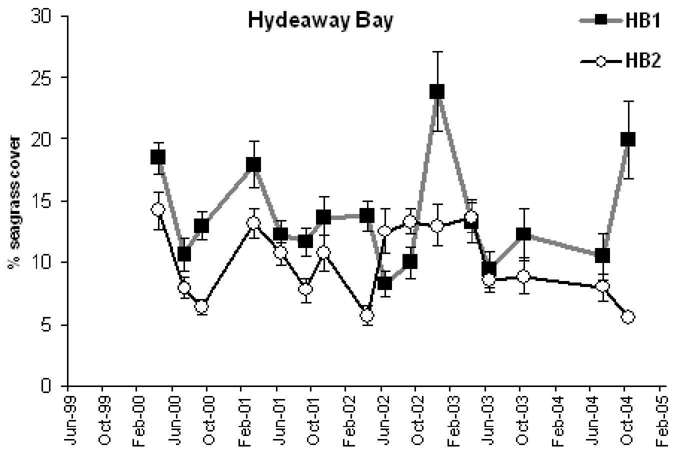

Figure 5.15 Seagrass cover (%) at the Hydeway Bay sampling sites in the Whitsundays region.

Laguna Quays

Location: Sand/mud flat in Repulse Bay, 3km south of O'Connell River mouth.
Potential pressures: Resort and marina development, vessel traffic, catchment inputs from agriculture, mangrove clearing.
Comments: Dugong and turtle feeding grounds.
Site summary: Meadows were dominated by *Halodule uninervis* (MP1), and *Z. capricorni* (MP4). The relative proportions of species remained stable over the monitoring period. Epiphyte cover (<30%) and macroalgal cover (<1%) were low. Dugong feeding trails were abundant at Laguna Quays and highest feeding activity was recorded in March and September 2000. Sediments were fine to medium sands, exposed to wave action, and generally had a low abundance of seagrass (<20% cover).

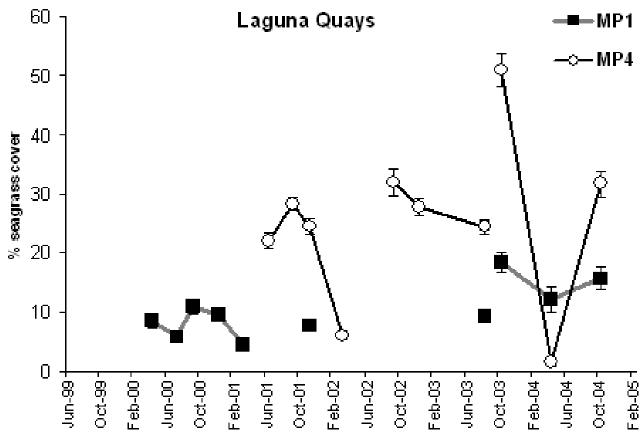

Figure 5.16 Seagrass cover (%) at the Laguna Quays sampling site in the Whitsundays region.

Midge Point

Location: sand/mud flat along coast of southern Repulse Bay.
Potential pressures: Low urban development, close to mangroves.
Comments: Fish, dugong and turtle feeding grounds.
Site summary: Meadows cover approximately 30ha, and are dominted by *Zostera capricorni* with low amounts *Halodule uninervis* and *Halophila ovalis*. The relative proportions of species at each site remained stable over the monitoring period. Epiphyte cover (<30%) and algal cover (<1%) was low. Dugong feeding trails were abundant. Sediments at tidal-dominated localities were composed of fine mud and sand with a high organic component. Disturbance to seagrass meadows may be caused by a number of factors. At Midge Point wave action from prevailing south-easterly winds and strong tides resulted in sediment movement where fine muds were displaced with coarse sands and shell.

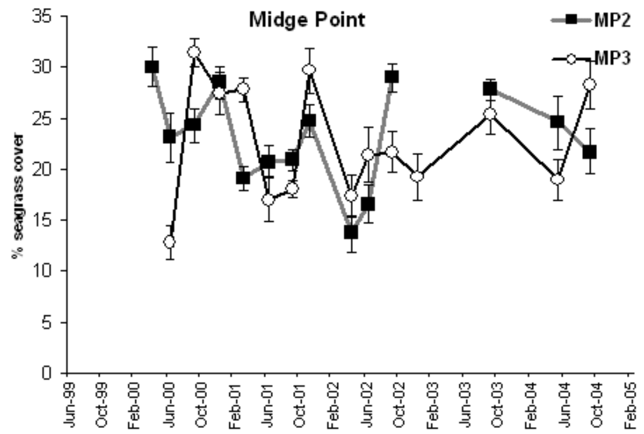

Figure 5.17 Seagrass cover (%) at the Midge Point sampling site in the Whitsundays region.

Midgeton

Location: Near mouth of Dempster Creek estuary south of Midge Point.
Comments: Dugong, turtle and fish feeding grounds.
Site summary: Meadows consist of an equal mix of *Z .capricorni, H. uninervis* and *H. ovalis.* The relative proportions of species at each site remained stable over the monitoring period. In 2000 and 2003 in seagrass cover declined from July (winter) to February (summer),

possibly due to disturbance from sediment movement associated with rainfall and freshwater inputs from Dempster Creek, together with strong wave action and south easterly winds. Epiphyte cover (40-70%) at Midgeton sites was high in spring-summer and low in winter, while macroalgal cover was low (<2%). Dugong feeding trails are abundant with highest feeding activity recorded in March and September 2000. Sediments at tidal-dominated localities were fine muds and sands with a high organic component.

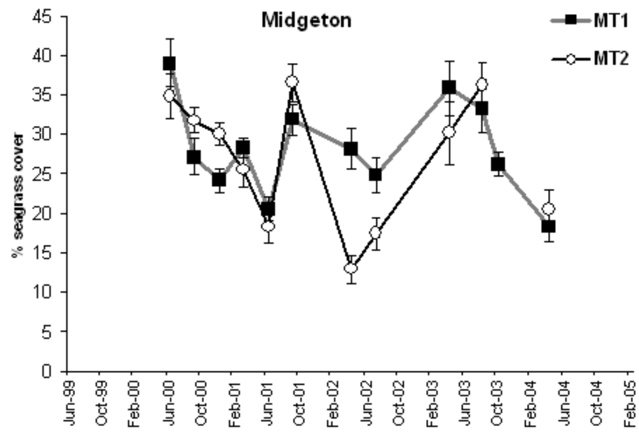

Figure 5.18 Seagrass cover (%) at the Midgeton sampling sites in the Whitsundays region.

Pigeon Island (Pioneer Bay)

Location: Intertidal sand/mud flats adjacent to Cannonvale in southern Pioneer Bay.
Potential pressures: Marina and urban development, inputs of treated sewage.
Comments: Dugong and turtle feeding grounds.
Site summary: Meadows cover approximately 60ha, and are dominated by *Halodule uninervis* and *Halophila ovalis* mixed with low amounts *Zostera capricorni*. Species composition remained stable over the monitoring period and indicated natural and/or anthropogenic disturbance. At two sites (PI1 and PI2) seagrass cover was relatively high, a possible consequence of elevated nutrients from an adjacent sewage outfall. Seagrass at the other sites (PI3 and PI4) were low (<15%) but highly variable. Epiphyte cover was high (30-70%) and persisted throughout much of the year. Macroalgal cover was high (10-50%) in winter, spring and summer and indicates possible nutrient enrichment from local sources and impact on seagrass meadows. Dugong feeding trails were abundant at Pigeon Island sites with the highest feeding activity was recorded in March and September. Anthropogenic disturbance (sewage inputs, stormwater runoff, boat discharges) results in accumulation of fine muds with a high organic component, however sediment mud levels have declined since 2003.

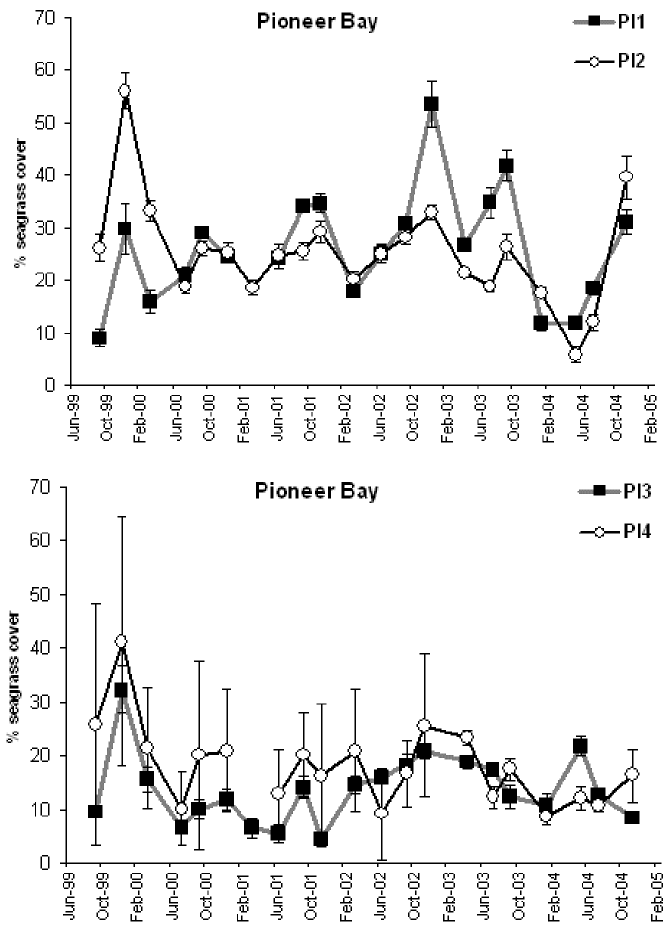

Figure 5.19 Seagrass cover (%) at the Pioneer Bay sampling sites in the Whitsundays region.

Whitehaven Beach

Location: Subtidal shores of beach on the eastern coast of Whitsunday Island.
Potential pressures: High boat usage, anchoring.
Comments: Turtle and fish feeding grounds.
Site summary: Subtidal meadows cover approximately 365ha, and are dominated by a mix of *Halodule uninervis, Halophila ovalis, Cymodocea serrulata* and *Syringodium isoetifolium*. Sites were established in relation to anchoring impacts: high impact site (WB3) and low impact site (WB2). WB3 had a higher proportion of *H. ovalis* and *S. isotetifolium* than WB2. Both species colonise disturbed areas and were in highest abundance from spring to summer when light and temperature are favourable for fast growth. From 1987 to 1999-2000 seagrass meadows at Whitehaven Beach increased in area, with the seaward edge extending up to 300m beyond the edge mapped in 1987. At initiation of monitoring, seagrass cover was significantly higher at WB2 than at WB3, suggesting that boat anchors cause a reduction in seagrass abundance. The abundance of epiphytic and non-attached algae at WB2 and WB3 was generally low (<10%). In March 2001 the blue green alga *Lyngbya majuscula* covered extensive areas (35-70%) of seagrass. Evidence of dugong and turtle grazing was low at Whitehaven Beach. Sediments were composed of fine mud, sand and shell with a high organic component.

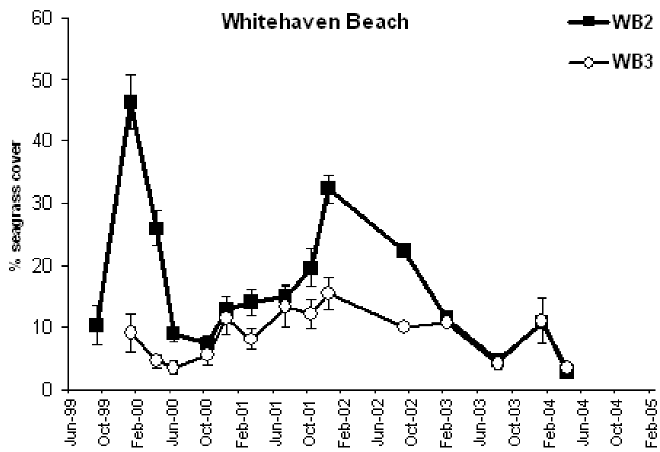

Figure 5.20 seagrass cover (%) at the Whitehaven Beach sampling sites in the Whitsundays region.

Mackay region

The Mackay region is on the dry tropical coast of central-eastern Queensland. Coastal waters adjacent to the large rivers and mangrove-lined inlets are generally very turbid and shallow, with predominantly muddy sediments. Tidal range in the region is large, and in some places has the effect of creating extensive tidal banks. The Port Newry region north of Mackay was declared a Dugong Protection Area in January 1998. The region receives rainfall between 500-3000 mm annually, which falls mostly from December to April. The major land use of each catchment is livestock grazing, and crops such as sugar cane.

Extensive meadows of *Halodule uninervis/ Halophila ovalis* or *Zostera capricorni* exist on the coastal intertidal flats of the region. Along much of the coastline, sheltered areas are few and generally small, and are exposed to south-east winds. Small *Halodule* or *Halophila* spp. meadows are found in the lee or in bays of islands. Strong tidal currents and associated high water turbidity in this region limit light penetration and therefore the depth to which seagrasses can grow. *Halophila ovalis, Halophila decipiens, H. spinulosa* and *H. tricostata* are found in deeper waters.

The seagrass meadows in the Mackay region were first mapped during a broad scale survey in 1987 with an estimated total cover of 490 ha. In 1999, approximately 2,450 ha of seagrass habitat was mapped on mud through to sand substrates and extending to 5.5 m below MSL in St Helens Bay.

Two Seagrass-Watch locations are have been monitored in the Mackay region since 2004. Monitoring has only recently been established and there are insufficient data to determine current condition or to describe long-term trends.

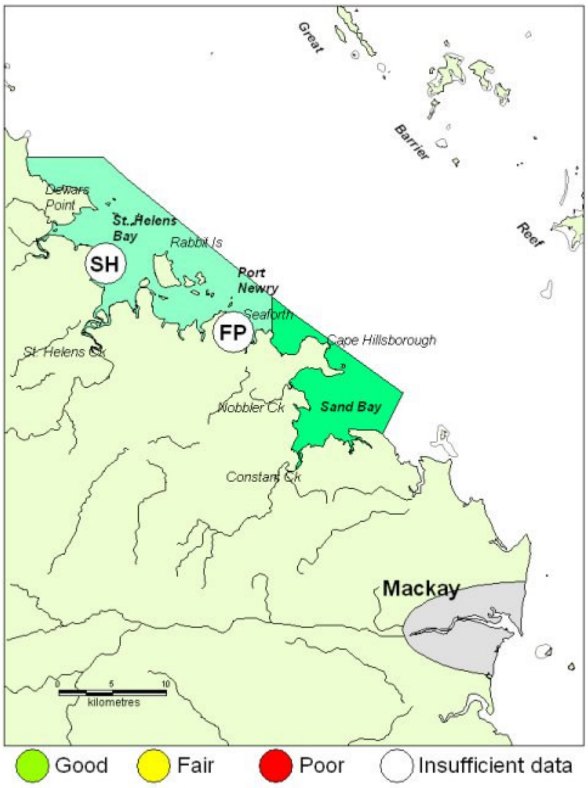

Figure 5.21 Status of seagrass meadows in the Mackay region in February 2005. SH= St Helen's Beach, FP= Finlayson Point.

St Helen's Beach

Location: Western shore of St Helen's Bay, 55km north west of Mackay.
Potential pressures: Coastal development, boat traffic, stormwater and sediment runoff.
Comments: Dugong and turtle feeding grounds. Important nursery areas for fish and prawns. A Mackay-based indigenous employment and training group, Diversity Queensland, has helped set up the site at St Helens Beach.
Site summary: The site has only been sampled once (Sep2004) and there are insufficient data to describe long-term trends. Seagrass cover is ~55% and the meadow appears healthy with very few epiphytes. Sediments at the site are very muddy.

Finlayson Point (Seaforth)

Location: Port Newry, south of Finlayson Point, 40km north west of Mackay.
Potential pressures: Coastal development, boat traffic, stormwater and sediment runoff.
Comments: Important nursery areas for fish and prawns. Dugong and turtle feeding grounds.
Site summary: The site has only been sampled once (Sep2004) and there are insufficient data to describe long-term trends. Seagrass cover is ~10%, consisting of *Zostera capricorni, Halophila ovalis* and isolated patches of *Halodule uninervis*.

Shoalwater Bay region

Shoalwater Bay is located in the southern section of the GBRWHA. The Shoalwater Bay Area covers 520,000 ha. There is relatively little coastal development. Shoalwater Bay was declared a DPA, in January 1998. The area has been reserved for defence force training since 1965. The most notable feature of the area is the massive tidal range – up to 7 metres, which in some places has the effect of creating extensive tidal banks. Rainfall is seasonal and highly variable between years. Winds are south easterly trade in the dry, cooler months of the year and light northerly winds occur during the summer monsoon season.

Seagrass meadows in and around Shoalwater Bay were first mapped during a broad scale survey in 1987 and large areas of patchy and dense seagrass were found on the shallow banks within the bay. In 1995/1996, approximately 13,000 ha of seagrass habitat were mapped in the bay by DPI&F (Lee Long *et al*., 1997). The area of subtidal seagrass habitat in Shoalwater Bay is small. Strong tidal currents and associated high water turbidity in Shoalwater Bay limit light penetration and therefore the depth to which seagrasses can grow.

Eight seagrass species (in 3 families) are found in the bay and most seagrass habitats are located on soft substrates, on intertidal flats. Four main seagrass habitats are identified: large continuous meadows on intertidal banks, patchy meadows restricted to drainage channels and pools on intertidal banks in the southern section of the bay, meadows in narrow bands on some creek banks in the southern section of the bay and inlets and a few subtidal meadows in the north eastern waters of the bay. Meadows dominated by *Zostera capricorni* are the most common and are generally more extensive and much higher above-ground biomass than other meadows. Seagrasses are present at depths from 0.7 m above MSL to 8.2 m below MSL.

Five Seagrass-Watch sites have been monitored in the Shoalwater Bay region since 2002. Seagrass meadows in the Shoalwater Bay region are rated as in a **fair** condition (as per Table 5.1), and results of monitoring indicate that seagrasses appear relatively healthy.

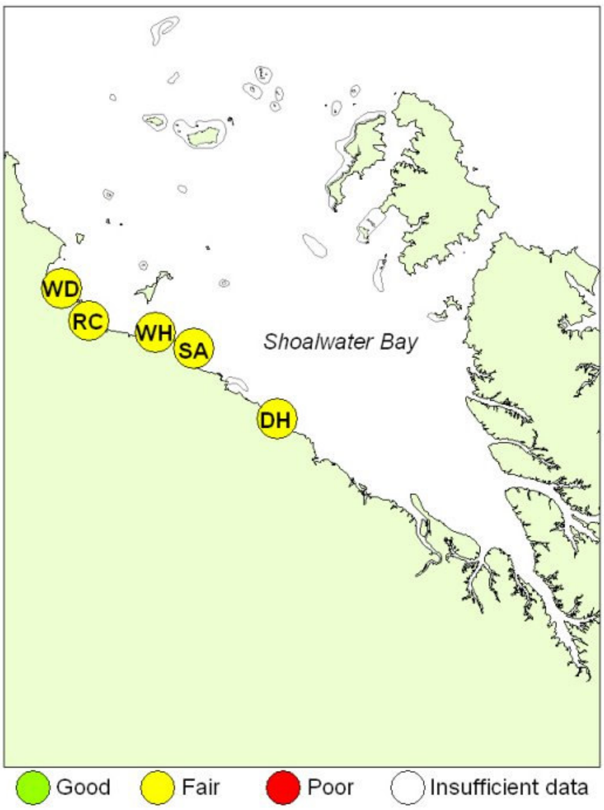

Figure 5.22 Status of seagrass meadows in the Shoalwater Bay region in February 2005. WD= Windmill Creek Headland, RC= Ross Creek, WH= Wheelans Hut, SA= Sabina Point, DH= Duck Hole Creek.

Duck Hole Creek, Ross Creek, Sabina Point, Wheelans Hut, Windmill Creek Headland

Location: Intertidal banks between McDonald Point and Raspberry Creek on the north western shores of Shoalwater Bay
Potential pressures: Freshwater and sediment runoff.
Comments: Due to Department of Defence restrictions regular access to seagrass habitats in all parts Shoalwater Bay is limited.
Site summary: The species compositions of each site were fairly similar between years with *Zostera capricorni* dominating three sites, *Halodule uninervis* dominating one while a more even mixture of the two species occurred at the fifth. Average seagrass cover ranged from ~15% to 40%, with higher cover in 2002 than in 2003.

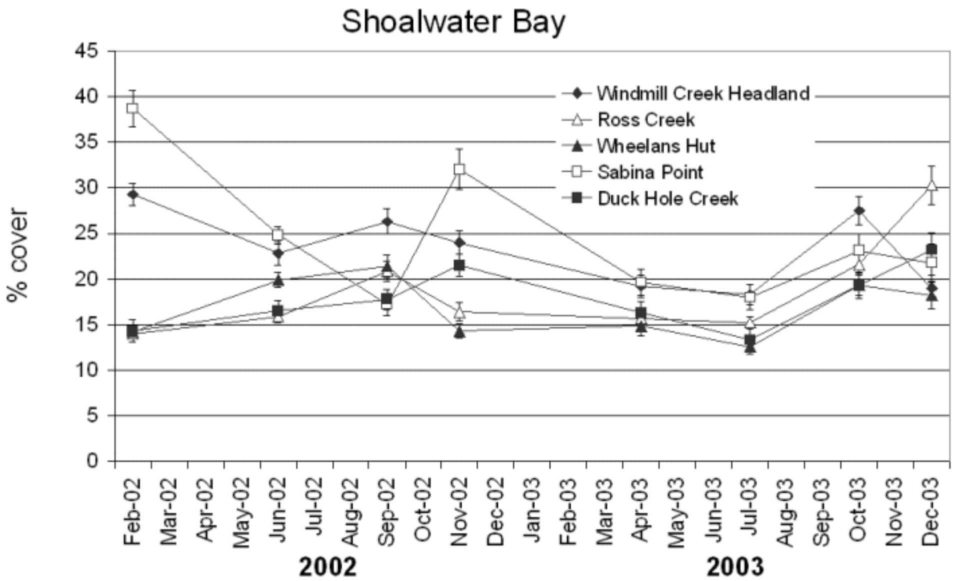

Figure 5.23 Seagrass cover (%) at the Finlayson Point sampling sites in the Mackay region.

Hervey Bay region

Hervey Bay is a large embayment (3,940 km^2) on Queensland's southern coast. Extensive intertidal banks (2,307 km^2) consisting of fine to medium grained sands fringe the landward component of the bay supporting extensive yet sparse seagrass meadows. The tidal range in this region is up to 4.1m.

Hervey Bay supports one of the largest areas of seagrass in eastern Australia. Meadows were first mapped during a broad-scale survey between Water Park Point and Hervey Bay in 1988. Seagrass distribution was estimated to be a least 1026 km^2 and mainly in large, dense meadows in the southern and western parts of the bay, extending from intertidal areas to 25 m depths in the centre of the bay.

Approximately 1000 km^2 of seagrasses in Hervey Bay was lost after two major floods and a cyclone within a 3 week period in 1992. The deeper water seagrasses died, apparently as a result of light deprivation caused by a persistent plume of turbid water that resulted from the floods and the resuspension of sediments caused by the cyclonic seas. Recovery of sub-tidal seagrasses (at depths >5m) began within two years of the initial loss, but recovery of inter-tidal seagrasses was much slower and only appeared evident after 4-5 years. The seagrasses appeared to be fully recovered in December 1998 (McKenzie *et al.*, 2000).

In December 1998 a detailed survey of Hervey Bay estimated ~2,300 km2 of seagrass in (McKenzie, 2000). Seagrass meadows extended from the intertidal and shallow subtidal waters to a depth of 32 m. The dominant (43%) deep water (>10 m) meadows in the southern section of Hervey Bay were large continuous meadows of medium-high biomass *Halophila spinulosa* with *Halophila ovalis* (high cover of drift algae). The south eastern section of the bay was generally barren substrate with isolated patches of *Halophila spinulosa/ H. ovalis/ H. decipiens*. In the south western section of the bay however, the subtidal seagrass meadows were generally patchy, medium to high biomass, *H. spinulosa* with *H. ovalis/H. decipiens* on sand down to 15 m. The shallow subtidal Dayman Bank, extending from near Urangan was covered with low biomass *H. spinulosa/ H. decipiens*. Seagrass meadows were also present on the intertidal sand banks between Burrum Heads and Eli Creek (Point Vernon). These meadows were generally low biomass *Zostera capricorni*, or *Halodule uninervis*, with *H. ovalis*. A narrow intertidal band of sparse (1-10% cover) *Z. capricorni* with *H. ovalis* was also present on the sand banks adjacent to the Esplanade from Pialba to Torquay.

The Mary River again flooded in February 1999 and produced a large freshwater plume of suspended sediments extending into Hervey Bay. Substantially reduced light conditions were measured by light meters for 19 days before returning to pre-flood levels (Ben Longstaff, UQ, Pers. Comm.). The flood had the greatest adverse effect on the intertidal and shallow subtidal seagrasses in the path of the flood plume. Seagrass had completely disappeared by November 1999 within the shallow sub-tidal monitoring area (2–10 m depth below MSL). Deepwater seagrasses in Hervey Bay within the path of the flood plume also declined significantly in abundance six months after the impact and remained significantly lower than outside the impact area after nine months.

Monitoring at Seagrass-Watch sites within the Booral wetlands found initial re-colonisation of seagrass occurring in November 2000, 21 months post-flood. Full recovery of meadows to pre-flood cover values (~20-40%) occurred by August 2002, 30 months post-flood. Monitoring sites exhibited seasonal tends in abundances with highest cover in November and lowest seagrass cover post-summer from April to June. This typical seasonal response coupled with a trend of increasing seagrass cover indicates a post-flood recovery.

In February 2002, deepwater seagrass abundances at monitoring sites within the impacted area had recovered to near pre-flood levels. The areas of seagrass that showed little recovery were the shallow sub-tidal seagrasses (2-4 m) immediately adjacent to the city of Hervey Bay. Only a few isolated patches of seagrass had recovered off the northern tip of the bank in February 2002 (see also Campbell and McKenzie, 2004).

Five Seagrass-Watch locations are have been monitored in the Hervey Bay region since 1999. Seagrass meadows in the Hervey Bay region are rated as in a fairly-good condition (as per Table 5.1).

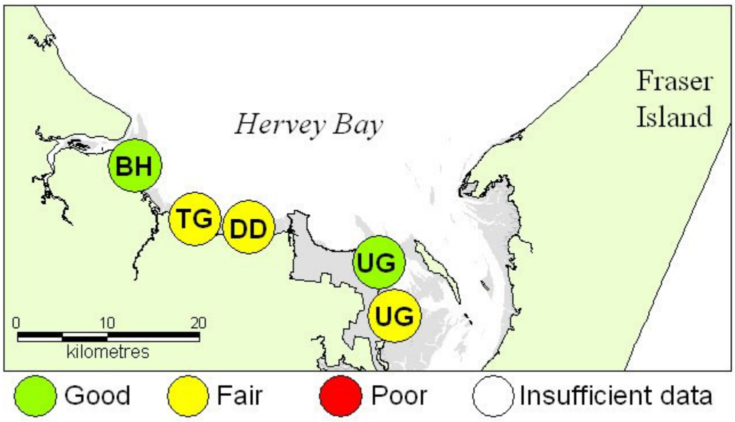

Figure 5.24 Status of seagrass meadows in the Hervey Bay region in February 2005. BH= Burrum Heads, TG= Toogoom, DD= Dundowran, UG= Urangan & Booral.

Burrum Heads

Location: Mouth of the Burrum River on the western shore of Hervey Bay.
Potential pressures: Urban development, stormwater and land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: The dominant seagrass species are *Halodule uninervis* (narrow form) and *Halophila ovalis*. Species composition varied over the monitoring period with losses of both *Z. capricorni* and *H. uninervis* due to burial by mobile sediments. From May 2000 seagrass cover improved and seasonal patterns emerged; the cover at BH2 was significantly higher than the other sites. Cover of associated macroalgae is generally low with episodic blooms, while epiphytes increase over late winter and spring and decline over summer months. The sediment grain size has become coarser over time, possibly improving water clarity. Dugong feeding trails were found at Burrum Heads (BH1) and were most abundant in May and August.

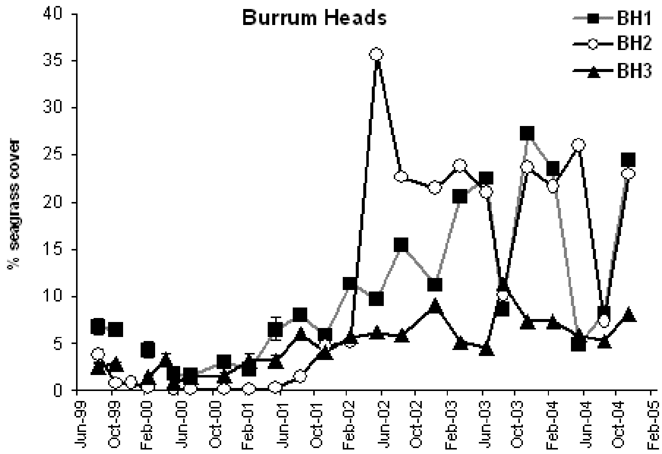

Figure 5.25 Seagrass cover (%) at the Burrum Heads sampling sites in the Hervey Bay region.

Dundowran

Location: On the western coastline of Hervey Bay.
Potential pressures: Urban development, stormwater and land runoff.
Comments: Dugong and turtle feeding grounds.

Site summary: The dominant species are *Halodule uninervis* (narrow leaf morphology) and *Halophila ovalis* and the species composition has remained stable. Seagrass cover has remained low, with no significant difference between sites. The decline in seagrass cover from August 1999 to May 2000 was due to burial by mobile sediments. Macroalgae and epiphytes showed episodic blooms, generally mid year. The sediment grain size has become slightly coarser across all sites over the monitoring period. The sites are influenced by wave action and tidal flows with high sediment movement observed throughout the monitoring period. A likely cause for change in seagrass cover at DD2 was smothering by sand movement. Nutrient sources from agricultural lands, unsewered developments and sewage inputs occur in proximity to seagrass sites at DD3.

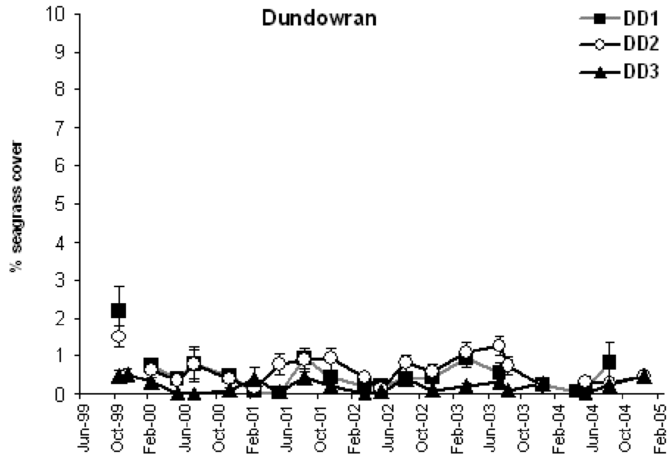

Figure 5.26 Seagrass cover (%) at the Dundowran sampling sites in the Hervey Bay region.

Toogoom

Location: On the western coastline of Hervey Bay.
Potential pressures: Urban development, unsewered stormwater and agricultural land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: The dominant species were *Halodule uninervis* (narrow form) and *Halophila ovalis*. Species composition varied over the monitoring period with losses of both *Z. capricorni* and *H. uninervis* due to burial by mobile sediments from August 1999 to May 2000. Seagrass abundance has not recovered to mid 1999 values, however is showing seasonal trends with significant increases each spring. Cover of associated macroalgae is generally low with occasional episodic blooms. Epiphytes increased dramatically in late 2002 and early 2003, however declined afterwards. Sediment grain size has remained stable with a prevalence of fine sands. The sites are influenced by wave action and tidal flows with high sediment movement observed throughout the monitoring period.

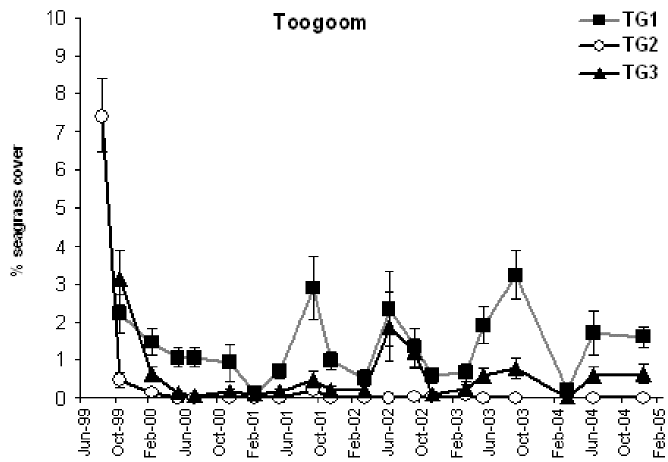

Figure 5.27 Seagrass cover (%) at the Toogoom sampling sites in the Hervey Bay region.

Urangan

Location: Adjacent to the marina and close to the Mary River mouth.
Potenatial pressures: Urban and marina development, input of treated sewage, stormwater and land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: The site is dominated by *Zostera capricorni*, with more *Halophila ovalis* early on and some *Halodule uninervis* now colonising. Following a major flood in 1999, seagrass was absent (0% cover) from August 1999 to May 2000. In July 2000 seedlings of *Zostera capricorni* appeared. Since then abundance has recovered significantly. A sudden and dramatic decline in early 2004 caused some concern, however abundance has since recovered. While macroalgal cover is relatively insignificant, epiphyte blooms regularly occur, suggesting acute high nutrient inputs. Sediment grain size has changed little over the monitoring period. Dugong feeding was absent until late 2001, coinciding with seagrass recovery, and feeding trails are now regularly observed.

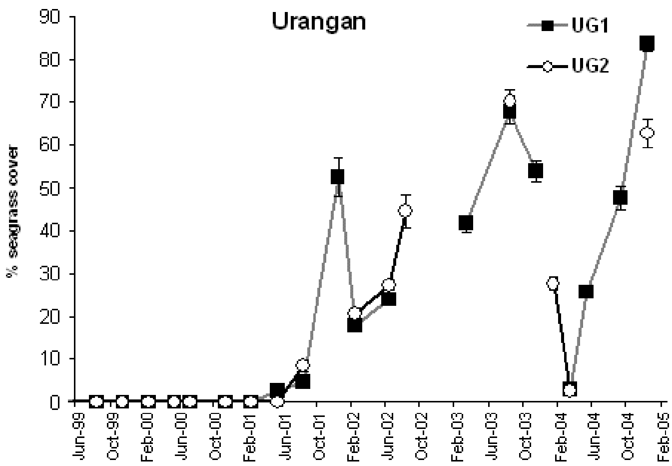

Figure 5.28 Seagrass cover (%) at the Urangan sampling sites in the Hervey Bay region.

Booral

Location: Adjacent to the mouth of the Mary River and Hervey Bay city.
Potential pressures: Urban and marina development, stormwater and land runoff.
Comments: Dugong and turtle feeding grounds.
Site summary: Seagrass abundance increased significantly since monitoring began in late 2000. Declines in both early 2003 and 2004 caused some concern, however abundance has recovered and variations are possibly seasonal. Episodic algal blooms have occurred in late 2001 and 2002, however, subsequently declined. Epiphyte abundance has continued to increase closely correlated with seagrass abundance.

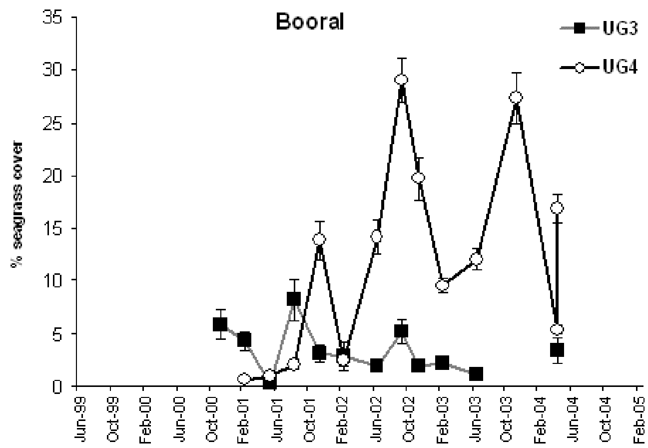

Figure 5.29 Seagrass cover (%) at the Booral sampling sites in the Hervey Bay region.

5.3 Water quality influences on seagrass

The most common cause of seagrass loss is the reduction of light availability due to chronic increases in dissolved nutrients which leads to proliferation of algae thereby reducing the amount of light reaching the seagrass (e.g. phytoplankton, macroalgae or algal epiphytes on seagrass leaves and stems), or chronic and pulsed increases in suspended sediments and particles leading to increased turbidity (Schaffelke *et al.*, 2005). In addition, changes of sediment characteristics may also play a critical role in seagrasses loss (refer also to Section 7). There is 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. These influences are interlinked in complex ways and it is expected that the Reef Plan MMP will support the process of understanding and quantifying these links.

Conclusions

This section provides a snapshot of the present status of seagrass meadows from seven regions as reported by Seagrass-Watch community members. Most seagrass meadows have been relatively stable over the last decade. Some regions, for example the Whitsunday region, are showing declines and the Seagrass-Watch program is monitoring and providing feedback to the relevant authorities on these changes. It is expected to find changes in seagrass meadow parameters as they respond to changes in weather, storms and water movement etc. A consistent pattern of decline is more of a concern. To provide a better quantitative response a longer time series is requires and additional environmental parameters need to be measured. The Reef Plan MMP is providing support for this to occur.

6. Bioaccumulation monitoring: mud crabs

Munro Mortimer, Queensland Environment Protection Agency and Andrew Negri, Australian Institute of Marine Science

Introduction

The direct measurement of pollutants is important for the assessment of river water quality and the estimation of export of e.g. herbicides and pesticides to the marine environment. However, these measurements do not provide an indication of how and to what levels pollution can enter and persist within estuarine and marine food webs. Measuring the body burdens of pollutants in key ecosystem organisms can provide this information, along with a potentially sensitive, early indication of the presence of some pollutants.

Recent research has demonstrated that several keystone marine organisms of the Great Barrier Reef including: corals (Jones and Kerswell, 2003; Negri *et al*., 2005), seagrass (Haynes *et al*., 2000b), mangroves (Bell and Duke, 2005) and crustose coralline algae (Harrington *et al*., 2005) are all sensitive to herbicides that are currently applied within Great Barrier Reef Catchments. Fish from tropical estuaries of the Great Barrier Reef are also susceptible to contemporary insecticides such as chlorpyrifos (Humphrey *et al*., 2004). Furthermore, inorganic pollutants including copper can affect sensitive life history transitions in corals such as fertilisation and settlement (Reichelt-Brushett and Harrison, 2000; Negri and Heyward, 2001).

Measuring pollutants within organisms is important to prove exposure in the field, and bioaccumulation levels of some pollutants can provide environmental managers with a proxy estimate for environmental pollution levels (Phillips and Rainbow, 1993). Measuring pollutants in macroinvertebrates may also alert management to otherwise unrecognised pollution levels that may accumulate up food webs.

Bioaccumulation of metals and pesticides in marine organisms of the Great Barrier Reef was comprehensively reviewed by Haynes and Johnstone (2000). Since then several relevant reports of metals and organic pollutants in Great Barrier Reef biota have emerged:

- The antifoulant herbicide Irgarol 1051 was discovered in seagrass sampled from 9 mostly urban inshore sites along the Queensland coast, including 4 sites within the Great Barrier Reef (Scarlett *et al*., 1999).
- The agricultural herbicide diuron was also detected in seagrass at inshore sites close to Cairns and Cardwell which receive both urban and agricultural runoff, as well as a site within Moreton Bay (Haynes *et al*., 2000a).
- An investigation into significant mangrove dieback within the Pioneer River catchment revealed the herbicide diuron in some mangrove leaf samples of diseased individuals (Duke *et al*., 2005).
- Stranded dugongs from the length of the Great Barrier Reef were analysed for pollutants and many individuals found to contain trace metals along with banned pesticides, DDT, dieldrin and heptachlor (or their breakdown products) (Haynes *et al*., 2005). The concentrations of these pollutants however were not considered detrimental to dugong health.
- Polychlorinated dibenzo-p-dioxins (PCDD) were identified in seagrass and dugong samples taken from 5 inshore sties between Mackay and Princess Charlotte Bay (McLachlan *et al*., 2001). The primary source of PCDDs remains unclear and may be natural.

- The filter-feeding barnacle species *Balanus amphitrite* was used successfully to monitor for cadmium within Ross Creek, which flows into the Great Barrier Reef (da Silva *et al.*, 2004).
- Cores from Great Barrier Reef massive corals provided a 10 year proxy records of dissolved concentrations of Sr, Ba and Mn when analysed by thermal ionization mass spectrometry and laser ablation inductively coupled plasma mass spectrometry (Alibert *et al.*, 2003).
- Two species of oysters were transplanted along sites in the Great Barrier Reef between catchments of the Herbert and Burdekin Rivers (Olivier *et al.*, 2002). The species used were demonstrated to be sensitive indicators for zinc and cadmium.
- Webster *et al.*, (2001) also demonstrated that a filter-feeding Great Barrier Reef sponge species is capable of accumulating high levels of copper in laboratory trials and may be a useful alternative species for bioaccumulation monitoring.

Crustaceans, and crabs in particular, are widely recognised as useful species for biomonitoring (Phillips and Rainbow, 1993). Most of the data collected on bioaccumulation in crustacea is concerned with metal concentrations in barnacles, which are sessile but provide little material for analysis of organic pollutants such as pesticides. The mud crab *Scylla serrata* was proposed as a biomonitor species for the current Reef Plan MMP because of its capacity to bioaccumulate a range of pollutants, and its significance as a target species for subsistence, commercial and recreational fisheries. It is readily collectable from most of the ten priority river systems, has a limited territorial range and is large enough to provide ample tissue for chemical analysis. There is limited data available on pollutant levels in the mud crab *S. serrata* from Queensland waters, and indeed very little work has been done on a systematic basis to monitor pollutants other than nutrients in tropical Queensland waters.

6.1 Results from relevant programs

Metals and metalloids

The Queensland Environmental Protection Agency (EPA) conducted a project between 1994 and 1998 analysing both muscle and hepatopancreas tissues from mud crabs (*S. serrata*) from the Brisbane River, including a major tributary (Oxley Creek), Maroochy and Pine Rivers in south-east Queensland and Port Curtis for a range of metals and metalloids (Mortimer 2000). Mud crab data from these collections are summarised in Figure 6.1 and Table 6.1. Most trace metals and metalloids preferentially accumulated in the hepatopancreas of *S. serrata* from these collections undertaken around Brisbane and the Southern Great Barrier Reef coast (Table 6.1). The highest levels of Pb and Sn were detected in the Brisbane River – fed by an urban catchment.

Mortimer (2000) also performed a survey of metals in the much smaller intertidal burrowing crab *Australoplax tridentate* at 22 rivers along the Queensland coast. In this case whole crabs were extracted, so the total metal concentrations are not directly comparable with those in *S. serrata*. Three of the sites (Burnett, Fitzroy and Johnstone Rivers) sampled by Mortimer, (2000) will also be sampled in the present study. The Johnstone stood out in terms whole body metal content with highest or equal highest levels of Cr, Fe, Ni, Ti and V (data not shown). The Johnstone also contained relatively high levels of Cu and Zn, lower only than sites affected by urban/industrial Brisbane and Townsville catchments.

Both muscle and hepatopancreas tissues from mud crabs (*S. serrata*) sampled from Port Curtis (Gladstone), the Fitzroy River, and the Burdekin River (Ayr) were analysed for a range of metals and metalloids as part of a study into shell disease in mud crabs conducted between 1998 and 2000 (Centre for Environmental Management, Central Queensland

University, Gladstone; Andersen and Norton, 2001). The results for mud crabs from these collections are summarised in Table 6.2. *Scylla serrata* hepatopancreas collected from Port Curtis contained significantly higher levels of total metals, in particular Cu and Zn, than those collected from the control site in the Burdekin River (Table 6.2). Input from industry in Port Curtis is a potential source of the elevated metal contamination. The control site near Ayr in the Burdekin region is primarily rural. Crabs from this region may be exposed to a different suite of pollutants, including pesticides (these were not analysed). Relatively low levels of Cu and Zn within the sediments at Port Curtis indicated that the *S. serrata* may have accumulated the elevated concentrations via the food chain (Andersen and Norton, 2001). A range of metals were monitored over a two year period but high within-site variability masked any significant changes between years. Results from the Burdekin collection may be comparable with those collected from the same site in the present monitoring program. Both of the above studies will provide good baselines for comparison with data collected in the present monitoring program.

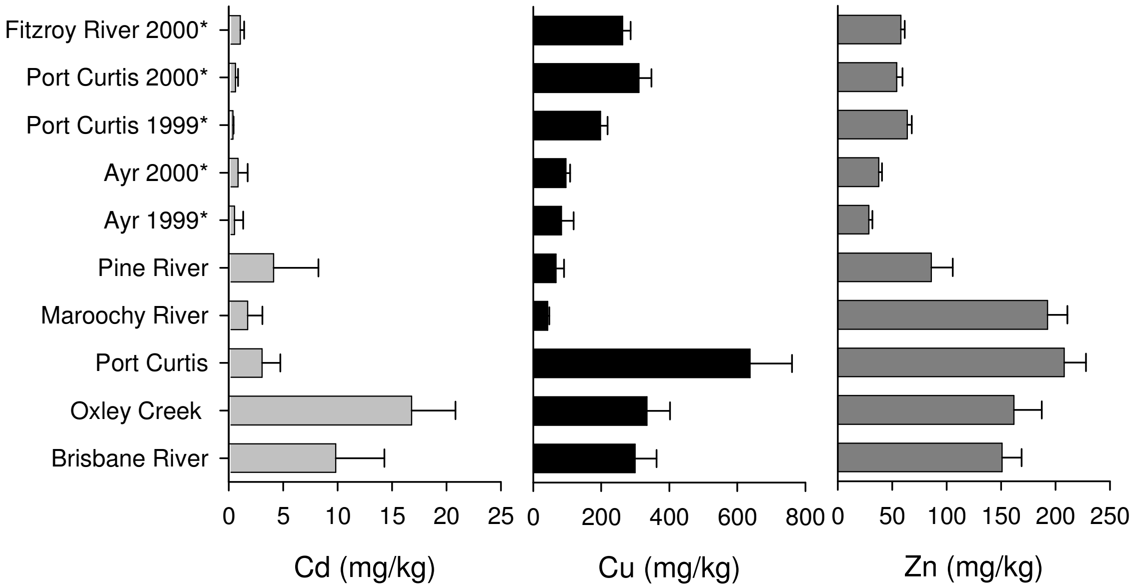

Figure 6.1. Selected average trace metal concentrations in Scylla serrata *hepatopancreas (+ SE) from 7 locations on the Queensland coast (Mortimer, 2000; Andersen and Norton, 2001). * indicates wet weight*

Muscle tissue from boiled *S. serrata* collected from Boigu Island off Papua New Guinea was analysed for a suite of trace metals as part of the Torres Strait Baseline Study by the GBRMPA (Gladstone, 1996). This study was undertaken to gauge the potential effects of mining operations in the Fly River and focussed upon primarily seafood contamination. The summary data for mud crabs from these collections are summarised in Table 6.3. This study focussed heavily upon seafood contamination and only the cooked edible muscle of five *S. serrata* specimens from the Boigu Island off Papua New Guinea were analysed (Gladstone, 1996). These results will not be comparable with those from the present monitoring program.

Insecticides and herbicides

Scylla serrata were sampled from both the Daintree and Johnstone Rivers by DPI&F personnel between 1990 and 1993 and muscle tissue (claw meat) analysed for a range of insecticides and herbicides. A report from this study was published by Russell *et al*. (1996). The paper presents data from bivalve and finfish species sampled as part of the study, but only sparse data are provided for crabs. The herbicides atrazine and 2,4D were detected in a small number of samples of *S. serrata* (Table 6.4). Although collected from the Johnstone River, which is a sampling site in the current Reef Plan MMP, scarce sampling information and low replication will make comparisons difficult.

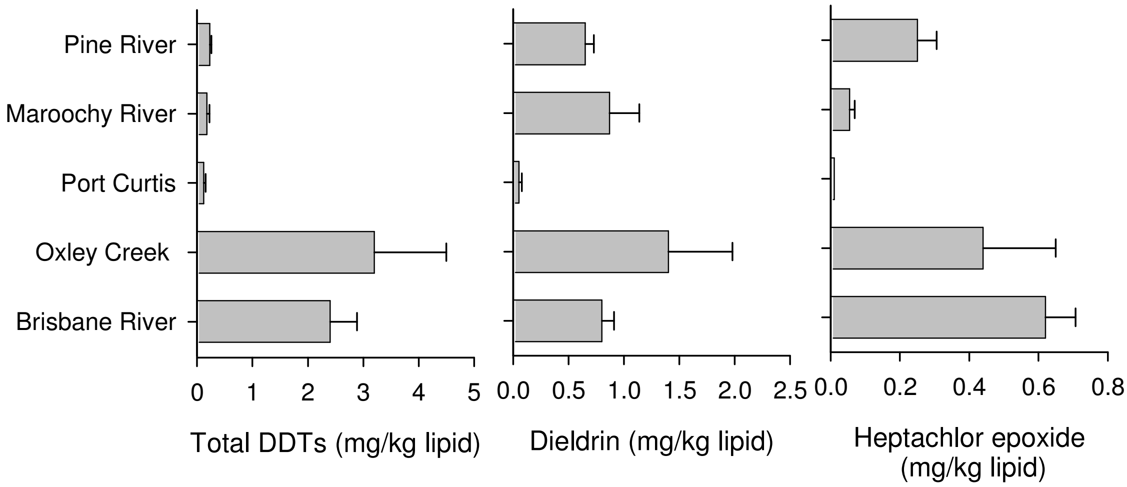

Figure 6.2. Chlorinated pesticides in Scylla serrata *hepatopancreas (average concentrations + SE) from 5 locations on the Queensland coast (Mortimer, 2000).*

Both muscle and hepatopancreas tissues from mud crabs sampled by Queensland EPA personnel from the Brisbane River, including a major tributory (Oxley Creek), two other rivers in south-east Queensland (Maroochy and Pine Rivers), and Port Curtis between 1994 and 1998 were analysed for a range of organophosphate and organochlorine insecticides (Mortimer 2000). The project also included analysis of insecticide concentrations in the intertidal burrowing crab *Australoplax tridentata*. The summary data for mud crabs from these collections are presented in Figure 6.2 and Table 6.5. The banned organochlorine pesticides DDTs and dieldrin were the only pesticides detected in *S. serrata* from southern Queensland sites, along with heptachlor epoxide, the breakdown product of the banned organochlorine heptachlor (Table 6.5). Each of these insecticides are extremely persistent and are likely to have resulted from historical use on crops, in industry and from urban application.

Mortimer, (2000) performed a survey of pesticides in composite samples (multiple crabs/analysis) of the small intertidal burrowing crab *A. tridentate* at 19 rivers along the Queensland coast. The data is not directly comparable to *S. serrata* as whole body samples were extracted, however when expressed on the basis of lipid content, some of confounding tissue difference effects should be minimised (Phillips and Rainbow, 1993). DDTs, Dieldrin and Heptachlor epoxide were each detected the Johnstone River crabs (0.053, 0.28 and 0.21 mg/kg respectively). Crabs from the Burnett contained both DDTs (0.38 mg/kg) and Dieldrin (5.5 mg/kg) and crabs from the Fitzroy River Dieldrin (0.84 mg/kg) and Heptachlor epoxide (0.048 mg/kg).

The current suite of organophosphates and carbamates are much less persistent and may be more difficult to detect in biota. Larger sample sizes and improved sample preparation procedures may enable detection of some of the currently applied pesticides. The sampling sites in the Reef Plan MMP are downstream of rural catchments and this may result in higher exposure and accumulation levels of relevant pesticides. The pesticides and metals targeted by the present monitoring program are listed in Table 6.6. Most of the pesticides targeted in the Reef Plan MMP are not naturally occurring so their appearance in crab tissue would positively reveal anthropogenic sources. Low levels of pesticides were detected in crab hepatopancreas in previous studies. Therefore, crab samples in the currently being implemented Reef Plan MMP should initially be pooled in order to obtain enough biomass to detect pesticides currently in use in catchments adjoining the GBRWHA. Sampling sites in this program are shown in Figure 6.3.

In summary, there are several advantages and disadvantages of using mud crabs for bioaccumulation monitoring. The advantages are:
- Crabs have been demonstrated to bioaccumulate a range of pollutants such as insecticides and metals
- Crabs are the largest macroinvertebrate in rivers and provide ample material for extraction, resulting in good detection limits
- Crabs have limited mobility and are not likely to travel between catchments
- Some compounds such as the very persistent organochlorines may be concentrate to higher levels in crabs than most other species/samples
- Crabs are an iconic species in tropical Queensland and are heavily fished both recreationally and commercially
- The data will be comparable with other data sets from Queensland (Mortimer, 2000).

The disadvantages of using mud crabs for bioaccumulation monitoring are:
- Crabs will not accumulate all pesticides or metals equally
- Relatively large variability in some pollutants has been recorded between individuals from the same site
- Unknown local factors may confound comparisons between rivers.

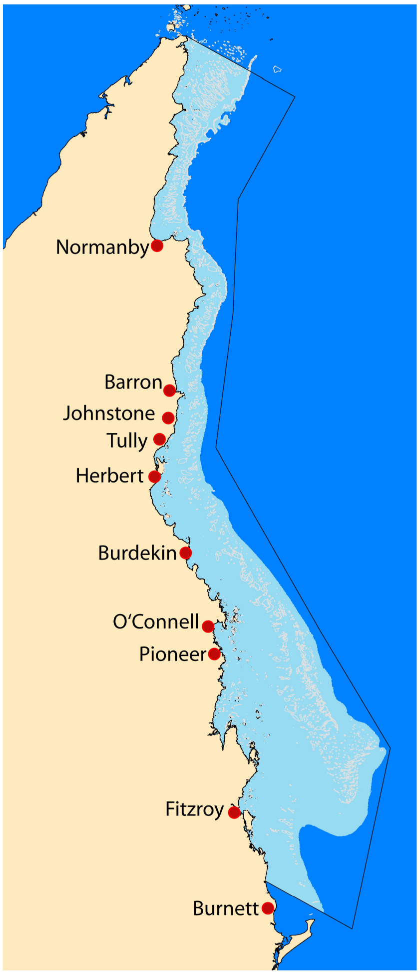

Figure 6.3 Sampling sites in the pollutant bioaccumulation monitoring using Scylla serrata *in the current Reef Plan MMP.*

Conclusions

Although rural runoff presents a potentially significant source of pesticides and trace metals to the Great Barrier Reef, no comprehensive survey of bioaccumulation within Great Barrier Reef Catchments has been performed previously. Several organisms such as oysters and seagrass can accumulate detectable levels of land-sourced pollution, but none of these organisms are present within all of the river systems targeted for monitoring in the Reef Plan MMP. The data from previous bioaccumulation studies that employed *S. serrata* in other Queensland catchments suggests that this species of crab would prove an appropriate and useful species for bioaccumulation monitoring in rivers and estuaries. Existing information highlights high within-site variability of metals and the need for high replication when comparing metals between sites and years. Trace metals also occur naturally in the environment; therefore it is difficult to positively link high levels in biota with potential anthropogenic sources. The pesticides detected in *S. serrata* from southern Queensland were primarily persistent chlorinated insecticides, which are banned from use.

Table 6.1 Trace metals and metalloids in body muscle (m) and hepatopancreas (h) of mud crabs S. serrata sampled between 1994 and 1988. Mean and (standard error) in mg/kg dry weight. n= sample size. Data from Mortimer (2000).

Location	**n**		**As**	**Cd**	**Co**	**Cr**	**Cu**	**Fe**	**Hg**	**Mn**	**Mo**
Brisbane River	16	h	12.5 (1.82)	9.84 (4.47)	0.913 (0.181)	0.842 (0.150)	300 (62.8)	124 (18.8)	0.294 (.0528)	17.9 (2.92)	0.529 (0.124)
	16	m	17.9 (1.81)	0.337 (0.112)	0.0681 (0.0150)	0.416 (0.0307)	67.5 (4.49)	19.9 (1.55)	0.263 (0.0551)	6.7 (1.44)	0.0429 (0.0105)
Oxley Creek (BRT)	8	h	4.4 (0.609)	16.8 (4.03)	3.69 (1.22)	0.451 (0.0350)	334 (68.3)	512 (72.1)	<0.001	30.8 (9.20)	0.838 (0.214)
	8	m	8.61 (3.75)	0.271 (0.113)	0.117 (0.0226)	0.222 (0.0522)	42.9 (5.95)	105 (16.3)	<0.001	11.7 (2.90)	0.0323 (0.00508)
Port Curtis	10	h	17.4 (2.71)	3.05 (1.67)	2.65 (0.463)	0.646 (0.0498)	637 (124)	186 (16.5)	0.0924 (.0512)	28.0 (12.2)	0.836 (0.149)
	10	m	20.0 (2.37)	0.0761 (.0143)	0.704 (0.382)	0.437 (0.0364)	194 (121)	33.8 (9.57)	0.0654 (0.0357)	8.20 (2.07)	0.0730 (0.0201)
Maroochy River	35	h	9.96 (0.782)	1.73 (0.136)	3.30 (1.08)	0.776 (0.0675)	42.6 (4.74)	272 (24.6)	0.0659 (0.00983)	45.7 (11.6)	0.955 (0.209)
	35	m	9.14 (0.895)	0.0302 (0.00953)	0.285 (0.0784)	0.347 (0.0336)	55.5 (5.30)	21.8 (1.79)	0.122 (0.0194)	12.2 (2.75)	0.0279 (0.00364)
Pine River	8	h	7.86 (0.251)	4.12 (0.545)	1.54 (0.228)	0.163 (0.0358)	67.4 (23.0)	182 (25.8)	<0.001	48.9 (17.6)	0.906 (0.184)
	8	m	7.55 (1.24)	<0.001	0.0887 (0.00958)	0.0634 (0.00525)	37.6 (2.62)	17.8 (4.85)	<0.001	10.5 (2.98)	0.0784 (0.0424)

Location	**n**		**Ni**	**Pb**	**Sb**	**Se**	**Sn**	**Ti**	**V**	**Zn**
Brisbane River	16	h	8.37 (1.04)	0.768 (0.314)	0.0384 (.0109)	6.38 (0.742)	0.997 (0.228)	2.36 (0.405)	32.5 (16.3)	151 (17.9)
	16	m	4.91 (0.490)	0.162 (0.0337)	0.00494 (0.000193)	6.56 (0.833)	0.290 (0.0566)	2.41 (0.285)	12.2 (5.99)	204 (9.45)
Oxley Creek (BRT)	8	h	12.8 (1.75)	0.117 (0.0172)	0.0278 (0.00674)	1.39 (0.446)	0.0968 (0.446)	NQ	0.473 (0.0576)	162 (25.4)
	8	m	2.43 (0.420)	0.0284 (0.00491)	0.00225 (0.00168)	2.50 (0.500)	0.0848 (0.0692)	NQ	0.145 (0.0226)	183 (10.2)
Port Curtis	10	h	9.61 (3.46)	0.200 (0.0266)	0.0455 (0.00479)	11.4 (1.20)	0.124 (0.0158)	54.8 (11.2)	1.26 (0.134)	208 (20.0)
	10	m	3.96 (0.0201)	0.0864 (0.0157)	0.0122 (0.000892)	9.31 (0.956)	0.0447 (0.0113)	23.9 (2.68)	0.291 (0.0624)	170 (12.3)
Maroochy River	10	h	5.99 (1.38)	0.143 (0.0254)	0.0283 (0.00745)	0.475 (0.372)	<0.01	0.719 (0.159)	0.434 (0.0619)	193 (18.2)
	10	m	3.61 (0.739)	0.0545 (0.00922)	<0.01	2.78 (0.258)	0.0196 (0.00916)	0.635 (0.112)	0.139 (0.0272)	236 (11.5)
Pine River	8	h	15.7 (2.78)	0.0436 (0.0177)	<0.001	6.16 (0.485)	<0.001	1.73 (0.456)	9.99 (1.31)	86 (19.6)
	8	m	7.16 (0.890)	0.0780 (0.0249)	<0.001	2.62 (0.152)	<0.001	0.420 (0.110)	7.84 (0.214)	172 (16.6)

BRT = Brisbane River tributary,
< values are limits of detection,
NQ = element not quantified

Table 6.2 Trace metals and metalloids in hepatopancreas tissue from mud crabs S. serrata sampled between 1999 and 2000. Mean (and SE) in mg/kg wet weight. n= sample size. Data from Andersen and Norton, (2001) and Andersen, (unpublished).

Location	n	Ag	Al	As	Ba	Cd	Ce	Co	Cr
Ayr 1999	10	0.635 (0.139)	1.76 (0.347)	11.7 (1.03)	0.256 (0.0645)	0.529 (0.0776)	0.097 (0.0143)	0.402 (0.0403)	0.263 (0.0563)
Ayr 2000	30	0.94 (0.100)	2.29 (0.170)	15.2 (1.54)	0.283 (0.0548)	0.859 (0.120)	0.109 (0.0193)	0.637 (0.0579)	0.215 (0.0200)
Port Curtis 1999	20	0.874 (0.0753)	2.11 (0.194)	8.08 (0.600)	0.374 (0.0755)	0.366 (0.0674)	0.092 (0.0124)	1.11 (0.128)	0.151 (0.0440)
Port Curtis 2000	45	1.16 (0.334)	3.00 (1.37)	7.40 (0.472)	0.290 (0.0903)	0.615 (0.253)	0.351 (0.118)	1.13 (0.0.380)	0.551 (0.181)
Fitzroy River 2000	9	2.34 (0.21)	1.24 (0.17)	1.24 (0.17)	0.46 (0.07)	1.07 (0.33)	0.18 (0.02)	1.02 (0.16)	0.23 (0.02)

Location	n	Cu	Fe	Hg	La	Mn	Mo	Ni	Pb
Ayr 1999	10	83.0 (36.1)	80.0 (11.7)	0.065 (.015)	0.069 (0.00971)	2.23 (0.362)	0.453 (0.0514)	0.62 (0.145)	0.302 (0.0882)
Ayr 2000	30	96 (11.8)	99.1 (13.6)	0.122 (0.0161)	0.089 (0.0147)	6.66 (0.752)	0.718 (0.0669)	1.10 (0.161)	0.055 (0.00348)
Port Curtis 1999	20	198 (20.5)	95.0 (11.4)	0.054 (0.0035)	0.065 (0.0135)	13.2 (4.48)	0.306 (0.0685)	1.90 (0.593)	0.156 (0.0240)
Port Curtis 2000	45	311 (36.6)	62.4 (7.71)	0.085 (0.0255)	0.250 (0.870)	6.77 (3.09)	0.0925 (0.436)	1.26 (0.499)	0.0684 (0.0184)
Fitzroy River 2000	9	263 (23)	50.7 (9.8)	0.09 (0.10)	0.14 (0.02)	4.07 (0.67)	0.62 (0.08)	3.01 (0.56)	0.05 (0.00)

Table 2 (continued)

Location	n	Rb	Sn	Sr	U	V	Zn
Ayr 1999	10	0.742 (0.0317)	0.058 (0.00800)	8.29 (.778)	0.149 (0.0360)	0.157(0.362)	28.53 (3.36)
Ayr 2000	30	<0.1	<0.1	52.9 (7.68)	0.182 (0.0566)	0.327 (0.0340)	37.7 (2.84)
Port Curtis 1999	20	0.641 (0.0269)	0.085 (0.0136)	29.1 (6.87)	0.314 (0.0627)	0.254 (0.0166)	63.9 (4.20)
Port Curtis 2000	45	<0.1	0.076 (0.0305)	30.6 (3.75)	0.342 (0.176)	0.529 (0.204)	54.2 (5.21)
Fitzroy River 2000	9	0.05 (0.00)	0.05 (0.00)	36.7 (9.1)	0.19 (0.04)	0.25 (0.01)	58.0 (3.5)

Table 6.3 Trace metals and metalloids in boiled muscle tissue from mud crabs S. serrata sampled between 1992 and 1993. Mean (and SE) in mg/kg wet weight. n= sample size. Data from Gladstone, (1996).

Location	n	As	Cd	Cu	Hg	Pb	Se	Zn
Boigu Is (PNG)	5	2.08 (0.62)	0.02 (0.01)	7.66 (2.64)	0.02 (0.01)	0.02 (0.01)	0.68 (0.43)	30.2 (4.6)

Table 6.4 Herbicide and insecticide reported in mud crabs S. serrata *sampled between 1990 and 1993 from the Johnstone and Daintree Rivers. Data from Russell* et al.*, (1996).*

Location	**Report**
Daintree	Small amounts dieldrin and DDT (concentrations not stated)
	2,4D in at least 1 sample (concentrations not stated)
Johnstone	Small amounts dieldrin and DDT (concentrations not stated)
	Atrazine in 2 crabs (20 •g/kg wet weight)
	2,4D in at least 1 sample (concentrations not stated)

Table 6.5 Pesticide concentrations in body muscle (m) and hepatopancreas (h) tissue of the mud crab S. serrata sampled between 1994 and 1988. Mean and (standard error) in mg/kg lipid weight. n= sample size. Data from Mortimer, (2000).

Location	**n**		**Total DDTs**[a]	**dieldrin**	**heptachlor epoxide**
Brisbane River	16	h	2.4 (0.49)	0.80 (0.11)	0.62 (0.087)
	16	m	2.8 (0.42)	0.66 (0.17)	0.43 (0.055)
Oxley Creek (BRT)	2 composites of 4	h	3.2 (1.3)	1.4 (0.58)	0.44 (0.21)
	8	m	0.67 (0.19)	0.40 (0.24)	<0.02
Port Curtis	10	h	0.12 (0.038)	0.053 (0.025)	<0.03
	10	m	<0.03	<0.03	<0.03
Maroochy River	35	h	0.18 (0.043)	0.87 (0.27)	0.054 (0.015)
	35	m	<0.03	0.45 (0.15)	<0.02
Pine River	8	h	0.23 (0.031)	0.65 (0.079)	0.25 (0.056)
	8	m	0.029 (0.0061)	0.026 (0.0035)	0.018 (0.011)

[a]Mostly the metabolite DDE,
BRT = Brisbane River tributary, < values are limits of detection

Table 6.6. Preliminary list of target pesticides and trace metals to be analysed in the present Reef Plan MMP Bioaccumulation Monitoring Program.

Pesticides	Metals
Aldrin	Al
Ametryn	
Atrazine	As
Chlordane cis	Ba
Chlordane Trans	Be
Chlordene	Ca
Chlordene Epoxide	Cd
DDD (op)	Co
DDD (pp)	Cr
DDE (op)	Cu
DDE (pp)	Fe
DDT (op)	Hg
DDT (pp)	Mg
Total DDT	Mn
Dicofol	Mo
Diuron	Ni
Dieldrin	Pb
Endosulfan alpha	Sn
Endosulfan beta	Sr
Endosulfan Sulphate	V
Endosulfan Ether	Zn
Endosulfan Lactone	
Total Endosulfan	
Endrin	
Endrin aldehyde	
HCB	
HCH alpha	
HCH beta	
HCH delta	
Heptachlor	
Heptachlor Epoxide	
Irgarol 1051	
Lindane	
Methoxychlor	
Nonachlor trans	
Oxychlordane	
Simazine	
Tebuthiuron	

7. Impacts of water quality changes on inshore marine ecosystems

There is now abundant evidence, primarily from locations outside of Australia, that the overall health of coral reef and seagrass ecosystems are affected by the quality of water in which they live. Poor water quality results in the loss or displacement of dominant or desirable species, reductions in coral or seagrass cover, loss of ecosystem amenity value, and in extreme cases, the destruction of the physical structure of the ecosystem.

Not surprisingly, the best documented cases and clearest relationships between poor water quality and the health of coral reefs or seagrasses have involved large inputs of sediment or nutrients to relatively small areas (e.g. Kaneohe Bay, Hawaii; Chesapeake Bay; Discovery Bay, Jamaica; Hervey Bay). Fortunately, these extreme situations have not appeared within the GBRWHA. However, the same processes and pressures that have caused large changes elsewhere are evident in the Great Barrier Reef and its Catchment. In almost every case study, major ecosystem degradation has begun with small, almost imperceptible changes, essentially indistinguishable from the 'normal' range of environmental variability.

The potential impacts of declining water quality on ecosystems in the Great Barrier Reef lagoon have been synthesised and reviewed in recent years (Hutchings and Haynes, 2000; Haynes, 2001; Williams, 2001; Baker, 2003; Furnas, 2003; see also Brodie *et al*., 2004; Brodie *et al*., 2005; Fabricius, 2005; Fabricius *et al.*, 2005; Schaffelke *et al*., 2005). Many of these documents were produced to support the development of the Reef Plan. Of particular relevance to this report, the 2005 Special Edition of the Marine Pollution Bulletin (Volume 51) was compiled to provide a benchmark of current information on broad range of aspects of the water quality issue for the Great Barrier Reef, and to provide a forum for publication of relevant data gathered by management agencies but not previously published (Hutchings *et al*., 2005a). Hutchings *et al*., (2005a) provides a concise overview of the content of the papers.

This section provides a brief overview of the current knowledge of the environmental implications of declining water quality in the Great Barrier Reef, which is related to (and in many cases, based on) the program data presented in this report. It is not within the scope of this report to provide a comprehensive review, although linkage of the key environmental implications to the reported data (where known) is recognised as being critical to support the integration of the subprograms in the Reef Plan MMP.

Inshore coral reefs
A major body of recent coral reef research funded through the CRC Reef has focused on the ways in which runoff can affect coral reefs, using field and laboratory experiments and surveys on gradients in water quality. The numerous ways in which the various components of runoff can potentially affect coral reef communities have been reviewed by Fabricius, (2005). Freshwater alone can kill reefs: low salinity flood plumes caused coral mortality at Keppel Island after Cyclone Joy in 1991 (van Woesik *et al*., 1995). The effects of suspended particles are diverse: they provide food for heterotrophic species; impose a metabolic cost in other species (e.g. for sloughing of tissue to remove settled particles); can reduce suitable substrate for settlement; and can smother small juveniles that have already settled. Turbidity reduces light levels, influencing photosynthesis. Nutrients can affect reproduction in corals (see below) and, in some circumstances, can enhance growth of algae that compete for space (Diaz-Pulido and McCook, 2003). Pollutants such as agricultural herbicides may have species-specific effects on different life-stages of corals (Negri *et al*., 2005). The different tolerances of species will cause community composition to change along water quality gradients; less tolerant species are lost as water quality decreases.

At the same time, reef communities on the Great Barrier Reef do change along water quality gradients and the kinds of changes that can occur at a regional scale have been described (van Woesik *et al*., 1999; Fabricius and De'ath, 2004; Fabricius *et al*., 2005). Reefs with high nutrient and/or particle availability (often reefs closest to shore or adjacent to agricultural areas of the northern Great Barrier Reef coast) have high macroalgal cover, low cover of hard and soft corals and lower abundance of reef fish, as well as community compositions that were different from less exposed reefs with reduced biodiversity of hard and soft corals (ibid.; Fabricius and De'ath, 2001a) and crustose coralline algae (Fabricius and De'ath, 2001b). Along a cross-shelf gradients, higher incidence of bioerosion by macroborers occurred on Great Barrier Reef inshore reefs, with possible consequences for the integrity of the reef framework at these sites (Hutchings *et al*., 2005b). Abundance of coral bioeroders has been linked to decreased water quality, e.g. at Tahitian reefs (Pari *et al*., 2002).

Active biological demand normally keeps dissolved nutrient concentrations in Great Barrier Reef lagoon waters well below levels that are likely to directly affect the health of corals and coral reefs. Nutrient inputs stimulate the production of additional organic matter which directly and indirectly affects corals and coral reef ecosystems. The limited available measurements of organic matter in Great Barrier Reef shelf waters show higher concentrations near the coast, in the area most directly affected by terrestrial runoff. There is a need to develop a better understanding of the quantity, nature and fate of organic matter in Great Barrier Reef lagoon waters, particularly with regard to how it influences the ecology of reef and benthic communities.

The full consequences of changes in land use that have already occurred may not yet have flowed through to communities on inshore reefs of the Great Barrier Reef. Coral reproduction and recruitment is sensitive to a number of components of runoff. Laboratory and field studies show that elevated concentrations of nutrients and high levels of suspended sediment and turbidity can affect one or more of gametogenesis, fertilisation, planulation, egg size, and embryonic development in some coral species (reviewed by Fabricius, 2005). High levels of sedimentation can affect larval settlement or net recruitment of corals. Similar levels of these factors may have only sub-lethal effects on established adult colonies. Corals are potentially long-lived organisms, so reefs may maintain high coral cover under conditions of declining water quality, but they may be relic communities made up of adult colonies that became established under more favourable conditions. Relic communities can persist until a major disturbance, but subsequent recovery will be slow if recruitment is reduced or non-existent. This can lead to degraded reefs since extended recovery times increase the likelihood that further disturbances will occur before recovery is complete (McCook *et al*., 2001). The presence of juveniles is necessary (but not sufficient) for coral communities to be sustaining.

The resilience and perseverance of coral reefs that are faced with both natural and increasing anthropogenic disturbances, ultimately depends on the successful settlement and survival of reef dwelling corals. Coral recruitment is directly vulnerable to the effects of sedimentation (Birkeland, 1977; Richmond, 1993; Babcock and Smith, 2002) and pollution (Negri and Heyward, 2000; 2001). Harrington *et al*., (2005) have shown that the indirect effects of sediments and pollution on reef substrata also have the potential to indirectly affect reef recovery and regeneration. Marine organisms rarely encounter only a single stressor, and multiple stressors pose significant effects to recruitment and the regenerative processes of marine assemblages (Hughes and Connell, 1999). Crustose coralline algae (CCA) is thought to play a key role in facilitating coral recruitment (Richmond, 1997; Morse *et al*., 1988, 1996; Heyward and Negri, 1999), therefore any disturbance to CCA can have subsequent effects on the resilience of coral reefs. Gilmour (1999) demonstrated that sediment deposition on CCA forms a barrier that reduced coral larvae settlement. Other studies have shown that live CCA is significantly more effective in inducing coral larvae to settle than dead CCA (Harrington *et al*., 2005). Harrington *et al*., (2005) represents the first attempt to understand the effects of environmentally relevant combinations of sediments and herbicides on critically important algae species. CCA is sensitive to environmentally relevant combinations of sediment (up to 116mgcm^{-2}) and diuron (> or equal to 0.79ugL^{-1}). These concentrations are exceeded in the Great Barrier Reef during flood events. For example, in the central Great Barrier Reef, river plumes affect large proportions of coastal coral reefs, carrying on average between 5 and 50mgL^{-1} of fine, organically and nutrient enriched suspended sediments with some concentrations reaching as high as 300mgL^{-1} close to river mouths (Devlin *et al*., 2001). The maximum concentration of diuron detected in rivers flowing into to the Great Barrier Reef is 8.5ugL^{-1} (White *et al*., 2002; Mitchell *et al*., 2005). Although concentrations on the near-shore reefs have not been measured, partitioning co-effcients and sediment concentrations have been used to estimate maximum water concentrations of between 0.1 and 1ug L^{-1} in coastal environments (Haynes *et al*., 2000a).

Another important nutrient-related interaction on reefs of the Great Barrier Reef, and through the Indo-Pacific generally, is that between the coral-eating COTS (*Acanthaster planci*) and reef condition. It is now believed that outbreaks of *A. planci* are associated with broad scale nutrient enrichment from land runoff and subsequent phytoplankton blooms leading to enhanced survivorship of *A. planci* larvae (Brodie *et al*., 2005). The critical chlorophyll *a* concentration range at which larval survivorship becomes significantly enhanced is 0.5–0.8 ugL^{-1} (Brodie *et al*., 2005). It is thus possible to use a chlorophyll *a* concentration of 0.5 ugL^{-1} in the larval period of *A. planci* (November to February) as a threshold guideline to ensure *A. planci* outbreaks are minimised. These concentrations are within the ranges recorded in the Long term Chlorophyll Monitoring Programs in the sectors south of (and including) Port Douglas (Section 3.3).

Biological effects of the most potent agricultural herbicides on coral larvae and reef-building algae begin to occur at concentrations in the 1-3 μg L^{-1} range (Harrington *et al*., 2005; Negri *et al*., 2005). Seagrass photosynthesis is inhibited at water concentrations of 0.1 to 100 μg L^{-1} of the herbicides diuron, atrazine and simazine, however, does usually recover after removal of the herbicide (Haynes *et al*., 2000b; Macinnes-Ng and Ralph, 2003). Highest concentrations of herbicides detected in Great Barrier Reef seagrass sediments are 1.7 μg kg^{-1} for diuron and 0.3 μg kg^{-1} for atrazine (Haynes *et al*., 2000a). Low concentrations of herbicides (e.g. diuron of up to 25 ng L^{-1}) were detected in surface waters over inter-tidal seagrass meadows in Hervey Bay and in sediments of these seagrass meadows (e.g. diuron and atrazine of up to1.1 μg kg^{-1} dry sediment) (McMahon *et al*., 2005). Herbicide concentrations increased in these surface waters during moderate river flow events (e.g. diuron of up to 50 ng L^{-1}; McMahon *et al*., 2005). Although no photosynthetic stress was detected in seagrass during low river flow, inshore seagrasses are considered at risk of being exposed to herbicide concentrations known to inhibit photosynthesis during higher river flow events. There is currently 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.

Coral reefs worldwide, including the Great Barrier Reef, are threatened by increased seawater temperatures and altered water chemistry caused by global climate change. The existance of healthy coral reefs into the future is dependent on minimising the rate and magnitude of further warming and maximising the resilience of reef systems. Maintenance of water quality in coral reef environments is likely to be critical to increase the chances that coral communities have the resilience to survive water temperature extremes (Hutchings *et al.*, 2005a).

When the reefs of Great Barrier Reef in general are grouped by latitude and position on the continental shelf into 'regions', the corals on the reefs within regions tend to show synchronous patterns of decline and recovery after disturbance. These cycles are not synchronised among regions, reflecting the scale of disturbances. Within a region, inshore reefs seem to be more variable in terms of coral cover and community types than is the case for mid-shelf and outer shelf reefs. Inshore reefs with quite different levels of coral cover can occur only a few km apart. It could be that impacts are more localised or there is much greater variation in recovery rates on inshore reefs. Environmental gradients in turbidity, salinity, exposure to floods and many other stresses may be steeper in inshore areas.

Assessment of ecosystem health will require measurements of such environmental factors and knowledge of the taxonomic composition and ecology of the coral communities that can be expected to develop in such conditions. Since these do not yet exist it is hard to give definitive assessments of health of inshore ecosystems. Much of the circumstantial evidence for impacts of runoff on inshore reefs is based on the apparent slow recovery from disturbances. Many inshore reefs on the Wet Tropics coast were extensively damaged by bleaching in 1998. Reporting in 2003 on information presumably gathered a year or two beforehand (Baker, 2003) considered that recovery was slow or non-existent. In 2004, six years after the bleaching event, the highest densities of coral recruits in any survey locations were to be found on Wet Tropics reefs. An assessment of what are realistic rates of recovery from various types and intensities of impact is clearly critical.

To address this, the assessment of inshore reefs ecosystems under the current Reef Plan MMP will measure coral cover, community structure and community demographics, including recruitment rates, to assess the balance between disturbance and recovery processes. Integrating these coral data with knowledge of environmental factors such as temperature and water quality parameters, which are measured at the same sites, and the disturbance history at a local scale will be critical to the interpretation of coral community changes over the next decade.

Seagrass and other marine plants

The responses of marine plants to changes in water quality in the Great Barrier Reef has recently been reviewed by Schaffelke *et al.*, (2005). The limited information of these responses limits the ability to make conclusions about responses across community types, however, there are clear indications that declining water quality negatively affects Great Barrier Reef macrophytes. 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 to altered benthic communities with high macroalgal cover on some Great Barrier Reef nearshore reefs. 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 Great Barrier Reef, and without also addressing deteriorated water quality such an intervention could only be an interim measure. Loss or disturbance of habitat-building macrophytes such as mangroves and seagrasses has serious downstream effects for coastal water quality due to their capacity to assimilate nutrients and to consolidate sediments.

The distribution and growth of seagrasses is dependent on a variety of factors such as temperature, salinity, nutrient availability, substratum characteristics, and underwater light availability (turbidity). Terrigenous runoff, physical disturbance, low light and low nutrients, respectively, are the main drivers of each of the four seagrass habitat types found in Queensland (Table 7.1), and changes to any or all of these factors may cause seagrass decline (Waycott *et al*., 2005). These factors can therefore be used to identify key areas of concern for monitoring impacts on these seagrass meadows (ibid.). In their natural state, these habitats are characterised by very low nutrient concentrations, are primarily nitrogen limited and are influenced by seasonal and episodic coastal runoff (Carruthers *et al*., 2001). All seagrass habitats in north east Australia are influenced by high disturbance and are both spatially and temporally variable. However, the spatial and temporal dynamics of the different types of seagrass habitat are poorly understood. Among these four seagrass habitat types in the Great Barrier Reef, both estuarine and coastal seagrass habitats are of primary concern with respect to water quality due to their location immediately adjacent to catchment inputs.

Loss of seagrass due to storms, flooding and cyclones has occurred in several regions due to the influx of freshwater and sediment in the water which cuts light penetration underwater. In 1999, about 3,000 hectares of seagrass in Hervey Bay was lost after flooding of the Mary River and local watercourses (Marsh and Lawler, 2001). Thousands of hectares of seagrass also appear to have been lost from northwest Torres Strait possibly due to flooding and sedimentation from Papua New Guinea in the mid 1990's. These changes may be part of longer-term natural cycles as both regions have subsequently recovered. A similar loss of seagrass appears to have occurred around Townsville in the early 1970s (Pringle, 1989), possibly associated with cyclone Althea (1971), but this was not fully studied at the time and the sequence of events is not as clear as in the Hervey Bay case.

As discussed in Section 3, inshore seagrass systems are episodically subjected to high dissolved nutrient and suspended loads during flood conditions. Abal and Dennison, (1996) predicted that detectable impacts on seagrass meadows may occur if higher sediment and associated nutrients were transported to the inshore regions of the Great Barrier Reef. It is intuitive that a continued decline in water quality will make plant growth increasingly difficult due to lack of light. However, the critical stress points for different seagrass species and their inherent adaptations to low light or variable light environments are not known; the extreme example of which is the seagrass species *H. ovalis* found inter-tidally, but also in deeper water habitats. This species occurs in high turbidity and clear water habitats, on reefs and in estuaries. How this same species survives in such a wide range of habitats is as yet unknown (Waycott *et al*., 2005). For most other seagrass species that occur in coastal habitats of the Great Barrier Reef, there is no data available to aid in the interpretation of their ability to adapt to both baseline and changing water quality. In addition to dynamic light environments, pulses of turbidity, particularly where they are derived from catchments, bring associated nutrients and pollutants (Brodie, 2002). These inputs are poorly understood with respect to seagrass meadow survival and require significant research effort to clarify effects before interpretation of change is possible.

Table 7.1. Summary of seagrass habitats in Queensland (from Carruthers et al., 2001).

Habitat	Limiting factor	Seagrass species	Feature/threats
River estuaries	Terrigenous runoff	*Cymodocea serrulata* *Enhalus acoroides* *Halophila ovalis*	Highly productive High density, low diversity Often anoxic

		Zostera capricorni	Highly threatened
Coastal	Physical disturbance	*Halophila ovalis* *Halophla spinulosa* *Halodule uninervis* *Syringodium isoetifolium* *Cymodocea serrulata* *Zostera capricorni*	Very diverse Highly productive Important for fisheries Supports dugongs Dynamic Threatened by development
Deep-water	Low light	*Cymodocea serrulata* *Halophila decipiens* *Halophila ovalis* *Halophla spinulosa* *Halophila tricostata*	15-58+m deep Monospecific High turnover Least known habitat Treats include trawling
Reef	Low nutrients	*Cymodocea rotundata* *Cymodocea serrulata* *Halodule uninervis* *Halophila ovalis* *Halophla spinulosa* *Syringodium isoetifolium* *Thalassia hemprichii* *Thalassodendron ciliatum*	Support high biodiversity Shallow unstable sediment Variable physical environment Little studied Least threatened

To date, no major decline in seagrass abundance in the Great Barrier Reef region has been recorded or attributed directly to increased nutrient availability, though localised declines have occurred in the Whitsunday and Hervey Bay areas possibly as a result of increased macro- and epiphytic-algae (see www.seagrasswatch.org). The presence of high or low, concentrations of nutrients in the environment is one of the stressors on seagrass survival. Seagrasses have the ability to act as a bio-sink for nutrients, sometimes containing high levels of tissue nitrogen and phosphorous (Mellors, 2003). Research to date in the Great Barrier Reef region has shown that nutrients do not appear to have a negative effect on seagrass growth and distribution, as reported in temperate regions (Mellors *et al*., 2005). This is not an unexpected observation as the region as a whole is in relatively healthy condition compared to many other regions globally (Furnas, 2003). However, Udy *et al*., (1999) observed an increase in seagrass cover at Green Island between 1936 and 1994 through aerial photographic analysis. The authors attribute this increase in seagrass to a net increase in the total nutrient pool available over 50 years of gradual build-up of nutrients in the Cairns region. This observation highlights the nature of gradual, diffuse sources of nutrients and sediments and the long term impact these may have. If, in fact, there has been such an increase in nutrients in the Cairns region, then short term sampling will not detect these differences. Recent data on seagrass tissue nutrient content (*Halophila ovalis*) collated by Mellors (2003) and Mellors *et al*., (2005) in Cleveland Bay shows an increase in tissue nutrients for a 25 year period which circumstantially reflects increases in fertiliser usage in the adjacent Burdekin catchment.

The direct effects of higher nutrient availability on seagrass in laboratory experiments have been observed. Moderate levels of nitrate additions (3.5 to 7.0 µM) promoted the decline of the temperate seagrass species *Zostera marina* (Burkholder *et al*., 1992; Short *et al*., 1995). Increased levels of ammonia (1.85–5.41 µM) and phosphate (0.22–0.50 µM) lead to a reduction in shoot density and biomass of *Z. marina* (Short *et al*., 1995). The concentrations measured in water samples taken in flood plumes have consistently recorded elevated dissolved inorganic nitrogen concentrations of 0.6 to 10 µM and phosphate levels of 0.13 to 1.98 µM (Brodie and Mitchell, 1992; Steven *et al*., 1996; Brodie and Furnas, 1996; Devlin *et al*., 2001). These nutrient levels have remained high in the inshore lagoon for periods of several days to weeks. Approximate ranges for (non-flood) inshore water nutrient concentrations have been measured between non-detectable and 2 µM for dissolved inorganic nitrogen (predominantly ammonia) and non-detectable and 0.2 µM for phosphate (Furnas *et al*. 1995; Furnas and Brodie 1996; Devlin *et al*. 1997).

The role of nutrients in seagrass survival in the Great Barrier Reef region has to date shown that seagrass growth is limited by nitrogen in the Great Barrier Reef region (Udy *et al*., 1999; Mellors, 2003). Both Udy and Mellors assessed the response of seagrass to enhanced nutrient levels and saw a response to both nitrogen and phosphorus but nitrogen was the primary limiting element. Thus at present seagrasses have the capacity to absorb additional nutrients enhancing their growth and it would appear that the current nutrient loadings in the Great Barrier Reef have not yet reached critical levels for seagrasses. However, the limits of the ability of seagrasses to continue to absorb nutrients is not known and additional experimentation that investigates the interaction between sediments, nutrients and the other limiters of plant growth light and temperature is required. In addition, nutrient analyses have been conducted primarily on the smaller more ephemeral species. Larger more persistent species may be more sensitive to additional nutrients in this region and this should be assessed.

There is 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. As a first step in elucidating the relationship between seagrass and water quality, Mellors *et al*., (2005) studied the sediment and nutrient status of intertidal seagrass meadows in the central region of the GBRWHA. The study showed that nutrients do not appear to be having a negative effect on seagrass growth and distribution. Data on seagrass tissue nutrient content (*H. ovalis*) showed an increase in tissue nutrients for a 20+ year period in Cleveland Bay, however a broader spatial survey revealed substantial heterogeneity in sediment nutrients and seagrass biomass even within species (Mellors *et al*., 2005). This heterogeneity indicates the significance of local site history. The geographic setting of a location dictates its sediment regime, while the frequency of disturbance dictates the structure of the meadow. In turn, differences in sediment mineralogy and grain size influence the nutrient regime at specific locations.

Herbicides (principally diuron) have also been found in coastal and intertidal seagrasses adjacent to catchments with high agricultural use at levels shown to adversely affect seagrass productivity (McMahon *et al*., 2005; Haynes, 2000b; refer also to Section 3.5). For example, diuron toxicty trials on three tropical seagrass species (*Halophila ovalis*, *Cymodocea serrulata* and *Zostera capricorni*) using Pulse-Amplitude-Modulated (PAM) fluorometry indicated that environmentally relevant levels of diuron (0.1-1.0 µg/l) exhibited some degree of toxicity to one or more of the tested seagrass species (Haynes *et al*., 2000b). Seagrasses are known to accumulate heavy metals, but appear to be moderately resistant to the direct effects of metals. However, the fauna associated with seagrass meadows is considered to be at great risk (Ward, 1989).

The ability of seagrass meadows to recover from large scale loss of seagrass cover observed during major events such as cyclones or due to anthropogenic disturbances such as dredging can be very slow, or nonexistent, because of chronic environmental pressures and poor dispersal capabilities of most seagrass species (Preen *et al*., 1995; Dennison and Kirkman, 1996). The capacity of seagrasses to recover requires either recruitment via seeds or through vegetative growth (Campbell and McKenzie, 2004). 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 (Waycott *et al*., 2005). The recovery of tropical seagrasses depends on the species and location; some plants are fairly resilient in unstable environments. Although structurally large seagrasses inhabit the Great Barrier Reef region, small species (e.g., *Halodule* and *Halophila*) comprise the majority of the coastal inshore seagrass meadows; they grow rapidly and survive well in unstable environments or places where sediments are continually being deposited. 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).

Knowledge Gaps

There are still many knowledge gaps relating to water quality and ecosystem health in the Great Barrier Reef. In particular, there is a lack of understanding of synergistic effects between declining water quality and other stressors on Great Barrier Reef ecosystems. A critical progression in understanding ecosystem health will be the development of water quality related pressure and condition indicators for the GBRWHA. This is examined by Moss *et al*., (2005); the relevance of these indicators with respect to regional ecosystem health and the availability of data on which to base guideline values is also discussed. Substantial progress is required to enable the development of disturbance and/or pollutant tolerance levels for ecosystems in the Great Barrier Reef.

While pollution effects on coral reefs at local scales are well understood, links at regional scales between increasing sediment and nutrient loads in rivers, and the broadscale degradation of coral reefs, have been more difficult to demonstrate (Fabricius and De'ath, 2004). This is due to a lack of large scale historic data and the confounding effects of other disturbances such as bleaching, cyclones, fishing pressure and outbreaks of the coral eating COTS, and is further complicated by the naturally high variability in monsoonal river flood events. The full extent of organism responses are poorly understood, as each of the numerous inshore species has its own tolerance limit at every life stage, and interactions between the organisms add to the complexity. Though considerable knowledge has been gained from single species exposure experiments in the laboratory, it is difficult to extrapolate from such laboratory studies to field settings and ecosystem responses. Detailed surveys at relatively fine taxonomic resolution, when cautiously interpreted in the context of available biophysical environmental data and biological knowledge of key species, can provide important information on the health and status of inshore coral reefs (Fabricius *et al*., 2005).

Causes for differences in assemblages are naturally difficult to determine definitively in ecological studies, especially if historic data are sparse. A framework based on epidemiological criteria can help synthesise and weigh available evidence to assess the likelihood of a causal association (Fabricius and De'ath, 2004). Both the regional differences in water quality and assemblages, and the existence of ecological gradients along the water quality gradients, added evidence that many of the responses were related to the differences in water quality. The changes along the water quality gradient that were consistent in direction with other studies (decreasing corals and increasing algae), the monotonic responses, and the large and ecologically relevant effect sizes, all added evidence that the inshore reef assemblages are strongly shaped by present-day water quality conditions.

The Reef Plan MMP provides an avenue to improving this understanding by integrating water quality monitoring with biological monitoring of the key inshore ecosystems. However, it is important to note that these sampling programs were initiated and have been continued for a variety of scientific objectives, mostly un-related to the monitoring objectives of the Reef Plan. It is anticipated that the design of the current and future Reef Plan MMP will enable integration of the datasets to build certainty in the understanding of causal links to ecosystem health.

Conclusions

As discussed in this Section, and throughout the report, there is sufficient evidence to demonstrate the threat of declining water quality to inshore coral reef and seagrass communities, together with evidence of a decline in their condition. For example, there is well-documented evidence within the Great Barrier Reef that benthic communities on inshore coral reefs vary along measured or presumed gradients of water quality/terrestrial influence. Observed changes include variations in the cover, composition and relative abundance of macroalgae, hard corals and soft corals, the recruitment of young hard corals and the abundance of coral bio-eroders. However, on a reef-wide scale the temporal and spatial dynamics of important ecosystems such as coral reefs and seagrass meadows are only beginning to be understood.

In order to assess the effects of land runoff on the Great Barrier Reef lagoon water quality and ecosystems with certainty, it is necessary to understand the assimilative capacity of the Great Barrier Reef lagoon and the critical thresholds of the ecosystem to ecological harm from the cumulative inputs of nutrients, sediments and pollutants. While dilution, sedimentation and biological uptake and transformation effectively remove nutrients and sediments from the water column, these materials stay in the system and are likely to slowly accumulate.

The highly variable (episodic, seasonal, inter-annual) monsoonal climate of the Great Barrier Reef region, and the episodic nature of disturbance processes operating within the Great Barrier Reef (cyclones, COTS, bleaching) produce high natural variability of river discharge, riverine sediment and nutrient loads, lagoonal water quality, biological community structure and community distributions particularly between wet and dry catchments. The variable influences of nutrient speciation, particularly nitrogen, require further consideration. Any successful monitoring program established to track progress toward achieving the goals of the Reef Plan needs to take a long-term perspective, be fit for purpose, and based on the existing reef-scale and long-term water quality and biological monitoring programs which have sufficient duration (5 to 15 years) and sampling density to provide useful baseline and trend/variability data.

In conclusion, much of the technical knowledge relating to water quality and ecosystem health in the Great Barrier Reef to date has been based on relatively small scale programs; the current research direction reflects a strong progression to extending this knowledge to a broader whole-of-system approach which is the foundation for the development of the Reef Plan. The Reef Plan MMP will assist to overcome this limitation by including inshore reef surveys and water quality monitoring at the same locations. In addition, passive samplers will be deployed within the corresponding regions to measure pesticide loads; sediment sampling will be undertaken to qualify these measurements. The Seagrass-Watch program will continue across the regions at sites that are complementary to river mouth and marine water quality monitoring programs. New and more extensive statistical analyses will be undertaken for the datasets. This improved capacity to integrate the subprograms in combination with ongoing research of the environmental implications of water quality on Great Barrier Reef ecosystems, will assist to improve our understanding of water quality influences and hence, changes in the Great Barrier Reef which may be attributable to implementation of the Reef Plan.

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Appendix 1: River sampling details

Date intervals within which pre-Reef Plan MMP nutrient sampling was carried out in Great Barrier Reef Catchment rivers by AIMS personnel and numbers of samples from which statistical summary values and statistical plots were calculated. NH_4 = ammonium, NO_2= nitrite, NO_3= nitrate, TDN= Total dissolved nitrogen, DON= dissolved organic nitrogen, PN= particulate nitrogen, PO_4= phosphate, TDP= total dissolved phosphorus, DOP= dissolved organic phosphorus, PP= particulate phosphorus, Si= silicate, SS= suspended solids.

River	Date Start	Date Finish	No. samplings	NH_4	NO_2	NO_3	TDN	DON	PN	PO_4	TDP	DOP	PP	Si	SS
Normanby R.	01-Jan-99	08-May-00	29	29	29	29	29	29	29	29	29	29	28	29	17
Barron R. (upper)	20-Nov-89	05-Aug-95	206	202	202	202	171	168	150	201	172	168	165	41	18
Mulgrave R. (upper)	10-Jun-88	30-Sep-94	41	40	40	40	40	39	11	40	40	39	34	4	20
Mulgrave R. (lower)	30-Nov-89	30-Sep-94	28	28	28	28	27	27	9	28	27	27	22	4	18
Russell R.	10-Jun-88	14-Dec-00	39	38	38	38	38	37	13	38	38	37	34	3	19
No. Johnstone R.	10-Jun-88	30-Sep-94	42	39	39	39	40	39	13	38	40	38	35	5	19
So. Johnstone R.	14-Dec-86	30-Sep-94	403	400	400	400	395	392	366	400	390	387	374	205	24
Tully R. (gorge)	18-Sep-87	13-Dec-00	109	109	109	109	104	104	100	109	104	104	101	39	1
Tully R. (lower)	19-Oct-86	13-Dec-00	344	305	337	337	337	329	323	326	336	325	318	320	212
Murray R.	10-Jun-88	06-Mar-95	39	35	35	35	37	33	11	35	37	33	37	2	17
Herbert R. (gorge)	22-Nov-89	03-Apr-95	155	146	146	146	148	139	90	146	148	139	140	39	107
Herbert R. (upper floodplain)	22-Nov-89	03-Apr-95	144	139	139	138	133	127	105	139	133	128	126	40	102
Herbert R. (lower floodplain)	10-Jun-88	03-Jul-00	288	247	246	247	254	234	212	247	252	232	268	123	141
Barratta Ck.	28-Sep-88	22-May-91	54	54	54	54	12	12	2	53	12	12	44	25	0
Burdekin R.	01-Apr-87	08-May-00	713	702	552	698	658	644	595	702	663	652	573	476	311
Fitzroy R.	14-Nov-92	18-Mar-97	160	160	160	160	136	136	159	159	135	135	160	77	18

Appendix 2: Water quality sampling and analytical methods

River water quality and export estimation

The primary objective of the river sampling programs is to estimate annual exports of fine suspended sediment (suspended load - largely silts [x-63 µm] and clays [<63 µm]) and nutrients (N, P). Export is calculated as the integration of the product of in-river concentrations and concurrent freshwater discharge -[• concentration x discharge]. Since most freshwater discharge from Great Barrier Reef Catchment rivers occurs during the summer wet season, and especially during brief high-flow or flood events, sampling was focused upon these events to constrain nutrient and suspended sediment concentrations during periods when maximum export takes place.

Sediment exports

Two approaches were taken to measure fine sediment exports.

In the Burdekin River, water samples were collected in mid-river by bucket from the highway bridge at Ayr. Samples were collected at periods ranging from daily intervals during high flow periods, increasing to several days as flow decreased, and to weekly or longer at low-flow periods. Daily sediment export fluxes were calculated as the product of the sediment concentration (g L^{-1}) and the daily discharge (the Department of Natural Resources and Mines and Energy (DNR&M) measured at Clare). Sediment concentrations in gaps between sampling events was estimated by linear interpolation.

In the Wet Tropics and Fitzroy rivers, suspended sediment concentrations were calculated from instrumental measurements of turbidity. The turbidity readings (I_{meas} relative to I in deionized water) were made at 30-minute intervals by a self-contained logger attached to a bridge pylon or other fixed structure in or immediately adjacent to the river. Each logger recorded light transmission through two light paths (85 mm - Wet Tropics rivers) and 15 mm (Fitzroy River), external pressure (water depth) and internal temperature. An integral brushing mechanism cleaned all of the optical surfaces on an hourly basis. The logger was capable of operating fully submerged during flood conditions. During deployments, loggers were positioned just above the water level at dry season low water. Deployments of individual loggers ran from before the wet season started (Nov-Jan) to after it finished (June-July). After instrument recovery, the retrieved data files were de-spiked and any instrument drift removed by inspection using an Excel spreadsheet. Gaps were filled by linear interpolation. The instrumental records of turbidity (30-minute sampling) were matched with records of river flow (cumecs) recorded by the DNR&M interpolated to 30-minute intervals. Suspended sediment concentrations were calculated from linearised relative turbidity ($ln[I_{meas}/I_o]$] using factors derived from empirical calibrations of turbidity vs suspended mass in graded suspensions of fine river mud. Calculated concentrations of suspended sediment concentration (g L^{-1}) were multiplied by the concurrent flow rate (m^{-3} sec^{-1}) and integrated to yield an estimate of transported sediment.

Nutrient exports

Dissolved and particulate nutrient exports were integrated from measurements made on discrete samples collected from selected rivers. Surface water samples were collected with a clean plastic bucket. Sampling intervals ranged from daily to weekly during the wet season, depending upon the river flow and ability of the sampler to get to the sampling site. During low-flow periods, sampling intervals were lengthened. Measured nutrient concentrations were combined with records of daily discharge obtained from DNR&M. Nutrient concentrations on days between sampling events were estimated by linear interpolation.

Daily exports for each nutrient species measured were calculated by multiplying the daily discharge and relevant concentration. Wet season exports were calculated by integration of the daily export estimates.

Sample collection and processing
After discrete samples were collected by bucket, subsamples for suspended sediment concentration, dissolved and particulate nutrient (N,P, Si) concentrations were processed for storage within a short period (usually within a few hours). After thorough mixing of the river water sample, duplicate aliquots were filtered onto pre-weighed polycarbonate membrane filters (Poretics 47 mm diameter, 0.4 µm pore size) for suspended sediment determinations. The volume filtered depended on the amount of suspended matter in the water. The filters were stored at room temperature in ashed scintillation vials for further processing. Duplicate aliquots (10 ml) of filtered (Minisart M filtration cartridges - 0.45 µm) river water were syringe filtered into pairs of acid-washed plastic test tubes (12 ml) with screw caps and frozen in a clean freezer. In all, 4 sub-samples of filtered water were taken for analysis (2 dissolved inorganic nutrients, 2 dissolved organic nutrients). Duplicate aliquots of river water were filtered onto pre-ashed glass fibre filters (Whatman GF/F, 25 mm), folded and stored frozen in envelopes of pre-ashed Al foil in a clean freezer for analysis of particulate N and particulate P. As with suspended sediment, the volumes filtered depended upon the suspended particulate load. Aliquots were dispensed with graduated cylinders or clean plastic syringes. Care was taken to thoroughly resuspend and mix all particulate matter in the water samples before taking each aliquot from the main sample.

Data on sample collection (date, time, location) and processing (sample ID, pre-weighed filter numbers, volumes filtered) were recorded in field notebooks by the sampler.

Lagoon Water Quality

Water Sampling
Vertical profiles of temperature and salinity were measured at most sampling sites (stations) with a Conductivity-Temperature-Depth (CTD) profiler. Salinity and temperature values derived from the CTD casts are routinely checked against temperatures measured at discrete depths with reversing thermometers or salinities of discrete water samples determined with a laboratory salinometer (Plessey 6220).

Water samples were collected at each station with a single cast of Niskin bottles. Between 2 and 6 depths were sampled depending on overall water column depth. Water samples taken from each Niskin bottle were syringe filtered through 0.45 µm filters (Sartorius Minisart) into 4 acid-washed plastic-capped test tubes (12 ml) for measurement of dissolved inorganic (NH_4^+, NO_2^-, NO_3-, PO_4^{-3}, $Si(OH)_4$) and organic (DON, DOP) nutrients. The water samples were immediately frozen in a clean laboratory freezer. Duplicate subsamples from each depth were drawn from the Niskin bottles and filtered onto pre-ashed glass fibre filters (Whatman GF/F, 25 mm) for determination of chlorophyll *a* (2 x 100 ml), particulate nitrogen (2 x 250 ml) and particulate phosphorus (2 x 250 ml). The filters were folded, wrapped in pre-ashed aluminium foil and frozen until analysis. Duplicate 1-litre subsamples from each sampling depth were filtered onto pre-weighed polycarbonate membrane filters (Poretics 47 mm, 0.4 µm) for determination of total suspended solids. The suspended sediment filters were stored at room temperature in pre-ashed scintillation vials.

Data Presentation
Because the water column at most sites in the Great Barrier Reef is well mixed vertically by wind stress and tidal action, station concentration data is summarized as a depth-weighted water column average. Depth-weighted average concentrations were calculated by

trapezoidal integration of water column concentrations down the sampling profile (summation of the average concentration between each two adjacent samples on a profile multiplied by the depth interval between the samples, the sum divided by the total profile depth).

Concentrations of most water quality parameters tend to have a log-normal distribution. Most values fall within a fairly narrow range of low concentrations, but with a tail of high concentrations due to inherent patchiness, zooplankton excretion or other factors. For this reason, medians provide the most reliable measure of central tendency for sets of data. Mean values are almost always higher, sometimes considerably so, than the median values due to a very small number of high samples.

Chlorophyll *a* Monitoring

Water Sampling
Duplicate surface water samples were collected with a clean bucket at each sampling site. Water transparency was measured with a Secchi disk. At some locations, temperature and salinity were recorded with a hand-held *in-situ* temperature-salinity probe. In other areas, surface water temperatures (to the nearest °C) were measured with a mercury thermometer immediately after sample collection. Salinities were measured with a hand-held refractometer. Site data (date, time, location, weather, sea state, presence of *Trichodesmium*) were recorded by the sampler. In most cases, 1-litre subsamples from the bucket water samples were placed on ice in an Esky and returned to the laboratory within several hours for filtration.

After mixing, duplicate subsamples (100 ml) were taken from each bottle and filtered for through Whatman GF/F filters (25 mm). After filtration, the filters were folded in pre-combusted aluminium foil envelopes and frozen. At intervals of 2-4 months, frozen, filtered chlorophyll *a* samples were transported frozen to AIMS for analysis. At AIMS, chlorophyll *a* samples were stored frozen until analysed.

Analytical Methods
Concentrations of dissolved inorganic nutrients (NH_4^+, NO_2^-, NO_3-, PO_4^{-3}, $Si(OH)_4$) were determined by standard wet chemical methods (Ryle et al., 1981) implemented on a segmented flow analyser. At a number of Great Barrier Reef lagoon water quality stations, ammonium concentrations were measured in freshly collected water at sea using modified colorimetric methods (Holmes et al. 1999)). Concentrations of total dissolved nitrogen (TDN) and phosphorus (TDP) were measured by irradiating filtered (0.45 µm) water samples (>8 hrs) with intense UV radiation in quartz test tubes, then re-analysing the samples for inorganic N and P species (Walsh, 1989). Concentrations of dissolved organic N (DON) and P (DOP) were calculated from the difference between total inorganic N and P after and before irradiation. Particulate N (PN) analysis of filtered samples was carried out by high-temperature combustion in an ANTEK 707/720 Nitrogen Analyser with a NO_x chemiluminescent detector. Analyses were standardized against aliquots of acetanilid blotted onto pre-ashed wedges of glass fibre filter. Sub-samples for particulate P analysis were refluxed to dryness in acid potassium persulfate. Final pH values prior to drying out were < 1. The digested filters were then homogenized in deionized water to dissolve the inorganic P. After the homogenate was cleared by centrifugation, the dissolved P content in an aliquot was determined colorimetrically (Parsons *et al.*, 1984). Total suspended sediment concentrations of the filtered material were determined gravimetrically. Chlorophyll *a* concentrations were determined fluorometrically after grinding in 90% acetone (Parsons *et al.*, 1984). With the exception of the shipboard ammonia measurements (3x), all analyses were run in duplicate.

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Appendix 3: River water quality statistical analysis

Source: De'ath (2005)

Table 1. Water quality variables used for statistical analyses.

Variable measured	Abbreviation	Units
Nitrite	no2	µmol L^{-1}
Nitrate	no3	µmol L^{-1}
Dissolved Inorganic Nitrogen	din	µmol L^{-1}
Dissolved Organic Nitrogen	don	µmol L^{-1}
Total Dissolved Nitrogen	tdn	µmol L^{-1}
Particulate Nitrogen	pn	µmol L^{-1}
Dissolved Inorganic Phosphate	dip	µmol L^{-1}
Dissolved Organic Phosphate	dop	µmol L^{-1}
Total Dissolved Phosphate	tdp	µmol L^{-1}
Particulate Phosphate	pp	µmol L^{-1}
Silicate	si	µmol L^{-1}
Suspended Solids	ss	mg L^{-1}

Tully	Temporal trend			Flow			Correlation		
	df	**Chi.sq**	**P**	**df**	**Chi.sq**	**P**	**Cor**	**LRT**	**P**
dip	1.01	0.73	0.3979	3.01	56.01	<0.0001	0.723	4.99	0.0256
din	1.31	0.74	0.4963	3.53	33.42	<0.0001	0.381	16.30	0.0001
no2	1.03	2.68	0.1089	2.21	4.39	0.1374	0.416	18.49	<0.0001
dop	1.00	1.15	0.2868	1.00	0.04	0.8349	0.393	15.33	0.0001
don	4.54	19.52	0.0018	2.03	2.65	0.2765	0.436	15.73	0.0001
no3	1.05	2.68	0.1104	3.75	48.40	<0.0001	0.381	17.38	<0.0001
tdp	2.70	11.23	0.0100	1.00	4.74	0.0317	0.249	5.73	0.0167
tdn	3.31	13.65	0.0064	3.61	22.93	0.0002	0.235	5.32	0.0211
si	5.92	92.48	<0.0001	3.38	69.08	<0.0001	-0.018	0.01	0.9217
pp	1.05	5.69	0.0202	1.02	35.32	<0.0001	0.198	3.89	0.0486
pn	1.23	21.23	<0.0001	1.04	40.84	<0.0001	0.194	3.63	0.0568
ss	3.21	14.46	0.0078	1.89	47.51	<0.0001	-0.090	0.17	0.6837
Burdekin	Temporal trend			Flow			Correlation		
	df	**Chi.sq**	**P**	**df**	**Chi.sq**	**P**	**Cor**	**LRT**	**P**
dip	5.18	35.45	0.0001	3.66	19.60	0.0016	0.195	8.44	0.0037
din	1.00	0.02	0.8927	2.46	29.44	0.0000	0.690	20.99	0.0000
no2	1.00	0.09	0.7652	1.01	10.48	0.0024	0.422	20.57	0.0000
dop	1.01	3.75	0.0590	1.01	1.64	0.2087	0.401	14.45	0.0001
don	1.85	1.75	0.3883	2.38	4.93	0.1288	0.423	14.58	0.0001
no3	1.01	0.77	0.3877	3.46	46.44	0.0000	0.717	22.66	0.0000
tdp	3.17	7.42	0.0831	6.56	40.66	0.0001	0.937	5.87	0.0154
tdn	2.81	12.66	0.0084	2.91	66.85	0.0000	-0.111	10.22	0.0014
si	1.00	5.67	0.0226	1.00	2.28	0.1397	0.153	2.15	0.1424
pp	6.53	79.57	0.0000	1.81	80.71	0.0000	0.085	8.16	0.0043
pn	6.04	36.20	0.0001	1.00	43.26	0.0000	0.462	4.53	0.0333
ss	2.46	12.83	0.0108	1.01	42.12	0.0000	-0.149	0.53	0.4682

Table 2. Analysis of water quality parameters (abbreviations in Table 1) for the Tully and Burdekin rivers over the period 1987 – 2000. All parameters were analysed and results subsequently plotted. The statistical model used for each analysis was a generalized additive model. The response was the log (base 2) transformed parameter (e.g. dip), and the explanatory variables were smooth terms in decimal years (e.g. 1991.083), log (base 2) of river flow, and months (for the Tully data only). The smoothness of the temporal and flow terms was selected by cross-validation. It was apparent that serial correlation over time was present for most responses and a first-order autocorrelation term was also included. For each river and each parameter the smoothness (df) and significance (Chi-squared test [Chi.sq] and P-value [P]) of smooth effects for the temporal trend are presented. Also shown is the estimate of the first-order

autocorrelation (Cor) and the likelihood ratio test (LRT) of significance (P); this is a 1 degree of freedom test.

Table 3. Analysis of temporal trends for site-averaged water quality variables (abbreviations in Table 1). Cross-validated smooths over years were fitted to the data. A df= 1 indicates a linear trend. Results are displayed in Figure 10.

Variable	df	Chi Square	P
nh4	1.00	1.89	0.177
no2	1.00	16.61	0.001
no3	2.84	10.28	0.023
tdn	2.11	12.37	0.005
pn	2.85	9.30	0.034
dip	2.74	15.51	0.003
tdp	2.01	26.92	<0.001
pp	1.00	2.11	0.155
si	1.85	2.40	0.282
chl	1.00	4.15	0.048
phaeo	1.00	0.83	0.367
ss	1.42	6.83	0.024

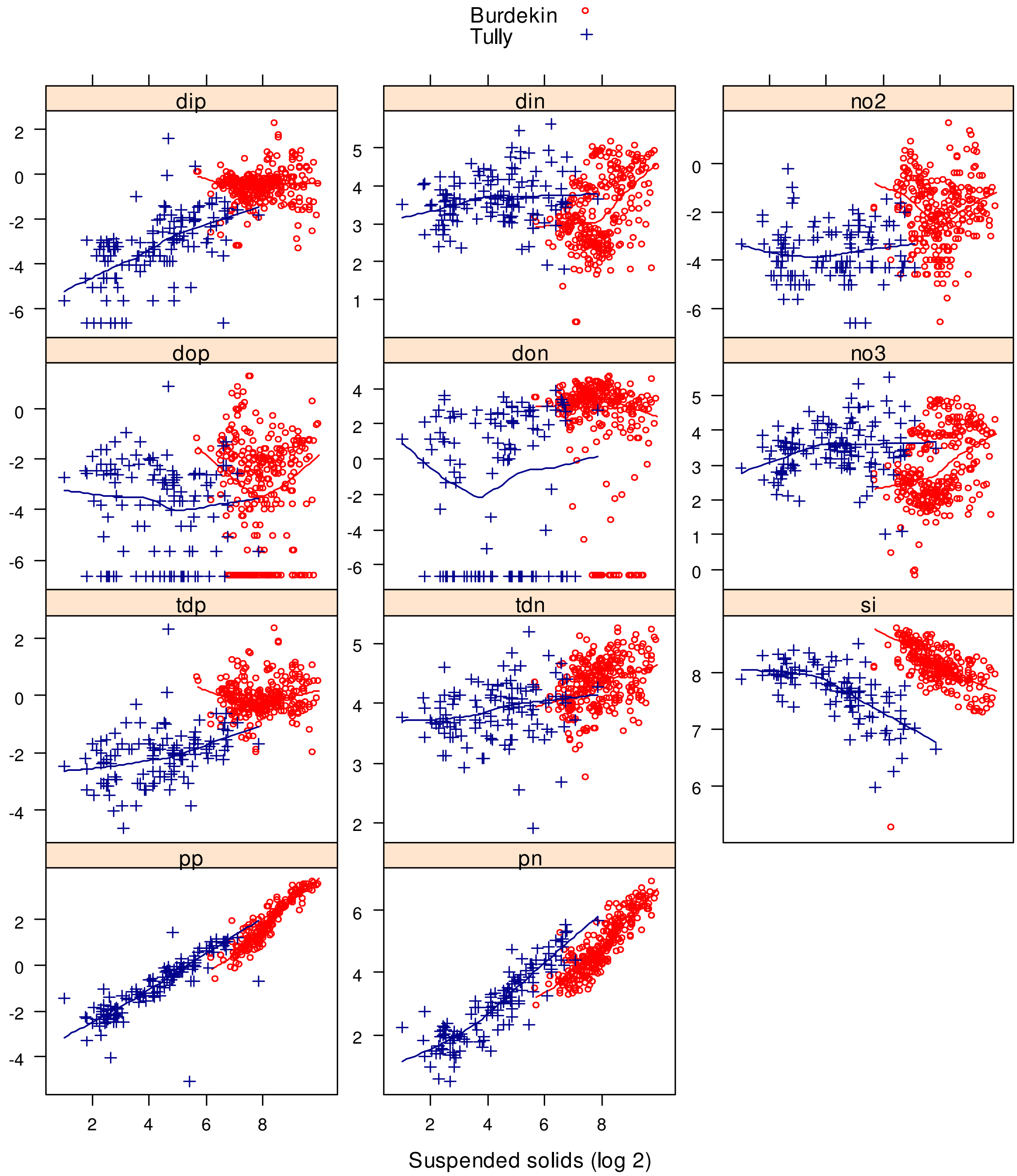


Figure 1. Plots showing the relationships of concentrations of water quality parameters to suspended solids for the Burdekin (red circles) and Tully (blue crosses) rivers. Data are restricted to the months January, February and March. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [DIP= PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. PP and PN are strongly related to suspended solids. Silicate shows negative relationships with suspended solids, and dissolved inorganic phosphorous a weak positive relationship.

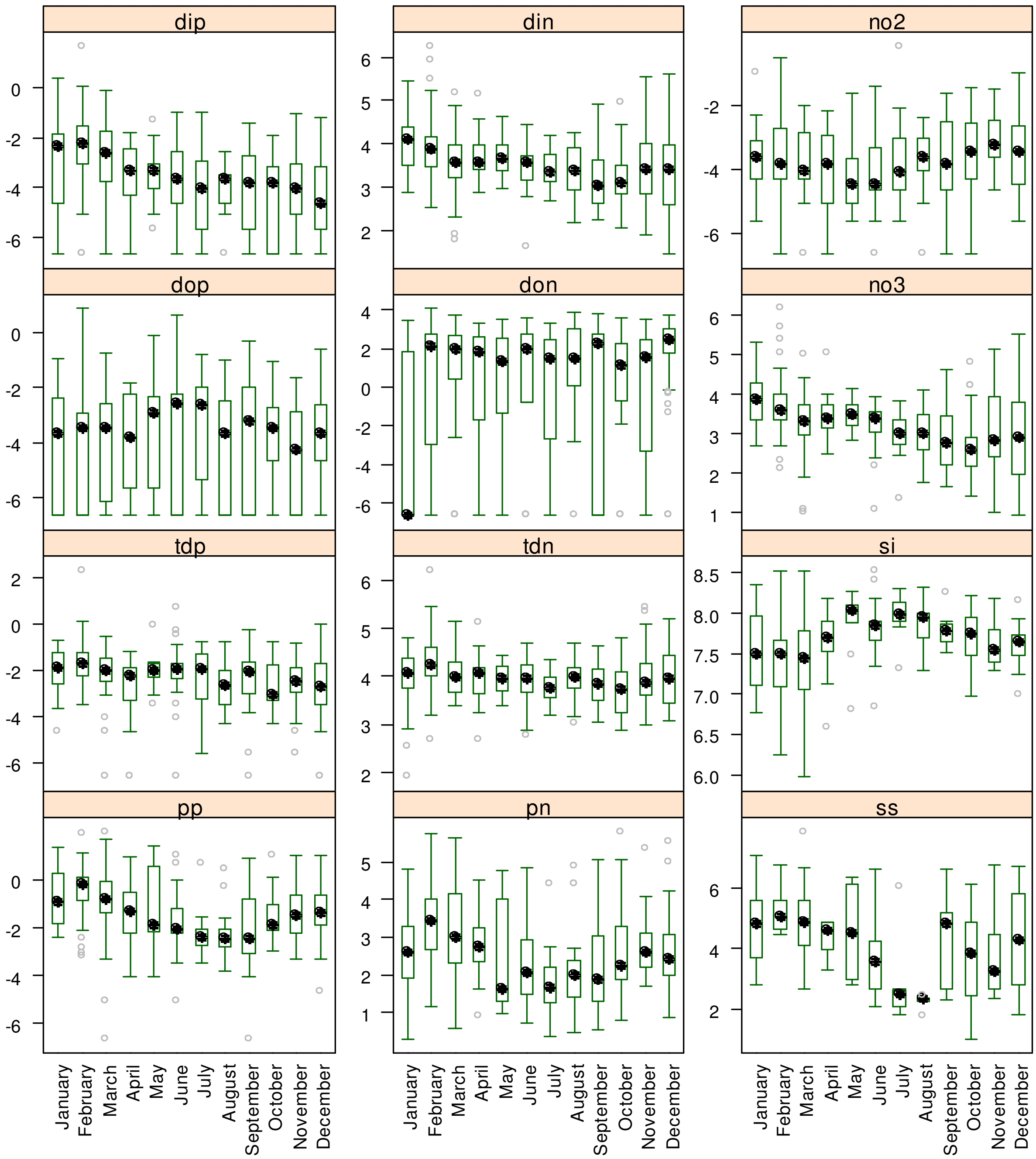


Figure 2. Monthly variation in concentrations of water quality parameters for the Tully River over the period 1987 – 2000. The boxplots indicate the median (filled circles), upper and lower quartiles (the box). The whiskers of the plot extend in both directions to either (i) the nearest of the most extreme data or (ii) twice the interquartile distance. Outliers are indicated as non-filled circles. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. There are strong seasonal differences for many of the parameters.

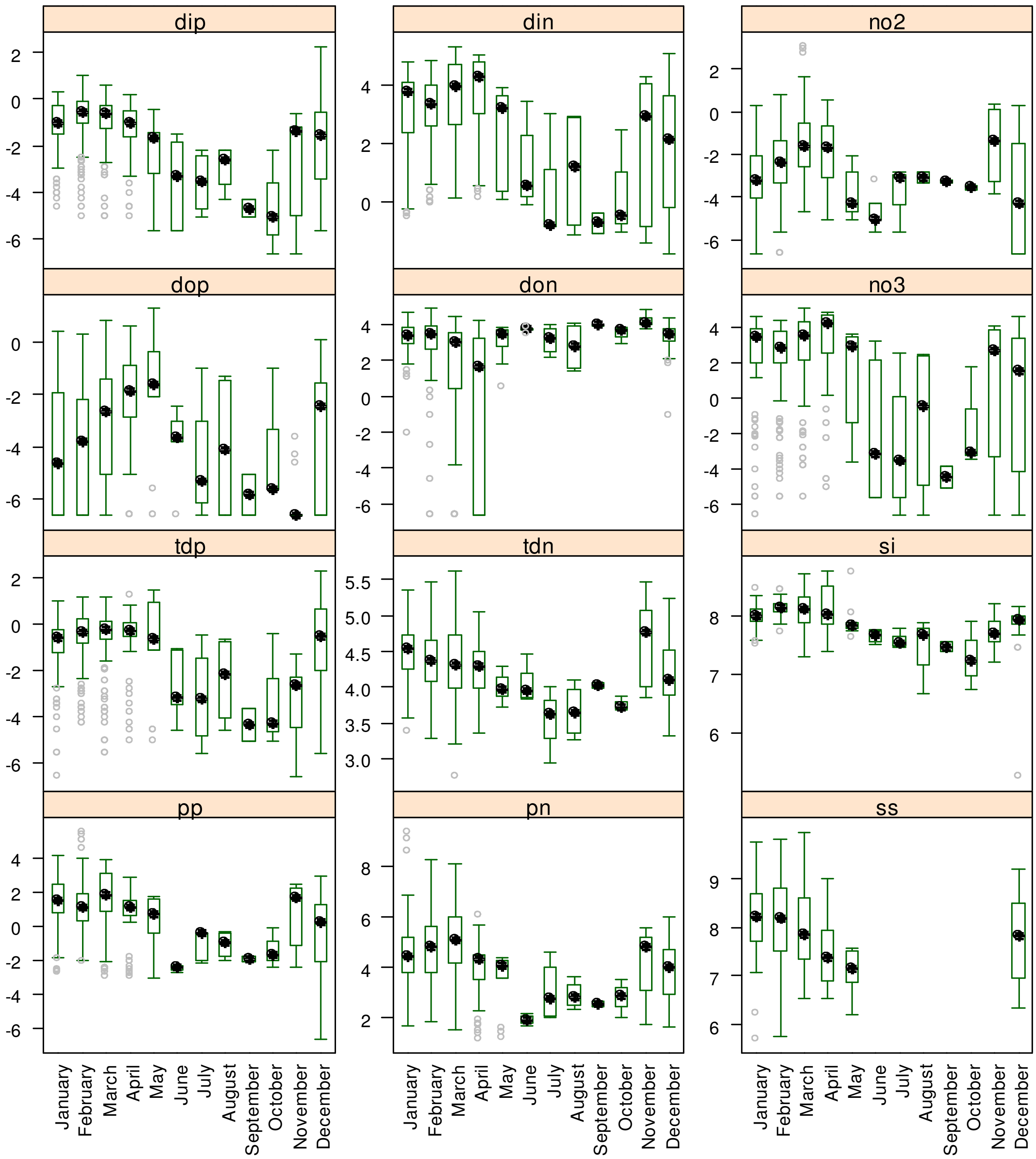


Figure 3. Monthly variation in water quality parameters for the Burdekin River over the period 1988 – 2000. The boxplots indicate the median (filled circles), upper and lower quartiles (the box). The whiskers of the plot extend in both directions to either (i) the nearest of the most extreme data or (ii) twice the interquartile distance. Outliers are indicated as non-filled circles. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3,, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. There are strong seasonal differences for many of the parameters.

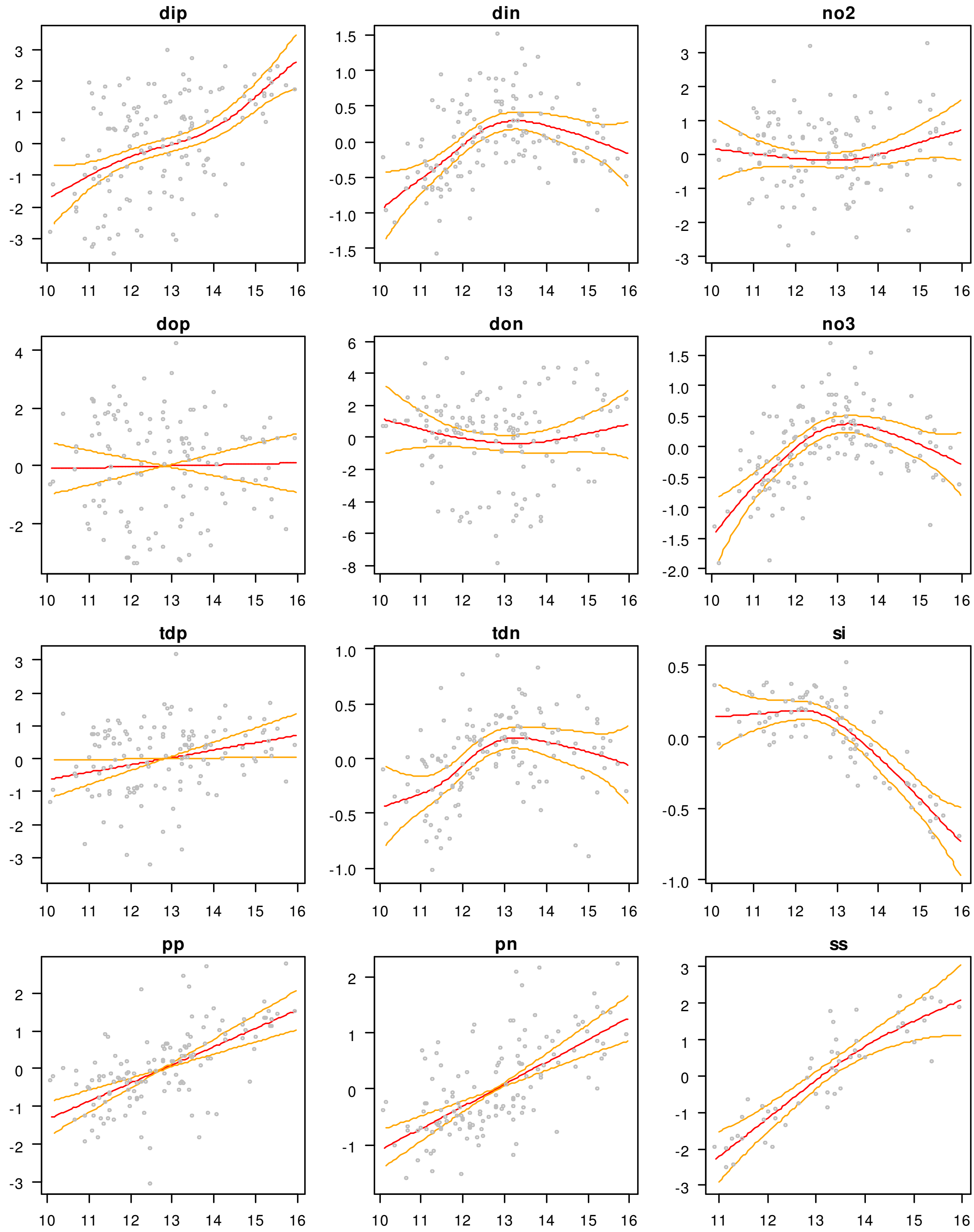


Figure 4. Estimation of the effects of flow (log base 2) on water quality parameters for the Tully River over the period 1987 – 2000. Flow effects are estimated using smooth terms in a generalised linear model. The flow effects are adjusted for monthly and temporal effects (smoothed) and autocorrelation between successive months. The smoothness of the temporal and river flow effects are estimated by cross-validation. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3,, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS, and ML d^{-1} for river flow. There are strong flow effects most of which are linear (PP, PN, and TDP), but some

are non-linear (DIN, TDN and NO^{3-}*). The non-linear effects are on dissolved nutrients and dilution is occurring at high flow rates.*

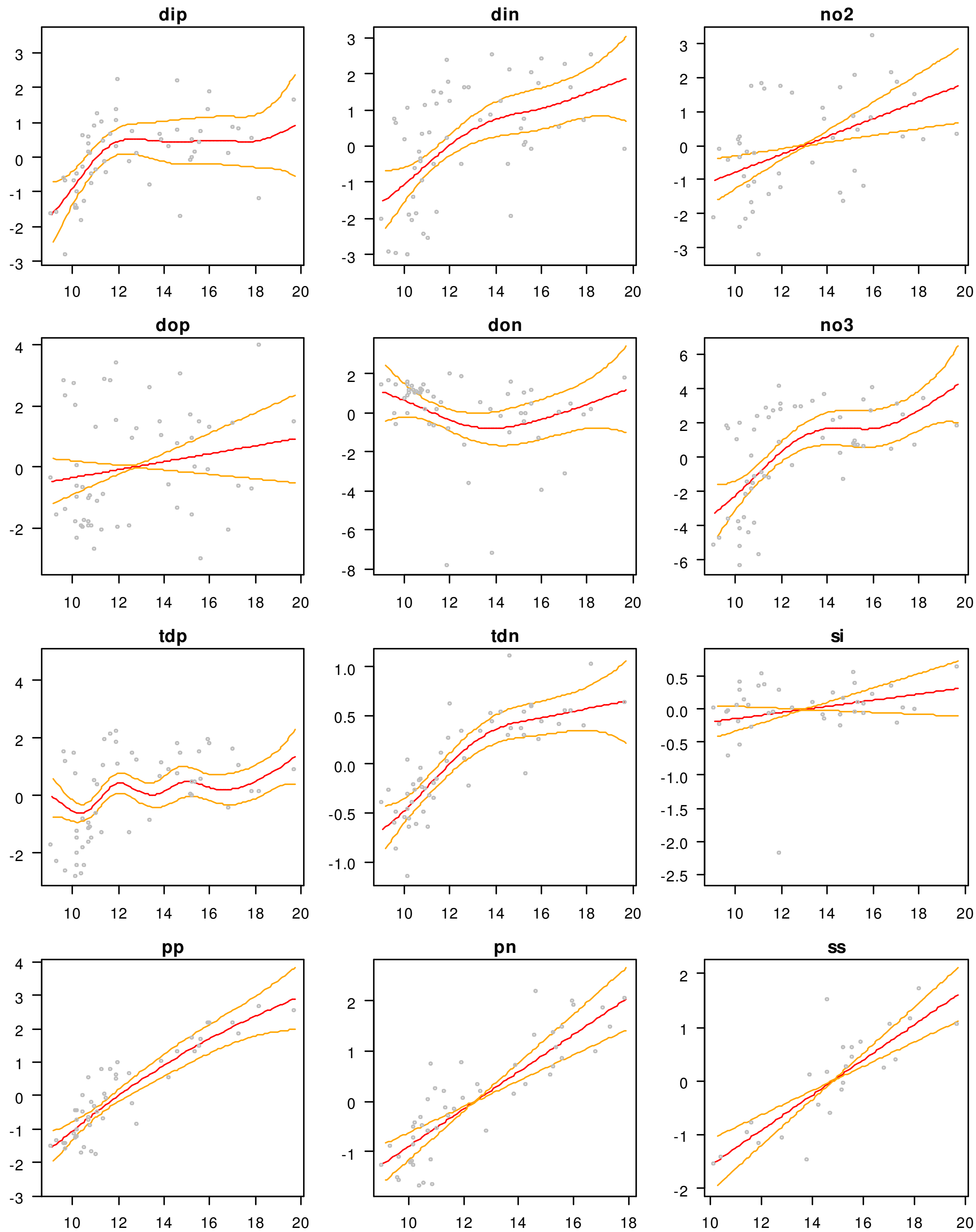


Figure 5. Estimation of the effects of flow (log base 2) on water quality parameters for the Burdekin River over the period 1988 – 2000. Flow effects are estimated using smooth terms in a generalized linear model. The flow effects are adjusted for monthly and temporal effects (smoothed) and autocorrelation between successive months. The smoothness of the temporal and river flow effects are estimated by cross-validation. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS, and ML d^{-1} for river flow. There are strong flow effects most of which are linear (PP, PN, SS, DIN, DOP

and NO^{2-}), but some are non-linear (TDP, TDN and NO^{3-}). The non-linear effects are on dissolved nutrients and dilution is occurring at high flow rates.

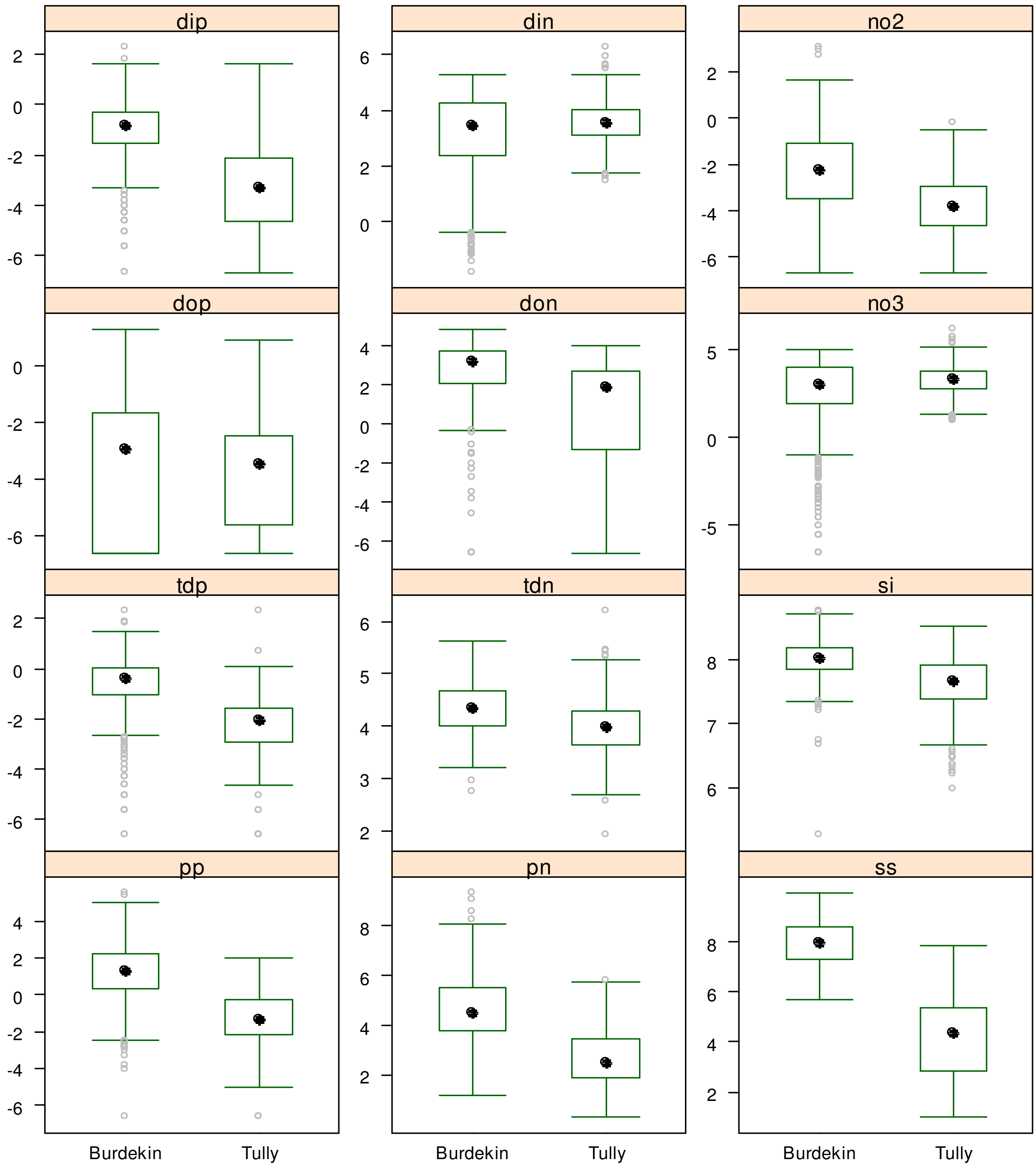


Figure 6. Comparison of concentrations of water quality parameters for the Burdekin and Tully Rivers averaged over the months January, February and March for the period 1987 – 2000. The boxplots indicate the median (filled circles), upper and lower quartiles (the box). The whiskers of the plot extend in both directions to the nearest of the most extreme data or twice the interquartile distance. Outliers are indicated as non-filled circles. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. Most parameters are higher in the Burdekin than the Tully. In particular, suspended solids are 15 times higher in the Burdekin.

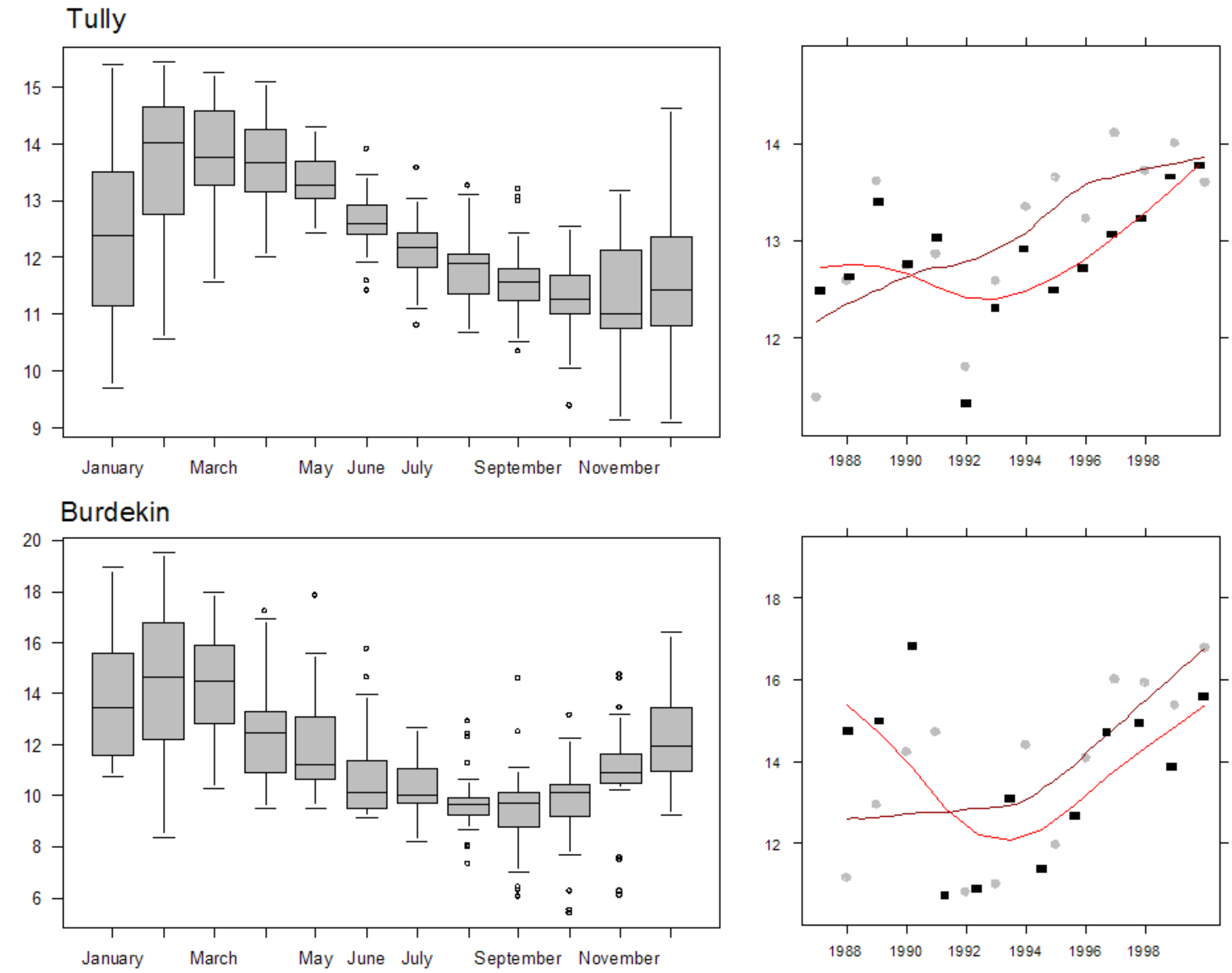


Figure 7. Seasonal and trend analyses of river flow for the Tully and Burdekin rivers over the period 1987 – 2000. All parameters are plotted on a log (base 2) scale. Units are ML d^{-1} for river flow. Seasonal (months) effects vary ~20 fold for the Burdekin compared to ~10 fold for the Tully. The red line (fitted trend) and black squares (annual means) show the mean flows for the two rivers. Weaker annual variation than seasonal variation is present for both rivers; ~3.5 fold for both the Burdekin and the Tully. For both rivers the trend in the mean flow is to some degree confounded with the year of survey and this makes it difficult to untangle changes in concentrations of water quality parameters due to trend and flow. The brown line (fitted trend) and grey circles (annual means) show the mean flows for the sampling occasions for two rivers. Apart from the first few years of sampling for the Burdekin, when river sampling was infrequent, the trends over years of both the mean river flows and mean flow at sampling times are quite similar.

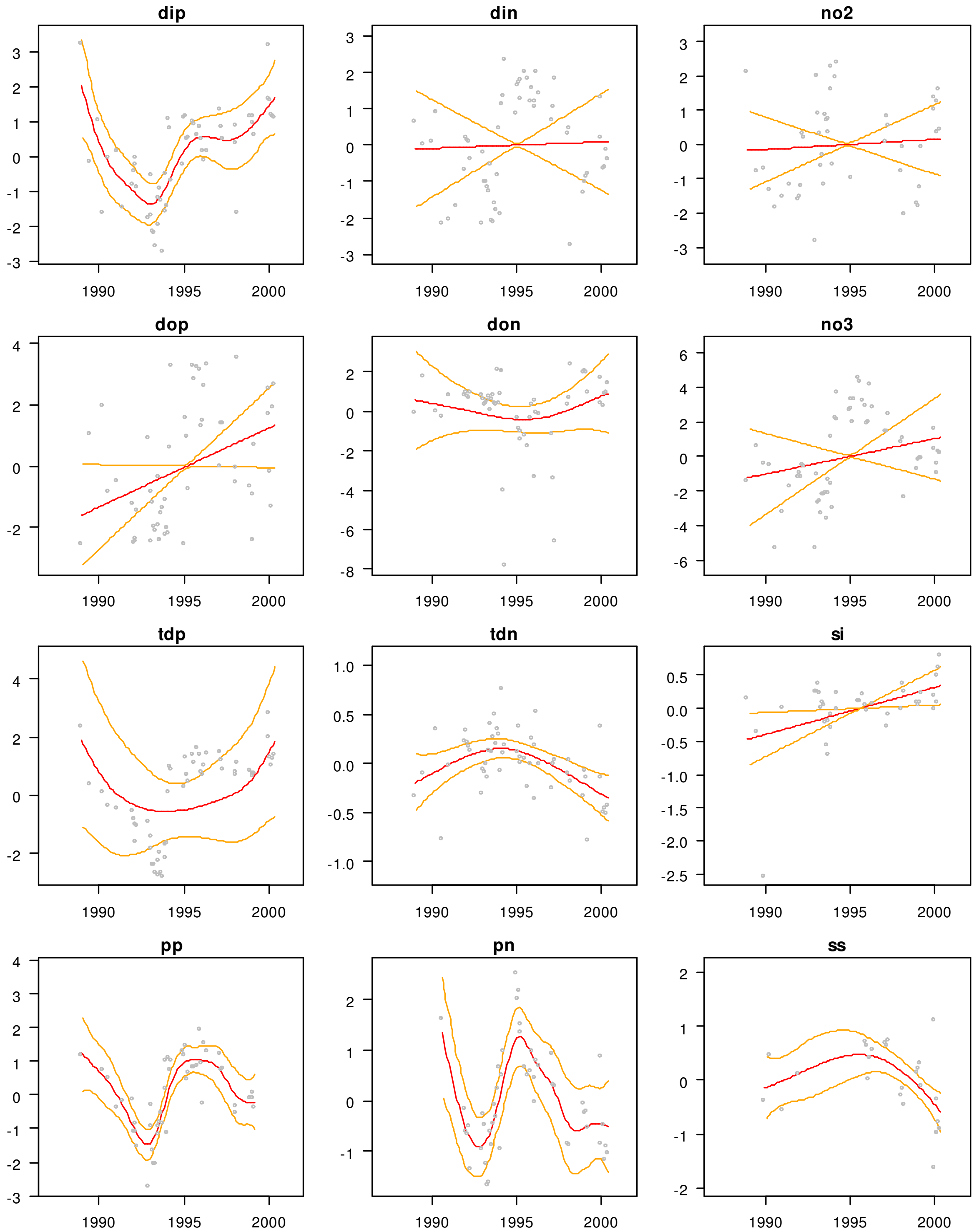
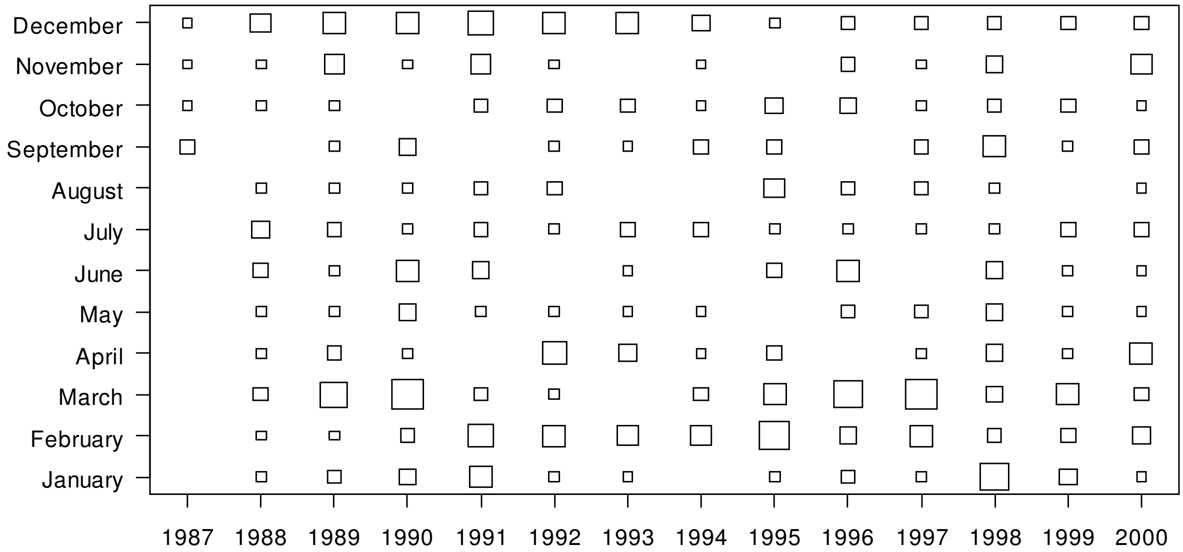


Figure 8. Estimation of temporal trends in water quality parameters for the Burdekin River over the period 1988 – 2000. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [PO_4^{3-}, DIN, NO_2, DOP, DON, NO_3,, TDP (total dissolved phosphorus), TDN (total dissolved nitrogen), Si, PP, PN] and mg L^{-1} for SS. Trends are estimated using smooth terms in a generalised linear model. The trends are adjusted for monthly and river flow effects (smoothed) and autocorrelation between successive months. The smoothness of the temporal and river flow effects are estimated by cross validation. The trends show systematic variation with a weak linear trend for Si, but all strong effects are highly non-linear (DIP, PN and SS). The explanation of the non-linear effects is not evident.

Tully



Burdekin

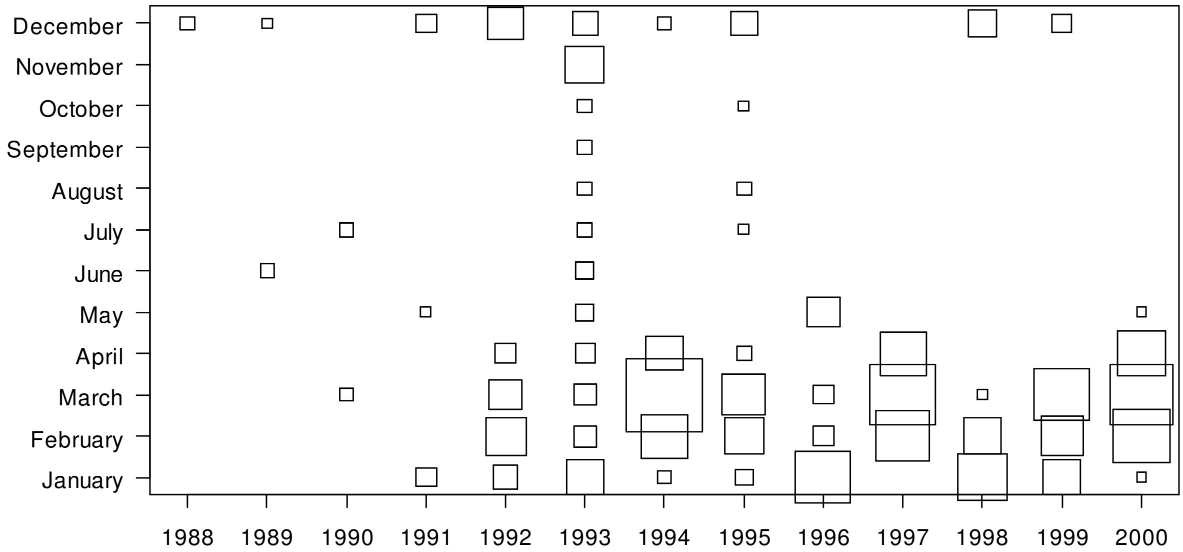


Figure 9. Illustration of the sampling intensity of the AIMS monitoring in the Tully and Burdekin Rivers. The size of each square of the plot is proportional to the square root of the sampling frequency for the given month of the given year. The smallest square represents one sampling occasion for both Tully (upper panel) and Burdekin (lower panel). The range for Tully was 0-9 samples in a month, and for Burdekin was 0-59 samples. For the Tully sampling is clearly more evenly distributed over time.

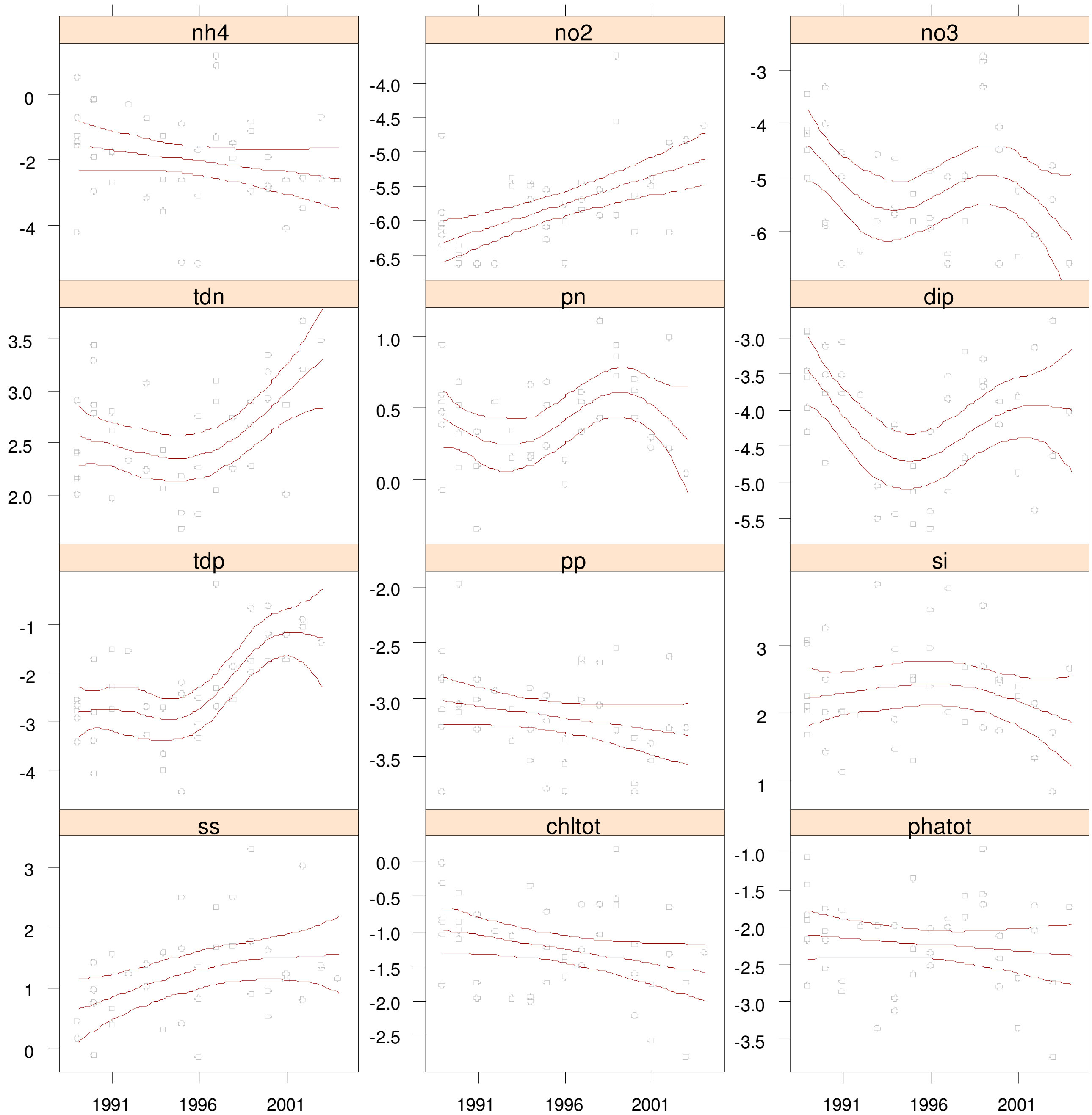


Figure 10. Great Barrier Reef lagoon water quality data analyses. All parameters are plotted on a log (base 2) scale. Units are µM for nutrient species [NH^{4+}, NO_2, NO_3, TDN (total dissolved nitrogen), PN, DIP= PO_4^{3-}, TDP (total dissolved phosphorus), PP, Si], mg L^{-1} for SS and µg L^{-1} for chlorophyll a *(chltot) and phaeophytin (phatot). The values of the 12 water quality parameters are site adjusted (averaged across sites for each visit). Systematic change is evident in NO^{2-}, NO^{3-}, TDN, PN, DIP, TDP and chlorophyll* a.

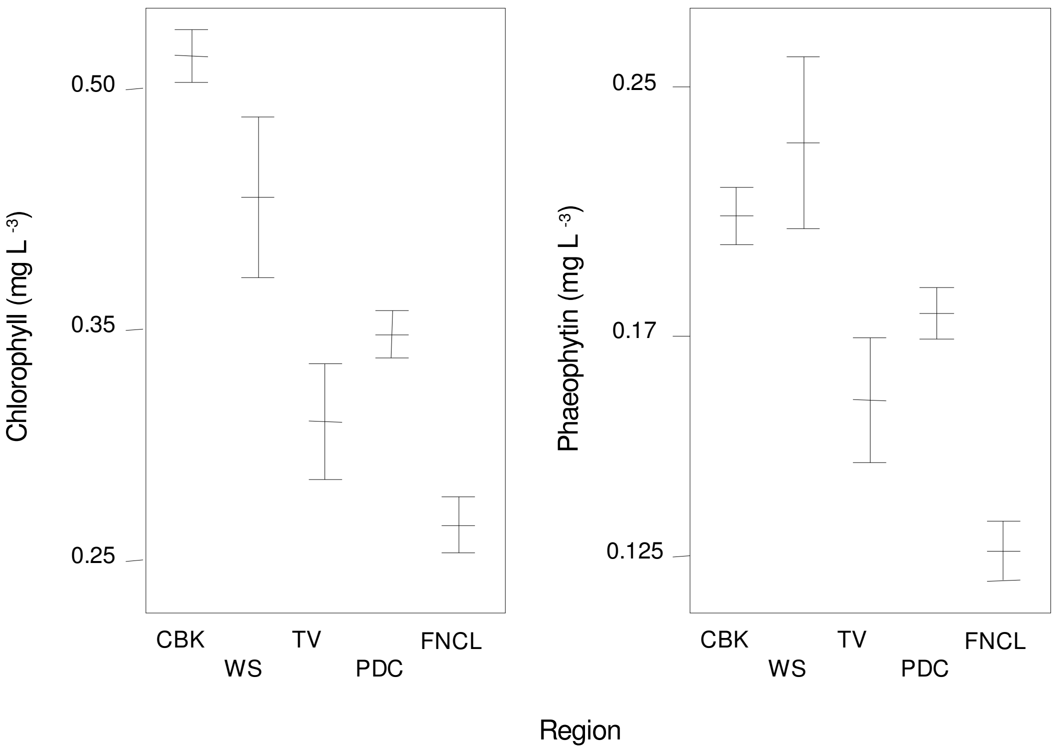


Figure 11. Partial effects plots plots (± 2SE) for chlorophyll a *and phaeophytin broken down by the five regions: Capricorn Bunker & Keppels (CBK), Whitsunday (WS), Townsville (TV), Port Douglas & Cairns (PDC), Far North, Cooktown & Lizard Island (FNCL). CBK is most southerly through to FNCL which is most northerly. A strong decrease from north to south is evident with values of chlorophyll* a *and phaeophytin being approximately twice as high in the south. The ratio of chlorophyll* a *to phaeophytin is typically 2:1.*

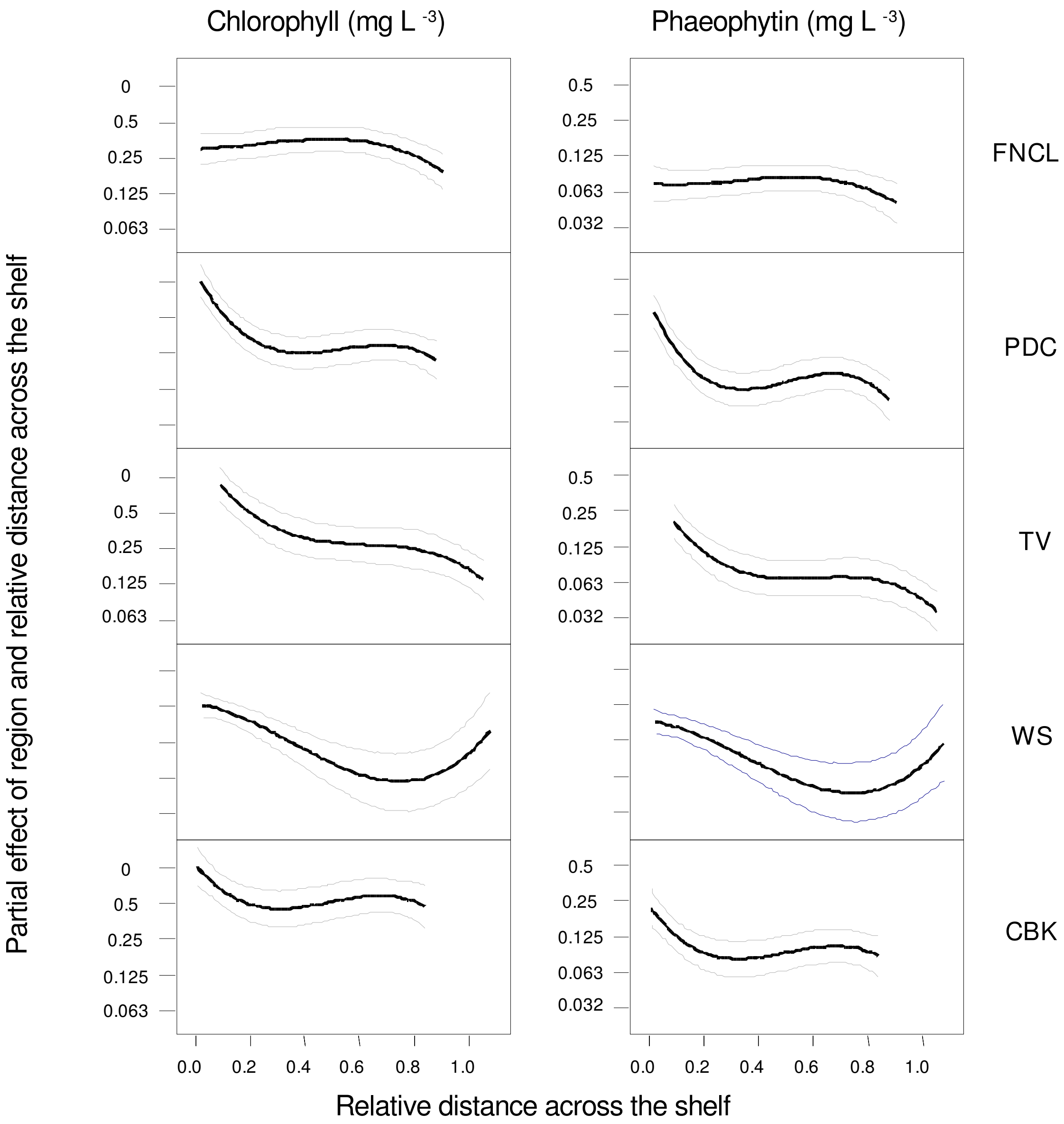


Figure 12. Partial effects plots showing trends across the shelf (± 2SE) for chlorophyll a *and phaeophytin broken down by the five regions. The patterns for chlorophyll* a *and phaeophytin are similar, with strong consistent declines across the shelf in PDC and TV, a quadratic change in WS with high values close to shore and off-shore, and no discernible change in FNCL and CBK. In PDC and TV the cross-shelf values change by a ratio of approximately 4:1.*

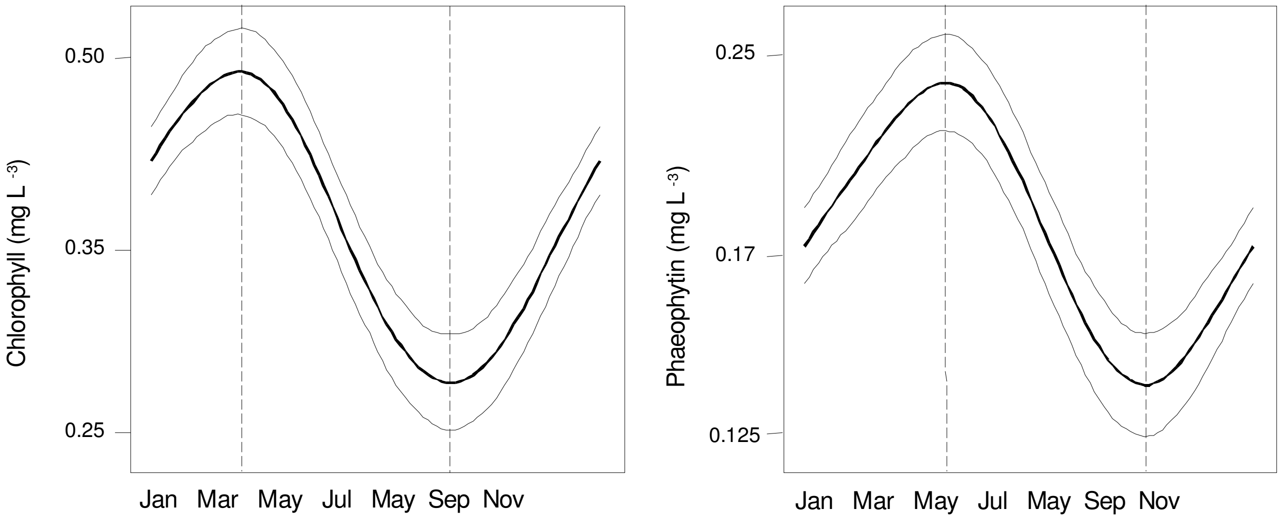


Figure 13. Partial effects plots showing cyclical change within years (± 2SE) for chlorophyll a *and phaeophytin. Patterns for chlorophyll* a *and phaeophytin are similar. Values of chlorophyll* a *are consistently twice that of phaeophytin. The change in phaeophytin lags that of chlorophyll* a *by approximately one month consistently throughout the year.*

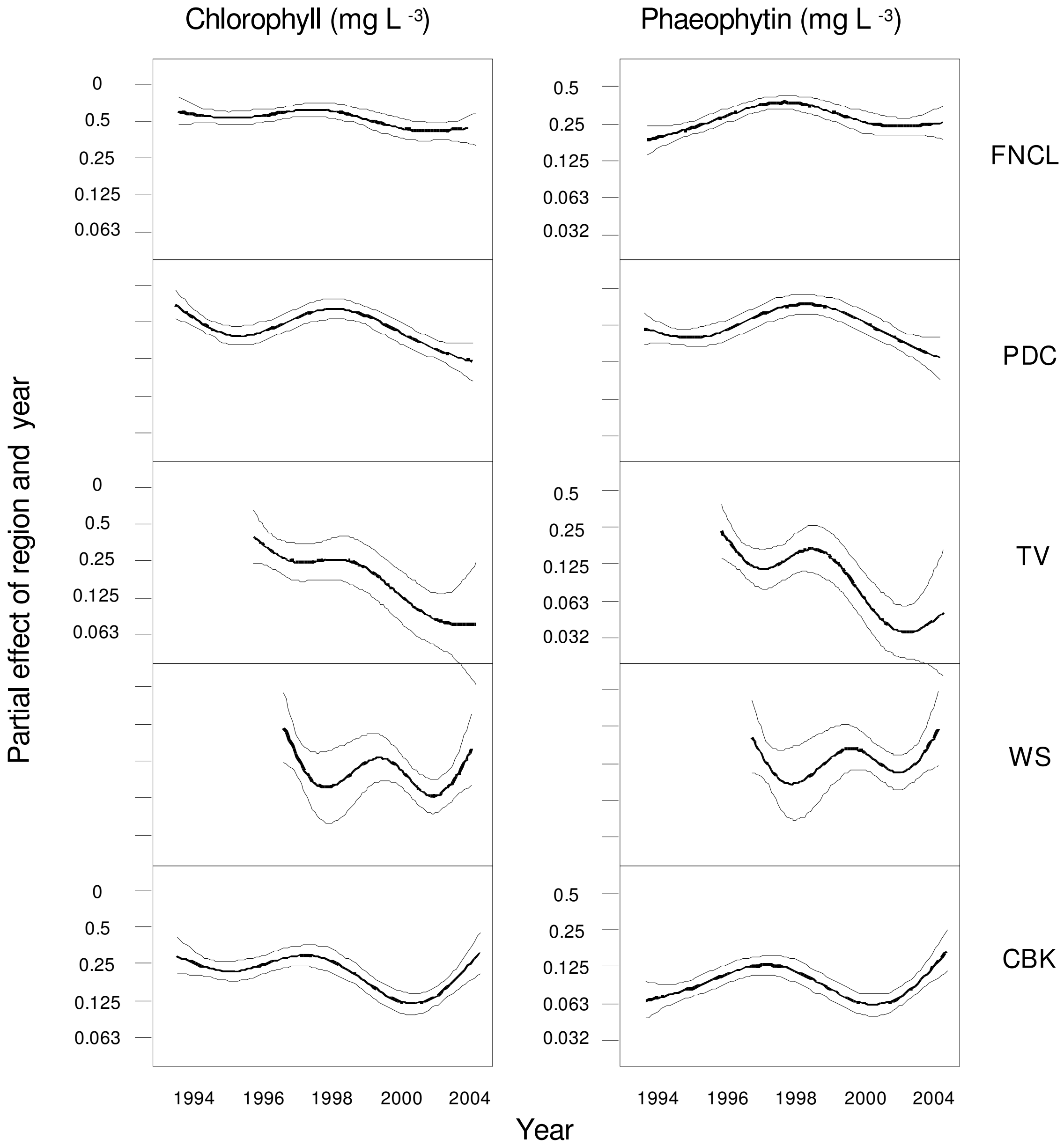


Figure 14. Partial effects plots showing the trends over years (± 2SE) for chlorophyll a *and phaeophytin broken down by regions. The patterns for chlorophyll* a *and phaeophytin are similar, but there are no consistent trends. FNCL, PDC and CBK show non-linear change over years whereas TV and WS show no discernible change.*

Appendix 4: Lagoon water sampling details

Numbers of stations used to compute regional and seasonal nutrient statistics for the Great Barrier Reef lagoon. Only stations where multiple depths were samples are included. All stations in depths < 80 m. Stations affected by river plumes, cyclonic resuspension or major upwelling have been excluded. Abbreviations as in Appendix 1.

Sector	In/Offshore	Seas	NH_4	NO_2	NO_3	TDN	PN	PO_4	TDP	PP	Si	SS	Chl	Pha
Torres Strait	Inshore	Dry	-	-	-	-	-	-	-	-	-	-	-	-
	Offshore	Dry	-	-	-	-	-	-	-	-	-	-	-	-
	Inshore	Wet	-	3	3	3	2	3	3	3	2	3	3	3
	Offshore	Wet	-	17	17	17	17	17	17	17	17	17	17	17
Shelburne Bay	Inshore	Dry	-	3	3	2		2	1		1	2	3	3
	Offshore	Dry	-	10	12	8	5	8	4	5	5	7	12	12
	Inshore	Wet	-	40	40	44	11	47	43	24	41	47	46	46
	Offshore	Wet	-	69	69	70	24	69	67	35	67	72	71	71
Princess Charlotte	Inshore	Dry	14	33	33	33	13	32	31	11	28	30	34	34
Bay	Offshore	Dry	7	59	59	55	9	55	52	9	47	54	54	59
	Inshore	Wet	12	91	92	68	15	93	61	20	94	64	99	74
	Offshore	Wet	4	141	143	92	9	142	80	12	157	95	160	104
Cooktown	Inshore	Dry	1	3	3	3	1	3	3	1	1	3	3	3
	Offshore	Dry	4	4	4	4	4	4	4	4	1	1	4	4
	Inshore	Wet	-	46	46	23	3	44	20	3	46	18	48	19
	Offshore	Wet	2	124	124	27	6	124	25	6	125	15	115	48
Cairns	Inshore	Dry	68	246	246	248	201	256	251	190	249	229	247	242
	Offshore	Dry	7	134	134	137	83	138	130	85	135	113	120	118
	Inshore	Wet	73	631	631	567	389	637	504	379	579	447	666	600
	Offshore	Wet	16	230	228	228	125	234	179	118	210	157	250	235
Innisfail	Inshore	Dry	25	42	42	41	29	48	41	24	48	32	44	44

	Offshore	Dry	18	165	165	36	35	168	33	33	156	27	156	159
	Inshore	Wet	34	65	65	52	27	65	52	30	51	48	56	55
	Offshore	Wet	86	283	282	53	48	290	51	48	276	62	322	322

Appendix 4 continued

Sector	In/Offshore	Seas	NH_4	NO_2	NO_3	TDN	PN	PO_4	TDP	PP	Si	SS	Chl	Pha
Townsville	Inshore	Dry	-	11	11	10	2	13	10	2	9	10	12	11
	Offshore	Dry	2	101	101	31	-	101	29	-	77	51	113	113
	Inshore	Wet	1	17	17	5	-	18	3	-	19	14	18	16
	Offshore	Wet	34	224	218	80	14	230	52	7	226	148	254	249
Pompey Reefs	Inshore	Dry	3	7	7	7	7	7	7	7	7	7	6	6
	Offshore	Dry	7	31	31	31	31	31	31	31	31	19	24	24
	Inshore	Wet	6	57	57	47	28	55	34	24	47	46	61	61
	Offshore	Wet	23	196	190	184	73	2-1	141	72	142	189	205	199
Swains Reefs	Inshore	Dry	-	2	2	2	2	2	2	2	2	2	1	1
	Offshore	Dry	7	37	37	38	26	36	36	26	26	34	37	37
	Inshore	Wet	-	5	5	5	5	5	5	5	5	4	5	5
	Offshore	Wet	2	187	189	203	37	197	137	34	137	206	192	192
Total			**456**	**3314**	**3306**	**2454**	**1281**	**3375**	**2139**	**1267**	**3064**	**2273**	**3463**	**3188**