Water Quality and Ecosystem Monitoring Program
Reef Water Quality Protection Plan

Final Report

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Contents

Acronyms

ACTFR	Australian Centre for Tropical Freshwater Research
AIMS	Australian Institute of Marine Science
AOP	Apparent Optical Properties
CDOM	Coloured Dissolved Organic Matter
COTS	Crown-of-thorns starfish
CRC Reef	CRC Reef Research Centre
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CTD	Conductivity Temperature Depth profiler
DNRMW	Queensland Department of Natural Resources, Mines and Water
DON	Dissolved Organic Nitrogen
DOP	Dissolved Organic Phosphorus
DPI&F	Queensland Department of Primary Industries and Fisheries
ED	Empore Disk (passive sampler)
EnTox	National Research Centre for Environmental Toxicology at UQ
EPA	Queensland Environmental Protection Agency
ERI	Ecosystem Risk Index
FNQNRM	Far North Queensland Natural Resource Management Pty Ltd
GBRMPA	Great Barrier Reef Marine Park Authority
GBRWHA	Great Barrier Reef World Heritage Area
GPC	Gel Permeation Chromatography
HDF	Hierarchical Data Format
HPLC	High Performance Liquid Chromatography
IOP	Inherent Optical Properties
JCU	James Cook University
LAT	Lowest Astronomical Tide
LOD	Limit of detection
LTMP	Long-term Monitoring Program
MERIS	Satellite sensor MEdium Resolution Imaging Spectrometer (operated by European Space Agency)
MODIS	MODerate resolution Imaging Spectrometer (operated by NASA) sensors
MODIS-Aqua	Satellite sensor launched in 2002– a nominal 13:30 equatorial overpass time
MODIS-Terra	Satellite sensor launched in 1999 – a nominal 10:30 equatorial overpass time
MWHW	Mackay Whitsundays Healthy Waterways program
NASA	National Aeronautics and Space Administration
NATA	National Association of Testing Authorities
NH_4	Ammonia
NO_2	Nitrite
NO_3	Nitrate
NRM	Natural Resource Management
PAH	Polyaromatic Hydrocarbons
PCBs	Polycyclic biphenyls
PN	Particulate Nitrogen
PO_4	Phosphate
PP	Particulate Phosphorus
PRC	Performance Reference Compound
QA/QC	Quality Assurance and Quality Control
QHSS	Queensland Health Scientific Services
QPWS	Queensland Parks and Wildlife Services
Reef Plan MMP	Reef Water Quality Protection Plan Marine Monitoring Program
Reef Plan	Reef Water Quality Protection Plan
RPI	River Pollution Index
SEADAS	SesWiFS Data Analysis System

$Si(OH)_4$	Silicate
SIOP	Spectral Inherent Optical Properties
SPMD	Semipermeable Membrane Device (passive sampler)
SS	Suspended Solids
TDN	Total Dissolved Nitrogen
TDP	Total Dissolved Phosphorus
TSS	Total Suspended Solids
UQ	University of Queensland

Executive Summary

The Reef Water Quality Protection Plan Marine Monitoring Programme (Reef Plan MMP) was designed and developed by the Great Barrier Reef Marine Park Authority and is coordinated through the CRC Reef Research Centre on behalf of a consortium of research partners. The consortium includes the Australian Institute of Marine Science, CSIRO, Queensland Department of Natural Resources, Mines and Water, Queensland Department of Primary Industries and Fisheries, Queensland Environmental Protection Agency, Sea Research and The University of Queensland.

This report summarises the activities of the first 17 months of the Reef Plan MMP. This first monitoring period achieved the implementation of all monitoring tasks and the completion of the first year of data collection.

Both the 2004-05 and 2005-06 wet seasons were characterised by freshwater discharges considerably below the long-term averages for all rivers, despite the occurrence of category 5 Tropical Cyclone Larry (TC Larry) in 2006. Very low discharges were recorded from central and southern Great Barrier Reef catchments (Burdekin, Pioneer, Fitzroy and Burnett Rivers). As sediment and associated nutrient exports are generally proportional to freshwater discharge, terrestrial inputs to the Great Barrier Reef lagoon within the last two years were 20 to 50 percent below the long-term average export values.

Concentrations in rivers draining the catchments south of Townsville were generally higher than the concentrations of the Wet Tropics and Cape York rivers. However, all rivers exceeded the Queensland Water Quality Guideline values for most nutrients and suspended sediment concentrations, except for the Pioneer and Burnett Rivers where the wet season flows were significantly lower than long-term averages.

The Burdekin River had the highest suspended sediment exports in 2005-06; on a discharge-weighted basis the Fitzroy had the highest sediment load, followed by the Burdekin River. Nitrogen and phosphorus exports were highest in the Tully and Herbert Rivers, respectively. However, the discharge-weighted loads were highest in the O'Connell River, which was the river with the lowest discharge volume in this year.

In the Reef lagoon, winter median water column concentrations of bio-available inorganic nitrogen were generally very low, while the summer values were elevated, mainly due to high ammonium concentrations. Dissolved organic nitrogen and particulate nitrogen were also higher during the summer sampling, indicating re-suspension and higher plankton biomass. During the 2005-06 winter sampling cruises, elevated levels of phosphate and dissolved organic phosphorus were measured while particulate phosphorus did not vary from the summer concentrations. It is likely that re-suspension of marine sediments led to these higher values, which were also reflected in the higher and more variable values of suspended sediment concentrations in winter.

General patterns that have been previously detected in the long-term chlorophyll dataset were confirmed in the sampling conducted under the Reef Plan MMP. There is a general southward increase in mean chlorophyll a concentration, especially in the coastal zone. There is no significant cross-shelf gradient in chlorophyll concentrations in the Cape York region, while in all sectors further south have significantly higher chlorophyll values inshore than offshore. The dataset to date has shown some long-term variations in mean chlorophyll concentrations, however, only weak linear trends in some regions.

The project was successful in providing satellite-based spatial and temporal information about near-surface concentrations of chlorophyll, suspended solids and vertical light attenuation coefficients in lagoonal and coastal waters of the Great Barrier Reef World Heritage Area. A regional remote sensing water quality algorithm was developed and a viable remote sensing water quality product is now available. The general spatial patterns of chlorophyll concentrations reflect the results from the long-term chlorophyll monitoring program. However, the level of detail from remote sensing is significantly higher than from traditional grab sampling and likely to be more cost-effective in the long term. This technique is likely to replace grab sampling for chlorophyll monitoring in the future, after some further validation and development.

Autonomous data logging instruments were used as a third way to measure chlorophyll and turbidity under the Reef Plan MMP. Test deployments indicated suitability of these loggers to deliver useful data time series. While remote sensing will allow the monitoring of large-scale patterns, autonomous instruments will have the benefit of obtaining high frequency data series at locations of particular interest, e.g. a reef or seagrass bed where long-term monitoring of biological status is undertaken.

The pesticide monitoring tasks demonstrated that the use of passive sampling techniques provides highly reproducible results and allows assessment of seasonal variability as well as variability between sampling sites. Overall, passive sampling results are available from eight inshore reef sites and nine river mouth sites. In total, successful sampling periods were achieved for 39 polar and 32 non-polar passive samplers at inshore reefs and for 35 polar and 21 non-polar passive samplers at river mouth sides. Polar samplers were analysed for a suite of 10 herbicides including two degradation products whereas non-polar samplers were analysed for more than 50 pesticides including degradation products.

The herbicides atrazine and diuron were typically found at the highest concentrations at the majority of sites. Concentrations of most herbicides were consistently higher in rivers and at inshore sites during the wet season. There was a greater variety of pesticides at Wet Tropics river mouth sites; however the total pesticide concentration was higher in rivers of the Mackay Whitsunday region. Conversely, at inshore reef sites, the total detected pesticide concentrations were higher in the Wet Tropics region in comparison to inshore sites further south. The passive sampling data were consistent with a limited number of grab water samples collected in the rivers. However, the passive sampling techniques allowed a broader range of chemicals to be quantified, and were more cost-effective, while the grab water sampling data showed substantial variability.

Mud crab biomonitoring proved complementary to passive sampler monitoring of the same rivers (Chapter 3) which accumulated a distinctly different suite of pesticides. Mud crabs from 7 of the 11 rivers samples (33% of all mud crabs sampled during the monitoring period) contained persistent organochlorine contaminants (OC) such as PCBs, dieldrin and the breakdown products of DDT. Pesticides such as chlorpyrifos and diuron were not detected in individual crabs, probably due to their comparatively short half-lives in the environment and higher polarity, which means they would be absorbed and accumulated less readily, as well as metabolised and excreted more rapidly than DDTs and dieldrin. Rivers with urban inputs such as the Burnett, Pioneer, Fitzroy and Barron contained the highest frequency and concentrations of OCs. Differences in metal and metalloid concentrations in crab

hepatopancreas between rivers were also observed, although these differences may have been related to local geography.

A high level of spatial variation between the Natural Resource Management regions, and between reefs within the regions, of the coral variables monitored here had limited relationship to observed differences in water quality parameters. Exceptions included the consistent finding of a higher cover of macroalgae close to river mouths and the systematic variation of the composition of hard coral communities in accordance with an estimate of risk of exposure to runoff and an index of water quality. In some regions, locations nearer to river mouths had lower densities of recruit-sized corals and the coral recruits comprised fewer genera.

Settlement of coral larvae was estimated in two regions using terracotta tiles as settlement plates. Settlement rates in 2005 were high in comparison to those measured in previous studies. Coral settlement did not show clear relationships with distance from rivers. Many nearshore reefs supported substantial numbers of adult corals of many genera and considerable numbers of juveniles. Exceptions to this were reefs in the Burdekin region, which had low coral cover and low numbers of recruits. To date it is unknown whether the current coral juveniles will survive to form future adult colonies and whether the current, often high-cover, coral communities will be resilient to future disturbance.

Seasonal fluctuations in seagrass cover were observed at the majority of the Seagrass-Watch sites between October 2005 and April 2006. Significant declines in seagrass cover and distribution were detected at sites believed to be effected by TC Larry. Severe declines in intertidal *Zostera* meadows in Gladstone and other southern/central Queensland locations may be related to atypical climate variables such as rainfall, wind and water temperature occurring in the region between October 2005 and April 2006. The assessment of indicators of seagrass reproduction (e.g. flowers and seeds) indicated that all seagrass meadows except those at Lugger Bay showed the capacity to recover from short term disturbance via seed banks.

Low but detectable concentrations of diuron in seagrass meadow sediments were recorded in both years. These levels were well below previously recorded concentrations. Longer term and potentially more frequent sampling of herbicide concentrations would be necessary to resolve spatial and temporal variability.

Sediment nutrients were found in levels similar to those previously recorded. The relationships between tissue nutrients with sediment nutrients, sediment type and the delivery of these nutrients remain unclear for Great Barrier Reef seagrasses and require future process studies.

Our understanding of the health of the GBR ecosystem has been enhanced by the data obtained during the reporting period. Some notable monitoring results were achieved:

- Eight of ten priority rivers exceeded the Queensland Water Quality Guideline values for most water quality variables.
- Elevated levels of nutrient and sediment in the near shore lagoon were found to be localised and weather-dependent (land run-off after flooding rain and re-suspension by storm events). The nearshore lagoon is well mixed along the coast, confirming that river inputs are likely to be widely distributed along shore resulting in generally low concentrations of nutrient and sediments in the nearshore lagoon during these low freshwater flow periods.

- The herbicides, atrazine and diuron are typically found at detectable levels at river mouths, inshore reef and intertidal seagrass locations, mostly with elevated concentrations during the wet season. The ecological consequences of low level chronic exposure are yet to be ascertained.
- Some coral reef health parameters could be directly linked to the measured water quality variables but not all coral health parameters.
- At present, hard coral cover and species number and seagrass cover was high at most locations, and both coral reefs and intertidal seagrass meadows showed capacity for recovery from short-term disturbances, which are important in shaping these ecological communities.

This first year of monitoring has strengthened our view that processes shaping biological communities are complex and may be based on local interactions of various factors, such as water quality, climate change and physical disturbance. More frequent and targeted monitoring of environmental parameters will further improve our understanding. The long-term monitoring under Reef Plan MMP, as well as complimentary process-oriented research of the environmental implications of water quality on Great Barrier Reef ecosystems, will be an active measure of changes in the Great Barrier Reef water quality status.

1. Introduction

CRC Reef Research Centre (CRC Reef) was contracted by the Great Barrier Reef Marine Park Authority (GBRMPA) on 24 March 2005 to provide a comprehensive Great Barrier Reef lagoon monitoring program, the Reef Water Quality Protection Plan Marine Monitoring Programme (Reef Plan MMP). The agreement between the GBRMPA and the CRC Reef is referred to as the Head Contract in this report.

The monitoring program is grouped into four subprograms:
Task 1: River Mouth Monitoring
Task 2: Inshore Marine Water Quality monitoring
Task 3: Marine Biological Monitoring
Task 4: Biomarker and Bioaccumulation Monitoring

To accomplish these tasks CRC Reef has sub-contracted seven monitoring providers (under a co-investment model) with a long-term track record of monitoring and research in the relevant areas required by GBRMPA. These are:

- The Australian Institute of Marine Science (AIMS) - Task 1 to 4.
- The University of Queensland - National Research Centre for Environmental Toxicology at UQ (EnTox) – Task 1, 2 and 4.
- The Commonwealth Scientific and Industrial Research Organisation (CSIRO) - Task 2.
- The Queensland Department of Primary Industries and Fisheries (DPI&F) – Task 3.
- The Queensland Department of Natural Resources, Mines and Water (DDNRMW) – Task 1.
- The Queensland Environmental Protection Agency (EPA) – Task 4.
- Sea Research – Task 3.

The first year of the Reef Plan MMP involved setting up of new sampling sites (river loggers, coral and seagrass monitoring sites) and community engagement for collection and preparation of samples (river mouth monitoring, chlorophyll monitoring- both cross-shelf transects and coastal sites, pesticide passive sampler deployment, mudcrab sampling) and for seagrass monitoring. Only a few tasks had components which continued already established monitoring activities projects, e.g. the cross-shelf chlorophyll monitoring transects and seagrass monitoring at a number of locations.

As an ongoing activity, AIMS Mudloggers were deployed in 6 and 10 of the priority rivers during the 2004-05 wet season. For the 2005-06 wet season, ten new AIMS mudloggers were constructed and deployed in all ten priority rivers. Water sampling was undertaken in all 10 priority river catchments by community volunteers, AIMS and DNRMW personnel and sampling frequencies were sufficient for statistical modelling of nutrient exports from 7 of the 10 priority catchments.

Water quality monitoring in the nearshore lagoon was carried out twice in 2005-06, at locations in proximity of the inshore reefs monitored under Reef Plan MMP and in open waters. There are very few long-term datasets available for comparisons of the nutrient concentrations measured in the inshore lagoon under the current Reef Plan MMP monitoring. The longest time series of water quality data for the Great Barrier Reef was collected by AIMS in coastal waters between Cape Tribulation and Cairns from 1989 to the present. Sampling of these stations was continued under Reef Plan MMP.

Surface chlorophyll concentrations in Great Barrier Reef waters have been measured since 1992 as part of a long-term monitoring program. The Reef Plan monitoring continued a large number of stations from the long-term chlorophyll program and added 11 additional locations in the coastal region. Chlorophyll and suspended solids concentrations were also obtained from analysing remote sensing imagery. The validation of remote sensing information with the long term chlorophyll monitoring dataset was carried out, and despite a number of compatibility issues showed a sufficient general agreement to consider using remote sensing-based water quality information in future for the variables that remote sensing can measure, e.g. chlorophyll and suspended solid concentrations.

Monitoring of trace organic pollutants in rivers and marine waters is challenging. The Reef Plan MMP activities have focused on the use of passive sampling techniques for assessing the delivery of pesticides to marine waters from the Great Barrier Reef catchment and provided a baseline for the assessment of long-term of trends of anthropogenic pollutants.

As a second tier to monitoring pesticides and other persistent contaminants, mud crabs (*Scylla serrata*) were collected by commercial fishers in 2005 and 2006 from all ten priority rivers. The digestive gland (hepatopancreas) was analysed for contaminants, including insecticides, herbicides and metals. The preferential accumulation of non-polar persistent organochlorine pesticides in *S. serrata* complemented other monitoring methods in the Reef Plan MMP such as passive samplers which better accumulate polar organic compounds.

The ecosystem monitoring included inshore coral reefs and intertidal seagrass beds. The coral monitoring program surveyed cover of benthic organisms, the numbers of genera and the size distributions of coral colonies at 35 inshore reef locations in four NRM regions.

Intertidal seagrass monitoring occurred at the 22 sites identified for the Reef Plan MMP long-term intertidal monitoring. Community-based monitoring (Seagrass-Watch) occurred in April and October of each year at sites that had been established prior to Reef Plan MMP. Mapping the 100m edge of each monitored seagrass meadow, assessments of reproductive health and collection of sediments for herbicide analysis was carried out twice during the monitoring period. Sediment and tissue nutrient samples from all 22 monitoring sites were collected once.

This report provides a detailed overview and findings of the activities undertaken as part of the Reef Plan MMP from June 2004 to May 2006.

2. River Sediment and Nutrient Loads

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Introduction

Freshwater runoff from rivers is the principal carrier of sediment, nutrients (e.g. nitrogen, phosphorus, silicon, iron) and other contaminants from the land to the Great Barrier Reef (Furnas, 2003). Each year, on average, approximately 70 km^3 of freshwater is discharged by rivers and streams into the Great Barrier Reef lagoon (Furnas, 2003). The total annual runoff volume may vary from year to year as much as 3-fold from the average due to climatic variations in overall catchment rainfall. The quantity of sediment and nutrients presently delivered to the coast by individual rivers is correlated with the volume of freshwater runoff (Furnas, 2003). This has likely always been the case. However, a variety of evidence clearly indicates that the delivery of sediments and nutrients from the Great Barrier Reef catchments to the Great Barrier Reef lagoon has increased on the order of 2- to 4-fold over the last 150 years (e.g. Great Barrier Reef Protection Interdepartmental Committee Science Panel, 2003; Furnas, 2003; Brodie *et al*., 2004). This increase means that the loads of sediment and nutrients carried by a given volume of runoff have correspondingly increased over this time period. As a consequence, land-management efforts to stabilize or reduce sediment and nutrient runoff to the Great Barrier Reef must focus upon reducing the volume-specific loads of these materials.

Nutrients are transported by rivers in a variety of forms. Most nutrients are in some type of particulate or particle-associated form (50-80% depending on the specific nutrient and the catchment), either as part of particulate organic matter, as inorganic ions bound to the surface of fine sediment particles, or as a constituent of the soil and mineral grains making up a sediment particle. The remaining nutrients are transported as dissolved inorganic ions and as dissolved organic matter. Slow, but significant increases in base-flow nitrate, phosphate (equivalent to filterable reactive phosphorus) and particulate nitrogen were observed in the lower Tully River (Wet Tropics) over a 10-year period (1990-2000) following an intensification of agricultural land-use in the catchment, indicating greater losses from land sources. Similar changes were not observed over the same period in the Burdekin River (Dry Tropics), in part because of the high natural variability in water flow, sediment and nutrient loads in the Burdekin, and because most nutrients carried by this river come from the unfertilised eroded soils of grazing lands (Furnas, 2005; De'ath 2005).

Many hundreds of rivers and streams discharge directly into the Great Barrier Reef World Heritage Area (GBRWHA), or the discharge is carried into the GBRWHA by coastal currents. These rivers and streams are regionally aggregated into 38 drainage basins. Thirty-five (35) of these drainage basins are located on the mainland. From these many streams and rivers, ten (10) priority rivers were selected for long-term monitoring (Figure 2.1). These rivers (Table 2.1) were chosen on the basis of their contribution to total catchment area and freshwater discharge, the extent and degree of agricultural development within the catchments (summarised in Table 2.1) and the degree with which estimates of nutrient, sediment and pollutant exports from these rivers could be usefully extrapolated to rivers of similar characteristics which could not be monitored due to cost and logistic constraints.

Collectively, these ten rivers account for 83 percent of the area of the Great Barrier Reef catchment and 54 percent of the average annual freshwater discharge.

Table 2.1. Priority rivers in the Great Barrier Reef catchment monitored under the Reef Plan MMP.

				Land use (% of catchment area)*				
NRM Region	**River**	**Catchment area (km^2)**	**Average annual discharge (km^3)§**	**Forest/ savannah**	**Grazing**	**Sugar**	**Other crops**	**Other**
Cape York	Normanby[2#]	24,408	4.9	24	76	0	0	1
Wet Tropics	Barron[1, 2]	2,136	0.8	43	45	0	6	5
	N Johnstone[1, 2]	925	1.8	55	20	20	2	3
	Tully[1, 2]	1,683	3.3	66	16	13	1	3
	Herbert[1, 2]	9,843	4.0	24	66	7	1	2
Burdekin	Burdekin[2]	130,126	10.3	4	95	0	1	0
Mackay Whitsunday	O'Connell[1]	2,387	1.5	30	50	16	0	3
	Pioneer[1]	1,570	1.2	37	32	27	0	3
Fitzroy	Fitzroy[1]	142,537	6.1	10	85	0	5	1
Burnett Mary	Burnett[1]	33,248	1.2	16	80	1	2	1

[1]Rivers where ambient sampling was carried out by community or other interest groups
[2]Rivers where sampling was undertaken under flood conditions by DNRMW staff during 2005-2006 wet season.
[#]A new gauging station was set up in the Normanby River at Kalpowar during the 2005-06 wet season.
§ For period 1969-1994 (Furnas, 2003)
* Land use data from Brodie et al. 2004, note that land use data for the N Johnstone River are those for the entire Johnstone catchment.

The river-mouth monitoring task is based on the straightforward presumption that delivery of sediments and nutrients to coastal waters in the Great Barrier Reef (GBR) lagoon is regulated by the volume of freshwater runoff and land-based factors such as terrain, soil, vegetation cover and land use which affect the quantity of freshwater, sediment and nutrients delivered to stream and river systems within the individual catchments.

Export = *function* (freshwater runoff volume, terrain, soil, vegetation cover, land-use)

Freshwater runoff is determined in a more localized and complex fashion by the quantity and distribution of rainfall within individual catchments, catchment size, vegetation cover and land use. In all catchments, the degree of vegetation cover has a significant influence on the retention of water within watersheds and the quantity of rainwater returned to the atmosphere through transpiration by plants. In most parts of the Great Barrier Reef catchment, vegetation cover is strongly influenced by both natural (rainfall, soils) and human (grazing, land-clearing, crop-cultivation) factors.

Cover = *function* natural factors (rainfall, soil)
+/- human factors (grazing, land clearing, crop cultivation)

It is the human element that the Reef Water Quality Protection Plan seeks to influence.

As a part of the Reef Water Quality Protection Plan (Reef Plan), the River Mouth Monitoring task was established to provide land users and natural resource managers with ongoing estimates of the delivery of nutrients, sediments and pollutant materials to the Great Barrier Reef.

The key aims of this task are:

- To detect long-term trends, if any, in concentrations and loads of sediments and nutrients at the mouths of significant or representative rivers entering the Great Barrier Reef lagoon, and thereby, to assist assessments of the effectiveness of land-based delivery reduction measures under the Reef Plan.
- To work closely with and involve community partners in the monitoring tasks to promote broad acceptance and ownership of the Reef Plan.

This report chapter reports results from the end of river monitoring at ten priority rivers, specifically freshwater discharge volumes, turbidity measurements of automated turbididty loggers, grab sampling to monitor nutrient and suspended sediment concentrations and the calculation of sediment and nutrient loads using the monitoring data.

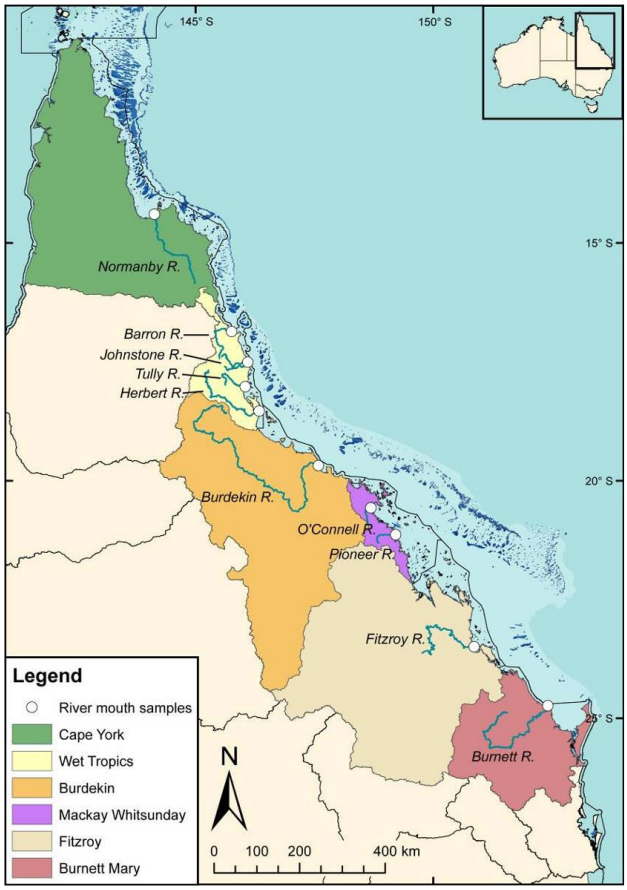


Figure 2.1. Location of river mouth sampling sites of the ten Reef Plan priority rivers monitored for end-of-catchment nutrients and suspended sediments during 2005-06.

Methods

Water sampling was carried out at the river mouths of the ten Reef Plan MMP priority rivers (Figure 2.1). All rivers had turbidity loggers installed and had some degree of grab sampling (Table 2.1).

River turbidity measured by AIMS Mudloggers

Autonomous turbidity loggers developed and constructed by the Australian Institute of Marine Science ("AIMS Mudloggers") were deployed in the lower reaches of 10 priority rivers over the 2005-06 wet season to continuously measure concentrations of fine suspended sediment carried by these rivers. In the first year of the Reef Plan MMP (2004-05 wet season), Mudloggers were also deployed in seven rivers, including six of the ten priority rivers. The turbidity records obtained by the Mudloggers were used in conjunction with time series of river discharge provided by the Queensland Department of Natural Resources, Mines and Water (hereafter DNRMW) to calculate loads of fine sediment carried by the rivers into the Great Barrier Reef lagoon.

The AIMS Mudloggers contain dual LED-based transmissometers (15mm and 85mm pathlengths) to measure *in situ* turbidity associated with suspended sediment over a 0-5 g L^{-1} concentration range (Mitchell and Furnas, 2001). The light beams of the two transmissometers in each logger are attenuated by sediment and other particles in the light path through absorption and scattering. Test measurements indicate that light absorption by coloured organic matter in river waters is small compared to absorption and scattering by particles. The turbidity readings were primarily affected by the concentration of very fine (<10 µm) silt and clay particles. Most of the particulate nutrients carried by rivers are attached to these small particles. The degree of light attenuation is used to calculate the fine sediment concentration. The loggers concurrently record water depth and internal temperature. Readings were taken at 30 minute intervals. Automatic wiper brushes cleaned the optical surfaces of the transmissometers on an hourly basis, allowing the Mudloggers to run unattended for long periods. The electronic and mechanical components of the Mudloggers are contained within a pressure housing that allows the logger to run fully submerged. Internal batteries provide sufficient power for deployments exceeding 6 months. At the deployment sites, the loggers are locked in a galvanised steel mesh cage for protection against floating debris and vandalism. During deployments, the cages were mounted on structures in the river (e.g. bridge support, water intake tower) which provide further protection from floating debris.

Sampling locations and periods

Mudloggers were deployed at sites in the lower freshwater reaches of the ten priority river of the Reef Plan MMP (Figure 2.1, Table 2.1).

AIMS Mudloggers have been deployed over a number of wet seasons, including the 2004-05 wet season, in the Normanby, Barron, North Johnstone, Tully, Burdekin and Fitzroy Rivers (Table 2.2). The Russell River was sampled once (2004-05). A number of deployments were previously made in the Herbert River between 1996 and 2000. Prior to the 2005-06 wet season new deployment sites were established in the O'Connell, Pioneer and Burnett Rivers, and the Herbert River site was re-established (Table 2.3). Site selection is determined by the availability of safe and legal access to a bridge or other secure structure in a freshwater reach close to the river mouth, and by proximity to the most downstream DNRMW gauging station in these rivers.

Most river discharge datasets obtained from DNRMW for the 2005-06 financial year are currently incomplete as validated data from the latter part of the wet season was not available when report preparation commenced. Differences between validated and un-validated flow data sets, where they occur, are typically small relative to overall seasonal and annual flow. For the purpose of analysing Mudlogger data sets, discharge records were interpolated by DNRMW to 30-minute intervals (cubic m per second – cumecs), matching the logger sampling rate.

Table 2.2. Details of AIMS River Logger deployments over the 2004-05 wet season.

NRM Region	River	Logger in	Logger out
Cape York	Normanby	15-Dec-04	20-Jul095
Wet Tropics	Barron	14-Dec-04	13-Jul-05
	Russell	02-Dec-04	12-Jul-05
	North Johnstone	02-Dec-04	12-Jul-05
	Tully	02-Dec-04	12-Jul-05
Burdekin	Burdekin	30-Nov-04	9-Jul-05
Fitzroy	Fitzroy	22-Dec-04	14-Jul-05

Table 2.3. Details of AIMS River Logger deployments over the 2005/2006 wet season.

NRM Region	River	Logger in	Logger out
Cape York	Normanby	16-Nov-05	7-Jun-06
Wet Tropics	Barron	18-Nov-05	2-May-06
	North Johnstone	15-Nov-05	6-Jun-06
	Tully	15-Nov-05	6-Jun-06
	Herbert	15-Dec-05	31-May-06
Burdekin	Burdekin	15-Dec-05	6-Jun-06
Mackay Whitsunday	O'Connell	23-Nov-05	22-May-06
	Pioneer	22-Nov-05	22-May-06
Fitzroy	Fitzroy	23-Nov-05	22-May-06
Burnett-Mary	Burnett	2-Nov-05	26-May-06

Calculation of wet season fine sediment exports

Mudlogger data records were downloaded in the field at the time of instrument recovery and the raw data records and associated metadata were stored in a database. Post-deployment checks were carried out with all instruments to determine levels (if any) of transmissometer drift over the course of the deployments. Raw instrumental records of light transmittance and pressure (instrument depth) recorded by the turbidity loggers were first de-spiked and corrected for instrumental drift, if required, using time-series data editing software (Whisky-

TV®, Kisters). After removal of spikes and other unsuitable data, gaps in the record were filled by linear interpolation.

Fine sediment concentrations (mg L^{-1}) over the durations of the acceptable data during the deployments were estimated from the Mudlogger records by a two-stage process.

First, time series of raw Suspended Sediment (SS) concentration (TSS_{raw}) were calculated from linearised records of relative transmittance (I_o = nominal 100% transmittance value for the individual transmissometer in air or distilled water) recorded by the Mudlogger using the equation:

$$TSS_{raw}\ (g\ L^{-1})_i = \mathbf{a}\ [-ln(I_{measured}/I_o)]_i + \mathbf{b}$$

Where **a** and **b** are laboratory-derived coefficients obtained by measuring relative transmittance with one or more loggers in a range of suspended sediment concentrations produced by serial additions to or dilutions of a sifted fine sediment suspension. The value of the factor (**a**) is defined by the sediment mass-turbidity characteristics of the particular batches of suspended sediment used in the calibration process. For the data presented herein, **a** values of 0.102 g L^{-1} and 0.961 g L^{-1} were used for the 85 and 15 mm channels, respectively (Furnas, in prep.). The use of relative transmittance ($I_{measured}/I_o$) means that slope calibrations for individual transmissometer are not required.

Second, the calculated time series of TSS_{raw} values produced from Mudlogger turbidity records using the above equation were then converted to in-river suspended sediment concentrations (TSS_{corr}) comparable to values determined by manual grab sampling and gravimetric analysis (see below for grab sampling methods) using an appropriate transfer function derived from correlations between suspended sediment concentrations measured in grab samples and concurrent Mudlogger-derived values of suspended sediment at the same site.

For the 2005-06 data, the transfer functions are defined by the relationships:

For TSS_{raw} values in the Tully River:

$$TSS_{corr}\ (mg\ L^{-1}) = 11.94\ \exp(0.0066\ TSS_{raw}) - 8.2 \quad r^2=0.89$$

For TSS_{raw} values in "wet" rivers other than the Tully:

$$TSS_{corr}\ (mg\ L^{-1}) = 0.553\ TSS_{raw} + 0.01 \quad r^2=0.96$$

For TSS_{raw} values in "dry" rivers:

$$TSS_{corr}\ (mg\ L^{-1}) = 0.356\ TSS_{raw} - 44.0 \quad r^2=0.92 \text{ (Burdekin River)}$$

$$TSS_{corr}\ (mg\ L^{-1}) = 0.385\ TSS_{raw} + 52.7 \quad r^2=0.90 \text{ (Fitzroy River)}$$

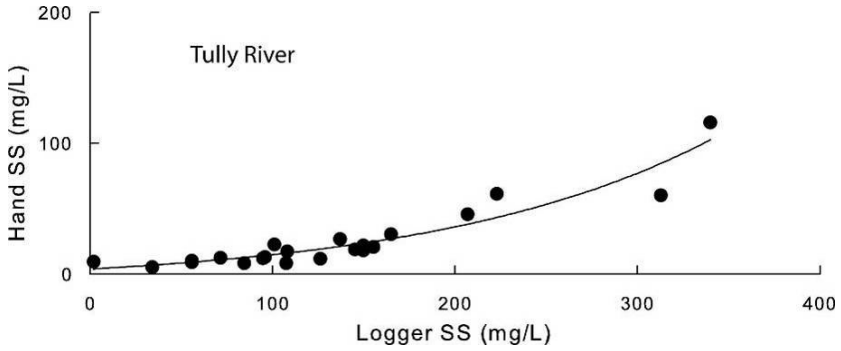


Figure 2.2. The relationship between total suspended sediment (TSS) concentrations in the Tully River determined by manual sampling/gravimetric analysis and initial-pass concentrations (TSS_{raw}) calculated from concurrent Mudlogger turbidity measurements. The fitted line was used to calculate corrected TSS values (TSS_{corr}) from the initial-pass TSS_{raw} values.

The transfer function for the Tully River exhibited a distinct non-linear form which was best fitted by an exponential relationship. In low-flow, low-sediment situations, the initial calculation of TSS_{raw} from logger turbidity values over-estimated the fine sediment concentration relative to hand-collected values. The difference diminished as stream energies and sediment loads increased.

For 'wet' rivers other than the Tully (Normanby, Barron, North Johnstone, Herbert, O'Connell, Pioneer), there was no obvious differentiation between rivers in their sediment mass-turbidity relationships. Accordingly, these data were pooled to derive a single transfer function.

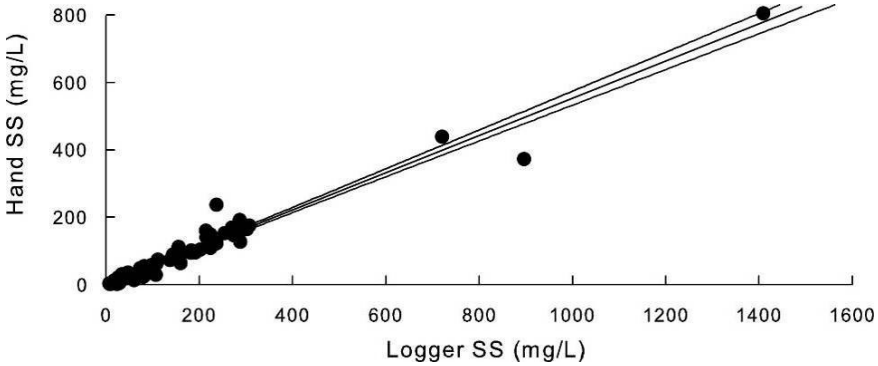


Figure 2.3 The relationship between total suspended sediment (TSS) concentrations in the other "wet" rivers determined by manual sampling/gravimetric analysis and initial-pass concentrations (TSS_{raw}) calculated from concurrent Mudlogger turbidity measurements. The fitted line (with 95% confidence band) was used to calculate corrected TSS values (TSS_{corr}) from the initial-pass TSS_{raw} values.

The sediment mass – turbidity transfer functions for the Fitzroy and Burdekin Rivers were different from those of the wet rivers, most likely because of the higher levels of dispersible clays in soils from these catchments.

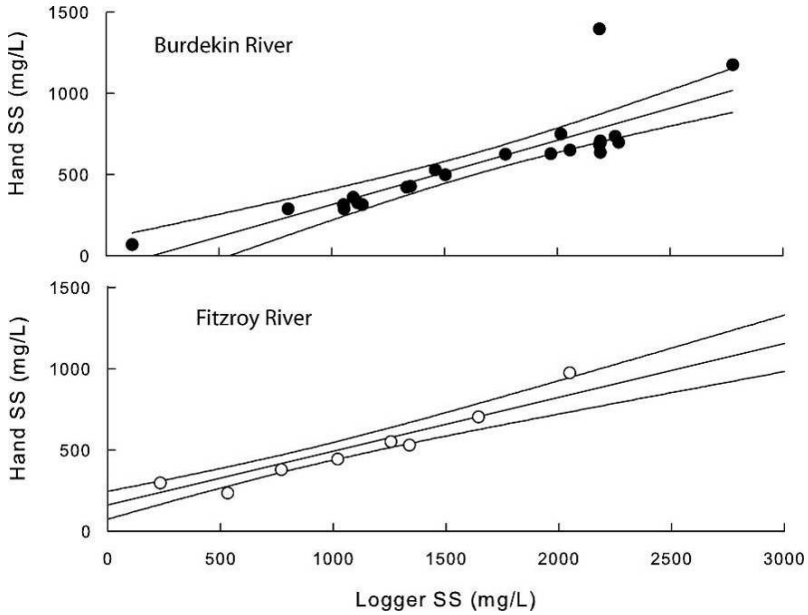


Figure 2.4. The relationships between total suspended sediment (TSS) concentrations in the Burdekin and Fitzroy Rivers determined by manual sampling/gravimetric analysis and initial-pass concentrations (TSS_{raw}) calculated from concurrent Mudlogger turbidity measurements. The fitted lines (with 95% confidence limits) were used to calculate corrected TSS values (TSS_{corr}) from the initial-pass TSS_{raw} values.

Instantaneous sediment fluxes (g sec^{-1}) are calculated as the product of the estimated suspended sediment load (g m^{-3} = mg L^{-1}) and the concurrent discharge rate (m^3 sec^{-1}). Total freshwater discharge (km^3)[1] and sediment load (10^6 tonnes) for the logger deployment period are then calculated by a progressive summation of the product of the half-hourly flows and the concurrent corrected Mudlogger suspended sediment (TSS_{corr}) time series by trapezoidal integration.

$$\text{Sediment load (tonnes)} = \sum_{\text{Nov-May}} [TSS_{corr}]_n \times Q_n$$

Where TSS_n is the average suspended sediment concentration (g m^{-3}) and Q_n is the concurrent average instantaneous river discharge (cumecs) in successive 30-minute time interval.

The total sediment load carried by individual rivers is herein expressed in tonnes (10^6 gm = 10^3 Kg). Because the total sediment load carried by a river in any year is strongly dependent upon the volume of freshwater discharge (related to rainfall and affecting the degree of in-catchment erosion), a more appropriate measure for comparison between rivers and between years is the volume-weighted average load (tonnes/Km^3 of discharge). This value is nominally equivalent to more commonly expressed (and directly measured) concentration units (mg L^{-1} or g m^{-3}). Direct comparisons between the volume-weighted average load over an annual or seasonal period and directly measured concentrations, however, should be avoided or done with the greatest caution as the former value is a derived parameter that smooths the natural temporal variability of sediment (or nutrient) concentrations found within rivers and streams.

[1] 1 Km^3 = 10^9 m^3 = 10^6 megalitres = 10^3 gigalitres

Several factors may influence the accuracy of estimates of wet-season sediment exports derived from instrumental records of turbidity. These include: instrumental variability and factors that affect instrument performance; uncertainties in estimate of instantaneous river flow, temporal variability in river turbidity and temporal variability in the turbidity-sediment mass relationship. To the extent possible, we have attempted instrumental errors through pre- and post-deployment performance checks and 'cleaning' of instrumental data records to remove dubious data affected by light path blockages or dirty optics. Instrumental performance is affected to some degree by temperature, but this affect is greatest when the loggers are out of the water and no fluxes are being recorded. This data is normally deleted. Temperatures are more stable and closer to calibration conditions when the instruments are underwater and recording useful data. Uncertainties in instantaneous river flow are related to the accuracy of DNRMW gauging and flow calibrations. DNRMW hydrographers have indicated that while the river remains within banks their data is most likely to be within 5% of the actual flow value with an outside error estimate of 10%. This will vary between rivers and over time as local erosion and deposition change the shape of the channel at the gauging site. Temporal changes in turbidity over the wet season and during individual events is well-covered by the 30-minute sampling rate of the turbidity loggers. Individual events are covered by >10 to several thousand data points. The accuracy of sediment mass – turbidity relationships is currently constrained by the number of simultaneous data points (see above). Existing correlations are quite tight (all with r^2 values close to or > 0.9), but the sampling is not intensive across all rivers and a range of events.

Concentrations of suspended sediment and nutrients in river water

2004-05 Wet Season

In 2004/05 (pre-contract), a limited number of water quality samples for analysed of nutrient species and suspended sediment concentrations were collected in the Burdekin, North Johnstone, South Johnstone and Barron Rivers by DDNRMW personnel during a number of wet season flood events in January and February 2005.

Water sampling in the Burdekin catchment was undertaken during a flood event from 23 January to 03 February 2005 (see Brodie *et al*., 2005). Sampling was undertaken at five sampling locations in different sub-catchments; only one site, Home Hill, corresponded to the Burdekin river mouth monitoring location prescribed under the Reef Plan MMP. The sampling period and frequency differed between sub-catchments but spanned 25 January to 2 February 2005. Results from this event sampling were reported in the Reef Plan MMP August 2005 Progress Report and in Brodie *et al*. (2005). This event sampling also included comparisons between samples collected and/or analysed by different agencies (AIMS, DNRMW, James Cook University (JCU) - Australian Centre for Tropical Freshwater Research (ACTFR)) (Bainbridge *et al*. in preparation).

The Barron, North and South Johnstone rivers were sampled during a flood event in March 2005, spanning a period from 12 to 14 March 2005 (12 and 13 March 2005 for the Barron River).

2005-06 Wet Season

Over 2005-06, community groups and interested agencies were sought out and engaged to carry out collection and initial preparation of water samples for the River Mouth Monitoring task under the Reef Plan MMP and to assist other regional water quality monitoring efforts.

An important and time-consuming component of this engagement was development of user-friendly sampling manuals (AIMS, 2005), including 1-page Quick Reference Guides (see Appendix 1), and the provision of hands-on training in sample collection and initial sample preparation (filtration before freezer storage). Training sessions included presentations of Reef Plan and the Reef Plan MMP overviews by GBRMPA and AIMS personnel.

Interested community groups were found for eight of the ten high priority rivers (see Table 2.1). We were unable to find suitable community volunteers or other agency personnel for regular sampling of the Normanby River and regular monthly sampling in this river was not achieved. Initial sampling of the Burdekin River was carried out by AIMS personnel, due to the lack of a suitable community volunteers in the Ayr-Home Hill region. Samples were stored for collection by AIMS personnel; toward the end of the 2005-06 wet season, a commercial arrangement was developed with a courier service to collect and deliver water samples to AIMS on a near-daily basis during flood events.

Regular monthly sampling was initiated in all but the Normanby River during late 2005. Several minor flood events were sampled in the Barron, North Johnstone, Tully, Herbert, Burdekin and O'Connell Rivers between January and April 2006. Flood event sampling was carried out by DNRMW personnel in six of the prescribed priority rivers (Table 2.1) and in three additional rivers (Russell, Mulgrave and South Johnstone Rivers. In particular, one dedicated DNRMW officer carried out extensive sampling in wet tropics rivers following Tropical Cyclone Larry (TC Larry) which crossed the coast near Innisfail on 20 March 2006. TC Larry caused extensive damage in the central Wet Tropics, including loss of electrical power for periods of several days to several weeks. As a result, a significant number of previously collected samples were lost or compromised due to freezer meltdowns. Community samplers in the cyclone-affected region were unable to collect and process samples for periods of several weeks due to loss of power or the need to deal with personal storm damage.

A large number of water samples were collected from each of the ten priority catchments over the 2005-06 year, covering some dry season flow and several flow events (Figures 2.5 and 2.6). Significant wet season flows were recorded in the Normanby, Barron, Johnstone, Tully, Herbert and O'Connell Rivers. In contrast, only a limited flow was recorded in the Burdekin River, and little or no discharge occurred from the Pioneer, Fitzroy and Burnett Rivers.

Manual sampling in the Normanby River was restricted to a short period in March 2006 when a DNRMW hydrographic team was in residence in Lakefield National Park to establish a rating curve for the new gauging station at the Kalpowar crossing site. A large flow event was sampled during this period.

In the other rivers of the Wet Tropics (Barron, North Johnstone, Tully, Herbert), manual sampling was conducted during both high and low flow conditions. One or more samples were obtained from most significant wet season flow events in these rivers.

Sample collection, preparation and analyses

Discrete surface water samples were collected with acid-washed buckets or flow-integrating P61 SS samplers. During flood events one sample per day were taken at selected sampling sites. Sub-samples were analysed for dissolved inorganic nutrients (NH_4, NO_2, NO_3, PO_4, silicate ($Si(OH)_4$)), Dissolved Organic Nitrogen and Phosphorus (DON, DOP), Particulate Nitrogen and Phosphorus (PN, PP), chlorophyll a (non-flood samples only) and SS. The sub-

samples for dissolved nutrients were filtered through a 0.45µm filter cartridge (Sartorius Mini Sart N) into acid-washed screw-cap plastic test tubes and immediately frozen in a freezer (-18ºC) for later analyses. The sub-samples for particulate nutrients were collected on pre-combusted glass fibre filters (Whatman GF/F). Sub-samples for SS were collected on pre-weighed 0.4µm polycarbonate filters. Filters were wrapped in pre-combusted aluminium foil envelopes and stored at -18ºC until analyses.

Inorganic dissolved nutrients (NH_4, NO_2, NO_3, PO_4, $Si(OH)_4$) concentrations were determined by standard wet chemical methods (Treguer and LeCorre, 1975) implemented on a segmented flow analyser (Bran and Luebbe, 1997).

Water samples for total dissolved nutrients, Total Dissolved Nitrogen (TDN) and Total Dissolved Phosphorus (TDP), were UV-irradiated for 12 hours to oxidise the organic matter (Armstrong *et al*., 1966; Walsh, 1989) before analyses as above. DON and DOP were calculated by subtracting the separately measured inorganic nutrient concentrations (above) from the TDN and TDP values. From 2006, analyses of TDN and TDP were carried out after persulphate digestion (Valderrama, 1981) instead of UV-oxidation. Persulphate digestion is likely to be more reliable for the higher nutrient concentrations expected from both river and coastal waters, and will facilitate comparison with water analyses results obtained under other monitoring programs, especially in rivers. This decision was based on the results of an inter-laboratory comparison of the methods undertaken by AIMS in January 2006.

Particulate nitrogen (PN) was determined by high-temperature combustion of filtered particulate matter on glass fibre filters using an ANTEK 707/720 Nitrogen Analyser (Furnas *et al*., 1995). The analyser was calibrated using AR Grade EDTA for the standard curve and marine sediment BCSS-1 as a control standard.

PP is determined spectrophotometrically as inorganic P (PO_4: Parsons *et al*., 1984) after digesting the particulate matter in 5% potassium persulphate (Furnas *et al*., 1995). The method was standardised using orthophosphoric acid and dissolved sugar phosphates as the primary standards.

SS concentrations were determined gravimetrically from the difference in weight between loaded and unloaded 0.4µm polycarbonate filters (47mm diameter) after the filters have been dried overnight at 60^{o}C.

Chlorophyll *a* and phaeophytin concentrations were measured fluorometrically using a Turner Designs 10AU fluorometer after grinding the filters in 90% acetone (Parsons *et al*., 1984). The fluorometer was calibrated against chlorophyll *a* extracts from log-phase diatom cultures (chlorophyll *a* and *c*). The extract chlorophyll concentrations were determined spectrophotometrically using the wavelengths and equation specified by Jeffrey and Humphrey, (1975).

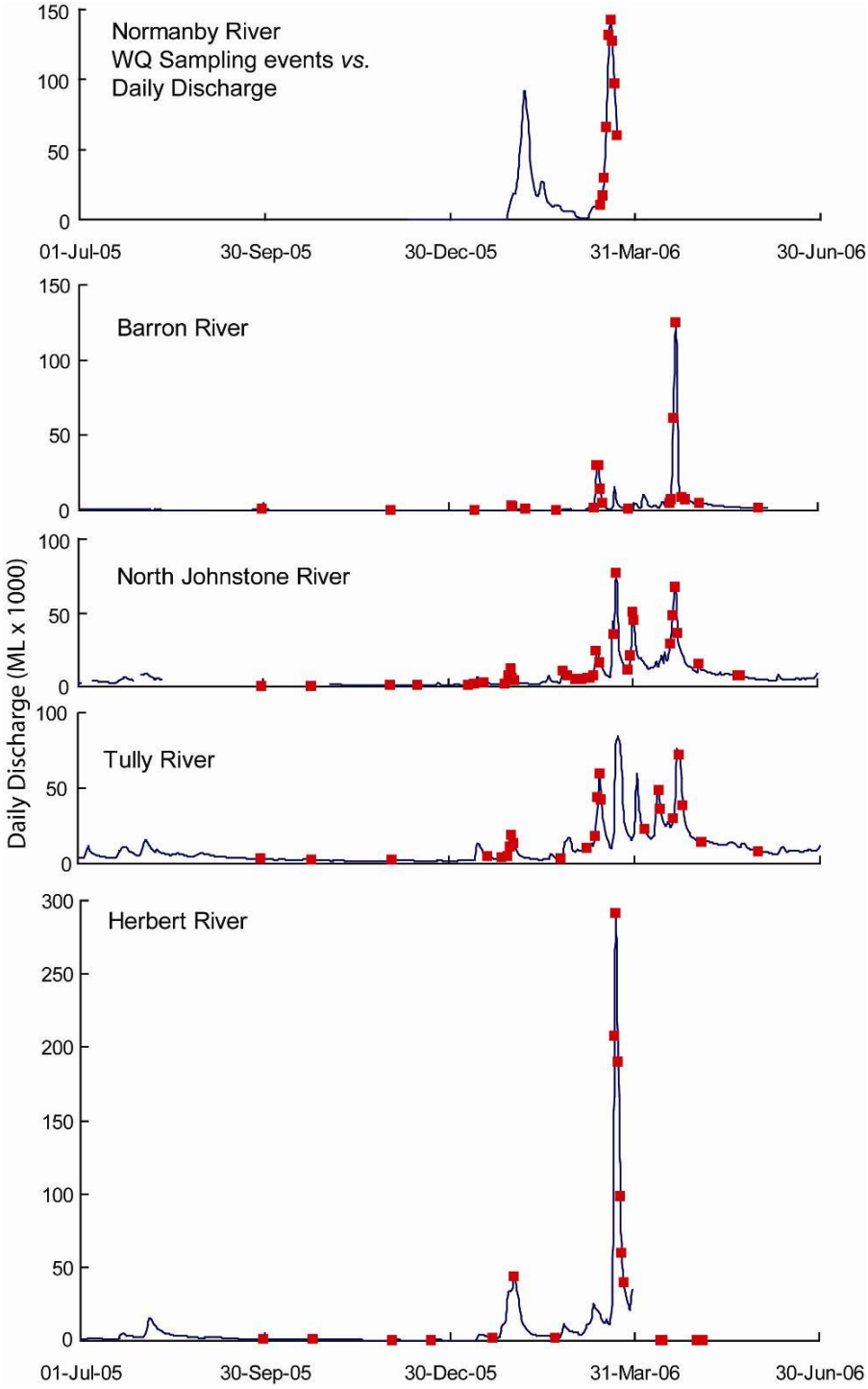


Figure 2.5. Manual sampling times in relation to discharge in the Normanby, Barron, North Johnstone, Tully and Herbert Rivers over 2005-06.

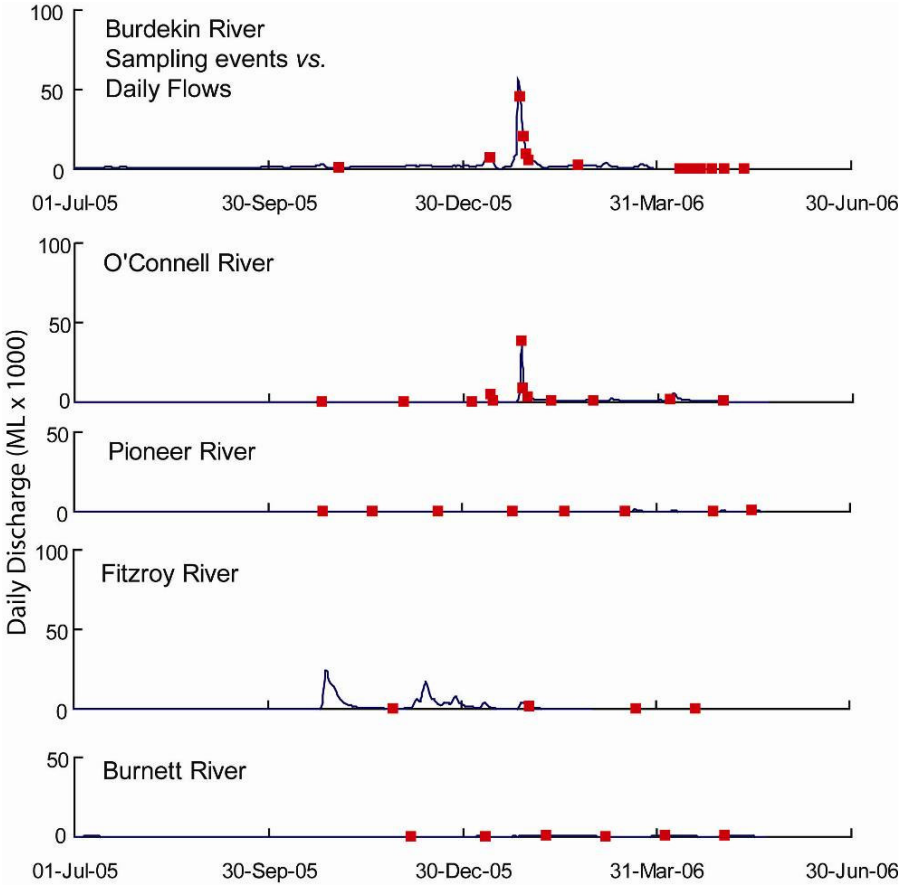


Figure 2.6. Manual sampling times in relation to discharge in the Burdekin, O'Connell, Pioneer, Fitzroy and Burnett Rivers over 2005-06.

Statistical analyses

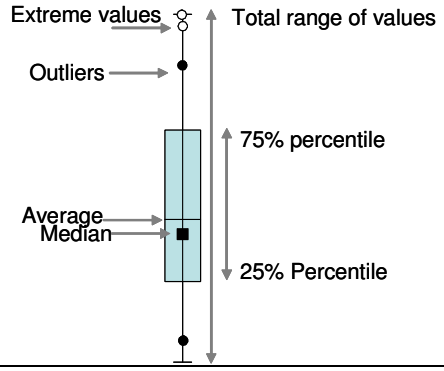
Seasonal summary statistics of concentrations of water quality parameters in river water are presented as medians, means, quartiles and maximum and minimum values.

Estimations of loads of water quality parameters in river water were carried out for all rivers, except for the Normanby, Fitzroy and Burnett Rivers. The Normanby River had only limited records of concentrations (only one flood peak was sampled in March 2006) and flow (the gauging stations was installed in December 2006, see Table 2.4). The Fitzroy and Burnett Rivers had very little discharge to the coast in 2005/06; in addition the Fitzroy River was poorly sampled for nutrient and suspended solids concentrations.

Estimations of loads are often based on simple flow-weighted average concentrations. It has been shown that concentrations of dissolved and particulates are flow dependent (e.g. De'ath, 2005). It follows that if sampled flow rates are not representative of the annual flow rates then load estimates could be substantially biased. The estimated loads presented here are based on the observed relationships between concentrations and flow. The relationships were derived using generalised linear models for each river and each water quality parameter. Concentrations were estimated for each daily mean flow rate of the whole year, and loads were then estimated as the sum of predicted concentrations multiplied by the observed daily

mean flow rate. Confidence intervals (90%) were derived by bootstrapping the whole procedure.

Summary statistics of water quality parameters for each river are presented as box and whisker plots (see box below for definitions).



Template for box plots in this chapter:

- The box contains 50% of the values= interquartile range (IQR)
- Outliers are defined as being >1.5 x IQR
- Extreme values are defined as being >3 x IQR

Results

Freshwater discharge

Monthly and annual freshwater discharges were estimated from the 10 Reef Plan MMP priority rivers and a number of additional catchments from which water samples were taken over the course of the year (Table 2.4). Records of daily discharge and 30-minute instantaneous flow were also provided by DNRMW for interpretation and integration of Mudlogger and manually sampled data sets. These higher frequency data records have been incorporated into the Reef Plan MMP end-of-river database.

Overall, both 2004-05 and 2005-06 were relatively low discharge years compared to the long-term average discharges for the priority rivers. Aggregate discharge for 9 of the 10 (excluding the ungauged Normanby) priority rivers in 2004-05 (11 km^3) was only 36 percent of the long-term average for these rivers. In 2005-06, the aggregate discharge (Normanby included) was 19.8 km^3 (52 percent of the long-term average = 38 km^3). Within both years, a few individual catchments within the Wet Tropics had runoff volumes close to their long-term average, but in the southern catchments (Burdekin to Burnett), discharge levels were significantly below average.

Much of the rainfall and runoff during the 2005-06 wet season came late in the season following TC Larry.

Table 2.4. Monthly freshwater discharge volume (ML) from target GBR Catchment rivers in 2004/05 and 2005/06. Bold print and values in red indicate rivers where AIMS Mudloggers were deployed for monitoring of suspended sediment concentrations. nd = no data available. 1969-94: Long-term discharge average as a comparison (in blue print). Data supplied by the Queensland Department of Natural Resources & Mines. Estimated error for flow data is generally <5% (G. Pocock, DNRMW; pers. comm.)

04/05	Normanby	Barron	Russell	N-Johnstone	S- Johnstone	Tully	Herbert	Burdekin	O'Connell	Pioneer	Fitzroy	Burnett	Total
July	nd	16,613	46,274	91,177	35,631	130,506	56,510	31,109	0	606	0	171	408,597
Aug	nd	14,063	24,365	53,488	23,709	93,32	36,874	27,763	0	598	0	599	265,291
Sep	nd	29,267	14,504	35,319	18,278	67,394	24,709	28,722	0	1,061	0	612	219,866
Oct	nd	16,879	10,435	25,008	11,630	60,562	14,608	45,635	0	1,122	0	624	186,503
Nov	nd	15,662	13,835	24,058	21,122	57,219	13,155	50,199	0	1,702	37,056	46,146	280,154
Dec	nd	14,901	25,118	52,925	29,603	7,545	38,005	66,045	821	927	252,192	66,225	644,307
Jan	nd	97,899	142,036	224,957	68,228	309,738	461,353	3,151,647	68,026	165,627	347,918	9,308	5,046,737
Feb	nd	33,820	54,441	123,007	42,665	145,567	238,974	712,284	6,584	2,760	273,748	1,319	1,635,169
Mar	nd	87,624	200,749	230,104	66,139	255,693	129,062	63,271	99	2,782	8,481	799	1,044,803
April	nd	14,708	163,551	216,632	61,262	322,859	173,840	62,568	106	13,013	310	1,018	1,029,867
May	nd	25,874	111,841	148,424	36,134	184,672	101,048	56,151	109	1,709	0	1,196	667,158
Jun	nd	24,913	64,048	99,579	19,254	151,404	53,742	31,054	29	1,433	0	816	446,272
Total	nd	392,223	871,197	1,324,678	433,655	1,876,991	1,341,880	4,326,448	75,774	193,340	919,705	128,833	11,874,724
1969-94	4,950,000	810,000				3,290,000	4,010,000	10,290,000	1,540,000	1,190,000	6,080,000	1,150,000	

Table 2.4 continued

05/06	Normanby	Barron	Russell	N-Johnstone	S- Johnstone	Tully	Herbert	Burdekin	O'Connell	Pioneer	Fitzroy	Burnett	Total
July	nd	28,190	107,162	124,018	18,425	194,426	59,201	24,589	54	2,039	0	8,013	566,117
Aug	nd	11,326	99,615	157,828	24,726	227,620	154,820	20,880	75	241	0	998	698,129
Sep	nd	10,168	31,532	89,920	12,022	127,474	43,008	28,102	50	2,762	0	536	345,574
Oct	nd	6,524	12,103	46,848	23,512	79,391	19,883	45,509	1,464	2,422	97,695	764	338,057
Nov	nd	5,491	9,278	31,201	14,479	53,399	12,853	42,167	47	1,128	53,196	677	223,916
Dec	New Site	6,191	7,845	26,819	12,372	61,238	9,509	60,669	48	1,290	144,664	1,945	332,590
Jan	83,325	14,669	42,585	82,519	39,582	167,894	238,021	229,912	62,308	2,466	38,516	7,177	1,008,974
Feb	711,847	9,977	60,534	106,184	41,095	156,150	226,860	63,949	11,382	1,021	4,973	22,043	1,416,014
Mar	1,010,382	133,778	355,009	540,823	257,175	842,228	1,409,617	54,000	9,283	5,920	44,888	4,271	4,667,374
April	1,412,390	357,599	218,207	625,236	221,364	931,998	1,864,885	1,353,367	15,043	10,553	267,629	12,648	7,290,920
May	172,520	84,784	105,496	248,807	98,276	375,065	574,999	186,258	1,905	10,885	4,231	11,226	1,874,452
Jun	28,502	34,968	119,593	149,607	64,986	256,185	280,549	69,627	1,137	15,379	22,374	4,321	1,047,228
Total	3,418,966	703,665	1,168,959	2,229,810	828,014	3,473,068	4,894,205	2,179,029	102,796	56,106	678,166	74,619	19,809,345
1969-94	4,950,000	810,000				3,290,000	4,010,000	10,290,000	1,540,000	1,190,000	6,080,000	1,150,000	

Fine suspended sediment exports (Turbidity Loggers)

Records of turbidity for estimating fine suspended sediment concentrations were obtained from nine of the ten priority rivers (Figures 2.7 to 2.10). A full data record was also obtained from the logger in the Burnett River, but it contained relatively significant short-term variability associated with temperature and water level fluctuations in the weir pool at Bundaberg where it was deployed. As there was very little discharge from the Burnett River over the 2005-06 wet season (*ca*. 0.06 km^3), it was decided not to fully process the turbidity record (Fig 2.10).

There were three significant flow events in the Normanby River over the period for which mudlogger records at the Kalpowar station are available (Figure 2.7 Top). The Kalpowar gauging station was established in December 2005. Estimates of flow over the 2005-06 wet season were made from the pressure (depth) data recorded by the Mudlogger (deployed November 2005) and the rating curve determined by DNRMW personnel. There was a strong, but non-linear correlation between Mudlogger pressure and measured discharge at the Kalpowar site. The largest discharge event occurred during late April 2006. The maximum suspended sediment concentration (ca. 400 mg L^{-1}) occurred during the first significant flow event of the summer. Thereafter, peak sediment concentrations were on the order of 150 to 300 mg L^{-1}. Over the 2005-06 wet season, the Normanby River had a volume-weighted average export of 67,000 tonnes per km^3 of discharge. This may be compared to a previously determined export of 100,000 tonnes per km^3 (Furnas, 2003). The 2004-05 wet season was particularly dry (Tables 2.4 and 2.5); however a direct comparison between the two years is difficult as there was no gauging station in the lower reaches of the Normanby River.

Two significant flow events occurred in the Barron River (Figure 2.7) over the 2005-06 wet season; the first in mid-March approximately one week before TC Larry, and a second much larger peak in late-April. TC Larry only caused a relatively small flow increase in the Barron River. The highest suspended sediment concentration (ca. 800 mg L^{-1}) was recorded during the mid-March flow event. The peak suspended sediment concentration associated with TC Larry was 450 mg L^{-1}, with a peak sediment concentration of 400 mg L^{-1} in the large late-April flood event. Estimates of freshwater discharge and fine sediment export are given in Table 2.5. Over the 2005-06 wet season, the Barron River had a volume-weighted average export of 158,000 tonnes per km^3 of discharge. This compares to a volume weighted export of 173,000 tonnes per km^3 during the 2004-05 wet season and is the highest export for all of the 'wet' rivers. Precise reasons for the high export were not resolved, but may be due to the large area of cultivated land on the nearby Atherton Tablelands.

The North Johnstone River was characterised by a number of large and small flow events over the 2005-06 wet season (Figure 2.7). The highest discharge and suspended sediment concentrations were associated with TC Larry in late-March. Rainfall continued in the catchment for the next two months and a number of significant flow events were recorded. The maximum suspended sediment concentration recorded in the North Johnstone River was 250 mg L^{-1}. In most cases, peak sediment concentrations were on the order of 75-150 mg L^{-1}. Estimates of freshwater discharge and fine sediment export are given in Table 2.5. Over the 2005-06 wet season, the North Johnstone River had a volume-weighted average export of 37,000 tonnes per km^3 of discharge. This compares to a volume weighted export of 63,000 tonnes per km^3 during the 2004-05 wet season. The lower export during the 2005-06 is most likely because a significant proportion of the seasonal rainfall fell in moderate size events over two months at the end of the wet season rather than in a massive flood event.

Five significant flow events occurred in the Tully River over the 2005-06 wet season (Figure 2.7). The largest flow was associated with TC Larry in late March, though significant rains continued to fall well into May. The highest recorded sediment concentration (420 mg L^{-1}), however, was measured during the small initial flow event of the wet season in mid-January. The Tully River had a volume-weighted average suspended sediment concentration for the wet season of 22,000 tonnes per km^3, which was approximately half of the volume-specific load (50,000 tonnes per km^3) recorded over the 2004-05 wet season (Table 2.5).

The 2005-06 wet season in the Herbert River catchment was characterised by three significant flow events, the first and largest of which was associated with TC Larry (Figure 2.7). The cyclone passed over the upper Herbert catchment as it moved inland. Maximum discharge rates exceeded 4,000 cumecs during this flood. The highest suspended sediment concentrations, approaching 800 and 500 mg L^{-1}, however, were measured during a number of small early season flow events. The peak suspended sediment concentration during the flood following TC Larry was 300 mg L^{-1}. Because of the large volume of water discharged, this event accounted for about half of the total wet season sediment export. Over the 2005-06 wet season, the Herbert River had a volume-weighted sediment export of 73,000 tonnes per km^3 (Table 2.5).

Only two flow events of note occurred in the Burdekin River over the 2005-06 wet season (Figure 2.8 Bottom); the first in late-January and the second, larger flow in mid-April. The second event was associated with the rains produced inland by the remains of TC Larry. Overall, it was a relatively low-flow year for the Burdekin River, with a total gauged discharge of 1.96 Km^3 for the first six months of 2006 (Table 2.5). The maximum recorded sediment concentration was slightly in excess of 1,000 mg L^{-1} in late January. During the late season flood event, the highest sediment concentration was 750 mg L^{-1}. Over the 2005-06 wet season, the volume-weighted sediment load was 316,000 tonnes per km^3. This value was considerably less than the 1,500,000 tonnes per km^3 calculated for the period when the logger was operational during the summer of 2004-05. Previously estimated sediment export for the Burdekin River was close to 360,000 tonnes per km^3 (Furnas, 2003).

The O'Connell River is a relatively small river in the Proserpine region draining low mountains and a coastal floodplain cleared for grazing and sugar cultivation. A logger was installed for the first time for the 2005-06 wet season in the O'Connell just below the confluence of the O'Connell River and its major tributary, Andromache Creek. The 2005-06 wet season was characterised by a single significant flow event in late-January (Figure 2.9 Top), though there were pronounced peaks in sediment concentration during seven other small flow events over the course of the wet season. Because of the small size of the O'Connell catchment and low rainfall over the 2005-06 wet season, relatively little sediment was exported. The catchment, however, had a moderately high volume-weighted export of 117,000 tonnes per km^3.

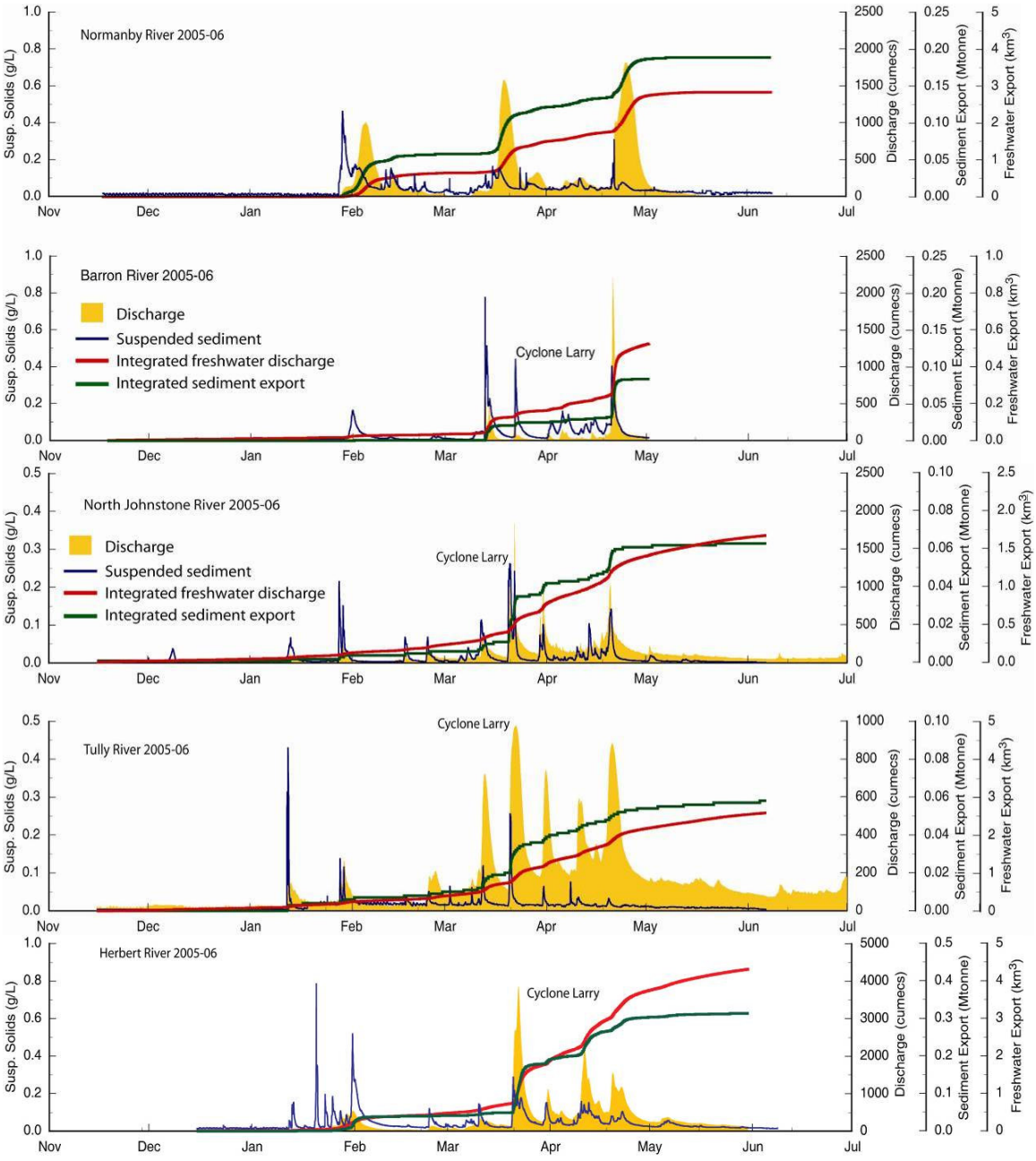


Figure 2.7. Time series of fine suspended solids concentrations in the Normanby, Barron North Johnstone, Tully and Herbert Rivers measured by AIMS Mudloggers over the 2005-06 wet season in relation to concurrent river flow and integrations of cumulative freshwater discharge and fine sediment export. Flow data was supplied by DNRMW. Note different axis scales.

A Mudlogger was installed for the first time in the Pioneer River during the 2005-06 wet season on the upstream side of the Dumbleton Weir (Mackay). The 2005-06 wet season was characterised by very low rainfall into and discharge from the Pioneer catchment (Figure 2.9 Bottom). Measured suspended sediment concentrations never exceeded 50 mg L^{-1} over the course of the deployment. As a result, very little suspended sediment was exported from the catchment during the 2005-06 wet season. A volume-weighted export of 10,000 tonnes per km^3 (Table 2.5) was calculated. Because of the extreme low discharge, this value should be regarded as very conservative.

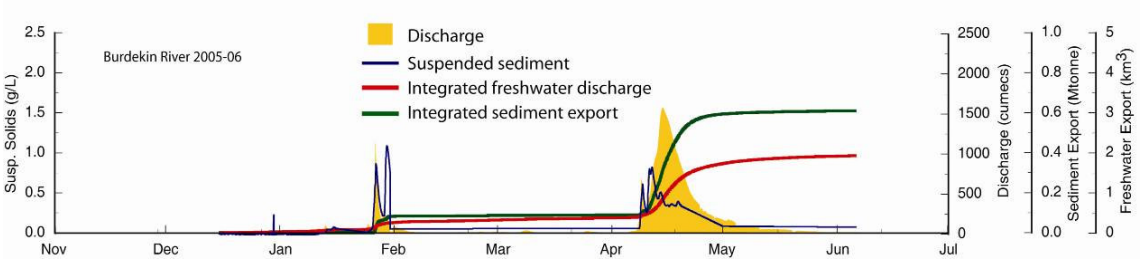


Figure 2.8. Time series of fine suspended solids concentrations in the Burdekin River measured by AIMS Mudloggers over the 2005-06 wet season in relation to concurrent river flow and integrations of cumulative freshwater discharge and fine sediment export. Flow data was supplied by DNRMW.

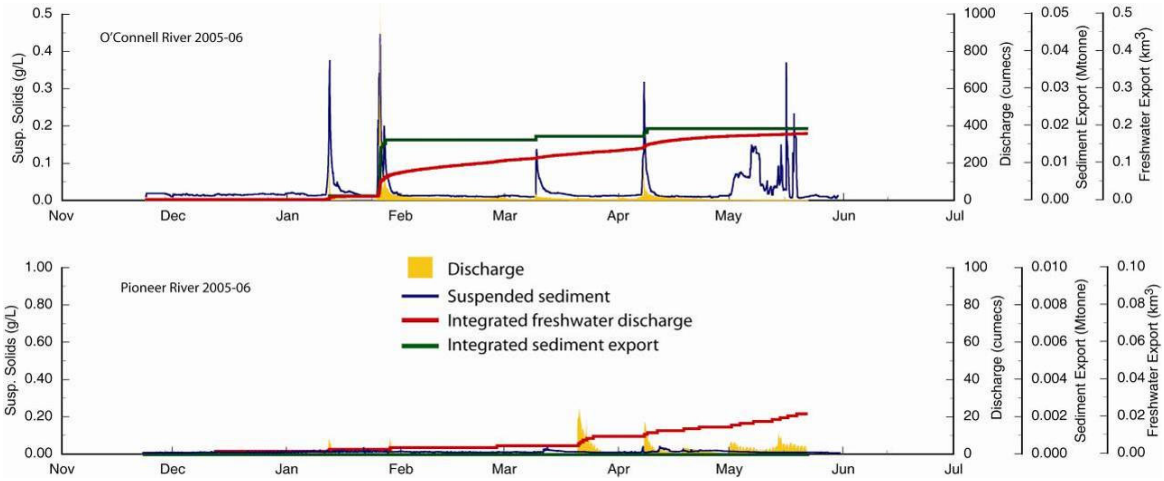


Figure 2.9. Time series of fine suspended solids concentrations in the O'Connell and Pioneer Rivers measured by AIMS Mudloggers over the 2005-06 wet season in relation to concurrent river flow and integrations of cumulative freshwater discharge and fine sediment export. Flow data was supplied by DNRMW.

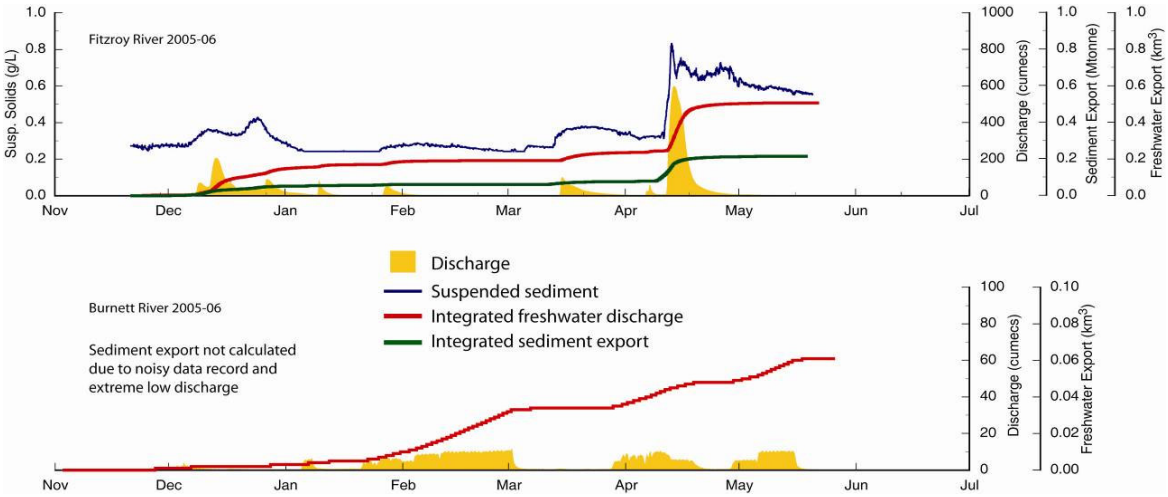


Figure 2.10. Top: Time series of fine suspended solids concentrations in the Fitzroy River measured by an AIMS Mudlogger over the 2005-06 wet season in relation to concurrent river flow and integrations of cumulative freshwater discharge and fine sediment export. Flow data was supplied by DNRMW. Bottom: Time series of freshwater flow in the lower Burnett River over the 2005-06. Mudlogger data from this deployment was not processed due to the level of instrumental signal variability and the very low flow. Flow data was supplied by DNRMW.

Table 2.5 Turbidity logger deployments in the ten Reef Plan priority rivers, estimated freshwater discharges during deployments and estimated loads of fine suspended sediment. Data for the 2004-05 wet season are shown in italics. Flow data from DNRMW.

River	Data record start	Data record end*	Freshwater discharge Km^{3}**	Sediment export 10^{6} tonnes	Discharge-weighted sediment export 10^{3} tonnes km^{-3}	Discharge-weighted sediment export tonnes km^{-2} catchment area
Normanby	16-Nov-05	7-Jun-06	2.84	0.189	67	3
Barron	*02-Dec-04*	*13-Jul-05*	*0.26*	*0.045*	*173*	*81*
	18-Nov-05	2-May-06	0.53	0.084	158	74
N Johnstone	*02-Dec-04*	*12-Jul-05*	*0.97*	*0.061*	*63*	*68*
	15-Nov-05	6-Jun-06	1.67	0.062	37	40
Tully	*02-Dec-04*	*12-Jul-05*	*1.25*	*0.063*	*50*	*30*
	15-Nov-05	6-Jun-06	2.59	0.058	22	13
Herbert	15-Dec-05	31-May-06	4.31	0.314	73	7
Burdekin	*30-Nov-04*	*8-Jul-05*	*4.09*	*6.220*	*1521*	*12*
	15-Dec-05	6-Jun-06	1.93	0.609	316	2
O'Connell	23-Nov-05	22-May-06	0.18	0.021	117	49
Pioneer	22-Nov-05	22-May-06	0.02	0.0002	10	6
Fitzroy	*22-Dec-04*	*14-Jul-05*	*0.66*	*0.740*	*1121*	*8*
	23-Nov-05	22-May-06	0.51	0.215	425	3
Burnett	2-Nov-05	26-May-06	0.06	***	***	***

* The end of a useful record depends on the logger recovery date, the duration of the 30-minute river flow record supplied by DNRMW or the loss of data due to fouling of the logger optics

** 1 Km^3 = 10^9 m^3 = 10^6 megalitres

*** No estimate calculated due to the very low discharge and high site noise in the logger data.

Nutrient and suspended sediment exports (grab samples)

The engagement of community volunteers to support the water sampling activities under this monitoring task was more difficult and time consuming than anticipated. Community sampling started only in late 2005. Dry season samples before this were collected, where possible, by AIMS staff. The collaboration between AIMS and GBRMPA was essential in the engagement of interested and suitable community samplers. In the Wet Tropics region sampling was undertaken by FNQ NRM staff and associated community volunteers. This arrangement proved extremely useful as it was tightly linked with other activities of the NRM Group in this catchment. However, we were unable to find community volunteers in two catchments, the Normanby and the Burdekin (see Table 2.6 for further details about community groups involved). While the sampling went smoothly once groups were set up and trained, a general comment from the volunteers was that the requirements for sampling during flood events (daily sampling) were too demanding for volunteer samplers. Community sampling activities were also hampered by the impact of cyclone Larry in the Wet Tropics catchments during March /Aril 2006.

Table 2.6 Summary of river mouth sampling activities in 2005/06. Shading indicates sites with community sampling undertaken.

NRM Region	River	Site name	Sampler affiliation	Monthly ambient sampling start	Flood sampling period (community)	Flood sampling period (NRMW)
Cape York	Normanby	Kalpowar	NRMW	n/a	n/a	14-22/03/06
Wet Tropics	Barron	Kamerunga/ Stratford Jetty*	FNQNRM/ volunteer	29/09/05	12-15/03/06 18-21/04/06	n/a
		Myola	NRMW	n/a	n/a	30-31/01/06 12-16/03/06 21&24/04/06
	N-Johnstone	Old Highway Bridge	FNQNRM	29/09/05	27/01-01/02/06 25/02-15/03/06	12-15/03/06 22/03-01/04/06
	Tully	Euramo	FNQNRM	29/09/05	30-31/01/06 12-24/04/06	28-31/01/06 08-16/03/06
	Herbert	Gairloch	FNQNRM/ volunteer	29/09/05	21-23/03/06	21-26/03/06
Burdekin	Burdekin	Home Hill Bridge	AIMS	12/01/06 22/02/06	11-26/04/06	26-30/01/06 11-21/04/06
Mackay Whitsunday	Pioneer	Dumbleton	MWHW	18/11/05	No flood event	n/a
	O'Connell	Vickers Farm Junction/ Bruce Highway Cr.*	MWHW	25/10/05	12-13/01/06 27-29/01/06	n/a
Fitzroy	Fitzroy	Fitzroy Water intake tower	Fitzroy River Water	28/11/05	No flood event	n/a
Burnett Mary	Burnett	Burnett Water treatment plant	Bundaberg City Council	02/11/05	No flood event	n/a

*Different site used for wet season sampling due to safe access issues and/or reach of saltwater in dry season. Note that for historical reasons NRMW flood sampling of the Barron River is undertaken at another site than the community monitoring.

Normanby River

The gauging station at Kalpowar was established by DNRMW personnel in December 2005. A hydrographic party was at the station to carry out surveys for establishing a rating curve

from mid- to late-March 2006. During that period a significant flood event occurred, allowing flow and height measurements to be made across almost the full height range available at that site. During their work, the DNRMW team collected seventeen (17) water samples for nutrient and sediment analysis. The results of the analyses are shown in Figure 2.11 and summarised in Appendix Table A1-2.1. A brief report on the establishment of the gauging station is presented in the Appendix.

SS concentrations exhibited a well defined peak during the mid-March flood event, with the highest concentrations on the leading edge of the hydrograph. DIN and DIP concentrations were low and fairly constant throughout the mid-March flood event, regardless of flow. In contrast, concentrations of both PN and PP varied directly with river flow and suspended sediment concentrations. DON and DOP concentrations were higher and more variable than inorganic N and P during the flood. Most of the dissolved N and P were in the dissolved organic pool. In contrast to N and P, silicate concentrations decreased sharply during the beginning of the flood event, then rose again, indicating dilution during the high flow period from low-Si rainwater and freshwater runoff. No chlorophyll or phaeophytin samples were collected.

Barron River

Water samples were collected from three sites in the Barron River over the course of the 2005-06 year: at the DNRMW Myola gauging station above the Barron Gorge, at the Kamerunga Bridge, immediately below the gorge, and at Stratford, several kilometres further downstream on the floodplain. These sites are relatively close together and concentrations are unlikely to change greatly between them due to additional inputs or in-river processes.

Intensive sampling was carried out during four flow events, including the flood associated with TC Larry and later in mid-April (Figure 2.12 and Appendix Table A1-2.2). Overall, sampling was undertaken on 21 occasions. Nutrient and suspended sediment data were obtained on 18 to 21 occasions, while chlorophyll and phaeophytin were measured on five (5) occasions. Only one sampling was conducted during the dry season, but a number of samples were collected under low-flow conditions during the summer. The intervals between sample collections during low-flow periods were sufficiently separated in time so that all of the samples can be used to derive useful estimates of in-stream concentrations and export.

SS concentrations exhibited sharp peaks during the flood event following TC Larry, and a larger event in late April, 2006, with a peak concentration of 1.16 g litre^{-1}. High dissolved inorganic phosphorus concentrations were recorded on two occasions associated with a small early wet season flood event in late January and the TC Larry flood. Apart from these two samples, measured concentrations of both inorganic nitrogen and phosphorus were low and relatively stable through the year. DON comprised nearly half of the estimated nitrogen exported from the Barron River over 2005-06. Particulate phosphorus was the primary form of P exported by the Barron. Silicate concentrations dropped sharply during the two major flood events of the summer, indicating significant dilution by low-Si rainwater and fresh runoff. Chlorophyll concentrations ranged from ca. 1 to 2.5 µg L^{-1} under non-flood conditions.

North Johnstone River

Thirty (30) sampling events were carried out in the North Johnstone River over 2005-06 by community volunteers and DNRMW personnel. Suspended sediment and particulate nutrient samples were obtained on 25-30 occasions and dissolved nutrients were collected on 15

occasions. Two sampling events were conducted during the nominal "dry" season, while a number of additional samples were collected under "wet" season low-flow conditions. The largest flow event of the year (late-March) was associated with TC Larry. Two additional flood events followed the TC Larry flood and significant flow continued up to the end of June 2006.

Suspended sediment concentrations (Figure 2.13) varied considerably, with peak concentrations occurring during both large and small flood events. A statistical summary is given in Appendix Table A1-2.3. The maximum SS concentrations (ca. 250 mg L^{-1}) was considerably lower than that recorded in the nearby Barron River. Both DIP and DIN were low and fairly stable over the course of the year. Prior to and during the early part of the wet season, DON comprised a significant proportion of the total N in the North Johnstone River. The DON concentration declined steadily over the course of the wet season. Both PN and PP exhibited peak concentrations during flood events. As before, silicate concentrations decreased during flood events due to dilution. Chlorophyll concentrations were variable, ranging from ca. 0.5 to 3 µg L^{-1}.

Tully River

Twenty-one (21) sample events were carried out in the Tully River over the 2005-06 year by community volunteers and DNRMW personnel. Suspended sediment was measured on 20 occasions and nutrient samples (dissolved and particulate) were taken on 16 occasions. Two sample events were undertaken during the nominal "dry" season. Results are presented in Figure 2.14 and summarised in Appendix Table A1-2.4.

The Tully River exhibited a seasonal flow pattern similar to that in the North Johnstone. Following a small flow event in late January, the wet season rains largely began in early March. The largest flow event (though not by much) occurred after TC Larry (20 March 2006). Unfortunately, no water samples were collected from the Post-TC Larry flood event.

With one exception, suspended sediment concentrations were < 100 mg L^{-1}. There was a short-lived peak in DIN and DON during a small flow event at the end of January and thereafter both DIN and DON concentrations were low and fairly constant over the wet season. After a small peak in late-January, PN concentrations were low throughout the summer. As in other rivers, PP concentrations peaked during flood events. Elevated DOP concentrations were measured in one sample during the end-of-January flow event. Reasons for this high value are unknown at this time. Thereafter, both DIP and DOP concentrations remained low through the wet season. During low-flow periods, silicate concentrations were relatively stable, but declined sharply due to dilution during flood events. Chlorophyll concentrations ranged between 0.25 and 1.3 µg L^{-1}.

Herbert River

Twenty-three (23) water samples were collected from the Herbert River over 2005-06. Two samples were collected during the "dry" season. Most samples were collected during the significant flood event in late March which followed TC Larry. At this time verified flow data is only available for the period to the end of March, although appreciable flows continued on through April. Results are presented in Figure 2.15 and summarised in Appendix Table A1-2.5.

With one exception (late-January 2006), SS concentrations were relatively low in the Herbert River over the summer period, with low-flow suspended loads < 25 mg L^{-1}. The peak

suspended sediment load during the large March flood event were < 200 mg L^{-1}.Apart from two samples collected at the end of April and early May, DIN concentrations were low and relatively constant throughout the summer period. Somewhat higher DIN levels were measured during the low flow period (Sept-Oct) of the dry season, but low DIN concentrations were measured at the end of the dry period (December). A very large fraction of the total N transported in the Herbert was in the form of DON (42%). Despite large variations in discharge, DON concentrations remained fairly constant through the wet season. PN concentrations peaked during the two significant flow events sampled. With the exception of one sample during the March post-Larry flood, concentrations of DIP and DOP were low and relatively constant over the wet season. As in other rivers, Si concentrations exhibited dilution during the two flood events sampled. Chlorophyll concentrations ranged from 1 to 3 µg L^{-1}.

Burdekin River

Twenty-eight (28) water samples were collected from the lower Burdekin River over the course of the 2005-06 wet season (Figure 2.16; Appendix Table A1-2.6). No dry season samples were collected due to the lack of flow. Chlorophyll concentrations were measured on four occasions.

Two significant flow events occurred during the 2005-06 wet season, in late January and mid-April (flow data not in hand at the time of writing).

Suspended sediment concentrations peaked during both summer flow events, with maximum concentrations slightly exceeding 1 g L^{-1}. Particulate nitrogen (PN) and phosphorus (PP) concentrations peaked in parallel with the suspended sediment load. DIN and DON concentrations were low during low-flow periods between the flood event, rising to ca. 250 µg N L^{-1} during the flood events, with a small number of higher concentrations. With a few exceptions, DIP and DOP were relatively low through the summer with slight elevations (to ca. 50 µg P L^{-1}) during the flood events. Silicate concentrations were relatively constant through the summer, with little or no dilution effects during the flood events. Chlorophyll concentrations during low-water conditions varied between 1.8 and 5.8 µg L^{-1}.

O'Connell River

The O'Connell is a small coastal river catchment. Twelve (12) water samples were taken over the course of 2005-06 (Figure 2.17). Statistical data are summarised in Appendix Table A1-2.7. One sampling was undertaken prior to the wet season. During much of the year, there is little flow in the O'Connell River.

The 2005-06 wet season was characterised by a small flow event in early January, a very significant flow event in late January and a third (small) event in early April.

The highest suspended sediment concentration (800 mg L^{-1}) was recorded during the first small flow event in early January. The peak sediment load during the larger late-January event was approximately half that level (400 mg L^{-1}). The highest concentrations of DIN (1,200 µg N L^{-1}) and DIP (77 µg P L^{-1}) were also recorded during the small first flush event. DON was a significant contributor (ca. 45%) to total N loads in the O'Connell River over the wet season. In contrast, DOP concentrations were relatively low during both high and low-flow conditions. Particulate N concentrations were low compared to DON through much of the wet season. PP concentrations, however, increased sharply during both early wet season flood events, regardless of magnitude. Silicate concentrations dropped significantly during the first two wet season flood events, but changed little in the final small flood event. Chlorophyll concentrations varied considerably (1 – 12 µg L^{-1}) between samples. High concentrations were recorded during early December (low-flow, pre-wet season) and shortly after the small initial flood event which likely brought additional nutrients into the river.

Pioneer River

The Pioneer River was characterised by very low flow rates over the 2005-06 wet season (Figure 2.18), with a maximum daily discharge of 1,600 ML. Summary statistics for nutrient data are given in Appendix Table A1-2.8. Five (5) water samples were collected from a site in the pool above the Dumbleton Weir near Mackay. Because of the low flow, little new sediment and nutrient material was washed into the river. The observed levels of nutrients and their lack of variation likely reflect the dominance of biological uptake, storage and recycling processes in the weir pool rather than catchment processes as a determinant of nutrient concentrations in this year.

With one exception (30 mg L^{-1}, early May), suspended sediment concentrations in the sampled waters were low (< 10 mg L^{-1}). Observed concentration ranges of all nutrient species were relatively narrow. DIN and DIP concentrations were consistently low through the course of the wet season, most likely from uptake by phytoplankton and aquatic plants and denitrification in river sediments. Most nitrogen was in a dissolved organic form (DON). DOP levels were also a significant proportion of total P. Chlorophyll concentrations varied ca. 8-fold between samples (3-23 µg L^{-1}).

Fitzroy River

Four small and one moderate-sized flow event occurred in the Fitzroy River over the course of the 2005-06 wet season (Figure 2.19). Summary statistics for nutrient data are given in Appendix Table A1-2.9. The moderate event (mid-April) followed widespread rain along the western margins of the catchment. Overall, only four samples were collected from the Fitzroy River (Water treatment plant, Rockhampton). None of the samples were collected during any of the peak periods of flow.

SS concentrations were fairly low (for the Fitzroy River) in three of the four samples collected, with the highest concentration found in the sample taken during the falling stage of the mid-April flood event. Concentrations of the major forms of nitrogen differed little between samples. Most of the N was in dissolved form (DIN and DON). Elevated concentrations of all forms of P were recorded during the small mid-March flow event. Chlorophyll concentrations varied between 0.2 and 6 µg L^{-1}. This range likely reflects both the levels of nutrients and turbidity of the waters in the weir pool above the Rockhampton barrage.

Burnett River

There was very little discharge from the Burnett River over the course of 2005-06 (Figure 2.20). Statistical summary data are presented in Appendix Table A1-2.10. Daily discharges were consistently less than 1,000 ML. Six water samples were collected from the Bundaberg weir pool in Burnett River over the 2005-06 wet season, including periods of relatively higher and lower flows.

Because of the relatively calm conditions in the river, suspended sediment concentrations were low (< 10 mg L^{-1}) and constant through the 'wet' season. Very little temporal variation was observed in either dissolved or particulate nutrient species. Concentrations of both DIN and DIP were very low, reflecting the importance of biological uptake by aquatic plants in the river and denitrification in river sediments. Significant amounts of DON were measured in the Burnett samples (ca. 400 µg N L^{-1}). No particulate N samples were taken. Despite the low energy levels in the river, most of the P was present in particulate form. Chlorophyll concentrations ranged between 15 and 38 µg L^{-1}, the highest of any river, most likely reflecting levels of nutrient loading relative to the slow flow of water in this impoundment.

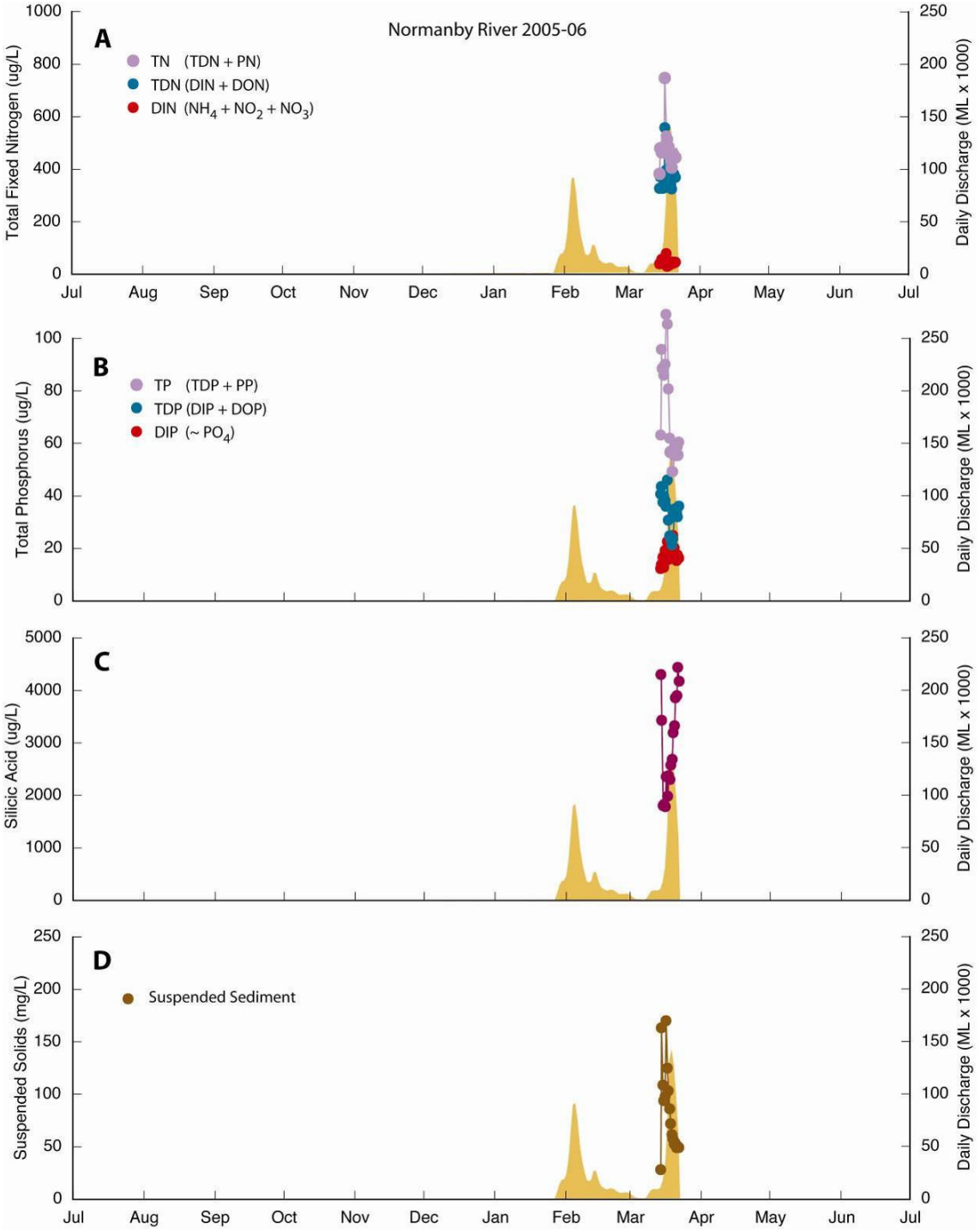


*Figure 2.11. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids in water samples manually collected from the **Normanby River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyse.*

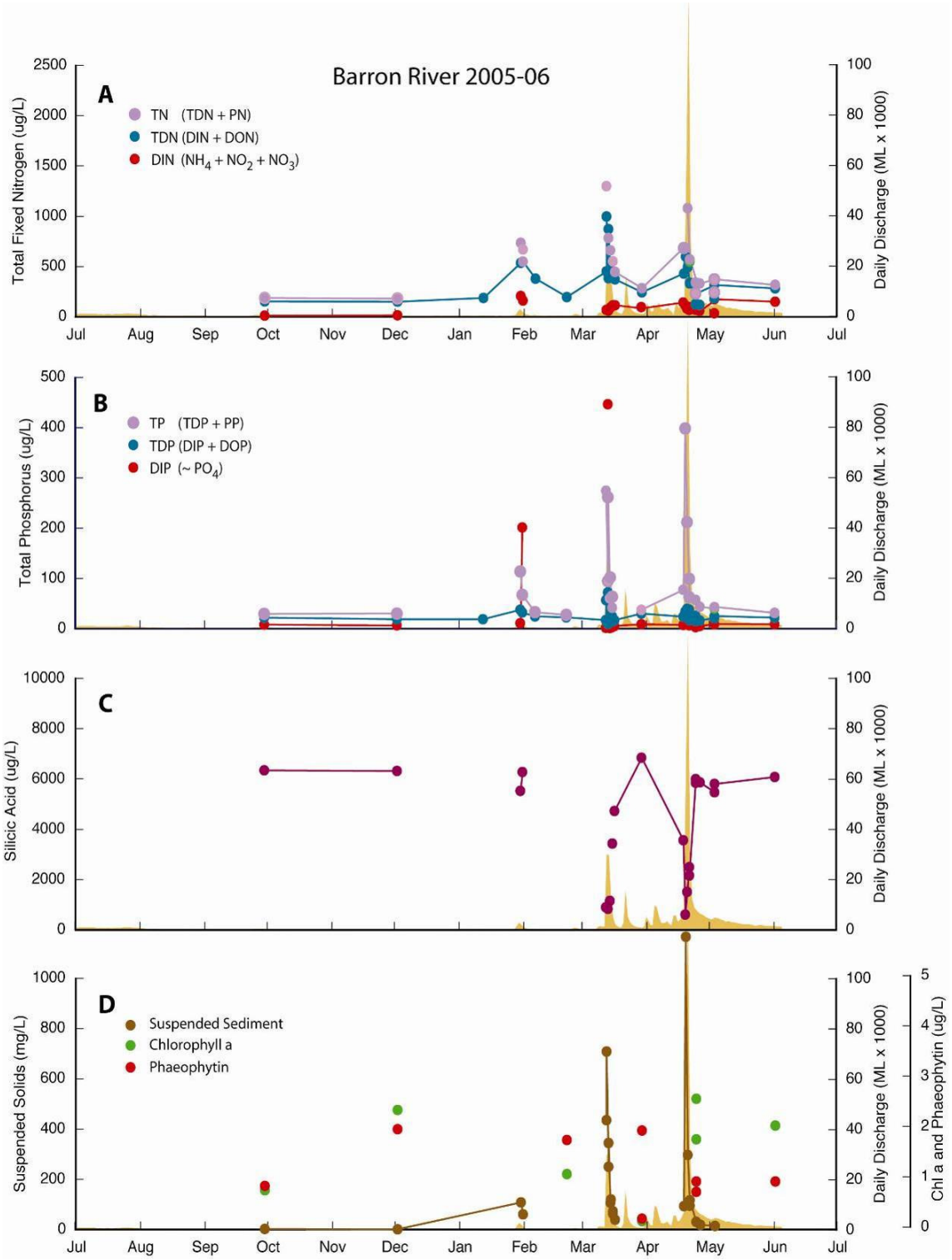


*Figure 2.12. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Barron River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

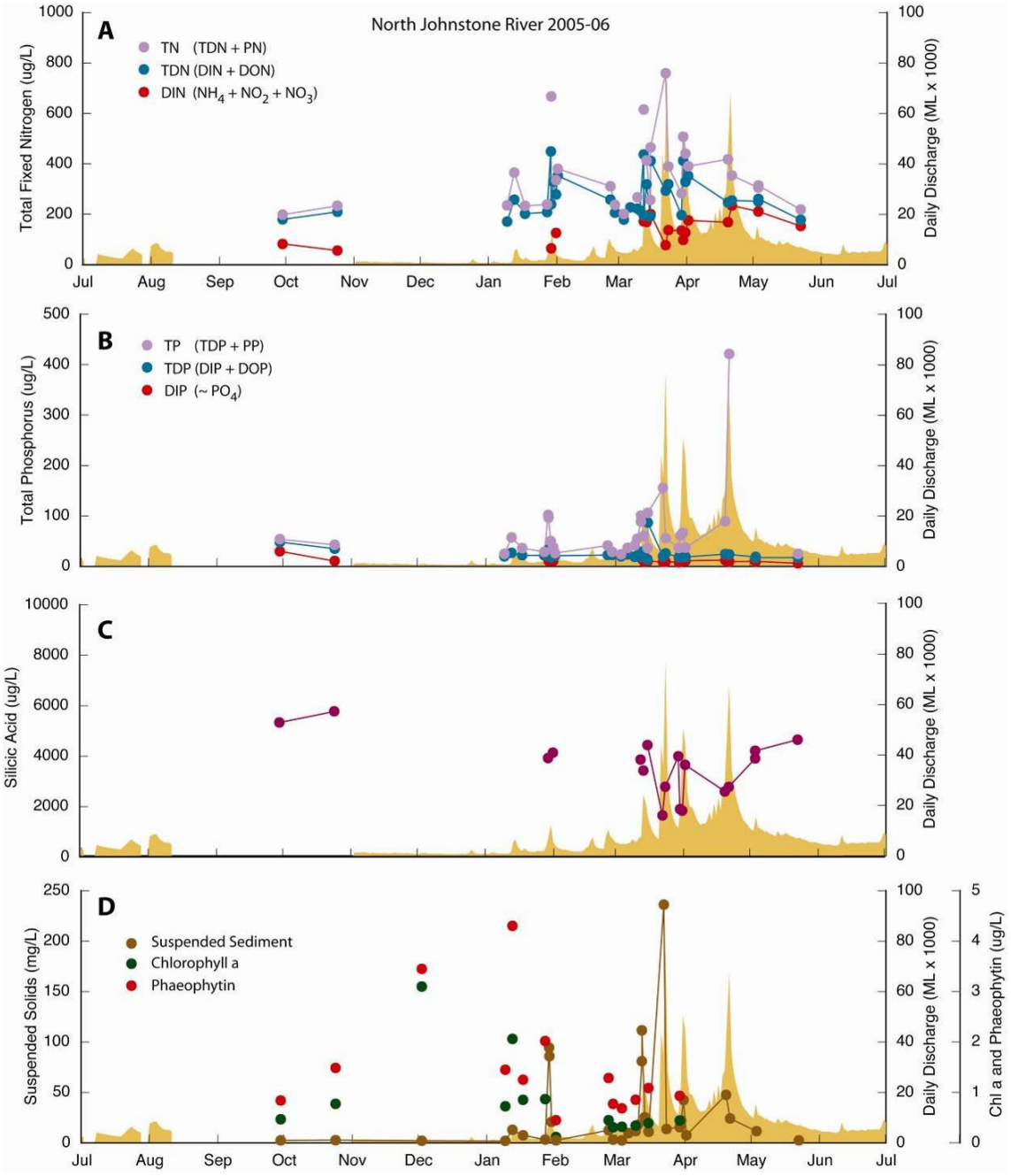


*Figure 2.13. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **North Johnstone River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

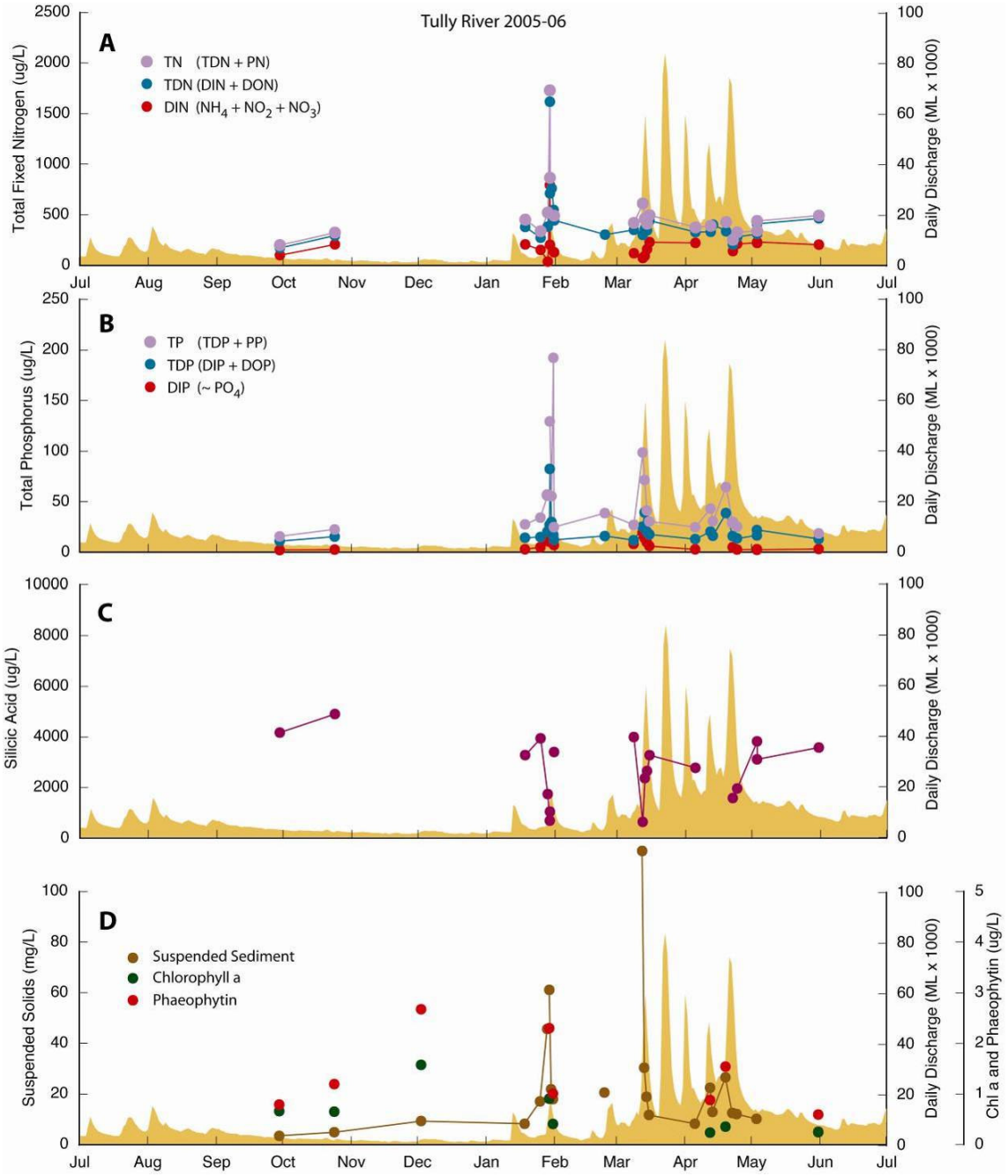


*Figure 2.14. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Tully River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

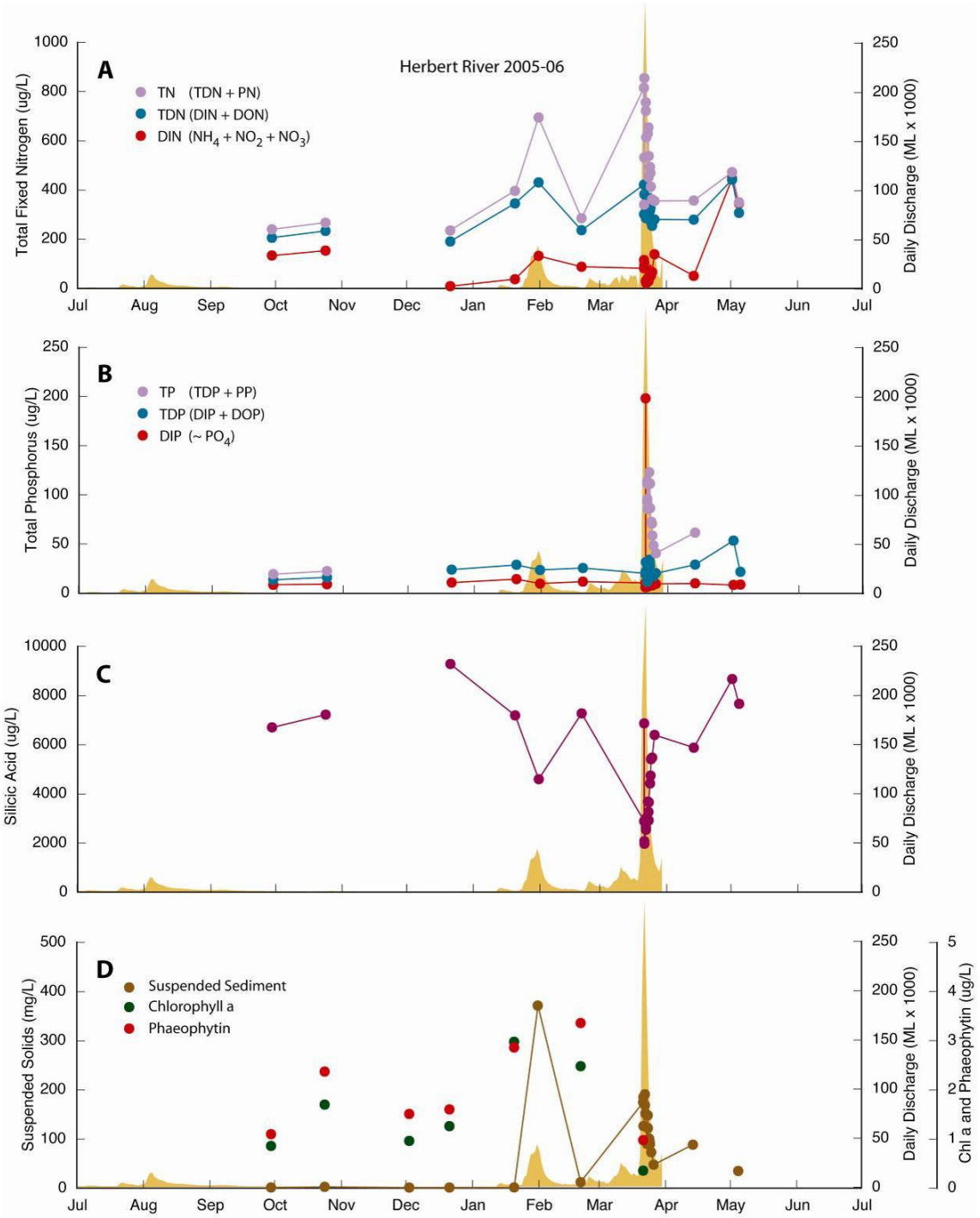


*Figure 2.15. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Herbert River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

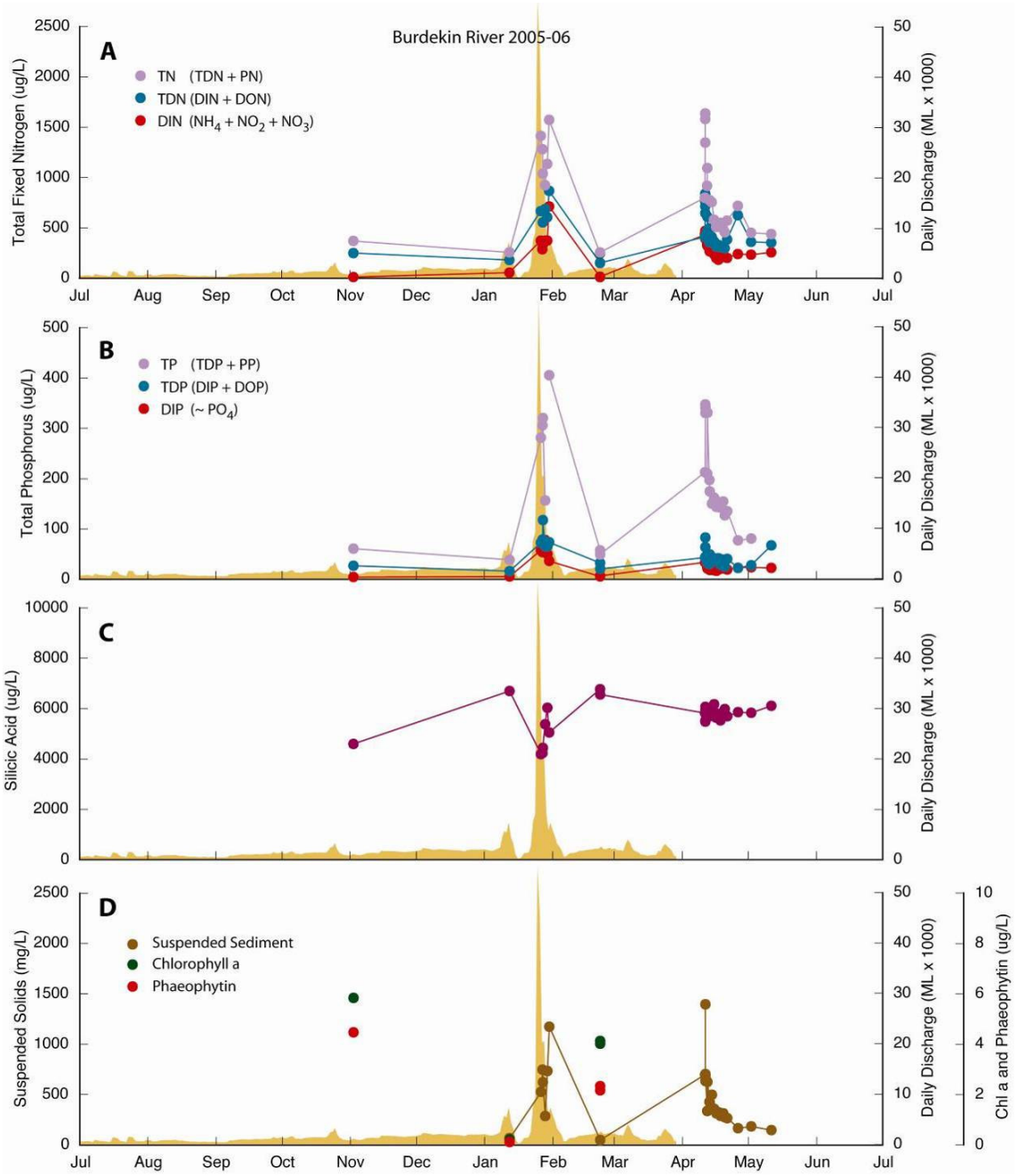


*Figure 2.16. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Burdekin River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

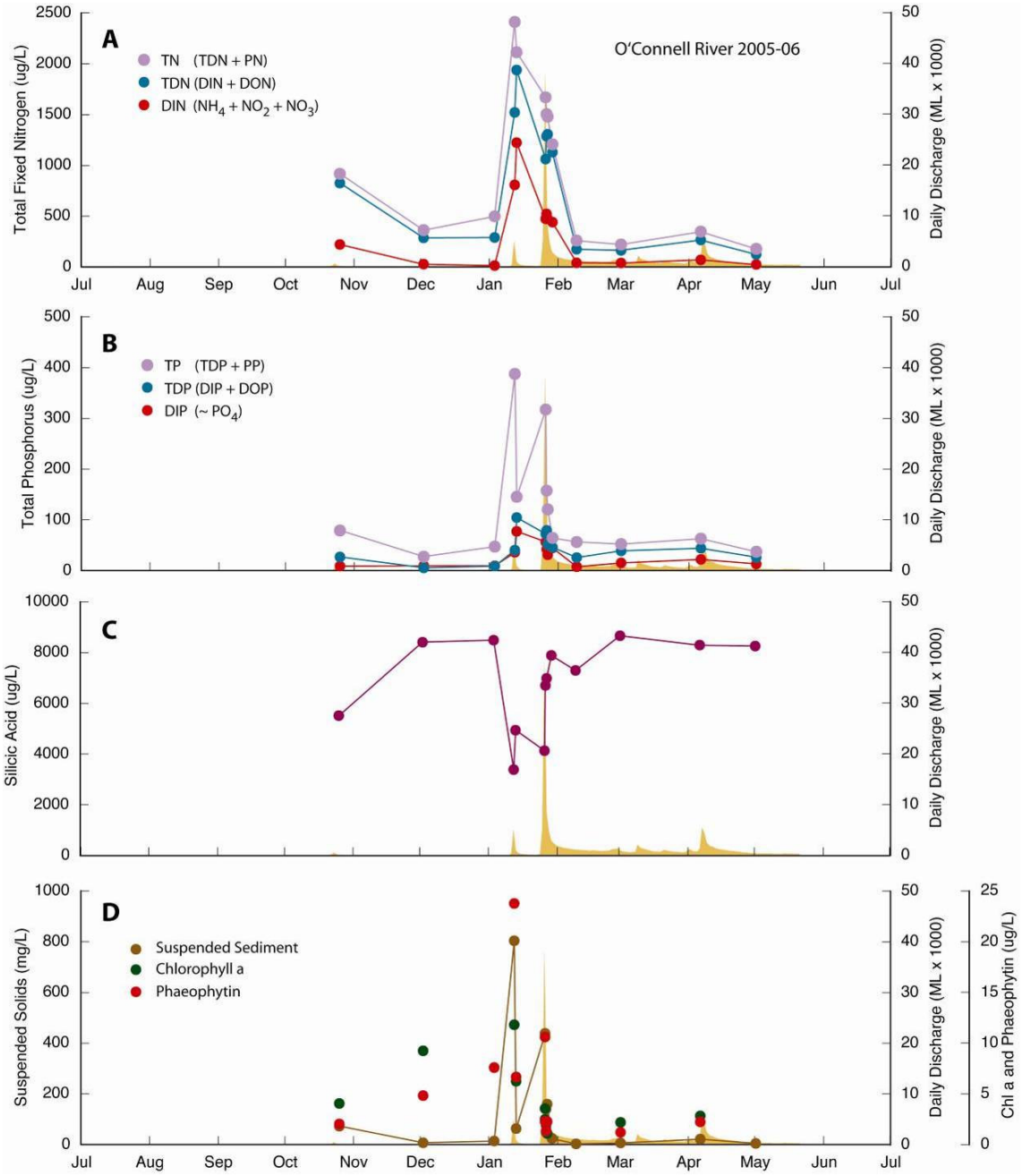


*Figure 2.17. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **O'Connell River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

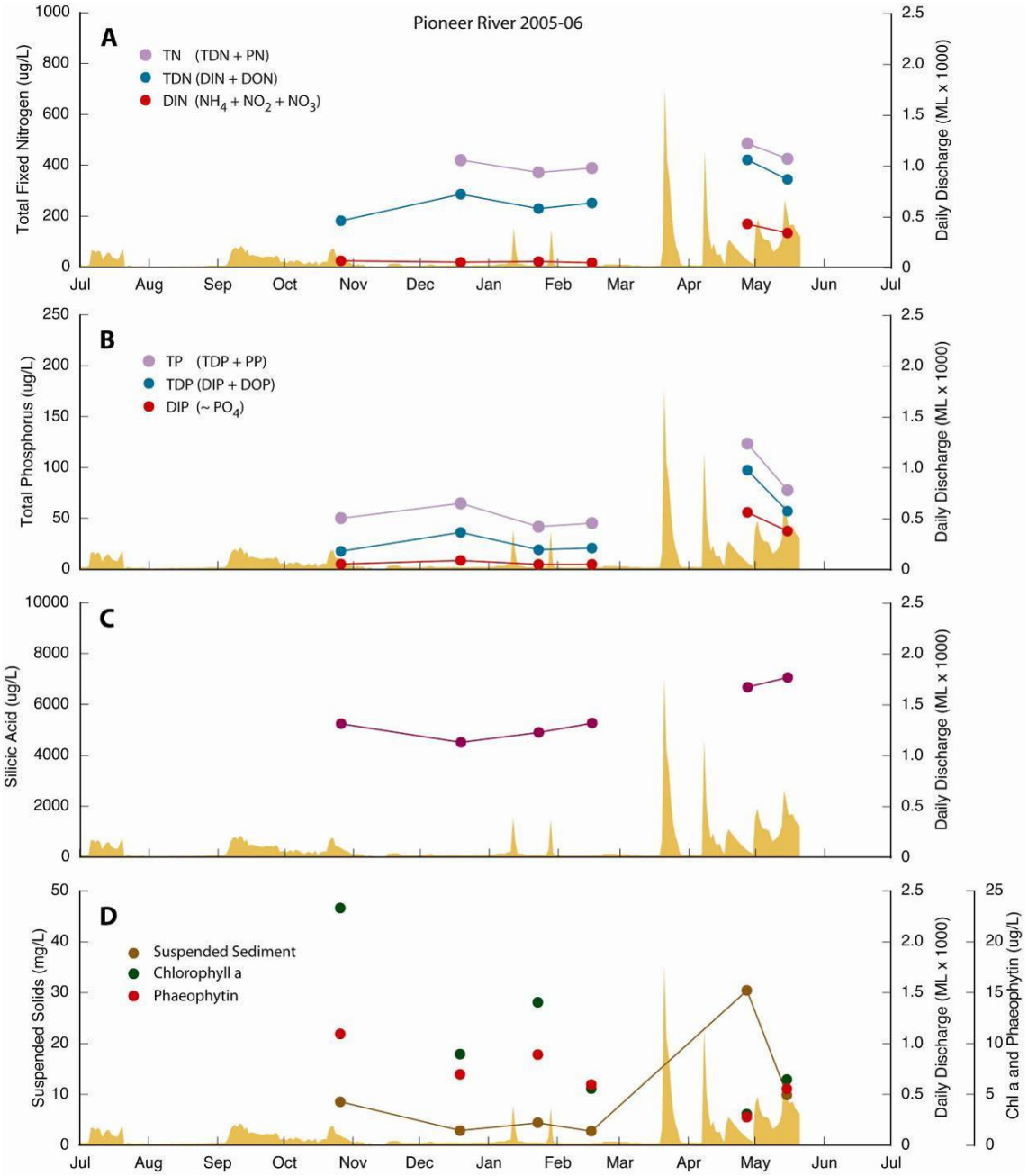


*Figure 2.18. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Pioneer River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

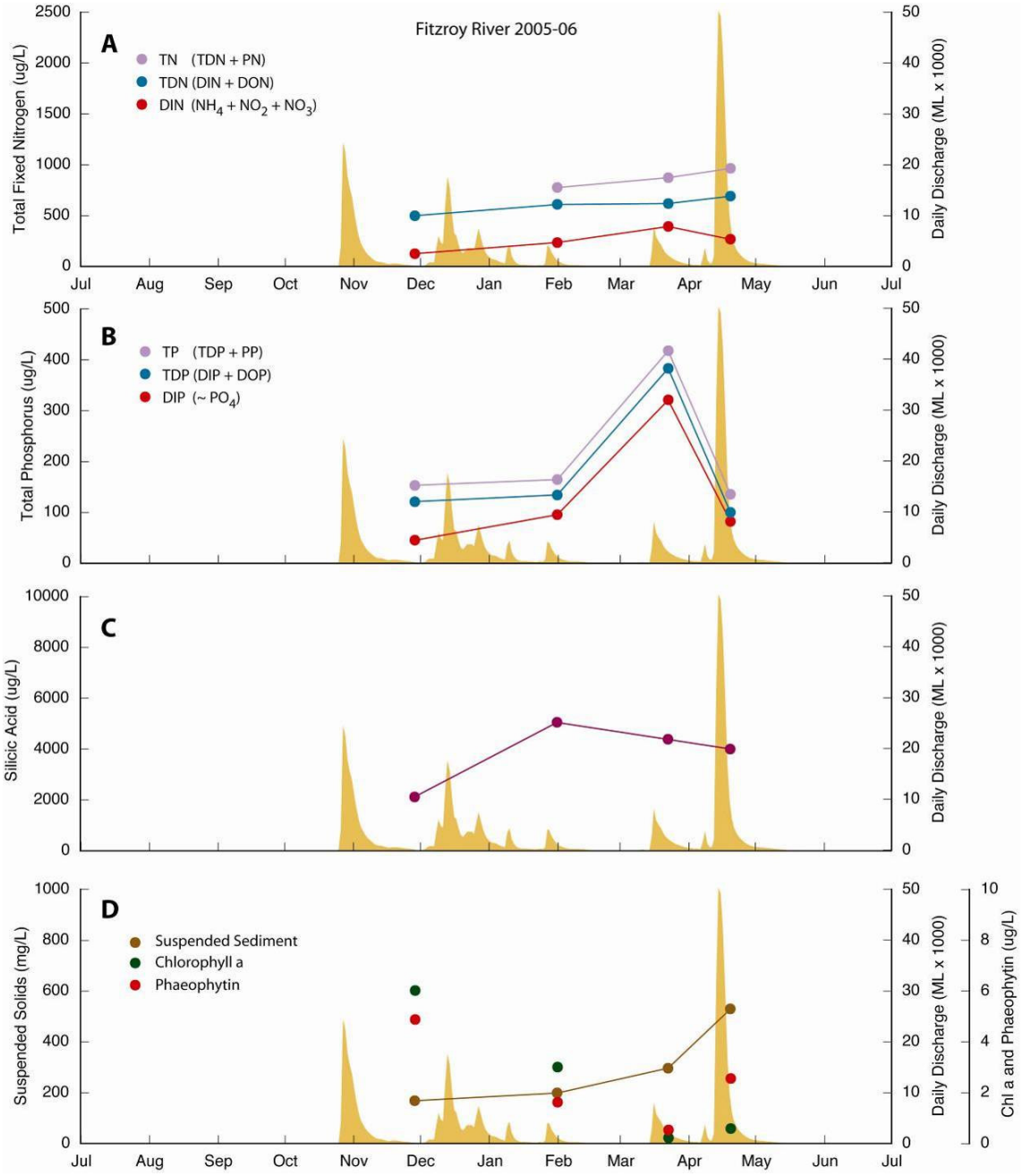


*Figure 2.19. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Fitzroy River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Values shown are the mean of duplicate analyses.*

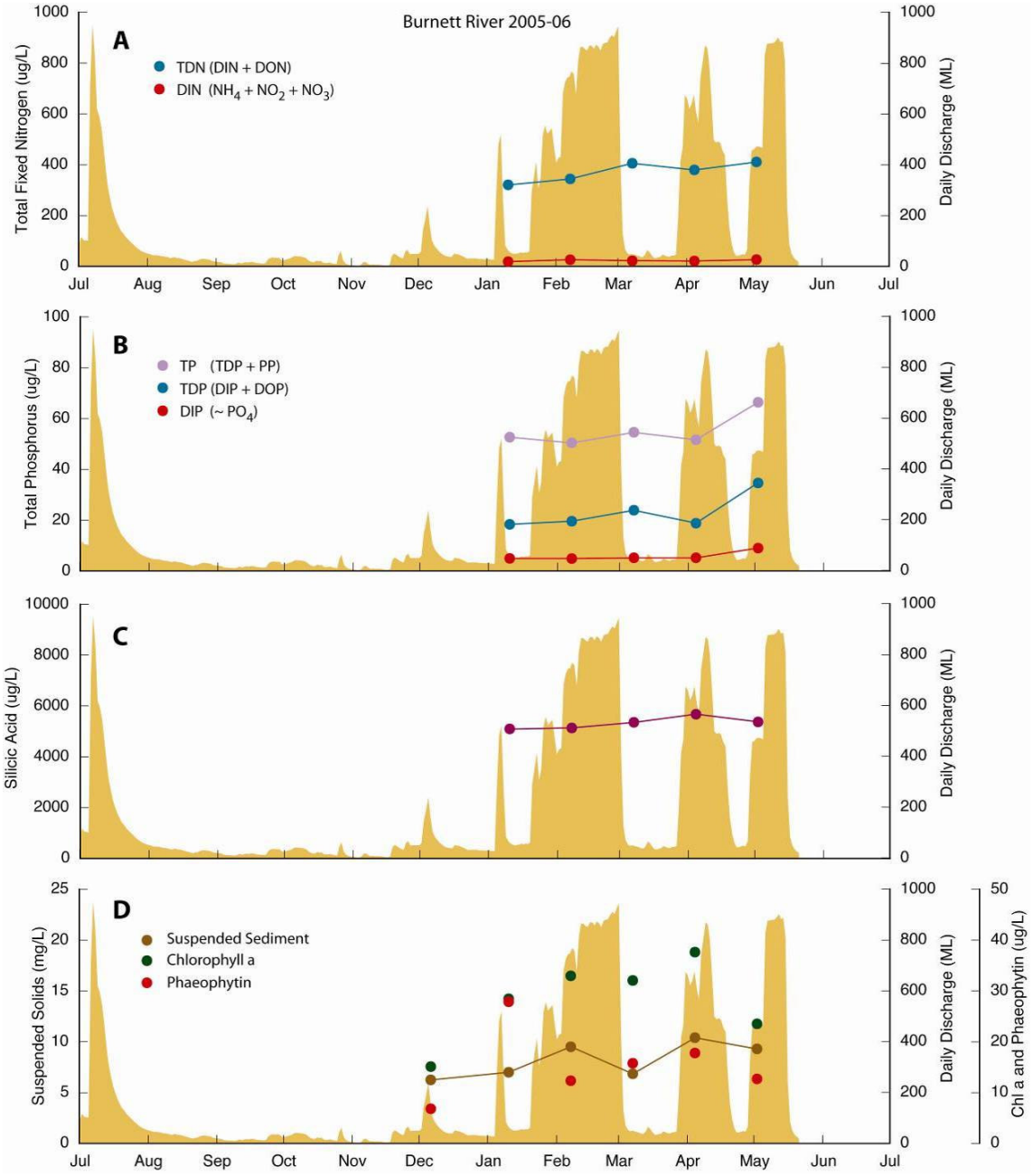


*Figure 2.20. Concentrations of (A) fixed nitrogen, (B) phosphorus, (C) dissolved silicic acid and (D) suspended solids, chlorophyll and phaeophytin in water samples manually collected from the lower **Burnett River** over 2005-06 in relation to concurrent daily discharge. Flow data was supplied by DNRMW. Note the change of the discharge scale relative to Figures 2.11 to 2.19. Concentration values shown are the means of duplicate analyses.*

Seasonal summaries of water quality parameters are presented in Figures 2.21 to 2.23. There are very few values for the dry season. In general the Dry Tropics rivers had higher concentrations of water quality parameters than Wet Tropics rivers. DIN was highest in the O'Connell and Burdekin Rivers, DON in the O'Connell and Fitzroy, PN and SS in the Burdekin and Fitzroy. Phosphate was highest in the Fitzroy and O'Connell, DOP in the Fitzroy and Pioneer, and PP in the Burdekin and Herbert Rivers.

Wet-season quartile ranges of suspended sediment concentrations were consistently higher than sample ranges observed in the much fewer dry season samples reflecting the generally higher levels of rainfall, catchment erosion and sediment transport (Figure 2.21a). The wet season sediment concentration ranges often overlapped the narrow dry season range as a number of low-flow samplings were carried out during the nominal wet season period, either before floods occurred, or during the period between flood events. Suspended sediment concentration quartiles in the Burdekin and Fitzroy Rivers were higher than those of the smaller rivers of the Wet Tropics and central Great Barrier Reef, in part due to the perennially higher turbidity levels in these rivers from very small clay particles.

Overall, silicic acid concentrations in regional rivers varied over a 5-fold range, from ca. 1,500 to 7,500 $\mu g\ L^{-1}$ (Figure 2.21b). Most concentrations fell into the 2,000 to 6,000 $\mu g\ L^{-1}$ range. In rivers where both wet and dry season samples were collected, wet season silicic acid concentration quartile ranges were generally lower than in the dry season, reflecting dilution during elevated flow events. This dilution indicates that weathering of catchment soils and rocks is at least partially de-coupled from the seasonal rainfall regime. Pools of Si do not appear to accumulate in catchment soils and groundwater and are rapidly flushed into the river systems by surface and sub-surface flows at a limited rate. This limited input is readily diluted by low-Si rainwater during brief storm events in the catchments.

Dry season summary statistics for concentrations of major nitrogen species are considerably attenuated due to the small number of samples taken (Figure 2.22a). Within the larger number of wet season samples, DO) was the principal pool of fixed nitrogen carried in eight of the ten priority rivers (all but the Herbert and Burdekin) (Figure 2.22b). PN was the next most important pool. DIN concentrations exceeded those of PN in two Wet Tropics (North Johnstone, Tully) and one central-Great Barrier Reef river (O'Connell).

The relative abundance of the three major types of phosphorus varied with river type and flow regime (Figure 2.23a, b). For the relatively small number of dry season samples, dissolved P forms (organic or inorganic) were the predominant forms measured in the four wet tropics rivers while particulate P was the most abundant form in the O'Connell and Pioneer Rivers. In low-flow conditions, the dry season samples had to be collected from a weir impoundment or a moderate sized, but slowly flowing billabong in the riverbed in the latter two rivers.

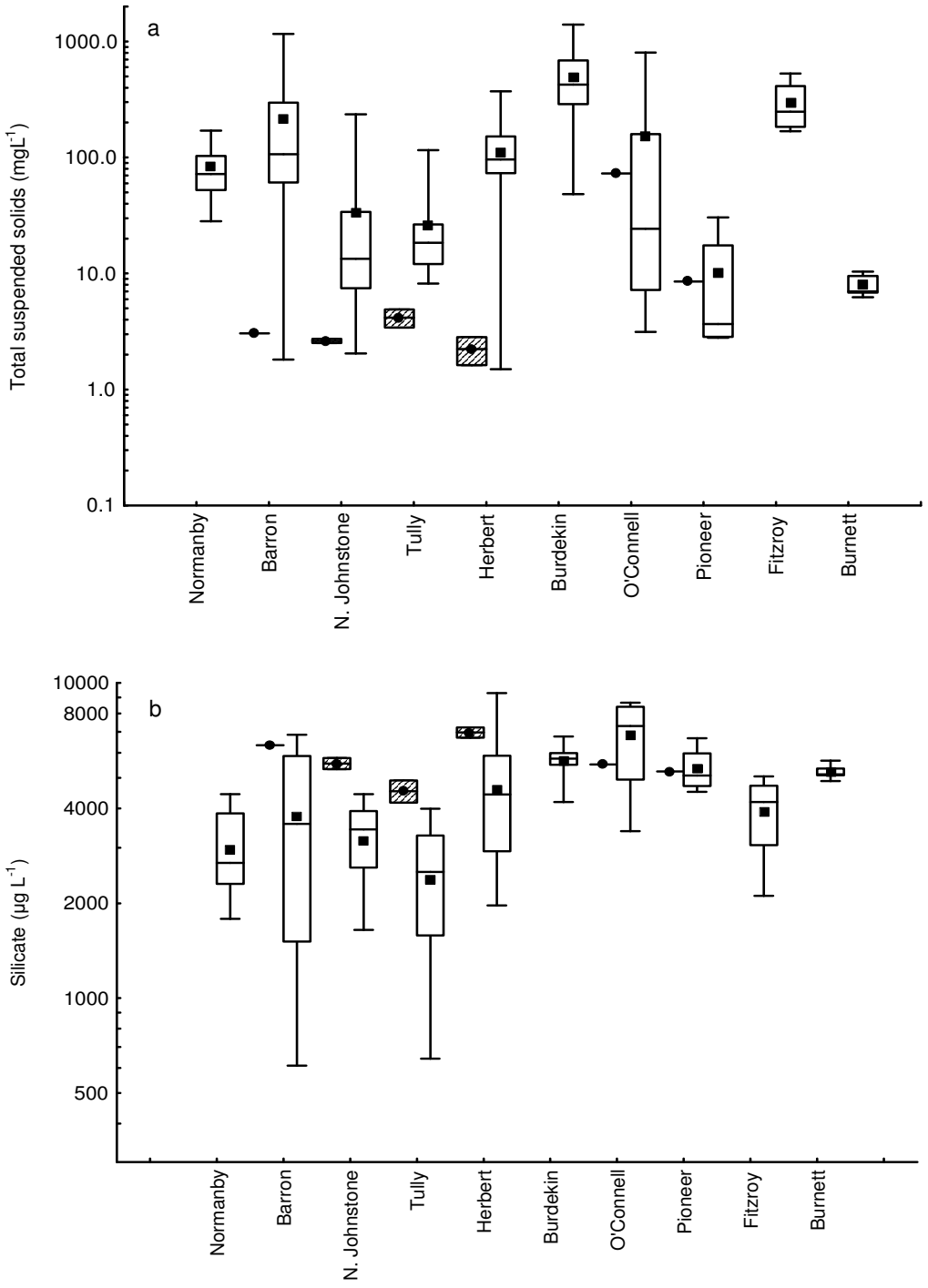


Figure 2.21. Summary of a) ***total suspended solids****, and b)* ***silicic acid*** *concentrations for the ten priority rivers for the sampling period 1 May 2005 to 30 April 2006. Data are for the dry season (May-Oct, grey boxes) and wet season (Nov-Apr, open boxes). See p. 25 for more information about the box plot representation. Note that dry season sampling was limited.*

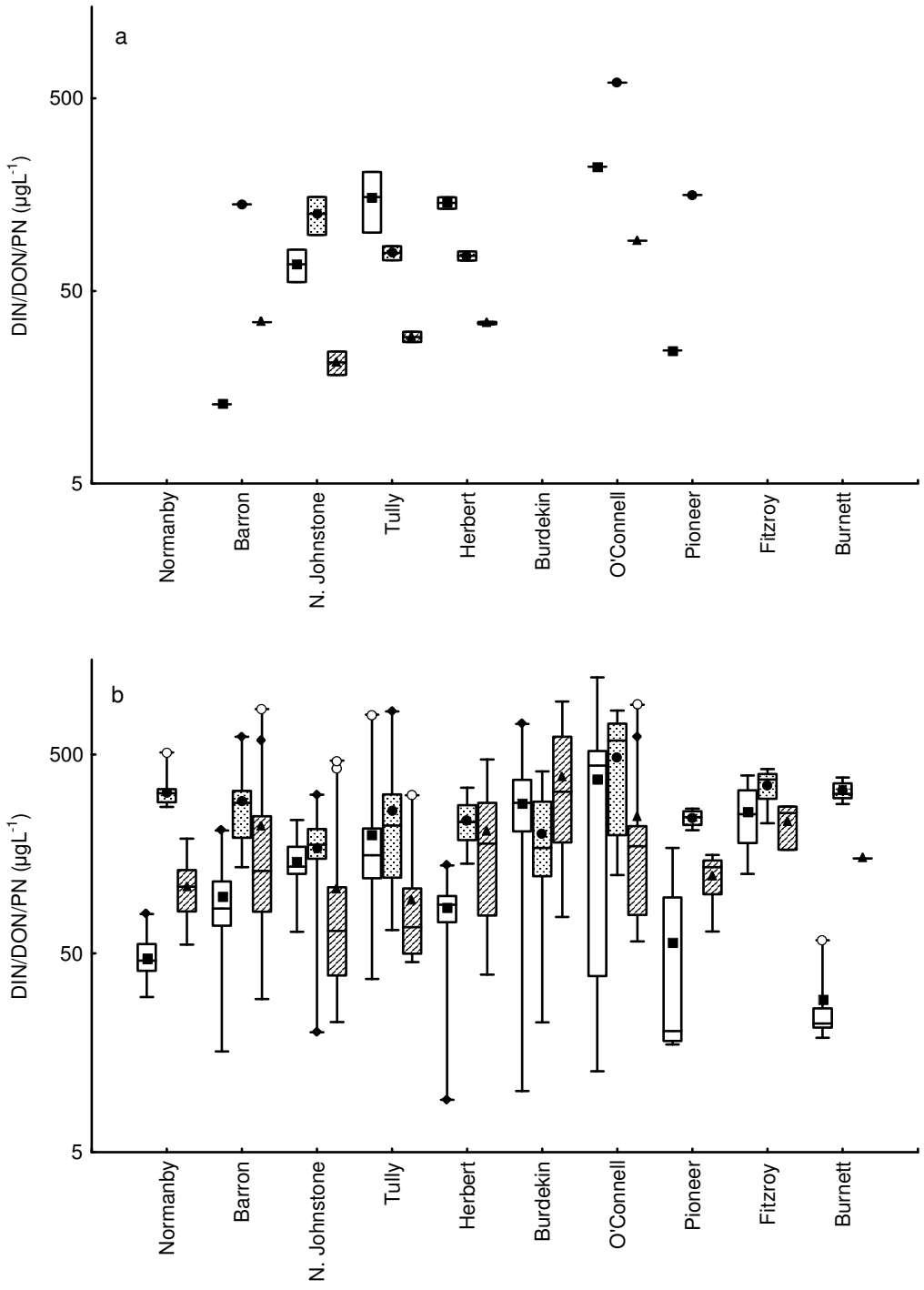


*Figure 2.22 Summary of measured concentrations of **nitrogen species** (Dissolved oxidised nitrogen – open boxes; dissolved organic nitrogen – light grey boxes; particulate nitrogen – dark grey boxes) for the ten priority rivers for the sampling period 1 May 2005 to 30 April 2006. Data are for the a) dry season (May-Oct) and Bottom) and b) wet season (Nov-Apr). See p. 25 for more information about the box plot representation. Note that dry season sampling was limited.*

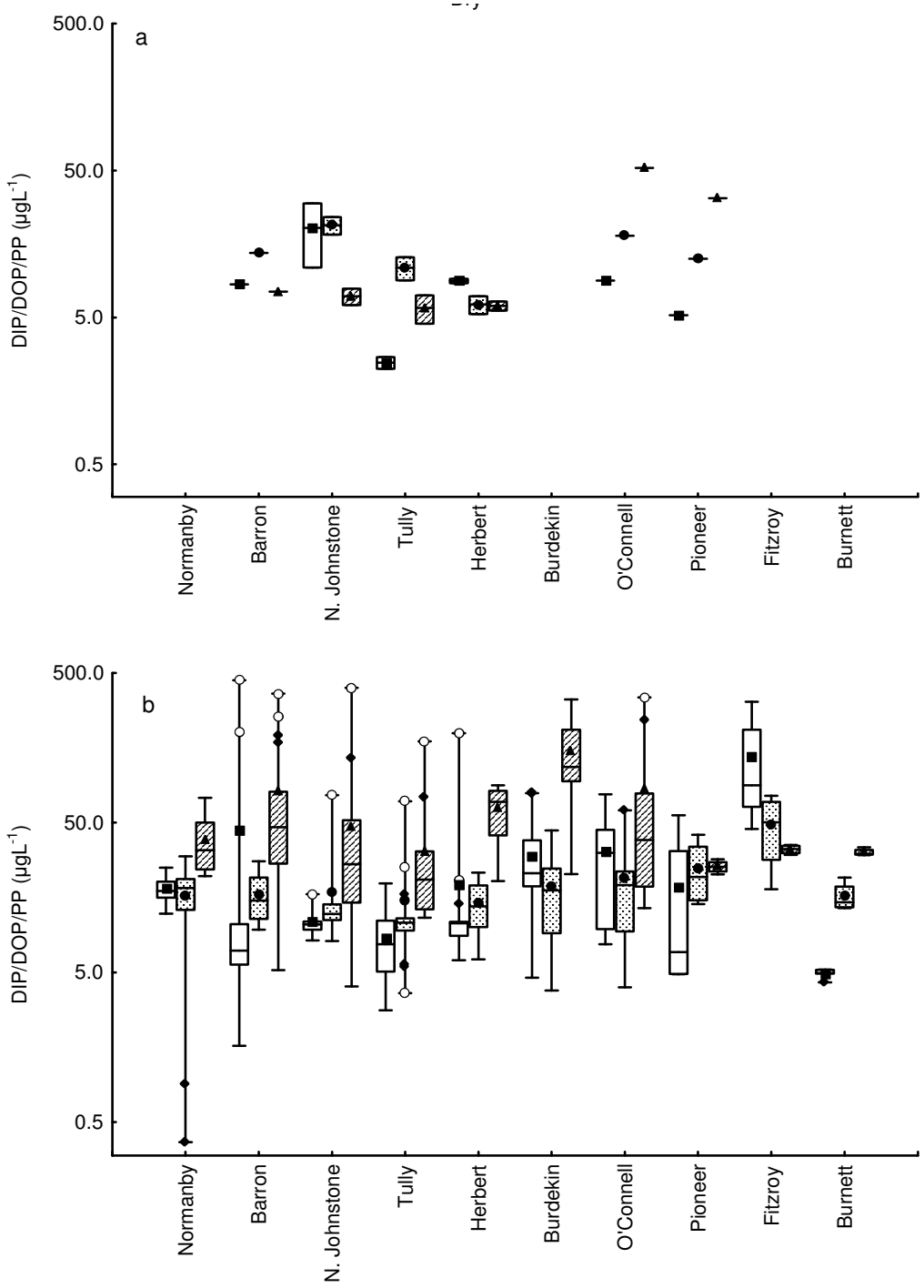


*Figure 2.23 Summary of measured concentrations of **phosphorus species** (Dissolved oxidised phosphorus – open boxes; dissolved organic phosphorus – light grey boxes; particulate phosphorus – dark grey boxes) for the ten priority rivers for the sampling period 1 May 2005 to 30 April 2006. Data are for the a) dry season (May-Oct) and b) wet season (Nov-Apr). See p. 25 for more information about the box plot representation. Note that dry season sampling was limited.*

Because of its close association with suspended sediment, particulate phosphorus was the predominant form of P in the wet season, run-of-river samples collected in all rivers between the Normanby and O'Connell. Concentration ranges of the two major dissolved forms (DIP, DOP) were relatively similar in these rivers. Wet season samples from the three southern-most rivers (Pioneer, Fitzroy, Burnett) were collected from weir impoundments under low to very-low flow conditions. Dissolved P forms were the most abundant in the Fitzroy River while particulate P (PP) was the predominant form in the Burnett River. Concentration ranges of major dissolved and particulate forms of P were very similar in the small number of Pioneer River samples.

Tables 2.7 and 2.8 summarise median concentrations of suspended sediment and the principal nitrogen and phosphorus species for the Reef Plan MMP priority rivers and compares these values to the Queensland Water Quality Guideline Values for these parameters in the Wet Tropics and Central Coast Guideline Regions (Environmental Protection Agency, 2006). In most rivers, most water quality parameters had median concentrations exceeding the Guideline values. Exceptions to this are the Pioneer River, with all parameters below the Guideline values, and the Burnett River with only ammonium and Total P exceeding.

Summary data for concentrations of water quality parameters during the limited 2004-05 wet season sampling in the Barron, North and South Johnstone, Burdekin and Fitzroy Rivers are in Appendix 1: Table A1-2.11 .

Table 2.7 Comparison of median concentrations of water quality parameters from limited wet season water sampling in 2004/05 with respective regional guideline values from the Queensland Water Quality Guidelines (Environmental Protection Agency, 2006). Shaded values exceed the guideline values. Nd= guideline value not determined.

River	NH_4	NO_x	DON	TN	PO_4	TP	SS
	µg N L^{-1}	µg N L^{-1}	µg N L^{-1}	µg N L^{-1}	µg P L^{-1}	µg P L^{-1}	mg L^{-1}
Wet Tropics Region Guideline values:							
	10	**30**	**200**	**240**	**4**	**10**	**nd**
Barron	17	102	230	629	14	83	81
North Johnstone	30	134	151	471	16	98	92
South Johnstone	17	138	90	390	19	63	49
Central Coast Region Guideline values:							
	20	**60**	**420**	**500**	**20**	**50**	**10**
Burdekin	42	307	276	1400	23	269	998
Fitzroy	32	469	384	1203	54	135	497

Table 2.8 Comparison of median concentrations of water quality parameters from water sampling in 2005/06 with respective regional guideline values from the Queensland Water Quality Guidelines (Environmental Protection Agency, 2006). Shaded values exceed the guideline values. Nd= guideline value not determined.

River	NH_4	NO_x	DON	TN	PO_4	TP	SS
	µg N L^{-1}	µg N L^{-1}	µg N L^{-1}	µg N L^{-1}	µg P L^{-1}	µg P L^{-1}	mg L^{-1}
Wet Tropics Region Guideline values:							
	10	**30**	**200**	**240**	**4**	**10**	**nd**
Normanby	17	27	320	465	18	62	72
Barron	15	98	233	561	10	76	82
North Johnstone	20	134	153	383	11	57	18
Tully	7	186	186	427	5	33	15
Herbert	16	70	227	470	10	85	91
Central Coast Region Guideline values:							
	20	**60**	**420**	**500**	**20**	**50**	**10**
Burdekin	32	282	227	1112	23	226	609
O'Connell	23	187	588	916	22	65	24
Pioneer	15	9	222	420	7	58	7
Fitzroy	25	394	375	1005	58	151	411
Burnett	23	1	338	491	5	54	8

Annual loads of SS and nutrients (± the 90% confidence interval) were statistically estimated for seven of the priority rivers where there were sufficient data to make useful calculations (Table 2.9). Estimated loads ranged from 327 tonnes in the Pioneer River to 437,000 tonnes in the Burdekin River. Estimated 90% confidence ranges vary from ca 85% (North Johnstone, Tully, Burdekin Rivers) to 213 % (O'Connell River) of the estimated export value.

Table 2.9 Estimates of river export loads of suspended solids and nutrients in tonnes per year. Load estimate is given in bold print, values above and below are the limits of the 90% confidence interval.

River		SS	DIN	TDN	PN	DIP	TDP	PP
Barron	5%	65000	60	260	60	6	19	25
	Load estimate	**144600**	**87**	**309**	**113**	**20**	**21**	**54**
	95%	254000	110	360	180	50	23	90
N-Johnstone	5%	40000	370	900	160	15	47	50
	Load estimate	**58310**	**478**	**1067**	**218**	**20**	**56**	**68**
	95%	90000	600	1300	300	25	68	90
Tully	5%	47000	500	1150	200	17	60	60
	Load estimate	**71640**	**617**	**1349**	**264**	**23**	**72**	**81**
	95%	110000	800	1600	360	30	90	110
Herbert	5%	66000	200	700	200	25	50	40
	Load estimate	**180100**	**260**	**772**	**321**	**39**	**57**	**83**
	95%	360000	310	860	440	60	60	130
Burdekin	5%	250000	180	340	210	25	40	75
	Load estimate	**437300**	**273**	**456**	**356**	**38**	**52**	**133**
	95%	620000	370	550	500	50	60	200
O'Connell	5%	5000	30	90	20	4	5	5
	Load estimate	**21660**	**67**	**152**	**38**	**5**	**8**	**6**
	95%	50000	110	200	70	7	12	8
Pioneer	5%	150	1	8	3	0.2	1	1
	Load estimate	**327**	**2**	**10**	**4**	**1**	**1**	**1**
	95%	570	4	11	4	1	2	1

Discussion

A variety of discharge regimes were observed in the ten Reef Plan MMP priority rivers over the 2005-06 year. Overall, however, gauged total discharge from the 10 priority rivers (ca. 19 km^3, Table 2.4) was approximately half of the long-term average for these drainage basins (ca. 38 Km^3, Furnas, 2003). As sediment and nutrient exports are largely proportional to freshwater runoff (Furnas, 2003), total exports of sediments and nutrients to the GBR lagoon in both 2004-05 and 2005-06 can be expected to be significantly lower than normal.

In the Wet Tropics, the largest flows occurred after TC Larry (20 March 2006). TC Larry, however, was a fast-moving and not particularly "wet" cyclone and the immediate post-cyclone floods were not very large compared to monsoonal and cyclone-associated floods in previous years. The cyclone, however, initiated an extended period of rainfall in the central Wet Tropics and much of the wet season discharge from the four wet tropics rivers (Barron, North Johnstone, Tully, Herbert) came in the two months following the cyclone (e.g. Figure 2.7, Table 2.7). Overall, discharges from the Tully and Herbert Rivers were close to the long-term average, while the Johnstone River discharged approximately 2/3 of its long-term average freshwater output.

A DNRMW gauging station was constructed on the Normanby River at Kalpowar Crossing (Lakefield National Park) in December 2005 and a rating curve was established in December 2005. This station provides the first good estimate of freshwater discharge from the Normanby River as previous estimates of discharge are based on stations (Battlecamp, Laura) with smaller catchments established much further upstream to support agricultural development in the upper Normanby basin. The relatively short flow, turbidity and nutrient records obtained during the 2005-06 should be regarded as provisional until more flow data and samples are obtained at this site with proper gauging. In addition, the relative contribution of the Normanby to total discharge from the many rivers and streams of this drainage basin need to be better resolved. Some further work is also required to extend the rating curve to low-flow conditions at Kalpowar; however, discharge and export under low-flow conditions only makes a small contribution to total annual discharge and export.

Discharges from the two large dry-catchment rivers (Burdekin, Fitzroy) were relatively moderate in both 2004-05 and 2005-06 and well below the long-term average discharges for these catchments. Measured discharge from the Burdekin River in 2005-06 was approximately 25% of the long-term average, while discharge from the Fitzroy was ca. 12% of the long-term average. The largest flow in the Burdekin (mid-April) followed the degeneration of TC Larry over the northwest corner of the catchment. Significant discharge from the Fitzroy followed late wet-season rains along the western boundary of the catchment. Most of the water from these rains, however, flowed westward into the Murray-Darling and Coopers Creek system.

Discharges from the two rivers in the Mackay Whitsunday NRM region (O'Connell, Pioneer) were relatively small. Only one significant flow event occurred in the O'Connell River which remains relatively dry for much of the year. Measured discharge from the Pioneer River was only on the order of 5% of the long-term average.

Little rain fell in the Burnett River catchment over 2005-06, and accordingly, there was little discharge and export from the catchment. Measured discharge was < 10% of the long-term average discharge.

Sediment and nutrient exports

Fine sediment exports were estimated from nine of the ten priority Reef Plan MMP rivers where useful turbidity logger records were obtained (the signal to noise ratio in the Burnett record was too small to extract a reliable export estimate). Sediment exports were calculated from trapezoidal integrations of the Mudlogger turbidity records (30 minute intervals) and from statistical analysis and integration of the manually collected samples. Calculated estimates of sediment export for the priority rivers from Mudlogger records for both the 2004-05 and 2005-06 wet seasons are given in Table 2.5.

The Mudloggers primarily measure light attenuation associated with very small particles (< 35 μm). Larger particles contribute less to attenuation, but proportionally more to sediment mass. Because of their ability to resolve short-term variability in turbidity, the primary constraints on estimates of sediment export derived from Mudlogger data are the accuracy and precision of discharge rate estimates and the turbidity: sediment-mass relationship for a given river. Examination of the match-ups between Mudlogger-derived and manual sediment samples indicates that there are significant differences in this relationship between "wet" and "dry" rivers, and between some "wet" rivers (Tully vs. the other "wets").

Factors affecting the accuracy of sediment export estimates from less frequent manually collected samples have been extensively discussed elsewhere (e.g. Fox, 2004 and references therein).

Table 2.10 shows a comparison between fine sediment export estimated derived from integration of the Mudlogger records (Table 2.5) and from statistical analysis of the manually collected samples. In all cases, the Mudlogger-derived export estimates fell within the broad 95% confidence intervals derived from statistical modelling with the manual samples. Particularly close matches were calculated for the North Johnstone, Tully, O'Connell and Pioneer Rivers. With the exception of the Pioneer samples, there was good coverage of both high and low-flow conditions in these river time series of samples. The modelled estimate of sediment export from the Burdekin River is almost certainly low as samples were not collected during high-flow, high-sediment conditions.

Table 2.10 A comparison between estimated wet season exports of fine sediment derived from Mudlogger turbidity time series and statistical models using manually sampled sediment concentrations.

River	Mudlogger derived (tonnes)	Statistical modelling (tonnes)
Barron*	84,000	145,000
North Johnstone	62,000	58,000
Tully	58,000	72,000
Herbert*	314,000	180,000
Burdekin*	609,000	437,000
O'Connell	21,000	22,000
Pioneer	200	330

* Incomplete wet season flow data available

There were strong positive correlations between concentrations of SS and PN and PP (Figures 2.24a, b). The close correlation indicates that reliable estimates of SS load for particular rivers can be useful for estimating particulate nutrient exports, and soluble exports if dissolved/total ratios are known and reasonably constrained.

Based on the regression slopes, sediment particles from the small catchments of the wet tropics and central Queensland averaged approximately 1200 mg/Kg N (0.12%) and 400 mg/Kg P (0.04%). In the poorer soils of the dry tropics catchments (Burdekin, Fitzroy), sediment particles contained approximately 650 mg/Kg N 0.065 percent N and 27 mg/Kg P (0.0027%). These ratios are similar to average surface soil N and P content calculated from regional soil analyses carried out by DNRMW (Furnas, 2003). They indicate that GBR catchment soils are relatively nutrient poor, particularly in the unfertilised soils of large grazing catchments. Significant exports of N and P from these catchments reflect the significant levels of erosion in these catchments.

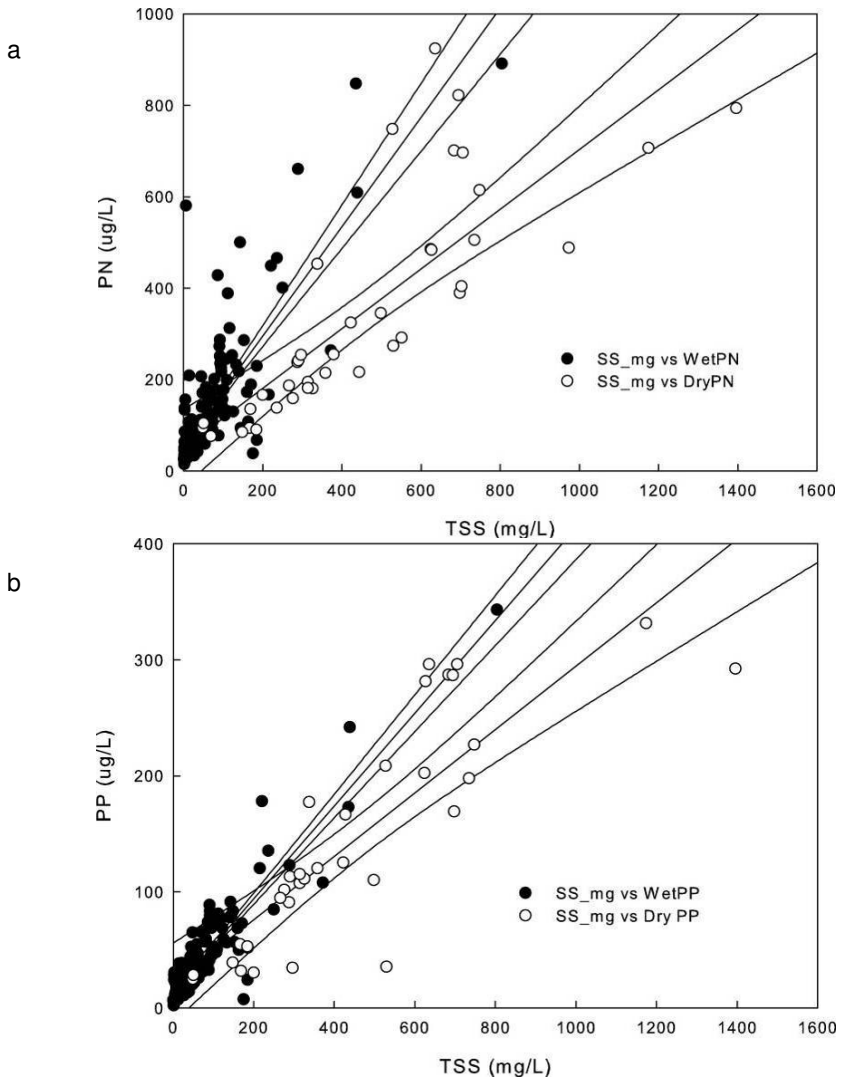


Figure 2.24. Relationships between measured concentrations of a) particulate nitrogen (PN) and b) particulate phosphorus (PP) with suspended sediment concentrations in wet and dry catchment rivers over 2005-06. Equations relating measured particulate nutrient concentrations to suspended sediment concentrations:

Wet Rivers (Normanby, Barron, N-Johnstone, Tully, Herbert, O'Connell, Pioneer):

$PN\ (\mu g\ L^{-1}) = 1.20\ (TSS\ [mg\ L^{-1}] + 56.2;\ r^2 = 0.66$

$PP\ (\mu g\ L^{-1}) = 0.40\ (TSS\ [mg\ L^{-1}] + 14.4;\ r^2 = 0.84$

Dry Rivers (Burdekin, Fitzroy)

$PN\ (\mu g\ L^{-1}) = 0.65\ (TSS\ [mg\ L^{-1}] + 50.1;\ r^2 = 0.66$

$PP\ (\mu g\ L^{-1}) = 0.27\ (TSS\ [mg\ L^{-1}] + 21.2;\ r^2 = 0.71$

Estimates of nitrogen and phosphorus exports for the main nutrient forms could be calculated for seven of the ten priority rivers (Table 2.9). Reliable nutrient export estimates could not be calculated for the Normanby River as only a partial wet season flow record was available at Kalpowar and nutrient sampling was limited to a single event. No export estimates were calculated for the Fitzroy and Burnett Rivers due to the low discharges from these rivers, the small number of samples taken and the temporal mismatches between sampling and flow in the Fitzroy River. Sediment and nutrient load estimates are summarised in Table 2.11 and as discharge-weighted loads in Table 2.12.

Table 2.11 Estimated total suspended sediment, total nitrogen and total phosphorus loads exported from the ten priority Reef Plan MMP rivers over the wet season 2005/06 from mudloggers records and over the 2005-06 year from modelled (*) estimates. nc = not calculated.

River	Mudlogger SS (10^3 t)	SS(10^3 t)*	TN (t)*	TP (t)*
Normanby	189	nc	nc	nc
Barron	84	145	510	95
North Johnstone	62	58	1760	115
Tully	58	72	2230	180
Herbert	314	180	1350	255
Burdekin	609	437	1100	225
O'Connell	21	22	105	125
Pioneer	0.2	0.3	15	5
Fitzroy	215	nc	nc	nc
Burnett	nc	nc	nc	nc

Table 2.12 Estimated discharge-weighted loads of total suspended sediment, total nitrogen and total phosphorus loads exported from the ten priority Reef Plan MMP rivers over the wet season 2005/06 from mudloggers records and over the 2005-06 year from modelled estimates. nc = not calculated.*

River	Mudlogger SS (10^3 t km^{-3})	SS (10^3 t km^{-3})*	TN (t km^{-3})*	TP (t km^{-3})*
Normanby	67	nc	nc	nc
Barron	158	207	729	136
N. Johnstone	37	26	789	52
Tully	22	21	643	52
Herbert	73	37	276	52
Burdekin	316	200	505	103
O'Connell	117	220	1050	1250
Pioneer	10	5	250	83
Fitzroy	422	nc	nc	nc
Burnett	nc	nc	nc	nc

During the 2005-06 year, the seven modelled rivers had an aggregate total freshwater discharge of 14.5 km^3. This compares to an estimated long-term average discharge of 25.8 km^3 from these rivers (56 %) (see Furnas, 2003 Table 32).

Estimated total fine sediment exports from the seven rivers calculated from Mudlogger wet season time series and statistical modelling using the manual sample data were 1,148 and

914 ktonnes, respectively (22 and 17 % of the estimated long-term average for these rivers; (see Furnas, 2003 Table 32). The Mudlogger derived sediment export estimates are conservative as full wet-season flow and turbidity records were not obtained, nor do they include export over the dry season, though this is likely only a small fraction of the total due to the low flows and low suspended sediment concentrations in this period.

Estimated total annual nitrogen and phosphorus exports were 7,070 and 995 tonnes, respectively, compared to estimated long-term average exports of 15,460 tonnes (46%) and 2.530 tonnes (39 %) from these rivers (see Furnas, 2003 Table 32).

There is a variety of evidence for long-term change in sediment and nutrient loads entering the Great Barrier Reef lagoon. The relatively wide confidence limits on the modelled estimates of sediment and nutrient inputs (Table 2.9) obviously constrain any estimate of change over a relatively short (ca. 5-10 years) time period. Furnas (2003) presented evidence for change in baseload concentrations of oxidised nitrogen (principally NO_3^-), particulate nitrogen (PN) and phosphate ($PO_4^{=}$) in the Tully River over a 14 year (1987-2000) time period. Between ca. 1990 and 2000, there was a slow increase in the low-flow concentrations of these nutrient species in Tully River waters. A more detailed statistical analysis of this data set by De'ath (2005) confirmed these overall upward trends for TDN, PN, TDP, PP and SS.

A continued trend of increase in nutrient concentrations between 2000 and 2005-06 was not observed, however, in the one year of nutrient data obtained under the Reef Plan MMP. River nutrient and sediment concentrations in the limited wet season sampling in 2004-05 and the more comprehensive sampling in 2005-06 were very similar (Tables 2.7 and 2.8). Concentrations in river catchments south of Townsville are generally higher than the concentrations of the Wet Tropics and Cape York rivers. However, with the exception of the Pioneer and Burnett Rivers, all rivers exceeded the Queensland guideline values for most nutrients and suspended sediment concentrations.

Baseflow concentrations of nitrate and particulate N in the Tully River over 2005-06 were similar to those recorded in the early 1990's. Baseflow phosphate concentrations were higher than those of the early 1990's, but not higher than concentrations measured at the end of the 1990's. Considerable caution should be taken in drawing conclusions about long-term trends or changes in catchment nutrient dynamics from this limited additional amount of data. Ranges of nutrient concentrations in all rivers (including the Tully) varied from year to year, so the 'return' to early-90's concentrations in one year of data could reflect natural variability in the system rather than overall improvement in export performance due to land management practices. The data clearly show that extended time series (>10 years) are needed to reliably estimated natural trends and variability in these systems.

Because of the large, natural, inter-annual variability in freshwater runoff into the Great Barrier Reef lagoon, estimated loads of sediment and nutrients carried in runoff will also vary considerably from year to year. This variability makes it difficult to objectively compare spatial patterns or define temporal trends in runoff and the material exports carried by runoff. Observed variations might be due to either changes in inputs to river systems (erosion, land use) or spatial/temporal variations in rainfall and the resulting volume of freshwater runoff. To resolve this problem, it is important that any comparisons between rivers, or over time in a single river, be based upon volume-normalised (discharge-weighted) estimates of runoff; e.g. tonnes of sediment, nutrients or pollutants per unit volume of freshwater runoff (tonnes Km^{-3},

tonnes per ML, mg L^{-1}). After normalisation, year-to-year changes in the volume of freshwater discharge should be mostly associated with proportional changes in the quantity of sediment and nutrients transported into the GBR lagoon. From the perspective of the Reef Plan, the objective of efforts to change land-use practices within the GBR catchment is therefore to stabilise and progressively reduce the quantity of sediment, nutrients and other pollutants carried in a specific volume of freshwater discharge.

However, it needs to be noted that data from the Tully and Burdekin rivers show that concentrations of particulate nutrients and suspended sediments show a strong positive correlation with river flow while dissolved nutrients seem to be diluted at higher flow rates (De'ath, 2005). Over the two wet seasons monitored under the Reef Plan MMP to date, discharge-weighted fine sediment loads in rivers with deployed Mudloggers varied on the order of 2- to 3-fold. This variability will necessarily constrain short-term (<10 year) estimates of change in land-derived exports under the Reef Plan. The relationship between river flow and concentrations of transported materials requires further research and may complicate the straight-forward use of discharge-weighted sediment and nutrient loads to compare between rivers and over time.

Discharge-weighted average exports (tonnes per km^3) are, at one level, analogous to concentrations (mg per L) directly measured in monitoring programs. Direct comparisons, however, should be avoided or used with great caution as natural concentrations of nutrients and other materials in rivers can vary widely in response to flow conditions and other factors.

Conclusions

A primary goal of the Reef Plan is to reduce loads of terrestrial sediment, nutrients and other pollutants carried into waters of the Great Barrier Reef lagoon where they influence water quality in and the health of coastal ecosystems. The size of sediment and nutrient loads carried by individual rivers depends to a large degree upon the volume of catchment rainfall which causes erosion of catchment soils and transports eroded materials through the river systems to the GBR lagoon. The extent of erosion for a particular amount of water is further influenced by land use within the catchment, particularly through the effect of land use upon the extent and distribution of vegetation cover.

The available data for the 2005-06 wet season indicates that despite the occurrence of the most-powerful cyclone to hit North Queensland in over 50 years (TC Larry), the total freshwater discharge to the lagoon from the Reef Plan MMP priority rivers was on the order of half of the long-term average (Furnas, 2003). Only the Wet Tropics rivers were close to their long-term average. Estimated N and P inputs from monitored rivers were about 50 percent, while fine sediment inputs were closer to 20 percent of the estimated longer-term average input. This variation illustrates the importance of using discharge-weighted loads to examine inputs and take account of natural variability in concentrations and loads (see Tables 2.5 and 2.12).

Given the relatively small amount of export data collected under the Reef Plan MMP to date, it is difficult to draw definitive relationships between materials export (sediment, nutrients) and land use in the Reef Plan MMP priority catchments. The observed relationships are likely to be strongly influenced by the current variation in runoff from long-term averages. Volume-specific loads of suspended sediment measured by Mudloggers were low in the three wet tropics rivers (Johnstone, Tully, Herbert), which might be expected given the higher

levels of vegetation cover in the wet parts of these catchments, but the very low export value for the Pioneer River and absence of significant export from the Burnett River reflect the lack of rainfall in these catchments over the 2005-06 wet season. The average volume specific sediment load for the 'wet' rivers (all rivers other than the Burdekin, Fitzroy and Burnett) of 69,000 tonnes per km^3 is of similar order to the average previously calculated for wet rivers (39,000 tonnes per km^3; Furnas, 2003). The average discharge-weighted load in the Burdekin and Fitzroy rivers (368,000 tonnes per km^3) is also very similar to that previously calculated for these rivers from several years of data (366,000 tonnes per km^3; Furnas, 2003). With the continued collection of sediment export data with mudloggers, additional persistent differences in discharge-weighted loads between different rivers are likely to emerge.

Differences in discharge-weighted sediment loads per unit of catchment area (Table 2.5) between the Reef Plan MMP catchments mainly reflect differences in rainfall per unit area over the sampling period. However, the data from 2004-06 also indicate that the Wet Tropics catchments exported higher sediment loads per square kilometre catchment area.

The broad confidence limits for modelled estimates of nutrient export (Table 2.9) preclude drawing relevant conclusions regarding relationships between land use and nutrient exports at this time. The identification of nutrient and sediment loads from different land uses by sub-catchment modelling (see Brodie *et al.*, 2004) remains the current benchmark. These modelling results show that suspended sediment exports (not discharge-weighted) to the GBR coast are greatest from grazing areas in the Normanby, Barron, Herbert, Burdekin, Fitzroy and Burnett rivers, while exports from the N-Johnstone and Tully catchments are predominantly from forested areas and in the Pioneer catchment from sugar cultivation areas (ibid.). These land uses also cover the greatest area in the respective catchments.

To date there are only few discharge-weighed estimates of Reef catchment river nutrient and sediment exports available. Due to the high natural climate (rainfall) variability of the region, river mouth load-monitoring, combined with sub-catchment monitoring and modelling of loads, should be sustained in the long-term to produce useful regional baselines against which the success of the Reef Plan can be evaluated.

Recommendations for future river monitoring:

- Automated monitoring of suspended sediment loads over the wet season should be continues as this approach provides high-density data most suitable for making accurate estimates of sediment exports.
- Additional effort should be made to increase the number of manual nutrient and sediment samples collected from Reef Plan MMP priority rivers. The accuracy and usefulness of current estimates is constrained by the low number of samples taken.
- Additional effort should be taken to better understand the relationships between dissolved nutrient loads and suspended sediment load in Reef Plan MMP priority rivers so that more readily measured sediment loads can be better used as a proxy for total nutrient exports from catchments. Changes in these relationships for individual rivers will need to be verified from time to time.
- The number of monitoring organisations, particularly with regard to manual sample collections, should be streamlined and minimized. Over the 2004-05 and 2005-06 wet seasons, data collection was hampered by confusion, particularly among community-

based groups by the number of separate monitoring efforts (4) and different methods used.

- Flood monitoring is unlikely to be widely supported by community samplers over a longer term, mainly because of the effort to accomplish daily sampling and OH&S issues associated with accessing flooding rivers. Alternative arrangements should be developed to achieve high frequency flood sampling.

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3. River Pesticide Loads and GBR Lagoon Pesticide Data

(Attachment A Task 2.6)

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Introduction

Cattle grazing and cropping (in particular sugarcane) account for significant land use in the Wet Tropics (Haynes, 2001). Pesticides commonly used in these industries include organophosphates (e.g. chlorpyrifos) and triazines (e.g. atrazine, simazine, ametryn, prometryn) as well as urea-based herbicides (e.g. diuron, tebuthiuron, fluometuron). Depending on the physical properties of these pesticides, their mobility varies, but those that are persistent have the potential to be transported from the sites of application in a catchment via rivers into the marine environment.

Anthropogenic pollutants such as pesticides and antifoulants have been detected in the Great Barrier Reef environment since the 1970s (Olafson, 1978). The effects from introducing land-based pollutants into the Great Barrier Reef are not well understood, however the potential for certain pollutants to impact on ecological processes and the health of reef ecosystems has been recognised (e.g. Brodie *et al*., 2001; Haynes, 2001; Bengtson-Nash *et al*., 2005).

Data on the concentrations of organic pollutants in rivers draining into the Great Barrier Reef have been gathered through short-term sampling efforts employing a range of sampling strategies which are unsuitable for estimating input loads. In addition analysis of biota or sediments have been used to assess exposure to contaminants in the ecosystem (von Westernhagen and Klumpp, 1995; Russell and Hales, 1993; Smith *et al*., 1985; Haynes *et al*., 2000; Müller *et al*., 2000; Bengtson-Nash *et al*., 2005). Overall, there is good evidence that land-sourced pollutants are entering waters of the Great Barrier Reef, but concentrations of pollutants are low, particularly in the offshore environment. Due to the sensitive nature and high conservation value of the Great Barrier Reef, concern remains for the potential consequences of continuous low exposure to these pollutants. This has been highlighted with the development of the Reef Plan, which aims to address long-term changes to pollutant concentrations and their effects on the Great Barrier Reef. To help achieve this aim, there is a need to monitor the concentrations of pollutants in Great Barrier Reef catchment waterways and in Great Barrier Reef inshore waters.

To assess whether environmental management practices are working, long term monitoring must be capable of detecting changes in water chemistry (Haynes, 2001). Further, it is important to be able to monitor pollutants at levels well below those which may have some impact on ecosystem health. Therefore, monitoring tools which are reproducible and highly sensitive are essential. These tools should be simple to use and produce data easy to interpret, incorporating sampling methods that are both cost and time effective. Many of the traditional sampling methods for trace pollutants are not reliable for monitoring long term trends. Typically, individual “grab” water samples are difficult to interpret if the variability of pollutants on a temporal scale is not known. Furthermore the method is insensitive and care is required to avoid degradation of chemicals between sampling and analysis. Analysis of biota or sediments may be more sensitive for persistent lipophilic pollutants; however,

interpretation of the results has remained a challenge. As a result, in the last decade(s) time-integrated passive sampling tools have become a practical tool for cost-effective time-integrated monitoring of pollutants (Huckins *et al.*, 1993). Samplers such as Semipermeable Membrane Devices (SPMDs) and Empore Disks (EDs) extract pollutants equivalent to that dissolved in several litres of water every day they are exposed. These techniques improve the feasibility of monitoring through increased sensitivity and reproducibility. Over the last decade, the University of Queensland's National Research Centre for Environmental Toxicology (EnTox) research team has developed, calibrated and evaluated a range of passive samplers for both polar and non-polar organic contaminants. This expertise has been utilised in the monitoring component of the Reef Plan.The Reef Plan MMP River Mouth Monitoring task will provide the primary indicator of the delivery of pollutants to the Great Barrier Reef and will assess, over time, trends in concentrations and loads of nutrients, sediments and pollutants that have the potential to adversely affect Great Barrier Reef ecosystems. The scientific aims of the River Pesticide Monitoring and Inshore Reef Pesticide Monitoring tasks are:

- To detect long-term trends in concentrations and loads of anthropogenic pollutants in river mouths and at inshore Reef sites of the Great Barrier Reef and
- To assist assessments of the effectiveness of measures under the Reef Plan to reduce the delivery of these pollutants.

In addition, by working closely with and involving community partners in the monitoring tasks it is hoped to promote broad acceptance and ownership of the Reef Plan.

The monitoring tasks in the Reef Plan MMP have primarily focused on the evaluation of organic pollutants using time integrated passive sampling techniques including ED and SPMD passive samplers as monitoring tools. Some grab samples and marine sediment samples have also been collected to provide an additional validation tool for the comparability of passive sampling tools with traditional water sampling techniques and to undertake preliminary load calculations during flood events. Sediment samples can also provide supporting information on contaminants that are bound to particulates and not sampled in the water column by the passive samples.

Passive sampling results are available from 8 inshore reef sites and 9 river mouth sites. In total 39 polar and 32 non-polar successful sampling periods for passive samplers at inshore reefs and 35 polar and 21 non-polar successful sampling periods for passive samplers at river mouth sides were undertaken. Polar samplers were analysed for a suite of 10 herbicides including 2 degradation products whereas non-polar samplers were analysed for more than 50 pesticides including degradation products (this number only includes pesticides that are known to accumulate in SPMDs and will not be eliminated as a result of the sample purification).

Methods

Logistics of the study

A key component of this first period of the Reef Plan MMP was organisation and logistics of the monitoring activities. From the onset of the program the study incorporated the participation of community groups and tourist operators. The involvement of these groups and their commitment was arranged through GBRMPA and/or AIMS (see Tables 3.1 and 3.2 for details about the groups assisting with sampler deployment). The start-up phase of the

study also provided a series of unexpected challenges including insurance of field personnel and the need to establish a system to mitigate the relatively high shipping costs of monthly samplers. Such challenges were not always well understood or their complexity was simply underestimated, leading to delays in the establishment of some sampling sites.

Table 3.1. Passive sampler monitoring at GBR inshore reef sites.

Site	Maintained by	GPS	Sampling details
Low Isles	Quicksilver Connections, Low Isles (Caretaker)	16°22.910'S 145°33.720'E	• Commenced deployment 23 June 2005
Normanby Island	Frankland Island Cruise & Dive	17°12.267'S 146°04.465'E	• Commenced deployment 24 June 2005; monthly changeover.
Fitzroy Island	Raging Thunder, Fitzroy Island Resort	Not available	• Commenced deployment 22 June 2005
Bedarra Island	Bedarra Resort	18°00.083'S 146°08.500'E	• Commenced deployment 13 July 2005*
Orpheus Island	Orpheus Island Resort	Not available	• Commenced deployment 12 July 2005
Magnetic Island	GBRMPA	19°10.831'S 146°50.304'E	• Commenced deployment 3 July 2005
Long Island	Long Island Dive and Snorkel	20°19.742'S 148°50.735'E	• Commenced deployment 14 July 2005*
North Keppel Island	North Keppel Island. Education Centre	23°05.197'S 150°53.336'E	• Commenced deployment 4 August 2005

* Organisation no longer able to assist with sampling

Table 3.2. Passive sampler monitoring at river mouth sites.

Site	Maintained by	Progress
Barron River	FNQNRM	• Commenced deployment 17 November 2005
Russell River*	GBRMPA	• Commenced deployment 25 June 2005
Mulgrave River*	GBRMPA	• Commenced deployment 25 June 2005
North Johnstone River	Far North Queensland Natural Resource Management Pty Ltd (FNQNRM)	• Commenced deployment 15 November 2005
Herbert River	FNQNRM	• Commenced deployment 02 December 2005
Burdekin River	AIMS	• Commenced deployment April 2006; deployments limited due to low flow.
O'Connell River	Whitsunday Catchment Landcare	• Commenced deployment October 2005, discontinued in December 2005 due to safety reasons.
Pioneer River	Mackay Whitsunday Healthy Waterways (MWHW)	• Commenced deployment 26 October 2005
Fitzroy River	Fitzroy Water	• Commenced deployment 28 November 2005
Burnett River	Bundaberg City Council	• Commenced deployment 3 November 2005

Note: No suitable deployment sites were found for the proposed Normanby or Tully Rivers. * Russell and Mulgrave Rivers were established by Melanie Shaw as part of her PhD.

Once the sampling sites and volunteer organisations were determined and agreed upon, each marine site was visited by EnTox and GBRMPA representatives to provide training in the use of passive sampling techniques and to ensure that the Passive Sampling Guidelines provided to the collaborating organisations were understood.

After the initial set up of each site, it was intended that samplers would be exchanged monthly. The success of this process varied depending on the nature of the sampling site and the commitment of partner organisations. Specific reasons for gaps in the data during the first year include, but are not limited to:

- Loss of samplers due to theft, vandalism, extreme weather or reasons that are unclear;
- Difficulties with deploying or retrieving samplers due to work commitments of volunteers and weather issues including TC Larry;
- Deployment staff on leave or not contactable for long periods without notice (major issue over December/January period);
- Loss of collaborating organisation or individual staff in a collaborating organisation without or with little notice; and
- Difficulties re-establishing contact after problems have occurred due to any of the above problems.

In spite of these challenges, this study has been successful in establishing sites, providing training in sampler handling and, at many sites, in maintaining an efficient sampling program.

Sample and Site Details

Passive samplers were deployed by tourism operators, community groups and agencies at a total of nine inshore reef sites and eleven river mouths between June 2005 and June 2006. The sites and the number of samples deployed and analysed up until April 2006 are identified in Tables 3.3 and 3.4. Two EDs and two SPMDs were exposed at most sites for approximately 1 month at a time. These were then retrieved and new samplers deployed. Both ED samplers were extracted and one of these was analysed; the second extract was stored at EnTox. Similarly, one SPMD sample was extracted and analysed for each deployment at river mouth sites, while the second was stored at EnTox. SPMDs deployed at inshore reef sites were both extracted and combined to improve detection limits. Replicates of both EDs and SPMDs, initially stored at EnTox, were later analysed to ensure reproducibility of results. Reproducibility data can be found in the QA/QC section in Appendix 2. For more comprehensive details on site status and problems regarding passive sampler deployments see the *Pictorial Deployments Chart* in Appendix 1. This table also shows the commencement of sampling and a timeline of sample change-overs and deployment lengths.

Sediment samples were collected from eight inshore reef sites (see Table 3.3). One sample from each site was analysed, with a replicate from Fitzroy Island also analysed for reproducibility. See the QA/QC information in Appendix 2 for details on the results of the replicates.

Grab water samples were collected at 7 river mouth sites (see Table 3.4). Additionally, samples were collected following TC Larry along transects from the mouth of the Tully River to Dunk Island, as well as from the Johnstone River mouth to Russell Island. All samples collected were extracted and analysed.

Table 3.3. Summary of sampling at the inshore reef sites for deployments from June 2005 to April 2006. The 'Sent' column details the number of sets of samplers that were sent to the site (a set comprises two EDs and two SPMDs). The 'Results' column details the number of sets of samplers returned to EnTox where results were obtained. The sediment sample column lists the number of samples analysed at each site.

NRM Region	Inshore Reef	ED		SPMD		Sediment Samples
		Sent	Results	Sent	Results	
Wet Tropics	Low Isles	9	7	9	5	1
	Fitzroy Island.	10	9	10	7	2
	Normanby Island.	8	2	8	2	1
	High Island.					
	Bedarra Island.	7	4	7	2	1
	Orpheus Island.	9	7	9	5	1
Burdekin	Magnetic Island.	8	3	8	3	1
Mackay Whitsunday	Long Island.	6	3	6	5	1
Fitzroy	North Keppel Island.	8	4	8	3	1
Total		*65*	*39*	*65*	*32*	*9*

Table 3.4. Summary of sampling at the river mouth sites for deployments from June 2005 to April 2006. The 'Sent' column details the number of sets of samplers that were sent to the site, (a set comprises two EDs and two SPMDs). The 'Results' column details the number of sets of samplers returned to EnTox where results were obtained. The grab sample column lists the number of samples analysed at each site.

NRM Region	River Mouth	ED		SPMD		Grab Samples
		Sent	Results	Sent	Results	
Cape York	Normanby R.					6
Wet Tropics	Barron R.	5	1	5	2	7
	Russell R.	7	7	7	4	
	Mulgrave R.	7	7	7	3	
	Nth Johnstone R.	5	3	5	2	6
	Tully R.					2
	Herbert R.	4	4	4	3	
Burdekin	Burdekin R.					3
Mackay Whitsunday	O'Connell R.	1	1	1	1	1
	Pioneer R.	6	6	6	3	1
Fitzroy	Fitzroy R.	4	1	4	1	
Burnett Mary	Burnett R.	6	5	6	2	
Total		*45*	*35*	*45*	*21*	*26*

Semipermeable Membrane Devices (SPMDs)

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

The methodology used for SPMD preparation, deployment and analysis was based on United States Geological Survey protocols (Huckins *et al*., 2000). Procedural, fabrication and field blanks were analysed with the samples to determine background levels of contamination

associated with preparation, storage and transport to and from the field. Details on the results of these blanks can be found in Appendix 2. SPMDs were prepared in the laboratory from pre-extracted Low Density Polyethylene and 99% triolein. Performance reference compounds (PRCs) were spiked into the triolein and their relative recovery was to provide a means for an in-situ adjustment of the uptake of target chemicals into the samplers. The samplers were mounted into stainless steel sampling devices prior to shipment. Two SPMDs were deployed in each sampling device with one device deployed per site. Samplers were transported in sealed tin cans and refrigerated. Following retrieval, samplers were returned to the can and kept cold during storage and transport.

In the EnTox laboratory, the surfaces of the SPMDs were scrubbed with water, dipped in hexane and 1 M hydrochloric acid, and rinsed briefly with acetone and isopropanol prior to undergoing accelerated solvent extraction using hexane with 10% acetone. The extracts were reduced in volume, transferred into DCM and subjected to size exclusion chromatography using an automated Gel Permeation Chromatography (GPC) Environgel with dichloromethane as the mobile phase. The samples were then reduced in volume again and transferred to Queensland Health and Scientific Services (QHSS) for final analysis. At QHSS the purified extracts from the SPMDs were analysed on a gas chromatograph that is coupled to a low resolution mass spectrometer using selective ion monitoring for 56 pesticides, table A2-3.10 in Appendix 2 has a complete list of pesticides that were targeted for the analysis. This table only shows pesticides that have sufficient partition coefficients to accumulate in SPMDs effectively and are not potentially removed during clean-up.

It should be noted that the uptake of chemicals into the sampler is expected to be primarily via the dissolved phase and thus water concentration (C_w) may be underestimated for extremely hydrophobic chemicals. Furthermore the assumption is made that chemicals (including the PRCs) are not degraded in the passive samplers. However for SPMDs deployed in shallow and very clean water degradation may well be an issue for some compounds such as PAHs. Work is underway to address this issue by modifying the technique to allow correction with the use of photodegradation PRCs that are spiked into the samplers as well as providing additional protection of these from sunlight.

Concentrations of contaminants sequestered in SPMDs (C_{SPMD}) were converted to C_W using a sampling rate (R_S in Ld^{-1}), determined by laboratory calibrations. For chemicals that were expected to be in a linear uptake phase, the concentration of the chemicals detected in the C_{SPMD} was used to predict the time averaged concentration of the respective chemical in the water using the following equation (Huckins *et al.*, 2000):

$$C_W = \frac{C_{SPMD} \times M_{SPMD}}{R_S \times t}$$

where: M_{SPMD} is the mass of the SPMD in grams, and
t is the sampling time in days.

Ideally, the laboratory R_S for each compound was to be adjusted for actual field conditions using the PRCs by calculating an environmental adjustment factor (EAF) (Huckins *et al.*, 2000):

$$EAF = k_{e-PRC} / k_{e-cal}$$

where: $k_{e\text{-}cal}$=elimination rate calculated in laboratory flow-through experiments
$k_{e\text{-}PRC}=\ln(C_{SPMD\text{-}0}/C_{SPMD})/t$.

where: $C_{SPMD\text{-}0}$ is the initial PRC concentration in the SPMD and
C_{SPMD} is the final PRC concentration in the SPMD.

It should be noted that the uptake of chemicals into the sampler is expected to be primarily via the dissolved phase and thus the total mass of chemical may be underestimated for extremely hydrophobic chemicals. Furthermore the assumption is made that chemicals (including the PRCs) are not degraded in the passive samplers. However for SPMDs deployed in shallow and very clean water degradation may well be an issue for some compounds such as PAHs and thus we are considering inclusion of photodegradation PRCs into the samplers as well as using additional protection of these from sunlight.

Empore Disk (ED) samplers

The polar samplers that were deployed are SDB-RPS 3M Empore^{TM} Extraction Disks. These were deployed in teflon manifolds designed by Kingston *et al*. (2000). For longer sampling periods (i.e. in excess of 1-2 weeks) requiring time-averaged sampling, the uptake needs to be controlled through a membrane that allows diffusion of polar chemicals (Stevens *et al.* 2005). In this study EDs were prepared by conditioning in methanol followed by MilliQ water. PRCs were added to the disk by slowly filtering water enriched with the PRC through each disk. After the disks were loaded into the teflon device, a diffusion limiting membrane was placed on the disk and MilliQ Water added to the device before it was sealed. All samplers were refrigerated and kept cold during transport to the site. Two EDs were deployed at each site, attached to the base of the SPMD deployment cage. When deployed, the lid of the teflon case was replaced with an open teflon ring to expose the sampler but hold the membrane in place. Upon retrieval the teflon ring was removed and the sampler filled with water and sealed with the lid. The samples were then kept cold and returned to EnTox.

Prior to extraction, polar samplers were spiked with deuterated simazine as a surrogate standard. The samplers were extracted in an ultrasonic bath or in an ED manifold using acetone and then methanol. The extracts were combined and reduced in volume before being filtered through a PTFE 45μm syringe driven filter unit. The extract was spiked with deuterated atrazine as a recovery standard, and the extracts transferred to QHSS for analysis by liquid chromatography-mass spectrometry (triple quadruple MS) for eight priority herbicides; diuron, atrazine, simazine, tebuthiuron, fluometuron, hexazinone, ametryn and prometryn. In addition, ED sampler extracts were analysed for two degradation products of atrazine and simazine; desethyl atrazine and desisopropyl atrazine. Polar sampler concentrations were converted into estimates of water concentrations using sampling rate (Ld^{-1}) obtained from laboratory studies (Stephens *et al*., 2005; Stephens *et al.* unpublished data, Shaw *et al.* unpublished data). Table A2-3.10 in Appenidx 2 lists the set of herbicides that samples were analysed for.

Grab samples for polar organic chemicals

Grab samples were collected directly from sites in solvent-washed 1L glass amber bottles. The samples were stored at $< 4^{o}C$ and transported cold to EnTox. These samples were collected to provide relevant information on the variability of pollutant concentrations in a system as well as to assist in the assessment of the accuracy of predicted pollutant concentrations from passive sampling data.

The water samples, including 1L blank samples consisting of ultra clean water, were extracted using a vacuum manifold through an Oasis HLB 12cc 500mg LP Solid Phase Extraction (SPE) cartridge. Prior to extraction the samples were spiked with deuterated simazine as a surrogate standard. The SPE cartridge was conditioned with HPLC grade methanol and ultra-pure water. The one litre sample was then extracted though the cartridge using a vacuum. The

cartridges were dried and then eluted with methanol. The eluate was reduced in volume and spiked with deuterated atrazine as a recovery standard before being analysed by QHSS LC-MS for the same ten herbicidesand degradation products as the ED sampler analysis. See table A2-3.11 in Appendix 2 for a complete list.

Sediment samples

Sediment samples were collected in proximity to each of the inshore reef passive sampler deployment sites using a sampling strategy developed for the National Dioxin Program (Müller *et al*. 2004). Ten sub-samples were collected from each sampling site over an area of ~ 1 km^2 using pre-cleaned aluminium tubes that are attached to a one-way valve. The samples were collected, stored and transported in the tubes to minimise contamination. In the laboratory the sediment was removed from the tubes and combined to form composite samples from each site prior to analysis at QHSS. For analysis, homogenised samples were transferred into stainless steel tubes for accelerated solvent extraction. The samples were spiked prior to extraction with a set of deuterated PAH surrogates.

Clean-up of the extracts included size exclusion chromatography using an automated Gel Permeation Chromatography (GPC) Environgel with dichloromethane as the mobile phase. The eluted sample was collected in two fractions. The first fraction was subjected to an additional adsorption chromatography clean-up step using florisil. The fractions were concentrated under a stream of nitrogen to final volumes of 1000 µL and 200 µL respectively. Samples were analysed by Gas Chromatography-Mass Spectroscopy using selective ion monitoring for 10 herbicides, 45 organophosphorus pesticides, 30 organochlorine pesticides and 17 PAHs. See table 3.7 and tables A2-3.8 and A2-3.9 in Appendix 2 for lists of the analytes and their LODs..

Data analysis and statistical methodology

The raw data for the ED and SPMD samples were used for the calculation of C_W (concentrations in water). Minima, maxima, averages and standard deviations during the wet season and dry season for each site were calculated and tabulated. Data was also graphed to facilitate comparison within and between NRM Regions.

Sediment data are presented on a dry weight basis.

Results

As discussed in the methodology section above, the establishment and maintenance of sampling sites through volunteer organisation proved challenging and was compounded by delayed establishment of sites, poor weather, and loss of samplers due to theft, vandalism and strong currents. For these reasons, there are many data gaps in this first sampling year.

Organic chemicals including pesticides were detected in a range of samples covering most sites, both in the Great Barrier Reef inshore lagoon and from river mouths during either or both the dry and wet season 2005-06.

Predicted concentrations of organic pesticides and degradation products in water from passive sampler deployments are summarised for the dry season (May to October 2005) and the wet season (November 2005 to April 2006) in Tables 3.5 and 3.6. Only data for compounds that were detected are shown. For a full list of the compounds for which the samples were analysed refer to Tables A2-3.10 and A2-3.11 in Appendix 2.

Please note that deployment month means the month the samplers were placed into the water. Samplers were generally exposed from mid-month to the next mid-month. For details on deployment dates see the raw data or the *Pictorial Deployments Chart* in Appendix 1.

Organic pollutants were detectable at relatively low concentrations at all inshore reef sites (Table 3.5). The key compounds detected at these sites were herbicides, in particular diuron, which was found at all sites except Normanby Island where deployments were not continued at the start of the wet season due to operational difficulties. Simazine, atrazine and hexazinone were the other pesticides that were detected routinely at selected sites (primarily Low Isles, Fitzroy Island and simazine at Normanby Island). The SPMD samplers covering more nonpolar pesticides were able to detect diazinon at a number of inshore reef sites as well as very low predicted levels of chlorpyrifos (~3 pg/L).

Inshore reef locations had mean diuron concentrations during the dry season of < 1 ng/L, except for Low Isles and Fitzroy Island. At Low Isles the concentrations were 1.3 ± 1.4 ng/L and at Fitzroy Island 2.5 ± 1.1 ng/L. Simazine was detectable in dry season samples at Fitzroy Island, Normanby Island and Bedarra Island as well as chlorpyrifos and diazinon (both detected in SPMDs) at Fitzroy Island in several samples. The predicted water concentrations of simazine and diazinon were on average in the sub-nanogram per litre range and several pg/L for chlorpyrifos.

Passive samplers were used for the evaluation of organic pollutants at river mouth sites and a range of organic pesticides were detectable at these sites (Table 3.6). For example, diuron concentrations at river mouth sites for the Pioneer River during one deployment period suggested a time-averaged concentration of about 1400 ng/L (1.4 ug/L) and even the minimum concentration of 160 ng/L can be considered high when compared to other sites in this study. In addition to diuron, other herbicides such as simazine, atrazine, hexazinone, ametryn and tebuthiuron were detected at various sites at different periods. Also, through the deployment of SPMDs, the sequestration of the nonpolar pesticides such as chlorpyrifos, diazinon, dieldrin and DDE allowed prediction of low levels of these chemicals in the water during various deployment periods.

Due to the difficulties in establishing and maintaining deployments at some sites, at this stage only a limited set of data are available that allow comparison between wet and dry season. Tables 3.5 and 3.6 and the *Pictorial Deployment Chart* in Appendix 1 indicate how many samplers were deployed, retrieved and analysed from each site. Evaluation of seasonal trends of pesticides is further complicated by a range of factors related to the use and retention of chemicals in a given catchment, the mobility of the chemicals as well as local sources near the sampling site (for example use of antifoulants) and finally the relatively vague categorisation of samples towards a given season (wet or dry). Figure 3.1 shows the average monthly rainfall from September 2005 to May 2006. The wettest months were clearly January-April 2006, with October-December 2005 showing a gradual increase in monthly rainfall. For detail on river flow see Chapter 2, Table 2.4 and Figure 2.8.

Table 3.5. Maximum, minimum, average and SD of estimated water concentrations (ng/L) from passive samplers deployed at Inshore Reef sites during the wet and dry seasons, June 2005-April 2006. Grey sections are results obtained from SPMDs, the remainder are from EDs.

Site		Diuron		Simazine		Atrazine		Hexazinone		Ametryn		Chlorpyrifos		Endosulfan		DDE pp	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Low Isles	Max	3.7	14	<1	<1	<1	5.6	<1	3.2	<1	<1	<0.03	<0.03	<0.6	<0.6*	<0.05	<0.05*
	Min	<1	1.4	<1	<1	<1	0.3	<1	0.5	<1	<1	<0.03		<0.6		<0.05	
	Mean	1.3	8.7	<1	<1	<1	2.8	<1	2.1	<1	<1	<0.03		<0.6		<0.05	
	SD	1.4	6.5				2.7		1.4								
Fitzroy Island.	Max	4	5.8	0.5	<1	<1	1.6	<1	1.6	<1	<1	<0.03	<0.03	<0.6	<0.6	<0.05	<0.05
	Min	0.4	0.9	<1	<1	<1	<1	<1	<1	<1	<1	<0.03	<0.03	<0.6	<0.6	<0.05	<0.05
	Mean	2.5	2.8	0.09	<1	<1	0.61	<1	0.62	<1	<1	<0.03	<0.03	<0.6	<0.6	<0.05	<0.05
	SD	1.1	2.1	0.18			0.69		0.62								
Normanby Island.	Max	5.0	#	1.8	#	<1	#	<1	#	<1	#	<0.03	#	<0.6	#	<0.05	#
	Min	<1		<1		<1		<1		<1		<0.03		<0.6		<0.05	
	Mean	1.7		0.92		<1		<1		<1		<0.03		<0.6		<0.05	
	SD	2.9		1.3													
Bedarra Island.	Max	0.9	#	0.5	#	<1	#	<1	#	<1	#	<0.03	#	<0.6	#	<0.05	#
	Min	<1		<1		<1		<1		<1		<0.03		<0.6		<0.05	
	Mean	0.39		<1		<1		<1		<1		<0.03		<0.6		<0.05	
	SD	0.46															
Orpheus Island.	Max	0.6	1.2	<1	<1	<1	<1	<1	<1	<1	<1	<0.03	<0.03*	<0.6	<0.6*	<0.05	<0.05*
	Min	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<0.03		<0.6		<0.05	
	Mean	0.14	0.58	<1	<1	<1	<1	<1	<1	<1	<1	<0.03		<0.6		<0.05	
	SD	0.28	0.58														
Magnetic Island.	Max	1.5	3.2*	<1	<1*	<1	4.5*	<1	1.6*	<1	<1*	<0.03	<0.03	<0.6	<0.6*	<0.05	<0.05*
	Min	<1		<1		<1		<1		<1		<0.03		<0.6		<0.05	
	Mean	0.75		<1		<1		<1		<1		<0.03		<0.6		<0.05	
	SD	1.1	0				0		0								
Long	Max	1.5	#	<1	#	<1	#	<1	#	<1	#	<0.03	<0.03	<0.6	<0.6	<0.05	<0.05

	Min	<1		<1		<1		<1		<1		<0.03	<0.03	<0.6	<0.6	<0.05	<0.05
	Mean	0.80		<1		<1		<1		<1		<0.03	<0.03	<0.6	<0.6	<0.05	<0.05
	SD	0.77															
North Keppel Island.	Max	<1	2.4	<1	<1	<1	<1	<1	<1	<1	<1	<0.03	<0.03*	<0.6	<0.6*	<0.05	<0.05*
	Min	<1	1.4	<1	<1	<1	<1	<1	<1	<1	<1	<0.03		<0.6		<0.05	
	Mean	<1	1.8	<1	<1	<1	<1	<1	<1	<1	<1	<0.03		<0.6		<0.05	
	SD		0.549104														
* results based on one sample; # No deployments were analysed in this season																	
Detected pesticides that did not pass quantification criteria (ie. 3 x baseline and blank concentration) are presented as a Limit of Detection (LOD) in bold.																	

Table 3.6. Maximum, minimum, average and SD of estimated water concentrations (ng/L) from passive samplers deployed at River Mouth sites during the wet and dry seasons, June 2005-April 2006. Grey sections are results obtained from SPMDs, the remainder are from EDs.

Site		Diuron		Simazine		Atrazine		Hexazinone		Ametryn		Tebuthiuron		Chlorpyrifos		Endosulfan		DDE pp	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Barron R.	Max	#	1.9*	#	8*	#	3.2*	#	1.0*	#	<1*	#	<1*	#	**<0.03**	#	<1	#	<0.05
	Min														**<0.03**		<1		<0.05
	Mean														**<0.03**		<1		<0.05
	SD																		
Russell R.	Max	78	430	<1	<1	29	150	52	77	1.4	<1	1.2	<1	**0.2**	#	<2	#	<0.05	#
	Min	15	260	<1	<1	8.2	37	14	65	0.0	<1	0.0	<1	<0.03		<2		<0.05	
	Mean	35	340	<1	<1	20	94	25	71	0.3	<1	0.2	<1	0.10		<2		<0.05	
	SD	26	120			7.6	81	16	8.7	0.6		0.5							
Mulgrave R.	Max	37	320	1.3	3.8	36	78	20	160	<1	5.7	<1	<1	**<0.03**	#	<2	#	<0.05	#
	Min	8.3	50	0.0	0	5.5	3.1	3.6	22	<1	0.0	<1	<1	**<0.03**		<2		<0.05	
	Mean	19	190	0.2	1.9	17	41	9.2	90	<1	2.8	<1	<1	**<0.03**		<2		<0.05	
	SD	10	190	0.5	2.7	11	53	5.5	95		4.0								
Nth Johnstone R.	Max	#	28	#	2.4	#	9.7	#	2.3	#	<1	#	6.4	#	**<0.03**	#	<2	#	<0.05
	Min		1.6		<1		<1		<1		<1		<1		**<0.03**		<2		<0.05
	Mean		9.0		0.80		3.2		0.84		<1		3.5		**<0.03**		<2		<0.05
	SD		12		1.0		3.9		0.85				2.0						
Herbert R.	Max	#	300	#	<1	#	140	#	80	#	<1	#	0.6	#	<0.03	#	<2	#	<0.05
	Min		4.6		<1		6		1.7		<1		<1		<0.03		<2		<0.05
	Mean		170		<1		79		48		<1		0.1		<0.03		<2		<0.05
	SD		140				62		39				0.27						
O'Connell R.	Max	130*	#	<1*	#	93*	#	110*	#	<1*	#	44*	#	<0.1*	#	<2*	#	<0.05*	#
	Min																		
	Mean																		
	SD																		

Site		Diuron		Simazine		Atrazine		Hexazinone		Ametryn		Tebuthiuron		Chlorpyrifos		Endosulfan		DDE pp	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Pioneer R.	Max	170*	1400	<1*	4.1	180*	1500	150*	520	5.2*	41	11*	6.1	<0.1*	**<0.03**	<2*	<2	<0.05*	<0.05
	Min		160		<1		95		140		5		<1		**<0.03**		<2		<0.05
	Mean		570		1.6		640		239		25		2.5		**<0.03**		<2		<0.05
	SD		470		1.7		514		135		13		2.7						
Fitzroy R.	Max	#	31*	#	11*	#	250*	#	14*	#	<1*	#	550*	#	<0.03*	#	<2	#	<0.05*
	Min																		
	Mean																		
	SD																		
Burnett R.	Max	2.2*	13	<1*	0.8	18*	31	5.4*	5.8	0.7*	1.7	2.3*	2.7	0.02*	<0.03*	<2*	<2*	<0.05*	<0.05*
	Min		4.4		<1		18		4.5		<1		1.9						
	Mean		10		0.16		24		5.3		0.99		2.3						
	SD		3.2		0.36		5.4		0.57		0.73		0.31						

* results based on one sample; # No deployments were analysed in this season

Detected pesticides that did not pass quantification criteria (ie. 3 x baseline and blank concentration) are presented as a LOD in bold

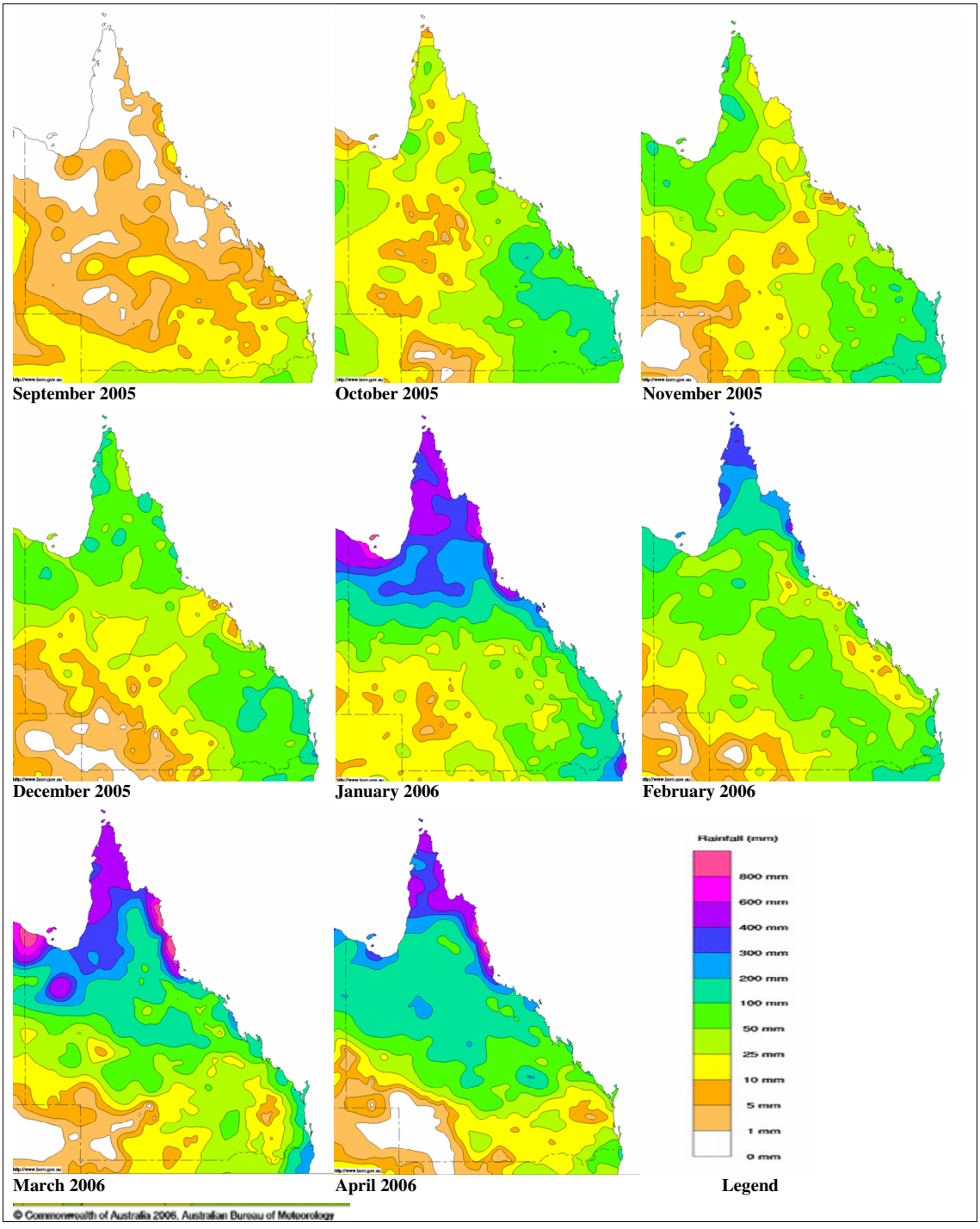

Figure 3.1.Monthly rainfall totals for Queensland, September 2005 – April 2006. Sourced from the Australian Government Bureau of Meteorology website, on 23 Oct 2006, http://www.bom.gov.au/climate.

Inshore Reefs

In summary, there is a consistent trend toward higher concentration of herbicides in the wet season at inshore reef sites (see Figures 3.2 and 3.3). The data indicate an increase in the total detected pesticide concentrations during the wet season compared with the dry at some sites. This is further highlighted by the fact that at a range of inshore reef sites, atrazine, hexazinone and diazinon were only detectable during the wet season sampling period (Low Isles, Fitzroy Island and Magnetic Island). Simazine was only detected in the dry season (Fitzroy Island and Normanby Island).

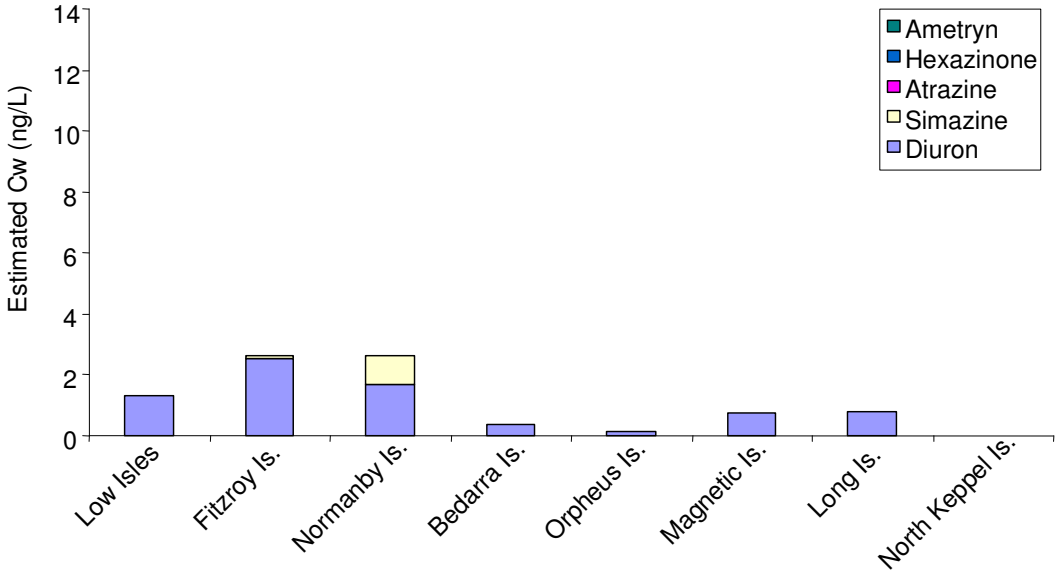

Figure 3.2. Average predicted concentrations of pesticides in water from passive samplers deployed at inshore reefs during the dry season, June 2005-October 2005.

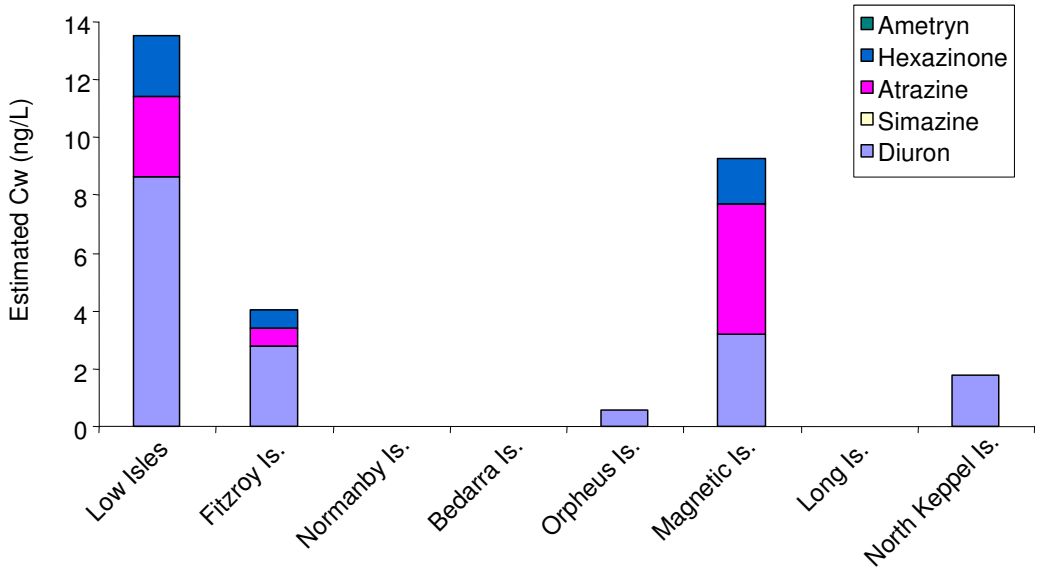

Figure 3.3. Average predicted pesticide concentrations in water from passive samplers collected at inshore reefs during the wet season, November 2005-April 2006.

Low Isles and Fitzroy Island are the inshore reef sites for which suitable data for seasonal comparison is available (see Figure 3.4). The data for Low Isles show a higher concentration of diuron during the peak wet season period (February-March deployment) compared to the earlier dry season sampling, and also the final sampling period, deployed in late March. This was also the only period when atrazine and hexazinone were detected.

At Fitzroy Island, the concentration of diuron was similar in both the wet and dry season. However the wet season was the only period where hexazinone and atrazine were detectable.

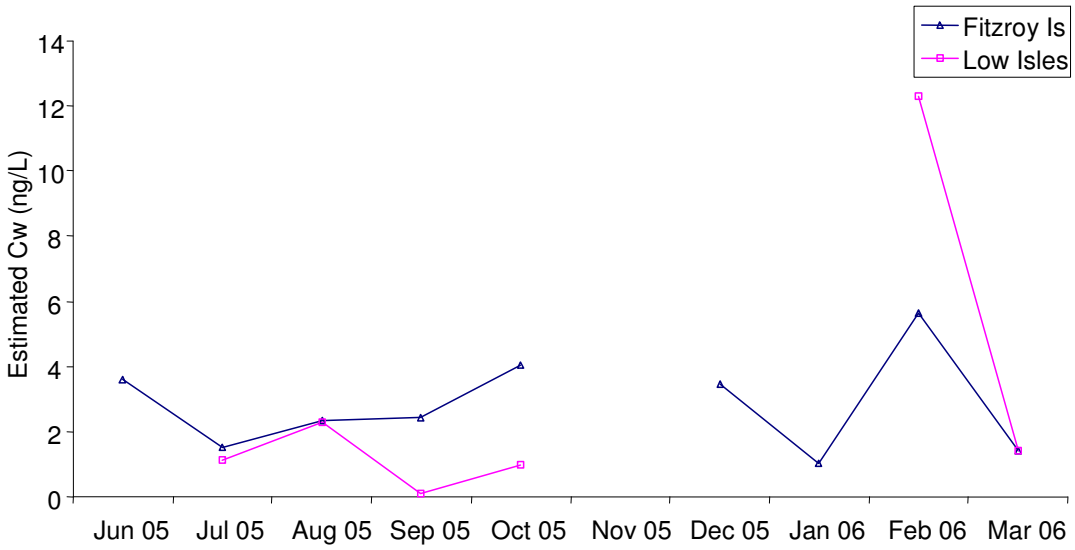

Figure 3.4. Predicted concentration of diuron at Fitzroy Island. and Low Isles based on passive sampling data for the period June 2005-March 2006. The SD is calculated from n=2 on deployments where 2 samplers were analysed.

The overall concentrations at Orpheus Island were consistently below the Limit of Detection (LOD) for all the organic pollutants considered except for diuron which was detectable at very low concentrations in one deployment from the dry season (C_w=0.6 ng/L in October) and two from the wet season (C_w=0.6 ng/L in November and 1.2 ng/L in February).

At Magnetic Island the highest diuron concentration was observed in the only successful deployment undertaken in the wet season (Cw=3.2 ng/L in December). Additionally, atrazine and diazinon were detected in the wet season samplers.

Long Island did not have ED results for the wet season; however diazinon was detected in the SPMDs for the wet season, but not during the dry season.

For Normanby and Bedarra Islands no results were obtained for the wet season, thus no comparisons can be made.

River Mouths

Due to delays and difficulties with site set-up for this section, river mouth monitoring commenced only late in 2005. The Barron, North Johnstone, Herbert and Fitzroy Rivers do not have dry season results (see Table 3.6 for details). O'Connell, Pioneer and Burnett Rivers provided only one set of samples. The O'Connell River has no wet season deployments, while the Burnett and Barron Rivers had only a single successful deployment. Consequently, seasonal comparisons are limited.

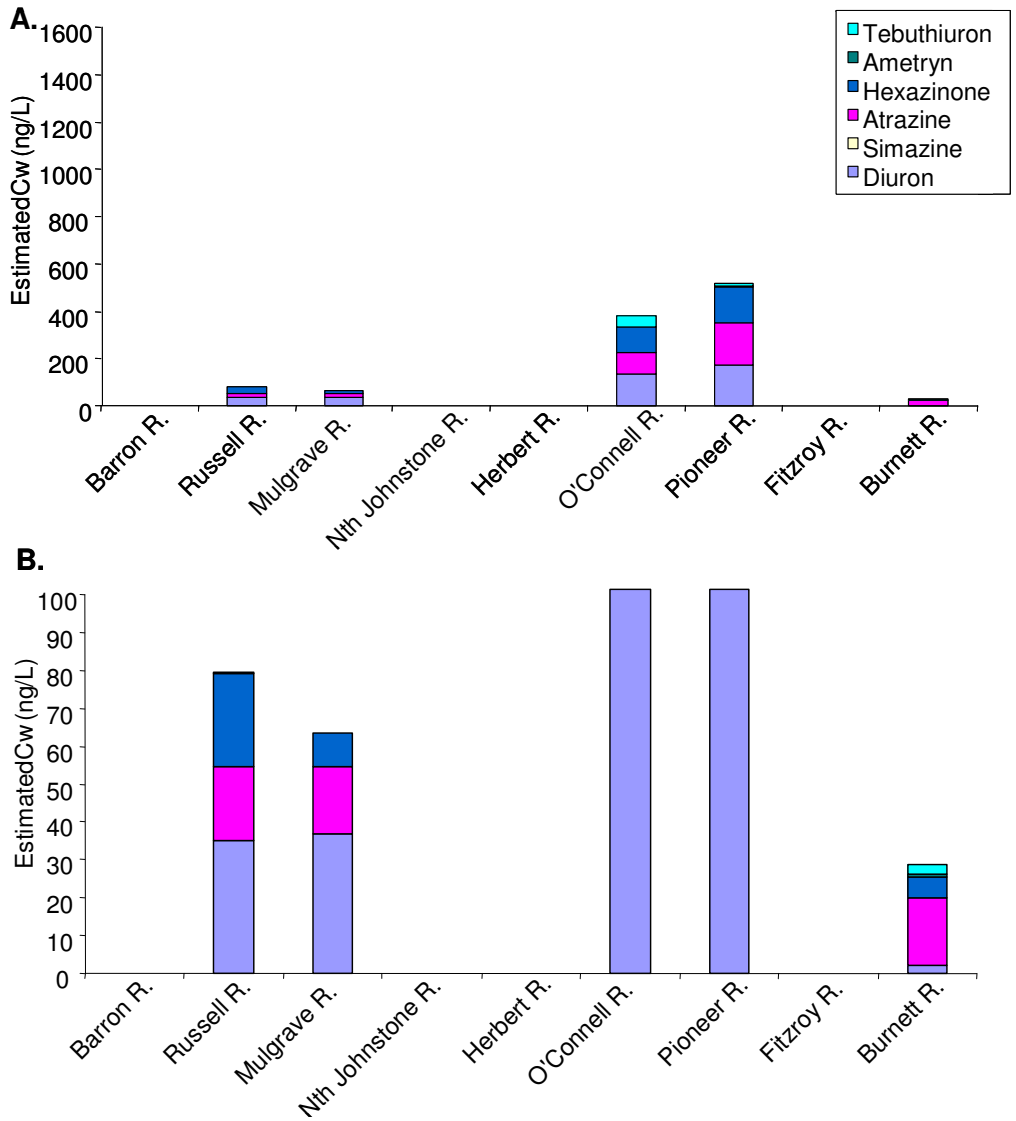

Figure 3.5. Averaged concentrations of pesticides at river mouth sites predicted from passive samplers during the dry season, June 2005-October 2005. a) Shows the compounds for all sites, b) the scale has been modified to allow visualisation of data in systems with lower observed pesticide concentrations.

Predicted concentrations of diuron, atrazine and hexazinone were greater in the wet season than the dry season (see Figures 3.5 and 3.6). Diuron was found at all sites for each season and was also the compound found at highest concentrations. The data indicate an increase in the total detected pesticide concentrations during the wet season compared with the dry season at sites where suitable data was obtained – the Burnett, Pioneer, Russell and Mulgrave Rivers.

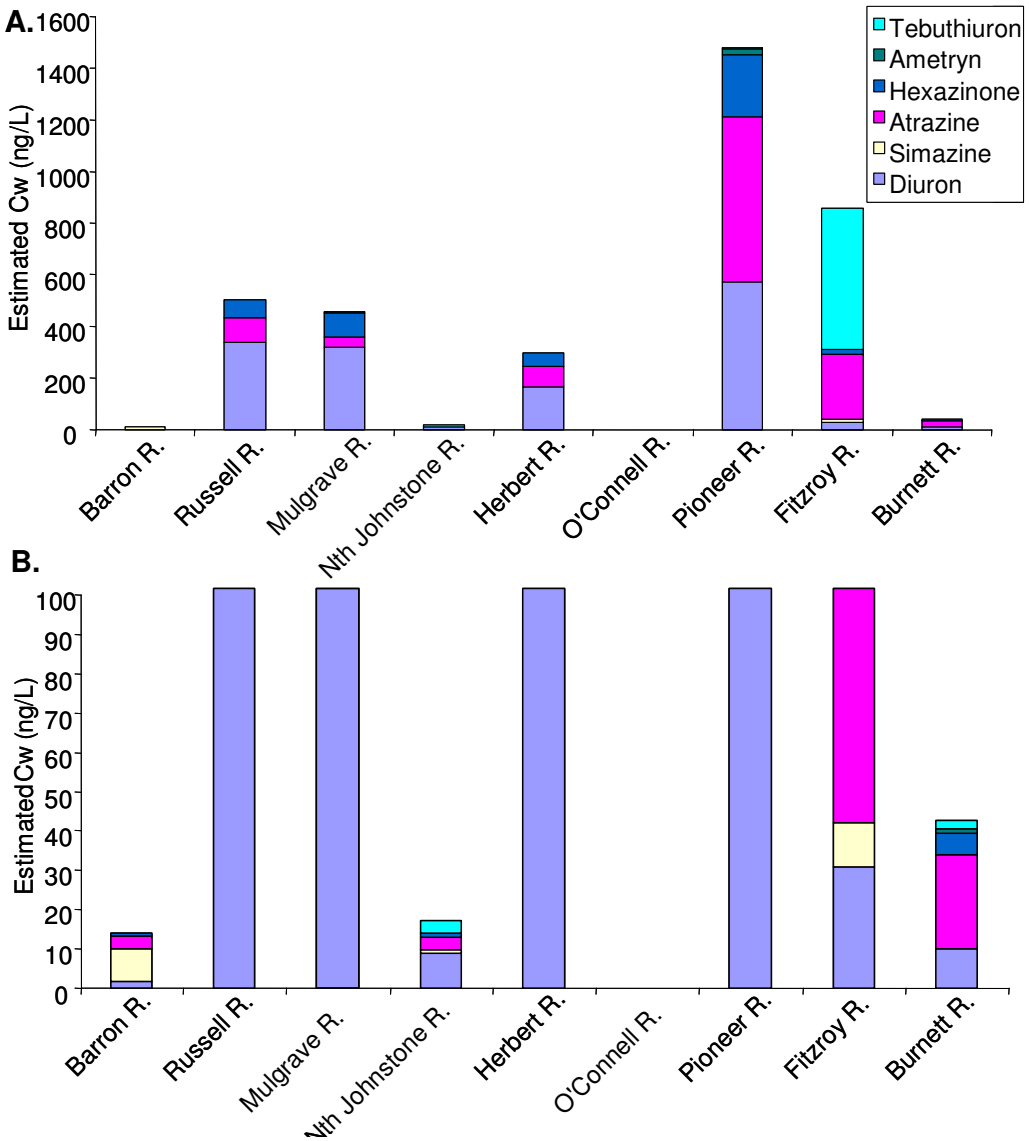

Figure 3.6. Averaged concentrations of pesticides at river mouth sites predicted from passive samplers during the wet season, November 2005-April 2005. a) Shows all compounds for all sites, b) the scale has been modified to allow visualisation of data in systems with lower observed pesticide concentrations. Note that no samples were collected from the O'Connell River.

In the Burnett River, diuron levels were much higher in December to March (C_w ranged from 11-12 ng/L) compared with October and November (C_w ranged from 2.2-4.4 ng/L). Atrazine and ametryn concentrations were also higher in the wet season deployments. Simazine was detected in the wet season but was not detectable in the dry season.

In the Pioneer River, sampling commenced in October 2005, thus only one dry season deployment occurred. Atrazine and diuron were the most dominant compounds in both their occurrence and predicted concentrations, followed by hexazinone. Concentrations of these three compounds increase from November with a peak in February-March, followed by a decline towards the onset of the dry season (see Figure 3.7a). Ametryn gradually increased in concentration throughout the entire sampling period. In contrast to other herbicides tebuthiuron appeared to decline throughout the wet season (see Figure 3.7b). Simazine, chlorpyrifos and diazinon were detected in the wet season but not in the dry.

Results from the Russell and Mulgrave Rivers followed a similar seasonal and compound profile. In both rivers diuron, hexazinone and atrazine were the key compounds detectable (Figures 3.8 and 3.9). A marked increase in diuron and atrazine levels was observed at both sites from the October-November deployment onwards, peaking in the December-January deployment. One of the differences between the two sites was a more pronounced increase of hexazinone towards the wet season in the Mulgrave River. Also Ametryn was detected only in the dry season in the Russell R (C_w=1.4 ng/L), and conversely only detected in the wet season in the Mulgrave R (C_w=5.69 ng/L). In the Russell River tebuthiuron was detected once but simazine not at all, whereas in the Mulgrave R, simazine was detected twice, but tebuthiuron was not present. The Russell R was the only site to have a confirmed detection of chlopyrifos (C_W=0.2 ng/L), which occurred during dry season sampling.

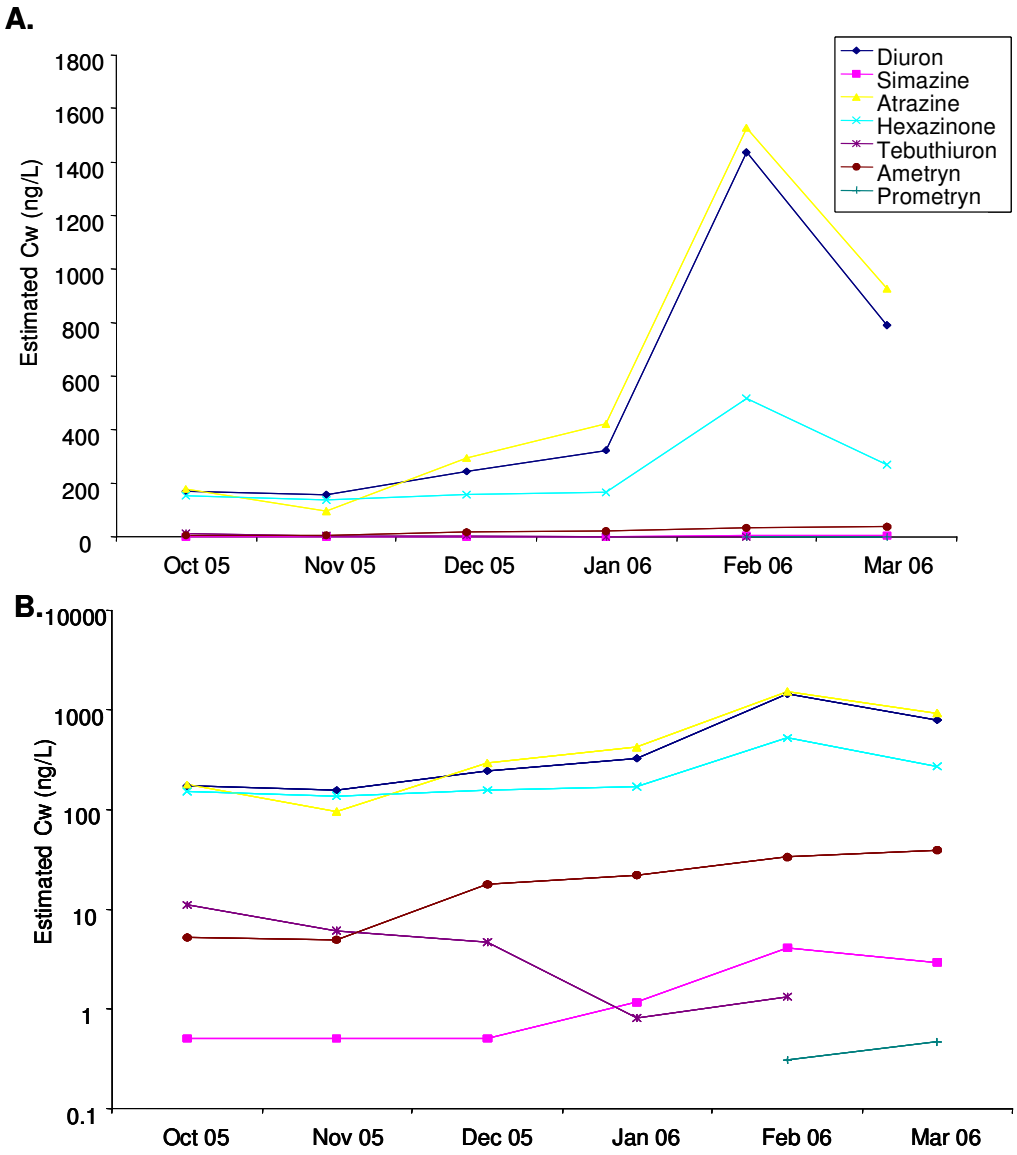

Figure 3.7. a) Predicted concentration of herbicides in Pioneer R based on passive sampling data for the period October 2005-March 2006. The error bars represent standard deviations calculated from n=2 on deployments where 2 samplers were analysed. b) Same data presented on a logarithmic scale to allow visualisation of lower observed pesticide concentrations

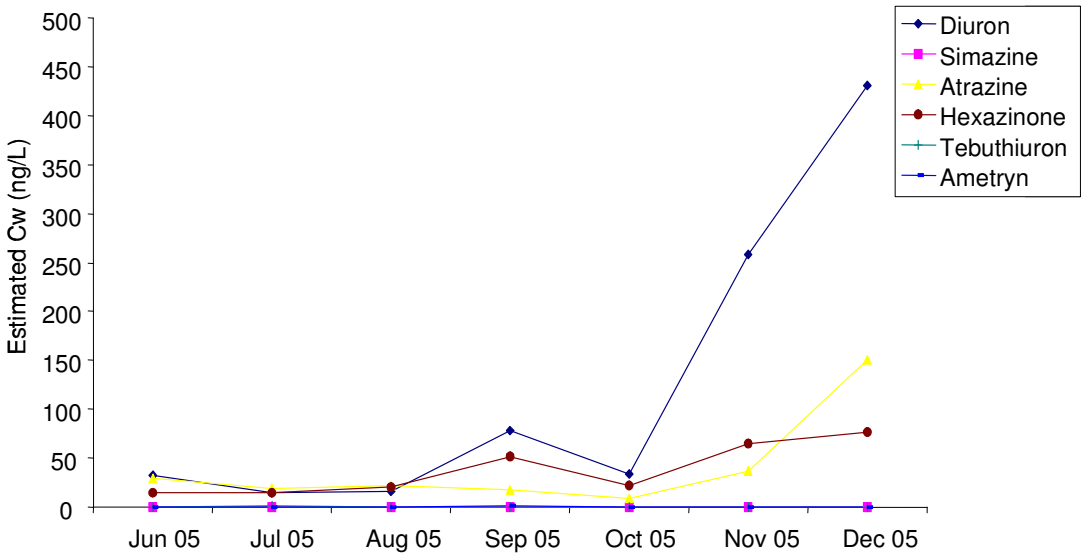

Figure 3.8 Predicted concentrations in water of herbicides in Russell River based on passive sampling data for the period June-December 2005. (Data collected in association with PhD project of Melanie Shaw).

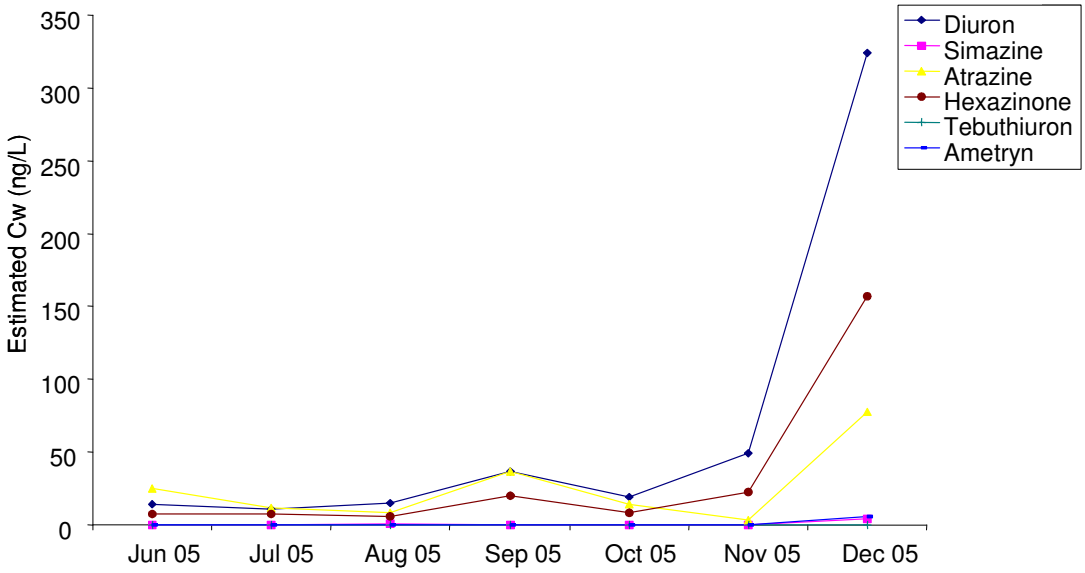

Figure 3.9 Predicted concentrations in water of herbicides in Mulgrave River based on passive sampling data for the period June-December 2005. (Data collected in association with PhD project of Melanie Shaw).

NRM Region Trends

Inshore Reef Sites

Diuron was the pollutant most commonly detected at all inshore reef sites, regardless of NRM region (see Figure 3.10). Note that no EDs were successfully deployed at Long Island in the wet season, thus only pesticides detected with SPMDs are presented. The total detected concentration of all herbicides and pesticides at inshore reefs appear to be higher at the more northern sites of Low Isles and Fitzroy Island, both of which are in the Wet Tropics NRM region (see Figures 2 and 3). Magnetic Island in the Burdekin region had the next highest levels of detection.

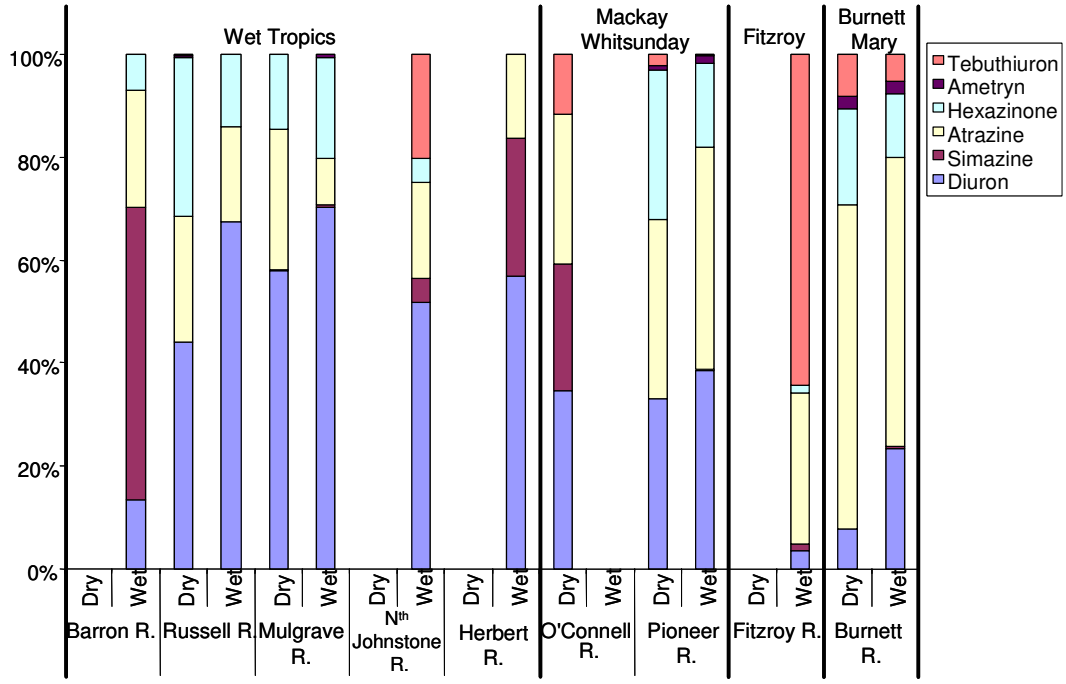

Figure 3.10. Comparison of the percentage contribution by mass of contaminants found in passive samplers across inshore reef sites grouped by NRM region.

River Mouth Sites

The southern NRM regions have a greater variety of compounds detected compared to the rivers in the Wet Tropics NRM region (see Figure 3.11). Diuron made up the greatest proportion of the pesticides found at sites in the Wet Tropics, with the exception of the Barron River. Meanwhile, atrazine made up a greater proportion of the total pesticides detected in the southern NRM regions than in the Wet Tropics NRM region, especially in the Fitzroy and Burnett Mary regions. Note however that although the Pioneer River appears to have a lower level of diuron than other rivers, it simply appears that way as it only makes up a lower percentage of the total due to the greater number of compounds detected. The O'Connell and Pioneer Rivers from the Mackay Whitsunday region had the highest levels of compounds detected.

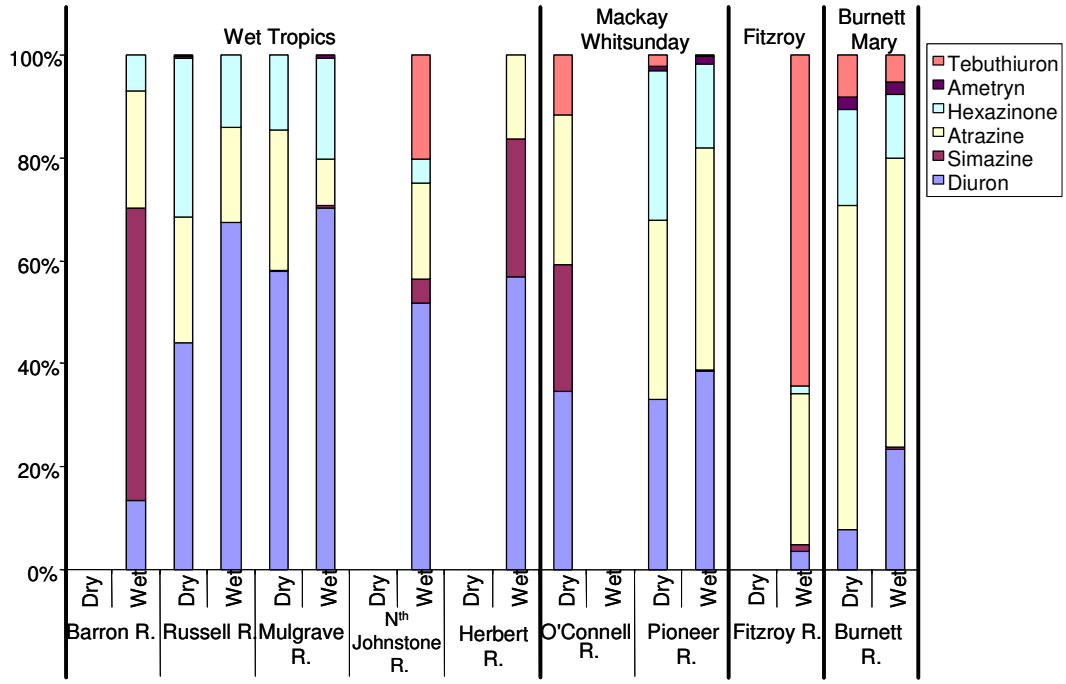

Figure 3.11. Comparison of the percentage contribution by mass of contaminants found in passive samplers across River mouth sites grouped by NRM region.

Grab sampling

Grab samples were collected at seven river mouth sites. These samples were largely collected around the time of flood events of varying sizes to provide information to allow assessment of the performance of passive sampling techniques during flood events. Additionally, samples were collected in transects post TC Larry; one transect was from the Tully River to Dunk Island and the other from the Johnstone River to Russell Island.

Of the grab samples collected, those collected from the Tully, Burdekin, O'Connell and Pioneer Rivers consisted of individual samples that were collected during or shortly after minor floods. The sample collected in the O'Connell River represented the first flow event of the wet season whereas the samples collected from the Pioneer did not coincide with a flow event. The sample in the Tully River was collected post peak flow (approximately 2 days after). A time series of grab samples were collected in the Normanby River, however except for the sample collected one day before peak flow, no pesticides were detected. The two key data sets that provide time series of concentrations originate from the North Johnstone (see Figure 3.12) and the Barron Rivers (see Figure 3.13). The results from these two systems suggest that there is no immediate relationship between a flood event and pesticide concentration in the water.

Diuron was found in all sites and was detected in higher concentrations than other compounds. Atrazine was the second most prevalent herbicide found, however it was not found in the Normanby River. Hexazinone was also reasonably represented, but was not detected in the Normanby or Burdekin Rivers. Simazine was detected at only three sites, the Barron, North Johnstone and Tully Rivers. From these results it

appears that the two southernmost rivers sampled, O'Connell and Pioneer, had the highest levels of herbicides, while the more northern rivers generally showed lower levels.

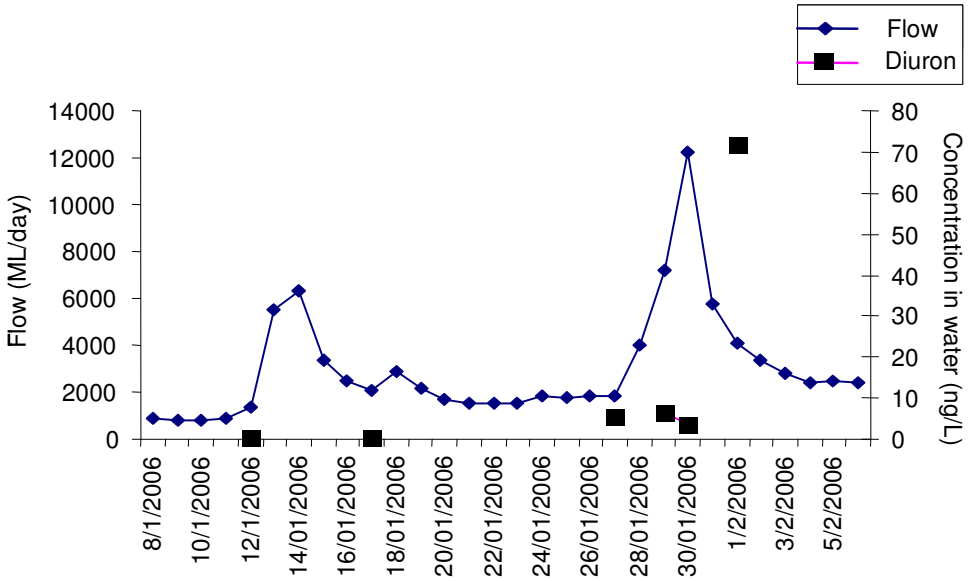

Figure 3.12. Comparison of flow rate and diuron concentration in water obtained from grab sample analysis from the North Johnstone River during a period with two minor floods.

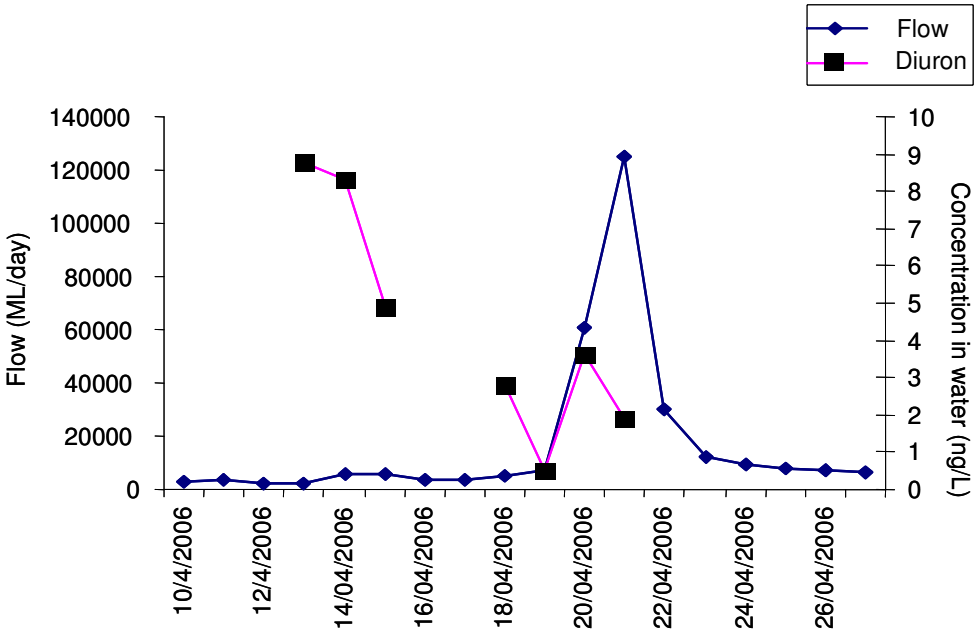

Figure 3.13. Comparison of flow rate and diuron concentration in water obtained from grab sample analysis from the Barron River during a period with a flood event.

Passive Sampling and Grab Sampling Comparison

An evaluation of passive sampling data compared to grab water sampling was carried out in January 2006, a period covering a minor flood event. This sampling period should provide some information that allows an assessment of the suitability of passive sampling techniques during flood events for assessing pollutant loads. Although grab samples were collected from seven rivers for comparison with ED passive samplers, only the Pioneer and North Johnstone Rivers have suitable data. In all other cases, the ED samplers were lost or were not deployed for the time period that the grab water samples were collected. In the case of the Normanby, Tully and O'Connell Rivers, no passive sampler results were obtained due to safety and logistical difficulties with deployment. For the Burdekin River, there are no passive sampler results for the periods the water samples were collected.

Passive samplers were deployed in the North Johnstone River from 19 January 2006 to 15 February 2006. Grab water samples were collected on six occasions from 12 January 2006 until 2 February 2006 (see Table 3.4). Overall, diuron and atrazine were detected in 4 of the 6 grab samples and hexazinone in three of the six grab samples whereas all three chemicals were detected in the passive sampler deployed (see Figure 3.12). Using a detection limit of 0.5 ng/L for grab samples, the average concentration of diuron, atrazine and hexazinone was 15, 5.5 and 1.8 ng/L respectively for the three herbicides in the grab samples. The average concentration is thus slightly higher than the predicted time averaged concentration based on passive sampling results (9.0, 3.2 and 0.84 for diuron, atrazine and hexazinone respectively). As the passive sampler deployment spanned a greater period (27 consecutive days) and the flood represents a short fraction of the entire period, it would be expected that the passive sampler result would be lower than the mean concentration of the grab water samples.

Passive samplers were deployed in the Pioneer River from 23 January 2006 to 16 February 2006 and a grab water sample was taken on the first day of this deployment. The ED detected higher levels of all compounds except for tebuthiuron (see Figure 3.15). The ED sampler was also able to detect simazine and ametryn whereas the grab sample did not.

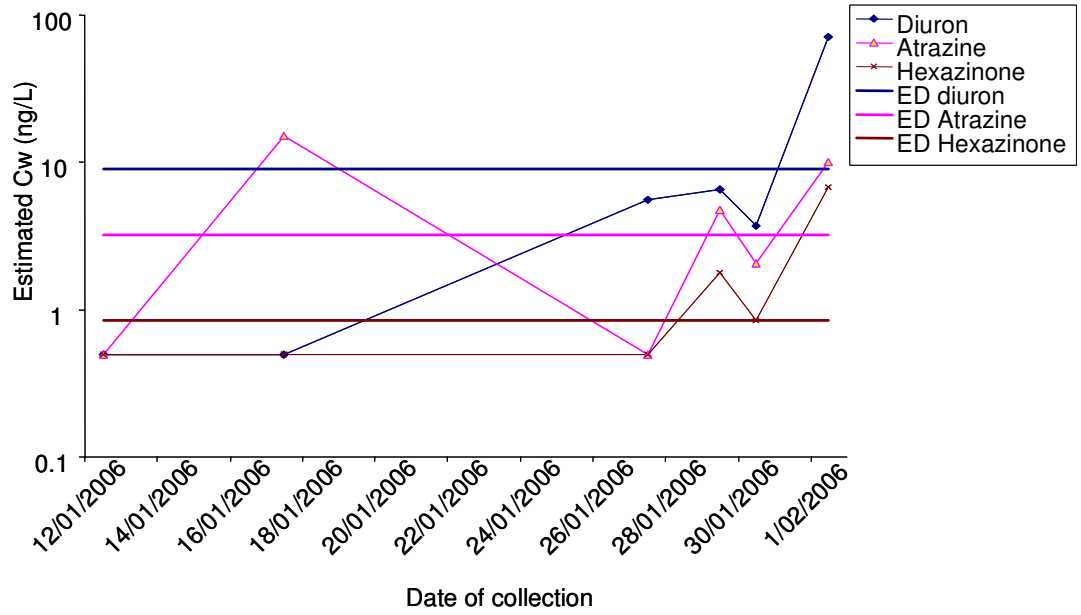

Figure 3.14. A comparison between the observed concentration of diuron, atrazine and hexazinone in six grab water samples collected from the North Johnstone River between 12 January and 1 February 2006, with estimated water concentrations obtained using ED passive samplers deployed from 19 January to 15 February 2006. Note the log scale. For values below the limit of detection, 0.5ng/L was used in the plot and to calculate the averages.

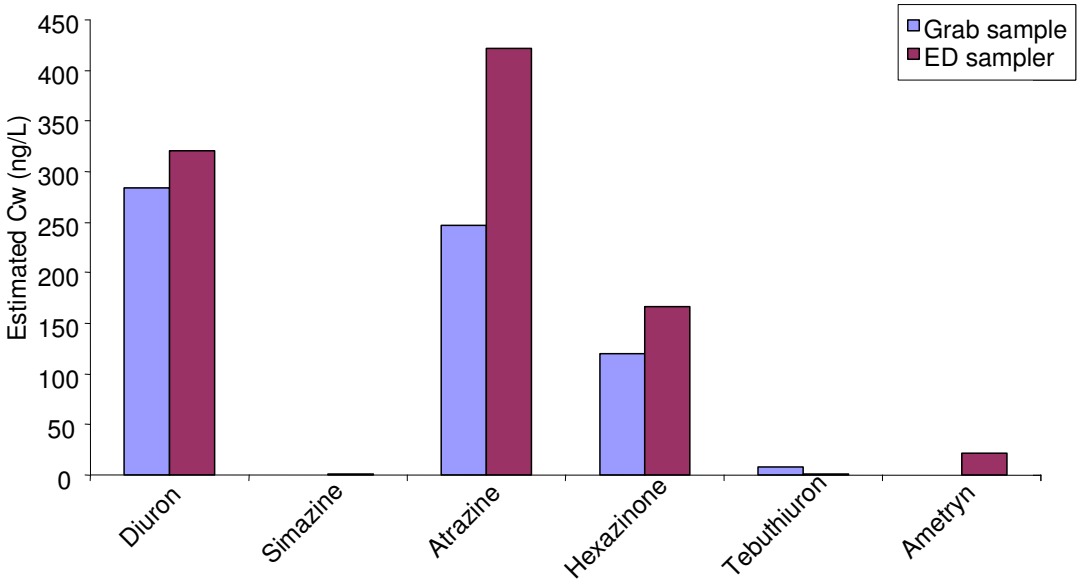

Figure 3.15. Comparison between the measured concentration of herbicides in one grab water sample collected from the Pioneer River on 23 January 2006 and water concentrations predicted from ED passive samplers deployed from 23 January to 16 February 2006.

Transects of Grab Samples after a major flood event

Grab sample data were collected during a flooding event from the Tully River to Dunk Island. A range of herbicides were detected during this study showing consistently that herbicides, in particular enter the marine environment.

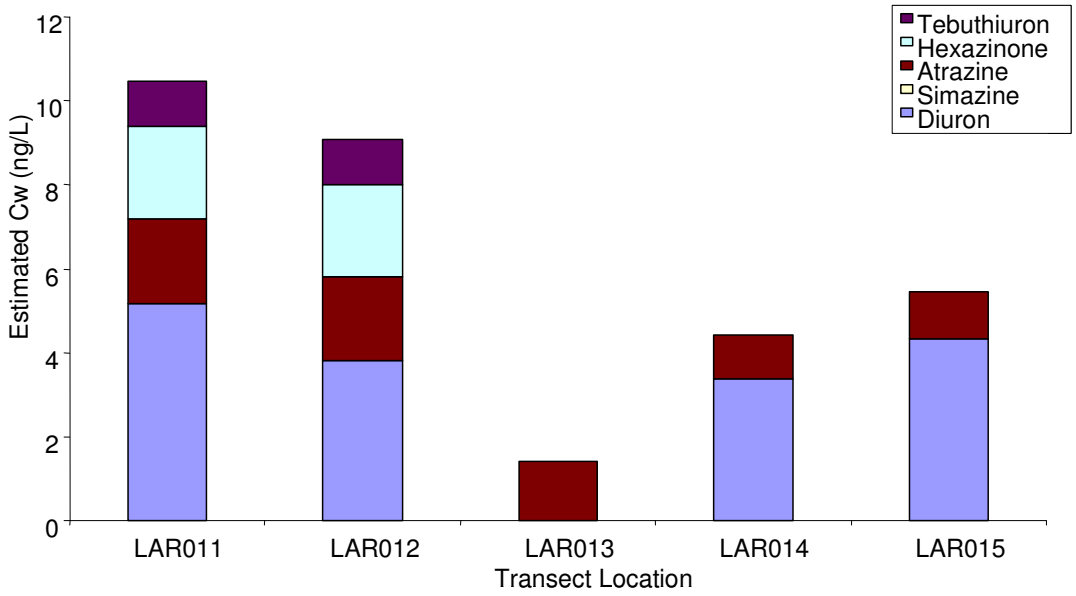

Figure 3.16. Concentrations of measured herbicides found in 1L grab water samples that were collected between North Johnstone River and Russell Island after TC Larry collected on 5 April 2006.

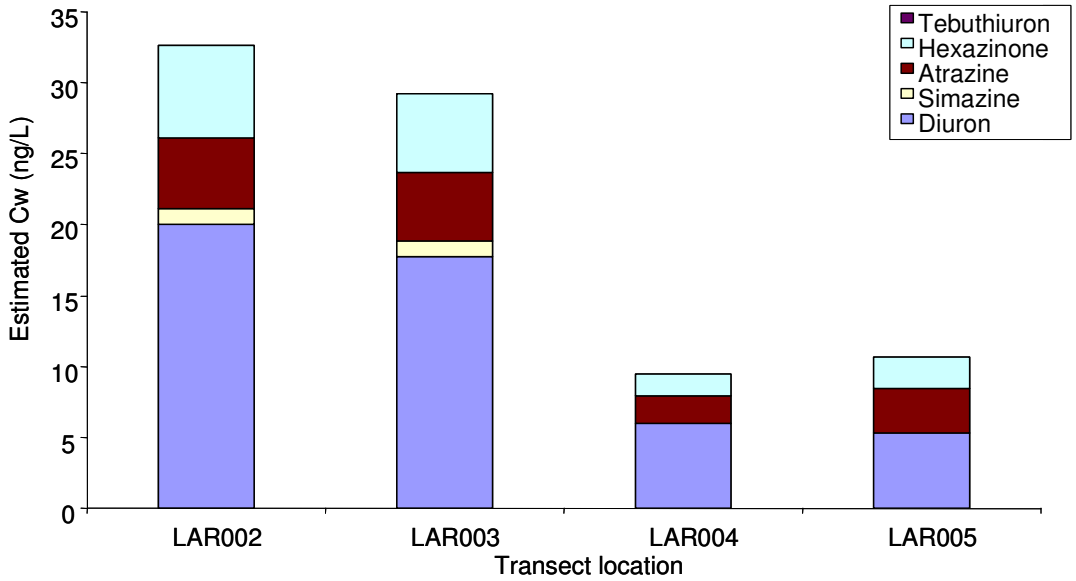

Figure 3.17. Concentrations of measured herbicides found in 1L grab water samples that were collected between Tully River and Dunk Island after TC Larry, collected on 4 April 2006.

Marine sediments

Sediment samples were collected from eight inshore reef sites in close proximity to the passive sampler deployment sites where pesticides were detected. One sample from each site was analysed with a replicate from Fitzroy Island analysed for reproducibility. See tables A2-3.5 and A2-3.6 in Appendix 2 for details on replicates and collection locations.

Few PAHs and pesticides were detected in sediments from inshore reef sites (Table 3.7). Only those compounds of special interest or found above the detection limit are listed. For the full list of PAHs and pesticides included in the analysis see Appendix 2, tables A2-3.8 and A2-3.9.

Pesticides were found at only three sites, with only three pesticides detected. Diuron was found at Long Island and Bedarra Island, while Low Isles contained ametryn and prometryn. Organophosphorus pesticides and organochlorine pesticides were all below the detection limits. In addition, Flouranthene was detected at Long Island. This was the only PAH detected and confirmed on full mass spectrometry.

Table 3.7. Concentrations of PAHs and pesticides (and selected degradation products) detected in sediments, ng/g. Note: the different limits of detection are the result of different sample masses used for extraction. Values in bold font were confirmed on full scan mass spectrometry. Dwt = Dry weight of analysed sample.

Compound	Low Isles	Fitzroy Island	Fitzroy Island (2)	Normanby Island	Bedarra Island	Orpheus Island	Magnetic Island	Long Island	N Keppel Island
Dwt (g)	*29.1*	*34.1*	*34.8*	*29.2*	*24*	*21.3*	*31.7*	*22.2*	*32*
Fluoranthene	<2.4	<2	<2.1	<2.4	<3	<3.3	<2.2	**2.9**	<2.2
Other PAHs	<2.4	<2	<2.1	<2.4	<3	<3.3	<2.2	<3.2	<2.2
Pesticides									
Fluometuron	<0.15	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Diuron	<0.15	<0.1	<0.14	<0.1	0.1	<0.1	<0.09	0.14	<0.13
Simazine	<0.15	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Atrazine	<0.15	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Hexazinone	<0.15	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Tebuthiuron	<0.15	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Ametryn	0.52	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Prometryn	0.14	<0.1	<0.14	<0.1	<0.11	<0.1	<0.09	<0.11	<0.13
Chlorpyrifos	<15	<10	<14	<10	<11	<10	<9	<11	<13
Endosulfan	<15	<10	<14	<10	<11	<10	<9	<11	<13
Diazinon	<15	<10	<14	<10	<11	<10	<9	<11	<13
DDEpp	<1.5	<1	<1.4	<1	<1.1	<1	<0.9	<1.1	<1.3
Organophosphorus pesticides	<10	<9	<9	<10	<13	<14	<9	<14	<9
Organochlorine pesticides	<1	<0.9	<0.9	<1	<1.3	<1.4	<0.9	<1.4	<0.9

Discussion

Pesticides were detectable at all inshore reef sites using passive samplers, especially diuron (Table 3.5). Overall the levels were low and in some cases were detected only at sub-nanogram levels. As could be expected, compounds were detected at substantially higher concentrations at river mouth sites compared to those at inshore reef sites (Table 3.6). A greater number of compounds were also detected at the river sites.

In general, passive sampling has allowed a very sensitive and low-impact evaluation of pesticides at inshore reefs. Continued use of such methods will be essential for future evaluation and the identification of seasonal, spatial and long-term trends relating to these pollutants. Despite the difficulties with the establishment and maintenance of sampling sites and resultant data gaps in the first sampling period, the use of passive samplers has resulted in arguably the most comprehensive data set to date of levels and trends of organic pesticides at the mouths of rivers that drain into the GBRWHA and at inshore reefs of the Great Barrier Reef.

Seasonal Trends

An examination of average monthly rainfall data (see Figure 3.1) shows that the wettest months were January-April 2006 and that during October-December 2005 there was only a gradual increase in rainfall. Although this slow and indistinct on-set of the wet season as defined in this report (from November 2005-April 2006) may have lowered the mean concentration of pesticides over the season, the levels are generally higher than for the dry season. Certainly, detection of pesticide residues on a monthly basis indicates that higher levels, and often more chemicals, are found in wetter months.

The detection of higher levels of pesticides in the wet season indicates that contamination most likely relates to input from land-based sources and that rainfall is increasing the mobility of these chemicals. For example, the diuron concentration at Low Isles peaked in the wet season and atrazine and hexazinone were only detected during a wet season deployment (see Figure 3.4). Conversely, at Fitzroy Island there is no discernable difference between wet and dry season diuron concentrations. However, hexazinone and atrazine were both detected in the wet season. While most herbicides are only introduced into the marine environment from agricultural practices, diuron is a common ingredient in marine antifoulant. The relatively constant concentrations of diuron at Fitzroy Island may indicate that diuron used on boats as an antifoulant may be contributing to a constant background concentration at this site, thus diluting seasonal variability. It should be noted that seasonal differences in pollutant concentrations were not detected at all sites (e.g. Orpheus Island).

A number of pesticides were present at inshore reefs in the wet season that were not detectable in the dry season. This may reflect changes in usage patterns of these chemicals or to increased mobility of these compounds during the wet season. However, simazine was only detected in the dry season at inshore reef sites (at Normanby and Fitzroy Islands, see Figures 3.2 and 3.3). Detection of simazine in the dry season at inshore reef sites has also been observed in a preliminary study by Shaw and Mueller (2005) (see ‘Comparisons with other data’ below).

In summary there is a consistent trend toward higher concentrations of herbicides and an increase in the number of herbicides detected in the wet season at inshore reef sites.

Seasonal comparisons are limited for river mouth sites due to gaps in the data. The data available for the Burnett, Pioneer, Russell and Mulgrave Rivers indicate differences in the pesticides detected and their concentrations between the wet and dry seasons (see Figures 3.5 and 3.6).

The Pioneer River had detectable levels of more pesticide compounds in the wet season compared to the dry season. Furthermore, the data showed increases in pesticide concentrations from December to a peak in February followed by a decline (see Figure 3.7). The Russell and Mulgrave Rivers also showed a trend of increased concentrations in the wet season, however, compared to the Pioneer River the onset of these higher levels began a month earlier, in November rather then December (see Figures 3.8 and 3.9). Additionally, a small peak occurred in the September-October deployment. The occurrence of this peak at both sites suggests that the source is not site specific, but due to a catchment-wide rainfall event or seasonal crop treatment pattern. The current data for the Russell and Mulgrave Rivers showed no seasonal trend for a change in the number of pesticides detected.

In summary, at river mouth sites there appears to be a general trend towards an increase in the total detected pesticide concentration during the wet season followed by a decline into the dry season.

NRM Region comparison

A comparison between pesticide concentrations in 2005/06 between NRM regions was inconclusive due to significant data gaps at some sites caused by delays in site set-up. Sampling at river mouth sites by task collaborators only began in the final month of the dry season.

While there was a greater variety of pesticides found in Wet Tropics river mouth sites, a greater total pesticide concentration was detected in the more southern rivers in the Mackay Whitsunday region (O'Connell and Pioneer rivers, see Figure 3.11). Conversely, at inshore reef sites, the total detected pesticide concentrations are higher in the Wet Tropics region (Low Isles, Fitzroy Island and Normanby Island) in comparison to sites in the southern regions (see Figure 3.10).

Grab sampling was added to this project at a late stage to serve two key objectives, namely i) an improved assessment of loads of chemicals from integration of the flow and concentration data and ii) a comparison with passive sampling data. Both objectives were combined and grab samples were collected primarily during flow events. The results from the grab samples collected from the two main flood events in the North Johnstone River and the Barron River, where herbicides were detected in multiple samples of a time series, showed that at least for the sampling period concentration of pesticides in the water did not correlate with river flow. The concentration and an increase in the baseline concentration of a given pesticide in a river are likely to be affected by many factors related to the physical-chemical properties of the compound. These include the sorption coefficient to the specific soils (KD), the use of the compound including specific application patterns in the catchment, rainfall patterns in the catchment and catchment topography. Therefore it is not surprising that pesticides, unlike water soluble nutrients and (re)suspended sediments (including associated phosphates), do not correlate with river flow. The data from the North Johnstone River suggest that for this particular flood, an increase in the pesticide concentration in the water has likely occurred post flood. It cannot be determined if a similar concentration increase also occurred in the Barron River because the post-flood period was not sampled.

The limited grab sample results from this task suggest that the timing of grab sample collection around flood events is crucial. A more thorough examination of the value of grab sampling for pesticide analyses would require collection of many samples over an extended period, which is very expensive and has substantial logistic problems. The grab sampling under this Reef Plan MMP task failed to collect useful grab sample data for the estimation of river pesticide loads. For systems such as the Tully River, with multiple flood events each season, it appears unlikely that resources would be available to maintain continuous grab sampling programs to cover these floods sufficiently.

Comparison between Passive Sampling and Grab Sampling Results

The evaluation of passive sampling data compared to grab water sampling was successful at only a few sites due to logistical and health and safety issues. The best data set in this study originates from the samples collected in the North Johnstone River which showed a relatively good agreement between the concentrations predicted from passive samplers and the mean concentration obtained from six grab water samples collected over a 20 day period. However, while we have shown the high reproducibility of the passive sampling data, the data from the grab samples vary up to 2 orders of magnitude. In the six collected samples, levels range from below the LOD (<1 ng/L) to high levels of 72 ng/L. As discussed above, the herbicide concentrations did not correlate with flow. In addition, the passive samplers were able to consistently detect a much wider range of chemicals.

In conclusion, the outcomes from this task provide strong evidence that pesticide concentration in water can be reproducibly predicted from passive sampling data and provide results that are similar to grab water sampling.

Transects of Grab Samples after a major flood event

Transects of grab samples were collected from near the Tully River mouth towards Dunk Island and also from the Johnstone River mouth towards Russell Island. The transect from the Tully River Mouth suggests a linear decrease in pesticide concentration with increasing salinity which allows a prediction of the water concentration near the river mouth assuming conservative dilution. The predicted concentration of diuron and atrazine at the Tully River mouth are 64 and 13 ng/L whereas the concentrations in a grab sample collected in the river on the 4 February 2006 were 39 ng/L of diuron and about 20 ng/L of atrazine. Pesticide concentrations along the transect from the North Johnstone to Russell Island showed no clear gradient and concentrations close to the river mouth were not elevated compared to those further offshore. This may indicate that the North Johnstone River contributed little to the herbicides found in the marine environment during this flood event, or that the sampling missed the peak concentrations which may have been present earlier during this post-cyclone flood event (sampling was conducted 2 weeks after the cyclone).

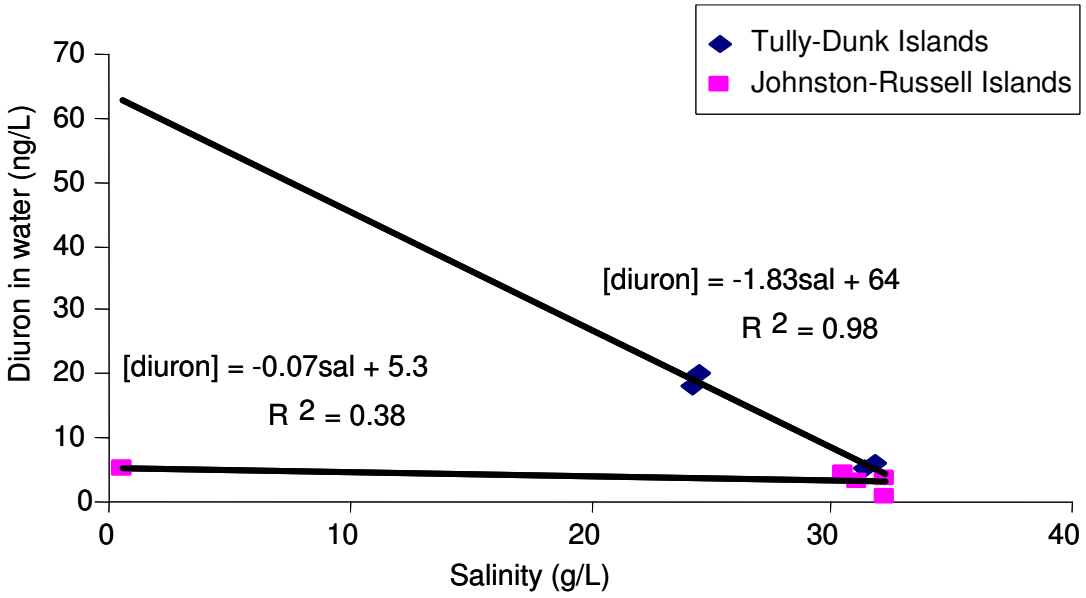

Figure 3.18. Concentration of diuron detected in 1L grab samples collected in a transect from river mouths (low salinity) to inshore reef islands (high salinity).

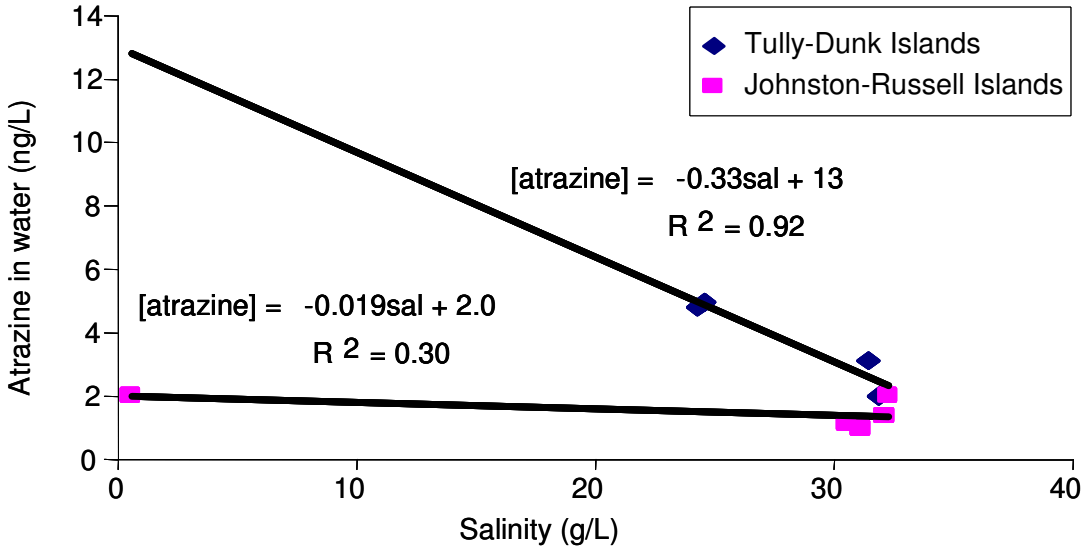

Figure 3.19. Concentration of atrazine detected in 1L grab samples collected in a transect from river mouths (low salinity) to inshore reef islands (high salinity).

Marine sediments

Only very few pesticides were detectable in sediments at inshore reef sites. Some pesticides were detectable but very close to the LOD. Overall, with current analytical methods and LOD's the analysis of pesticide concentrations in marine sediment provides little valuable information for the assessment of long-term trends or regional distribution of land-based chemicals.

Comparisons with other data

Inshore Reefs

The present monitoring task is among the first that provides results on organic pollutants in water of the marine environment of the Great Barrier Reef (a summary of results is presented in Table 3.8). To our knowledge, pesticides were first detected in marine waters along the Queensland coast by Bengtson-Nash *et al*. (2003; also reported in McMahon *et al*. 2003) who were able to detect and quantify diuron in grab samples collected at sites in the Great Sandy Strait of Hervey Bay (<LOD – 25 ng/L) including an offshore site (<LOD – 5 ng/L). The first set of data that are directly comparable to those from this study were obtained by Shaw and Mueller (2005) as part of a study that aimed to evaluate the feasibility of passive sampling techniques for monitoring herbicides in the Great Barrier Reef. Shaw and Mueller deployed samplers at four and seven sites during a dry and wet season respectively. The results demonstrated that with the use of passive sampling techniques, low levels of herbicides (1 – 4 ng/L for the sum of various herbicides) are detectable throughout inshore reef sites in the Wet Tropics. Shaw and Mueller's results also suggested seasonal variation in the combination of pesticides that can be found with a higher fraction of simazine in the dry season. The data by Mueller and Shaw are similar to those observed in this study although arguably a seasonal pattern to the presence of atrazine and simazine are yet to be confirmed. Recently, Rohde *et al*. undertook sampling following a flood in the Mackay Whitsunday and detected atrazine, diuron, hexazinone and tebuthiuron in grab water samples, with concentrations of diuron as high as 440 ng/L. The authors also found an inverse linear relationship between herbicide concentration and salinity suggesting conservative mixing of the river plumes. To our knowledge no data are available on chlorpyrifos or other nonpolar pesticides in water from the Great Barrier Reef.

Other relevant data on pesticides at marine sites along the Great Barrier Reef are primarily from sediment analyses. Cavanagh *et al*. (1999) investigated organochlorine pesticides in sediments from the Great Barrier Reef but was unable to detect any pollutants. Haynes *et al*. (2000) subsequently collected samples from 16 intertidal sites along the east coast of Queensland and detected diuron in intertidal sediments collected near Cairns and Cardwell with concentrations of 0.5 and 1.7 ng/g dwt, respectively, which is higher than the detection of diuron in sediments from inshore reefs in the present study (diuron could only be detected at Long Island and Bedarra Island). Haynes *et al*. also analysed sediments from 26 subtidal sites, where possible adjacent to river mouths, and detected a range of herbicides including diuron (< LOD – 10 ng/g), atrazine (<LOD – 0.3 ng/g) as well as dieldrin, lindane, DDT and DDE all at subnanogram per gram levels. From the data in the sediment, the authors predicted diuron concentrations in the water for various rivers ranging from 100 – 1000 ng/L. These predicted results are similar to at least some of the results from river mouths. However, results from the present study suggest these predictions overestimated the concentration in the water. Overall, the comparison with published data highlights the scarcity of information available and shows that the results from this study will provide the basis for future monitoring of these chemicals in the Great Barrier Reef.

River Mouth Sites

Levels of herbicides and other pesticides in rivers have been part of a range of monitoring projects. The objectives of these studies were often different to those of this Reef Plan MMP task, consequently comparison is complex. Most importantly, other studies have usually employed event grab sampling with limited information on the variability of concentrations over a longer period, which further complicate any meaningful comparison of results.

A summary of studies on herbicides in Queensland rivers over the last 10 years is presented below (Table 3.8). The herbicides atrazine and diuron are typically found at the highest concentrations at the majority of sites. Other commonly detected herbicides in rivers include simazine, hexazinone, ametryn and tebuthiuron. Diuron is widely found in rivers along the Great Barrier Reef coastline from the Mary River in the south (Bengtson *et al*. 2003) to the wet tropics rivers in the north (Shaw and Müller, 2005), a trend that is also observed in the data from this task. We found the highest simazine concentrations in the Barron River, which concurs with other studies that found simazine was found only in the Wet Tropics and the Mary River (this river was not monitored in this task).

Table 3.8. Overview of key studies sampling levels of organic pollutants in the Great Barrier Reef.

Pollutant	Sampling strategy	Location	Level	Reference
Herbicides				
Atrazine	Sediments (dw)	Wet tropics	<0.1-0.3 µg/kg	Haynes *et al.,* 2000
	Seagrass (dw)	Wet tropics	<0.5 µg/kg	Haynes *et al.,* 2000
	Water (ground)	Johnstone Basin(1995)	0.2-0.7 µg/L	Hunter *et al.*, 2001
	Water (surface)	Johnstone Basin(1995)	0.2-0.3 µg/L	Hunter *et al.*, 2001
	Water (surface)	Johnstone Basin(1996)	0.1-0.3 µg/L	Hunter *et al.*, 2001
	Water (surface)	Nth Johnstone, Johnstone Basin (1997)	0.16-0.61 µg/L	Hunter *et al.*, 2001
	Water (surface)	Hervey Bay Region	0.004-0.11 µg/L	Bengston Nash *et al.,* 2005
	Water (ground)	Lower Pioneer River Catchment	<0.01-0.12	Baskaran *et al.*, 2001
	Biota	Johnstone River	nd-20 µg/kg	Russell and Hales, 1993
	Water (surface)	Gregory River	0.04-4.2 µg/L	Rohde *et al.,* 2006
	Water (surface)	Proserpine River	0.1 µg/L	Rohde *et al.,* 2006
	Water (surface)	O'Connell River	0.01-0.87 µg/L	Rohde *et al.,* 2006
	Water (surface)	Pioneer River	0.09-1.2 µg/L	Rohde *et al.,* 2006
	Water (surface)	Plane Creek	0.34 µg/L	Rohde *et al.,* 2006
	Water (surface)	Carmila Creek	0.02 µg/L	Rohde *et al.,* 2006
	Water (marine)	Inshore GBR - Mackay Whitsunday region	<0.01 – 0.1 µg/L	Rohde *et al.,* 2006
	Water (surface)	GBR Wet Tropics River Mouths and Inshore Marine Waters	0.1-0.3 ng/L	Shaw and Müller, 2005
	water(surface)	Pioneer River (Dumbleton Weir)	0.29-1.3 µg/L	Mitchell *et al.,* 2005
	water(surface)	Gooseponds Creek	0.67-4.1 µg/L	Mitchell *et al.,* 2005
Diuron	Sediments (dw)	Wet tropics	<0.1-10 µg/kg	Haynes *et al.,* 2000
	Seagrass (dw)	Wet tropics	<0.5-1.1 µg/kg	Haynes *et al.,* 2000
	Water (surface)	Johnstone Basin(1995)	0.34-2.3 µg/L	Hunter *et al.*, 2001
	Water (surface)	North Johnstone, Johnstone Basin(1997)	0.43-1 µg/L	Hunter *et al.*, 2001
	Water (ground)	Lower Pioneer River Catchment		
	Water (surface)	Gregory River	0.33-6.5 µg/L	Rohde *et al.,* 2006

Pollutant	Sampling strategy	Location	Level	Reference
	Water (surface)	Proserpine River	0.39 µg/L	Rohde *et al.,* 2006
	Water (surface)	O'Connell River	0.02-1.5 µg/L	Rohde *et al.,* 2006
	Water (surface)	Andromanche River	<0.01-0.03 µg/L	Rohde *et al.,* 2006
	Water (surface)	Pioneer River	0.32-3.3 µg/L	Rohde *et al.,* 2006
	Water (surface)	Plane Creek	0.95 µg/L	Rohde *et al.,* 2006
	Water (surface)	Carmila Creek	0.2 µg/L	Rohde *et al.,* 2006
	Water (marine)	Inshore GBR - Mackay Whitsunday region	<0.01-0.44 µg/L	Rohde *et al.,* 2006
	Water (surface)	GBR Wet Tropics River Mouths and Inshore Marine Waters	0.2-1.6 ng/L	Shaw and Müller, 2005
	Water (surface)	Pioneer River (Dumbleton Weir)	0.9-8.5 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Gooseponds Creek	0.56-5.3 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Carmila Creek	0.6 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Sandy Creek	0.6-1.6 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Hervey Bay Region	0.0017-0.079 µg/L	Bengston Nash *et al.,* 2005
Ametryn	Water (surface)	Johnstone Basin(1995)	0.2 µg/L	Hunter *et al.*, 2001
	Water (surface)	Pioneer River	0.01-0.14 µg/L	Rohde *et al.,* 2006
	Water (surface)	Pioneer River (Dumbleton Weir)	0.10-0.30 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Gooseponds Creek	0.12-0.71 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Hervey Bay Region	0.002-0.011 µg/L	Bengston Nash *et al.,* 2005
Organochlorines	Sediments (dw)	Wet tropics	<0.05-0.26 µg/kg	Haynes *et al.,* 2000
Hexazinone	Water (surface)	Gregory River	0.04-0.45 µg/L	Rohde *et al.,* 2006
	Water (surface)	Proserpine River	0.04 µg/L	Rohde *et al.,* 2006
	Water (surface)	O'Connell River	0.01-0.08 µg/L	Rohde *et al.,* 2006
	Water (surface)	Pioneer River	0.06-0.41 µg/L	Rohde *et al.,* 2006
	Water (surface)	Plane Creek	0.26 µg/L	Rohde *et al.,* 2006
	Water (surface)	Carmila Creek	0.04 µg/L	Rohde *et al.,* 2006
	Water (marine)	Inshore GBR - Mackay Whitsunday region	<0.01-0.09 µg/L	Rohde *et al.,* 2006
	Water (surface)	GBR Wet Tropics River Mouths and Inshore Marine Waters	0.08-0.45 ng/L	Shaw and Müller, 2005
	Water (surface)	Pioneer River (Dumbleton Weir)	0.11-0.3 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Gooseponds Creek	nd-1.0 µg/L	Mitchell *et al.,* 2005
	Water (surface)	Hervey Bay Region	0.0011 µg/L	Bengston Nash *et al.,* 2005
Tebuthiuron	Water (surface)	Andromanche River	<0.01-0.51 µg/L	Rohde *et al.,* 2006
	Water (surface)	O'Connell River	0.04-1.4 µg/L	Rohde *et al.,* 2006
	Water (marine)	Inshore GBR - Mackay Whitsunday region	<0.01-0.08 µg/L	Rohde *et al.,* 2006

Pollutant	Sampling strategy	Location	Level	Reference
	Water (surface)	Hervey Bay Region	0.054 µg/L	Bengston Nash *et al.,* 2005
	Seagrass (dw)	Wet tropics	<1 µg/kg	Haynes *et al.,* 2000
	Water	Nearshore	nd-34 µg/kg	von Westernhagen and Klumpp, 1995
Simazine	Water (surface)	GBR Wet Tropics River Mouths and Inshore Marine Waters	nd-2.5 ng/L	Shaw and Müller, 2005
	Water (surface)	Hervey Bay Region	0.002-0.049 µg/L	Bengston Nash *et al.,* 2005
PAHs				
	Sediments (dw)	Green Island	<0.01-15 µg/kg	Smith *et al.,* 1985
	Water	Green Island	<0.3-140 ng/L	Smith *et al.,* 1985
nd – Below the limit of detection.				

Conclusions

The study demonstrated that the use of passive sampling techniques provide highly reproducible results and allows assessment of the seasonal variability as well as variability between sampling sites. One interesting aspect of this study is that, in contrast to other more traditional water quality parameters in the rivers, the concentrations determined using passive sampling techniques did not correlate with flow. This may be related to the many factors that affect the retention/transport of chemicals from a catchment into a river as well as other factors that may relate to the application of the pesticide (ie. where, when and how), the topography of the catchment and river, and dilution from the water input. However, more work may be required to investigate this result.

The passive sampling data are in good agreement with a limited number of grab water samples collected in the rivers where the passive sampling techniques allowed a broader range of chemicals to be quantified. However, the grab water sampling data show substantial variability over short periods and for a comprehensive study the analytical costs may be prohibitive particularly when only few chemicals are likely to be detected. Similarly, the analysis of sediments from inshore reef sites provided little information on these chemicals since most chemicals were not detected.

Another key component of the project relates to the involvement of community partners. This was a challenge that led to the delay in the establishment of sites and commencement of the project. Furthermore it resulted in the loss of many samplers, loss of entire sites and the variability in sampler deployment and retrieval including periods where samplers were not deployed at all. Typically for this type of study, the first year's data are patchy, and have resulted in the implementation of new strategies to minimise the loss of samplers/samples in future (ie. deployment with new security systems, selection of partners and development of better communication strategies).

Despite the loss of samplers and samples and the patchiness of data, the results from this study provide the most comprehensive assessment of anthropogenic pollutants entering and being present in the Great Barrier Reef, which will be essential for the assessment of long term trends.

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4. Nearshore Marine Water Quality Monitoring

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Introduction

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. Land runoff is the largest source of new nutrients to the Reef (Furnas *et al.* 1995). However, most of the inorganic nutrients used by marine plants and bacteria come from recycling of nutrients already within the Great Barrier Reef ecosystem (Furnas *et al.*, 2005).

Extensive water sampling throughout the Great Barrier Reef over the last 25 years has established the typical range of concentrations of nutrients, chlorophyll *a* and other water quality parameters and the occurrence of persistent cross-shelf and seasonal variations in these concentrations (summarised in Furnas, 2005b). While concentrations of most nutrients, suspended particles and chlorophyll *a* are normally low, water quality conditions can change abruptly and nutrient levels increase dramatically for short periods following disturbance events (cyclonic mixing, river flood plumes). However, nutrients introduced, released or mineralised into Great Barrier Reef lagoon waters during these events are generally rapidly taken up by pelagic and benthic algae and microbial communities (Alongi and McKinnon 2005), sometimes fuelling short-lived phytoplankton blooms and high levels of organic production (Furnas, 2005).

Analyses of best available long-term time series in Cairns coastal waters identified long-term net increases in suspended particulate matter, dissolved organic nitrogen and dissolved organic phosphorus concentrations (De'ath, 2005; Furnas, 2005). No long-term changes in concentrations of dissolved inorganic nutrients (nitrate, phosphate, silicate), particulate nutrients or chlorophyll *a* have been identified (*ibid.*). Regional-scale monitoring of surface chlorophyll *a* concentrations in Great Barrier Reef waters since 1992 shows consistent regional (latitudinal), cross-shelf and seasonal patterns in phytoplankton biomass, which is regarded as a proxy for nutrient availability (Brodie *et al.* in press). In the mid- and southern Great Barrier Reef, higher chlorophyll *a* concentrations are usually found close (< 25km, within 20m depth) to the coast. Overall, however, no long-term *net* trends in chlorophyll *a* concentrations have been found (Brodie *et al.* in press).

The objectives of this Task were to continue the collection of long-term water quality data series (in particular the long-term chlorophyll monitoring), to initiate regular water sampling at fixed sites in the inshore area of the GBRWHA, and to explore the value of remote sensing as a future water quality monitoring technique.

The objectives of this task are to:

- Determine spatial patterns and long-term (decadal) trends in inshore water quality within the Great Barrier Reef lagoon, particularly in inshore habitats directly affected by river runoff.

- Provide satellite-based information on near-surface chlorophyll, suspended solids concentrations and vertical attenuation of diffuse downwelling light coefficients in lagoonal and coastal waters of the GBRWHA.
- Explore the usefulness of autonomous instruments for high-frequency measurements of local water quality

Marine Water Quality Samples

(Attachment B Task 2.1)

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Sampling locations

In 2004/2005, marine water quality data were collected only along the AIMS 'Cairns Transect' (sampling sites in coastal waters off Douglas Shire and Cairns, Table 4.1; see Furnas (2005) for more information), which was has been biannually sampled by AIMS since 1989. Only two locations, Cape Tribulation (equivalent to "Daintree Reefs" in the contract) and Snapper Island, correspond with the sampling locations prescribed under the Reef Plan MMP. Sampling of the other Reef Plan MMP sites was not undertaken during 2004/2005 because of delays in contract finalisation.

Table 4.1. Locations of water sampling along the AIMS Cairns transect (Wet Tropics NRM Region) in the dry and wet seasons of 2004/05 and 2005/2006. Sampling locations are listed from north to south. Only Snapper Island and Cape Tribulation correspond with the sampling locations prescribed under the Reef Plan MMP Contract.

Location Name
Cape Tribulation
Snapper Island
Daintree River
Port Douglas
Double Island
Yorkeys Knob
Cairns Airport
Cairns Fairlead
Mission Bay
Green Island
Cape Grafton

In 2005-06, biannual sampling of all locations specified under the Reef Plan MMP was carried out in August/September 2005 and January 2006, including stations of the Cairns Coastal Transect. Sample locations were both close to the reefs selected for the inshore coral reef surveys under Reef Plan MMP, (see Table 4.2 and Chapter 5) and in open coastal waters (Figure 4.1).

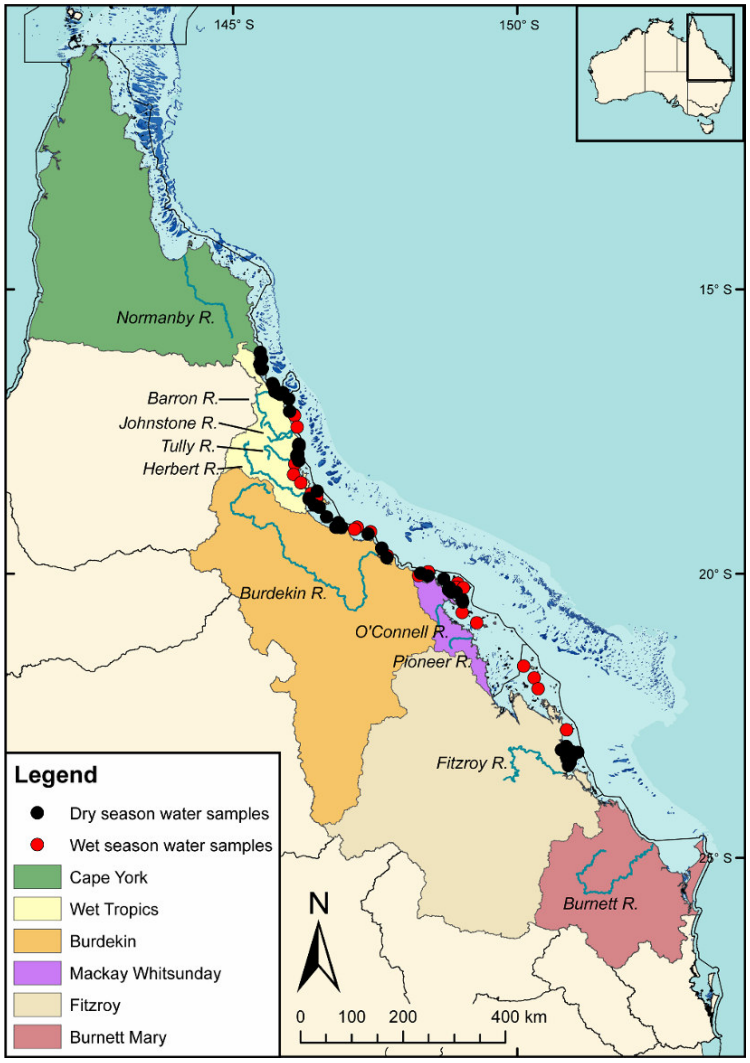


Figure 4.1. Great Barrier Reef lagoon water quality sampling locations during 2005/06 research cruises in the nearshore zone.

Sample collection, preparation and analyses

Discrete water samples were collected from two to four depths through the water column with Niskin bottles. Sub-samples taken from the Niskin bottles were analysed for salinity (using a HYTEC 6220 Salinometer), dissolved inorganic nutrients (NH_4, NO_2, NO_3, PO_4, $Si(OH)_4$), DON, DOP, PN, PP, SS and plant pigments (chlorophyll *a*, phaeophytin). Temperatures were measured with reversing thermometers from at least 2 depths.

The sub-samples for dissolved nutrients were immediately filtered through a 0.45µm filter cartridge (Sartorius Mini Sart N) into acid-washed screw-cap plastic test tubes and frozen in a lab freezer (-18ºC) for later analysis ashore. The sub-samples for particulate nutrients and plant pigments were collected on pre-combusted glass fibre filters (Whatman GF/F). Sub-samples for suspended solids were collected on pre-weighed 0.4µm polycarbonate filters. Filters were wrapped in pre-combusted aluminium foil envelopes and stored at -18ºC until analyses.

Inorganic dissolved nutrients (NH_4, NO_2, NO_3, PO_4, $Si(OH)_4$) concentrations are determined by standard wet chemical methods (Treguer and LeCorre, 1975) implemented on a segmented flow analyser (Bran and Luebbe, 1997).

Table 4.2. Locations selected for inshore water quality monitoring. Water samples were collected at all locations during research cruises in August/September 2005 and January 2006.

Great Barrier Reef Sector	NRM Region	Catchment	WQ monitoring locations
Cairns	Wet Tropics	Daintree	Cape Tribulation*
			Snapper Island
		Russell-Mulgrave Johnstone	Fitzroy Island
			High Island
			Russell Island
			Normanby Island
Innisfail		Tully	North Barnard Island
			King Reef
			Dunk Island
Townsville	Burdekin	Herbert	Orpheus Island
			Lady Elliot Reef
		Burdekin	Pandora Reef
			Havannah Island
			Middle Reef
			Geoffrey Bay
Whitsundays	Mackay Whitsunday	Proserpine O'Connell Pioneer	Double Cone Island
			Daydream Island
			Shute Island
			Pine Island
			Hook Island
			Dent Island
			Seaforth Island**
			Whitsunday Island***
Capricorn-Bunkers	Fitzroy	Fitzroy	Peak Island
			Barren Island
			Pelican Island
			Humpy Island
			Middle Island
			North Keppel Island

Variations of survey locations from contract:
*A location off Cape Tribulation was chosen as equivalent to the required "Daintree Reefs"
**Seaforth Island. was chosen instead of Lindeman Island., because it is an inshore coral monitoring site (see Chapter 5).
***Whitsunday Island. (2 sites) was only sampled in January 2006.

Water samples for total dissolved nutrients (TDN and TDP) were UV-irradiated for 12 hours to oxidise the organic matter (Armstrong *et al*., 1966; Walsh, 1989) before analyses as above. DON and DOP are calculated by subtracting the separately measured inorganic nutrient concentrations (above) from the TDN and TDP values. From 2006 (i.e. wet season cruises 2006), analyses of TDN and TDP were carried out after persulphate digestion (Valderrama, 1981) instead of UV-oxidation. Persulphate digestion is likely to be more reliable for the higher nutrient concentrations expected from coastal waters, and will facilitate comparison with water analyses results obtained under other monitoring programs.

Particulate nitrogen (PN) was determined by high-temperature combustion of filtered particulate matter on glass fibre filters using an ANTEK 707/720 Nitrogen Analyser (Furnas *et al*., 1995). The analyser is calibrated using AR Grade EDTA for the standard curve and marine sediment BCSS-1 as a control standard.

PP was determined spectrophotometrically as inorganic P (PO_4: Parsons *et al*., 1984) after digesting the particulate matter in 5% potassium persulphate (Furnas *et al*., 1995). The method is standardised using orthophosphoric acid and dissolved sugar phosphates as the primary standards.

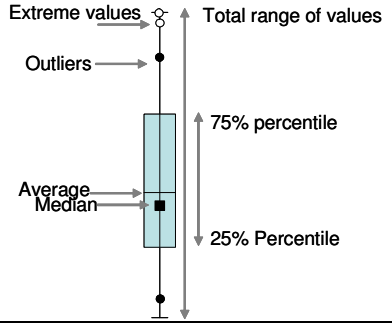
Chlorophyll *a* and phaeophytin concentrations were measured fluorometrically using a Turner Designs 10AU fluorometer after grinding the filters in 90% acetone (Parsons *et al*., 1984). The fluorometer was calibrated against chlorophyll *a* extracts from log-phase diatom cultures (chlorophyll *a* and *c*). The extract chlorophyll concentrations were determined spectrophotometrically using the wavelengths and equation specified by Jeffrey and Humphrey, (1975).

SS concentrations were determined gravimetrically from the difference in weight between loaded and unloaded 0.4 µm polycarbonate filters (47mm diameter) after the filters had been dried overnight at 80°C.

Statistical analysis

To report nearshore lagoon samples by NRM region, the marine boundaries of each NRM region were applied. The nearshore/offshore boundary was defined as the 20m depth contour. This is the depth where regular resuspension of sediments by surface waves occurs, creating water quality conditions dissimilar to deeper water (M. Furnas pers. comm.). All reported lagoon water quality stations, biannually sampled during research cruises, are within this depth contour.

Values for water quality parameters at each station were calculated as depth-weighted averages. Summary statistics of these values are presented as box and whisker plots (see box below for definitions) for the whole Great Barrier Reef nearshore lagoon and by NRM region.



Template for box plots in this chapter:

- The box contains 50% of the values= interquartile range (IQR)
- Outliers are defined as being >1.5 x IQR
- Extreme values are defined as being >3 x IQR

Temperature, salinity, total suspended solids, PP, PN, POC and chlorophyll and phaeophytin concentrations were analysed with a Principal Component Analysis (PCA) to classify the water column at various sampling stations along the GBR coast (Figure 4.1). Most parameters were strongly right skewed (a few extremely large outlying values) and a log transformation was used to covert parameters to normality. To place all parameters on a common scale, each variable was standardised by subtracting its mean from each value and dividing by the

standard deviation. The results from the PCA were illustrated in two-dimensional biplots that combined parameter vectors with the distribution of sampling stations in 2D space.

Data from the 'Cairns Coastal Transect', which was regularly sampled by AIMS since 1989, is the only available long-term dataset for water quality parameters in the GBR lagoon (other than chlorophyll, see below) to conduct a temporal trend analysis. This dataset was analysed to assess the temporal trend of water quality parameters in the GBR lagoon over the observation period. Water quality parameters were measured at 11 locations on 49 occasions from 1989 – 2006. Data were screened for outliers and then averaged across duplicates and across sites. Trends in particulate phosphorous, particulate nitrogen, total dissolved nitrogen and suspended solids were assessed using log-linear models with linear and natural splines with 4 degree of freedom. The significance of the terms were based on F-tests.

Results

Nearshore weather quality data are summarised for the whole GBR and for each NRM region, separately for the dry and wet season (Figures 4.2 to 4.6). Detailed data for each station in proximity to the coral reefs survey locations are in Tables A1-4.1 and A1-4.2 (Appendix 1).

Phosphate and DOP values were high in the dry season samples and the whole GBR and regional median values of total phosphorus exceeded the Queensland Water Quality Guideline value of 20 µg L^{-1} (Environmental Protection Agency 2006). Median values of all other water quality parameters were within the guideline values for both seasonal samplings.

In general, higher values were found for most water quality parameters during the wet season, e.g. for DIN, DON, PN, PP, chlorophyll and phaeophytin (Figures 4.4, 4.5 and 4.6). Higher and more variable values during the dry season were measured for suspended solids, DIP and DOP (Figure 4.3 and 4.5).

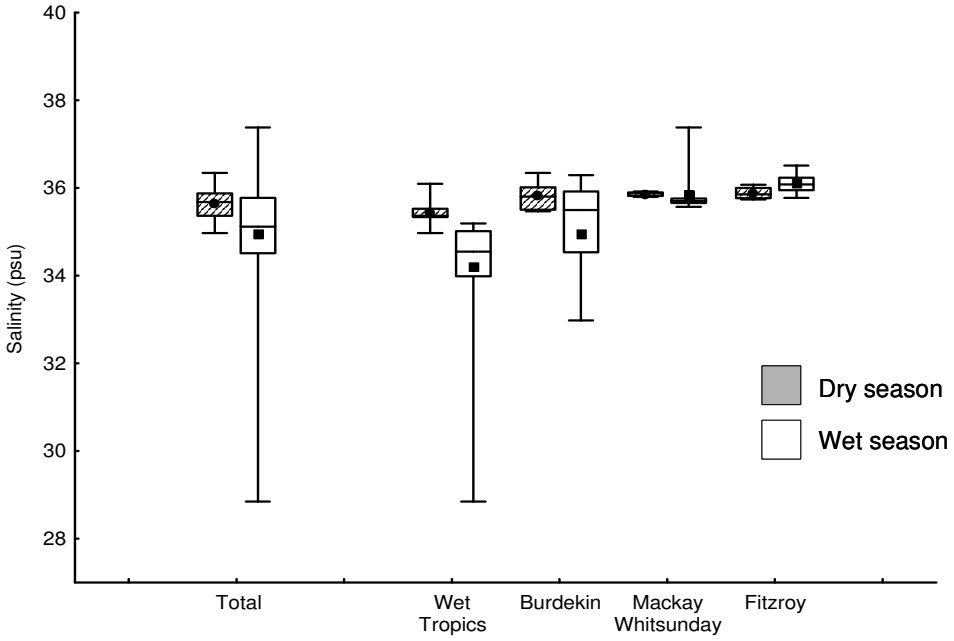


Figure 4.2. Summary of salinity values for the whole Great Barrier Reef (Total) and by NRM region for the sampling period May 2004 to April 2006. Dry season (May- Oct)= shaded boxes, wet season (Nov-Apr)= white boxes. See page 103 for more details about the box plot presentation.

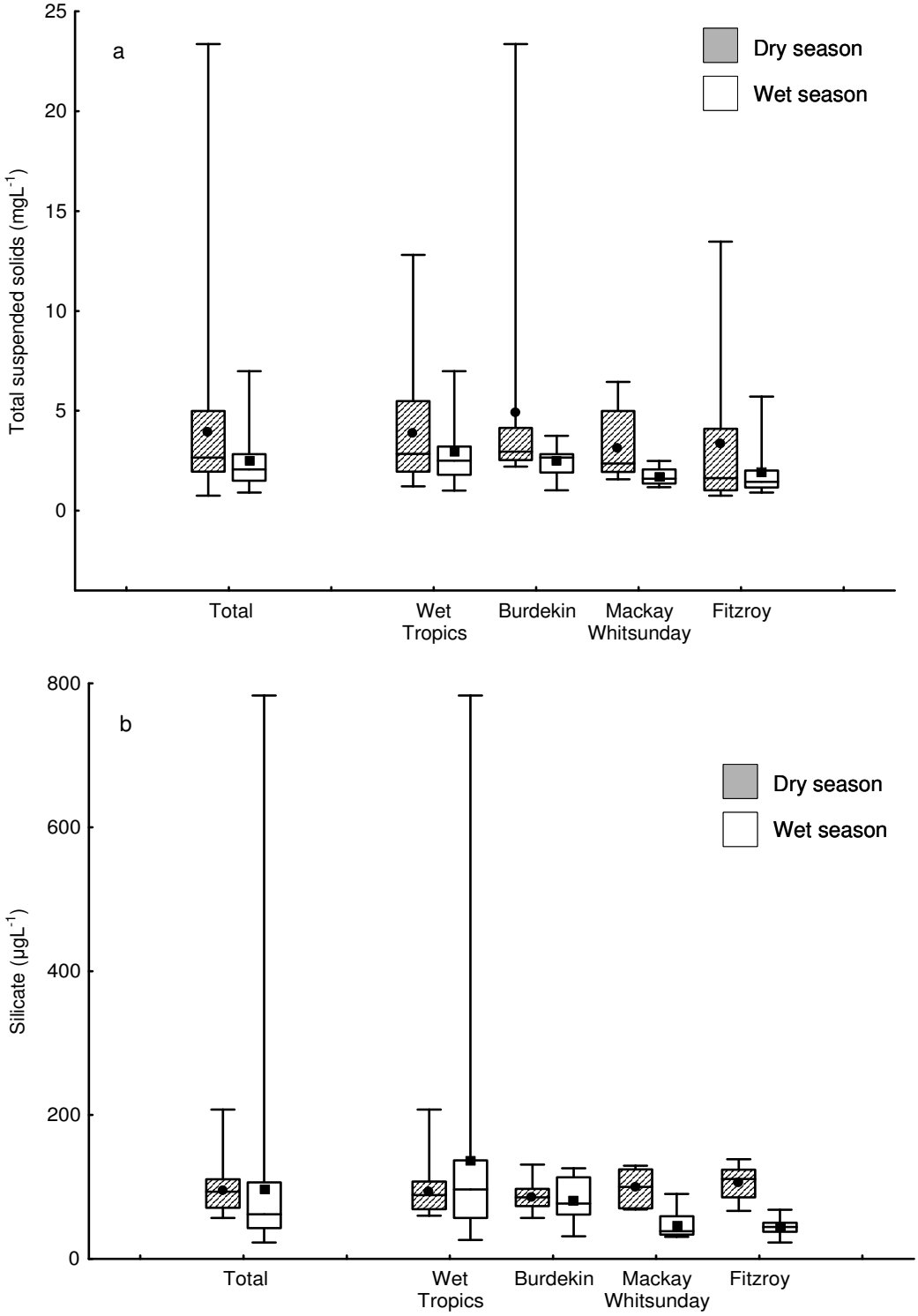


Figure 4.3. Summary of a) suspended solids and b) silicate concentrations for the whole Great Barrier Reef (Total) and by NRM region for the sampling period May 2004 to April 2006. Dry season (May- Oct)= shaded boxes, wet season (Nov-Apr)= white boxes. See page 103 for more details about the box plot presentation.

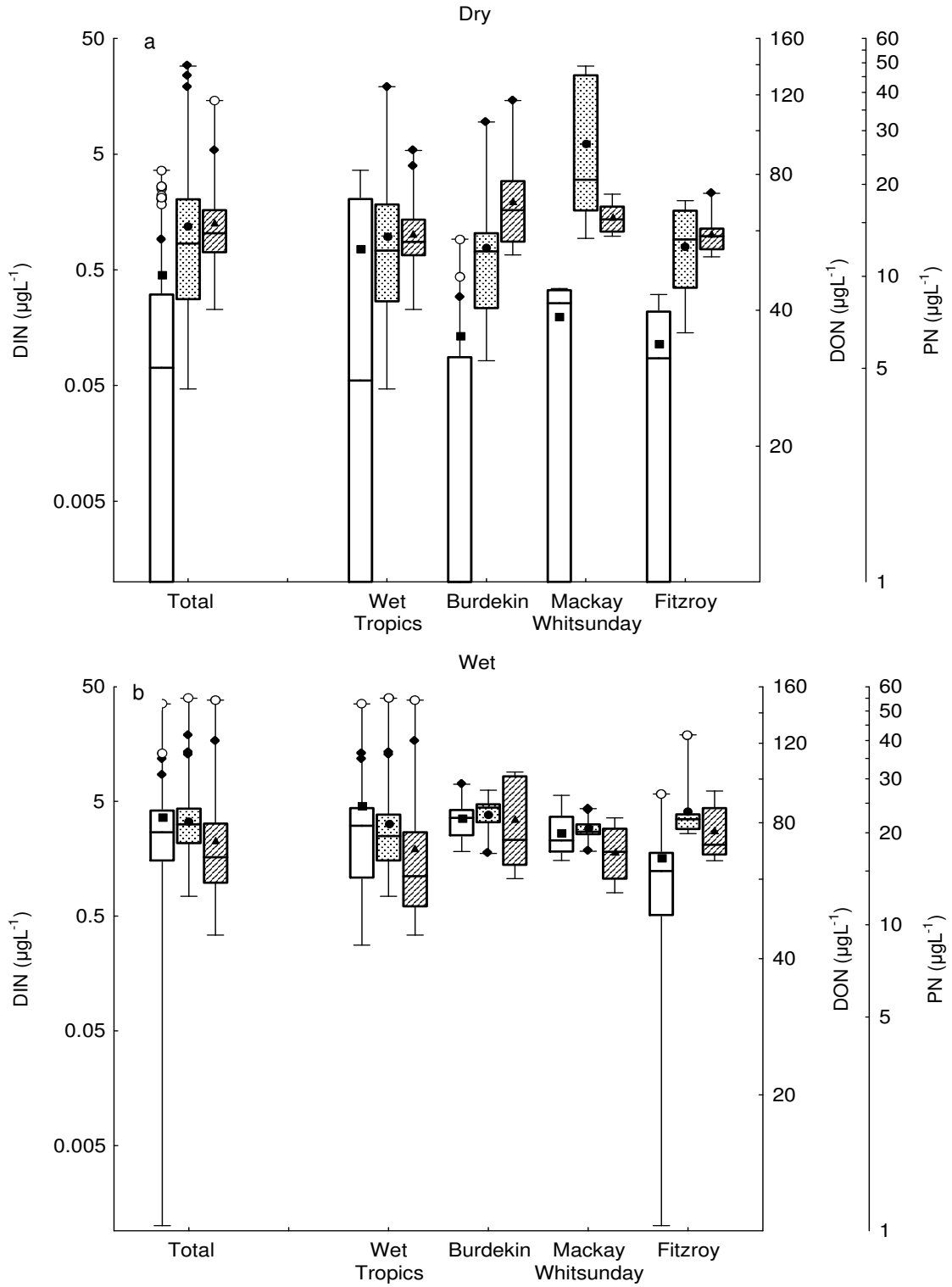


Figure 4.4. Summary of concentrations of nitrogen species for the whole Great Barrier Reef (Total) and by NRM region for the sampling period May 2004 to April 2006. a) Dry season (May- Oct) and b) wet season (Nov-Apr). Dissolved inorganic N= white boxes, Dissolved organic N= light shaded boxes, particulate N= dark shaded boxes. See page 103 for more details about the box plot presentation. Note the logarithmic scale.

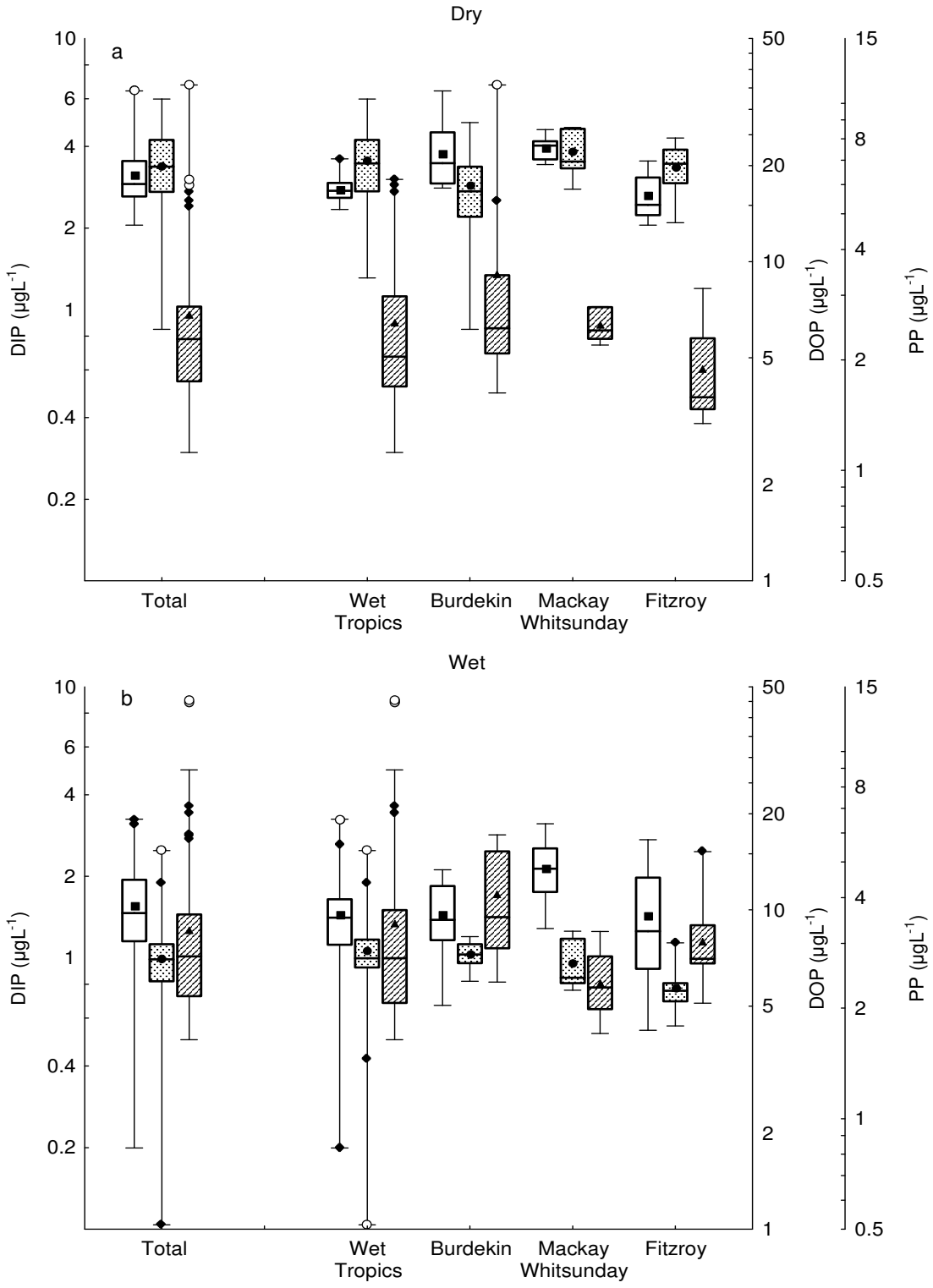


Figure 4.5. Summary of concentrations of phosphorus species for the whole Great Barrier Reef (Total) and by NRM region for the sampling period May 2004 to April 2006. a) dry season (May- Oct) and b) wet season (Nov-Apr). Dissolved inorganic P= white boxes, Dissolved organic P= light shaded boxes, particulate P= dark shaded boxes. See page 103 for more details about the box plot presentation. Note the logarithmic scale.

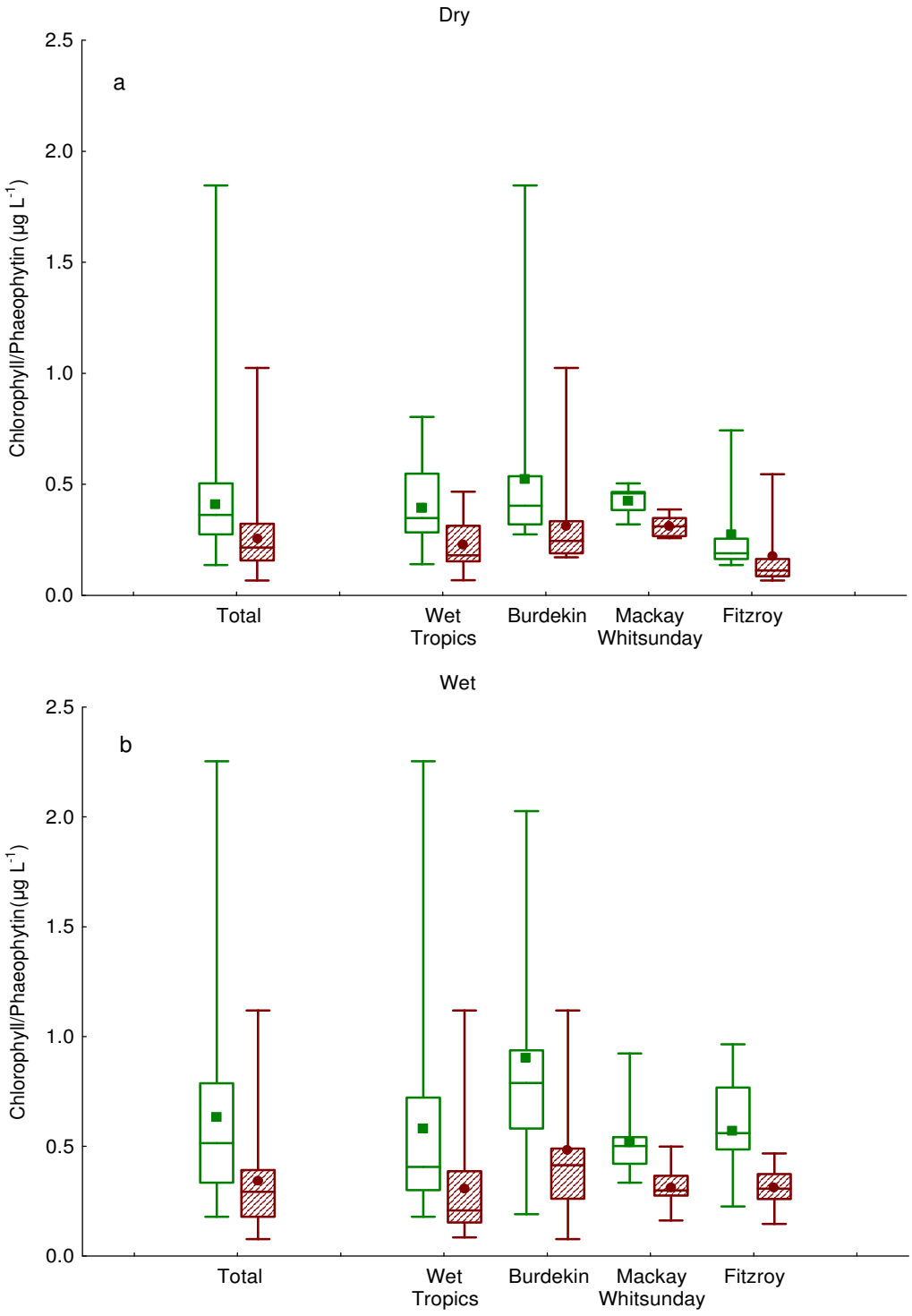


Figure 4.6. Summary of chlorophyll (white bars) and phaeophytin (shaded bars) concentrations for the whole Great Barrier Reef (Total) and by NRM region for the sampling period May 2004 to April 2006. a) Dry season (May- Oct) and b) wet season (Nov-Apr). See page 103 for more details about the box plot presentation.

Spatial patterns of a number of water quality parameters were explored by principal component analysis. The PCA on physical, biological and chemical variables separated the water quality sampling stations by wet and dry season (Figure 4.7). Sites are predominantly grouped by season (driven by differences in temperature and salinity) and there is now clear spatial pattern of water quality parameters along the nearshore Great Barrier Reef. The exception are samples collected at coastal sites between Dunk and Magnetic Islands from the 28th and 31st of January 2006 (within ellipse, Figure 4.7), which had higher values of all parameters (e.g. PN ranging from ~10-54 µg L^{-1}, PP: 0.9-9 µg L^{-1}, chlorophyll: 0.2-2.4 µg L^{-1} and SS: 0.9-9 mg L^{-1}). A number of these stations were within Hinchinbrook Channel and followed a period of heavy rains and minor flooding in the Herbert and Tully River catchments (see Chapter 2 for river flow information and Figure 4.2 for associated low salinity values).

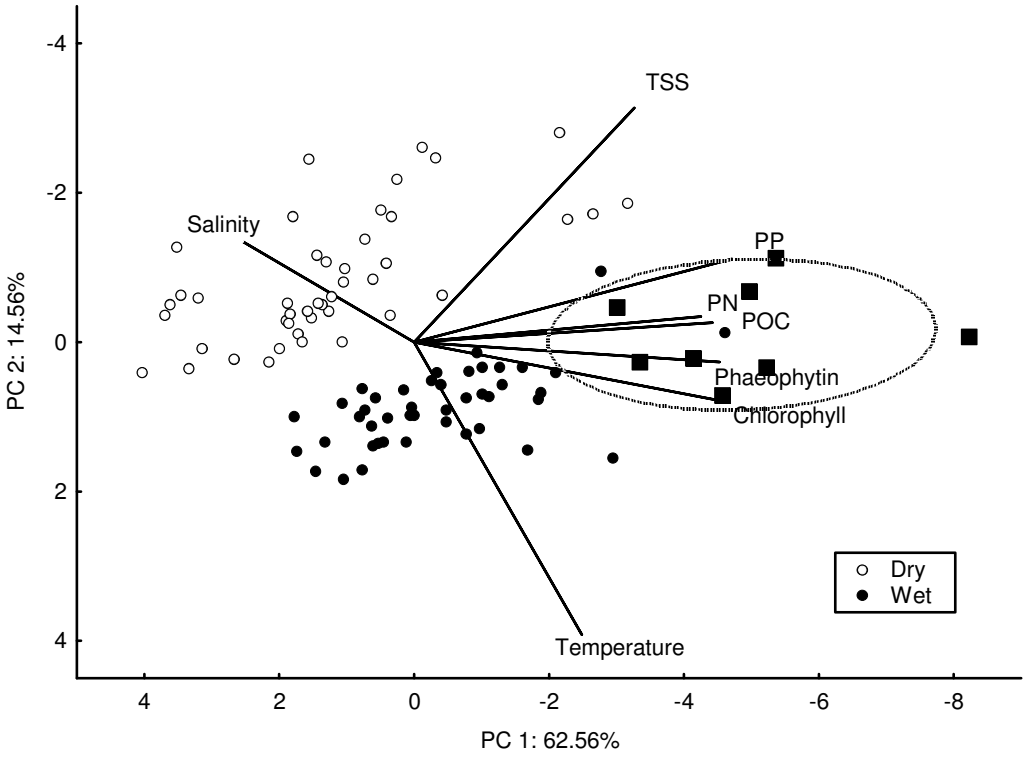


Figure 4.7. Principal component analysis on water quality parameters collected in the dry season 2005 and the wet season 2005/06 in the nearshore Great Barrier Reef lagoon from the Daintree to Keppel Bay. Open symbols are dry season sampling stations, black symbols are wet season stations. Ellipse is centred on the bivariate mean for coastal stations sampled between Dunk and Magnetic Islands from the 29 and 31 January 2006 and encompass the range of these sites.

Water quality parameters sampled since 1989 along the 'Cairns Coastal Transect' (Figure 4.8 for sampling locations) were analysed for temporal trends. Linear temporal trends were significant for PP, TDP and SS (Figure 4.9, Table 4.3). While TDP and SS were increasing over time, PP slightly decreases. For PN and TDN the trends were non-linear with no linear trend effect for the former parameter (Figure 4.9, Table 4.3).

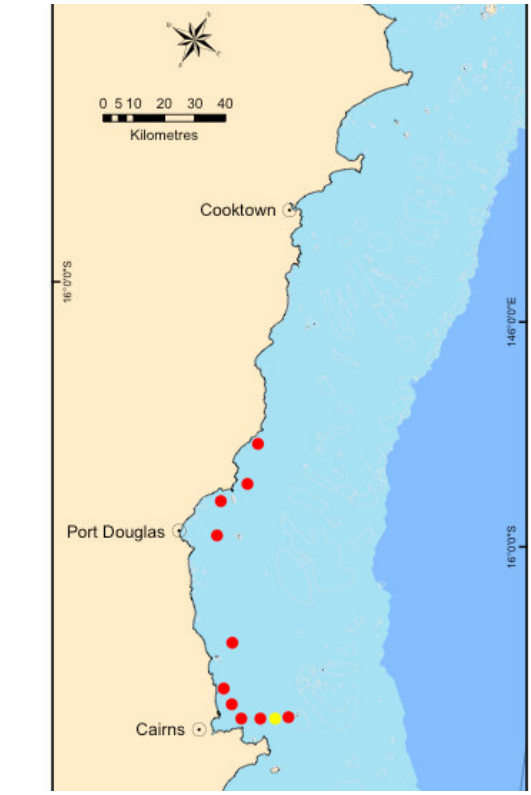


Figure 4.8. Locations of coastal stations in the Cairns region that have been repeatedly sampled by the Australian Institute of Marine Science since 1989. The station marked by a yellow dot is not included in the analysis below as its characteristics vary between those of coastal stations and those near Green Island.

Table 4.3. Cairns Coastal Transect. Analyses of variance assessing the significance of trends over time. ns= Natural spline.

Response variable	Source	df	F	Pr(>F)
PP	time	1	6.557	0.014
	ns(time,4)	3	0.462	0.710
	Residuals	43		
PN	time	1	7.038	0.011
	ns(time,4)	3	4.812	0.006
	Residuals	44		
TDP	time	1	41.615	<0.001
	ns(time,4)	3	0.459	0.714
	Residuals	42		
TDN	time	1	1.302	0.260
	ns(time,4)	3	5.972	0.002
	Residuals	42		
SS	time	1	5.645	0.023
	ns(time,4)	3	1.323	0.281
	Residuals	38		

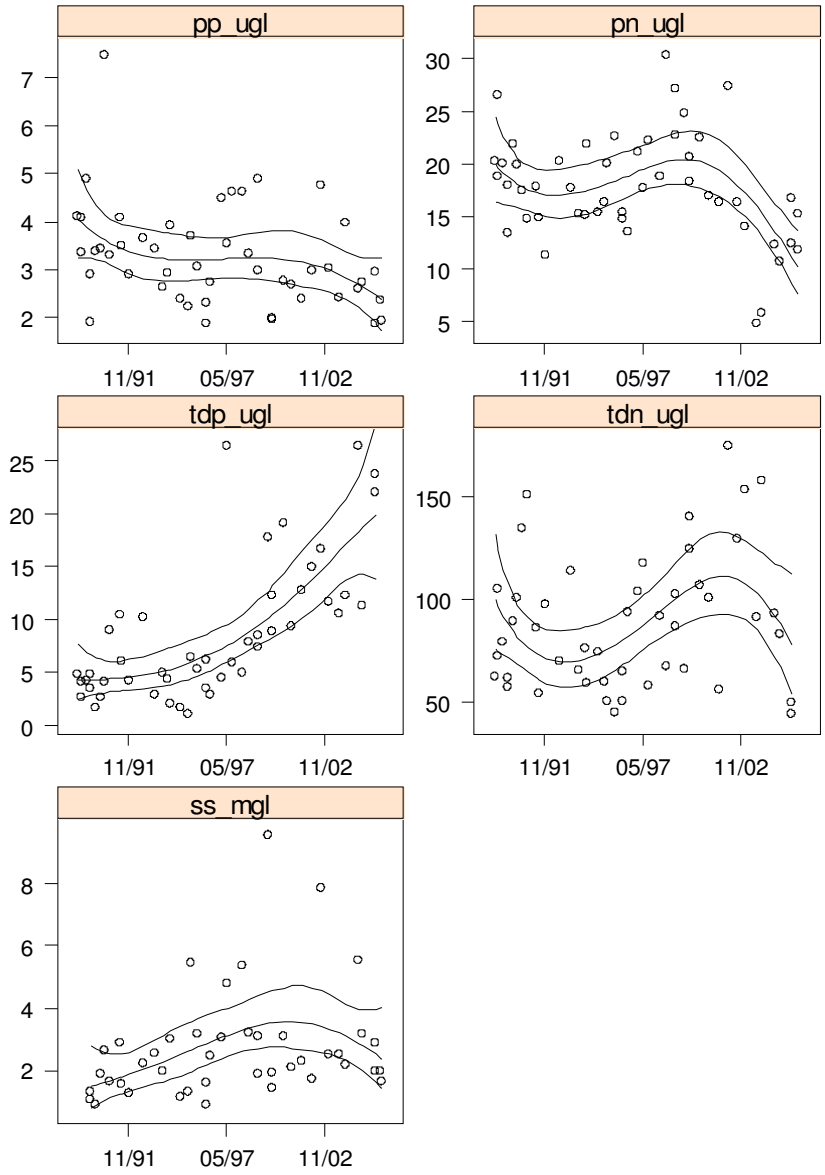


Figure 4.9. Smooth trends over time (1989-2006) for the water quality parameters total dissolved nitrogen, particulate nitrogen, total dissolved phosphorus, particulate phosphorus (all in µg L-1) and suspended solids (mg L-1).

Coastal and Lagoon Chlorophyll a Concentrations

(Attachment B Task 2.4, 2.5; Attachment E Task 2.3, 2.4)

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Collections of surface water samples for chlorophyll *a* analyses for the Reef Plan MMP are achieved by two components involving community engagement: the Long-Term Lagoon Chlorophyll Monitoring Program (ongoing monthly since 1992) and a new component for collection of monthly[2] coastal water samples. A map of sampling locations is in Figure 4.10.

Sampling locations

The Long-Term Lagoon Chlorophyll Monitoring Program has involved, in most cases, monthly sampling at stations along inshore-offshore transects. Under the Reef Plan MMP, nine transects were sampled (Table 4.4). Seven of the nine required transects have been a continuation of existing arrangements. Collection and filtration of samples was carried out by officers of the Queensland Parks and Wildlife Service (QPWS) (currently 3 transects) or by tourism operators (currently 4 transects). Two additional transects were established in the central Wet Tropics and Hervey Bay, sampled by a tourism operator and a seafood industry operator.

In 2005/06 community and other interest groups were engaged to carry out collection and initial preparation of water samples for the Coastal Chlorophyll Monitoring under the Reef Plan MMP, in collaboration with GBRMPA. Interested community groups have been engaged in all eleven suggested coastal sites (see Table 4.5 for details). Regular monthly sampling generally commenced in late 2005, some sites had earlier sampling undertaken.

[2] Note that the initial contract specification was for weekly coastal chlorophyll sampling. This was varied to monthly sampling in March 2006.

Table 4.4. Details of the Long-term chlorophyll monitoring: cross-shelf transect sampling in 2004-05 and 2005-06. Shading indicates sampling carried out by community.

NRM Region	Transect name	No of sites	Sampler	Sampling details
Cape York	Far Northern	5	Undersea Explorer	Continuation of long-term transect, sampling about 4 times per year.
	Cooktown-Osprey	8	Undersea Explorer	Continuation of long-term transect, monthly sampling from Feb 2005
Wet Tropics	Port Douglas	5	Undersea Explorer	Continuation of long-term transect, monthly sampling from Feb 2005
	Cairns	7	QPWS	Continuation of long-term transect, bi-monthly sampling from Feb 2005.
	Wet Tropics (Dunk Island.)*	3	Quick Cat Scuba Diving Adventures	New transect, monthly sampling commenced Dec 2005
Burdekin	Townsville	2	Sunferries	Irregular sampling since Dec 2003. Monthly sampling recommenced in Nov 2005
Mackay Whitsunday	Whitsunday	3	QPWS	Continuation of long-term transect, monthly sampling from Feb 2005
Fitzroy	Keppel Bay	5	QPWS	Continuation of long-term transect, monthly sampling from Feb 2005
Burnett-Mary	Hervey Bay	3	QLD Sea Scallops Ltd	New transect, monthly sampling commenced in Mar 2006.

* Samples collected along the new Wet tropics transect were not analysed because they were defrosted due to a long-lasting power outage on the island after TC Larry.

Table 4.5. Details of the coastal chlorophyll monitoring carried out by community groups during 2005-06.

NRM Region	Location	Community Group	Sampling details
Cape York	Cooktown	Cook Shire Council	Sampling started in Dec 2005, some months not sampled due to river flood dominating water at the site
Wet Tropics	Port Douglas	Undersea Explorer	Monthly sampling since Sep 2005
	Fitzroy Island	Fitzroy Island resort	Monthly sampling since Dec 2005
	Bedarra Island*	Bedarra Island Resort	Monthly sampling since Dec 2005
	Cardwell	Cardwell State School	Monthly sampling since Dec 2005
Burdekin	Magnetic island	Volunteer	Monthly sampling since Oct 2005
Mackay Whitsunday	Shute Harbour	MWHW	Monthly sampling since Oct 2005
	Mackay Marina	MWHW	Monthly samples since Nov 2005
Fitzroy	Rosslyn Bay	Cap Reef	Monthly samples since Nov 2005
Burnett-Mary	Gladstone, Tannum-Boyne coast (6 sites)	Tannum Sands Coastcare	Monthly sampling since Sep 2005
Burnett-Mary	Burnett coast** (5 sites)	Woongarra Marine Park Monitoring & Education Project	Continuation of an ongoing Program, monthly sampling from Feb 2005

* Samples collected at the Bedarra location were not analysed because they were defrosted due to a long-lasting power outage on the island after TC Larry.
** No suitable community group was found to sample the required site at Urangan. After discussion with GBRMPA this site was replaced by a number of sites along the Burnett coast, for which also previous data exist.

Sample collection, preparation and analyses

Two replicate (only one sample for coastal chlorophyll) surface water samples were collected at each site. Each sample was subsampled and filtered onto 2 replicate GF/F filters and stored at -18ºC until analysis (refer to methods for lagoon water quality, above).

The following parameters were also measured at each site at the time of sampling: salinity (with a refractometer), water temperature (with a manual thermometer), secchi depth, water depth (depth sounder), the presence of *Trichodesmium*, and information about the weather, wind and tides.

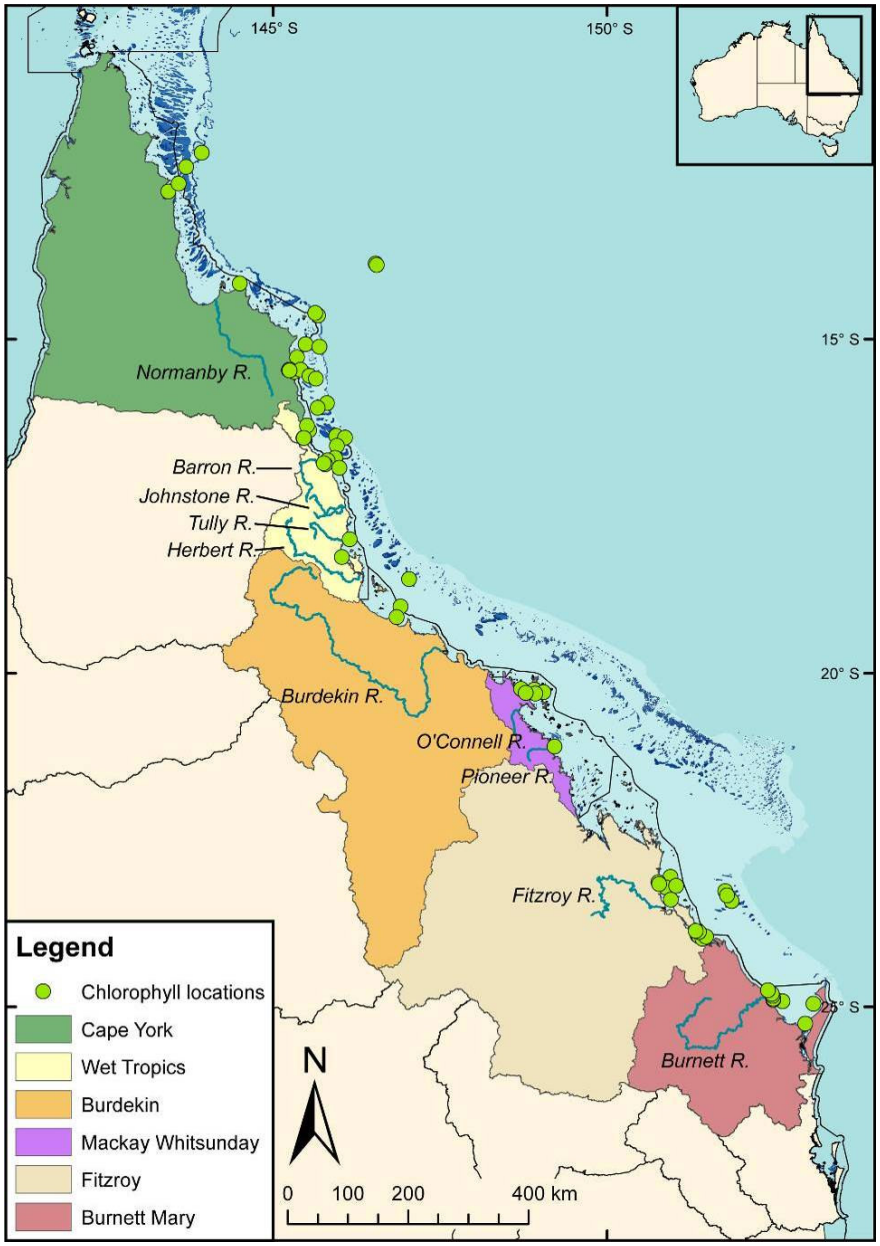


Figure 4.10. Long-term chlorophyll monitoring locations sampled during 2005-06.

Statistical analyses

Chlorophyll data along transects has been collected since 1992 under the GBRMPA and CRC Reef-funded 'Long-term Chlorophyll monitoring program'. Several transects are still being sampled under the current Reef Plan MMP. This long-term dataset was used to conduct a temporal trend analysis over the observation period (1993- 2006). Data on concentrations of chlorophyll have been collected on the GBR in at 95 locations in six sectors: Far Northern (FN), Cooktown to Lizard Island (C-L), Port Douglas to Cairns (CA-PD), Townsville (TV), Whitsunday (WH) and Keppel Bay to Capricorn Bunker reefs (CB-K). The distributions of the sampling effort are shown in Table 4.6 and Figure 4.11. The objective of the analyses was to assess the effects of transect, years, relative distance across the shelf and month of year on the levels of chlorophyll.

Table 4.6. Long-term Chlorophyll monitoring program. Frequency of sampling across transects.

Sector	FN	C-L	CA-PD	TV	WH	CB-K
Sampling occasions	76	906	1608	303	222	1217

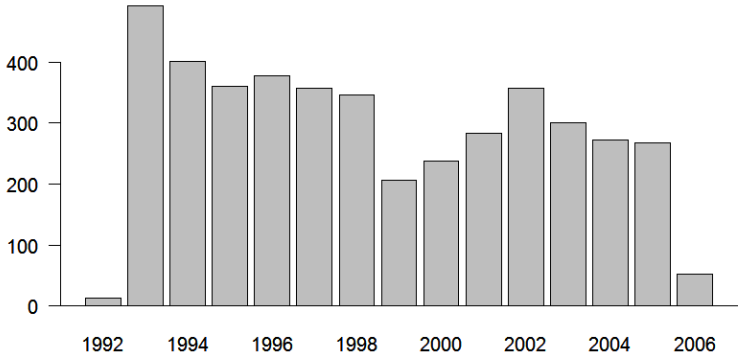


Figure 4.11. Long-term Chlorophyll monitoring program. Frequency of sampling across years.

Data were averaged over replicates and duplicates. The chlorophyll data are problematic to analyse since they have a highly skewed distribution, the sampling design is unbalanced over time and space and locations were visited repeatedly. The requirement was to estimate mean concentrations and hence transforming the data to remove skewness was not used since this results in substantial downwards bias of chlorophyll estimates by ~10-30%. To cater for these complexities, generalised additive mixed models were used for the analyses. Random effects of the sampling locations were included in all models and a "quasipoisson-log link function" was used (this assumes the variance of the observation is proportional to the mean). Smooth effects in decimal years (eg April 1st 1998 ~1998.25) and relative distance across the reef shelf (0 on the coast and 1 on the 60m isocline). The smoothness of temporal and spatial trends was estimated by cross-validation. Spatial smooths were fitted to four composite regions.

Results

To represent the spatial patterns of chlorophyll concentrations in Great Barrier Reef waters the results from the transect and coastal monitoring and the biannual lagoon water sampling were aggregated over the sampling period for each NRM region (Figure 4.12).

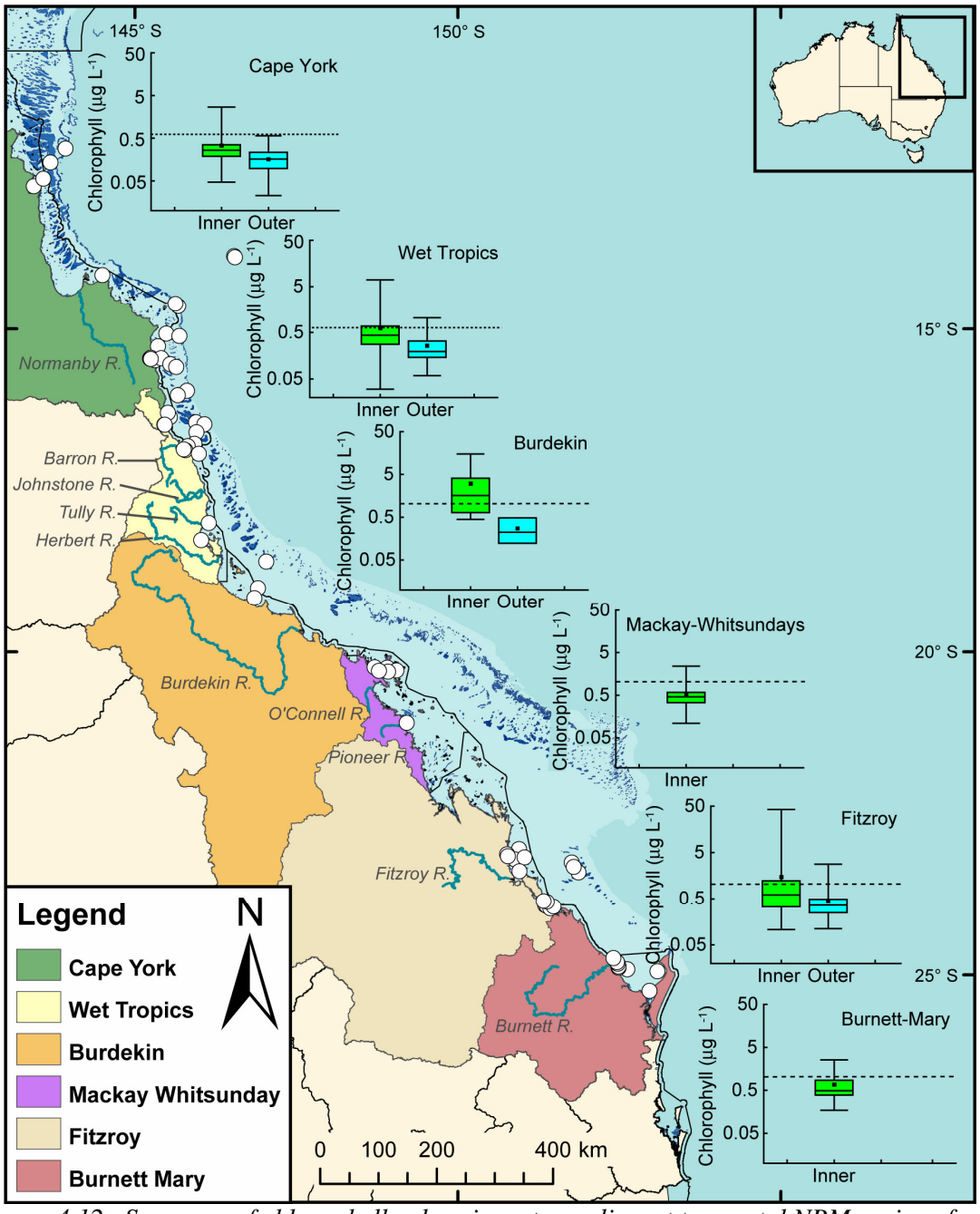


Figure 4.12. Summary of chlorophyll values in waters adjacent to coastal NRM regions for the sampling period 01 May 2004 to 30 April 2006. 'Inner'= stations in water less than 20 m deep, 'outer'= stations in water deeper than 20 m. Horizontal lines= median, square symbols= average, boxes=quartiles, whiskers=maximum and minimum values. Dotted and dashed lines represent the Queensland Water Quality guideline values for coastal waters (see text). Note the logarithmic scale.

In all regions chlorophyll values outside the nearshore zone ('outer') are distinctly lower than inshore values (Figure 4.12). Nearshore values in the Cape York region are less than half than in other regions. Averages and medians over the reporting period did not exceed the respective Queensland Regional Guideline Values (open coastal waters: Wet Tropics region: 0.6 µg L^{-1}, Central Coast region: 1 µg L^{-1}, Environmental Protection Agency 2006), except for the inshore Burdekin region. These values are, however, based on relatively few samples. Detailed seasonal values for chlorophyll, phaeophytin, temperature and secchi depth for the transect and coastal sites are in Tables A1-4.5 to A1-4.9 (Appendix 1).

Chlorophyll values measured along the inshore-offshore transects were generally below the Queensland Regional Guideline Values (open coastal waters: Wet Tropics region: 0.6 µg L^{-1}, Central Coast region: 1 µg L^{-1}, Environmental Protection Agency 2006; see Table A1-4.5 in Appendix 1 for detailed data). Seasonal averages exceeding the respective guideline values were measured within the Cairns transect at the stations close to the coast ('inner'; dry seasons 2004 and 2005 and wet season 2005-06) and stations away from the coast ('outer'; wet season 2005-06). A similar pattern was found within the Keppel Bay transect, with 'inner' stations exceeding the guideline value in the dry seasons 2004 and 2005 and wet season 2005/06 (Table A1-4.5).

Coastal chlorophyll values were mostly below the Queensland Regional Guideline Values (open coastal waters: Wet Tropics region: 0.6 µg L^{-1}, Central Coast Region: 1 µg L^{-1}; enclosed coastal waters: 2 µg L^{-1} in all regions, Environmental Protection Agency 2006; see Table A1-4.9 in Appendix 1 for detailed data). 2005/06 wet season averages exceeding the respective guideline values were found at: Port Douglas, Fitzroy Island., Cardwell, Magnetic Island., Rosslyn Bay, Gladstone-Oyster Rocks and the Burnett River mouth (Table A-4.9). Dry season averages exceeded guideline values at Cardwell, Gladstone-Oyster Rocks and Colosseum Inlet and the Burnett River mouth (Table A1-4.9). High values were predominantly caused by presence of *Trichodesmium* in a few samples for that season or the occurrence of flood waters affecting the site, indicated by low salinity readings (salinity data not shown) (e.g. Magnetic Island).

An important and time-consuming component of the coastal chlorophyll monitoring was the engagement of community groups. This involved the development of user-friendly sampling manuals (AIMS 2005a, b), incl. 1-page Quick Reference Guides (see Appendix 1), and the provision of hands-on training in sample collection and initial sample preparation (filtration before freezer storage). Training sessions also included the presentation of overviews of the Reef Plan and the Reef Plan MMP.

Long-term monitoring of chlorophyll concentrations in the Great Barrier Reef

The long-term dataset for Great Barrier Reef chlorophyll concentrations (1993 to 2006) confirms the strong cross-shelf pattern of decreasing chlorophyll concentrations with increasing distance from the coast for all regions except the Far Northern region of the Great Barrier Reef (equivalent to the Cape York NRM region) (Figure 4.13 and Figure A1-4.1 in Appendix 1, Table 4.7). The cyclical variation over months of the year was strong, with high values in summer and lowest values in winter (Figure A1-4.2 in Appendix 1, Table 4.7).

Long-term variations in mean chlorophyll concentrations were also observed (Figure 4.14). For example, mean chlorophyll concentrations in the Far Northern region doubled from 1997 to 2000 and then declined back to 1997 levels by 2006, and chlorophyll in the Keppels to Capricorn Bunkers region increased ~3.5 fold from 2001 to 2006. Temporal trends were

linear for the Cooktown-Lizard region and Townsville regions, with a slight increase in the former and no significant change in the latter (Figure 4.14, Table 4.7). The Far Northern, Port Douglas to Cairns and Whitsundays regions fluctuated over time with no net linear trend. There was a steep increase from 2001-2006 in the Keppels to Capricorn Bunkers region (Figure 4.14, Table 4.7).

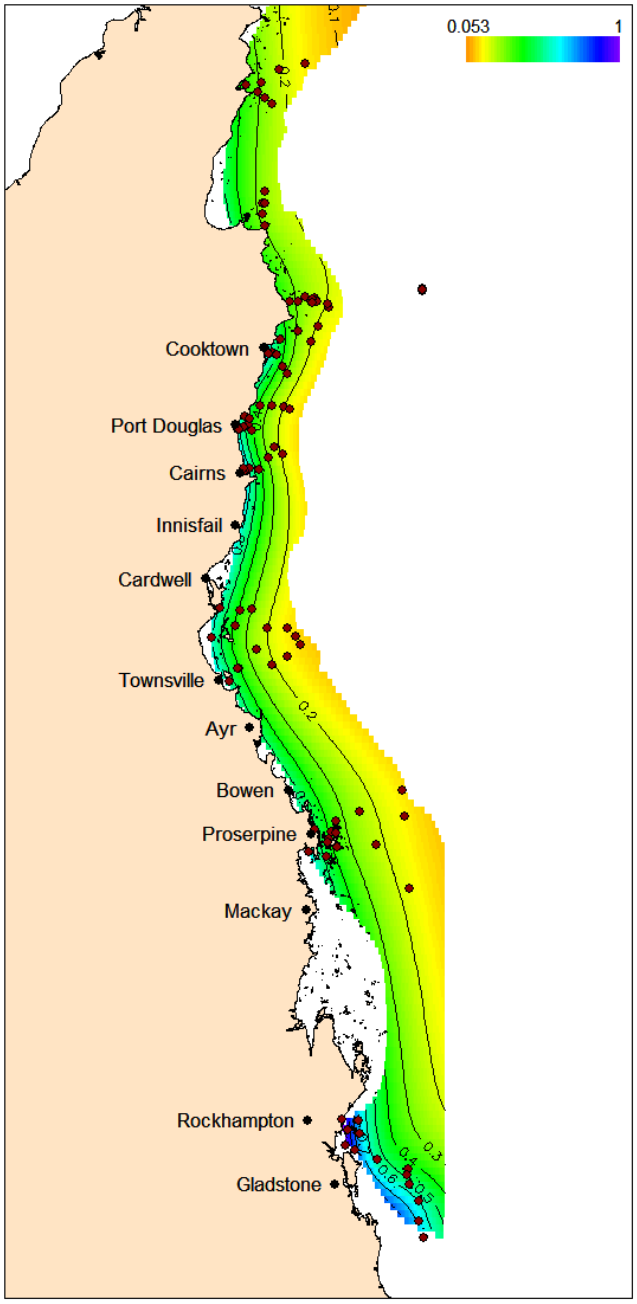


Figure 4.13. Locations of sites sampled 1993 to 2006 (note not all sites were sampled throughout the period) and spatially smoothed chlorophyll values averaged for each site over time. This representation is descriptive since it does not adjust for imbalances in sampling over time and sites. Strong cross-shelf trends are suggested with higher values in the inshore regions.

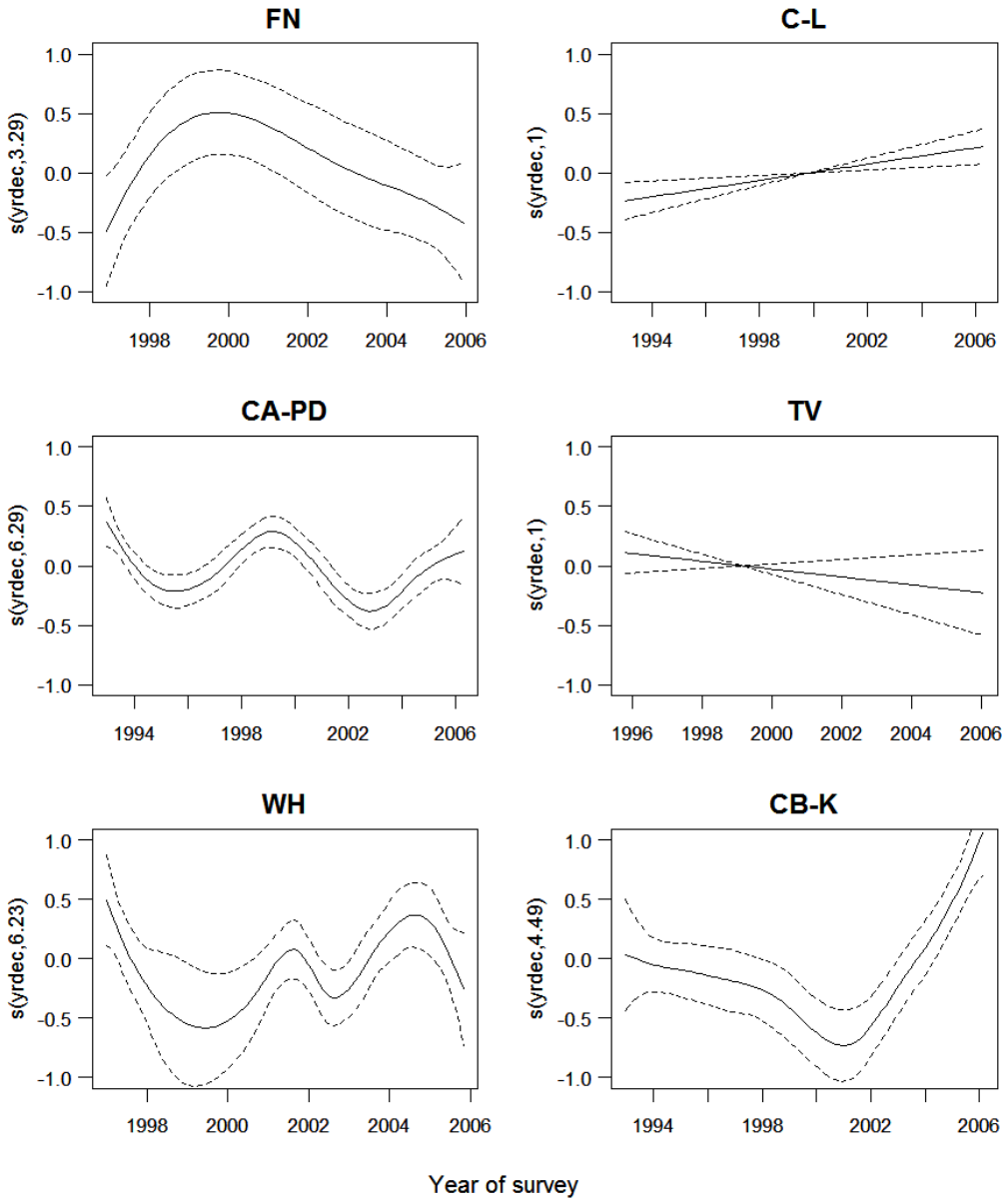


Figure 4.14. Estimated trend effects of chlorophyll concentrations in the Great Barrier Reef lagoon from 1992-2006. These partial effects are presented on a log2 scale, and thus a change of one unit represents a halving or doubling of chlorophyll. The bands indicate 95% confidence intervals. GBR regions: Far Northern (FN), Cooktown to Lizard Island (C-L), Port Douglas to Cairns (CA-PD), Townsville (TV), Whitsunday (WH) and Keppels to Capricorn Bunker reefs (CB-K).

Table 4.7. Approximate significance of smooth terms of partial effects represented in Figures 4.14, A1 4.1 and A1 4.2. Variation has been partitioned for each region by year [s(yrdec)] and location across the shelf [s(across)], and for all regions and shelf positions by sampling month for seasonal effects [s(Months)].

	df	F	p-value
Far Northern			
s(yrdec)	3.29	2.474	0.0252
s(across)	1.00	0.008	0.9295
Cooktown to Lizard Island			
s(yrdec)	1.00	8.880	0.0029
s(across)	1.00	13.58	<0.0001
Port Douglas to Cairns			
s(yrdec)	6.29	7.908	<0.0001
s(across)	2.95	7.520	<0.0001
Townsville			
s(yrdec)	1.00	1.571	0.211
s(across)	2.91	10.80	<0.0001
Whitsundays			
s(yrdec)	6.23	3.731	0.0002
s(across)	2.79	2.694	0.0154
Keppels to Capricorn Bunker			
s(yrdec)	4.49	8.111	<0.0001
s(across)	2.49	3.823	0.0019
s(Months)	2.01	29.36	<0.0001

Water Quality Monitoring using Remote Sensing

(Attachment B Task 2.2, 2.3)

This subtask provides satellite based spatial and temporal information on near-surface concentrations of chlorophyll (Chl) and suspended solids (SS), and on vertical attenuation of diffuse downwelling light coefficients (K_d) in lagoonal and coastal waters of the GBRWHA (see Figure 4.21). To achieve this goal MODerate resolution Imaging Spectrometer (MODIS) satellite imagery data was acquired, processed, archived, validated and interpreted. The work presented is focused on the Reef Plan MMP project deliverables but some components have been funded through the Fitzroy Agricultural Contaminants Remote Sensing Project of the Coastal CRC or the CSIRO Wealth from Oceans Biogeochemistry and Remote Sensing Project.

We present here an introduction to remote sensing of coastal waters in Queensland and the Great Barrier Reef, based on Oubelkheir *et al.* (2006). Coastal systems such as the waters of the Great Barrier Reef lagoon are dynamic and complex, acting as boundary zones for large scale reef to oceanic circulation and tides as well as receiving inputs of material from the land and the reef. Within the coastal zone, land-derived material undergoes a range of transformations including flocculation, aggregation-disaggregation, biological uptake, diagenesis, and photochemical processes before eventually reaching the open ocean, or being deposited. Thus, both particulate and dissolved materials occurring in coastal waters are highly diverse. The impact of anthropogenic activities on coastal ecosystems is not always well known. Consequently, there is a clear need for implementing research and monitoring programs for proper management of near-shore coastal areas. The recent development of ocean colour remote sensing techniques for synoptic mapping of natural waters provides a powerful tool for monitoring coastal areas. The surface spectral reflectance used to quantify ocean color, an apparent optical property (AOP), is a function of the light conditions (geometrical and spectral structure of the light field) and of the inherent optical properties (IOPs) of the water. In turn, IOPs depend on the concentration and type of optically-significant constituents which absorb and scatter light including phytoplankton, non-algal particles (biogenic detritus, heterotrophic organisms and minerals) and colored dissolved organic material (CDOM). There is considerable interest in how changes in the characteristics of particulate and dissolved materials impact on seawater IOPs, and ultimately on AOPs, and in the relationships between IOPs and concentrations of biogeochemical stocks. A prerequisite for providing accurate maps of biogeochemical properties *in situ* relies on the improvement of the algorithms used to invert the coastal-ocean color signal measured at the satellite.

The development of multi- and hyperspectral optical instruments for profiling through the water column allows the characterization of AOPs and IOPs at vertical and temporal scales not previously accessible using discrete sampling strategies. These new instruments have particularly powerful potential applications in coastal system studies where biogeochemical and optical properties vary at shorter horizontal and vertical scales (patchiness), and temporal scales (as a result of tides in macro-tidal systems, waves, flood events, wind gusts or internal waves). Better defining these various scales of variability is critical also for optimizing sampling and modeling grids [Chang and Dickey, 2001; Chang *et al.*, 2002].

While there have been numerous *in situ* studies on the optical variability of oceanic waters [*Morel and Maritorena*, 2001 and references therein], there have been fewer studies of coastal

waters, which are more complex than oceanic waters. The latter have mainly focused on the continental shelf break, where a large fraction of land-derived material is transferred to the open ocean. Only recently results have been published on the optical variability of Southeastern Queensland waters (Phinn and Dekker, 2005), and Fitzroy Estuary and Port Curtis waters (Dekker and Phinn, 2005, Oubelkheir, 2006) examining the optical variability in shallower (< 30 m) near-shore coastal areas. This study has significantly enhanced the scope of this prior research to encompass all the waters from Cape York down to Port Curtis as well as from the estuaries to the outer reefs. The results presented here are unique and are template for further research and development that will continue in to the future.

For this monitoring subtask, daily satellite images were obtained from the MODIS sensor. The MODIS images are obtained every day at about 13:30. All the MODIS products presented here are based on these daily images but are combined into multi-temporal binned data products as much as possible to reduce the number of images. In consultation with GBRMPA/AIMS it was decided to deliver monthly median maps of Chl, SS and K_d as well as a dry season and a wet season median map for the GBRWHA. To keep both the size of MODIS satellite datasets to be processed reasonable and to provide image graphics with sufficient resolution, and to enable reporting for the NRM Regions, the GBRWHA was divided into 6 areas and the resulting products are produced for each of the six areas (note that the most southern area is not part of the GBRWHA). As the information products are geo-coded they can be reassembled into one entire file for the GBRWHA if so required.

Analysis of global algorithm products for selected waters in GBRWHA has demonstrated that the global MODIS algorithms in the SeaWiFS Data Analysis System (SEADAS) 4.8 may be invalid in near shore Great Barrier Reef lagoonal waters (based on previous work in the Fitzroy Estuary and the Mossman –Daintree: see Qin *et al.*(submitted)). The level of disagreement is at least twofold of the concentrations of Chl (above 2 $\mu g\ L^{-1}$), worse for SS and can run up to ten-fold or more at higher concentrations. Therefore CSIRO developed and implemented a different type of algorithm that can cope with the significant variability in the specific inherent optical properties of concentration specific light absorption and scattering encountered in these waters-the CSIRO Australian Regional MODIS Algorithm for the GBRWHA. The principles of these new algorithms (based on Singular Value Decomposition matrix inversion) have been published for other sensors (Brando and Dekker, 2003; Dekker *et al.*, 2004; Phinn and Dekker, 2005; Dekker and Phinn, 2005). These algorithms were adapted to MODIS and applied to one year of MODIS data for the GBRWHA covering the November 2004 to October 2005 period, thereby encompassing a full wet season and a full dry season.

In order to parameterise and validate these new algorithms, additional measurements were made of surface and water column apparent (reflectance) and inherent optical properties (light absorption and scattering) and associated concentrations of algal pigments, SS, and CDOM. The measurements were collected during the four AIMS Reef Plan MMP cruises between August 2005 and February 2006. Earlier campaigns carried out by CSIRO of relevance to the waters in the GBRWHA are also included in the parameterisation and validation datasets thus ensuring the most comprehensive dataset to parameterise the algorithms is applied. For detailed information about the *in situ* measurements and their analysis see Appendix 1 in Volume 2.

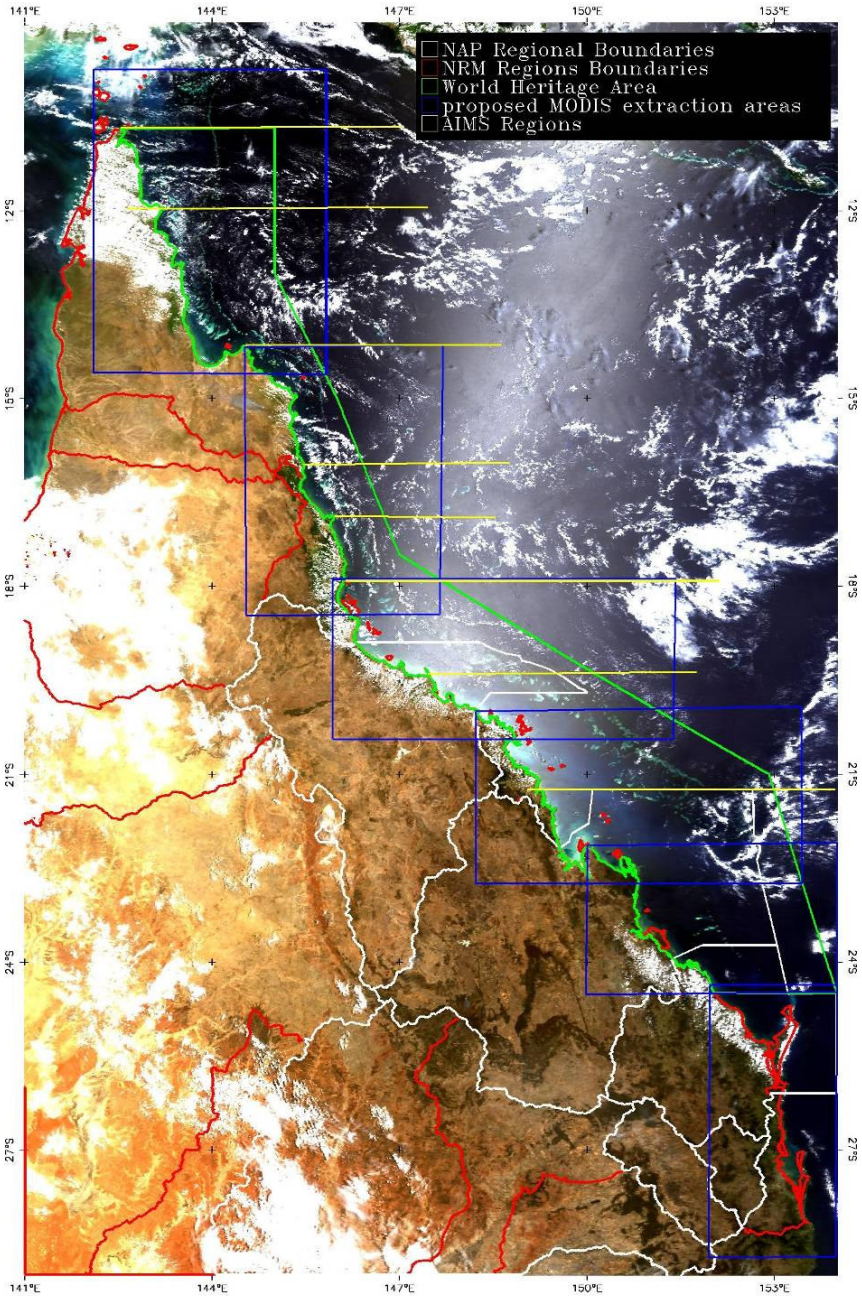


Figure 4.21. Overview of the various regions and boundaries. The MODIS Aqua data sections for image delivery are presented in the blue outlines from Cape York down to Port Curtis. The NAP (white), NRM (red) and AIMS (yellow) boundaries are added to enable referencing to those boundaries. The World Heritage Area (green outline) is the GBRWHA boundary for which the satellite data needed to be provided. See text for more detail.

Methods

Field and laboratory data

For quantitative assessment of optical water quality parameters retrieval from remote sensing imagery, it is necessary to use high quality *in situ* data to parameterise the underlying bio-optical model that inverts satellite measured reflectance to the desired water quality variables of Chl, SS and K_d. The measurement methodology for the *in situ* optical measurements is described in detail in Oubelkheir *et al.*, (2006)

The spectral reflectance R(λ) or normalised water leaving radiance nLw(λ) used to quantify the "water colour", an AOP, is a function of the geometric structure of the light field and of the optical properties of the medium itself. The optical properties are defined as the IOPs composed of the light absorption, scattering and backscattering coefficients of optically-significant constituents (in addition to pure water itself): phytoplankton, non-algal particles (biogenic detritus, heterotrophic organisms and sediments) and CDOM. Total suspended matter (SS) is the gravimetric sum of phytoplankton biomass and non-algal particles.

Discrete optical and bio-optical measurements

At each station, discrete samples were collected from Niskin bottles for the determination of in vitro phytoplankton pigments, SS, CDOM and particulate absorption coefficients. For phytoplankton pigments and particulate absorption, water samples were respectively filtered through 47mm and 25mm diameter GF/F glass-fibre filter (Whatman) and stored in liquid nitrogen until analysis. Glass bottles containing the water samples for CDOM analysis were kept in cool and dark conditions. For SS measurements, water samples were filtered through 47mm pre-weighted Millipore Durapore membrane filters, pre-ashed at 450°C and then pre-washed in 500ml of MilliQ water to remove friable fractions that can be dislodged during filtration. After filtration SS filters were then rinsed with distilled water to remove dissolved salts and stored flat in a Petri slide. All these samples were further analysed by L. Clementson from the CSIRO Division of Marine and Atmospheric Science Hobart for the September and August 2005 cruises and M. Slivkoff from AIMS for the January and February 2006 cruises.

In situ continuous optical measurements

Inherent optical properties (IOPs): A WETLabs ac-9 (www.**wetlabs**.com) spectral absorption meter was used for the acquisition of vertical profiles of total particulate matter and CDOM (instrument equipped with a 0.2 µm Pall Corporation filter) absorption and attenuation coefficients at 9 wavelengths (412, 440, 488, 510, 432, 555, 650, 676, 715nm} (Figure 4.22.a). A spectral backscattering meter HOBILabs Hydroscat 6 (HS-6) (www.**hobilabs**.com) was coupled to the ac-9 for *in situ* vertical profiling of light backscattering coefficients at 6 wavelengths {442, 488, 555, 589, 676, 852 nm} (Figure 4.22.b).

Apparent Optical Properties (AOPs): For the calculation of the vertical attenuation coefficient Kd(λ), the subsurface reflectance R(0-, λ) and the bottom reflectance, upwelling radiance and downwelling irradiance were measured through the water column from two hyperspectral Ramses spectroradiometers (manufactured TriOS http://www.**trios**.de/, spectral measurement intervals: 3.3 nm) and the above-water downwelling irradiance (sun and sky light) was measured using a deck Ramses downwelling irradiance sensor, as free as possible from ship's structure shading or reflection effects (Figure 4.22.c).

Together with these *in situ* measurements, a set of ancillary data and information on weather conditions were recorded (including sky and water digital photos) for subsequent image correction procedures.

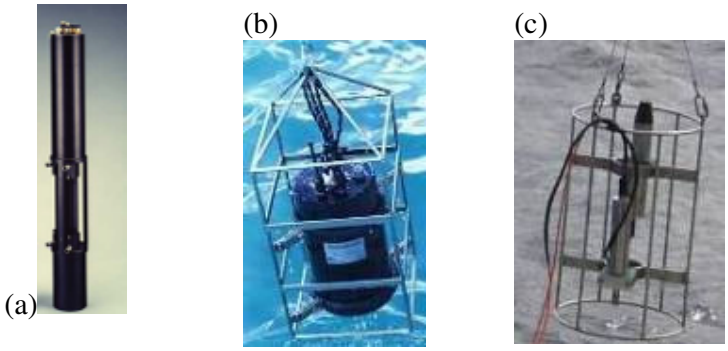


Figure 4.22 Optical instruments deployed for vertical in situ continuous sampling during field campaigns ((a) ac-9, (b) HS-6, (c) TriOS/Ramses)

Phytoplankton pigments: Phytoplankton pigments were analysed by HPLC following the method of Wright *et al.*, (1991). Please note that the data used for validating the remote sensing data results for chlorophyll are based on the fluorometric method (applied at AIMS). Thus, the method used to parameterise chlorophyll in the remote sensing bio-optical model is different to the methods used to obtain chlorophyll data in the long term Chl monitoring data set.

SS: SS filters were weighted on a high precision micro-balance.

CDOM: Water samples were filtered through 0.22µm polycarbonate filter (Millipore) into a 10-cm path length quartz cell. Optical densities were then determined using a GBC 916 UV/VIS spectrophotometer and baseline corrected optical densities were multiplied by the ratio of 2.3 to the cuvette path length for each wavelength to calculate the spectral absorption coefficient of CDOM (acdom(λ)). CDOM spectra were then fitted to an exponential function over the range 350-680nm.

Particulate absorption coefficients: Optical densities (OD) of total particulate material retained on GF/F filters and detrital matter after pigment extraction using the method of Kishino *et al.*, (1985) were determined by spectrophotometry from 350 to 680 nm. The spectral absorption coefficient of phytoplankton was calculated from the difference between the OD of the total particulate and non-algal fractions. The optical density scans of total particulate and detrital material were converted to absorption spectra by first normalizing the scans to zero at 850 nm and then correcting for the path length amplification using the coefficients of Mitchell (1990).

In situ optical profiles of absorption, attenuation and backscattering coefficients: The ac-9 has been calibrated at CLW prior to each field campaign following the method described by WETLabs. Ac-9 data of spectral attenuation and absorption coefficients of total and dissolved material were first corrected for *in situ* temperature and salinity effects using the CTD data according to Pegau *et al.*, (1997) and then for incomplete recovery of the scattered light in the ac-9's absorption tube by using the proportional method described in Zaneveld *et al.*, (1994). As per the ac-9, the HS-6 was calibrated at CLW prior to each campaign using the calibration system provided by HOBILabs: the signal response is measured through the sample volume

over a Lambertian reflective (TeflonTM) plaque (Maffione and Dana, 1997). For an accurate estimation of the backscattering coefficient ($b_b(\lambda)$), a correction for incomplete recovery of backscattered light (i.e., sigma correction –polynomial–, (Maffione and Dana, 1997) was applied using the absorption and attenuation coefficients measured *in situ* simultaneously by the ac-9.

Computation of $K_d(\lambda)$, $R(0-, \lambda)$: As mentioned in the previous section, in-water downward irradiance [$Ed(z, \lambda)$], upward nadir radiance [$Lu(z, \lambda)$] profiles as well as above-water downward irradiance [$Ed(0+, \lambda)$] were measured by the TriOS/Ramses sensors. For the estimation of the below-water reflectance $R(0-, \lambda)$, the in-water profile of $Lu(z, \lambda)$ is extrapolated to the sea surface to estimate $Lu(0-, \lambda)$. K_d is calculated as the negative logarithm of the attenuation of light with depth.

Remote sensing data

The MODIS sensor on board of the Aqua satellite platform with a overpass time of 13:30 images a swath of 2300 pixels of 1 km resulting in a swath of 2300 km per overpass. MODIS circles the earth in 100 minutes. Thus an image over Australia is built up in a few minutes.

As it was required to process all daily images for the GBRWHA acquired during a full year (wet and dry season, November 2004 to October 2005), we divided the GBRWHA region (see Figure 4.21) into five areas to obtain i) a manageable data file size of approximately one gigabyte for applying the algorithm and ii) image maps that still had sufficient resolution to be displayed on an A4 paper size.

In a MODIS image each pixel represents a spectrum (of 7 spectral bands in the visible light). In order to create an image of for example of Chl it is necessary to translate each spectrum (and thus each pixel) into a Chl concentration. When these pixels are subsequently mapped into an image a two-dimensional image of the chlorophyll concentration is given. A prerequisite for the accurate inversion of spectra into biogeochemical quantities (e.g., concentrations of Chl, SS and CDOM) or physical quantities (e.g. K_d, turbidity or Secchi Disk transparency) relies on an estimation of the relationship between constituent concentrations and the IOPs of light absorption and light (back)-scattering. The essential parameters are the specific inherent optical properties (SIOPs), i.e., IOPs normalised by the constituent concentration. Once the SIOPs are established it is possible to generate any spectra that are a combination of naturally occurring concentrations of Chl, SS and CDOM. This family of representative spectra can then be matched to actually measured MODIS spectra or be used to invert each MODIS pixel (spectrum) using the new inversion algorithms (CARMA-GBRWHA) that are based on water-leaving radiances in the MODIS spectral bands. The algorithm estimates simultaneously the concentration of Chl, SS and CDOM as well as calculates the vertical attenuation coefficient Kd. (If a bottom effect is visible they can, in principle, also estimate the bottom depth, for the 1km MODIS pixels, however, estimating bottom depth still needs to be proven for MODIS). This simultaneous estimation of concentration of Chl, SS and CDOM as well as calculates the vertical attenuation coefficient K_d makes this method much more reliable than the global algorithm approach by NASA as the resultant concentrations always together create a spectrum (colour of water) that is very close to reality. The main errors that may remain are errors due to incorrect correction for the atmosphere or sun and sky-glint of the satellite image or the presence of water types with an algal, non-algal particulate matter or coloured dissolved organic matter composition that is not adequately represented in our optical properties dataset.

As the waters in the GBRWHA vary from the very turbid river estuaries such as Fitzroy and Burdekin draining mostly agricultural land to dry to wet tropics estuaries draining sugarcane or rainforests such as the Proserpine, Pioneer and Daintree through to resuspended near coastal mud-laden waters through mixed waters into clear lagoon waters to reef waters to outer reef and oceanic waters significantly varying SIOPs were found during all our field sampling campaigns. Take as an example the Fitzroy Estuary waters: these contain fine grained mineral sediments originating from significant flow events of the Fitzroy River which are subsequently reworked through tidal action often having a transparency of 50 cm or less. These waters also contain CDOM from the catchment as well as from the mangroves in the estuary and from autochthonous production by algae. The algal composition is quite different from that found several kilometres towards the Coral Sea (see Oubelkheir *et al.*, 2006). Within the same MODIS image that covers the Fitzroy estuary oceanic waters occur with a transparency of 30 meters or more. These waters are dominated by picoplankton and their breakdown products. It is easy to see that the light absorption and light scattering characteristics of these water as a function of Chl and SS concentration will vary significantly. Thus an inversion of a reflection spectrum back to Chl or SS from a satellite image has to take these variations into account for each pixel in the image (as within each image the a priori distribution of water types is not known).The analysis of the entire GBRWHA *in situ* optical data set found that of all 82 complete and reliable samples 9 specific samples adequately represented all water types from Cape Tribulation down to the Fitzroy Estuary. Presumably these 9 samples also represent the waters from Cape Tribulation to Cape York (however these waters have not been sampled within the Reef Plan MMP). Thus the inversion method first calculates which of the 9 *in situ* samples is most representative for each MODIS pixel and only then the associated concentrations for Chl, SS and CDOM are calculated. From this information it is possible to calculate the correct K_d for each pixel. We also provide a map representing how close our calculated reflectance is to the measured satellite image reflectance, calculated as % reflectance difference; the smaller this value, the more reliable is the resultant map of Chl, SS and K_d. This reflectance closure (or error) map identifies those areas in each image where the results are less certain as they will show a higher discrepancy in modelled versus measured spectra. In future fieldwork these areas with poor spectral reflectance closure can be targeted for *in situ* sampling to try to enhance the dataset used for inversion. Thus the MODIS image results can be used for optimising future fieldwork for parameterising the MODIS remote sensing algorithm, till a point is reached where no improvements are likely – then no further fieldwork is required till anomalies are found in the reflectance closure maps. At this point remote sensing will have become entirely independent op *in situ* sampling. It is estimate that within 2 to 3 years this point may be reached in the GBRWHA.

The accuracy of the calculated normalised radiance leaving the water (i.e. the definition of ‘reflectance’ in MODIS) from a MODIS image is also dependent on the accuracy of the atmospheric correction. It is known that the standard atmospheric correction in SEADAS 4.8 is not accurate (especially in the blue region of the spectrum) in natural waters that reflect significantly above zero in the nearby infrared (as the nearby infrared is used in SEADAS 4.8 to estimate the aerosol contents). Initially it was intended to test and implement one out of two to three published SEADAS code adaptations that improve the atmospheric correction over highly reflecting waters. However a theoretical and applied study by Qin *et al*., (submitted) has demonstrated that the atmospheric correction, although not accurate, is not easy to improve and that most gain lies in improving the in-water algorithms. A sensitivity analysis proved that focus was required first on capturing the SIOP variability before concentrating on the atmospheric correction in this first year of this Subtask. It is

recommended to pay more attention to atmospheric correction improvement in the next year(s) of this Subtask. If possible in conjunction with the Wealth from Oceans Remote Sensing Project where atmospheric correction for all of Australia will be further addressed and improved.

Within this RWQPP Subtask project a strict selection rule was applied for those MODIS pixels failing in the atmospheric correction: they have all been blanked out in the final products as grey pixels. Many of these occur in the very near coastal areas and estuaries. Clouds and cloud shadows have also been blanked out. The coral reef vectors (provided by GBRMPA) have been overlayed as white pixels in the final products.

Results

The results for the six reporting regions (the blue boxes in Figure. 4.21) are presented as wet and dry season log-median maps of chlorophyll (Figures 4.23.a and b), total suspended matter (as non algal particulate matter, Figures 4.23 c and d) and the vertical attenuation coefficient of light (Figures 4.23 e and f). Also presented are maps for these dry and wet periods that present the number of valid pixels used for calculating the log-median values (Figures 4.23.g and h.). Here maps are presented only for the Fitzroy Estuary Keppel Bay region. The other regions (Cape York, Cairns, Burdekin, and Mackay Whitsunday) are presented in the accompanying CDROM.

Guide to interpreting the maps:

All maps have a similar layout: land is presented as dark grey and the coastal boundary is based on a standard coastline vector. Main rivers are presented in blue lines. Coral reefs including a 1 km buffer zone (to avoid mixed land or reef and water pixels) are presented as white. The heavy red lines in the East represent the GBRWHA boundary. The two pink lines (one thin and one thick) represent a rough description of the 20 m bathymetry line and the 80 m bathymetry line. These lines are loosely based on a bathymetry vector map provided by GBRMPA. As one agreed deliverable was box plots showing the statistics of all the valid pixels per season per region, subdivided into the zero to 20 m bathymetry zone and the 20 m to 80 m bathymetry zone, some assumptions had to be made how to implement this requirement. For example, the 20 m and 80 m bathymetry vectors/contours may contain missing values as occurs at site 22 of the long term chlorophyll monitoring program in Keppel Bay located directly next to a rock island, which not visible on any bathymetry map. Also encountered were contours that with a narrow corridor suddenly extend from the subtidal zone into large areas far offshore. Therefore a rough contour was drawn that followed the first zero to 20 m bathymetry line but that necessarily skipped or joined to highly detailed features. The underlying idea for separating the satellite image results into the zero to 20 m and the 20 m to 80 m bathymetry was that re-suspension will occur often till about 20 m depth and not beyond.

The final information in the maps is the long term monitoring stations, presented as pink numbers. The images show that many of these stations are situated very near to islands or coral reefs, rendering them less suitable for remote sensing product validation as all international remote sensing product validation protocols stipulate that validation match-ups will only be accepted if they are at least 5 km away from the nearest exposed land mass or submerged but still visible substratum.

The maps in Figure 4.23.g and 4.23.h. depict the number of image pixels per pixel location available for calculating the log-median values for each season. The maps show that this amount varies from 2 to about 60 for each season for each pixel location. Theoretically about 180 images should be available for each season. The reason these pixel frequency values are less is because

the currently available archives at CSIRO still contain some MODIS Aqua data gaps over the period of 1st November 2004 till 31 October 2005 (most notably a gap between 1 November 2004 and 13 November 2004 –see the titles of Figures 4.23.a, c, e and g respectively). These gaps need filling in, which is time consuming as it entails matching thousands of file names with the same filenames at NASA to understand which files are missing. For the above period about 250 of possible 360 images were available. A lower number of reliable pixels in the coastal zone were available because of the stringent quality control criteria applied: e.g., pixels were flagged and omitted with cloud or cloud shadow, pixels where the atmospheric correction failed and pixels where the error between modelled and measured spectra was too high. Please note that these results are only for the MODIS Aqua sensor data with its nominal 13:30 overpass time. The other MODIS sensor on board of Terra has been operational since 2000 and passes the GBRWHA at 10:00 in the morning when cloud cover is less. We decided to initially work with Aqua data, because the sensor is better characterised and therefore it was the most suitable for us to begin with. In the future, incorporating MODIS Terra data and improving the atmospheric correction over near coastal muddy waters should increase the amount of reliable pixels per wet or dry season to over 120 per season, possibly even up to 200 per season.

Figures 4.23.g and h show more image pixels per pixel location in the dry season then in the wet season which will mainly be due to increased cloud cover during the wet season.

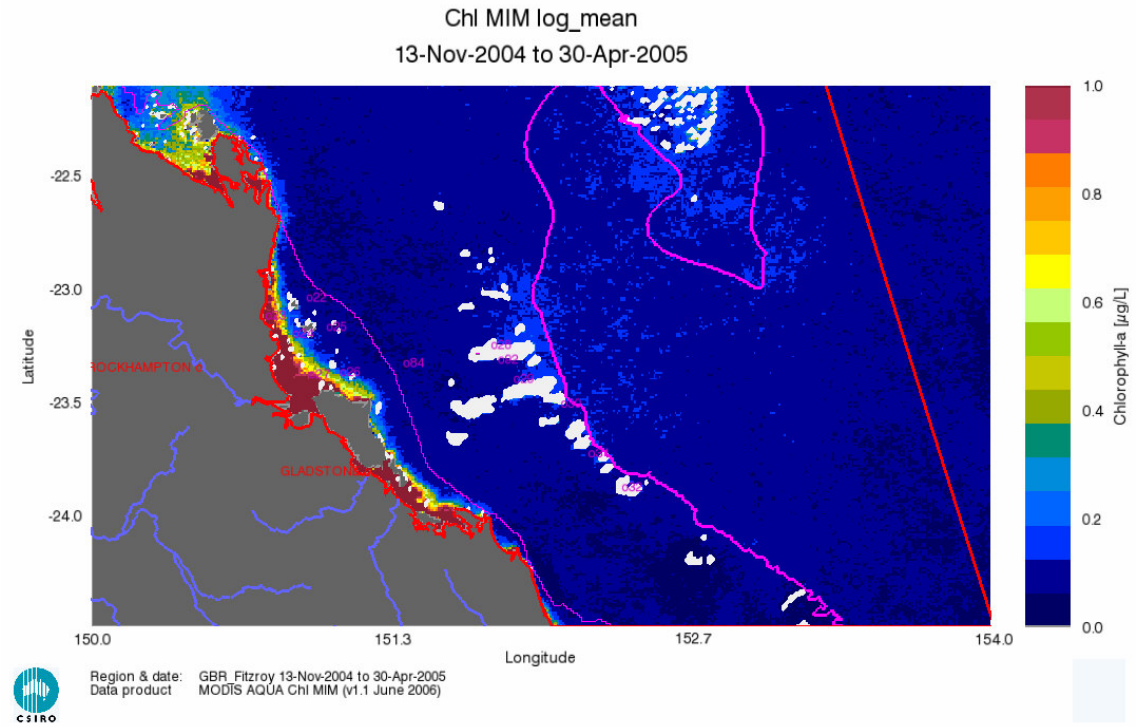
Irrespective of these potential improvements the current dataset is already much more data-rich then any *in situ* measurement programme: the boxplots presented in Figures 4.30 (a) to (l) show that *in situ* measurements are limited in scope and coverage of the actual concentration ranges. In essence, *in situ* programmes deliver 10 to a hundred spot samples in one of these regions, whereas this first satellite dataset already delivers hundreds of thousands of measurements in each square kilometre, covering virtually all occurring concentration ranges and events.

Wet and dry season maps

As an example of the wet and dry season log-median maps of chlorophyll the Fitzroy Estuary – Keppel Bay region is presented (Figures 4.23.a and b). The maps show high chlorophyll levels near the coast and in the estuary to lower concentrations towards the East. The zero to 20 m bathymetry contour is indeed the boundary for higher chlorophyll levels. In waters deeper than the 20 m bathymetry line chlorophyll concentrations are almost everywhere close to their minimum. Around the coral reefs a slight increase in Chl is visible at the scale chosen here to show chlorophyll concentrations. It would be possible to scale this image from zero to 0.5 $\mu g\ L^{-1}$ in which case the slight increase in chlorophyll around the reefs would be more visible. Figure 4.24 presents an example of the effect of scaling the images differently. The wet season log-median values near the coast show markedly higher values than the dry season log-median Chl for Shoalwater Bay, from the northern end of Keppel Bay to the Fitzroy Estuary and for Port Curtis. The higher Chl levels from the Fitzroy Estuary to Keppel Bay across the northern end of Curtis Island indicate an increased export of chlorophyll (and thus nutrients) to the south-east during the wet season.

The wet and dry season log-median maps of non-algal particulate matter (as a measure of total suspended matter) (Figures 4.23.c and d) for the Fitzroy Estuary –Keppel Bay region show similar gross patterns as for the chlorophyll distribution, although locally there are differences such as in towards the northeast of Shoalwater Bay where increased levels of non-algal particulate matter reach out further into the lagoon.

The wet and dry season log-median maps of vertical attenuation of light (Figures 4.23.e and f) for the Fitzroy Estuary –Keppel Bay region show similar gross patterns as for the chlorophyll and on-algal particulate matter distribution. Differences may be caused by the fact that coloured dissolved organic matter (CDOM) also contributes to the K_d, which is not presented here. The difference in dark blue to light blue colours between the wet and dry season for K_d is due to the K_d being slightly dependent on average sun-angles during the satellite overpass- the reason is that sun light coming in at higher slant angles during the winter months is scattered more in the first meters of the water column. Care must be taken in interpreting the Shoalwater Bay results as these waters have never been parameterised for the bio-optical model (and they are presumably very different from our previously sampled waters) and there may be a bottom visibility issue too.



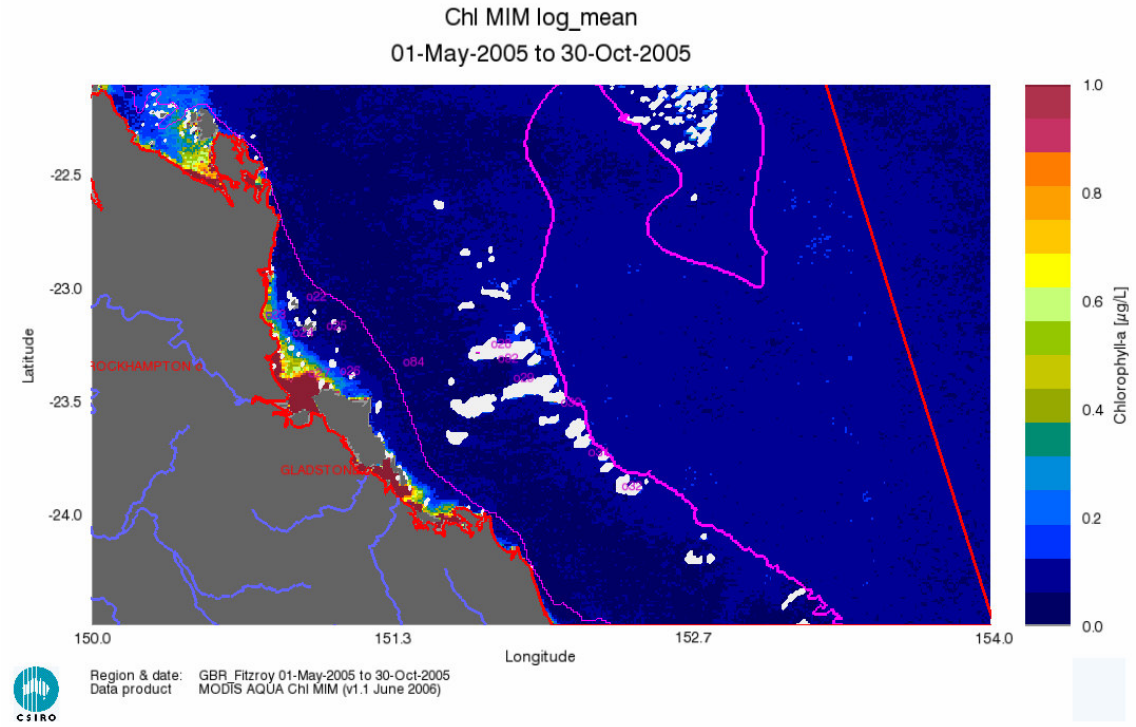
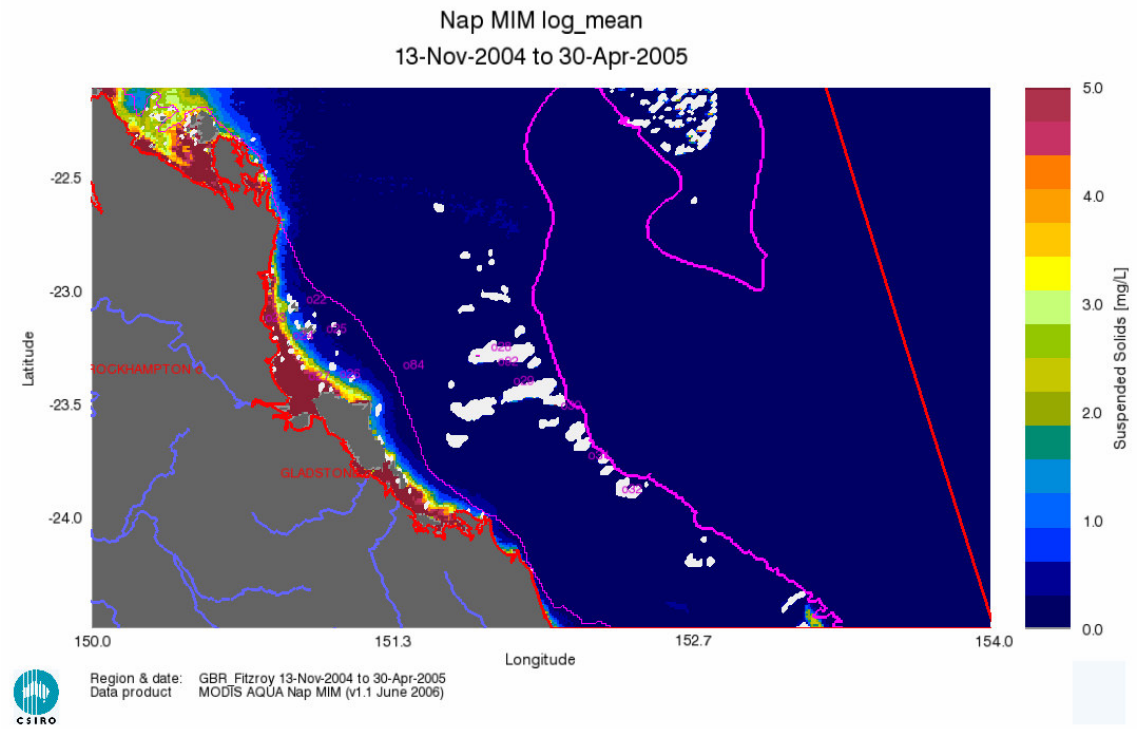


Figure 4.23 a (top) and 4.23 b (bottom). The chlorophyll log-median values for the wet and dry season (November 2004 to April 2005 and May 2005 to October 2005, respectively). See text for annotation explanation. The chlorophyll log median concentrations are higher in the Keppel Bay and Port Curtis Bays for the wet season. A slight increase in chlorophyll levels is visible around the coral reefs for the wet season.



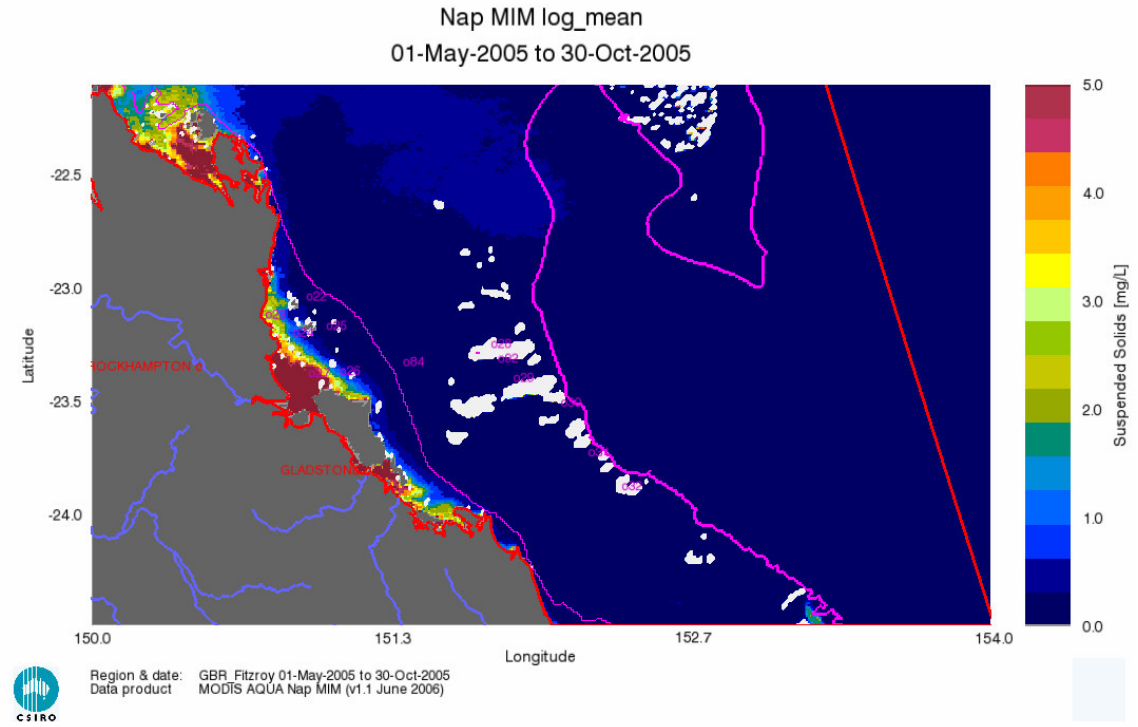
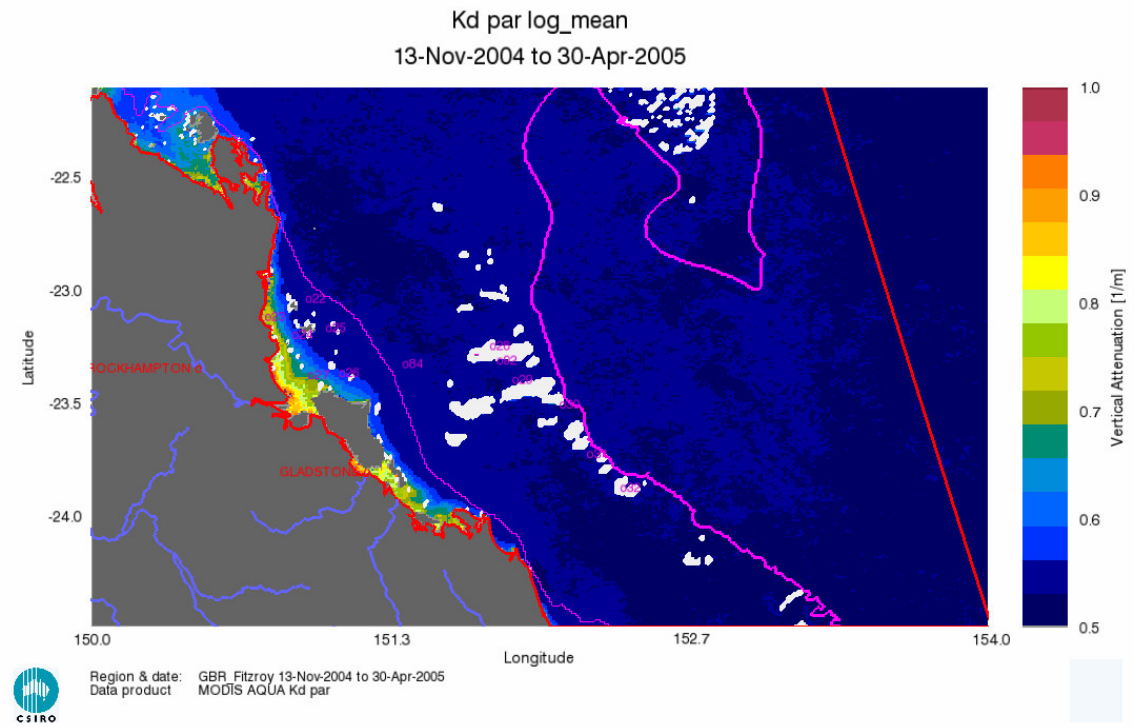


Figure 4.23 c (top) and 4.23 d (bottom). The non-algal particulate matter (Nap as a measure of SS) log-median values for the wet and dry season (November 2004 to April 2005 and May 2005 to October 2005, respectively). See text for annotation explanation. The Nap log median concentrations are higher in the near coastal areas of Shoalwater Bay, Keppel Bay and Port Curtis Bays for the wet season. A slight increase in Nap levels is visible in the northern part of this image between the Swains Reef and the coast.



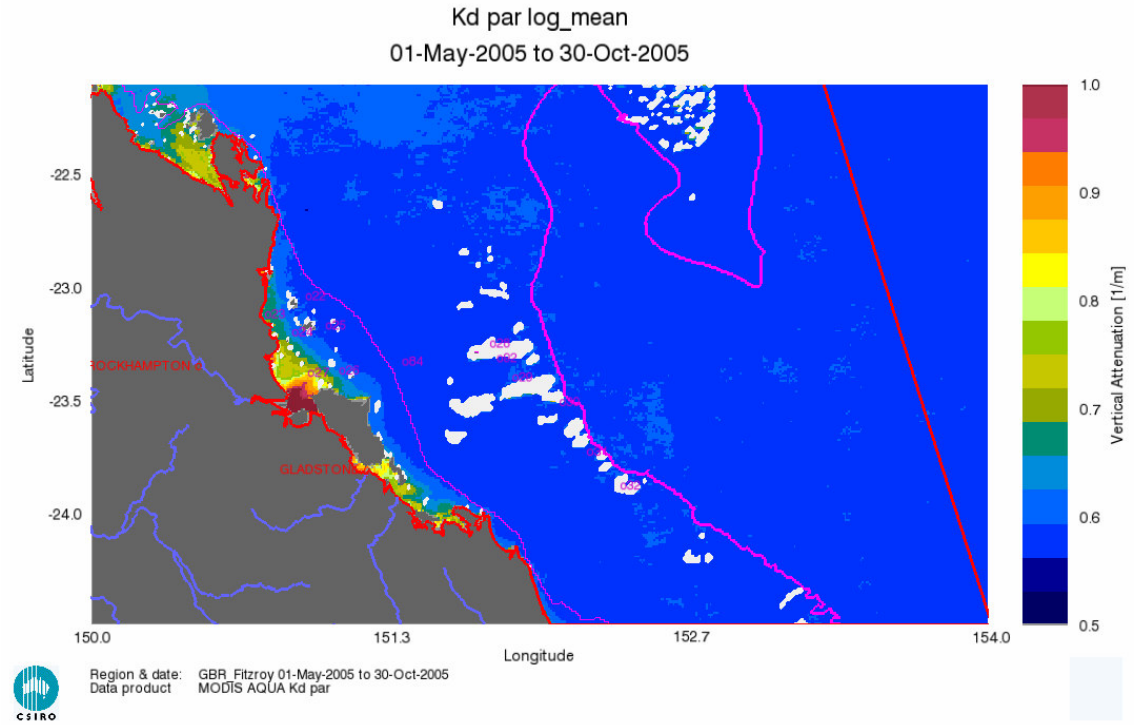
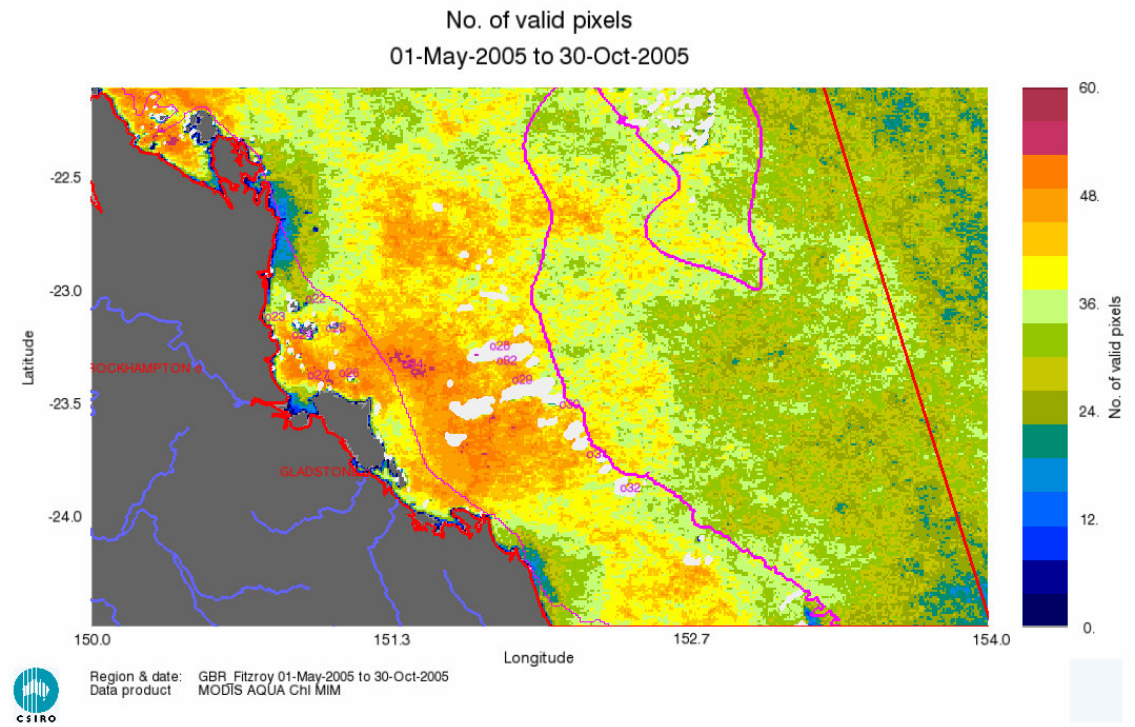


Figure 4.23 e (top) and 4.23 f (bottom). The vertical attenuation of light (K_d) log-median values for the wet and dry season (November 2004 to April 2005 and May 2005 to October 2005, respectively). See text for annotation explanation. The K_d log median values are higher overall for the dry season as the sun angles are lower causing a more diffuse underwater light field. The K_d values are slightly higher in the Keppel Bay and Shoalwater Bay area in the dry season.



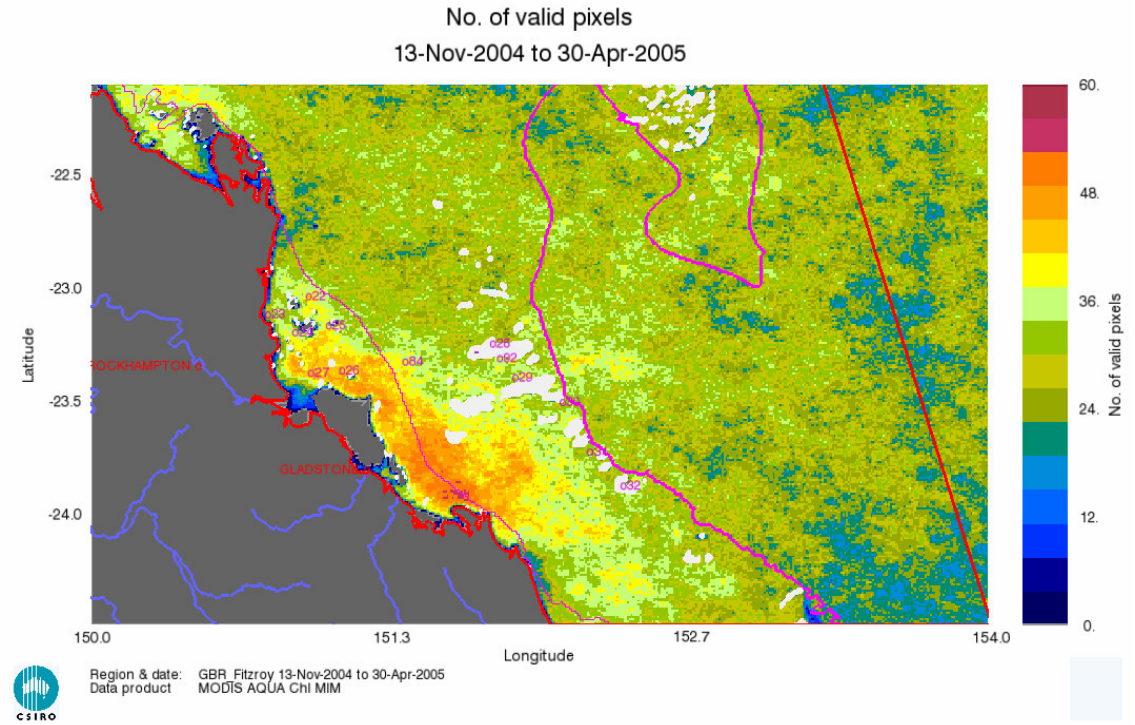


Figure 4.23 g (top) and 4.23 h (bottom). The spatial frequency distribution of pixels used for the log median calculations of the previous images for the wet and dry season (November 2004 to April 2005 and May 2005 to October 2005, respectively). See text for annotation explanation. The range varies from 60 pixels for a half year season to less than 2. Most frequencies are 24 to 48 pixels. These numbers are based on the currently available data from MODIS Aqua, if MODIS Terra data with its 10:30 overpass time were to be included these values would more than double. See text for further explanation.

Effects of scale

Figure 4.24 a, b, c, and d show the effect of changing the scale of the legend of non-algal particulate matter maps on the interpretation of the image. In:

- Figure 4.24.a the legend scales from zero to 1 mg L^{-1} in 16 (colour) steps
- Figure 4.24.b the legend scales from zero to 2 mg L^{-1} in 16 (colour) steps
- Figure 4.24.c the legend scales from zero to 5 mg L^{-1} in 16 (colour) steps
- Figure 4.24.d the legend scales from zero to 10 mg L^{-1} in 16 (colour) steps
- Figure 4.24.e the legend scales from zero to 20 mg L^{-1} in 16 (colour) steps

The information in the zero to one mg L^{-1} map focuses on the extent of the non-algal particulate matter (SS) from a minor flow event in the Fitzroy River into the Coral Sea. This effect clearly extends into the reef system of the Capricorn Bunker Group. The sediment plumes from Shoalwater Bay, Fitzroy Keppel Bay and Port Curtis Bay are all flowing south eastwards. At this scaling however no structural information on these near coastal waters is visible as the concentrations are generally above the 1 mg L^{-1} range.

The following four maps represent increasing ranges of non-algal particulate matter (SS) and clearly show that the information of the extent of the (diluted) sediment plume of the Fitzroy River-Estuary disappears in this representation, but more hydrodynamic features and flow patterns close to the coast become visible.

In a similar fashion chlorophyll maps intended to give an indication of the primary productivity of near coral reef waters would need to be scaled between zero to 0.5 µg L^{-1}, whereas maps intending to represent the increased near shore chlorophyll would need to be scaled to 2 µg L^{-1}. In the event of a phytoplankton, e.g. caused by a nutrient enrichment event, it may be necessary to scale to 20 µg L^{-1} Chl.

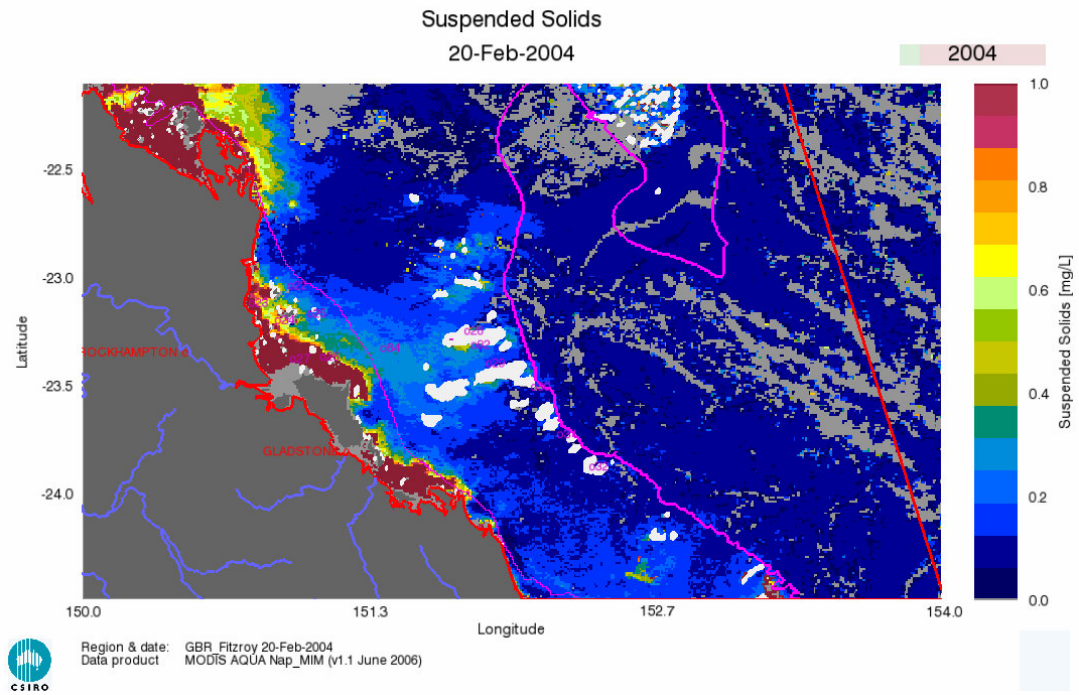


Figure 4.24.a. The legend for suspended solids (as non-algal particulate matter) scaled from zero to 1 mg L^{-1} in 16 (colour) steps. The Fitzroy River plume extent can be traced into the Capricorn Bunker group reef system. Near coastal concentrations saturate.

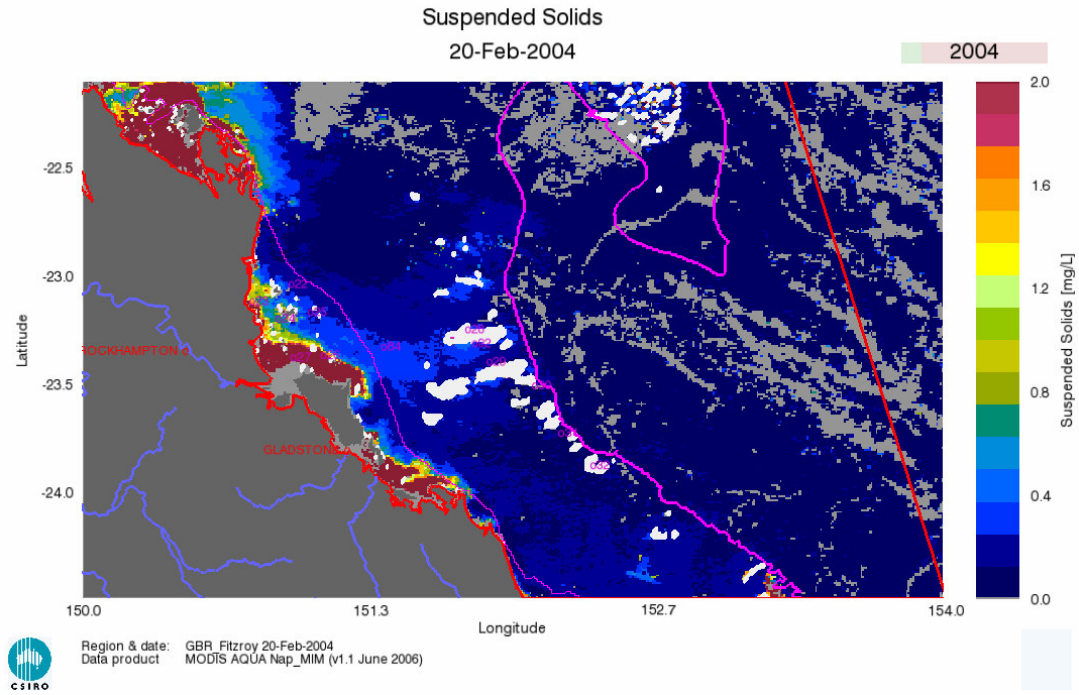


Figure 4.24.b. The legend for suspended solids (as non-algal particulate matter) scaled from zero to 2 mg L^{-1} in 16 (colour) steps. The Fitzroy River plume extent can still be just traced into the Capricorn Bunker group reef system. Near coastal concentrations saturate less.

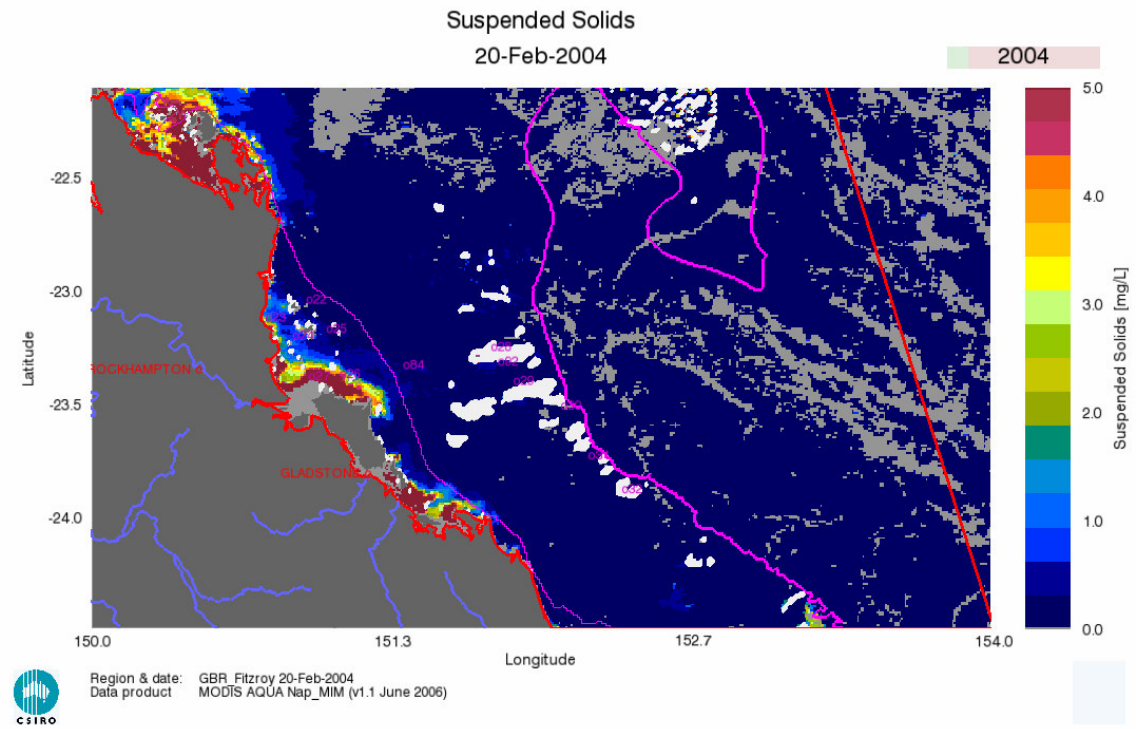


Figure 4.24.c. Legend for suspended solids (as non-algal particulate matter) scaled from zero to 5 mg L^{-1} in 16 (colour) steps. The Fitzroy River plume extent cannot be seen anymore. Near coastal concentrations show more patterns.

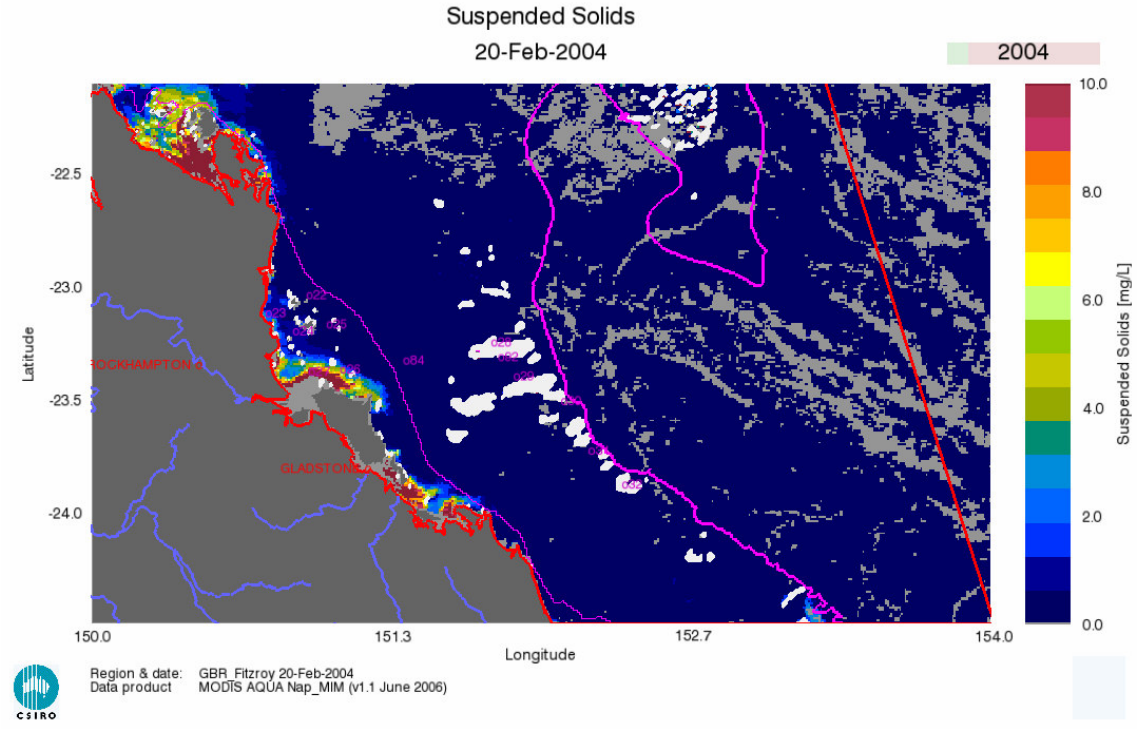


Figure 4.24.d. The legend for suspended solids (as non-algal particulate matter) scaled from zero to 10 mg L^{-1} in 16 (colour) steps. The Fitzroy River plume extent cannot be seen anymore. Near coastal concentrations show much more patterns.

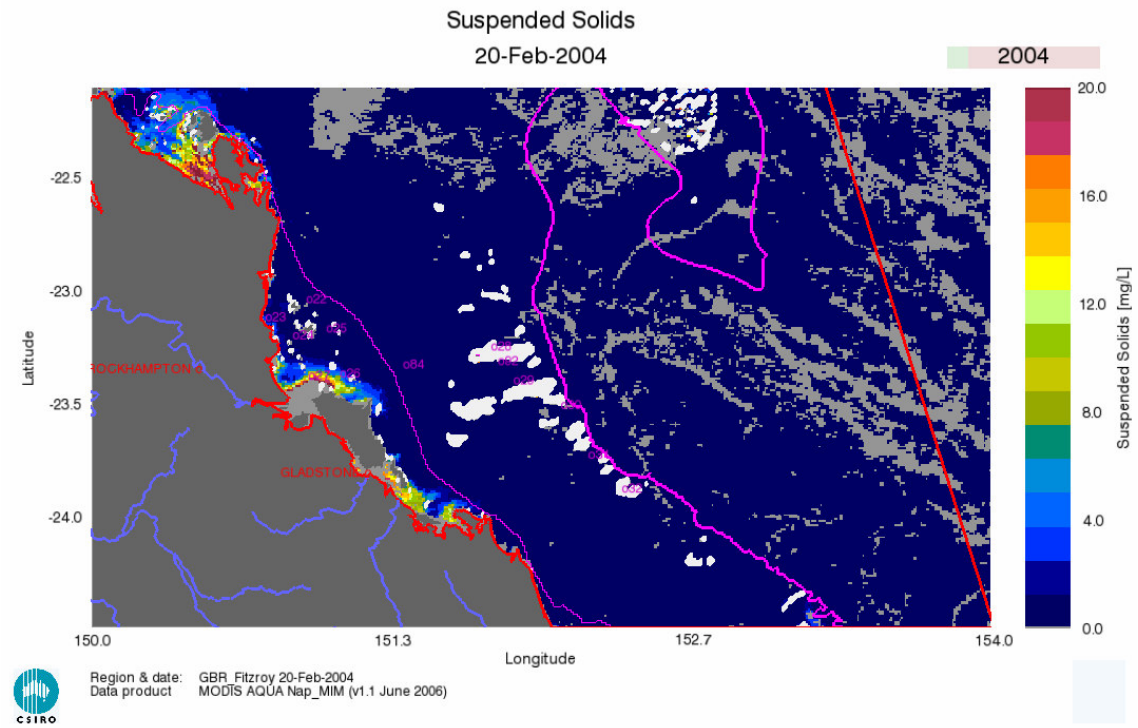


Figure 4.24.e. Legend for suspended solids (as non-algal particulate matter) scaled from zero to 20 mg L^{-1} in 16 (colour) steps. The Fitzroy River plume extent cannot be seen. Near coastal concentrations show most patterns.

Depicting remote sensing errors

Figure 4.25 shows the "optical closure" map for the same date as the Figures 4.24. "Optical closure" is the average error between the modelled spectrum that most closely measures the measured MODIS spectrum (defined as the root mean square over the MODIS Aqua spectral bands of the difference between the modelled and the measured spectrum). As this error becomes larger, the results for concentrations become less reliable as the bio-optical model can find no single combination of concentrations of Chl, SS and CDOM and SIOPs that correctly creates a spectrum close to the measured spectrum by MODIS. The reasons can lie in the model parameterisation (the water type is different from any present in our *in situ* bio-optical database) or it can be due to poor atmospheric correction (or any source of error) in the MODIS spectrum for that pixel or group of pixels. Figure 4.25 shows that the best match between modelled and measured spectra is in the lagoonal waters between the reef and the coast. Much poorer match is found in the far oceanic waters and then near coastal waters, specifically Shoalwater Bay. The *in situ* measurements used for parameterising the bio-optical model are mostly located in those waters where the closure is excellent. No *in situ* characterisation in the far oceanic waters exists, nor for Shoalwater Bay. Significant *in situ* data sets do exist for the parameterisation in the Fitzroy Estuary and Keppel Bay due to Coastal CRC remote sensing focus there in the past. This confirms that major reasons for higher error lies in either the poor atmospheric correction in very turbid waters or unavailability of proper water type optical characterisation in our bio-optical model.

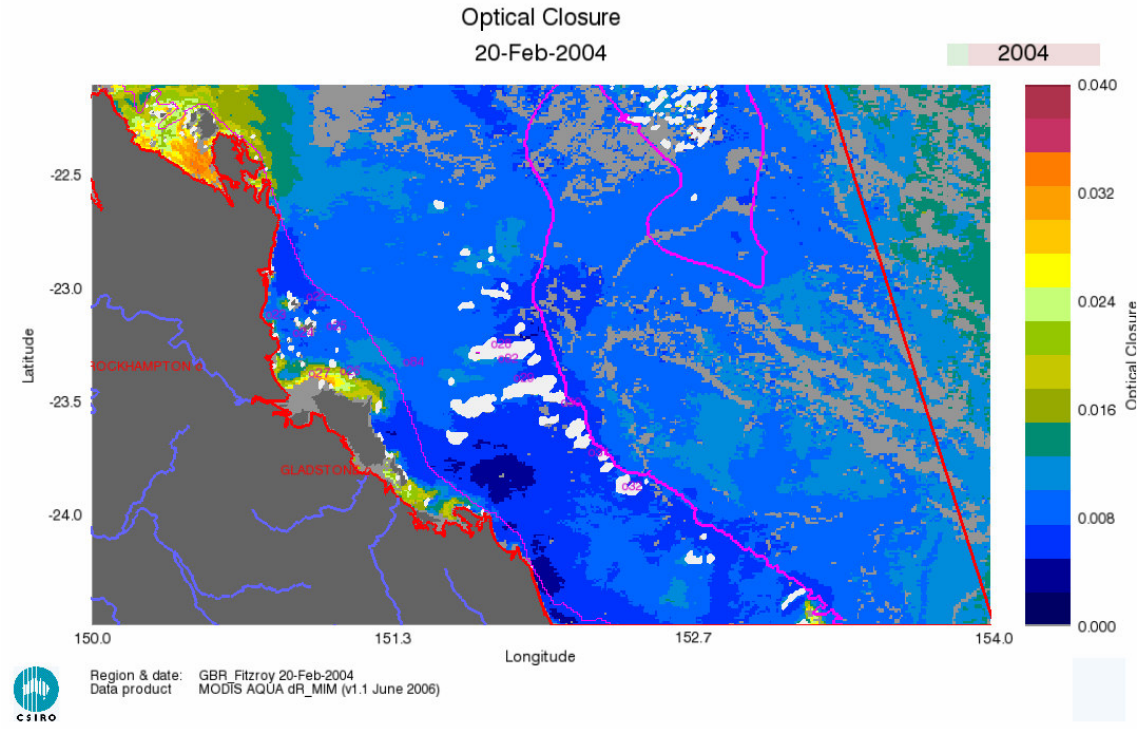


Figure 4.25. The "optical closure" map for the same date (20 February 2004) as Figure 4.24. High values indicate higher errors. From light green to red the errors become significant.

Matching data location and analysis: Keppel Bay case study

A more detailed analysis of using *in situ* data from the long term chlorophyll monitoring program for remote sensing product validation was carried out in the Keppel Bay area. Figure 4.26 shows the location of long term monitoring plan field sampling locations used for match-up analysis between remote sensing chlorophyll estimates and *in situ* chlorophyll estimates. Except for point 84 they are all located in the vicinity of islands or coral reefs making them less suitable for remote sensing product validation. Point 22 is located near a rocky island outcrop that does not exist in the bathymetry information provided to CSIRO.

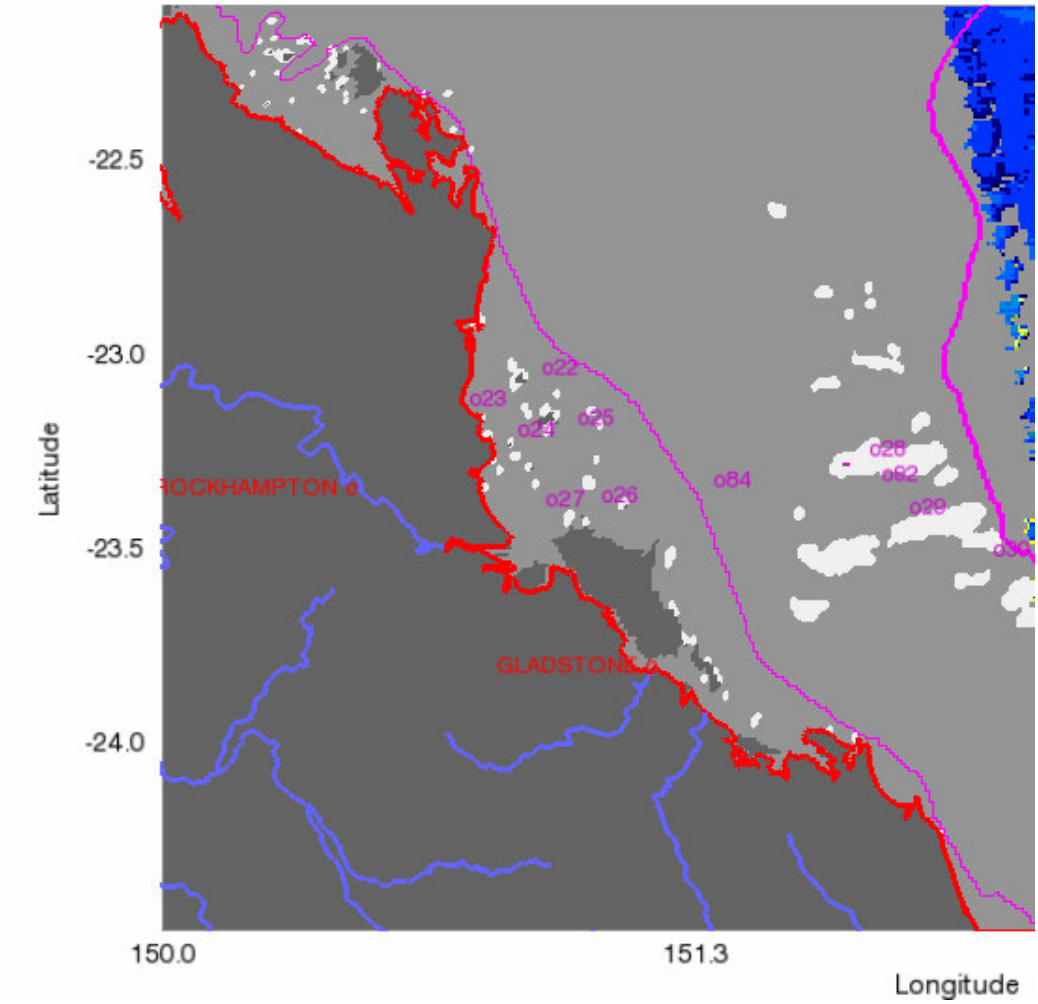


Figure 4.26. Locations of Long term chlorophyll monitoring stations n the Fitzroy Keppel Bay area.

Figures 4.27 a, b and c show the individual station match-up data for chlorophyll estimates from the long term monitoring plan and from the MODIS Aqua images from December 2002 till November 2005. The match-ups are calculated on a 3 by 3 pixel estimate surrounding the *in situ* sampling station, but with all reef vectors plus one kilometre buffer zone filtered out.

Figure 4.27.a shows the match-up for station 84 halfway between the Keppel Bay and the Capricorn Bunker reefs. The long term monitoring programme only has samples here till July 2003. There is a reasonable agreement between the two different datasets between December 2002 and the July 2003 at the higher level of chlorophyll measured (~0.15 to 0.40 µg L^{-1} chlorophyll). The remote sensing images also present a significant amount of lower chlorophyll values down to almost zero. Currently, values below about 0.04 µg L^{-1} chlorophyll are considered as below the verifiable detection limit as *in situ* samples cannot be determined with accuracy below those levels. The remote sensing algorithms do allow estimation below those values but the correctness cannot be validated at these low values. It is recommended to analyse this low chlorophyll estimation issue in future work. Figure 4.27.a does illustrate the strength of remote sensing data in data density (even on one pixel location) and in the fact that it measures everywhere geographically.

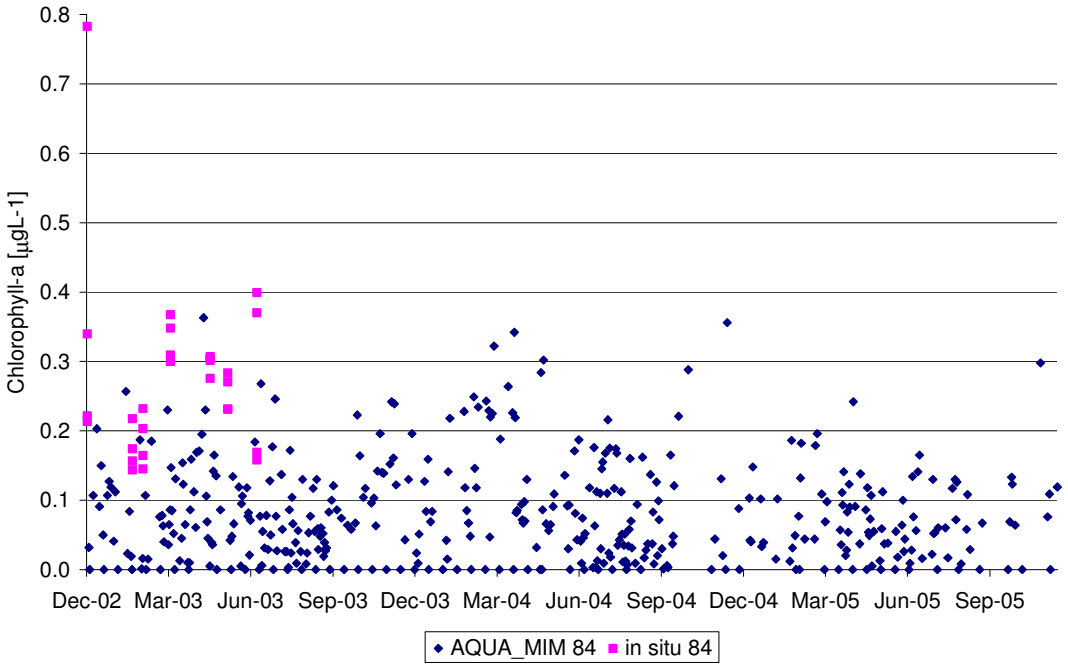


Figure 4.27 a. The match-up for station 84 halfway between the Keppel Bay and the Capricorn Bunker reefs. The individual station match-up data for chlorophyll estimates are from the long term monitoring plan compared to the MODIS Aqua images from December 2002 till November 2005. The match-ups are calculated on a 3 by 3 pixel estimate surrounding the in situ sampling station, but with all reef vectors plus one kilometre buffer zone filtered out.

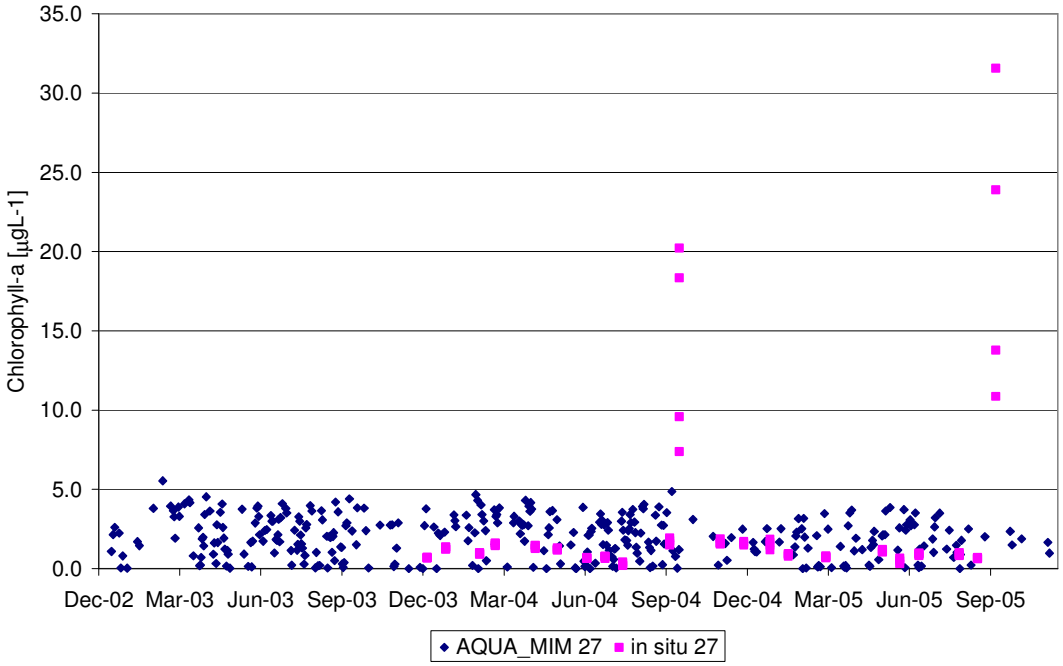


Figure 4.27 b. Match-up for station 27, just to the North of Curtis Island. The individual station match-up data for chlorophyll estimates are from the long term monitoring plan compared to the MODIS Aqua images from December 2002 till November 2005. The match-ups are calculated on a 3 by 3 pixel estimate surrounding the in situ sampling station, but with all reef vectors plus one kilometre buffer zone filtered out.

Figure 4.27.b shows the match-up relationship at site 27, just to the North of Curtis Island. Once again the advantage of the remote sensing data coverage and continuity is evident as this site had *in situ* samples only from December 2003 onwards. There is a reasonable concordance between the two different measurement methods, although the range of remote sensing measured concentrations is larger as is to be expected as it covers a 9 square kilometre region and has many more samples than the *in situ* dataset. Spikes in chlorophyll occur in the *in situ* data set in September 2004 and in October 2005 most likely due to *Trichodesmium* in the *in situ* samples. The remote sensing estimates do not show such high values indicating that the Trichodesmium may have been very patchy and only present locally. The actual effect of a Trichodesmium bloom on the MODIS retrieved chlorophyll signal requires further study.

Figure 4.27.c shows the match-up results for point 24 close to Keppel Island. There is a reasonable concordance between the two chlorophyll estimates; once again the remote sensing derived estimates show a larger range than the *in situ* data. The values in Figure 4.27.c show a seasonal cycle in the chlorophyll value for both measurement methods. Also in this match-up data set there is a substantial amount of remote sensing estimates at the very low range of chlorophyll requiring further study as to its veracity.

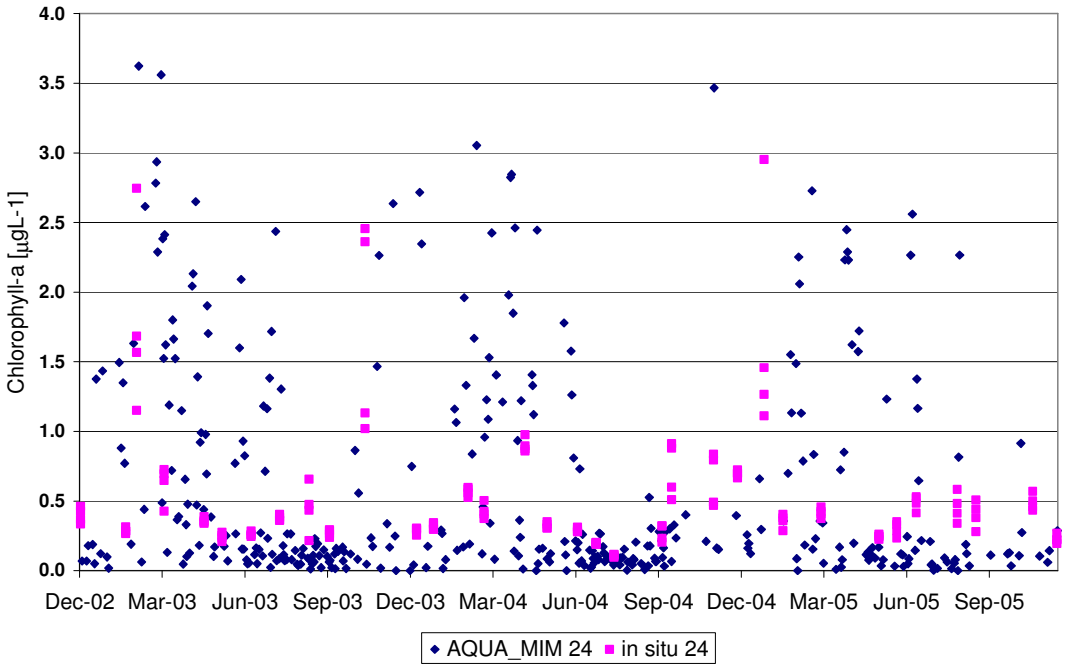


Figure 4.27 c. The match-up for station 24, close to Keppel Island scaled between zero and 4 µg L^{-1} chlorophyll. The individual station match-up data for chlorophyll estimates are from the long term monitoring plan compared to the MODIS Aqua images from Dec 2002 till November 2005. *The match-ups are calculated on a 3 by 3 pixel estimate surrounding the in situ sampling station, but with all reef vectors plus one kilometre buffer zone filtered out.*

Figure 4.28 shows a bar graph organised in steps of 0.2 µg L^{-1} (thus all values of zero to 0.2 µg L^{-1} are binned into the 0.2 bar, and so on) chlorophyll that contains all the match-up data from December 2002 till November 2005 of MODIS Aqua pixels and long term monitoring plan *in situ* measurements. The graph shows a similar trend in the two datasets but with noticeable variations. There are many more very low chlorophyll estimates from MODIS Aqua in the 0.2 µg L^{-1} chlorophyll range (see Figure. 4.28) of the chlorophyll scale than from the *in situ* measurements. This is reversed at the 0.4 µg L^{-1} chlorophyll level. At higher levels the number of match-ups decreases rapidly making it more difficult to assess the relationship. At concentrations of more than 2 µg L^{-1} chlorophyll there is no match-up between satellite and *in situ* values. As was mentioned previously the *in situ* data set is not well suited for the purposes of match-up comparison with remote sensing data from 1 by 1 km pixels (averaged to 3 by 3 km in order to reduce pixel to pixel variability and to capture the water mass sampled *in situ*). The overall similarity in chlorophyll ranges measured is promising, but more research is required to understand the exact nature of the relationship between these two very different types of chlorophyll estimates. Adding MODIS Terra data to this dataset will significantly enhance the number of match-ups. Figure 4.29 present the regression statistics for this match-up dataset and the correlation coefficient is 0.72 for a total number of 118 match-ups between long term monitoring plan *in situ* measurements and MODIS Aqua regional algorithm chlorophyll estimates. This assessment gives confidence that the remote sensing method is on the right track, especially considering all the issues involved in comparing these two independent datasets. In all likelihood further improvements possible within a year will enhance the confidence in the remote sensing estimates to the degree that remote sensing can become the prime detection and monitoring tool for chlorophyll, SS and K_d estimates in the GBRWHA.

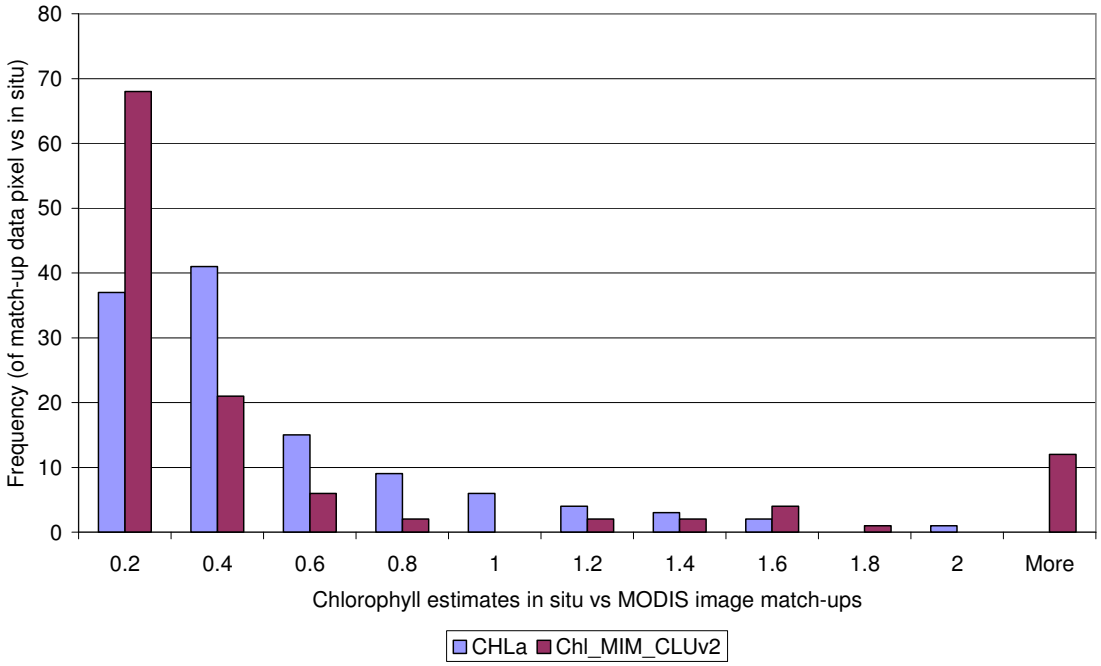


Figure 4.28. MODIS Aqua pixels and the long term monitoring plan in situ measurements. Bar graph organised in steps of 0.2 µg L^{-1} (thus all values of zero to 0.2 µg L^{-1} are binned into the 0.2 bar, and so on) chlorophyll that contains all the match-up data from December 2002 till November 2005 of MODIS Aqua pixels and the long term monitoring plan in situ measurements.

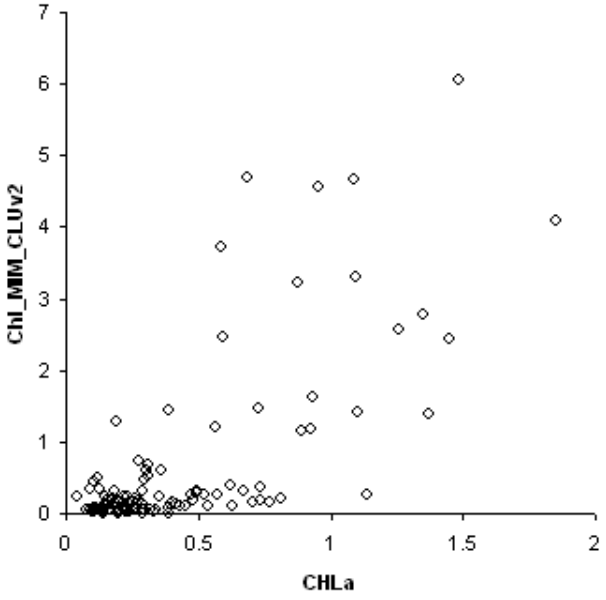
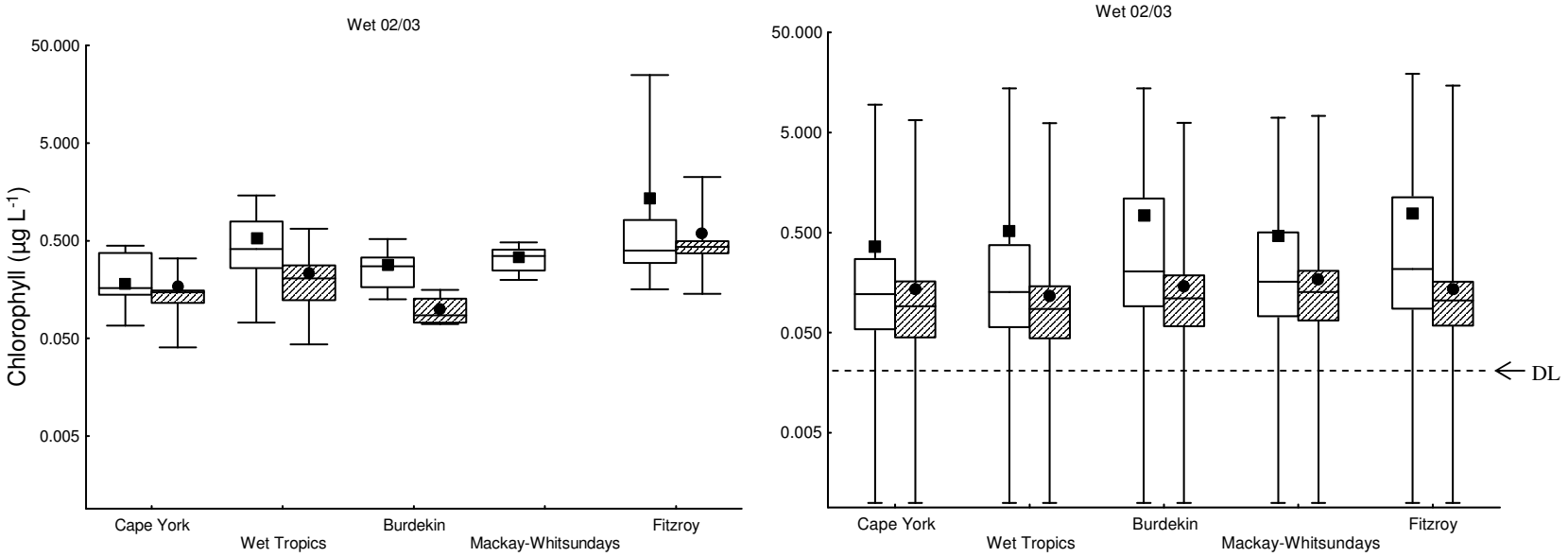


Figure 4.29. Scatter plot and correlation coefficient of match-up dataset. Number of matchups is 118, the Pearson correlation coefficient r is 0.72.

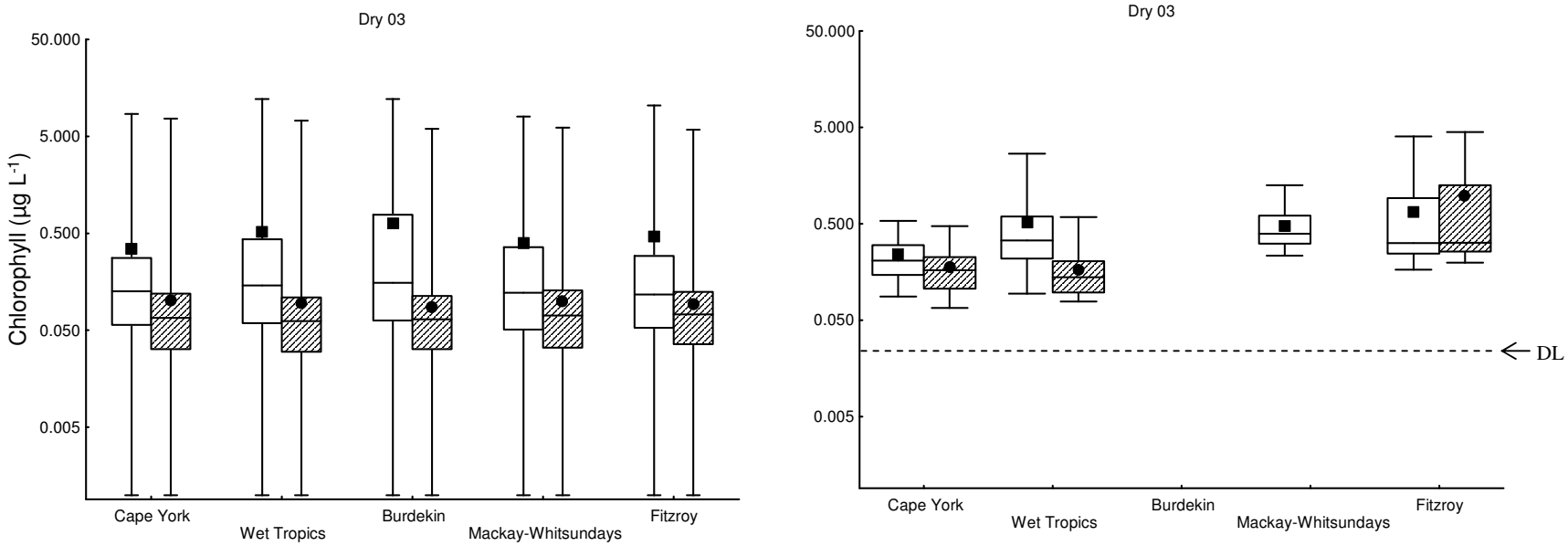
Figures 4.30 (a) through to (l) are box plots comparing the *in situ* data distributions of the long term monitoring plan with the remote sensing data distributions. The amount of data points varies from ten to a hundred for the *in situ* data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry area. Please note when comparing these plots that the spatial and temporal scales of the underlying datasets is vastly different between *in situ* data and remote sensing data. On a clear day one remote sensing image can already provide tens of thousands of measurements for a 20 km wide and 100 km long strip of coastal waters compared to 5 samples on a transect for boat based sampling. Under full overcast sky this reduces to zero versus 5 respectively. The remote sensing data set also includes pixels in shallow waters outside the reach of larger vessels. Also, the *in situ* sampling locations are by definition not randomly distributed (as they were chosen to be close to reefs, in calm water situations under normal conditions and/or based on historical site locations) whilst the distribution of pixels in remote sensing images is complete on clear days (full sampling), partially complete (but quite random) on partially clouded days or not at all available on overcast days. The differences in distribution between these two data sets indicate that conclusions relating or comparing these data sets to each other must be seen to be indicative.

The distribution of the *in situ* samples indicates that they are based on limited data (narrow range or missing values). The remote sensing data show that many pixels are estimated having very low chlorophyll values. These values are being annotated as being below a verifiable detection limit, as the threshold for reliable *in situ* chlorophyll measurements is ~0.04 $\mu g\ L^{-1}$ chlorophyll. The low chlorophyll value estimates from the remote sensing data needs further validation (note that the objective of the 2005-06 remote sensing monitoring was to focus on the near shore coastal waters). Specifically, attention should focus on the detection limits and accuracies of the various methods of chlorophyll detection as well as on the remote sensing algorithms, the atmospheric correction algorithms and the correct representative parameterisation of the bio-optical model.

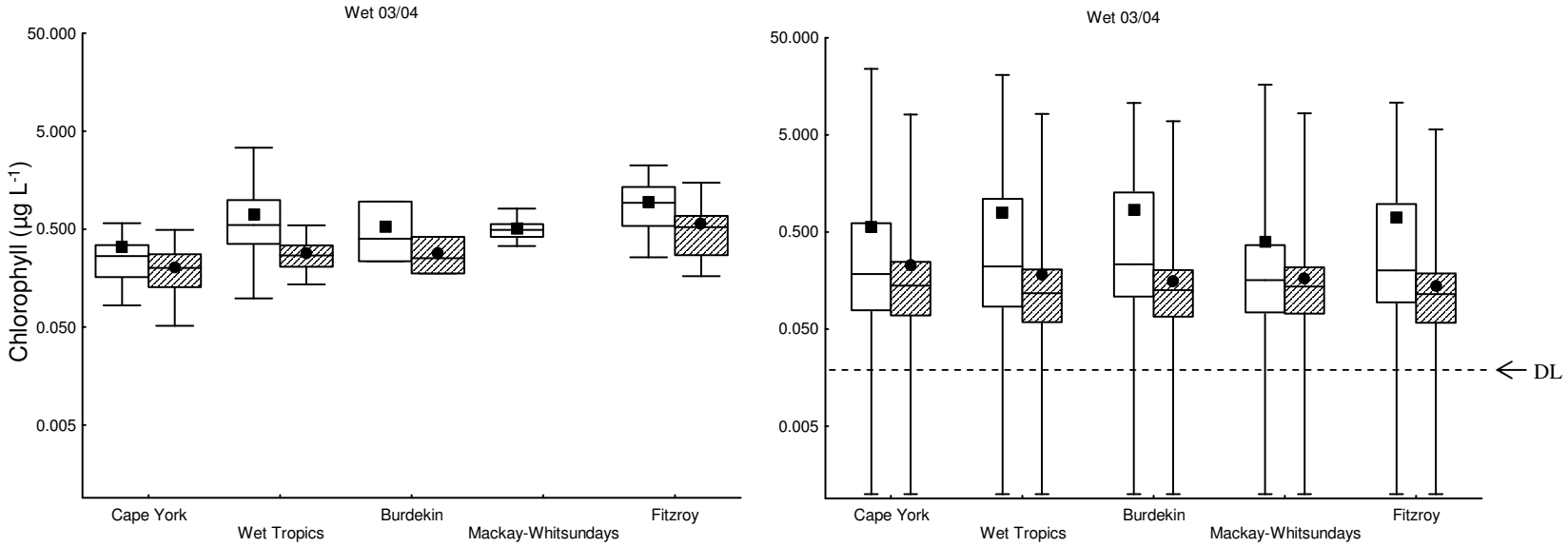
In general, the box plots show higher chlorophyll values expressed as medians and 25 to 75% percentiles for the *in situ* samples and the remote sensing estimates of the waters in the zero to 20 m bathymetry zone than for the waters in the 20 m to 80 m bathymetry zone. The ranges of the *in situ* samples generally fall within the ranges of the remotely sensed values. This is to be expected because the remote sensing data are based on (sometimes) full coverage of the surface waters over the zero to 20 or 20 to 80 m bathymetry zones. However, the medians and 25 to 75 percentiles do differ. For the Fitzroy region this is mostly likely to be due to chlorophyll concentrations of oceanic water sites being under-represented in the *in situ* dataset. The remote sensing data contain much more oceanic water type pixels than the *in situ* sampling stations.



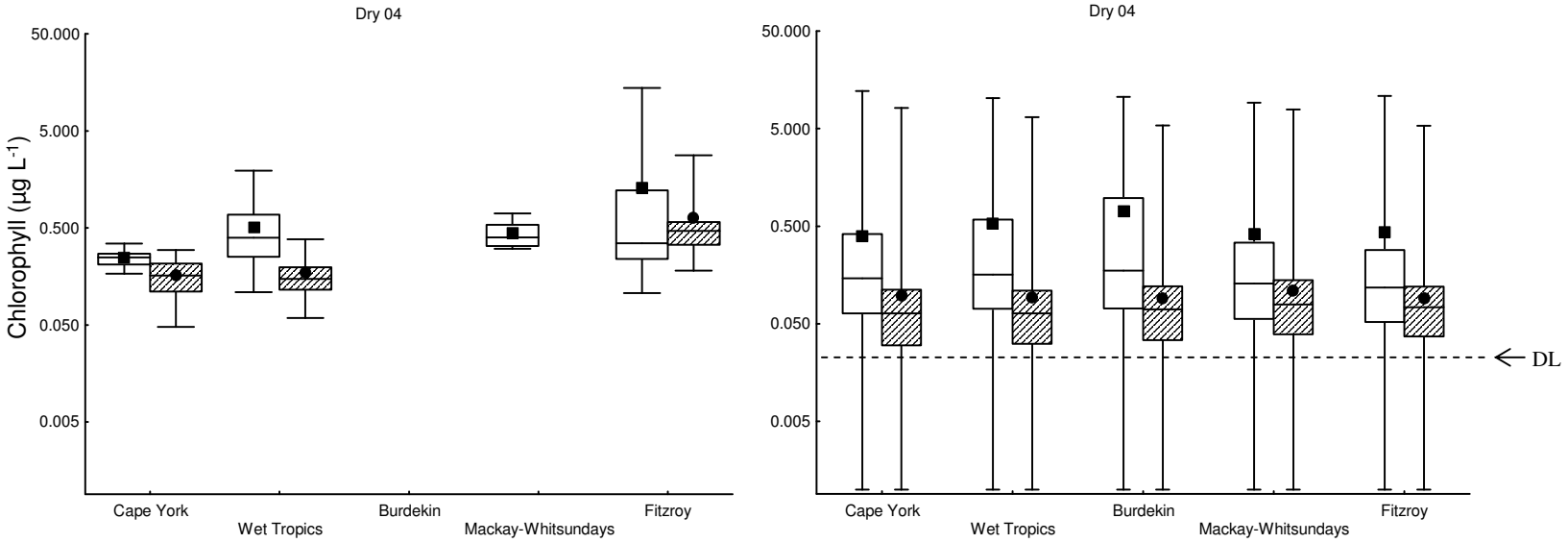
Figures 4.30 a (left) and 4.30b (right). Comparison of chlorophyll in situ data with remote sensing data for the 2002-2003 wet season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.



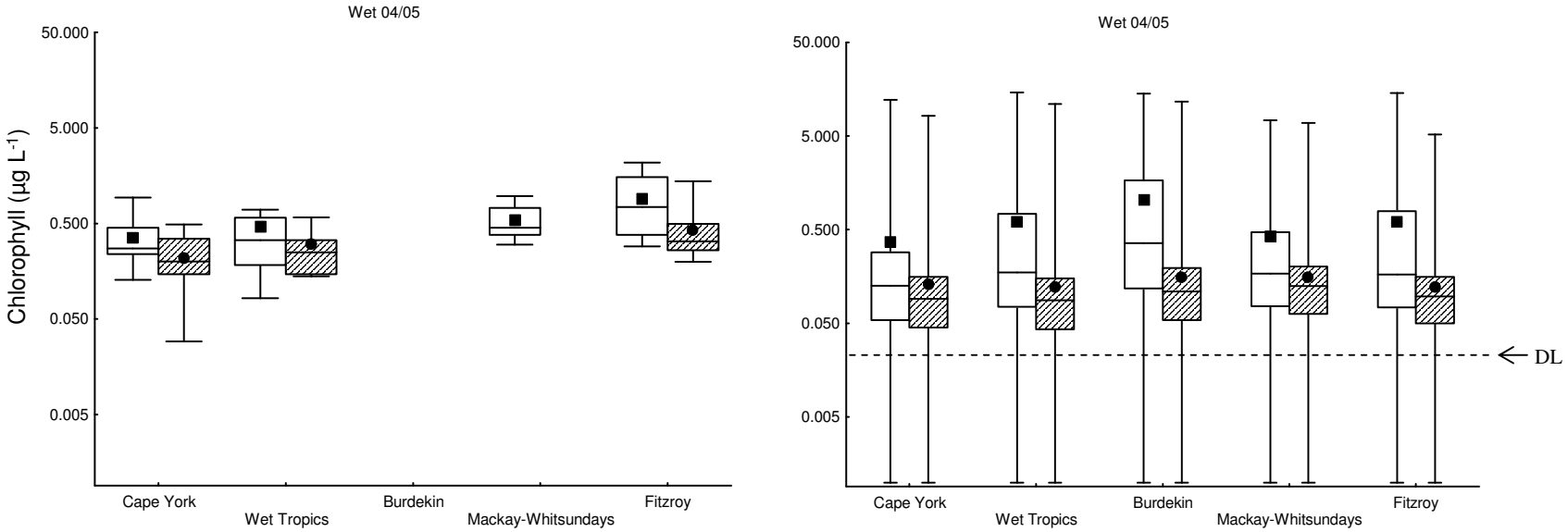
Figures 4.30 c (left) and 4.30d (right). Comparison of chlorophyll in situ data with remote sensing data for the 2003 dry season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.



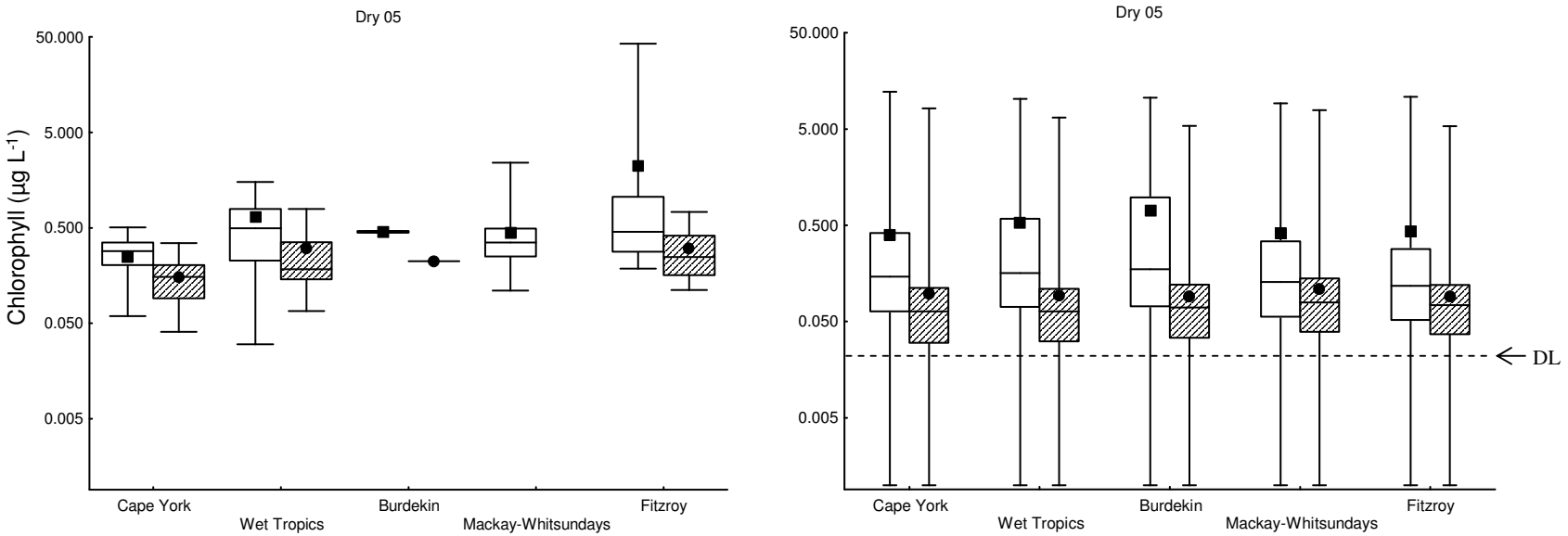
Figures 4.30 e (left) and 4.30f (right). Comparison of chlorophyll in situ data with remote sensing data for the 2003-2004 wet season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.



Figures 4.30 g (left) and 4.30h (right). Comparison of chlorophyll in situ data with remote sensing data for the 2004 dry season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.



Figures 4.30 i (left) and 4.30j (right). Comparison of chlorophyll in situ data with remote sensing data for the 2004-2005 wet season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.



Figures 4.30 k (left) and 4.30 l (right). Box plots comparing the in situ data distributions of the long term monitoring plan with the remote sensing data distributions for the 2005 dry season. The amount of data points varies from ten to a hundred for the in situ data to hundreds of thousands for the remote sensing data. The left box (white) contains the data for the samples collected in the waters in the zero to 20 m bathymetry area, the right box contains the data collected for the 20 m to 80 m bathymetry areas. DL = Detection limit for fluorometric analysis; approximately 0.04 µg L^{-1}.

Autonomous Environmental Loggers

(Attachment B Task 2.6)

The original intent of this component was to test two different types of autonomous environmental loggers for their suitability to provide time series of chlorophyll a concentrations and turbidity in Great Barrier Reef waters. However, one supplier (JCU) has ceased to manufacture this type of instrument. This task component has thus evolved into an operational test of the performance of one available logger type, the WET Labs Eco FLNTU Combination Fluorometer and Turbidity Sensor, under Great Barrier Reef *in situ* conditions. Three FLNTUs were purchased for these operational tests.

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

The FLNTU loggers perform *in situ* measurements of chlorophyll fluorescence and turbidity. The combination fluorometer and turbidity sensors of these loggers simultaneously measure chlorophyll fluorescence at 470nm and turbidity at 700nm. The fluorometer monitors chlorophyll concentration by directly measuring the amount of chlorophyll *a* fluorescence emission, using LEDs (centred at 455nm and modulated at 1 kHz) as the excitation source. A blue interference filter is used to reject the small amount of red light emitted by the LEDs. The blue light from the sources enters the water volume at an angle of approximately 55–60 degrees with respect to the end face of the unit. Fluoresced light is received by a detector positioned where the acceptance angle forms a 140-degree intersection with the source beam. A red interference filter is used to discriminate against the scattered blue excitation light. The red fluorescence emitted is synchronously detected by a silicon photodiode. Turbidity is measured simultaneously by detecting the scattered light from a 700nm LED at 140 degrees to the same detector used for fluorescence. The turbidity measurement is performed at the same 140 degree angle as the chlorophyll fluorescence.

FLNTU loggers were deployed at three inshore sites (determined by operational constraints, especially easy accessibility of sites from jetties, access to islands by commercial ferries) in the Wet Tropics, Burdekin and Whitsunday regions (Table 4.9). The trials measured the performance of the loggers in terms of sensitivity, temporal stability, battery life, reliability (mechanical and electronic) and ability to resist bio-fouling. Development of standard operational procedures for the calibration, deployment, retrieval and downloading of the WET Labs Eco FLNTU Combination Fluorometer and Turbidity Sensors have been part of this task, guided by the manufacturer's User's guide.

Table 4.9 Details of test deployments of combined chlorophyll a/turbidity loggers.

NRM Region	Data logger site	Deployment status
Wet Tropics	Dunk Island	18/12/05-25/01/06 25/01/06-04/04/06
Burdekin	AIMS Wharf	04/10/05-13/10/05
Mackay Whitsunday	Long Island	16/12/06 – 24/4/06

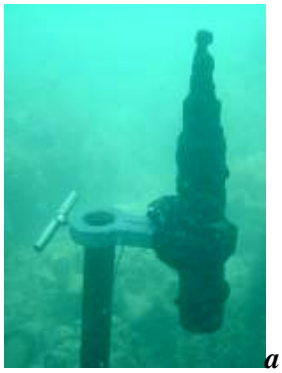
Two match-up trials were conducted to compare instantaneous FLNTU readings with concentrations of chlorophyll analysed in concurrently collected water samples (collection,

preparation and analyses of chlorophyll as before, see Material and Methods section at the beginning of this chapter).

An extensive calibration program was also undertaken. This included comparisons of *in situ* readings with chlorophyll *a* and suspended sediment values obtained from analyses of discrete water samples, as well as the construction of a laboratory calibration set-up to obtain calibration curves for chlorophyll a and turbidity. These calibration curves are based on the relative transmittance response of the FLNTU loggers to different phytoplankton densities (added as different volumes of a pure diatom culture) and concentrations or the reagent Formazin, respectively. Formazin is a standard calibration solution which is used widely for calibration of turbidity meters. The chlorophyll concentrations in the calibration solution were quantified by filtering 100ml of the solution on to a glass fibre filter and analysing Chlorophyll *a* and phaeophytin concentrations fluorometrically as before (see Material and Methods section at the beginning of this chapter).

Results

The FLNTU loggers were tested during deployments of 1 to 10 weeks. To decrease fouling of the instrument, the logger was wrapped in plastic and electrical tape and attached with a clamp to a star picket with the measurement window pointing downward (Figure 4.16a). Fouling in shorter deployments was minimal and did not affect the loggers' performance. However, the longest deployment of 10 weeks resulted in serious fouling of the logger 'face'. Data delivered under these conditions were still reliable because the measurement window was being kept clean, despite the in-built wiper being fouled as well (Figure 4.17a and b). Examples of typical records for chlorophyll concentration and turbidity are in Figure 4.18.



a

b

Figure 4.16. Wet Labs FLNTU combined chlorophyll a and turbidity logger
a) Logger deployed at Dunk Island (wrapped in plastic for protection)
b) Logger immersed in calibration chamber.

Figure 4.17. Fouling of logger after 4 months deployment at Long Island from December 2005 to April 2006.

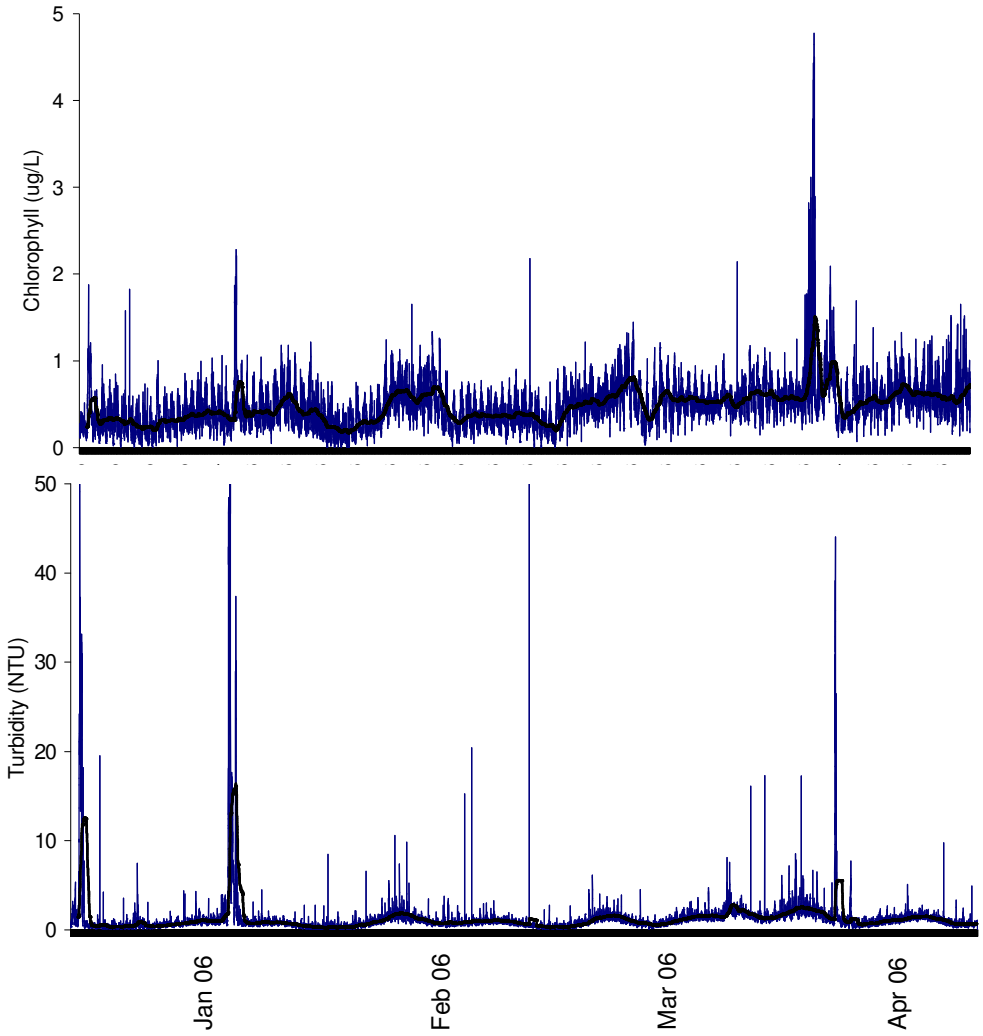


Figure 4.18. Example records of FLNTU measurements of a) chlorophyll and b) turbidity during a four month deployment at Long Island. The spikes in the record were interpreted as the logger being emersed during extreme low tides.

Laboratory calibration procedures were developed for the FLNTUs in a custom-made calibration chamber (Figure 4.16b). A typical calibration line for chlorophyll is in Figure 4.19.

Chlorophyll concentrations measured in field collected water samples also matched well with concurrent instantaneous measurements with an FLNTU (Figure 4.20).

The FLNTU deployed at Dunk Island was damaged during the passage of tropical cyclone Larry across the Island on 20 March 2006. The other two FLNTUs also showed some problems after one or two deployments each. The measurement window seems to delaminate. All three instruments were returned to the manufacturer and have since been repaired and upgraded with copper face plates to reduce fouling.

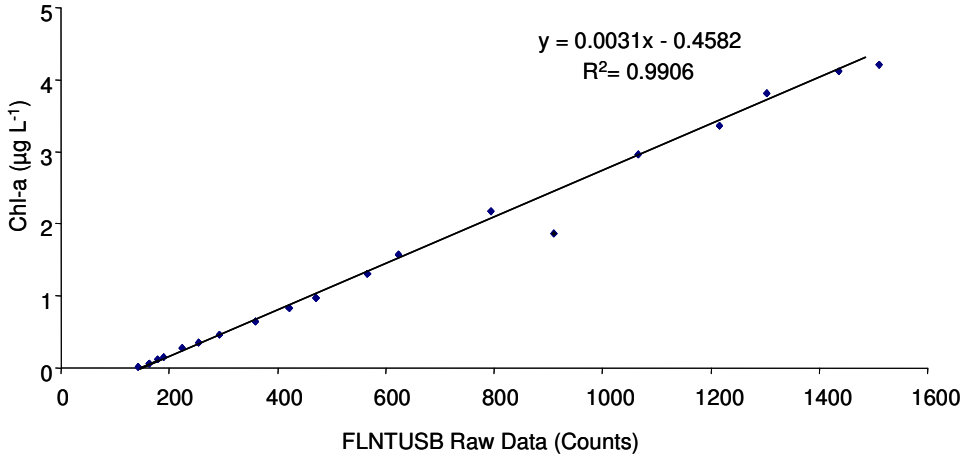


Figure 4.19. Example of calibration readings of the FLNTU in dilutions of pure plankton culture in filtered seawater.

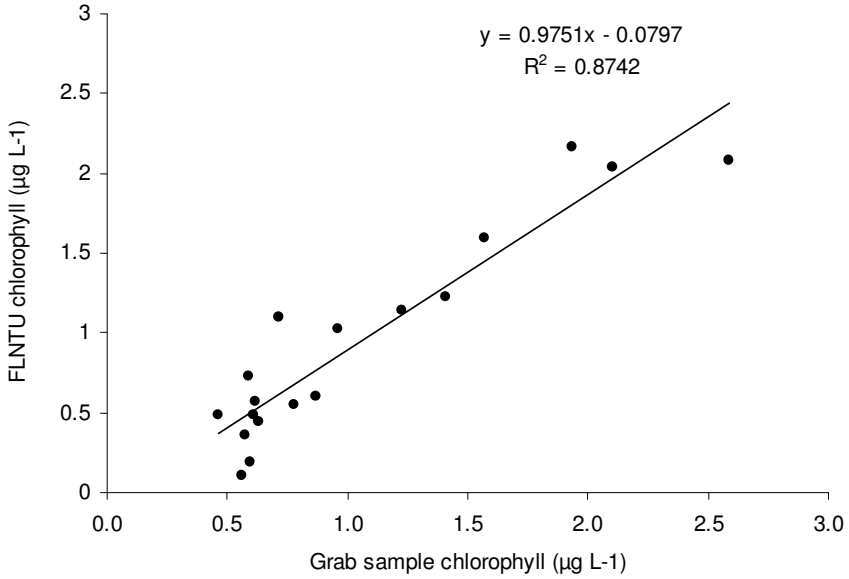


Figure 4.20. Match-up of FLNTU-measured chlorophyll and chlorophyll concentrations measured from natural water samples.

Temperature Loggers

(Attachment B Task 2.7)

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Data loggers (Odyssey, Dataflow Systems NZ) instantaneously record sea temperatures every 30 minutes, log data to an inbuilt memory which is downloaded every 6 to 12 months, depending on the site. Loggers are double- or triple- calibrated against a certified reference thermometer after each deployment and are generally accurate to ± 0.2°C.

Table 4.8. Location of temperature loggers for monitoring of sea temperatures. All "new logger" locations are directly at inshore coral reef monitoring locations.

NRM Region	Location	New logger	Existing logger
Cape York	Night Island		√
	Wallace Islet		√
Wet Tropics	Coconut Beach Reef		√
	Black Rocks Reef		√
	Low Isles		√
	Green Island		√
	Fitzroy Island	√	
	High Island	√	
	Normanby Island	√	
	North Barnard Island	√	
	Dunk/Bedarra Island	√	
Burdekin	Pioneer Bay		√
	Cattle Bay		√
	Northeast Bay		√
	Florence Bay		√
	Geoffrey Bay		√
	Nelly Bay		√
	Middle Reef		√
Mackay Whitsunday	Double Cone Island.	√	
	Hayman Island		√
	Daydream Island		√
	Dent Island	√	
	Pine Island	√	
	Seaforth Island	√	
Fitzroy	Barren Island	√	
	Pelican Island	√	
	Peak Island	√	
	Halfway Island		√
	Halftide Rocks		√

Autonomous temperature loggers were deployed at 29 inshore reef sites. Sites identified in the Head Contract are listed in Table 4.8 and include sites of the AIMS Long-term Temperature Monitoring Program (SeaTemps) funded by various sources, and new sites established under the current Reef Plan MMP. The new temperature logger locations have been matched with the reefs where coral reef benthic monitoring under Task 3 (see Chapter 5) has been carried out to optimise integration and interpretive capacity of the Reef Plan MMP. At most sites two temperature loggers have been deployed, at 2m and 5m depth, to correspond with the inshore coral monitoring carried out at these depths and provide redundancy in case of possible losses of loggers or data. Seventeen existing sites relevant to the Reef Plan MMP are being continued under the AIMS SeaTemps Program. For more details about logger deployment sites and times please see Appendix 1 (Table A1-4.10).

Temperature data from the ongoing AIMS Long-term Temperature Monitoring Program which has been operational since 1992, are reported in summary form (average daily, weekly and monthly temperatures) through an interactive webpage that allows data visualisation and download at http://www.reeffutures.org/topics/bleach/loggers.cfm. This site is intermittently updated when new data become available from retrieved loggers.

Results

Temperature data are reported for the period of October 2003 to October 2005 (Figure 4.15), spanning the two wet and dry seasons before (and including) the period when surveys of inshore coral reefs under Reef Plan MMP were undertaken (see Chapter 5). The temperature monitoring included existing loggers, which were close to but not directly at the reef survey locations, and new loggers, which were installed at twelve reef survey locations (Table 4.8). Results from these loggers will be reported in the future. Please refer to Appendix 1 for detailed data and deployment locations and times (Tables A1-4.10, A1-4.11, Figures A1-4.3 to –A1-4.5).

Temperatures follow a seasonal pattern with lowest temperatures occurring during the winter months (June, July, August) and highest temperatures during the summer months (December, January, February). Also obvious is a latitudinal pattern with decreasing average values from north to south (Figure 4.15).

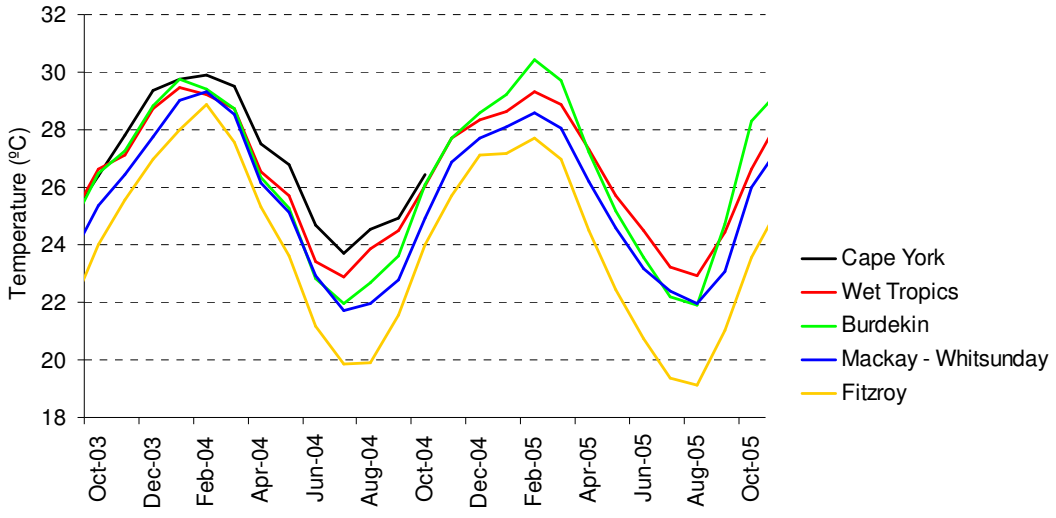


Figure 4.15. Average monthly seawater temperatures measured with temperature loggers at reef locations in five NRM regions.

Discussion

Nearshore lagoon water quality

At most lagoon monitoring locations in winter, median water column concentrations of bio-available inorganic nitrogen were very low (< 1 μg N L-1), while the summer values were elevated, mainly due to high ammonium concentrations (Figure 4.4). Because ammonium and nitrate in Great Barrier Reef lagoon waters are turned over on time scales ranging from hours to days (Furnas *et al.*, 2005), elevated nutrient, especially ammonium, concentrations in inshore waters usually indicate nutrient release from re-suspension of coastal sediments (during strong winds) and nutrient input from rivers (Devlin *et al.*, 2001; Devlin and Brodie, 2005). The weather during the sampling may have a significant impacts on the measured water quality concentrations, because lagoon nutrient sampling is undertaken only twice per year. The summer cruises in 2006 had many windy days and samples in the central Wet Tropics region were taken after a short period of heavy rainfall. Suspended sediment values were higher and more variable during the winter cruises and do not simply correlate with the nutrient parameters.

DON and PN are the largest pools 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, while large proportion of the PN is associated with detritus, plankton and bacteria. Both forms were also higher during the summer sampling, indicating re-suspension and higher plankton biomass. Sediment trap measurements indicated that particulate organic matter in the water column of the Great Barrier Reef lagoon is easily resuspended from the bottom by wave action (Furnas *et al.*, 1995).

In general, concentrations of dissolved and particulate phosphorus concentrations in Great Barrier Reef waters are also very low (Furnas, 2005), again largely due to rapid uptake by phytoplankton, bacteria and macrophytes. During the 2005/06 winter sampling cruises, elevated levels of phosphate and DOP were measured, while PP did not vary from the summer concentrations (Figure 4.5). It is likely that re-suspension of marine sediments lead to these higher values, which are also reflected in the higher and more variable values of suspended sediment concentrations in winter (see Figure 4.3).

There are very few datasets available for comparisons of the nutrient concentrations measured in the inshore lagoon under the current Reef Plan MMP monitoring (Furnas *et al.* 1997, Schaffelke *et al.* 2003 and references therein, Cooper *et al.* in preparation). The data from the current Reef Plan MMP monitoring period were generally within previously reported ranges. The longest and most detailed time series of water quality data for the Great Barrier Reef was collected by AIMS in coastal waters off Douglas Shire (Cape Tribulation) and Cairns (Cape Grafton) from 1989 to the present. Sampling of these stations was continued under Reef Plan MMP. Over time increases in TDP and SS have been identified, while PP decreased and PN and TDN show non-linear fluctuations (Figure 4.9, Table 4.3). This is similar to the results of a previous analyses of this dataset to 2004 by De'ath (2005), with the exception of TDN, which shows a declining trend over the past two years..

Surface chlorophyll concentrations in Great Barrier Reef waters have been measured since 1992 as part of a long-term monitoring program (Brodie *et al.*, in press). The Reef Plan MMP continued a large number of stations from the long-term chlorophyll program and added 11 additional locations in the coastal region. General patterns that have been found in the long-term dataset (Brodie *et al.*, in press; De'ath, 2005) are also obvious in the sampling conducted

under the Reef Plan MMP in 2004/05 and 2005/06. There is a general southward increase in mean chlorophyll a concentration, especially in the coastal zone (Figure 4.12). There is no significant cross-shelf gradient in chlorophyll concentrations in the Cape York region, while in all sectors further south have significantly higher chlorophyll values inshore than offshore (Table 4.7). The lack of a cross-shelf gradient in the North most likely indicates the smaller terrestrial nutrient inputs and perhaps, a greater degree of cross-shelf mixing (Brodie *et al.*, in press).

The dataset to date has shown some long-term variations in mean chlorophyll concentrations (Figure 4.14). However, the Far Northern, Port Douglas to Cairns and Whitsunday regions fluctuated over time with no net linear trend. Long-term trends were linear for the Cooktown-Lizard region and Townsville regions, with a slight increase in the former and no significant change in the latter. Notable is the steep increase from 2001-2006 in the Keppels to Capricorn Bunkers region. De'ath (2005) estimated that the long-term chlorophyll dataset should provide a sufficiently precise estimate of long-term change of this parameter; he calculated that an increase of approximately 32% could be detected, assuming linear change, over the monitoring period 1992-2004.

Long-term variations in mean chlorophyll concentrations were also observed (Figure 4.14). For example, mean chlorophyll concentrations in the Far Northern region doubled from 1997 to 2000 and then declined back to 1997 levels by 2006, and chlorophyll in the Keppels to Capricorn Bunkers region increased ~3.5 fold from 2001 to 2006. (Figure 4.14, Table 4.7).

High chlorophyll concentrations and high levels of variability in data from the coastal stations in the Burdekin and Fitzroy reflect the more frequent appearance of the pelagic cyanobacterium *Trichodesmium* in samples, but also increased chlorophyll concentrations supported by nutrients released by re-suspension and land run-off. Some inshore estimates appear higher than in previous years, which is likely to be a factor of the new coastal stations being added to the dataset.

Water quality remote sensing

Chlorophyll and suspended solids concentrations were also obtained from analysing remote sensing imagery. This technique is very promising to replace grab sampling for chlorophyll monitoring in the future, after some further validation and development.

The remote sensing task was successful in providing satellite based spatial and temporal information about near-surface concentrations of chlorophyll (Chl), suspended solids (SS) and vertical attenuation of diffuse downwelling light coefficients (K_d) in lagoonal and coastal waters of the GBRWHA. Specific algorithms for the GBRWHA were developed and applied to a year of daily MODIS Aqua satellite data encompassing a wet and dry season (2004-2005).

The amount of information generated was large (underlying satellite image data set is composed of 6 areas by 250 MODIS Aqua images by 3 variables and an error map = 6000 images) and it was necessary to develop smart ways of combining the data into an accessible format as presented here. In order to constrain the amount of images to be presented it was agreed to focus on presenting the log mean averages of Chl, SS and K_d over the 6 months of daily images for the wet and the dry season of 2004-2005.

The validation with the long term chlorophyll monitoring *in situ* dataset was carried out, but many problems were encountered with respect to the incompatibility of the types of information of these two datasets. In summary:

Variable	Point Data	MODIS Remote Sensing Data
No. of samples per transect	10 to a hundred	100,000 or more
Total No. of samples	Hundreds	Millions
Spatial coverage	Limited, fixed points	Unlimited
Temporal coverage	Limited	Unlimited (mainly cloud cover related)
Random distribution	No	Yes (with exceptions near to coast)
Amount of water sampled	Litres	Square kilometres per pixel, nr of pixels in the millions
Spatial representativity	Low	High
Accuracy per sample	High	Moderate
Accuracy per region	Low	High
Costs per sample	Very high	Low (but many samples)
No. of variables	Unlimited	Chlorophyll, SS, CDOM, Kd and derived products such as primary productivity, eutrophication, algal bloom detection and monitoring, transparency and turbidity assessments.

It is most likely more cost-effective to focus on improving the remote sensing detection and monitoring methods and products for those variables measurable by remote sensing (chlorophyll, SS, CDOM, K_d and derived products such as primary productivity, eutrophication, algal bloom detection and monitoring, transparency and turbidity assessment) and to focus *in situ* measurements on those variables that cannot be determined by remote sensing (nutrients, organic micro-pollutants, heavy metals etc.) as well as providing suitable validation data for the remote sensing information products.

This is a unique project both in Australia and internationally. Firstly, because the large size of the GBRWHA remote sensing based monitoring would therefore useful for water quality monitoring as it provides daily information on the lagoonal, reef and outer reef waters. Secondly, in the past year a regional algorithm method was developed which is also unique. As a result this subtask combined a relatively new optical remote sensing end-user (GBRMPA), an experienced GBRWHA research institute (AIMS) and innovative remote sensing derived information (by CSIRO). This required incremental development of both the remote sensing product by CSIRO and the desired (and possible) information delivery for management purposes for GBRMPA in combination with the aquatic ecosystem knowledge from AIMS. At the end of this one year project a viable remote sensing water quality product is available and can be used by GBRMPA to assist them in their monitoring responsibilities. However, it must be seen as a first stage of a more valuable form of management relevant information delivery method. A unique opportunity for collaboration now exists between GBRMPA, AIMS and CSIRO to develop parts of a reef water quality information system of unequalled sophistication nationally and on a par with or exceeding international developments. Please see Appendix 1 for further recommendations for future remote sensing work in the Great Barrier Reef.

Autonomous instrumentation for water quality monitoring

Autonomous data logging instruments were used as a third way to measure chlorophyll and turbidity under the Reef Plan MMP activities. The test deployments gave a high confidence in these loggers for being suitable to deliver useful time series of chlorophyll concentrations and turbidity readings. The bio-fouling after the longest deployment to date of 10 weeks was extreme, even though the data records were unaffected. Testing at reef sites will be necessary to determine the extent of biofouling in open water conditions. The upgrade of the instrument with a copper plate surrounding the measurement window and the planned wrapping of the instrument in copper tape is likely to further reduce biofouling. While remote sensing will allow the monitoring of large-scale patterns, autonomous instruments will have the benefit of obtaining high frequency data series at one location of particular interest, e.g. a reef or seagrass bed where long-term monitoring of biological status is undertaken.

Autonomous instruments to measure sea temperature have been long used in climate change research in the Great Barrier Reef (e.g. Berkelmans, 2002) and have been incorporated into the Reef Plan MMP. Sea temperatures were not significantly elevated during the two summers preceding the Reef Plan MMP activities. No major bleaching event occurred during this time. The 2003-04 summer had elevated temperatures at most GBR locations which caused mild stress and bleaching (5-10% coral cover showing bleaching or paling) at a number of reefs between Cooktown and the Keppel Islands (GBRMPA unpublished Bleachwatch Current Condition Report 2003-04). The 2004-05 summer had a hot water anomaly over much of the Coral Sea and northern Australia. Minor bleaching was observed in shallow areas at some reef sites during the course of the 2004-05 summer (GBRMPA unpublished Bleachwatch Current Condition Report 2004/05).

At this stage, the temperature monitoring data are only descriptive. These data will become more important in the future of the Reef Plan MMP, when they will serve as a correlative environmental variable. Long-term temperature data measured directly at reef monitoring locations will assist to explain changes in the status of inshore coral reefs, e.g. caused by mortality or stress due to coral bleaching caused by high temperatures.

Conclusions

The lagoon water quality sampling showed no distinct spatial patterns during the sampling period, apart from samples that were affected by a local flood event. This implies that the water of the coastal zone is in general well mixed and that it may be difficult to trace inputs from particular catchments, at least using the water column parameters. The more frequent chlorophyll sampling, which has been maintained for more than a decade, confirmed the distinct spatial pattern of higher concentration close to the coast (except for the Far North), indicting higher nutrient availability. Dissolved nutrient concentrations are inherently highly variable and concentrations that are mostly close to detection limits. Because the dissolved nutrients are rapidly assimilated into phytoplankton biomass and rapidly recycled, plant pigments such as chlorophyll a and particulate nutrients are therefore more useful proxy indicators of the quantity of nutrients which are circulating within the Great Barrier Reef ecosystem. This is indicated by the long-term time series north of Cairns, which showed an increase in particulate phosphorus and suspended solids (and total dissolved phosphorus). It remains to be seen if the twice yearly sampling of the newly established water quality monitoring sites under Reef Plan will show similar trends in the long term. At this time the sampling can only serve as a rough indicator of the water quality surrounding the surveyed reefs. A promising avenue is also the application of autonomous instruments to measure local

environmental parameters to relate to changes in biological communities such as coral reefs or seagrass meadows and the application of remote sensing to obtain more frequent data for chlorophyll and suspended solids on a whole GBR scale.

This one year project was a proof-of-concept stage for water quality monitoring by remote sensing. We successfully provided satellite-based spatial and temporal information about near-surface concentrations of chlorophyll, suspended solids and vertical light attenuation in Reef lagoonal and coastal waters.

To develop a more cost-effective water quality monitoring framework we suggest to focus on improving the remote sensing methods and products for those parameters measurable by remote sensing (e.g., chlorophyll, SS, CDOM, K_d), improve the use of automated instrumentation for local high-frequency monitoring, and to focus *in situ* monitoring activities on i) those variables that cannot be measured by these techniques (e.g., nutrients and pesticides) and ii) on the provision of suitable validation data for the remote sensing and instrumentation approaches.

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5. Inshore Coral Reefs Monitoring

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Introduction

The biological monitoring of inshore reefs is intended to document trends in the benthic reef communities of selected inshore reefs. These changes may be due to acute disturbances such as cyclonic winds, bleaching and crown-of-thorns starfish as well as those related to runoff (e.g. floods), which disrupt processes of recovery such as recruitment and growth. The reef monitoring sites are close to the sampling locations for lagoon water quality, allowing the possibility of correlating changes in reef communities to changes in water quality as well as the other, more acute impacts.

One salient attribute of a healthy ecological community is that it should be self-perpetuating and 'resilient', that is: able to recover from disturbance. One of the ways in which water quality is most likely to shape reef communities is through effects on coral reproduction and recruitment. Laboratory and field studies show that elevated concentrations of nutrients and other agrichemicals and 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 sub-lethal effects on established adult colonies. Because adult corals can tolerate poorer water quality than recruits and colonies are potentially long-lived, reefs may retain high coral cover even under conditions of declining water quality, but have low resilience. These may be relic communities made up of adult colonies that became established under more favourable conditions. Such relic communities would persist until a major disturbance, but subsequent recovery will be slow if recruitment is reduced or non-existent. This can lead to long term degradation of reefs, since extended recovery times increase the likelihood that further disturbances will occur before recovery is complete (McCook *et al*., 2001). For this reason, the surveys for the Reef Plan MMP estimate cover of various coral taxa and also collect information of size-distribution of colonies as evidence for the extent of past and ongoing recruitment. In addition, settlement and early survival of corals are measured using settlement plates in two localities that are considered at high risk of exposure to land runoff; the Whitsundays and the Wet Tropics coast off Innisfail.

The key aims of the inshore coral reef monitoring are to provide:

- Annual time series of habitat status for inshore reefs as a basis for detecting changes related to water quality.
- Information about past and ongoing coral recruitment on Great Barrier Reef inshore reefs as a measure for reef resilience.

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Survey locations were prescribed in the Head Contract and were chosen to represent reefs along a wide area of the coastline (and six Regional NRM regions, Figure 5.1), and to

represent the downstream effects of the major waterways flowing into the Great Barrier Reef lagoon (Table 5.1). In some instances survey locations were changed from the prescribed ones because sampling could not be implemented at the prescribed sites for reasons of reef morphology etc. (Table 5.1 for details).

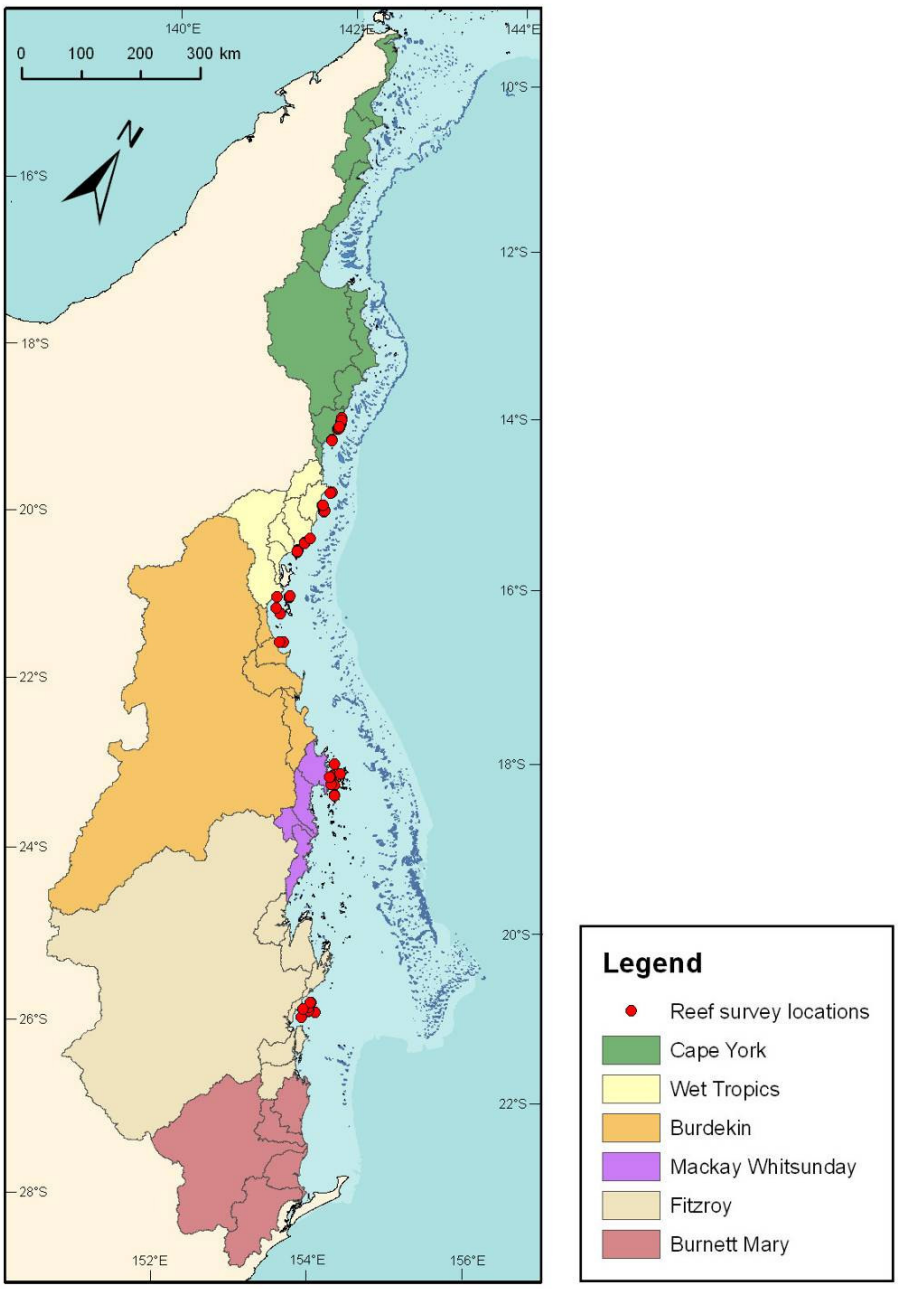

Figure 5.1. Distribution of survey locations among NRM regions (see Table 5.1 for details).

Two replicate sites are surveyed at each survey location. Ideally each site consists of a set of five 20 m transects, separated by about 5m, laid along depth contours on the reef slope at each of two depths: 2 and 5m below Lowest Astronomical Tide (LAT). At several locations there were no coral communities at or deeper than 5 m below LAT so no deeper surveys of reef communities were possible. The five transects were permanently marked and GPS waypoints

are recorded. The start- and end-points are marked with star-pickets and transects were laid out following the depth contour as precisely as possible. Compass bearings for each change in transect direction aid in tracking the path along the depth contour between star-pickets.

Table 5.1 Inshore reef monitoring locations.

NRM Region	Primary Catchment	Coral monitoring locations
Wet Tropics	Daintree	Daintree North[2]
		Daintree Central[2]
		Daintree South[2]
		Snapper Island North
		Snapper Island South
	Russell-Mulgrave, Johnstone	Fitzroy Island West[R]
		Fitzroy Island East[R]
		High Island West[R]
		High Island East[R]
		Frankland Group West[R]
		Frankland Group East[R]
	Tully	North Barnard Group*
		King Reef
		Dunk Island North
		Dunk Island South
Burdekin	Herbert	Orpheus Island East
		Pelorus and Orpheus Island West
		Lady Elliot Reef
	Burdekin	Pandora Reef
		Havannah Island
		Middle Reef[2]
		Geoffrey Bay
Mackay Whitsunday	Proserpine	Double Cone Island[R]
		Daydream Island[R]
		Shute & Tancred Island
		Pine Island[R]
		Hook Island
		Dent Island
		Seaforth Island**
Fitzroy	Fitzroy	Peak Island
		Barren Island
		Pelican Island
		Humpy & Halfway Island
		Middle Island
		North Keppel Island

[2] indicates locations where surveys were only at 2m depth.

Variations of survey locations from contract:

*North Barnard Island was chosen instead of South Barnard Island., because it had reef communities at the required depths, and gave a better distribution of locations in this region.

**Seaforth Island was chosen instead of Lindeman Island., because of the availability of previous survey data (KE Fabricius, unpublished).

Other locations in the Whitsundays: We were unable to find any reef development along the western sides of Whitsunday or Cid islands other than next to Dent Island where sites were already established.

[R] indicates locations where recruitment surveys by settlement plate deployment were carried out.

Six types of data were collected along each transect:

1. *Benthic cover*: For the first year of sampling cover of benthic organisms was estimated from five 20m video point intercept transects. The organisms beneath 5 points on each of 32 evenly-spaced frames are identified to highest possible taxonomic resolution (governed by image quality). This technique was altered for the second years sampling when still digital photographs taken at 50cm intervals were substituted for the 32 evenly-spaced video frames. The primary reason for the change was the enhanced image quality of digital photographs compared to captured video frames.
2. *Community size-frequency/demographics*: Colonies of hard and soft corals that fell wholly or partially within areas of a belt transect 34cm (data slate width) wide and totalling 50m in length were identified to genus and then classified into one of six size categories. The 50m of the demographic belt was made up of the first 10m of each of the five video point-intercept transects, however, all the data were pooled. Colonies falling within the survey area were classified into the following size categories: <2cm, 2cm to <5cm, 5cm to <10cm, 10cm to <20cm, 20cm to < 50cm, 50cm to < 100cm and 100cm or greater.
3. *Agents of coral mortality*. All new scars (identified as bare white skeleton) that were encountered along a 2m wide belt centred on the 5 video point transects were scored according to the perceived cause of the mortality. Potential agents of mortality included *Drupella* spp., crown-of-thorns starfish (*Acanthaster planci*), several categories of disease and unknown causes. Bleached and physically damaged corals were also recorded as a proportion of the living coral cover.
4. *Developmental indicators of water quality*. The Catchment to Reef joint research program of the CRC Reef Research Centre and the Rainforest CRC is currently developing indicators of the sub-lethal effects of poor water quality on coral reefs. Data on two of these indicators were collected during the surveys:
    - The shade of colour of all colonies of massive *Porites spp*. that occurred along the transects was estimated using a six point scale from a standardised colour chart (Siebeck *et al.* 2006); and
    - Collection of small samples of sediment for examination of the taxonomic composition of foraminiferan communities.

    Note: Measurements of another developmental indicator, rugosity of *Porites* spp colonies, was trialled, but proved too time-consuming to allow sites to be both set up and surveyed in the required time. This variable is unlikely to change rapidly and so baseline data may be collected during future surveys.
5. *Sediment sampling*. Sediment samples were collected from the reefs that were revisited in May 2006 for analysis of grainsize and terrestrial mud component. At each 5m deep site six 1cm deep cores were collected haphazardly along the length of the site from available deposits. Grainsize fractions are to be estimated by sieving for larger particles then optically for smaller grainsizes. The proportion of inorganic and organic carbon and the chlorophyll concentration will also be estimated.
6. *Coral settlement*. Coral settlement rates were estimated at reefs downstream from the Johnstone / Russell-Mulgrave rivers (Wet Tropics region) and at three reefs in the Mackay Whitsunday region. Cleaned terracotta tiles (11.5 x 11.5cm) were deployed as settlement plates in early September 2005, prior to the estimated time of coral spawning on inshore reefs, and retrieved in mid December 2005 (Table 5.2). Groups of six tiles were deployed at each reef, one group near the star-pickets marking the start of 1st, 3rd and 5th transects at

each site and depth (a total of 72 tiles per reef). The base plates to which tiles were attached were left in place for re-use in future years. After collection the tiles were bleached, dried and the number and taxonomic identity (to genus level where possible) of coral recruits was recorded, along with their position on the tile. Coral recruits were identified and categorized into five groups: non-isoporan Acroporidae, isoporan Acroporidae, Pocilloporidae, Poritidae, other and unknown.

Sampling

Sites at all locations in Table 5.1 have been marked with star-pickets and surveyed at least once. Two sites were marked out in each location. Where reef development allowed, surveys were made at two depths (2 m and 5 m below LAT) and five transects were surveyed at each depth in each site. All locations were surveyed once in the period April 2005 to January 2006. In May 2006 the second annual surveys of all locations were undertaken in the Tully and Johnstone / Russell-Mulgrave sub-regions of the Wet Tropics NRM region and also the Mackay Whitsunday region. Surveys of *Porites* colour were not repeated in the May 2006 surveys. Some of these locations were sampled in a similar manner in 2004, these results are referred to where appropriate.

In addition to the sampling specified in the Head Contract, sediment samples have been collected from all re-surveyed locations and analysis is underway. This will give an indication of the local re-suspension/deposition regime which may explain more of the variation in the structure of coral communities.

Table 5.2. Locations of coral settlement studies.

NRM Region	Catchment	Coral monitoring locations	Coral settlement tile deployment
Wet Tropics	Russell-Mulgrave Johnstone	Fitzroy Island West	06/09-13/12/05
		Fitzroy Island East	06/09-13/12/05
		High Island West	04/09-11/12/05
		High Island East	04/09-11/12/05
		Frankland Group East	03/09-12/12/05
		Frankland Group West	03/09-12/12/05
Mackay Whitsunday	Proserpine	Double Cone Island	09/09-19/12/05
		Daydream Island	08/09-18/12/05
		Pine Island	08/09-17/12/05

Univariate analyses

Variation in univariate summary variables (cover hard coral, soft coral and macroalgae and the density of recruit-sized colonies, overall richness of genera, richness of genera represented by recruit-sized colonies and the number of coral recruits found on settlement tiles) was analysed using linear mixed-effects models. Fixed effects were NRM region, Reef (nested within NRM region), Depth and Year; random effects were Site (nested within Reef) and the interactions with other fixed factors and the mean square error (Figure 5.2). Prior to analysis, data were averaged over transects, therefore this term does not feature in subsequent analyses.

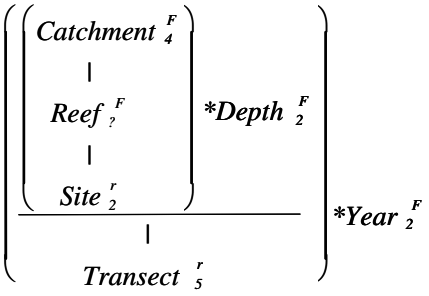

Figure 5.2. Schematic representation of the sampling design. "Catchment" refers to the NRM region. Terms linked by an asterisk are crossed; the hierarchy of nested terms is linked by a bar. The superscript F indicates a fixed effect and superscript r indicates a random effect; subscripts represent the number of levels of the factor terms.

Initial analyses considered the subset of locations that were surveyed at both depths in both years. This balanced analysis provided estimates of temporal effects and their interactions. A second set of analyses used a balanced subset using data from locations surveyed at both depths in Visit 1 only. This second analysis allowed estimation of depth effects and their spatial interactions based on all regions. Finally a set of analyses included the data from surveys at 2m depth only, as these gave the most complete spatial coverage.

To investigate the relationship between the univariate response variables and environmental data, generalised linear models with a log link function and errors proportional to the mean response were fitted. These models related the response variables to regional locations and environmental variables (Table 5.3). As preliminary water quality data was available at a spatial resolution roughly equating to individual reefs, univariate response variables were averaged to the reef level for these models. This averaging included averaging over depths and also over locations with differing aspects when they occurred on the same island.

Table 5.3 Description of environmental variables included in both univariate and multivariate models.

Water Quality Index	An index constructed from the sum of relative concentration (z-scores) of chlorophyll and particulate nitrogen, particulate phosphorus and organic carbon
Ecosystem Risk Index (ERI)	Estimated risk of exposure to river borne contaminants, from Devlin *et al.* (2003)
Distance from shore	Distance from the reef to nearest point of the mainland
Predicted WQ gradient	The ranking of reefs within each NRM region in terms of estimated risk of exposure to runoff. Values based largely on the relative proximity to the major rivers within each NRM region. Relative distance from shore was also considered.

Data manipulations prior to analysis

For the univariate response variables (cover hard coral, soft coral and macroalgae and the density of recruit-sized colonies) it was necessary to fourth-root transform the data. The numbers of recruits on settlement tiles required natural log transformation. Estimates of the overall richness of genera and the richness of genera represented by recruit-sized colonies did not require transformation prior to analysis.

The density of recruit-sized colonies refers to the numbers of colonies<10cm on demography transects, standardised to the area of substrate considered "available" to settling hard coral larvae. Available substrate was taken as the proportion of the substrate that was classified as "Algae" or "Rubble" in the point-intercept analysis. Densities of recruit-size colonies were standardised per unit area of available substrate. The mean densities of recruit-sized colonies at each sampling location are presented graphically in the descriptions of the NRM regions. The plots include a reference line showing the mean density over all regions. Barren Island was not included in the calculation of the overall mean as the very high coral cover and hence very low area of available substrate caused estimates here to deviate greatly from those observed elsewhere.

Multivariate analyses

The composition of hard and soft coral communities was investigated using data from the demography surveys. The approximate area covered by each colony was estimated assuming that the diameter of a colony was equal to the mid point of its size category, ie. a colony in the 10-20cm category was assumed to have a diameter of 15cm and hence an area of $0.017m^2$. The effective transect width was also adjusted for each size category by adding the diameter of the mid value of each category to the width of the intersect belt (34cm). These two calculations allowed us to derive an estimate of cover for all genera of hard and soft corals occurring on demography transects. We used these estimates rather than those from video transects because better taxonomic resolution was possible in the field compared with that from video images. Groups of locations with similar coral community types were then identified from hierarchical cluster analysis using the complete linkage method. The optimal number of clusters was defined from the cluster solution that minimised the average silhouette width (Rousseeuw 1987).

Assessments of the relationships between hard coral and soft coral communities and environmental variables were based on redundancy analysis. Environmental variables are listed in Table 5.3; their relative importance as predictors of the composition of coral communities was determined by the backward elimination of explanatory variables based on permutation tests.

Results

Results are presented in two sections. In the first the results of spatial and limited temporal analysis of various community attributes are presented. The aim of this section is to identify possible links between spatial variation in coral communities and perceived gradients in water quality. The second section provides more detailed descriptions of the coral communities at survey locations associated with each NRM region. Incorporated into this second section are references to existing data from the regions and summaries of past disturbance events.

Because the rivers feeding into the Great Barrier Reef lagoon are so diverse in size and flow pattern, it makes sense to discuss many results in terms of sets of survey locations that are downstream of, and influenced by, major rivers.

Summary of benthic communities

The general picture of the reef communities on nearshore reefs was of great variability, both between the NRM regions and among the reefs within regions. The cover of hard corals, of soft corals and of macroalgae and the density of recruit-sized colonies all varied substantially among NRM regions (Appendices 1.5.1b, 1.5.4, 1.5.5, 1.5.6, 1.5.7) as did the overall number of genera and the number of genera present as recruit-sized colonies (Appendix 1.5.8).

Regional patterns for each of these variables and any average differences between depths or between the surveys in 2005 and 2006 are described in the next sections. After accounting for regional differences, there was no strong evidence for relationships between community attributes and variation in the limited available water quality data or in any of the indices related to water quality (Appendix 1.5.1a). The only exception was slight evidence that, within each NRM region, the cover of macroalgae tended to be higher on the reefs that were closer to the mouths of the main river. This result is explored further in the section concerning cover of macroalgae (below).

Variation among the reefs within NRM regions is presented separately in the following section describing the benthic communities associated with each NRM region.

The composition of both hard coral and soft coral communities also varied substantially among regions (Table 5.7, Appendix 1.5.2). Once regional differences had been accounted for, the composition of hard coral communities varied in relation to our constructed Water Quality Index (Table 5.7, Figure 5.6). These results are explored further in the section concerning variation in coral communities (below).

Cover of hard corals

There were significant differences in the average cover of hard corals among NRM regions (Appendix 1: Table A1-5.1a and b). Average cover was highest for the Fitzroy region (44.2%); this was substantially higher than the average cover in any of the other NRM regions. Cover was also higher in the Mackay Whitsunday region (34.4%) than in either the Herbert/Tully sub-region (26%) or Burdekin (15.5%) regions. The hard coral cover on reefs in the Burdekin region was significantly lower than in any of the other regions.

Hard coral cover also varied substantially within NRM regions (Appendix 1: Table A1-5.1b, and descriptions of reefs in individual regions). Within NRM regions, differences in cover among reefs showed little relationship with their proximity to major rivers. The Fitzroy region was an exception: cover was significantly lower on Pelican Island and Peak Island than on the four reefs further from the river.

Average cover of hard corals varied with depth. Overall, average cover was higher at 5m (31%) than at 2m (26.6%). This relationship was consistent among NRM regions, though it did vary among reefs within regions.

There was an overall reduction in coral cover from an estimated 30% to 23.6% on reefs that were surveyed in both 2005 and 2006, though this varied significantly among NRM regions and among reefs within regions. The average cover on re-surveyed reefs in both the Mackay Whitsunday region and Johnstone / Russell-Mulgrave sub-region remained largely unchanged between years, while average coral cover in the Herbert / Tully sub-region fell from 26% to 8.6% because several reefs were badly damaged by TC Larry (see section Herbert / Tully sub-region for more detail).

Cover of soft corals

There were differences in the average cover of soft coral among the NRM regions. The cover on reefs in the Mackay Whitsunday region (8.3%) and Johnstone / Russell-Mulgrave sub-region (6.8%) was higher than at all other NRM regions. The cover in the Burdekin region (3.3%) and the Barron-Daintree sub-region (3.2%) was also higher than the average from either the Fitzroy region (1.3%) or Herbert / Tully sub-region (1.2%). Once again there was

substantial variation in the cover of soft corals among reefs within the different NRM regions. The variation among reefs within NRM regions bore no consistent relation with distance from rivers. The cover of soft corals showed no consistent variation with depth though did vary between depths at individual reefs.

There was an overall reduction in soft coral cover in the three NRM regions that were surveyed twice from 5.1% in 2005 to 3.7% in 2006, though this varied significantly among regions and among reefs within regions. The average cover on reefs in the Mackay Whitsunday region increased marginally over this period while cover decreased marginally on reefs in the Johnstone / Russell-Mulgrave sub-region and decreased significantly on reefs in the Herbert / Tully sub-region. Several reefs in the Herbert / Tully sub-region were badly damaged by TC Larry (see description of the coral communities of the Herbert / Tully sub-region below for more detail).

Cover of macroalgae

There were significant differences in average cover of macroalgae among NRM regions. Average cover was highest in the Herbert / Tully sub-region of the Wet Tropics (7.9%) and in the Burdekin region (5.5%). Average cover in the Johnstone / Russell-Mulgrave sub-region (0.4%) and Mackay Whitsunday region (0.6%) was very low. Macroalgal cover varied substantially among reefs within NRM regions and tended to be higher on reefs that were nearer to river mouths or very close to the coast. Over all, cover was higher at 2m depth than at 5m, and this pattern held in all NRM regions except the Johnstone/Russell-Mulgrave sub-region (where cover was very low). The magnitude of this difference varied among regions.

There was an overall reduction in the cover of macroalgae between 2005 and 2006, but this was driven by changes in the Herbert / Tully sub-region of the Wet Tropics, where average cover dropped from 7.9% to 1.5%, probably due to TC Larry (see description of the coral communities of the Herbert / Tully sub-region for more detail).

Density of recruit-sized colonies

The density of colonies in the recruit size classes (<10cm) varied among NRM regions. Survey locations in the Mackay Whitsunday region and the Johnstone / Russell-Mulgrave and Herbert / Tully sub-regions (12.5 -14 colonies per square metre of available substrate) had significantly higher densities than the remaining NRM regions, where densities range from 6.4m^{-2} on reefs in the Burdekin region to 6.0m^{-2} in the Barron / Daintree sub-region. The densities varied significantly among reefs within each NRM region and although there was no consistent difference in density of recruits between sites at 2m and those at 5m, there was substantial variability between depths from reef to reef.

Comparing mean densities of recruit-sized colonies among reefs within regions found no strong relationship between recruit density and proximity to river. However, the lowest density of recruit-sized colonies in the Fitzroy region (Peak Island), Herbert / Tully sub-region (Dunk Island South and King Reef), Johnstone / Russell-Mulgrave sub-region (western reefs of both High Island and Frankland Group) and the Barron / Daintree sub-region (Snapper Island South) in the Wet Tropics, were all on reefs that were close to river mouths.

The average density corals in the <10cm size class declined from 15.7 to 11.1m^{-2} in the three NRM regions that were surveyed in both 2005 and 2006. Declines varied among reefs, with significant declines recorded on only four of the 17 reefs that were surveyed in both years. The density was significantly higher on only one reef (Dunk Island South). The largest

decline was at the North Barnard group, where the estimated mean density of colonies dropped from 34.4 to 3.1m^{-2} of available substrate following severe damage to the coral communities from TC Larry.

Richness of hard coral genera

The average number of genera recorded on demography transects showed overall differences among NRM regions (Appendix 1: Table A1-5.7). The richest was the Mackay Whitsunday region with an average of 28.4 genera. This value is significantly higher than all other regions. Conversely, average richness was significantly lower on reefs in the Barron-Daintree sub-region (17.2) and Fitzroy region (9.2) than all other regions. There was however substantial variation around these means among the reefs within the regions. Richness showed no consistent pattern with distance from river mouths. Consistently more genera were recorded at 5m than at 2m in all NRM regions with the exception of the Fitzroy region where the richness was only marginally higher at 2m (9.4) than at 5m (9.1).

There was no significant overall difference in the richness of genera on demography transects in the three NRM regions that were surveyed in both 2005 and 2006. There was some variability in number of genera among regions, with an overall decline in the Herbert / Tully sub-region but no net change at reefs in either the Johnstone /Russell-Mulgrave sub-region or Mackay Whitsunday region. Within the Herbert / Tully sub-region, damage by TC Larry caused significant declines in richness at 5m at Dunk Island North (from 31.5 genera per site in 2005 to 25 in 2006) and the North Barnard Group (from 34 to 20). Richness also changed at High Island East at 5m depth. There was coral bleaching on our transects in 2006 and the number of genera dropped from 29.5 per site to 23. Richness increased from 24.5 to 31 genera at 2m at Seaforth Island.

Richness of recruit-sized (<10cm) colonies

On average the number of genera represented by colonies <10cm in diameter varied among NRM regions (Appendix 1; Table A1-5.7). Differences among NRM regions followed the same pattern as the overall richness of all size classes, with significantly higher average numbers of genera represented in the Mackay Whitsunday region (20.7) and the Johnstone / Russell-Mulgrave (19.6) and Herbert / Tully (19.2) sub-regions than in the Burdekin (14.7), Barron / Daintree (11.7) or Fitzroy (6.2) regions. Again there was considerable variation among the reefs within the NRM regions. Comparing mean values for reefs within NRM regions and considering the relative proximity of reefs to the major rivers shows that for the Burdekin, Johnstone / Russell-Mulgrave and to lesser degree Herbert / Tully regions, reefs with lowest richness of recruit-sized colonies are most proximate to the rivers. The opposite was true in the Fitzroy region where the richness of recruits was low overall but highest at Peak Island and Pelican Island.

The richness of recruit-sized colonies was lower at 2m than at 5m (average of 14.1 vs. 18 genera). This relationship varied marginally among NRM regions though only in magnitude not direction.

For the three NRM regions that were surveyed in both 2005 and 2006 there was a very slight reduction in average number of genera represented by colonies <10cm in diameter on demography transects from 19.9 in 2005 to 18 in 2006.

The number of recruits to tiles

The average number of coral recruits settling onto tiles was higher at reefs in the Wet Tropics than in the Mackay Whitsunday region (Table 5.4). There was significant variability among reefs within the regions, and between depths though this relationship was not consistent for the two regions (Appendix 1: Table A1-5.1b). Comparing estimated means for each reef showed that there was no difference in the average number of recruits per tile among reefs in the Wet Tropics. In the Mackay Whitsunday region there were significantly fewer recruits per tile at Double Cone Island than at either Pine Island or Daydream Island (Table 5.4). Settlement was greater at 5m than at 2m on all reefs, though these differences were only significant at Daydream Island and Double Cone Island, both in the Mackay Whitsunday region. Comparing the exposed (East) and sheltered (West) sides of the Wet Tropics reefs showed that significantly higher numbers of corals settled on tiles on the exposed sides of the islands. This pattern was not consistent on all three reefs; significantly greater numbers settled in the sheltered sites at Fitzroy Island.

Non-isoporan Acroporidae made up 89.5% of all coral spat recorded on the tiles (Table 5.4). Pocilloporidae (6% of all spat) and Poritidae (3.2% of all spat) were well represented at some reefs. The remaining taxonomic groups: isoporan Acroporidae, other and unknown represented only 1.1% all spat recorded (Table 5.4).

There have been two coral settlement studies in the Wet Tropics region that are very comparable with the data collected in 2005 (Table 5.5). The average number of recruits per tile observed in 2005 exceeded the levels recorded in any previous study. There was substantial variability in recruitment among years; on Wet Tropics reefs there was approximately a 6-fold difference between the highest level of recruitment recorded previously and observations in 2005. The western reefs of the Frankland Group and High Island do have the lowest recruitment of the reefs in the Wet Tropics but recruitment was low on eastern side of Fitzroy Island where water quality should have been the best of any of these reefs. In summary there was not a consistent increase in recruitment with distance from the mouth of the Russell / Mulgrave River. Data were too few to allow a formal analysis of relationship between settlement density and Water Quality Index, but none was apparent (Figure 5.3).

Recruitment in 2005 in the Mackay Whitsunday region was 10 times that recorded in 1993, though spatial patterns were consistent in that the numbers of recruits per tile were lower at Double Cone Island than at Pine Island in both years. There was no clear increase in recruitment with distance from the major rivers influencing the Whitsunday transect. Unlike the locations in the Wet Tropics, the site with the highest Water Quality Index had the lowest settlement (Figure 5.3), though the sparse data did not allow formal analysis.

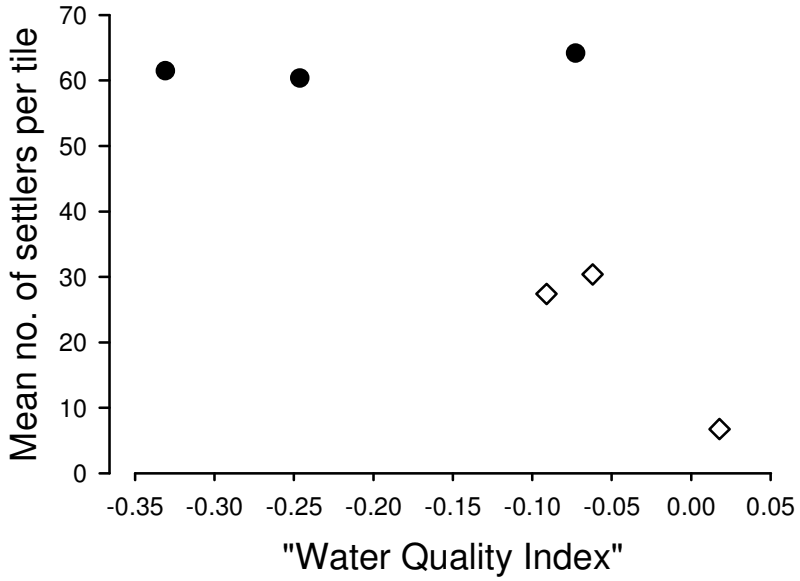

Figure 5.3 Mean number of coral larvae (all taxa) settling per tile in 2005 plotted against Water Quality Index. Filled circles represent means for reefs in the Wet Tropics; diamonds represent means for reefs in the Mackay Whitsunday region. Higher values of the Water Quality Index indicate higher concentrations of the contributing variables.

Table 5.4. Total numbers of coral recruits recorded on terracotta tiles at each reef and depth. Values in bold are average numbers of recruits per tile.

NRM Region	Primary catchment	Reef	Depth	non-isoporan Acroporidae	Pocilloporidae	Poritidae	Isoporan Acroporidae	other	Unknown	Total	**Mean**	**Reef mean**	**Island mean**	**Region mean**
Wet Tropics	Russell/Mulgrave and Johnstone	High Island (West)	2	1170	42	16	0	0	13	1241	**34.5**	**41.2**	**64.1**	**61.9**
			5	1589	74	20	0	5	38	1726	**47.9**			
		High Island (East)	2	2781	312	49	0	0	4	3146	**87.4**	**87.1**		
			5	2902	153	37	0	2	31	3125	**86.8**			
		Frankland Group (West)	2	503	130	12	0	0	4	649	**18**	**24.1**	**60.3**	
			5	926	25	130	2	1	1	1085	**30.1**			
		Frankland Group (East)	2	3153	160	36	15	0	11	3375	**93.8**	**96.6**		
			5	3334	201	37	0	1	10	3583	**99.5**			
		Fitzroy Island (West)	2	2747	215	168	1	0	8	3139	**87.2**	**80.9**	**61.4**	
			5	2330	95	245	0	1	14	2685	**74.6**			
		Fitzroy Island (East)	2	1365	28	10	0	0	5	1408	**39.1**	**42**		
			5	1308	262	39	1	0	8	1618	**44.9**			
Mackay Whitsunday	Proserpine	Pine Island	2	556	6	48	0	39	23	672	**18.7**	**27.4**	**27.4**	**21.5**
			5	1117	10	111	1	25	38	1302	**36.2**			
		Daydream Island	2	443	14	7	2	8	2	476	**13.2**	**30.4**	**30.4**	
			5	1608	39	22	2	19	21	1711	**47.5**			
		Double Cone Island	2	38	63	1	0	0	0	102	**2.8**	**6.8**	**6.8**	
			5	272	98	4	0	11	0	385	**10.7**			

Table 5.5. Recruit densities recorded in previous studies on the Great Barrier Reef.

NRM Region	Primary catchment	Reef	1993	1999	2000	2002	2003	2005
Wet Tropics	Johnstone	High Island (West)				**12.3**	**3.3**	**41.2**
		High Island (East)		**3.3**	**1.8**	**2.3**	**4.4**	**87.1**
		Frankland Group (West)				**3.1**	**5.3**	**24.1**
		Frankland Group (East)		**11**	**3.8**	**6.3**	**2.7**	**96.6**
		Fitzroy Island (West)				**22.6**	**3.5**	**80.9**
		Fitzroy Island (East)		**27.1**	**5.4**	**10.4**	**4.2**	**42**
Mackay Whitsunday	Proserpine	Pine Island	**2.3**					**27.4**
		Daydream Island						**30.4**
		Double Cone Island	**0.6**					**6.8**

Sources: Data from 1993 are the average number of all recruits from tiles at 2m (van Woesik *et al*., 1999). Data from 1999 and 2000 are the average numbers of Acroporidae recruits from tiles placed at 4m depth at four locations around each reef (Smith *et al*., 2005). Data for 2002 and 2003 are the average number of all coral recruits to tiles placed at 5m and 8m on the eastern and western sides of each reef (K Fabricius unpublished data).

Coral colour

The colour of massive *Porites* colonies was estimated at all survey locations except those in the Daintree / Barron sub-region or Fitzroy region. Massive *Porites* colonies were very rare on most reefs in the Fitzroy region so the colour of the dominant branching *Acropora* species was estimated instead. There were very few massive *Porites* on reefs in the Herbert / Tully sub- region (with the exception of the southern side of Dunk Island) and on reefs at Pine Island, Shute and Tancred Island and at 2m depth at Daydream Island in the Mackay Whitsunday region. These data were not analysed formally; the following patterns are based on summary plots (Figure 5.8).

- The intensity of colour of branching *Acropora* spp. did not vary either among reefs or between depths within reefs in the Keppel Island group.
- There is some evidence that *Porites* colonies nearer the coast (more turbid waters) in the Burdekin region were darker than those in sites further from shore (compare Geoffrey Bay, Lady Elliot and Pandora Reefs to those further offshore (Figure 5.4). The low values from 5m depth at Lady Elliot Reef were due to a consistent bias by a first time observer.
- *Porites* colonies were darker at 5m than at 2m at most reefs in the Mackay Whitsunday region. The exceptions were Double Cone Island and Pine Island, though the sample size at Pine Island was very low.

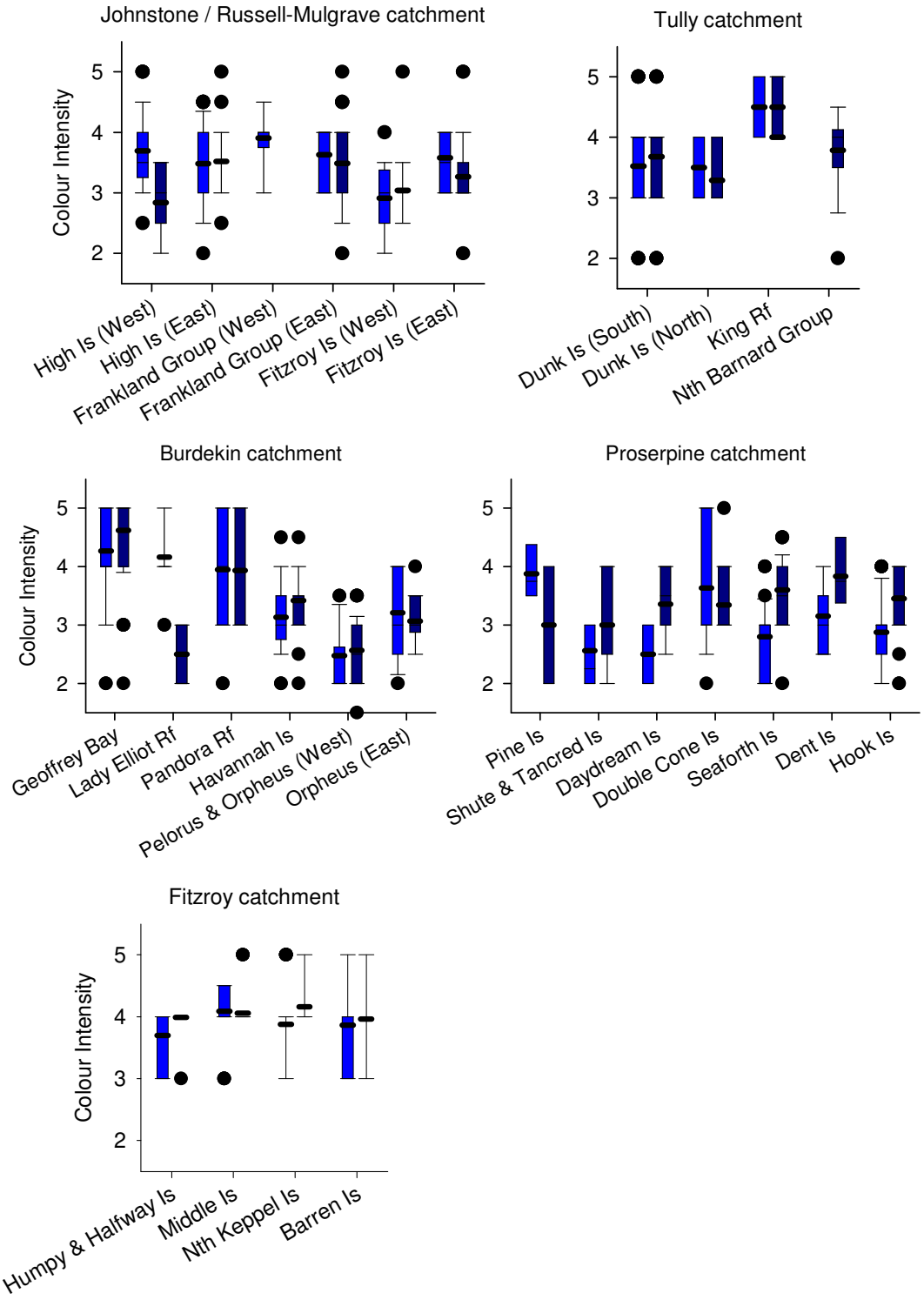

Figure 5.4. Colour intensity of colonies of massive Porites spp (or branching Acropora spp in the Fitzroy region). Mid-blue bars represent samples from 2m depth; dark blue bars from 5m depth. Heavy dark horizontal lines give the mean values and boxes include the 25th to 75th percentiles of the observations, with error bars including the 5th to 95th percentiles. Observations beyond these limits are indicated by solid dots.

Variation in structure of coral communities

Five broad types of nearshore coral communities were distinguished based on the relative proportions of hard coral and soft coral genera at each reef and depth (Figure 5.5). The most distinct group of communities was found only on reefs close to the Fitzroy River (Cluster group 5 in Figure 5.5). These reefs had substantially higher average proportions of the hard coral genera *Psammocora*, *Goniastrea*, *Cyphastrea* and *Hydnophora* and the soft coral genera *Sarcophyton*, *Junceella* and *Cladiella* (Table 5.6). Another relatively discrete community type was dominated by *Porites* spp. with a relatively high proportion of soft corals in the genus *Lobophytum* (Cluster group 4 in Figure 5.5, Table 5.6) This community type was found at Pandora Reef in the Burdekin region, on the western reefs of both High Island and the Frankland Group in the Johnstone / Russell-Mulgrave sub-region and at the southern reef of Snapper Island in the Barron / Daintree sub-region of the Wet Tropics. The communities at the majority of reefs had high proportion of the hard corals in the genus *Acropora* and soft corals in genera *Xenia* and *Efflatournaria*. Communities of this type did not occur near the mouths of rivers.

Table 5.6. Major constituents of coral communities in each cluster group. Values represent the average proportion of the hard and soft coral communities identified by cluster analysis that was contributed by each genus. Genera were included if they constituted more than 5% of the community in at least one of the cluster groups. Shaded cells indicate the cluster group where the proportion of that genus was greatest; boldface indicates cluster groups where that genus was common (>5% of the community).

Hard Corals	Cluster groups				
	1	2	3	4	5
Goniopora	**12**	0.6	2.1	4.3	**6.7**
Turbinaria	**8.9**	1.1	2.1	0.1	**5.4**
Pachyseris	**5.4**	0.2	1.2	3.9	
Acropora Branching	4.5	**35.3**	3.3	1.7	**6.4**
Acropora Other	**9.6**	**26.1**	**15.4**	3.6	**10.5**
Montipora	**8.8**	**14**	**6.3**	0.3	**7.2**
Pocillopora	0.5	1.8	**5.6**	0.4	1.7
Porites Other	1.7	2.7	**10.4**	**25.8**	0.3
Porites Branching	3	0.4	**8.6**	**20.8**	
Porites Massive	**5.5**	2.9	**8.2**	**19.2**	0.5
Heliopora				**7.6**	
Psammocora	1.1	0.3	0.1	0.2	**22.8**
Goniastrea	1.3	0.7	2.5	0.9	**14.4**
Cyphastrea	3.3	0.7	1.4		**11.7**
Hydnophora	1	0.7	0.5		**5.2**

Soft Corals	Cluster groups				
	1	2	3	4	5
Briareum	**40.8**	**26.3**	**9.4**	**30.4**	
Klyxum	**14.2**	**9.5**	**8.6**	3.4	0.6
Xenia	0.2	**15.9**	0.1		**12.6**
Efflatournaria	0.2	**6.5**			0.5
Sinularia	**18.1**	**11.2**	**41.8**	**25.3**	**19.5**
Rhytisma		0.2	**17**		
Lobophytum	1.5	2.3	**9.9**	**23.5**	3.2
Clavularia	**6.3**	**8.5**	0.9	**11.3**	0.2
Sarcophyton	**12.7**	**10.3**	**11.4**	**5.8**	**36.4**
Junceella	3.7	1.1			**14.1**
Cladiella	0.4	1.7	0.5	0.1	**7.3**

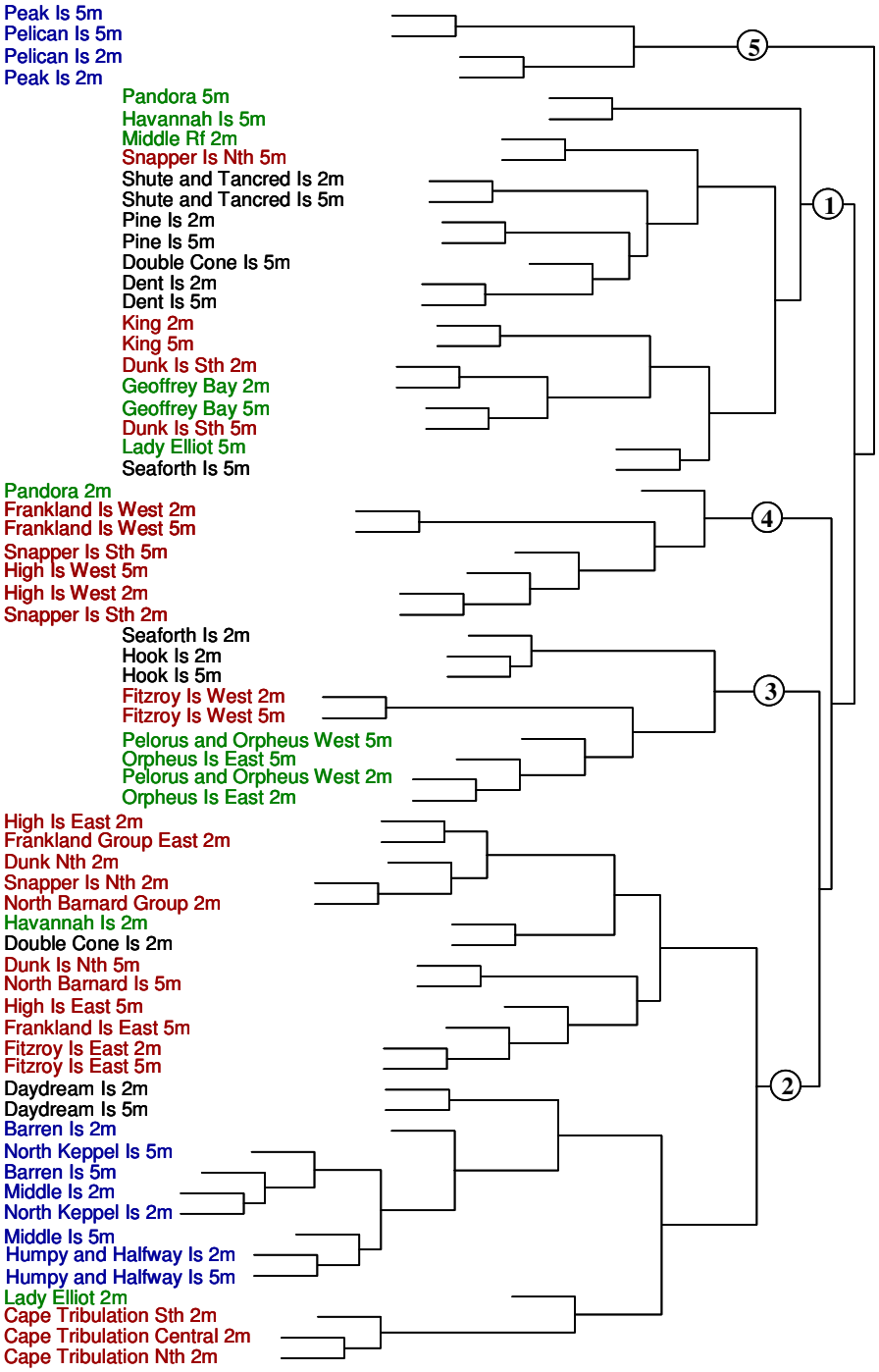

Figure 5.5. Groups of similar survey sites based on the relative cover of hard and soft coral genera in reef communities. Numbers on main branches correspond to groups in Table 5.6. Proportional cover for genera were estimated from demography transects. Cover estimates were converted to proportions of the hard coral / soft coral community. Data were fourth-root transformed; clustering was based on the Manhattan metric. Reefs are colour coded by NRM Regionas follows, Wet Tropics (Red), Burdekin (green), Mackay Whitsunday (black) and Fitzroy (blue).

Both hard and soft coral communities varied in composition among NRM regions (Table 5.7, Appendix 1: Figure A1-5.1). There was a significant relationship between community structure of hard coral communities and the Water Quality Index. There was weaker evidence for differences in community composition with distance to shore (Table 5.7). Communities of both hard corals and soft coral in the Fitzroy region differed most from those in other regions (Appendix 1: Figure A1-5.1). After accounting for the regional effects, distance from shore and the Water Quality Index appear as largely compensatory: concentrations of the water quality variables increased as distance from shore decreased (Figure 5. 4). While the relationship between the composition of soft coral communities and the Water Quality Index was not significant, the variation in the Water Quality Index did explain 5% of the variability in soft coral communities (Table 5.7). Note that, once regional differences have been accounted for, most soft coral genera with the exception of *Junceella* show a negative relationship the Water Quality Index (Figure 5.6).

Table 5.7. Summary of redundancy analyses.

Hard corals				
Predictive variable	**df**	**f ratio**	**permuted P**	**%SS**
NRM Region	3	5.105	**0.0005**	35.885
Water Quality Index	1	2.687	**0.0015**	6.296
Distance to shore	1	1.676	0.0555	3.928
Soft corals				
Predictive variable	**df**	**f ratio**	**permuted P**	**%SS**
NRM Region	3	3.323	**0.0005**	28.401
Water Quality Index	1	1.746	0.0915	4.974
Distance to shore	1	0.383	0.942	1.092

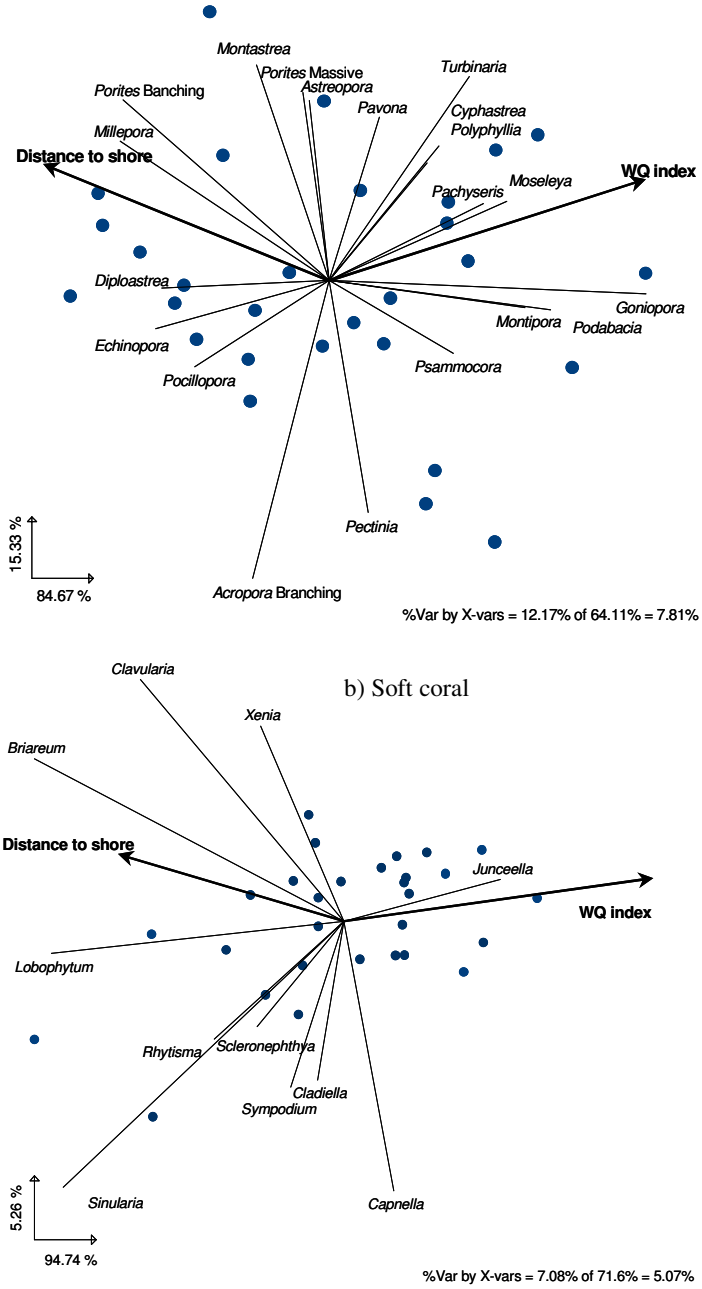

Figure 5.6. Biplots showing relationships between (a) hard coral and (b) soft coral communities and environmental variables at survey sites after the regional differences among communities have been removed by partial redundancy analysis. Vectors indicate the dimension of the greatest variation in values of an environmental variable or relative abundance of a genus. Higher values of the Water Quality Index indicate higher concentrations of the contributing variables. Genera whose vectors align most closely with the vector for an environmental variable are those whose abundances vary most closely with that variable; vectors in opposing directions indicate negative associations; vectors at right angles indicate little or no association.

Descriptions of coral communities on the survey reefs in each region

Information on the coral reef communities at sites in each NRM region is summarised in two subsections. The first focuses on the regions and describes the physical location of the survey reefs, describes any historical data sets available for reefs within the region, documents disturbances to local reef communities (Appendix 1: Table A1-5.2) and highlights any aspects of the coral communities that may relate to effects of run-off. The second sub-section provides more detailed descriptions of the coral communities at individual locations.

Because of the number and spread of sites, the Wet Tropics NRM region has been divided into three sub-regions: the Herbert / Tully, the Johnstone / Russell-Mulgrave and the Barron-Daintree sub-regions

Summary plots for the cover of major benthic groups and the density of recruit-sized coral colonies at each location are presented on a map of the region. Please note that regional averages are also presented in each location plot.

Finer scale taxonomic details of hard coral and soft coral communities and the absolute numbers of recruit-sized colonies are provided in Appendices 5.4 to 5.7.

These results represent a baseline for future studies.

Wet Tropics NRM region: Barron and Daintree sub-region

Five reefs were selected to represent a gradient of possible exposure to river-borne contaminants, which was assumed to be largely a function of their distance from the mouths of the Daintree and Barron Rivers. The location closest to the river mouth was the fringing reef on the southern side of Snapper Island, followed by reefs on the northern side of Snapper Island, then three groups of coastal fringing reefs extending from south of Cape Tribulation to just south of the Bloomfield River.

Reefs were also selected if any historical data on coral community composition and dynamics were available. All five survey locations have been monitored by Sea Research since 1985 (Cape Tribulation fringing reefs) or 1995 (Snapper Island reefs). With the exception of the Daintree North location a subset of the sites included in this study were surveyed with similar methods in 2004. The 2004 survey also included compilation of hard coral species lists (Sweatman et al. 2006). These historical observations provide background to the current status of the reefs by identifying and quantifying previous disturbance events. For example, in 2006 coral cover at Snapper Island was lower at sites on the southern side of Snapper Island compared with those on the northern side. Surveys by Sea Research show that coral cover had been steadily increasing at both southern and northern sites following disturbance from bleaching, cyclones and flooding until 2004 when the southern site was again inundated with freshwater during a flood, causing a marked decline in coral cover. There had been little recovery by 2006 and cover remained 30% lower than in the northern sites. Dynamics and composition of communities on these reefs are described in a series of reports to the GBRMPA (most recently: Ayling and Ayling (2005)). These historical sites were at a depth of approximately 2m below datum. All shallow (2m) sites included in this study use the exact same sites as monitored historically by Sea Research.

Monitoring of the reefs in this NRM region shows the frequent and localised nature of disturbances in this region. Between 1994 and 2006 disturbance events removed substantial proportions of the living coral at all five reefs (Ayling and Ayling 2005). The most severe recorded disturbance was caused by flooding of the Daintree River in 1996. During this flood existing hard coral cover was reduced by 87% on the southern side of Snapper Island and 20% on the northern side. There was no change in coral cover on the Cape Tribulation reefs 24km further north, possibly because the flood plume did not extend that far. Only one of the five reefs (Cape Tribulation south) has been infested by crown-of-thorns starfish.

The cover of hard coral at 2m depth is significantly lower at Daintree South and Snapper Island South than at other reefs. At Daintree South macroalgae cover was significantly higher than at all other reefs; cover of macroalgae at Daintree North was also higher than that observed at either side of Snapper Island (Figure 5.7). The cover of soft coral at 5m was significantly higher on the southern face of Snapper Island than on the northern reef, this pattern was reversed at 2m; the cover of soft coral at the Daintree Central was also higher than at Snapper South.

At 5m depth both the density of recruit-sized colonies and the overall richness of genera was lower on the southern compared to northern side of Snapper Island (Figure 5.8 and Appendix 1: Tables A1-5.5 and A1-5.7). At 2m the lowest density of recruit-sized colonies was at Daintree South though this differed statistically from Daintree North only. The low density of recruit sized colonies observed in early 2006 were lower than observed during surveys in

2004. The richness of genera was significantly lower at Snapper North than other reefs with the exception of Daintree South.

Low cover of hard coral at 2m at Snapper Island South is almost certainly the result of mortality associated with freshwater inundation during floods of the Daintree River in 1997 and 2004. The soft coral community was also impacted by these flood events (Appendix 1: Table A1-5.2, Ayling and Ayling 2005). Low coral cover at Daintree South was the result of a localised crown-of-thorns outbreak as was the ensuing high cover of macro-algae (Ayling and Ayling 2005). This high cover of macroalgae may be influencing the observed low density of recruit-sized colonies and as such limiting the rate of recover of the community.

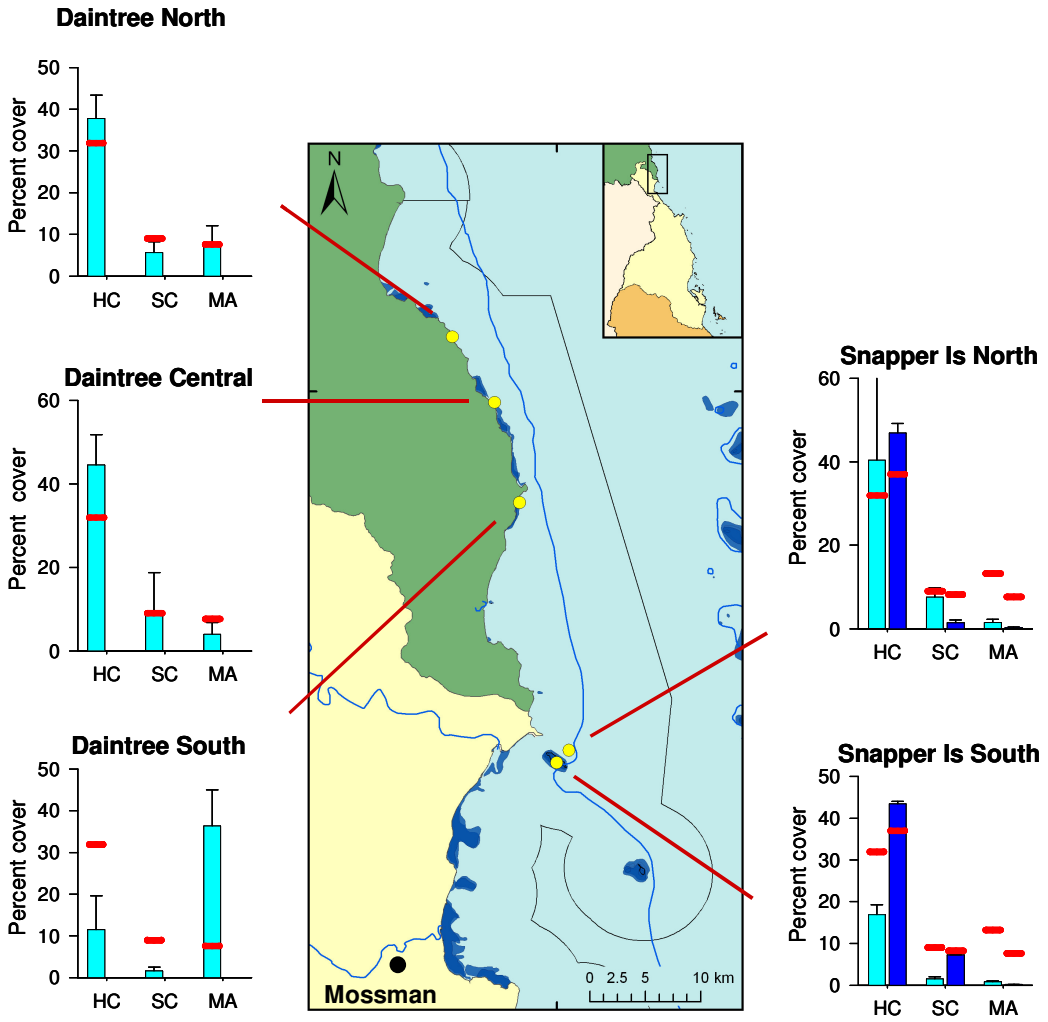

Figure 5.7. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in the Barron-Daintree sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

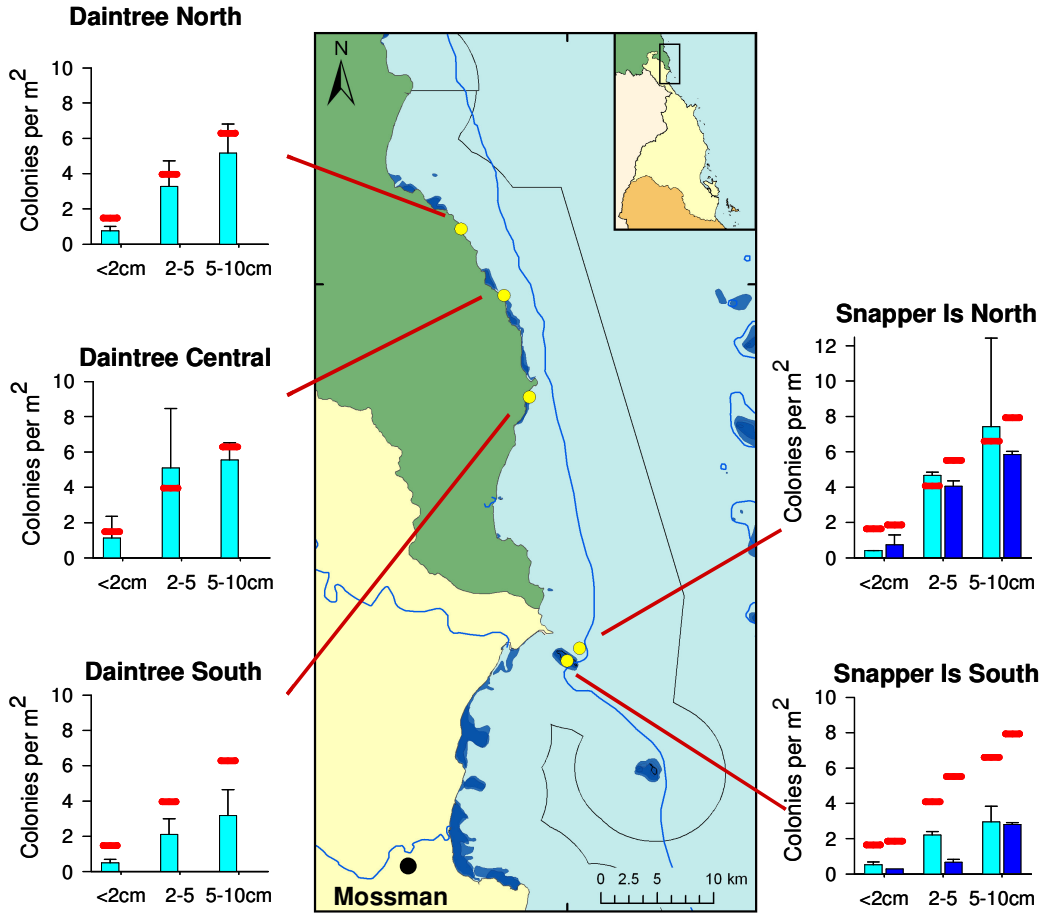

Figure 5.8. Density of recruit-sized hard coral colonies by size class for reefs in the Barron-Daintree sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines . For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Snapper Island South

The reef on the southern face of Snapper Island was severely damaged by flooding associated with cyclone Ethel in March 1996 that killed 87% of the hard coral community. The majority of corals killed by the flooding were in the genus *Acropora;* the absolute cover of this genus was reduced from 58% to just 1%. Other taxa that suffered substantial mortality included *Montipora* spp and members of the Pocilloporidae, though these were relatively uncommon. Cover of these taxa then increased until a second flood in 2004 reduced their cover again. The direct impact of these floods was restricted to the upper 2-3m of the reef as the deeper communities were largely unaffected. Our observations in 2006 confirm this, recording higher coral cover and a more diverse community, including higher proportions of flood-sensitive taxa such as *Acropora spp*. and *Pocillopora* spp., at 5m than at 2m (Figure 5.7, Appendix 1: Table A1-5.3).

Recruit densities were low in both deep and shallow sites. The average number of recruits per square metre was approximately half that recorded for all survey reefs (Figure 5.8). There was twice the density of recruits in the 0-5cm size range at shallow sites than at the deep sites, while densities of the 5 to 10cm size class were similar (3 per m^2) at each depth. This

contrasts observations in 2004 when the density of recruits was substantially higher at 5m. In light of a slight increase in cover at 5m between 2004 and 2006 it is possible some of this reduction in recruit density is due to growth of colonies into larger size classes. In 2006 the majority of recruit-sized colonies (<10cm) were either Poritidae (35%) or Faviidae (21%) while the combined genera *Acropora*, *Montipora* and *Pocillopora* contributed only 20% of recruits. The low densities of recruits at both depths suggest that recovery potential at this site is limited.

Snapper Island North

The reef on the northern face of Snapper Island was severely affected by bleaching in 1998 and by cyclone Rona in 1999. Hard coral cover fell from 80% in early 1998 to 70% immediately after the bleaching event in 1998, then to 20% after cyclone Rona. There has been steady recovery from these disturbances since 2001 (Ayling and Ayling 2005).

In 2005 hard coral cover was high at both 2m (40%) and 5m (47%) depths (Figure 5.7). The hard coral cover at 2m was dominated by Acroporidae (95% of observed cover) with small representation of Faviidae, Fungiidae and Oculinidae (Appendix 1: Table A1-5.3). In contrast, the community at 5m was more diverse with a more even mix of Agariciidae (29.6%), Poritidae (22.3%) and Acroporidae (24.1%, Appendix 1: Table A1-5.3). Recruit densities at both the shallow and deep sites was close to the average for all survey reefs (Figure 5.8). Recruit densities in the three size classes (0-2cm, 2-5cm and 5-10cm) were similar at both the 2m and 5m depths, but the proportion of recruits of Acroporidae was greater at the shallow site (Appendix 1: Table A1-5.5), possibly related to the higher proportion of *Acropora* spp. in the adult communities.

The difference in coral community structure between shallow and deep sites is likely to reflect differing disturbance regimes. Previous studies show the 2m site has been frequently damaged by both cyclones and floods, while the 5m site has been largely unaffected by floods (Ayling and Ayling 2005). The hard coral community at 2m was less diverse as a result, and in 2006 was dominated by fast growing *Acropora* spp. In comparison, the community at 5m had a more equal distribution of taxa, indicative of a site that has been disturbed less frequently. Coral cover at both shallow and deep sites is still below 1994 levels (Ayling and Ayling 2005). The numbers of recruits recorded in 2006 suggests that recovery is continuing.

Daintree South

Long-term data show distinct declines in coral cover corresponding to coral bleaching in 1998 and a crown-of-thorns starfish outbreak in 1999. Hard coral cover was 53% in 1997 but had fallen to 20% after the 1999 crown-of-thorns starfish outbreak and remained low up to 2006 (Ayling and Ayling 2005). Hard coral cover (11%) in this site was the second lowest and macroalgae cover (85%) was the highest for any reef surveyed in 2006 (Figure 5.7). The hard coral community was comprised predominantly of the family Acroporidae (75% of cover, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was low though 68% of the colonies were Acroporidae (Figure 5.8, Appendix 1: Table A1-5.6).

The coral community at South Daintree has remained largely unchanged since 2001. Previous surveys show that coral cover fell dramatically after the crown-of-thorns starfish outbreak in 1999 and has failed to recover (Ayling and Ayling 2005). Community composition is similar to the middle and northern Daintree reefs, but the lower numbers of coral recruits, combined with a higher macroalgae cover, suggest recovery at the southern site may be slow.

Daintree Central

Coral cover has remained around 60% since 1995 after increasing from a low point of around 40% following cyclone Manu in 1986 and coral bleaching in 1987 (Ayling and Ayling 2005). The only disturbance in the decade 1995 to 2005 was due to coral bleaching in 1998 (14% decrease, Appendix 1: Table A1-5.2). Community composition changed with the bleaching in 1998: cover of *Pocillopora* spp decreased significantly. The more abundant *Acropora* spp and *Montipora* spp recovered quickly from both events (Ayling and Ayling 2005).

In early 2006 the cover of hard coral (45%) was below the long term average though this is in part due the video technique returning slightly lower estimates of coral cover than the line intercept technique (LIT) used historically. Comparison to LIT data shows only a minor reduction in cover between 2005 and 2006 (Ayling pers. comm.). Cover of soft coral and macroalgae were close to the average for all survey reefs (Figure 5.7). Recruit density was similar to the overall average from all near-shore reefs (Figure 5.8).

Historical and current observations indicate Daintree Central reefs are very stable and unlikely to change dramatically in the short term. Despite a range of past disturbances including cyclones, floods and coral bleaching, overall hard coral cover has quickly recovered to pre-disturbance levels and, with the exception of bleaching in 1998, community composition has also remained stable. Recruitment levels in 2005 were average for all three size-classes and overall hard coral cover may be returning to the long-term average.

Daintree North

As at Daintree Central, the benthic communities on Daintree North reefs have remained stable over the past 20 years. Hard coral cover has remained around 60%, with brief decreases associated with cyclone Manu in 1986 (29% decrease) and Coral Bleaching 1998 (9% reduction) 2002 (8% reduction). Community composition changed with the bleaching event in 1998 when *Pocillopora* spp decreased significantly. The more abundant *Acropora* spp and *Montipora* spp recovered quickly (Ayling and Ayling 2005) and have increased their dominance.

Observations in 2006 showed that the community has remained relatively stable over previous year. Though hard coral cover (38%) appears to have fallen substantially below the long-term average (60%) this is due in part to consistent differences between the image based method used in this program and the line intercept technique used historically. LIT data collected in 2005 and 2006 suggest only a slight reduction in coral cover (Ayling pers. comm.) Soft coral and macroalgae cover remained similar to the average for all survey reefs (Figure 5.7). Recruit density was marginally below the average for all near-shore reefs in all size classes (Figure 5.8).

Daintree North reefs seem to have very stable benthic communities: they have experienced a range of disturbances in the past two decades including cyclones, floods and coral bleaching, but total hard coral cover has quickly recovered to pre-disturbance levels. Community composition has also remained stable except for the decrease in *Pocillopora* spp. from the bleaching event in 1998.

Wet Tropics NRM region: Johnstone and Russell - Mulgrave sub-region

The influence of the Johnstone and Russell-Mulgrave Rivers extends in a northerly direction as plumes and sediments are transported by prevailing winds. There are not many reefs in this area; those closest to the rivers are those on the western side of High Island, followed by the eastern fringing reefs. Reefs on the western and then eastern sides of the Frankland group are less likely to be influenced due to their greater distance from shore. The fringing reefs on the western and eastern sides of Fitzroy Island are further again from the mouths of Johnston and Russell-Mulgrave Rivers. While the relatively small Wet Tropics rivers are closest to these reefs and are consistent sources of runoff, infrequent major floods from the Burdekin River can also extend this far north as occurred in 1994.

The coral reef communities in this region have been studied relatively intensively. The Frankland group and Fitzroy Island have been monitored since 1995 (Ayling and Ayling 2005) and 1992 (Sweatman *et al.* 2005) respectively. Two studies have related gradients in water quality to rates of coral settlement: Smith *et al.* (2005) examined communities at one depth (5m) over a two year period (1999 and 2000) at High Island, the Frankland group and Fitzroy Island. Fabricius (AIMS unpublished data) studied coral recruitment at two depths (5 and 8m) on fringing reefs of the same islands during 2002 and 2003. Sites at both the Frankland group and High Island included in this study are essentially the same as those surveyed in 2004 using a similar suite of methods though also including the compilation of species list for hard corals (Sweatman et al. 2006)

Three major disturbances have affected survey reefs in this NRM region in recent times, coral bleaching in 1998 and in 2002, and crown-of-thorns starfish (COTS) in 1999-2000 (Appendix 1: Table A1-5.2). In 1998 bleaching is known to have affected all coral communities on the target reefs in this NRM region. The eastern reef of the Frankland group suffered the greatest coral mortality with a 44% decrease in cover followed closely by the western reef were cover decreased by 43%. Fitzroy Island and the Frankland group both suffered a major reduction in coral cover due to COTS in the period 1999-2000: western reef slope communities at Fitzroy Island lost 78% of their hard coral and the eastern reef communities of the Frankland group lost 68%. Bleaching in 2002 was less severe than in 1998 but still affected most coral communities in some way. Freshwater plumes associated with major flooding were recorded at most reefs in 1994, 1995, 1996, 1997 and 1999 (Devlin *et al.* 2005) though there were no marked impacts on coral cover directly attributable to these events at the depth of monitoring sites. Temporal profiles of coral cover for Fitzroy Island and the Frankland group are presented in Sweatman et al. (2006).

There are significant variations in estimates of community summary statistics among reefs in this NRM region. In 2005 the western reefs of both High Island and the Frankland Group stood out generally as having lower densities of recruit-sized colonies (Figure 5.9), richness of genera represented by recruit-sized colonies and richness over all (Appendix 1: Table A1-5.7). The average number of coral recruits settling to tiles was also lowest at Frankland Group West (Table 5.4) and while low or absent on most reefs the cover of macroalgae was highest at this reef (Figure 5.8). In contrast the cover of hard corals was highest on the western then eastern reefs of High Island and lowest at Fitzroy Island; the western reefs at Fitzroy Island had significantly higher cover of soft corals than all other reefs (Figure 5.9).

The western reefs of High Island and the Frankland Group are likely to be most exposed to river borne contaminants due to a combination of their proximity to rivers and likely higher

sedimentation regimes than reefs on the eastern sides of these islands (Wolanski *et al.* 2005) and this may be reflected in the generally lower richness and density of recruits at these reefs.

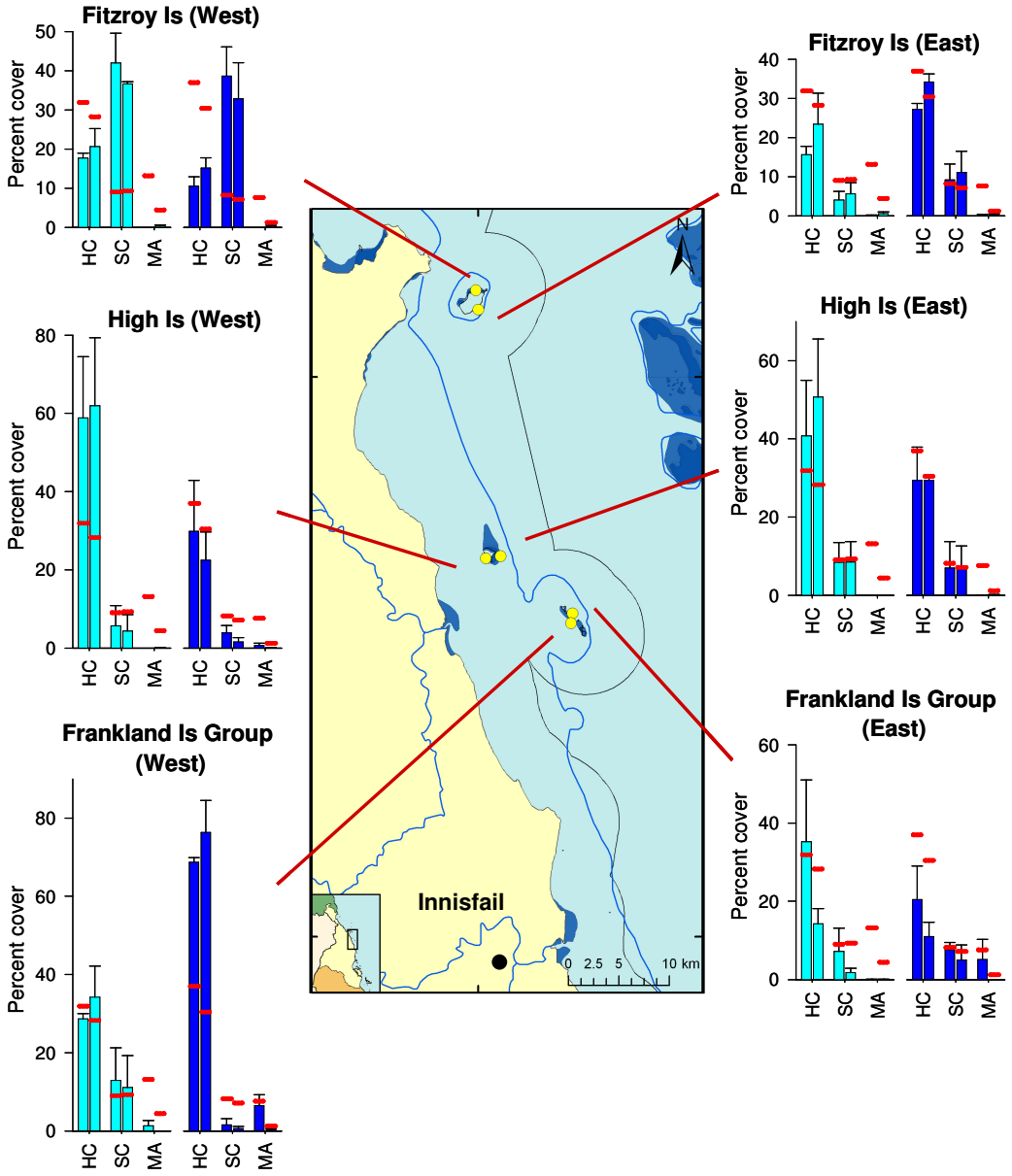

Figure 5.9. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in the Johnstone / Russell-Mulgrave sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

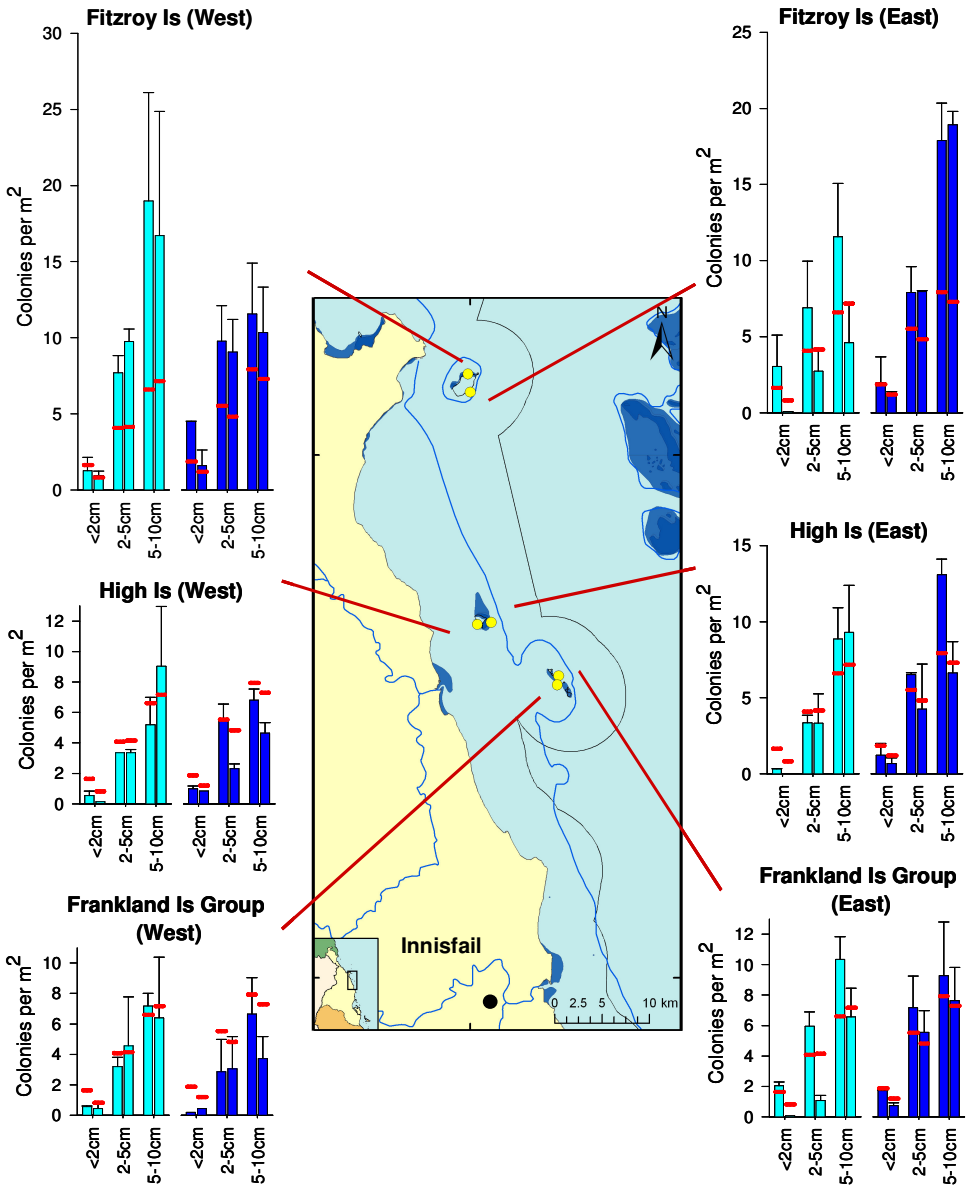

Figure 5.10. Density of recruit-sized hard coral colonies by size class for reefs in the Johnstone / Russell-Mulgrave sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines .For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Comparing between surveys in 2004 (Sweatman et al. 2006) and 2005 indicates a period of recovery at both the Frankland group and High Island. This was especially evident on eastern reefs where hard coral cover increased notably and especially at 2m were there was a high component of the fast growing *Acropora* genus. There was also a general increase in the density of recruit sized colonies over this period indicating continued recruitment. The western reef of the Frankland group was an exception with little change in coral cover.

Between surveys in 2005 and 2006 there was a substantial reduction the density of recruit-sized colonies at 2m at Frankland Group East and Fitzroy Island East and 5m at High Island East (Figure 5.9). There was a corresponding reduction in the richness of genera represented by recruit-sized colonies at 5m at High Island East and 2m at Frankland Group East (Appendix 1: Table A1-5.7). Hard coral cover also declined at Frankland Group East 2m. The declines at Frankland Group East can be explained by the severe disturbance to site 1 caused by the passage of TC Larry. At Fitzroy Island East there was little obvious damage from the cyclone and the increase in hard coral cover supports the supposition that some of the decline in recruit density may be the result of growth of a strong cohort into large sized classes. Bleaching was more prevalent at High Island East in 2006 than at other reefs in this NRM region and it is possible that some mortality of small colonies occurred.

High Island West

The coral reef community along the, more sheltered, western side of High Island has consistently been inundated by fresh water plumes from the Russell-Mulgrave and Johnstone rivers. This reef appears to have been badly damaged by coral bleaching in 1998 and 2002 and the COTS outbreak in 1999-2000 may have caused additional mortality, however there is no data on effects of these disturbances from this reef.

Historical observations of coral recruitment to settlement tiles have been variable ranging from <2 recruits per tile in 2000 to 41.2 in 2005 (Table 5.5). In 2005 there were 34.5 recruits per tile at 2m; predominantly Acroporidae (94%), Pocilloporidae (3.4%) and Poritidae (1.3%) (Table 5.4). Tiles at 5m recorded 47.9 recruits on average and had a similar composition of taxa; Acroporidae (92%), Pocilloporidae (4.3%) and Poritidae (1.2%). This numerical dominance by the family Acroporidae was common to past observations with the exception of 2003 when Acroporidae numbers were low resulting in roughly similar proportions of Acroporidae and Pocilloporidae (K.Fabricius unpublished).

In 2005 the benthic communities at 2m had less than average cover of soft coral and very high cover of hard coral (59%, Figure 5.9). The hard coral community on the shallow slope was dominated by massive *Porites* spp. representing 83% of all corals; only 13% of corals were Acroporidae (Appendix 1: Table A1-5.3). The density of recruits in all size classes was below average (Figure 5.10) with colonies from the families Faviidae, Acroporidae and Poritidae most common (Appendix 1: Table A1-5.5). The benthic communities at 5m had less than the average cover of hard coral (30%), 85% of which consisted of large colonies of massive and branching *Porites* spp. (Figure 5.9, Appendix 1: Table A1-5.3). The density of recruits of all size classes was at or slightly below average (Figure 5.10). The majority of these recruits were from the families Faviidae and Poritidae (Appendix 1: Table A1-5.5).

In March 2006 TC Larry passed close to these sites but appeared to have only slight impact on the cover of the more fragile *Acropora* spp. on the shallow slope. At 5m several large colonies of *Porites cylindrica* were dislodged and tumbled down the slope, reducing the cover of this species, and contributing to an overall drop in cover of Poritidae.

Resurveys in May 2006 found that soft coral cover had changed very little; hard coral cover had increased marginally at 2m and decreased at 5m (Figure 5.9). The density of recruit-sized colonies had also increased marginally at 2m and decreased at 5m (Figure 5.10). The numbers of recruits from the families Poritidae and Faviidae showed the largest decline at 5m (Appendix 1: Table A1-5.6).

There is an obvious disjunction between the dominance of the adult coral community by large colonies of *Porites*, a high proportion of *Porites* amongst the recruit-sized colonies and the vastly higher numbers of Acroporidae settling to tiles. These data suggest that Porites are more resilient to the environmental conditions at this location. While high numbers of settling, and increasing numbers of recruit-sized, Acroporidae indicate a likely increase of this family in the short term it is equally likely that these colonies are fated to being a transient part of the community in the current environmental setting.

High Island East

The coral reef community along the eastern side of High Island has a similar disturbance history to the western reef consistent inundation by fresh water plumes as well as a high probability of mortality from bleaching (Appendix 1: Table A1-5.2).

In 2005 the benthic communities at 2m had moderate cover of soft coral (8%) and high cover of hard coral (Figure 5.9). The hard coral community on the shallow slope was dominated by large *Acropora* spp. which made up 74% of all corals; only 16% of corals were Poritidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for the <5cm size classes but above average for the 5-10cm size class (Figure 5.10) and dominated by Faviidae, Acroporidae and Poritidae (Appendix 1: Table A1-5.5). The benthic communities at 5m had similar cover of soft coral (7%) and a below average cover of hard coral (29%, Figure 5.9). The composition of the hard coral community differed from that at 2m with 59% consisting of mostly large colonies of massive and sub-massive *Porites* spp and 24% from the family Acroporidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for the >2cm size classes but above average for the large size classes especially those 5-10cm in diameter (Figure 5.10). Very large numbers of recruits from the Poritidae, Faviidae and Acroporidae were present (Appendix 1: Table A1-5.5).

There appeared to be little or no impact of TC Larry at 2m. In 2006 the soft coral cover at had not changed while hard coral cover had increased by 24% (Figure 5.9). The hard coral community on the shallow slope was dominated by large tabulate, corymbose and branching *Acropora* spp. that now made up 80% of all corals; only 11% of corals were Poritidae, a reduction from the previous year (Appendix 1: Table A1-5.3). The density of recruits was very similar to the previous year, dominated by Acroporidae, Faviidae and Poritidae (Figure 5.10, Appendix 1: Table A1-5.3). At 5m there was some bleaching most likely due to a prolonged period high turbidity due to strong winds (including TC Larry) and local flooding. Neither hard coral or soft coral cover had changed from levels observed in 2005 (Figure 5.9) thought the composition of the hard coral community saw an increase in the family Acroporidae and comparative decrease in Poritidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for all size classes, with a large drop in 5-10cm recruits, probably because they grew beyond the size class (Figure 5.10).

The presence of large colonies of various growth forms of *Acropora* spp indicated that the reef was recovering rapidly from previous disturbances. The coral community appears to be shifting from one dominated by *Porites* spp to one including a high proportion of *Acropora* spp.

In 2005, there were an average of 87.4 recruits per tile at 2m and 86.8 per tile at 5m, at both depths the family Acroporidae made up the bulk of recruits recorded with the families Pocilloporidae and Poritidae also represented (Table 5.4). This number of recruits is more than an order of magnitude higher than recorded in previous studies between 1999-2003

(Table 5.5). With such high levels of recruitment in 2005 it will be interesting to see how this translates into the number of recruit-sized colonies in future surveys.

Frankland Group West

Studies of the back reef slope community date from January 1995 (Ayling and Ayling 2005). The hard coral cover then was 80% and bottlebrush *Acropora* spp made up 60% of all hard corals. Coral cover remained high until bleaching in 1998 when the community shifted from domination by *Acropora* spp. to one dominated by *Porites* spp. After bleaching in 1998 and a COTS outbreak in 1999-2000 the more resilient Poritidae flourished and the cover of this family continued to increase as cover of Acroporidae diminished. In the latest surveys of these sites in March 2005, hard coral cover was recorded as 40% with 80% of hard corals represented by Poritidae and only 2% Acroporidae (Ayling and Ayling 2005).

Shallow sites used in this program correspond closely in location and depth to sites that have been surveyed annually by Sea Research (Ayling & Ayling 2005). In 2005 the benthic communities at 2m had above average cover of soft coral (13%) and below average cover of hard coral (29%, Figure 5.9). Eighty percent of the hard coral community on the shallow slope was from the family Poritidae; only 8% was from the family Acroporidae (Appendix 1: Table A1-5.3). The density of recruits was below average for colonies <2cm in diameter and near average for the larger recruit size classes (Figure 5.10). Most recruits were from the family Poritidae (Appendix 1: Table A1-5.5). The benthic communities at 5m had high cover of hard coral, 96% of which of the family Poritidae (mostly large colonies of *Porites rus*, Figure 5.9, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average and again these were mostly from the family Poritidae (Figure 5.10, Appendix 1: Table A1-5.5). No Acroporidae recruits were found.

In March 2006 TC Larry passed almost directly over the Frankland Group but appeared to cause little damage to the western reef. In May 2006 at 2m the soft coral cover had not changed and hard coral cover had increased by 19% and there was a decrease in macroalgae cover from 4% to below 1% compared to that observed in 2005 (Figure 5.9). The hard coral community on the shallow slope was still dominated by *Porites* spp. though to proportion represented by the family Acroporidae fell from 8% to 4% (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for the <2cm and 5-10cm size classes, but above average for the 2-5cm size class (Figure 5.10). Most recruits were Poritidae, but the number of Acroporidae recruits had increased threefold since 2005 (Appendix 1: Table A1-5.5). The cover of hard coral at 5m had increased by 11% from the previous year and community composition was largely unchanged (Figure 5.9, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was again below average (Figure 5.10).

An average of 18 recruits per tile settled at 2m in 2005 these mostly of the families Acroporidae (78%), Pocilloporidae (20%) and Poritidae (1.9%). At 5m the average number of recruits per tile was 30.1 recruits per tile again these were mostly from the families Acroporidae (86%), Poritidae (12%) and Pocilloporidae (2.3%, Table 5.4). These values are well above those recorded from this reef in previous studies (Table 5.5) though the lowest among reefs surveyd in this region in 2005.

Large stands of *Porites rus* and *Porites cylindrica* have continued to expand and dominate the reef slope communities especially at 5m. These species have proved resilient to the various disturbances imposed on the coral communities at this reef. That the supply of coral larvae form other taxa was high it will be informative to see if this translates into the development of

a more diverse community or if disturbance and competition for space continue to limit the community diversity.

Frankland Group East

The eastern reef slope community was first surveyed in April 1998 (Ayling and Ayling 2005). At this time hard coral cover was 53% and dominated by corymbose plate and branching *Acropora* spp (62% of all hard corals). Bleaching in 1998 removed 43% of the coral cover with the families Acroporidae and particularly Pocilloporidae suffering the greatest losses. During 1999-2000 crown-of-thorns starfish caused a further 68% decrease in coral cover and again mainly affected the family Acroporidae by December 2000 coral cover was low at just 10%. Cover of the more bleaching-resistant and unpalatable Poritidae and Faviidae remained unchanged. In the most recent surveys in March 2005 hard coral cover had recovered to 25% of which 60% was of the family Acroporidae (Ayling and Ayling 2005). The sites surveyed in this study correspond closely to the monitoring sites surveyed annually by Sea Research (Ayling and Ayling, 2005).

In 2005 the benthic community at 2m had marginally below average cover of soft coral (7%) and an above average cover of hard coral (35%, Figure 5.9). The hard coral community was dominated by the family Acroporidae which made up 84% of coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average and dominated by Acroporidae and Faviidae (Figure 5.10, Appendix 1: Table A1-5.5). At 5m the cover of SC was similar and hard coral cover well below average (Figure 5.9). As at 2m the coral community was dominated by *Acropora* spp. (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average and included a diverse range of families though most abundant were Poritidae, Faviidae, Acroporidae and Oculinidae (Figure 5.10, Appendix 1: Table A1-5.5).

In March 2006 the benthic communities suffered a major setback due to TC Larry. In May 2006 the cover all major benthic groups was markedly reduced at both depths, the most notable change was a 60% reduction in cover of hard corals at 2m (Figure 5.9). The hard coral community on the shallow slope was still dominated by *Acropora* spp but the cover of Acroporidae dropped substantially (Appendix 1: Table A1-5.3). The density of recruit-sized colonies at 2m was also reduced to very low levels in the smaller <5cm sizes (Figure 5.10).

Coral settlement in 2005 averaged 93.8 recruits per tile at 2m and 99.5 recruits per tile at 5m; the highest recorded for any of the study reefs (Table 5.4). At both depths settling corals were mostly of the family Acroporidae, with Pocilloporidae and Poritidae also represented (Table 5.4). The number of recruits per tile in 2005 greatly exceeded the levels recorded past studies (Table 5.5). The highest level of settlement observed previously was in 2002 when an average of 6.3 recruits per tile was recorded. At this time the composition of recruits was similar to that observed in 2005; Acroporidae (80%), Pocilloporidae (11%) and Poritidae (3.1%). Given observations of recovery from past disturbance in monitoring data and average numbers of recruit-sized colonies remaining at 5m following TC Larry it is likely this community will begin to recover in the near future.

Fitzroy Island West

Broadscale monitoring of the whole perimeter of Fitzroy Island using manta tow was first carried out in 1986 (Sweatman *et al.,* 2005). More detailed monitoring sites were set up by AIMS on the western side of Fitzroy Island in 1994 at which time hard coral cover was 25%. Coral cover increased to 35% by 1997 before a series of disturbances (Appendix 1: Table A1-

5.2). Hard coral cover declined by 13% following bleaching in 1998, then 78% over 1999-2000 as a result of a COTS outbreak, and a further 15% following bleaching in 2002. Most of the decrease was through mortality of Acroporidae that are more susceptible to bleaching than some other taxa and are preferred prey of COTS, though a similar pattern of decline was observed in many other hard coral families with the exception of Poritidae. In 2005 hard coral cover had increased to 15% and soft coral cover was 25%.

The AIMS Long-term Monitoring Program (LTMP) sites are located on the reef slope just below the deep (5m) slope sites of this study. In 2005 the benthic communities at 2m had high cover of soft coral and below average cover of hard coral (Figure 5.9). The hard coral community at 2m was dominated by the families Acroporidae and Poritidae that represented 54% and 22% of the cover respectively (Appendix 1: Table A1-5.3). At 5m the benthic community was dominated by soft corals (39%), the cover of hard coral was low (Figure 5.9). The hard coral community was principally made up of colonies of *Porites* spp (60%); only 13% of all corals were from the Acroporidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies across all size classes was above average at both depths (Figure 5.10) At 2m the majority of recruit-sized colonies were from the families Poritidae, Faviidae and Acroporidae, while at 5m recruits from a diverse range of families were present (Appendix 1: Table A1-5.5).

In May 2006 the cover of soft corals had declined slightly at both depths with compensatory increases in the cover of hard corals (Figure 5.9). At 2m the increase in coral cover resulted in a slight increase in the proportion of the family Acroporidae in the community while at 5m the family Faviidae increased the most (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was again well above average at both depths (Figure 5.9).

In 2005 coral settlement averaged 87.2 recruits per tile at 2m and 74.6 recruits per tile at 5m; at both depths the family Acroporidae was most common with Pocilloporidae and Poritidae also in reasonable numbers (Table 5.4). The numbers of recruits per tile recorded in 2005 greatly exceed those recorded previously (Table 5.5).

The high abundance and diversity of recruit-sized colonies, coupled with the increasing cover of Acroporidae (especially at 2m) indicate that the back reef at Fitzroy Island. is in a state of recovery. The availability of larva as evidenced by the level of settlement to tiles should further enhance this process.

Fitzroy Island East

The reef community on the eastern side of Fitzroy Island has been monitored by AIMS using manta tow since 1986. This data shows two large disturbances to the coral community; 75% of the coral cover was lost in the period early 1989 and January 1990 most likely due to the close passage of cyclone Felicity, then, after a period of slow recovery, an estimated 90% of the coral cover was lost following bleaching in 1998 (Appendix 1: Table A1-5.2).

In 2005 the benthic communities at 2m had below average cover of both hard coral and soft coral and no recorded macroalgae (Figure 5.9). The hard coral community was dominated the family Acroporidae with Poritidae and Faviidae well represented (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average for all size classes with the families Faviidae, Poritidae and Acroporidae all abundant (Figure 5.10, Appendix 1: Table A1-5.5). At 5m soft coral cover was moderate and hard coral cover below average (Figure 5.9). The density of recruit-sized colonies was very high (Figure 5.10). The families Acroporidae,

Poritidae and Faviidae were well represented by both relative cover and numbers of recruit-sized colonies (Appendix 1: Tables A1-5.3, A1-5.5).

In May 2006 the cover of both hard corals and soft corals had increased at both depths (Figure 5.9). At 2m the increase in hard coral cover was due mostly to increase in the cover of the family Acroporidae that now accounted for 79% of the cover (Appendix 1: Table A1-5.3). In contrast to the increase in cover the density of recruit-sized colonies declined, though some of this is almost certainly due to growth into larger size classes (Figure 5.10). At 5m increase in coral cover was again due primarily to an increase in the family Acroporidae though Poritidae and Pocilloporidae also increased (Appendix 1: Table A1-5.3). The density of recruit size colonies was still very high and included a range of families (Figure 5.10, Appendix 1: Table A1-5.5).

In 2005 there was an average of 39.1 recruits per tile at 2m and 44.9 recruits per tile at 5m; at both depths the family Acroporidae made up the bulk of the recruits, though at 5m there was also a large number of Pocilloporidae (Table 5.4). The average number of recruits per tile was well above that recorded in previous studies (Table 5.5).

The high abundance and diversity of recruit-sized coral colonies coupled with the increasing cover of Acroporidae (especially on the shallow slope site) and the bleaching sensitive Pocilloporidae (on the deep slope) indicates that the front reef at Fitzroy is in a state of recovery. In the absence of any major disturbances in the near future the cover and diversity of the hard coral community should continue to increase.

Wet Tropics NRM region: Herbert and Tully sub-region

Four reefs were selected to represent a gradient of likely exposure to river borne contaminants, which is assumed to be largely a function of their distance from the mouths of the Herbert, Murray, Hull and Tully Rivers. The location the most proximate to riverine inputs is the fringing reef on the southern side of Dunk Island, followed by reefs on the northern side then King Reef and lastly the North Barnard Group. Plumes from these rivers are known to flow predominantly northwards. In very wet years all four reefs may also be subject to flooding from the Burdekin river located 250km further south (Devlin & Brodie 2005).

There is little available information describing coral communities of the North Barnard Group. Sites at King Reef and Dunk Island North are very close to but not exactly the same as sites surveyed in 2004 using a similar suite of methods with the added inclusion of hard coral species lists (Sweatman et al. 2006). Sweatman et al. also reports surveys at Dunk Island South though site 1 of that survey does not correspond to site 1 under this study. Comparison between these 2004 data and data collected in 2005 indicate little change in the coral communities over the intervening year. AIMS also holds unpublished data from both sides of Dunk Island (K. Fabricius pers. com).
Flood plume observations by Devlin *et al*. (2001) show all reefs were subject to flood events on three or more occasions between 1991 and 2001 (Appendix 1.5.3). Dunk Island reefs were inundated by flood plumes annually between 1994 and 1998, North Barnard Group reefs on three occasions between 1994 and 1997 and King Reef on four occasions between 1994 and 1997. The impacts on the benthic communities are unknown.

Recent modelling work indicates hard coral communities in this sub-region were all likely to have been impacted by coral bleaching in 1998 and 2002 (Appendix 1.5.3). Similar reductions in hard coral cover (43%) to those observed by Ayling and Ayling (2005) at the Frankland Island Group in 1998 are quite possible.

In 2005 the communities at both 2m and 5m at King Reef had significantly lower estimates of hard coral cover, lower density of recruit-sized colonies and significantly higher cover of macroalgae than either reefs at either Dunk Island north or the North Barnard Group (Figure 5.11,. The richness of genera represented by recruit-sized colonies was also low at King Reef (Appendix 1: Table A1-5.7). The cover of hard coral at 2m and density of recruit-sized colonies at 5m was also lower at Dunk Island south that at either Dunk Island north or the North Barnard Group while the cover of macroalgae was higher. The reef at Dunk Island South is closest to the rivers that might affect these reefs. While King Reef is further from the main rivers than the sites at Dunk Island north, it may be affected by a local creek.

The impact of TC Larry (March 2006) on these coral communities varied dramatically (Figure 5.11, Appendix 1: Table A1-5.2). Coral cover was significantly reduced at both the North Barnard Group (95% decline at 2m, Appendix 1: Table A1-5.2) and Dunk Island north (65% loss at 5m). Soft coral cover was also significantly reduced at these reefs (Figure 5.11). The density of recruit-sized colonies was also significantly reduced (Figure 5.12). In addition to loss of cover, the richness of genera both over all and for those represented by recruit-sized colonies was also reduced at these heavily impacted reefs (Appendix 1: Table A1-5.7). The cover of macroalgae was reduced at King Reef and Dunk Island north (Figure 5.11). It will be interesting to relate the recovery trajectories for these impacted reefs to measures of local water quality regimes over the coming years.

Dunk Island North

In March 2006 TC Larry passed close to Dunk Island and had a dramatic impact on the benthic community. The most notable changes were substantial reductions in; hard coral cover, cover of macroalgae at 2m and cover of soft corals at 5m. Following the cyclone the cover of all major benthic components was well below the average for near-shore reefs (Figure 5.11). There was also a general decline in the density of recruit-sized colonies (Figure 5.12) but this was due to the marked increase in area of available substrate caused by the loss of coral cover.

In 2005, prior to TC Larry, the benthic community at 2m featured high cover of hard coral, moderate cover of macroalgae and very low cover of soft corals (Figure 5.11). Hard coral cover was mostly of the family Acroporidae (85%, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above the overall average for near-shore reefs in all size classes (Figure 5.12). Coral cover was markedly reduced following TC Larry, with the large stands of Acroporidae reduced to dead rubble and scattered fragments of live coral. It is the presence of these fragments that largely explains why there was little change in the density of recruit-sized colonies given the vastly increased area of available substrate following the removal of coral cover. In addition to the high number of surviving fragments there was also a substantial increase in the number of recruit-sized colonies in the family Faviidae (Appendix 1: Table A1-5.5), presumably these small robust colonies were present in 2005 though over looked due to the over-growth of the now removed Acroporidae.

On the deeper slope (5m) prior to TC Larry hard coral cover was slightly lower than at 2m and the community more diverse though still dominated by the Family Acroporidae (Figure 5.11, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was also very high with recruits in the families Dendrophylliidae and Faviidae well represented. Following the cyclone the cover of the major benthic groups and also the density of recruit-sized colonies were markedly reduced (Figure 5.11, 5.10). Of the families represented by a high number of recruit-sized colonies prior to the cyclone Dendrophylliidae suffered the greatest decline with 88% of colonies lost (Appendix 1: Table A1-5.5).

That there is no quantitative data on the dynamics of this community prior to 2004 (Sweatman *et al*., 2006) limits our ability to determine the recovery potential at this reef. However, the high number of surviving *Acropora* fragments at 2m should see a reasonably rapid recovery toward the pre cyclone *Acropora* dominated community at this depth as these fragments grow. Similarly that there was still near average numbers of recruits at 5m indicates the recovery potential inherent in this community.

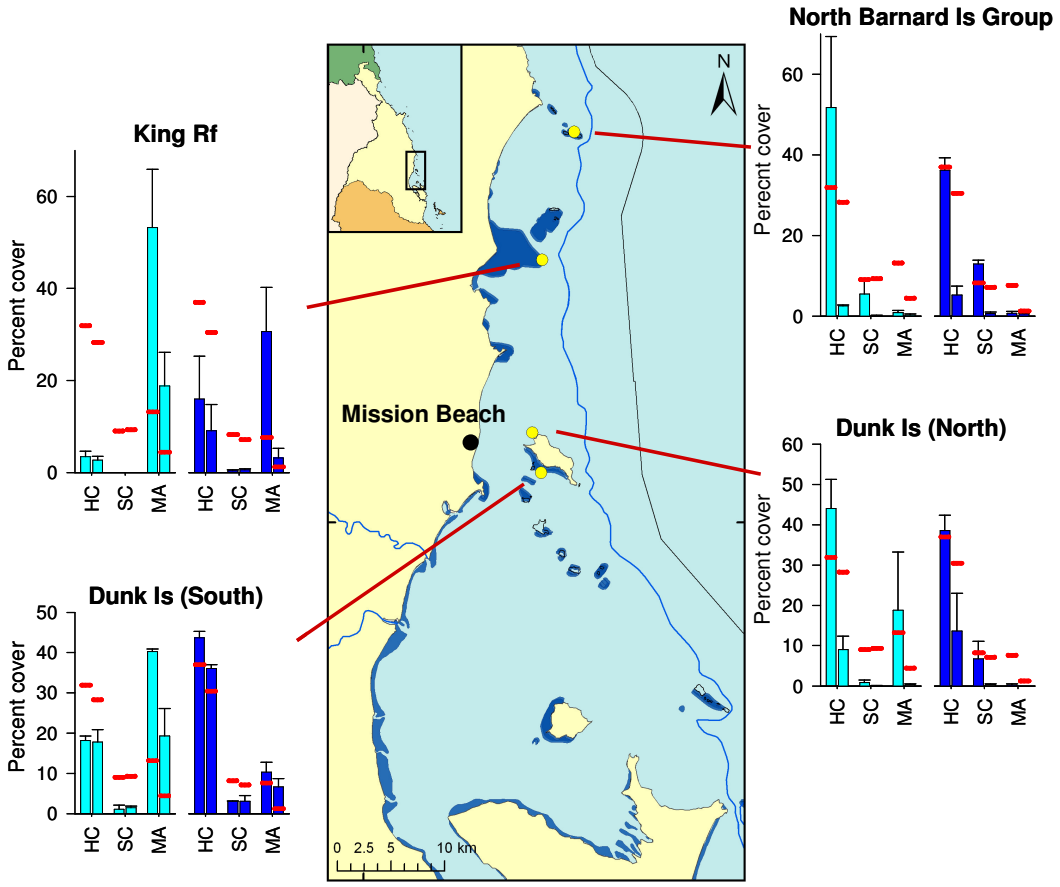

Figure 5.11. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in the Herbert and Tully sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

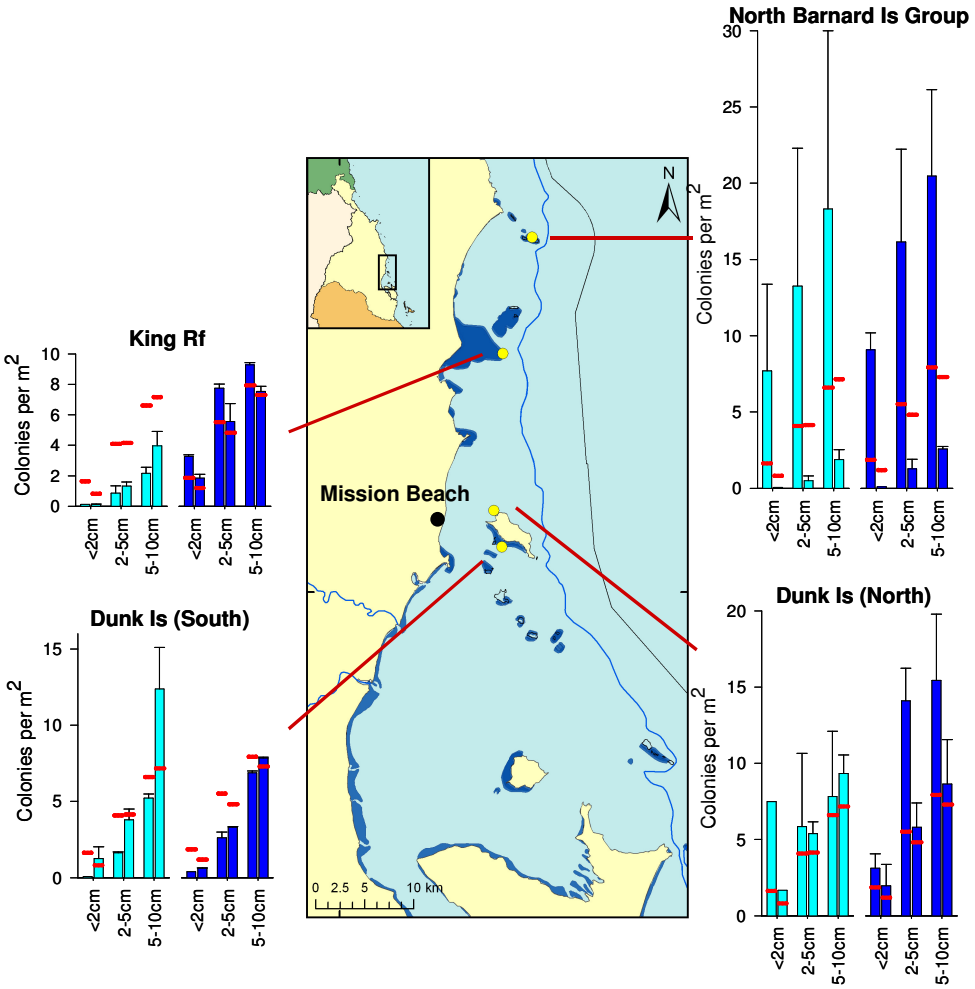

Figure 5.12. Density of recruit-sized hard coral colonies by size class for reefs in the Herbert and Tully sub-region of Wet Tropics NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines. For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Dunk Island South

There are no published data on the dynamics of the benthic community of this reef prior to 2004 (Sweatman *et al.* 2006) however AIMS holds unpublished data (Fabricius,). While the direct evidence of disturbance has not been quantified, flood plumes were observed to inundate this reef annually between 1994 and 1998 it is also likely that the reef was impacted by bleaching in 1998, and to a lesser degree in 2002 (Appendix 1: Table A1-5.2). Most recently the passage of TC Larry in 2006 is likely implicated in a slight decline in coral cover at 5m (Figure 5.11).

In 2005 the benthic communities at 2m had very low cover of soft coral, below average cover of hard coral and very high cover of macroalgae (Figure 5.11). The hard coral community on the shallow slope was dominated by colonies from the families Acroporidae, Poritidae and

Dendrophylliidae, which combined represented 79% of all corals (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average in all size classes (Figure 5.12); colonies from the families Faviidae and Acroporidae were most abundant (Appendix 1: Table A1-5.5). At 5m depth the cover of hard coral was markedly higher and macroalgae markedly lower than at 2m, soft corals were again relatively rare (Figure 5.11). The hard coral community at 5m was diverse with the families Faviidae (21%), Acroporidae (17%) and Dendrophylliidae (14%) best represented (Appendix 1: Table A1-5.3). As at 2m the density of recruits was below average in all size classes (Figure 5.12) though here colonies of the families Faviidae and Poritidae were most abundant (Appendix 1: Table A1-5.5).

Compared to other reefs in this NRM region the benthic community was not severely impacted by TC Larry. There was a decline in the cover of macroalgae though this could be due to unrelated seasonal or inter-annual fluctuations typical of this group (McCook *et al*., 2001). Lower cover of macroalgae may also explain the slightly higher density of recruit-sized colonies at both depths as macroalgae tends to obscure some colonies from observation. The decline in hard coral cover at 5m was due to coral bleaching most likely as a response to an extended period of high turbidity due to both re-suspension of sediments and local flooding associated with TC Larry and also an extended period of strong SE trade winds following the cyclone. On the whole the community appears stable however marginally low density of recruit-sized colonies and high macroalgae cover may be limiting the rate of coral cover increase especially at 2m. Given the proximity of this reef to local rivers it is possible that runoff is impacting this community.

King Reef

There are no published data on the dynamics of the benthic community of this reef prior to 2004 (Sweatman *et al.* 2006). Consistent anecdotal reports indicate that the reef has in the past supported complex coral communities with large stands of branching *Acropora* spp, it is however likely that these observations are from a different area of reef to the sites used in this program. While the direct evidence of disturbance has not been quantified, flood plumes were observed to inundate this reef annually between 1994 and 1997, it is also likely that the reef was impacted by bleaching in 1998, and to a lesser degree in 2002 (Appendix 1: Table A1-5.2). Most recently the passage of TC Larry in 2006 caused a substantial reduction in the already depauperate coral community (Figure 5.11).

Coral cover has remained low since first surveyed in 2004 (Sweatman *et al*., 2006). Between 2004 and 2006 coral cover fell from 5% to 2% at 2m depth and 23% to 9% at 5m. The majority of this decline is attributable to the effects of TC Larry. The largest effect of the cyclone may have been the removal of macro-algae (Figure 5.11).

At 2m there is very little accretion of a carbonate substrate the substrate rather comprised largely of igneous rock. This rocky substrate along with the observation of very low coral cover, low density of recruit-sized colonies and no soft corals (Figure 5.11,5.10) all suggest limited potential for coral growth both currently and historically. At 5m there is more carbonate substrate though again it is limited suggesting that although recruit densities (Figure 5.12) suggest potential for community recovery, in the longer term such communities are likely transient.

North Barnard Group

There is little prior data relating to the coral communities at this reef, though modelling suggest a high likely hood that the reef was impacted by bleaching particularly in 1998 (Appendix 1: Table A1-5.2).

In March 2006 this reef was severely impacted by TC Larry. In May 2006 the cover of all major benthic groups had been reduced to very low values (Figure 5.11).

Prior to the cyclone benthic communities at 2m had above average cover of hard coral, very low cover of soft coral, and very low cover of macroalgae (Figure 5.11). The hard coral cover at 2m was dominated by colonies from the family Acroporidae (mostly large stands of branching *Acropora*), which represented 94% of all corals (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was well above average in all size classes (Figure 5.12); colonies from the families Acroporidae and Faviidae were most abundant (Appendix 1: Table A1-5.5). On the deeper slope (5m) the cover of hard coral was lower and soft coral markedly higher than at 2m, macroalgae were again relatively rare (Figure 5.11). The hard coral community on the deep slope was slightly more diverse however still dominated by colonies from the family Acroporidae (80%, Appendix 1: Table A1-5.3). As at 2m the density of recruits was well above average in all size classes (Figure 5.12) with colonies from all families except Agariciidae abundant (Appendix 1: Table A1-5.5).

Surveys immediately following the cyclone found that hard coral cover and the density of recruit-sized colonies had declined dramatically. Hard coral cover fell 95% at 2m and 80% at 5m (Figure 5.11). The density of recruits fell approximately 90% at both depths and cover of macroalgae and soft coral remained low (Table 5.10, 5.9).

The lack of quantitative data on prior the dynamics of this community limits our ability to determine the recovery potential of this reef. Very low overall coral cover combined with low recruit densities suggest future increases in hard coral cover will need to be driven by sexual recruitment alone. This, combined with the almost total lack of fast growing *Acropora* spp fragments remaining, suggests recovery may take many years.

Burdekin NRM region

The influence of the Burdekin River extends in the north to Orpheus and Pelorus Island. During the 1997 flooding of the Burdekin River the prevailing northerly winds forced the flood plume to extend to the south influencing the reef communities in the northern Whitsunday region. This is however a rare occurrence, with the main influence of the Burdekin extending to the north with the prevailing south-easterly flow. The closest northerly near-shore reefs to the Burdekin mouth chosen for this study are Middle Reef, adjacent to Townsville followed by the fringing reef located at Geoffrey Bay on Magnetic Island. Both of these reefs are also influenced by Ross River. Five reefs to the north of these locations were chosen to reflect a decreasing influence of the Burdekin River and coastal rivers that flow into the Halifax Bay area, such as Cattle Creek and occasionally the Herbert River. Lady Elliot Reef is situated the closest to the coast and the influence of Cattle Creek, Pandora Reef then the fringing reef on the southern side of Havannah Island, western reefs on Orpheus Island and Pelorus Island and finally the eastern side of Orpheus Island complete the gradient of decreasing likelihood of exposure to runoff.

There is a wealth of historical data for many of the sites chosen. These historical observations substantially increase our understanding of the dynamics of these coral reef communities. Coral communities in Geoffrey Bay have been monitored since 1989 (Ayling & Ayling 2005), initially as part of an impact study into the effects of construction of a harbour in the adjacent Nelly Bay, and subsequently with the ongoing monitoring of the effects of the harbour and singular events such as cyclone Tessi in 2000. Middle Reef is part of the AIMS LTMP and benthic communities have been studied since 1992. The AIMS LTMP sites at Middle Reef are situated approximately one meter below the sites constructed for this program. Pandora Reef has been the focus of several studies over the last two decades (AIMS LTMP and Dr Terry Done [AIMS]); in particular the deep site on the south east side of the reef corresponds to a site surveyed by Dr Terry Done since the 1980's. Havannah Island is also an AIMS LTMP survey site; however the sites chosen for this study are on the opposite side of the island. The AIMS LTMP has broad scale data on the site in the form of manta tow data collected from the entire island since 1992. Orpheus Island and Pelorus Island sites have been studied extensively. The shallow sites chosen formed part of a long term monitoring program instigated by QPWS in 1993 for the Palm Island Group (Thompson and Malcolm 1999). There were no deep sites in the original study which is still carried out intermittently by AIMS LTMP. The temporal profiles of hard coral cover from these monitoring studies can be found in Sweatman *et al.* (2006). Sweatman et al. (2006) also includeds coral community data from 2004 for four reefs in this region derived from a similar suite of methods as used in this study. Sites at two of these (Geoffrey Bay and Middle Reef) are essentially the same sites as used in this study. Comparison between 2004 and 2005 surveys indicates the community changed little at Middle reef while cover of hard coral increased at both 2m and 5m at Geoffrey Bay. [

The extended period of monitoring of the reefs in this catchment highlights the intense and frequent nature of disturbances to some reefs (Ayling and Ayling 2005, Sweatman *et al*., 2006, Appendix 1: Table A1-5.2). The largest disturbance since monitoring began in 1989 was coral bleaching in 1998. This event affected all coral communities on the target reefs in this NRM region (Appendix 1: Table A1-5.2). The 2002 bleaching event was less severe than 1998 but still affected the majority of coral communities (Appendix 1: Table A1-5.2). Cyclonic disturbances in 1990 (Joy), 1996 (Justin) and 2000 (Tessi) variously impacted reefs, and a large decrease in coral cover attributed to cyclone Tessi at Havannah Island may also

include the effects of a local COTS outbreak there in the same year. During the period 1991-1999 flood plumes extended to most reefs in 1994, 1997 and 1998 (Devlin et. al 2001). Monitoring studies (Ayling and Ayling 2005, Sweatman *et al.,* 2006) found no discernable direct effects of these flood plumes on the coral communities at the depths monitored. However, surveys on Pandora Reef after the major flooding event of 1998 found around 80% of the corals were bleached to a depth of about 10 metres hence the effects of the flood plume may have exacerbated the impacts of high temperature during this period (Devantier, Fabricius unpublished).

Given the frequency and severity of disturbances to reefs in this region over the preceding decade it is not surprising hard coral cover was lower on average than in the other NRM regions. In contrast to low hard coral cover, reefs in this region had, on average, higher cover of macroalgae (compare cover estimates for these groups against overall means shown in Figure 5.13). While these broad community measures differed from other regions along with the cover of soft corals they also differed substantially among reefs within the region. Reefs that were close to river mouths and the coast that had suffered disturbances tended to have high cover of macroalgae while cover of macroalgae was very low on the reefs that were furthest from rivers, even though coral cover was markedly reduced in the mid 1990's (see low cover at Pelorus Island and Orpheus Island Figure 5.14). This is a possible effect of runoff. Interestingly the cover of soft corals on these reefs was exceptionally high.

In addition to low cover of hard coral in this region, the average density of recruit-sized colonies was also low, though this also varies substantially among reefs (Figure 5.14). Very low recruit densities at both Havannah Island and Pandora Reef may indicate a lack of recovery potential on these reefs in particular. Regionally, neither the richness of hard coral genera nor the richness of genera represented by recruit- sized colonies were significantly low however again this varies among reefs (Appendix 1: Table A1-5.7). The richness of genera represented by recruit-sized colonies is significantly lower at Pandora Reef than at any other reef in the region further questioning the recovery potential of the coral communities.

Middle Reef

In the period 1992-2005 coral cover on the AIMS LTMP sites fluctuated from 27% in 1993 to 40% in 1999. Cyclone Tessi in 2000 was responsible for a large decrease in coral cover and signalled the start of a decline from 40% in 1999 to a low of 27% in 2002 after the bleaching event of that year. Recovery of the coral community since 2002 has been rapid and in 2005 coral cover was recorded as 40% at the LTMP sites (Sweatman *et al*., 2006).

Due to the shallowness of the water surrounding the reef 5m below datum sites were not established at this reef. The 2m below datum sites are located a meter or so shallower than, and adjacent to, sites 1 and 2 of the AIMS LTMP.

At 2m hard coral cover was 50.8%, soft coral cover 7% and macroalgae largely absent comprising <1% cover of the substrate (Figure 5.13). The cover of hard coral was the highest in the region and dominated by corals from the families Poritidae and Agariciidae which when combined represented 84% of hard coral cover (Appendix 1: Table A1-5.3). The presence of very large colonies of *Pachyseris* spp and *Goniopora* spp indicates the resilience of these colonies to major disturbances such as bleaching and cyclones. The density of recruit-sized colonies was well above the average (Figure 5.14) with the families Dendrophylliidae, Faviidae and Acroporidae all well represented (Appendix 1: Table A1-5.5).

The large numbers of recruits from these families suggests their presence may rapidly increase within a community now dominated by large colonies of other taxa.

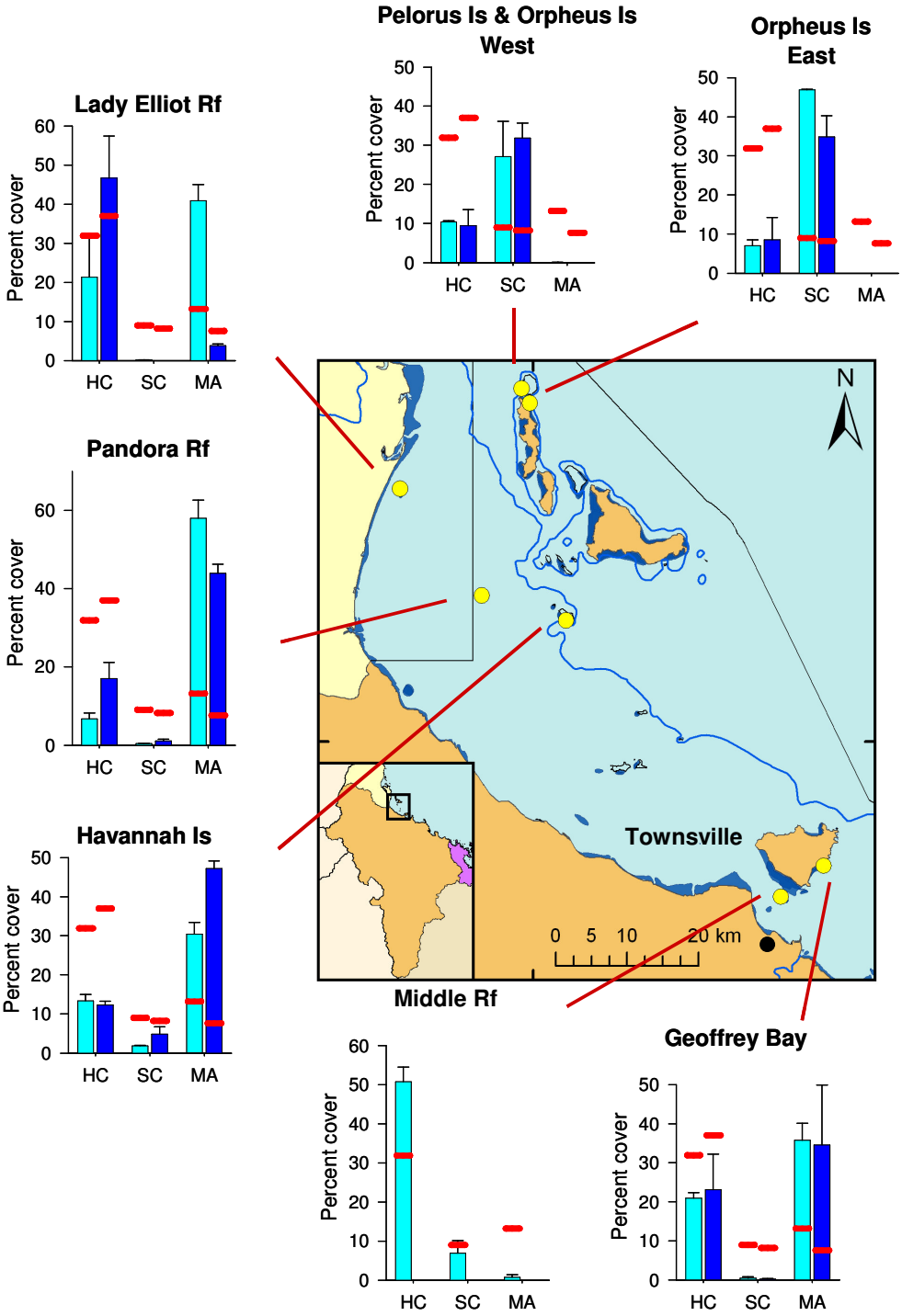

Figure 5.13. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in Burdekin NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

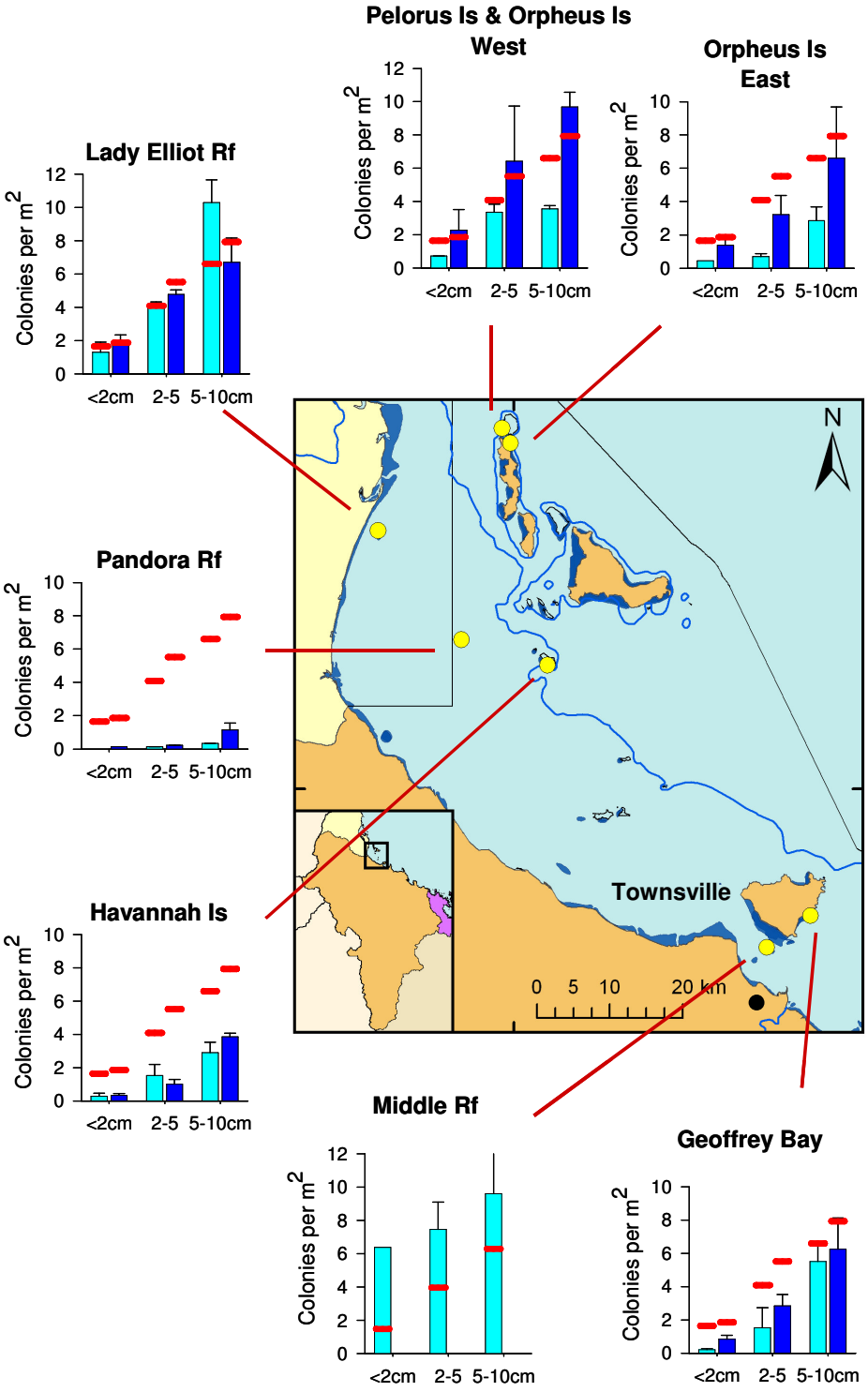

Figure 5.14. Density of recruit-sized hard coral colonies by size class for reefs in the Burdekin NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Geoffrey Bay

Reef slope communities in Geoffrey Bay have been monitored since 1989 (Ayling and Ayling, 2005). Hard coral cover during this period peaked at 50% in 1997 with the community dominated by corals from the family Acroporidae. Subsequent declines in coral cover have occurred following coral bleaching in 1998 and 2002 and cyclone Tessi in 2000. Hard coral cover decreased by 24% after bleaching in 1998 and then 37% after bleaching in 2002. In 2000 cyclone Tessi caused a decrease in cover of 18%. In 2003 total hard cover for this reef was only 22% and the makeup of the coral community had shifted to be dominated by the more resilient corals from the families Poritidae and Faviidae. The most recent surveys in Geoffrey Bay in June 2005 by Sea Research recorded coral cover at 26%.

Survey sites used for this program are in the vicinity of previous monitoring sites. The benthic communities at both 2m and 5m had a high cover of macroalgae and relatively low cover of hard corals (Figure 5.13). Hard coral cover at 2m was largely comprised of the families Acroporidae and Poritidae (Appendix 1: Table A1-5.3). At 5m the hard coral community was more diverse with a number of families represented though the families Acroporidae, Faviidae and Poritidae still represented the majority of coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average (Figure 5.14). At 2m the community appears to be recovering from previous disturbances with high numbers of Acroporidae recruits (Appendix 1: Table A1-5.5). At 5m recruit-sized colonies were spread over a more diverse range of families with small colonies of Faviidae, Dendrophylliidae and Merulinidae most abundant.

The coral community at Geoffrey Bay appears to be recovering from previous disturbances with an increase in Acroporidae likely at 2m and an increase in cover across a number of hard coral families at 5m. This recovery appears to be slow when compared to several reefs in the vicinity of Geoffrey Bay, such as Nelly and Florence Bays. An explanation for this may be linked to the relative effect of beaching in 2002. Florence Bay recorded a drop in hard coral cover of only 3% and Nelly Bay 9% compared to 34% at Geoffrey Bay. The slow recovery may also be related to low densities of recruit-sized colonies, future surveys will provide information of this relationship. The high cover of macroalgae at these sites may also play a part in limiting recovery potential.

Lady Elliot Reef

There is little data on the makeup of the coral community at this reef prior to these surveys. Lady Elliot Reef is situated only 2km offshore from Taylors Beach and close to the mouth of Cattle Ck. Bleaching in 1998 and 2002 are highly likely to have affected this community as are flood events during the last decade (Appendix 1: Table A1-5.2). It is likely that these disturbances have influenced the benthic community especially on the shallow slope where macroalgae cover was high compared to the cover of hard and soft corals (Figure 5.13).

Hard coral cover at 2m was 21% and dominated by corals from the Acroporidae and Oculinidae families (Figure 5.13, Appendix 1: Table A1-5.3). The density of recruit-sized colonies at 2m was slightly above average and included a high number Acroporidae recruits along with an extraordinary number of Fungiidae recruits (Figure 5.14, Appendix 1: Table A1-5.5). In contrast, at 5m the cover of macroalgae was very low, hard coral cover high (47%) and no soft corals were recorded (Figure 5.13). The families Oculinidae, Poritidae, Mussidae and Pectiniidae were all common components of the hard coral community (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average across all size classes with recruits of the families Faviidae, Dendrophylliidae and Fungiidae all well

represented (Figure 5.14, Appendix 1: Table A1-5.5). The adult cover of Dendrophylliidae was low at this depth (1%) and future surveys should see an increase in the cover of this family as smaller colonies grow.

The coral community appears to be healthy and diverse with good recruitment levels. The presence of high numbers of small colonies from the fast growing Acroporidae family and few larger colonies at 2m suggests this community is recovering from past disturbances.

Pandora Reef

The northern side of Pandora Reef has been studied in detail for two decades as part of the ongoing AIMS LTMP (Sweatman *et al*., 2005) surveys and as part of a detailed photographic transect and community survey by AIMS (Done unpublished data). Only one site of the later surveys provides historical data on the reef community from the vicinity of the sites chosen for this study. During the period 1992-2005 the maximum hard coral cover on AIMS LTMP was 58% in 1997 following a period with no observed disturbances. Bleaching in 1998 and 2002 and cyclone Tessie 2000 in reduced cover to a low of 39% in late 2002, cover has been slowly increasing from this point. Monitoring of photo quadrats in a similar habitat to our deep slope sites from 1985 showed the community was dominated by branching *Acropora* and *Montipora* until declining through the 1990's there has been very little recovery of the coral communities since this decline. Bleaching in 1998 followed by extensive flooding resulted in the observation that around 80% of the corals on Pandora Reef were bleached to a depth of around 10 metres (Devantier, Fabricius unpublished). These two major events likely contributed to the mortality of the large stands of *Acropora* spp and *Montipora* spp on the southern face of the reef.

In 2005 our sites on the southern reef face were dominated by of macroalgae (Figure 5.13). At 2m hard coral cover was 7% and the lowest observed in this region, while higher at 5m (17%) this is still low for near-shore reefs. Corals from the Poritidae and Siderastreidae families contributed most to the low cover at 2m while the family Faviidae had the highest cover at 5m (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was well below average and the lowest for the region at both depths (Figure 5.14). The majority of recruits were primarily from the Oculinidae family at 2m and the families Faviidae and Oculinidae at 5m (Appendix 1: Table A1-5.5). There was only one Acroporidae recruit found at 5m and none at 2m. With the adult Acroporidae population at both depths occupying <5% cover there is little remaining of the large stands of this family recorded in the early 1990's. There was no evidence for the recovery of the coral community at this location.

Havannah Island

The northern reef community of Havannah Island has been studied in detail since 1997 as part of ongoing AIMS LTMP surveys (Sweatman *et al.* 2005). In 1998 bleaching was responsible for a 49% decrease in hard coral cover at this location. The remaining coral cover was then reduced by 66% in 2000 from a combination of cyclone Tessie and crown-of-thorns starfish and then bleaching in 2002 caused a further 21% decrease (Appendix 1: Table A1-5.2). In 2005 very little coral was left at the northern Havannah Island sites with cover down to 10%. The cover of macroalgae at the AIMS LTMP sites in 2005 was 49%.

Sites for this program were set up on the southern side of Havannah Island were an extensive reef flat has developed. In 2005 cover of macroalgae was well above the average contrasting the low cover of hard corals (Figure 5.13). At 2m hard coral cover was primarily composed of the families Acroporidae, Poritidae, Faviidae and Dendrophylliidae (Appendix 1: Table

A1-5.3). The coral community at 5m was more diverse, the families Faviidae, Poritidae and Dendrophylliidae accounted for the highest proportions of the observed coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was low at both depths (Figure 5.14). At 2m small colonies of the families Poritidae, Faviidae and Acroporidae were most abundant while at 5m most recruit-sized colonies were from the families Fungiidae, Poritidae, Oculinidae and Faviidae (Appendix 1: Table A1-5.5). There were only two recruits from the family Acroporidae at 5m.

The shallow sites appear to be recovering from the severe disturbances experienced at Havannah Island with hard coral cover and recruit-sized colonies including a high proportion of the fast growing Acroporidae family. However, the potential for recovery of reefs surrounding Havannah Island appears to be compromised due to the low recruitment densities and low cover of hard coral overall. The high levels of macroalgae may also play a part in limiting the recovery of hard coral communities.

Orpheus and Pelorus Island West

The shallow back reef community of Orpheus Island (Cattle Bay) and Pelorus Island have been studied extensively and formed part of a long term monitoring program instigated by Queensland Parks and Wildlife Service in 1993 for the Palm Island Group (Thompson and Malcolm 1999). There were no deep sites in the original study. These sites are still surveyed intermittently by AIMS. Hard coral cover at Cattle Bay and Pelorus Island was estimated at 32% and 26% respectively in surveys conducted in 1994. Bleaching in 1998 was responsible for an 83% decrease in hard coral cover at these sites (Appendix 1: Table A1-5.2). Decreases in coral cover were measured in surveys in 1997 and are thought to have been caused by cyclone Justin which passed near the area in 1996. The latest survey data from these sites in 2005 measured hard coral cover at Cattle Bay at 3% and Pelorus Island 10% (Sweatman *et al.* 2006).

Shallow sites in this program use the first half of the existing QPWS/AIMS monitoring sites. At both 2m and 5m the benthic community has a high cover of soft coral, low cover of hard coral and no or very little macroalgae (Figure 5.13). At 2m there was roughly even cover of Acroporidae and Pocilloporidae that combined account for 71% of the hard coral cover (Appendix 1: Table A1-5.3). At 5m the coral community was more diverse though Acroporidae, Faviidae and Poritidae were the main components of coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was slightly below average at 2m with colonies from the families Faviidae, Acroporidae and Oculinidae most numerous (Figure 5.14, Appendix 1: Table A1-5.5). At 5m the density of recruit-sized colonies was higher with the families Poritidae and Acroporidae by far the most abundant (Figure 5.14, Appendix 1: Table A1-5.5).

The reef community appears to be recovering from the devastating disturbances experienced in the last decade. The presence of many colonies of bleaching intolerant species such as *Acropora* spp and *Pocillopora* spp and the recruitment of a range of hard coral families is a good indicator of this recovery. The high cover of soft corals may however be limiting the rate of recovery.

Orpheus Island East

The shallow front reef community of North East Bay on the seaward side of Orpheus Island also formed part of a long term monitoring program instigated by QPWS in 1993 for the Palm Island Group (Thompson and Malcolm 1999). Hard coral cover at North East Bay was

estimated at 19% in surveys conducted in 1994. Bleaching in 1998 and then 2002 reduced hard coral cover to as low as 2.5% in 2003; there had been some recovery with cover up to 8% by 2005 (Sweatman *et al*., 2006). The 2m depth at site 1 of this study follows the first half of the existing monitoring site.

In 2005 the cover of soft coral was high, cover of hard coral low and macroalgae absent (Figure 5.13). The hard coral cover at 2m included a high representation from the families Acroporidae and Faviidae these were also well represented at 5m as was the family Poritidae (Appendix 1: Table A1-5.3). The density of recruit-sized hard coral colonies was low at both depths though particularly at 2m (Figure 5.14). Low numbers of recruits from the Poritidae, Faviidae and Acroporidae families were recorded at 2m (Appendix 1: Table A1-5.5). At 5m there were three times the number of recruit-sized colonies as at 2m with the families Poritidae, Faviidae and Acroporidae most numerous (Appendix 1: Table A1-5.5).

The reef community appears to be slowly recovering from the devastating disturbances experienced in the last decade. However, with low recruitment and low cover of adult colonies and the presence of large colonies of soft corals the recovery is likely to be slow.

Mackay Whitsunday NRM region

The influences of the Proserpine, O'Connell and Pioneer Rivers are most likely to extend in a northerly direction as plumes and sediments are transported by a northerly near-shore drift. It is worth noting that during the 1997 flooding of the Burdekin River the prevailing northerly winds forced the flood plume to extend to the south influencing the reef communities in the northern Whitsunday region. The same prevailing winds also forced the flooding of the Proserpine River in 1997 to extend predominately south. This is however a rare occurrence. The reefs chosen to form the gradient of decreasing influence of runoff are broken into two gradients of influence based on their distance from the shore. The 'inshore' gradient consists of four fringing reefs all located within 10km distance from the shoreline spread along a northerly direction of decreasing influence. The 'off shore' gradient consists of three reefs located more than 10km from shore also in a northerly direction of decreasing influence. Of the 'inshore' fringing reefs in the most proximate to the influence of the rivers are those on the western side of Pine Island followed by the eastern fringing reefs on Shute and Tancred Island this is followed by the reef on the northern side of Daydream Island and finally the fringing reef on the southern side of Double Cone Island. Of the 'offshore' fringing reefs the most proximate to the influence of the rivers is that on the eastern side of Seaforth Island followed by the eastern side Dent Island with a reef on the south-western tip of Hook Island completing the 'offshore' gradient.

There is historical data available for coral communities for most of the locations selected. Van Woesik *et al*., (1999) found physio-chemical and biological gradients were evident across a gradient from the mouth of the Proserpine River to Double Cone Island and surmised that anthropogenic effects may be evident at reefs near the river output. This study also examined coral larvae supply at 7 reefs (including Pine and Double Cone Island) using terracotta settlement tiles and compared the results in reference to the gradient. Van Woesik and DeVantier (1992) contain a substantial data set on hard and soft coral communities at 54 sites in the Whitsunday region, including Shute and Tancred Island and Hook Island. Daydream Island was the focus of several studies: Harriott and Fisk (1990) surveyed coral communities on the back reef as part of a report on the extent of anchor damage in the Whitsunday Isands, Fisk (1991) carried out more detailed surveys of this community which provides good

baseline data on major coral groups. In April 1998 and September 1998 coral communities on the back reef of Daydream Island at two depths were surveyed using video transects to ascertain the effect of the February 1998 bleaching event. Similar bleaching and post bleaching surveys were undertaken on the back reef of Dent Island. DeVantier *et al*., (1998) re-examined data collected in the Whitsunday region in the surveys of 1992 and 1995 and provides a good synthesis of results from this data. In 2004 Sweatman *et al.* 2006 conducted surveys using essentially the same techniques as this study though with the addition of compiling hard coral species list from six reefs in this area. Of these six reefs sites at Pine Island and Shute and Tancred Islands and one of the two sites at Daydream Island are essentially the same as sites surveyed in this study.

The largest disturbances in recent history were coral bleaching events in 1998 and 2002 at likely affected all target fringing reefs in this region (Appendix 1: Table A1-5.2). In 1997 flood plumes from the Burdekin River drifted south due to the prevailing northerly winds but extended only to the 'inner' gradient reefs (Devlin *et al.* 2005). Comparison between surveys in 2004 and 2005 for those reefs included in both years indicate no major changes over the period. Similarly, a comparison between all reefs surveyed in 2005 and 2006 showed no substantial changes for any of the community attributes measured. Some minor changes are detailed below for individual reefs.

The cover of both hard corals and soft corals, the density of recruit-sized colonies and the richness of genera, both over all and of recruit-sized colonies, vary significantly among reefs. The cover of macroalgae is significantly higher at Seaforth Island and Pine Island than on the other reefs in this region; these reefs are the most proximate to the rivers potentially influencing these reefs. There was no clear association with proximity the major rivers and the variation in the other community statistics. In addition to variation among reefs in community summary statistics there is are also marked differences in community composition among reefs.

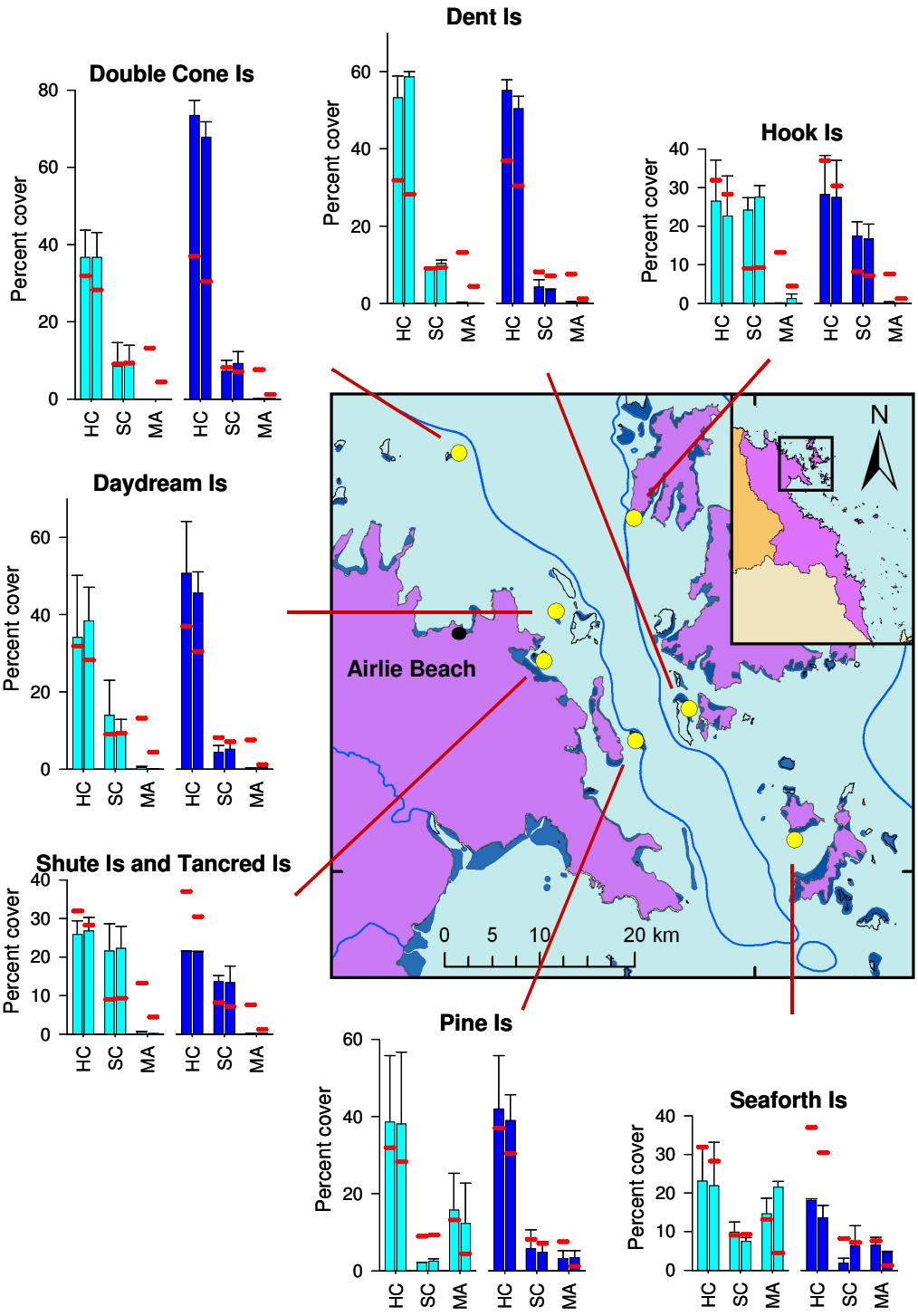

Figure 5.15. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in the Mackay Whitsunday NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

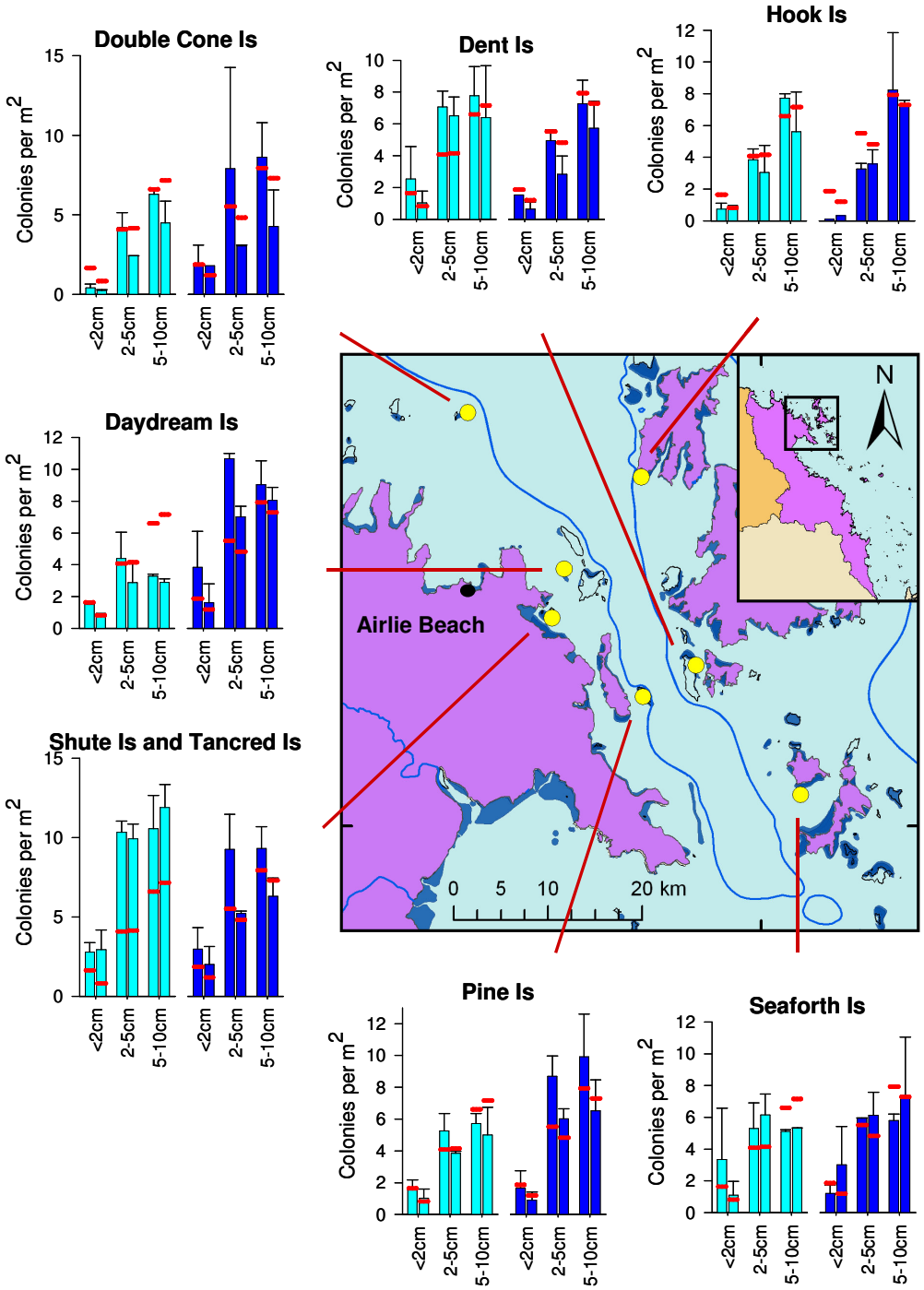

Figure 5.16. Density of recruit-sized hard coral colonies by size class for reefs in the Mackay Whitsunday NRM region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines. For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Pine Island

The benthic community situated on the western fringing reef of Pine Island was first examined in 1993 (Van Woesik *et al*., 1999). In 1993 the reef slope sites (2-4m) were dominated by macroalgae (36%) and hard coral cover was only 11%. The reef was described as 'sparse scleractinian coral communities, dominated by faviids, *Montipora* spp and encrusting *Porites* spp colonies'. It is likely there was some affect on coral communities due to bleaching in 1998 and highly likely the 2002 bleaching event had some detrimental consequences on the coral community (Appendix 1: Table A1-5.2). Coral recruitment in summer 1993 onto deployed settlement tiles was the highest recorded for the gradient reefs with an average of 2.3 recruits per tile (Van Woesik *et al*., 1999). The recruits were dominated by Acroporidae (45%), Poritidae (22%) and Pocilloporidae (18%).

The sites chosen for this study are situated in the vicinity of sites used by the 1993 study. In 2005 the benthic communities at 2m had an above average cover of macroalgae (16%) and hard coral (39%, Figure 5.15). The hard coral was dominated by massive *Galaxea* spp which represented 55% of all corals; 32% of corals were Acroporidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average (Figure 5.16), with the families Acroporidae and Poritidae most abundant (Appendix 1: Table A1-5.5). At 5m the cover of macroalgae was lower (3%) and hard coral higher (42%, Figure 5.15). The hard coral cover included high representation of the families Oculinidae (28%), Pectiniidae (22%) and Poritidae (16%), Acroporidae represented 12% of the cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average or above for all size classes and recruits from a range of coral families were recorded however the vast majority of these were from Poritidae and Acroporidae (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 the cover of all major benthic groups had remained largely unchanged from those observed in 2005 (Figure 5.15). At 2m the density and community composition of recruit-sized colonies was also similar to that in 2005 (Figure 5.16). At 5m the density of recruit-sized colonies had slightly declined with abundance of the family Poritiidae most notably reduced (Figure 5.16, Appendix 1: Table A1-5.3). The two sites at this reef have totally different coral communities although they are only a few hundred meters apart and in very similar physical settings. Site 2 is dominated by large *Galaxea* colonies that have obviously seen little or no impact for many years whereas the rubble slope of Site 1 indicates a past dominance of *Acropora* spp. The community composition of recruit-sized colonies differs between sites: site 1 has 2 times the number of *Acropora* and *Montipora* recruits and 5 times the number of Porites recruits when compared to site 2. It is highly likely that these sites will maintain this community difference with recruitment at Site 1 indicating a return to an *Acropora/Montipora/Porites* community type. This location demonstrates differences in community types that can occur over very small scales.

In 2005 the coral recruitment to terracotta settlement tiles was the highest recorded for the three reefs examined in the region. There was an average of 18.7 recruits per tile at 2m (Table 5.4) which is up to nine times the recruitment measured by Van Woesik *et al.* (1999) (Table5.5). At 2m recruits settling to tiles were mostly Acroporidae (83%), other/unknown (9.2%) and Poritidae (7.1%, Table 5.4). At 5m an average 36.2 spat recruited per tile and these were mostly Acroporidae (86%), Poritidae (8.5%) and other/unknown (4.8%, Table 5.4). The high numbers of other/unknown recruits at both depths may be from Oculinidae or Pectiniidae - which are well represented in the adult community (Appendix 1: Table A1-5.3).

Shute and Tancred Island

A single fringing reef site on the southern tip of each island was selected and these combined to form the survey location. Sites 1 and 2 in this study correspond to site 30 on Tancred Island (closely) and 31 on Shute Island (general area) surveyed in 1992 and reported by Van Woesik and DeVantier (1992). The report found some degradation of corals in this area possibly caused by the freshwater plume associated with cyclone Joy in 1991. In 1992 the Shute Island fringing reef was reported as being dominated by macroalgae (40%), hard coral (29%) and soft coral (21%), while the benthic community on Tancred Island was dominated by soft coral (42%), hard coral (11%) and macroalgae (3%).

In 2005 the benthic communities at 2m sites had above average cover of soft coral (22%), below average cover of hard coral (26%) and almost no macroalgae (Figure 5.15). The hard coral community had a high proportion of the family Acroporidae (61%), a further 21% was from the Poritidae family (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average for all size classes (Figure 5.16), with recruit-sized Acroporidae, Faviidae and Poritidae all numerous (Appendix 1: Table A1-5.5). At 5m there was again above average cover of soft coral (14%) and below average cover of hard coral (22%) and virtually no macroalgae (Figure 5.15). The hard coral community had a high proportion of the family Acroporidae (48%), a further 34% of corals were equally from the Poritidae and Pectiniidae families (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average for all size classes with Acroporidae, Faviidae and Poritidae colonies all numerous (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 the cover of major benthic groups was largely unchanged at 2m or at 5m (Figure 5.15). The coral community composition was also similar though with a slight increase in the representation of the family Pectiniidae and reduction in Poritidae at 2m with the reverse occurring at 5m (Appendix 1: Table A1-5.3). At 2m the density and community composition of recruit-sized hard coral colonies was similar to the previous year (Figure 5.16, Appendix 1: Table A1-5.5). At 5m the density of recruit-sized colonies was reduced in all size classes (Figure 5.16). The majority of the recruits were Acroporidae, Faviidae and Pectiniidae, with the number of Poritidae recruit was markedly lower than observed in 2005 (Appendix 1: Table A1-5.5). This reduction in Poritidae recruits explains the decrease in the density of the 5-10cm size class and the increase in the representation Poritidae which have likely grown to adult sizes.

The high cover of *Acropora* spp indicates a lack of recent disturbances. The high diversity and high recruitment density indicate the potential for future increases in coral cover at this location, limited on the shallow slope somewhat by the prolific soft coral community.

Daydream Island

In 1990 surveys of the shallow reef slope (2-3 m) reported a community with 45% hard coral cover that was dominated by branching *Acropora* spp. (Fisk 1991). At this time some damage to the coral community from anchoring was noted. Surveys nearby (GBRMPA bleaching surveys) of the shallow reef slope in April 1998 found hard coral cover was down to 28% and by September 1998 cover was down to 16% due to mortality associated with the 1998 thermal bleaching event. The deep community was not affected by bleaching in 1998, with cover estimates similar in April (54%) to post bleaching in September (59%). In 2002 large scale bleaching was observed on benthic communities at Daydream Island (GBRMPA Unpublished Bleaching Surveys (2002)) but there appeared to be little mortality associated with this event.

The permanent sites surveyed in this project coincide with sites 1 and 2 videoed by GBRMPA in April 1998 in response to coral bleaching. The shallow sites also coincide with the general area surveyed as site 6 in Fisk (1991). In 2005 the benthic communities at 2m had above average cover of soft coral (14%) and hard coral (34%) and almost no macroalgae (Figure 5.15). The hard coral community at 2m was dominated by Acroporidae (mostly large stands of branching *Acropora* spp) which represented 93% of the coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average or above for the <5cm size classes but below average for the 5-10cm size class and were primarily Acroporidae and Faviidae (Figure 5.16, Appendix 1: Table A1-5.5). The benthic communities at 5m had below average cover of soft coral (4%) and above average cover of hard coral (51%) and no macroalgae (Table 5.13). The hard coral community was dominated by Acroporidae (again mostly large stands of branching *Acropora* spp) which represented 88% of the recorded cover of hard corals (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average for all size classes and included recruits from a wide range of families but dominated by Acroporidae, Mussidae and Pectiniidae (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 the soft coral cover had decreased to 9% and the hard coral cover increased to 38% at 2m (Figure 5.15). The coral community was still dominated by Acroporidae which represented 93% of hard coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was slightly lower than compared to the previous year with the numbers of Acroporidae and Faviidae most reduced (Figure 5.16, Appendix 1: Table A1-5.5). At 5m the cover of soft coral had not changed, hard coral cover had however decreased to 46%, still well above average (Figure 5.15). The coral community was still dominated by *Acropora* spp which represented 88% of coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was reduced for all size classes, however, was still above average (Figure 5.16). The majority of the recruits were Acroporidae, Faviidae and Pectiniidae with the number of Mussidae decreasing in 2006 (Appendix 1: Table A1-5.5).

In 2005 the coral recruitment to terracotta settlement tiles at 5m (mean of 47.5 per tile) was the highest recorded for the three reefs examined in the Whitsunday region; however the recruitment at 2m (13.2 per tile) was second to Pine Island (Table 5.4). At both depths recruits from the family Acroporidae (94%) were by far the most abundant with much lower numbers of Pocilloporidae and Poritidae recorded (Table 5.4).

The very large stands of branching *Acropora* spp. that grow from above 2m down the slope beyond 5m indicate a lack of recent disturbances. The slight decrease in cover at 5m could however be the result of disease that was prevalent during surveys in 2005, though largely gone by 2006. The high coral cover, high density and diversity of recruit-sized colonies and high numbers of spat settling to tiles at 5m are all indicative of a healthy coral community that will continue to expand given no major disturbances in the near future. The large stands of branching Acropora spp at 2m should develop further given no disturbances.

Double Cone Island

The fringing reef on the southern side of Double Cone Island was first examined in 1995 (DeVantier and Turak 1995). They described the shallow community as having a coral cover of 1-10%, patches of macroalgae (*Sargassum* spp), isolated medium sized (>50cm) colonies of *Acropora* and *Montipora* spp and a very low soft coral cover. The deep sites were described as having a coral cover of 11-30% and dominated by a large *Acropora* spp colonies >50cm and colonies of sub-massive *Goniopora* spp; soft coral cover on the deep site was

estimated at between 11-30%. The sites chosen in this study are located to the north-east of survey sites described by DeVantier and Turak.

The bleaching events of 1998 and 2002 and cyclone Joy are likely to have affected coral communities at Double Cone Island (Appendix 1: Table A1-5.2); however there is no quantitative data on the affect of these disturbances.

In 2005 the benthic communities at 2m had an average cover of soft coral (9%) and above average hard coral cover (37%) and no macroalgae (Figure 5.15). The hard coral community was dominated by Acroporidae (mostly branching *Acropora* and encrusting *Montipora* spp) which represented 59% of the coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average or below for all size classes and were primarily Acroporidae, Faviidae and Poritidae (Figure 5.16, Appendix 1: Table A1-5.5). At 5m soft coral cover (7%) was below average and hard coral cover very high (73%) macroalgae was absent (Figure 5.15). The hard coral community was dominated by large stands of sub-massive Goniopora spp and *Acropora* spp colonies which represented 73% and 12% of the hard coral cover respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average and above for all size classes and included low numbers of recruits from a wide range of families but dominated by Poritidae and Faviidae and Oculinidae (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 at 2m the cover of the three major benthic groups and composition of the hard coral community remained largely unchanged (Figure 5.15). The density of recruit-sized colonies was lower in 2006 for all size classes and the number of recruits recorded decreased; especially Acroporidae and Oculinidae (Figure 5.16, Appendix 1: Table A1-5.5). At 5m there was a slight increase in the cover of hard coral and a similar decline in soft coral (Figure 5.15). The hard coral community on the deep slope was still dominated by large stands of sub-massive *Goniopora* spp and *Acropora* spp. (Appendix 1: Table A1-5.3). The density of recruit-sized colonies decreased in the 2-5 cm and 5-10 cm size classes as did the number of recruits recorded from several of the coral families that dominated in 2005 (Figure 5.16, Appendix 1: Table A1-5.5). The majority of the recruits were Faviidae and Pectiniidae, with the numbers of Poritidae recruits decreasing from 36 in 2005 to only 8 in 2006 (Appendix 1: Table A1-5.5).

In 2005 coral recruitment to terracotta settlement tiles was the lowest recorded for the three reefs examined in this region (Table 5.4). The average number of recruits per tile at 2m was very low compared to all other locations studied in 2005 though higher than the 0.6 recruits per tile recorded in 1993 (Table 5.5). The majority of recruits were from the families Pocilloporidae (62%), Acroporidae (37%) and Poritidae (1%) recruits (Table 5.4). At 5m recruitment to tiles was higher though still low compared to other reefs (Table 5.4). Most abundant amongst recruits to tiles at 5m were Acroporidae (71%), Pocilloporidae (25%) and Poritidae recruits (1.3%) (Table 5.4).

The two depths contain very different coral communities; the deep slope is dominated by large fields of Goniopora spp and the shallow slope dominated by *Acropora* and *Montipora* spp. The aggressive nature of the sub-massive *Goniopora* spp may serve to maintain this community separation by actively killing or at least suppressing the growth of other nearby corals from different coral genera. The presence of a high cover of *Acropora* and *Montipora* spp on the shallow slope indicate a lack of recent disturbances. The low recruitment densities and recruit numbers may limit the increase in the coral cover at 2m.

Seaforth Island

We know of no previous data on the coral communities from this location. Modelling data however suggest coral communities were likely to have been more influenced by bleaching in 2002 than in 1998 (Appendix 1: Table A1-5.2).

In 2005 the benthic communities at 2m had average cover of soft coral (10%), average cover of macroalgae (15%) and below average hard coral cover (23%, Figure 5.15). The hard coral community was dominated by Poritidae (mostly branching *Porites* spp) and the foliose *Pavona cactus* which represented 43% and 40% of all corals respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average; recruits were primarily Faviidae, Poritidae, Acroporidae and Mussidae (Figure 5.16, Appendix 1: Table A1-5.5). At 5m cover of soft coral (2%) and hard coral (18%) were low while cover of macroalgae was near average (Figure 5.15). The hard coral community was dominated by massive and branching *Porites* spp and the occasional Faviidae colony which represented 49% and 19% of all recorded hard corals respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was near average; there were a low numbers of recruits from a range of families but Faviidae, Poritidae and Mussidae were most abundant (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 cover of hard and soft coral at 2m was very similar to 2005 as was the composition of the hard coral community (Figure 5.15, Appendix 1: Table A1-5.3). The over all density of recruits was slightly lower due to fewer colonies in the <2cm size class (Figure 5.16). At 5m the cover of soft coral increased slightly and hard coral decreased (Figure 5.15). The hard coral community on the deep slope was still dominated by massive and branching *Porites* spp and the occasional Faviidae colony (Appendix 1: Table A1-5.3). The density of recruit-sized colonies increased to above average levels for all size classes and the dominant coral recruits were from the same of families as in 2005 (Figure 5.16, Appendix 1: Table A1-5.5).

Some colonies of *Porites cylindrica* on the shallow slope were observed to have split apart in 2006 and this may account for the drop in the representative cover of this genera. There were very few colonies of *Acropora* spp representing only 6% and 11% of all corals at 2m and 5m. There was an above average density of recruit-sized colonies both depths indicating some growth potential of this community.

Dent Island

This reef was included in a survey in 1991 (van Woesik and Ayling, 1992) when HC cover was 64% at 3m and 40% at 6m; Poritiidae accounted for approximately 40% of the community at both depths. The reef was also included in bleaching surveys by GBRMPA in April 1998 when hard coral cover was 49% on both the reef crest and slope. Following bleaching cover was reduced to 30% on the reef crest and 39% on the slope. The 2002 bleaching is likely to have affected reef communities at this location, however there no quantitative data available (Appendix 1: Table A1-5.2). The sites chosen for this study correspond closely to areas surveyed in 1991 and 1998.

In 2005 the benthic communities at 2m had an average cover of soft coral (9%), high cover of hard coral (53%) and very little macroalgae (Figure 5.15). The hard coral community was dominated by Acroporidae mostly (branching *Acropora* spp) and Poritidae which represented 43% and 39% of the coral community respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was above average with Acroporidae and Poritidae most abundant (Figure 5.16, Appendix 1: Table A1-5.5). At 5m the cover of soft coral was low (4%), cover of hard coral high (55%) and again there was very little macroalgae (Figure 5.15). The hard

coral community was again dominated by Acroporidae (mostly branching *Acropora* spp) and Poritidae which represented 34% and 31% of all corals respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for all size classes and recruits were primarily Poritidae, Mussidae and Acroporidae (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 hard coral cover at 2m increased to 59% while at 5m there was a slight decrease to 50%, the cover of soft coral remained similar to that in 2005 (Figure 5.15). At both depths there was a decline in the density of recruit-sized colonies, at 2m the density was still around average though at 5m the density in 2006 was low (Figure 5.16). Numbers of recruits of the family Poritidae were the most reduced at both depths (Appendix 1: Table A1-5.5).

The main change in the community since first surveyed in 1991 appears to be an increase in the cover and proportional representation of the Acroporidae. The high cover observed in 2005 and 2006 suggest the impacts of bleaching have been largely negated. The high cover of Acroporidae and *Acropora* spp recruits indicates the continued increase in dominance of this family given no disturbances in the near future.

Hook Island

The fringing reef on the south-western side of Hook Island (False Nara Inlet) was first examined in 1995 (DeVantier and Turak 1995). They described the shallow community as having a coral cover of 11-30%, dominated by large (>50cm) colonies of *Acropora* spp and massive/branching *Porites* spp and a diverse soft coral community with 11-30% cover. The deep sites were described as having a coral cover of 31-50% coral cover and dominated by massive/branching *Porites* spp and sub-massive *Goniopora* spp with a few *Acropora* spp colonies >50cm; soft coral cover on the deep site was estimated at between 11-30%. While no disturbances to this community have been documented it is likely that bleaching in 1998 and 2002 would have caused some mortality (Appendix 1: Table A1-5.2).

The sites chosen for this study are in the vicinity of sites 14a (shallow) and 14b (deep) described in DeVantier and Turak (1995). In 2005 the benthic communities at 2m had an above average cover of soft coral (24%), below average hard coral cover (26%) and very little macroalgae (Figure 5.15). The hard coral community was dominated by massive and branching *Porites* spp and Acroporidae (mostly branching *Acropora* spp) which represented 52% and 22% of all corals respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was near average with the families Faviidae, Poritidae and Acroporidae most abundant (Figure 5.16, Appendix 1: Table A1-5.5). At 5m the benthic communities had above average cover of soft coral (18%), below average cover of hard coral (28%) and again very little macroalgae (Figure 5.15). The hard coral community on the deep slope was dominated by Poritidae (mostly massive and branching *Porites* spp, Faviidae and Acroporidae (mostly branching *Acropora* spp) which represented 56%, 17% and 11% of all corals respectively (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was slightly below average with recruits of the families Faviidae, Poritidae and Mussidae most abundant (Figure 5.16, Appendix 1: Table A1-5.5).

In 2006 at 2m there was a slight decline in the cover of hard coral, an increase in the cover of soft coral (Figure 5.15). The main change to the hard coral community was a reduction in the representation of Poritidae from 51% to 42% of the coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was lower with abundance of recruits declining in most

families (Figure 5.16, Appendix 1: Table A1-5.5). At 5m there was little change in the benthic community (Figure 5.15).

Benthic communities in 2006 differed little from those described in 1995.

Fitzroy NRM region

Six reefs at different distances from the mouth of the Fitzroy River were selected to represent a gradient of likely exposure to river borne contaminants (see Figure 5.17). Plumes from the Fitzroy River flow predominantly northwards with limited vertical mixing during spring tides (Van Woesik 1991) though can flow out to the east and south during northerly winds (Devlin *et al*. 2001).

Historical data are available for three of the six reefs selected in this study. Humpy, Halfway and Middle Island reefs were first monitored in 1989 and 1991 as part of an impact study into the effects the 1991 Fitzroy River flood (Van Woesik 1991). Sites on these reefs have been monitored by staff of Queensland Parks and Wildlife Service (QPWS) from 1993 (Middle Island) or 1996 (Halfway Island). North Keppel Island reefs have been monitored by QPWS since 1995. The QPWS sites at Halfway Island do not correspond closely with sites in this present study. The QPWS site at Middle Island corresponds closely with the Middle Island site in this study though QPWS transects run transversely across the reef slope rather than parallel to the slope. The QPWS site at North Keppel Island coincides closely with the 5m depth at site 2 in this study. In addition Sweatman *et al.* 2006 report surveys from 2004 for each reef included in this study with the exception of Barren Island. These surveys used the same methods as this study and also included the compilation of a species list of hard corals found at each site. In 2004 sites were not fix and as such may vary marginally from those used in this study. Sites at Peak Island and Humpy Island do not correspond between the two studies.

Between 1991 and 2006, disturbance events substantially reduced the hard coral cover at three of the six study sites and may have affected the other three sites similarly, though this was not documented (Appendix 1: Table A1-5.2). The most severe disturbance was the Fitzroy River flood in 1991. At depths of less than 1.5m hard coral cover declined by 85% at Humpy, Halfway and Middle Island; mainly the dominant Acroporidae and Pocilloporidae were lost (Van Woesik 1991). Subsequent declines in hard coral cover were associated with coral bleaching in 1998, in 2002 and again in 2006 (QPWS, R. Berkelmans pers com.). Coral cover showed rapid recovery following bleaching in 1998 on monitored reefs (Sweatman *et al.* 2006). No crown-of-thorns starfish have been observed on these reefs.

Comparing between surveys in 2004 (Sweatman *et al.* 2006) and 2005 do not indicate any great change in coral communities. One possible exception maybe the marked increase in cral cover at 2m at Pelican Island from an estimated 14% in 2004 to 28% in 2005. Slight increases or decreases in cover or density of recruits sized colonies at other reefs may be as much an artefact of transect placement as due to real differences among years as sites were not marked in 2004.

There is a clear distinction in the benthic communities between those closest to the Fitzroy River (Peak Island and Pelican Island) and those farther offshore. The reefs closer to river have significantly lower cover of hard corals and markedly higher cover of macroalgae. It is not only the cover of hard corals but also the community composition that differs among these

reefs. The four reefs most distant from the river have coral communities dominated by large stands of branching *Acropora* spp., with the cover of the family Acroporidae constituting at least 96% of the community at these reefs (see cluster group 2, Figure 5.5, Table 5.6). In contrast branching *Acropora* spp. are less common at Peak Island and Pelican Island, especially at 5m were the family Acroporidae constitutes only 1.2% and 3.4% of the communities respectively. The communities at Peak Island and Pelican Island are unique amongst all reefs surveyed in this study (see cluster group 5 Figure 5.5, Table 5.6). Interestingly the richness of genera over all and also of genera recorded in the recruit-sized categories is higher on the two reefs closest to the river, though this is largely a reflection of the total dominance of the family Acroporidae on the other reefs. Regionally overall genus level richness and the richness of genera represented by recruit-sized colonies is significantly lower than NRM regions to the north.

Peak Island

There are no data available describing the reef community at this location prior to 2004 (Sweatman *et al*. 2006). Peak Island is the closest reef to the mouth of the Fitzroy River, so it is likely that the major flood of the Fitzroy River that caused substantial loss of coral on reefs further north (Humpy and Halfway Islands, Van Woesik (1991)) also inundated this reef. Modelling suggests that Peak Island reefs experienced conditions that lead to coral bleaching in 1998 and again in 2002 (Appendix 1: Table A1-5.2).

The benthic community at 2m had low cover of soft coral (2%) and hard coral (16%) and very high cover of macro-algae (64%) (Figure 5.17). Cover of hard corals consisted mostly of the families Faviidae and Acroporidae, which together made up 75% of coral cover (Appendix 1: Table A1-5.3). At 5m the cover of soft coral was near average, (8%) the cover of hard coral higher (25%) than at 2m though still below average and the cover of macroalgae was lower than at 2m though still very high (Figure 5.17, Appendix 1: Table A1-5.3). The hard coral community at 5m was unusually dominated by Siderastreidae that accounted for 47% of coral cover and Faviidae (30%, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was very low at both depths (Figure 5.18). At 2m recruit-sized colonies of the families Acroporidae and Faviidae were most abundant while at 5m *Porites* were the most abundant (Appendix 1: Table A1-5.5).

Proximity to the Fitzroy River mouth, below average coral cover and lack of a substantial carbonate reef base all suggest Peak Island provides a marginal habitat for hard corals. Low densities of recruit-sized colonies suggest hard coral cover will remain low at least in the short term.

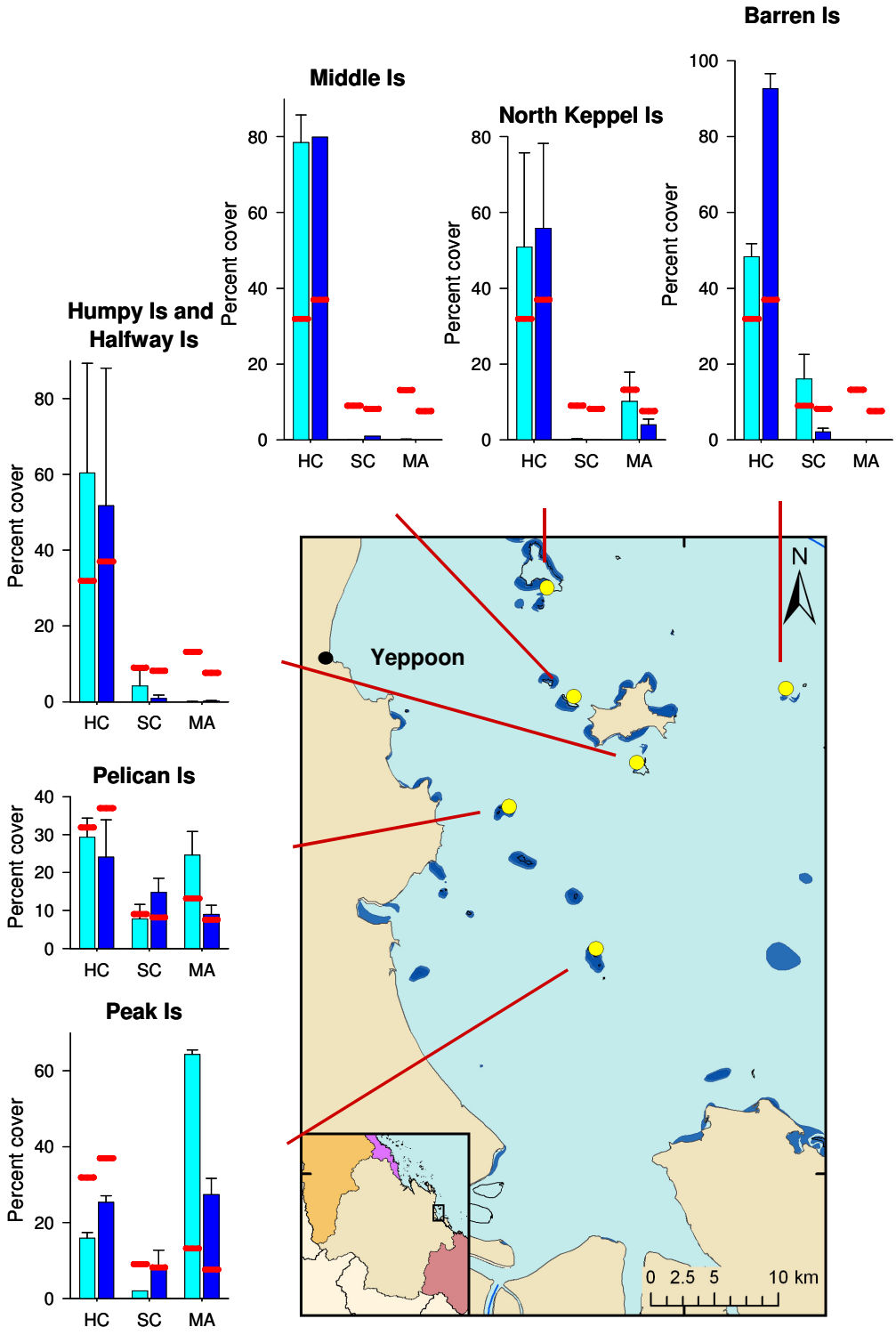

Figure 5.17. Percent cover estimates of major benthic groups, hard coral (HC), soft coral (SC) and macroalgae (MA) on reefs in the Fitzroy region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each group and depth from all reefs and NRM regions combined are indicated by red lines. For each benthic group at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

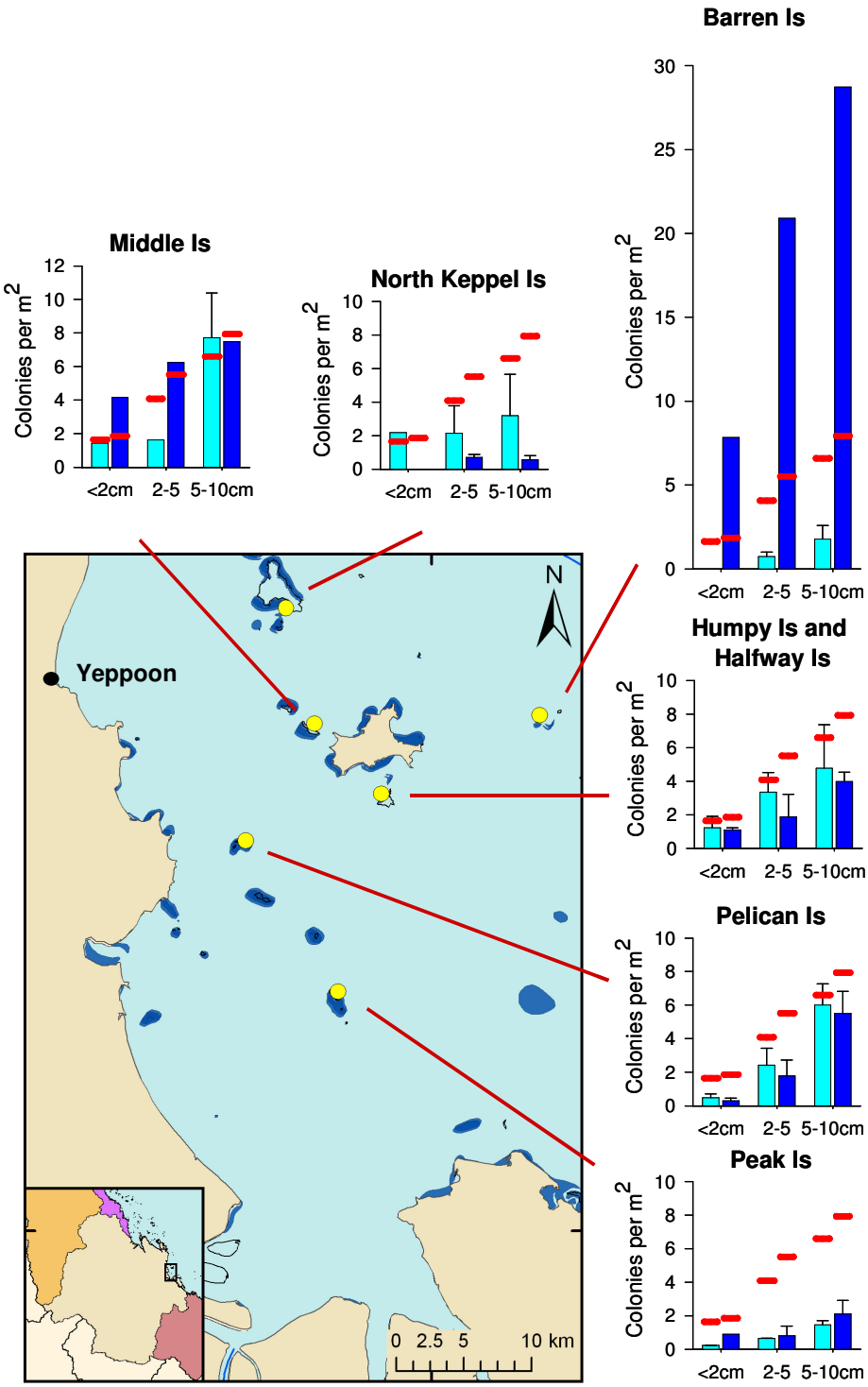

Figure 5.18 . Density of recruit-sized hard coral colonies by size class for reefs in the Fitzroy region. Pale blue bars represent values for 2m depth and mid blue bars for 5m depth. Average values for each size class and depth from all reefs and NRM regions combined are indicated by red lines. For each size class at each depth the left hand bar represents 2005 data and the right hand bar 2006 data.

Pelican Island

There are no data describing changes to the reef community at Pelican Island reefs prior to 2004. Being close to the mouth of the Fitzroy River means the reefs are exposed to flooding. The large flood in 1991 which caused dramatic loss of hard coral cover (85%) on reefs further north (Humpy and Halfway Islands, Van Woesik [1991]) would also have inundated these reefs. Modelling work suggests that coral communities on these reefs are likely to have been affected by coral bleaching in 1998 and 2002 (Appendix 1: Table A1-5.2).

In 2005 the benthic community at 2m had average cover of both soft coral (8%) and hard coral (28%) and above average cover of macroalgae (Figure 5.17). The families Acroporidae and Faviidae together comprised 85% of the cover of hard corals (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average (Figure 5.18). Recruits were mainly Acroporidae and Faviidae (Appendix 1: Table A1-5.5). The benthic community at 5m had above average cover of soft coral cover (15%) and less than average cover of hard coral (24%); 32% of the hard coral cover was Faviidae and 22% was Merulinidae (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was below average for all three size-classes and the majority were Faviidae and Dendrophylliidae (Figure 5.18, Appendix 1: Table A1-5.5).

The hard coral cover at 2m is double that observed in 2004 from very similar sites (Sweatman *et al.* 2006). The majority of this increase can be attributed to an increase in the cover of the family Acroporidae. The increasing cover and ongoing recruitment of this family indicates likely further increases in cover, though this is subject to disturbance. There is little accumulation of carbonate substrate at this location a fact suggesting coral communities are likely to be transient in the longer term.

Humpy and Halfway Island

The coral reef communities along the western side of Humpy and Halfway Island have previously been heavily damaged by floods and coral bleaching. The most significant of these impacts was the 1991 Fitzroy River flood, which caused 100% coral mortality down to a depth of 1.3m (below datum) and saw hard coral cover decline from 66% to 0% (Van Woesik, 1991). Below this depth coral cover was very high between 1996 and 2002 (QPWS monitoring, in Sweatman *et al*., 2006). There were small declines due to coral bleaching in 1998 (6%) and 2002 (22%, Appendix 1: Table A1-5.2)

In 2005 benthic communities at 2m had average cover of soft coral (4%) and well above average cover of hard coral (60%) (Figure 5.17). The hard coral community on the shallow slope was dominated by large branching *Acropora* spp which made up 99.5% of all hard coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average for the 0 - 5cm size-class but below average for the 5 -10cm size-class (Figure 5.18). Most recruits were Acroporidae (68%), Pocilloporidae (16%) and Faviidae (13%) (Appendix 1: Table A1-5.5). Benthic communities at 5m also had below average cover of soft coral (1%), above average cover of hard coral (51%) and the dominant hard coral family being Acroporidae (99.5%) (Figure 5.17, Appendix 1: Table A1-5.3). The density of recruits was below average for all three size-classes and dominated by recruits from the Faviidae (38%) and Acroporidae (35%) families (Figure 5.17, Appendix 1: Table A1-5.5).

Monitoring of sites nearby on this reef has shown similar branching *Acropora* dominated communities to be resilient to disturbance once below the influence of floods. Though

impacted by bleaching recovery is typically rapid. Given low numbers of recruit-sized colonies it is not clear how rapidly these communities could recover should the large stands of *Acropora* be removed by a future disturbance.

Middle Island

Middle Island reefs have been damaged by flooding and coral bleaching in the past. Van Woesik recorded a 27% decline in hard coral cover associated with the flooding of the Fitzroy River in 1991. Hard coral cover declined from 35% to 8% and all *Acropora* spp and *Pocillopora* spp in less than 1.5m depth (below datum) were killed; the *Acropora* spp community below 2m depth was badly bleached (Van Woesik, 1991). The hard coral community then recovered quickly and had increased to 68% by 1993, when QPWS surveys began. Hard coral cover then increased slowly reaching 78% in 1997. Cover declined by 40% due to coral bleaching in 1998 but had recovered to 60% by 2003 (Sweatman *et al*., 2006)

In 2005 the benthic communities at 2m had below average cover of soft coral (0.1%) and well above average cover of hard coral (78%), macroalgae were all but absent (Figure 5.17). The hard coral community was dominated by Acroporidae (almost exclusively large branching *Acropora* spp) which represented 99.4% of coral cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was average (Figure 5.18). Most recruits were Acroporidae (Appendix 1: Table A1-5.5). The benthic communities at 5m also had below average cover of soft coral (1%) and very high hard coral cover (80%), 96.1% of the hard coral cover was again made up Acroporidae (Figure 5.17, Appendix 1: Table A1-5.3). The density of recruit-sized colonies was marginally above average and again dominated by Acroporidae (Figure 5.18, Appendix 1: Table A1-5.5).

Middle Island reef communities appear to have recovered well from previous disturbance events. High coral cover and substantial numbers of *Acropora* recruits suggest the effects of the 1998 and 2002 coral bleaching events were short term.

North Keppel Island

North Keppel Island has been monitored since 1995. There is no record of flood damage but the 1991 flood plume from the Fitzroy River did extend to this reef (Devlin *et al*., 2001). Between 1995 and 1997, QPWS surveys recorded hard coral cover around 39%. Following the 1998 bleaching event cover declined to 34% before increasing to 52% just prior to the 2002 coral bleaching. Cover had dropped to 33% in October 2002 (Sweatman *et al*., 2006). The sites used in this study are on the same stretch of reefs as monitored by QPWS.

In 2005 the benthic communities at 2m had below average cover of soft coral (0.1%), well above average cover of hard coral (51%) and macroalgae cover was marginally below average (Figure 5.17). The hard coral community on the shallow slope was dominated by large branching *Acropora* spp which represented 98.5% of all corals; only 1.5% of corals were from other families (Appendix 1: Table A1-5.3). The density of recruit-sized hard coral colonies was below average (Figure 5.18); recruits from family Acroporidae were most numerous. The benthic communities at 5m also had below average cover of soft coral (0.1%), above average cover of hard coral (56%) and slightly below average cover of macroalgae (Figure 5.17). The hard coral community again consisted of large colonies of branching *Acropora* spp which accounted for 96.8% of the cover (Appendix 1: Table A1-5.3). The density of recruits was very low with only Acroporidae and a single Fungiidae recruit observed (Figure 5.18, Appendix 1: Table A1-5.5).

Previous observations in combination with the results of this study highlight the stability of North Keppel Reefs to date. Previous disturbances have had obvious impacts on the coral community however coral cover has not fallen below 33% during the 13 years of monitoring. The dominance of large *Acropora* spp colonies indicate previous disturbance events have failed to result in widespread mortality and subsequent quick increases in coral cover suggest community recovery has been rapid. Below average recruit densities indicate that like other *Acropora* dominated reefs in the region, North Keppel reef communities may be particularly susceptible to disturbances which result in large scale mortality amongst the adult population. Recovery from such events would likely be very slow given the very low numbers of recruits.

Barren Island

There is no data on benthic community prior to this survey. Between 1991 and 1999 flood plumes extended to this reef in 1991 however the impact on coral communities was not documented (Devlin *et al*., 2001). Previous work 9km to the south-west at Halfway Island revealed dramatic declines in hard coral cover in shallow (<1.5m) habitats associated with flooding of the Fitzroy River in 1991 and smaller declines with coral bleaching in 1998 and 2002 at deeper depths (Van Woesik, 1991; Sweatman *et al*., 2006). It is likely that this reef was similarly impact by these bleaching events (Appendix 1: Table A1-5.2).

In 2005 the benthic community at 2m had above-average cover of both soft coral (16.1%) and hard coral (48%) and no macroalgae (Figure 5.17). The hard coral community was dominated by Acroporidae which represented 96% of the cover (Appendix 1: Table A1-5.3). The density of recruit-sized colonies was very low (Figure 5.18) with only a few recruits from the families Acroporidae, Faviidae and Pocilloporidae observed (Appendix 1: Table A1-5.5). At 5m the cover of hard coral was extremely high (92.6%), cover of soft coral was low (2.1%) and macroalgae were absent (Figure 5.17). The hard coral community was again dominated by Acroporidae (branching *Acropora* spp 99.9%, Appendix 1: Table A1-5.3). The density of recruits was also calculated as being extremely high however this was largely an artefact of the very high coral cover that effectively limited the available substrate for recruitment; 82% of recruits were from the family Acroporidae (Figure 5.18, Appendix 1: Table A1-5.5).

The large difference in cover between the 2m and 5m sites suggests differing disturbance regimes at the two depths. The coral communities at both depths were thriving.

Discussion

The coral communities on nearshore reefs of the Great Barrier Reef varied greatly in all the characteristics that were measured. Total coral cover, which is the simplest and most commonly-used indicator of reef status, varied from very high to very low. It was generally high on the reefs of the Fitzroy region and in the Mackay Whitsunday region and lower on reefs in the Wet Tropics and Burdekin regions. Diversity, as measured by number of genera, was highest on reefs of the Mackay Whitsunday region and low on the reefs of the Fitzroy region, which had high coral cover but heavy dominance of a few *Acropora* spp. The densities and diversity (number of genera) of colonies <10cm in diameter were highest in the Wet Tropics region and the Mackay Whitsunday region. Reefs in the Wet Tropics have relatively low coral cover, most probably due to mortality from bleaching in 1998, but the numbers and diversity of recruit-sized colonies suggests that recovery is proceeding. In summary, the surveys gave an optimistic view of the status of the majority of nearshore reefs in most regions. Reefs in the Burdekin region give the greatest cause for concern, having both relatively low coral cover and relatively low densities of recruit-sized colonies.

The general lack of clear relationships between characteristics of coral communities and the existing indices of water quality is not surprising for several reasons. One reason is the limited data on local water quality regimes and the relative crudeness of the predictive indices. Limited numbers of water samples have been taken from near to the survey reefs in only one wet season and one dry season as part of the Reef Plan MMP. Even so, the composition of hard coral communities did show a relationship with the Water Quality Index based on the Reef Plan MMP water quality data. This is a long-term program and a better picture of both the average and the range of local conditions should emerge from more sampling supplemented by the use of loggers and remote sensing information. Among the other indices, Distance to Shore will only show the most general relationship to water quality. The Predicted Water Quality Gradient is a simple ranking within regions. The Ecosystem Risk Index (ERI, Devlin *et al*., 2003) is a more sophisticated attempt to estimate likely exposure to runoff on the Great Barrier Reef. It is based on a River Pollution Index (RPI) that includes a simplistic estimate of sediment loads and weights all proxies for pollution, except urban development, equally on linear scales. The RPI is combined with rough estimates of extents of flood plumes and a simplistic linear dilution factor to give the ERI. While the resulting estimates show qualitative correspondence to preconceptions of regional risk to near-shore habitats from terrestrial runoff, the scaling of ERI values is very dubious. Knowledge of the effects of components of runoff on reef organisms (as summarised by Fabricius 2005) could be used to scale the component functions in the RPI. The revised RPI could then be combined with basic hydrodynamic models for the Great Barrier Reef lagoon to give more a realistic estimate of ecosystem risk.

A second reason is that, even if good estimates of local water quality regimes were available, corals are long-lived organisms and the response of coral communities to sub-lethal changes in conditions may be slow. Land use and agricultural practices have changed significantly in the past 40 years and the present communities of large colonies could be relics that settled and became established when conditions were more favourable. Many components of aggravated runoff are particularly harmful to coral larvae and juvenile corals (Fabricius, 2005) while effects on large established colonies are not so obvious. Existing communities, particularly those made up of large colonies may not be representative of the community that would be re-established in the current surroundings if existing communities were removed by a major disturbance.

A third reason is that disturbances are likely to disrupt relationships between the coral reef communities and the water quality regime. Disturbances to nearshore reefs occur frequently (Appendix 1: Table A1-5.2) but effects of events like cyclones, though severe, may be localised (e.g. Cheal *et al*., 2000). Time since the last major disturbance is going to have an overriding influence on measures such as coral cover on a reef. Since coral taxa have different growth rates, the rate of recovery will vary among the community types.

These surveys were specifically designed to include information on recruits and juvenile coral colonies, because of their known vulnerability to components of runoff. The surveys found diverse cohorts of juvenile corals in many regions. There are logical reasons why reefs in the Johnstone / Russell-Mulgrave sub-region are judged to be at high risk of exposure to aggravated runoff. These reefs were badly damaged by coral bleaching in 1998, but now many areas have large numbers of small colonies and the densities of recruits recorded on settlement tiles in 2005 are among the highest that have been recorded anywhere on the Great Barrier Reef. While fears of complete recruitment failure seem unfounded, the big difference between the density of newly-settled corals on the tiles and the density of juvenile colonies found on the transects implies that there is massive mortality of corals in their first year on the reef. At present we lack any clear reference points for what is a reasonable rate of recovery after disturbance for the different types of nearshore coral community and what settlement and survival rates would be required to produce such recovery.

Conclusions

Two aspects of the variability in nearshore coral reef communities stand out. The first is just how much reef communities varied. This is particularly striking because one of the selection criteria for the survey sites was the presence of a substantial platform of calcareous rock derived from persistent reef building corals; we rejected sites where coral colonies were present, but the lack of any substantial accumulation of calcareous rock suggested that their presence was ephemeral (we sought "Coral reefs" rather than "Coral communities" *sensu* Van Woesik and Done, 1997). The second salient characteristic was the fine spatial scale of the variation – neighbouring reefs could have quite different coral communities. This fine spatial variation may be instructive; it suggests that dissolved contaminants play a minor role in determining community structure as their concentrations would be expected to vary along regional dilution gradients, but not in complex patterns over such small distances. Sedimentation and associated turbidity relate to local hydrodynamics, which can vary on a fine scale. The broad scientific question that emerges is: which environmental variables cause this high level of fine-scale variability in communities? The questions for Reef Plan are: how much of this variation is due to differences in water quality and, in particular, how much of that can be reduced through improvements in land use practice?

These surveys represent a baseline for assessing future changes in coral communities on nearshore reefs of the Great Barrier Reef. Targeted monitoring of water quality at local scales, enhanced by instruments and remote sensing, will provide more precise characterisation of the averages and the ranges of conditions that the survey sites experience. The objective of the Reef Plan is to improve water quality entering the reef, so these conditions are expected to change. Detecting corresponding changes in reef communities depends on a clear understanding of the relation between communities and the water quality regime, a comprehensive record of the timing and intensity of disturbances and an understanding of the processes and time-scales of recovery. The ability to recover from disturbances is fundamental to the persistence of communities in the long term. The inclusion

of studies of settlement and collection of information on colony size structure in the surveys is based on assumptions about the dynamics of communities: that sites with higher rates of settlement will in general later support greater numbers of juvenile corals and then of adult colonies. These assumptions need to be substantiated by careful analysis of future monitoring results, perhaps backed up by studies of the processes.

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6. Intertidal Seagrass Monitoring

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Introduction

Monitoring of the major marine ecosystem types in the Great Barrier Reef region most at risk from land based sources of pollutants is being conducted to ensure that any change in their status is identified. Monitoring sites are located in proximity to river mouths with some coastal chlorophyll monitoring sites to enable comparisons with water quality information.

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. For this reason seagrasses have been nominated as an ecosystem type to be monitored.

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. 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.

One of the paramount requirements of the Reef Plan MMP, apart from being scientifically robust, is that its findings must have broad acceptance and ownership by the community. It was identified very early in development of the Reef Plan, that the existing Seagrass-Watch program was a platform on which the inshore seagrass monitoring component could be based.

The key aims of this task in the Reef Plan MMP are to:

- Detect long-term trends in seagrass abundance, community structure, distribution, reproductive health and nutrient status from representative intertidal seagrass meadows in relation to large river inputs into the GBRWHA.
- Detect long-term trends in levels of ecologically significant herbicides and nutrient pollutants from representative intertidal seagrass meadows in relation to large river inputs into the GBRWHA.
- Work closely with and involve community partners (Seagrass-Watch) to ensure broad acceptance and ownership of the Reef Plan by the Queensland and Australian community.

Inshore seagrasses in Queensland and the western Pacific appear to be in a fair to good condition with most impacts localised (McKenzie and Coles, 2005). Regional climate appears the most likely driver of observed changes but local characteristics of the seagrass populations and physical characteristics at sites are also likely to have had a significant effect on the observed seagrass changes. This report provides the intertidal seagrass monitoring component

of the Reef Plan MMP. For detailed reports on each location/region visit the Long-term monitoring section of the website at www.seagrasswatch.org.

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Inter-tidal seagrass monitoring

Survey methodology followed Seagrass-Watch standard methodology (www.seagrasswatch.org, McKenzie *et al*., 2004, 2005a, b). At each sampling location, sampling includes two sites nested in location and three 50m transects nested in each site. A site is defined as a 50m x 50m area within a relatively homogenous section of a representative seagrass community/meadow (McKenzie *et al*., 2000).

Community-based monitoring at the 22 sites identified for the Reef Plan MMP long-term intertidal monitoring (Table 6.1) in April and October of each year is supervised on-site by a qualified and trained scientist. Monitoring conducted outside these months, is conducted by trained community volunteers.

Sites are monitored for seagrass cover and species composition. Additional information is collected on canopy height, algae cover and epiphyte cover and macrofaunal abundance. Sites that were monitored, their position and species community types are presented in Table 6.1.

Edge mapping

Mapping the edge of the seagrass meadow within 100m of each monitoring site was conducted at all sites in the October 2005 and April 2006 monitoring periods. Training and equipment (GPS) were provided to personnel involved in the edge mapping.

Seagrass reproductive health (status)

Seagrass reproductive health was assessed from *Halodule uninervis* seed bank monitoring (refer to www.seagrasswatch.org for detailed methods) and from samples collected in October 2005 and April 2006 at locations identified in Table 6.1. At each location, 15 seagrass samples were haphazardly collected from an area of approximately 100 m surrounding the Seagrass Watch site using a PVC corer, washed to remove sediment, and frozen. In the laboratory the reproductive structures of the plants were identified and counted. After processing, samples were retained for future verification if required. Data collected from each core included: number of nodes with shoots for each species present, number of flowers and fruits for each species present. Averages per site were calculated based on the presence per core of each parameter.

Table 6.1. Seagrass-Watch sites selected for Reef Plan MMP intertidal seagrass long-term monitoring

GBR region	NRM Board	Catchment	Seagrass monitoring location	Site		Latitude		Longitude		Seagrass community type
Far Northern	Cape York	Endeavour	Cooktown	AP1	Archer Point	15°	36.5	145°	19.143	*Halodule univervis/ Halodule ovalis* with *Cymodocea/T. hemprichii*
				AP2	Archer Point	15°	36.525	145°	19.108	*H. univervis/H. ovalis* with *C. rotundata*
Northern	Wet Tropics	Daintree	*NA*							
		Russell / Mulgrave Johnstone	Green Island	GI1	Green Island	16°	45.789	145°	58.31	*C. rotundata/T. hemprichii* with *H. uninervis/H. ovalis*
				GI2	Green Island	16°	45.776	145°	58.501	*C. rotundata/T. hemprichii* with *H. uninervis/H. ovalis*
			Cairns	YP1	Yule Point	16°	34.159	145°	30.744	*H. uninervis* with *H. ovalis*
				YP2	Yule Point	16°	33.832	145°	30.555	*H. uninervis* with *H. ovalis*
		Tully	Mission Beach	LB1	Lugger Bay	17°	57.645	146°	5.61	*H. uninervis*
				LB2	Lugger Bay	17°	57.674	146°	5.612	*H.uninervis*
Central	Burdekin	Herbert	*NA*							
		Burdekin	Magnetic island	MI1	Picnic Bay	19°	10.734	146°	50.468	*H. uninervis* with *H. ovalis & Zostera/T. hemprichii*
				MI2	Cockle Bay	19°	10.612	146°	49.737	*H. ovalis* with *C. serrulata* and *T. hemprichii/H. uninervis*
			Townsville	SB1	Shelley Beach	19°	11.046	146°	45.697	*H. uninervis* with *H. ovalis & Zostera/H. spinulosa*
				BB1	Bushland Beach	19°	11.028	146°	40.951	*H. uninervis* with *H. ovalis*
	Mackay / Whitsunday	Proserpine	Whitsundays	PI2	Pioneer Bay	20°	16.176	148°	41.586	*H. uninervis/H. ovalis* with *Zostera & H. spinulosa*
				PI3	Pioneer Bay	20°	16.248	148°	41.844	*H. uninervis/H. ovalis* with *Zostera/H. spinulosa*
		Pioneer	Mackay	SI1	Sarina Inlet	21°	23.76	149°	18.2	*H. uninervis* with *H. ovalis (Z. capricorni)*
				SI2	Sarina Inlet	21°	23.712	149°	18.276	*Zostera capricorni*
Southern	Fitzroy Basin Association	Fitzroy	Shoalwater Bay	RC	Ross Creek	22°	22.953	150°	12.685	*Zostera capricorni*
				WH	Wheelans Hut	22°	23.926	150°	16.366	*Zostera capricorni*
	Burnett-Mary	Burnett	Gladstone	GH1	Gladstone Hbr	23°	46.005	151°	18.052	*Zostera capricorni* with *H. ovalis*
				GH2	Gladstone Hbr	23°	45.874	151°	18.224	*Zostera capricorni* with *H. ovalis*
		Mary	Hervey Bay	UG1	Urangan	25°	18.053	152°	54.409	*Zostera capricorni* with *H. ovalis & H. uninervis*
				UG2	Urangan	25°	18.197	152°	54.364	*Zostera capricorni* with *H. ovalis*

Herbicide Sampling

Sediment samples for analysis of herbicide concentrations were collected from all sites in April/May 2005 and March/April 2006, with the exception of Shoalwater Bay, Gladstone and Urangan in 2005.

In 2005, Shoalwater Bay sampling was not conducted until September 2005 as access to the area was not permitted due to defence forces activity in the area. Similarly, Gladstone and Hervey Bay sites were not sampled until October 2005 as sites were either not established or there was insufficient time to train on-ground staff. Sites could not be established sooner at Gladstone Harbour as there was insufficient information available to determine suitable site locations; sites need to be within representative meadows for that region. A survey by DPI&F for the Port of Gladstone in October 2005 provided this advice and suitable sites were sampled in October.

For herbicide sampling, three replicate sediment samples were collected at each site. Collection, storage and analysis methods are detailed in the Reef Plan MMP methodologies. Results are presented as µg/kg Dry Weight.

Sediment and Seagrass tissue nutrients

In September/October 2005, samples were collected from all 22 monitoring sites for analysis of sediment and tissue nutrient (*Halodule uninervis*, *Halophila ovalis* and *Zostera capricorni*).

Five haphazardly placed 0.25m^2 quadrats were harvested from an area adjacent (<25m), of similar cover and species composition, to each monitoring site. Five sediment cores were also collected from each 0.25m^2 quadrat prior to harvesting, for the analysis of sediment porosity.

In the laboratory, seagrass leaves from each harvested sample were separated and epiphytic algae removed by scraping. Samples were oven dried at 60°C to a constant weight. Dried biomass samples of leaves was then homogenised by milling to fine powders prior to nutrient analyses. An inconsistent problem occurred during the grinding phase of the seagrass processing. Some samples contained glass bead fragments from the grinding process. As the determination of %C, %N and %P is calculated on a *w/w* basis, the extra weight provided by the glass fragments gave erroneous measurements. These contaminated samples were eliminated from the reporting of % C, % N and % P. Clean samples (i.e. those not contaminated with glass beads) had tissue nitrogen and phosphorus extracted using a standardised selenium Kjeldahl digest. Percent tissue C was determined by atomic absorption spectrometry. The N and P concentrations of these samples were determined with an automatic analyser using standard techniques at a Quality Assured and NATA certified laboratory. The uncontaminated samples from which % C, % N and % P were derived are presented in Table 6.9 and all locations sampled are represented.

Comparing deviations in the ratios of carbon, nitrogen and phosphorus (C: N: P) retained within the plant tissue has been used extensively as an alternative means of evaluating the nutrient status of coastal waters (Duarte 1990, Johnson *et al*. 2006). Contamination within the sample would have been consistent, for all elemental determinations, the atomic ratio would have remained unaffected (pers. com. Dan Wruck, QHSS). C: N: P ratios were therefore calculated for all samples using atomic

weights to allow for a statistical testing of tissue nutrient status with replication across all locations.

Nitrogen and phosphorus were extracted using a standardised selenium Kjeldahl digest and the concentrations determined with an automatic analyser using standard techniques at a Quality Assured and NATA certified laboratory. N:P ratios were calculated using atomic weights.

Adsorbed exchangeable ammonium (NH_4) and nitrogen oxides (NO_3^-, NO_2^-) were extracted using KCl. To extract adsorbed phosphate (PO_4), the method described in Colwell (1963) was used. This technique provides information on the total amount of adsorbed PO_4 regardless of the mineralogy or pH of the sediment. Chemical analyses of all inorganic nutrients were determined using a Skalar segmented flow auto-analyser using standard water quality techniques.

Sediment volume was measured from the sediment cores and particle size density (PSD) and porosity (Φ) calculated for converting adsorbed nutrients units ($\mu molkg^{-1}$) to equivalent units ($\mu mol\ L^{-1}$ sediment) to enable the molar ratios of the total sediment nutrient pool to be calculated.

Statistical analysis

Sediment physical characteristics

All sediment physical characteristic variables were analysed using a General Analysis of Variance with Location and Site as factors and Site nested within Location. Normality of the data was checked using standardised residual plots. Where data showed non-normal tendencies, the variables were transformed accordingly (See Table 6.2). GenStat® was used to detect data outliers (i.e. observations having unusually large residual values that fell outside the range of the response data and the model design). Where there were outliers in the data, analysis was re-run excluding outliers to determine if there was any influence on the ANOVA outcome. The results of these reruns showed the outliers had no influence and all data was included. Results indicated that variation within locations was negligible compared to the differences between locations. Hence, Sites were considered as replicates. Results are reported for locations and graphed as mean (x) ± 95% Confidence Interval (CI).

Table 6.2 Transformation performed on each sediment characteristic parameter prior to analysis.

Parameter	Transformation
NH_4	Log
PO_4	Log
NO_2+NO_3	No analysis
N:P	Log
Particle Size Density	Log
Porosity	Square root

Seagrass tissue nutrients

Residual maximum likelihood (REML analysis) showed that differences in tissue nutrients between species was highly significant ($p<0.001$). However, two of the

species *(Halodule uninervis* and *Zostera capricorni*) were almost confounded with location, therefore nutrient data was analysed separately for each species. Analysing species separately is further supported by the knowledge that all seagrasses do not have the same environmental requirement or responses to environmental conditions as proposed by the "Seagrass Functional Form Model" (Walker *et al.* 1999).

All variables were analysed using Analysis of Variance (ANOVA) with Location as treatments and Sites nested within Location as the blocking structure. Normality of the data was checked using standardised residual plots. Transforming the data had no effect on the residual plots as the plots were heavily influenced by outliers. Analyses were re-run leaving out the outliers to ascertain their influence on the analysis outcome. Outliers were not deleted for analysis but their influence on the analyses is reported. Because of the variable nature of this data results are reported for sites and graphed as mean (x) ± CI(95%)

Reproductive effort

Reproductive effort was calculated as the number of reproductive structures per node (leaf cluster emerging from the rhizome) as each of the three species examined (*Halophila ovalis*, *Halodule uninervis* and *Zostera capricorni)* have different reproductive structures (Figure 6.1). For comparative purposes only the presence of a reproductive structure per node was counted rather than the relative number of flowers, fruits or seeds. The number of nodes counted reflects the number of shoots found in the core. Thus cores with larger numbers of nodes contained more shoots. The average number of reproductive structures per node reflects the per unit area occurrence of reproductive output and this is the reproductive effort (i.e. average number of flowers per core).

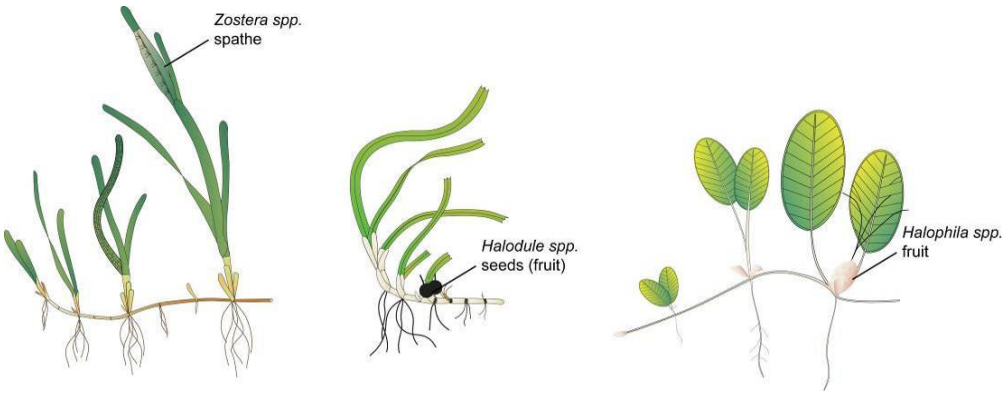


Figure 6.1. Form and size of reproductive structure of the three seagrasses collected – Halophila ovalis, Halodule uninervis and Zostera capricorni.

The production of flowers and fruit were analysed with respect to the sediment nutrient loads and tissue nutrient concentrations. Correlations were performed in SPSS® and presented as Pearson correlation coefficients (p).

Results

Seagrass Presence – Absence

The dominant species of seagrass that were present along the east coast of Queensland within the monitoring sites were *Halophila ovalis, Halodule uninervis* and *Zostera capricorni. Halophila ovalis* occurred ubiquitously, *Halodule uninervis* was found at nine of the 11 locations monitored, while *Zostera capricorni* was collected from five locations (Table 6.3). Although *Zostera* communities are found all along the coast of Queensland, they only dominant southern intertidal meadows and are not common or representative of intertidal meadows in the north (www.seagrasswatch.org). Tissue nutrient and reproductive assessments were restricted to these dominant species.

Table 6.3 Presence (■) of Halophila ovalis, Halodule uninervis and Zostera capricorni in monitoring locations sampled in Reef Plan MMP for plant tissue and reproductive health.
** indicates presence adjacent, but not within, 50m x 50m site.*
+ only found at Picnic Bay

GBR region	NRM Board	Catchment	Seagrass monitoring location	Site name	*H. ovalis*	*H. uninervis*	*Z. caprcorni*
Far Northern	Cape York	Endeavour	Cooktown	Archer Point	■	■	
Northern	Wet Tropics	Daintree	*NA*				
		Russell / Mulgrave Johnstone	Green Island	Green Island	■	■	
			Cairns	Yule Point	■	■	
		Tully	Mission Beach	Lugger Bay	■*	■	
Central	Burdekin	Herbert	*NA*				
		Burdekin	Magnetic island	Picnic & Cockle Bays	■	■	■+
			Townsville	Shelley Beach Bushland Beach	■	■	
	Mackay Whitsunday	Proserpine	Whitsundays	Pioneer Bay	■	■	■
		Pioneer	Mackay	Sarina Inlet	■		■
Southern	Fitzroy	Fitzroy	Shoalwater Bay	Ross Creek Wheelans Hut	■*	■*	■
	Burnett Mary	Burnett	Gladstone	Gladstone Hbr	■	■*	■
		Mary	Hervey Bay	Urangan	■*		■

Seagrass cover and composition in each NRM region

Cape York

The two Cooktown sites (AP1 and AP2) were located on a fringing reef platform in a protected section of bay, fringed by mangroves, approximately 15km south of Cooktown. The sites were dominated by *Halodule uninervis* and *Halophila ovalis* and seagrass cover was between 20% in winter and 35% in spring. Monitoring was established at one site in late 2003, an additional site was established in May 2005. Although sites were only 50m apart, AP2 had slightly more *Cymodocea* and *Thalassia* present. Species composition remained relatively stable over the past 12 months (Figure 6.2). Seagrass cover over the past 12-24 months appeared to follow a seasonal trend with higher abundance in late spring/early summer (Figure 6.6).

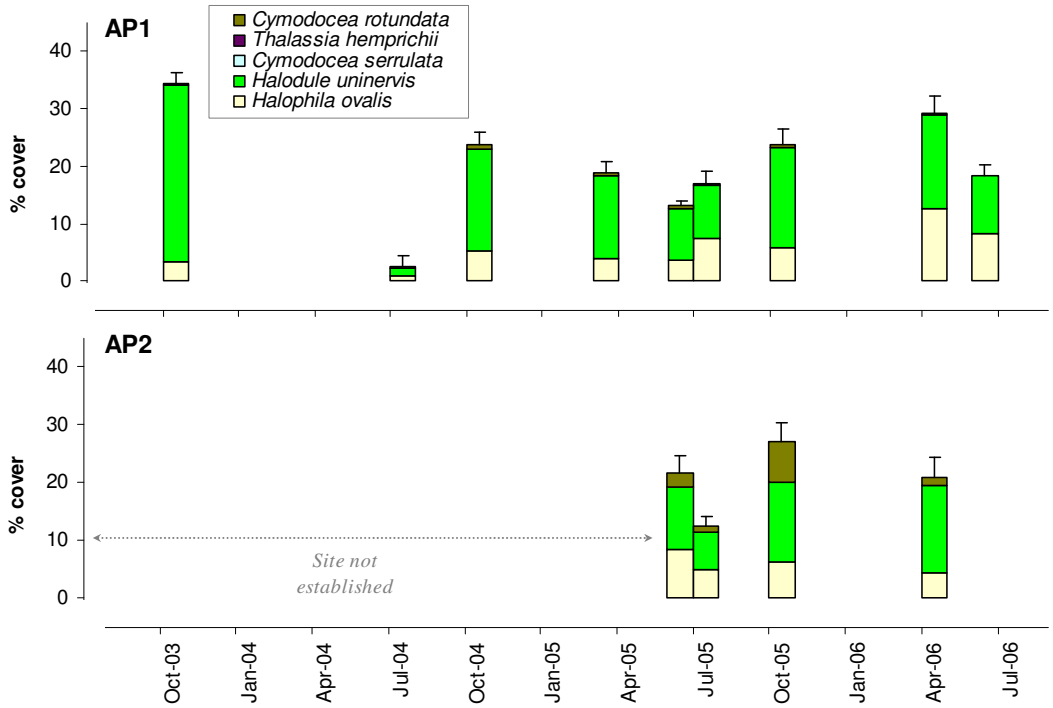


Figure 6.2. Mean percentage cover for each seagrass species at Cooktown Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Wet Tropics

The coastal sites selected for monitoring in the Cairns region were at Yule Point, Green Island and Lugger Bay, Mission Beach. Ellie Point on the north of Cairns Harbour, adjacent to the mouth of the Barron River was not selected for the Reef Plan MMP as access to the site is restricted (via Cairns Airport) and did not comply with the requirements set out by the Head Contract.

The Yule Point sites (monitored since 2000) were representative of inshore seagrass communities in the region, and dominated by *Halodule uninervis* and *Halophila ovalis* (Figure 6.3). *Zostera capricorni* was reported from YP1 in 2002, but was absent during the period of the Reef Plan MMP. Seagrass cover over the past 12-24 months appeared to follow a seasonal trend with higher abundance in summer (Figure

6.6). Associated macroalgae generally increased in spring/early summer, but were not at levels that suggested a decline in water quality. Finer sediments (e.g. fine sand) increased over 2004, but there were no detected changes in the taxonomic composition of the monitored seagrass meadows. Overall, the sites appeared to differ little from 1967 when den Hartog (1970) described the species present and sediment condition.

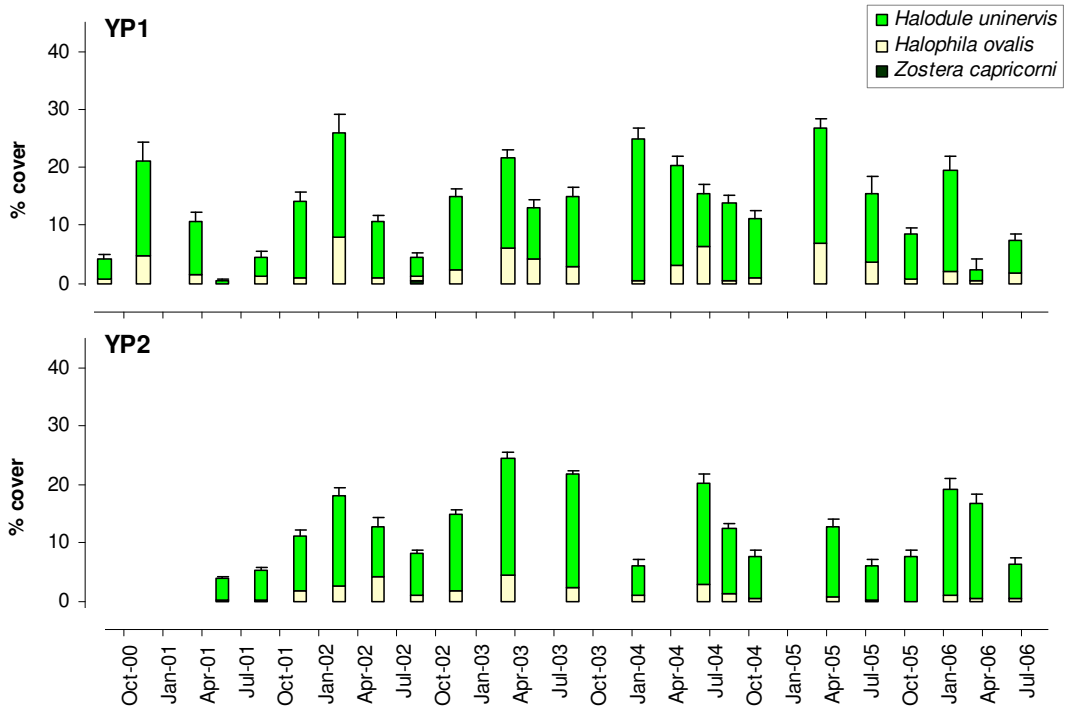


Figure 6.3. Mean percentage cover for each seagrass species at Cairns Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Green Island sites were on a reef-platform on mid shelf reef, approximately 27 km north east of Cairns. One site was established in late 2001, however the standard Seagrass-Watch sampling protocols were not implemented until late 2003 (the original sampling included only one transect with 12 replicate quadrats). An additional site was established in April 2005. The sites are located south west of the cay and dominated by *C. rotundata* and *T. hemprichii* with some *H. uninervis* and *H. ovalis*. The sites appeared to follow a seasonal pattern in abundance, with high cover in the summer and low cover in winter, and no significant changes in species composition were observed (Figures 6.4, 6.6).

In the Mission Beach region the only suitable inter-tidal seagrass meadow that could be located was in Lugger Bay. This meadow is only exposed as very low tides (<0.4m) and composed of *H. uninervis*. This site was also selected for monitoring as some historical data is available on seagrass abundance and sediment and plant nutrients from previous research at this location. The monitoring site was established in May 2005. Seagrass cover was generally low (< 10%) (Figure 6.5), which is similar to observations in the early 90's at this location (Mellors *et al.* 2005). No seasonal trends in seagrass cover were apparent due to the paucity of data (Figure 6.6).

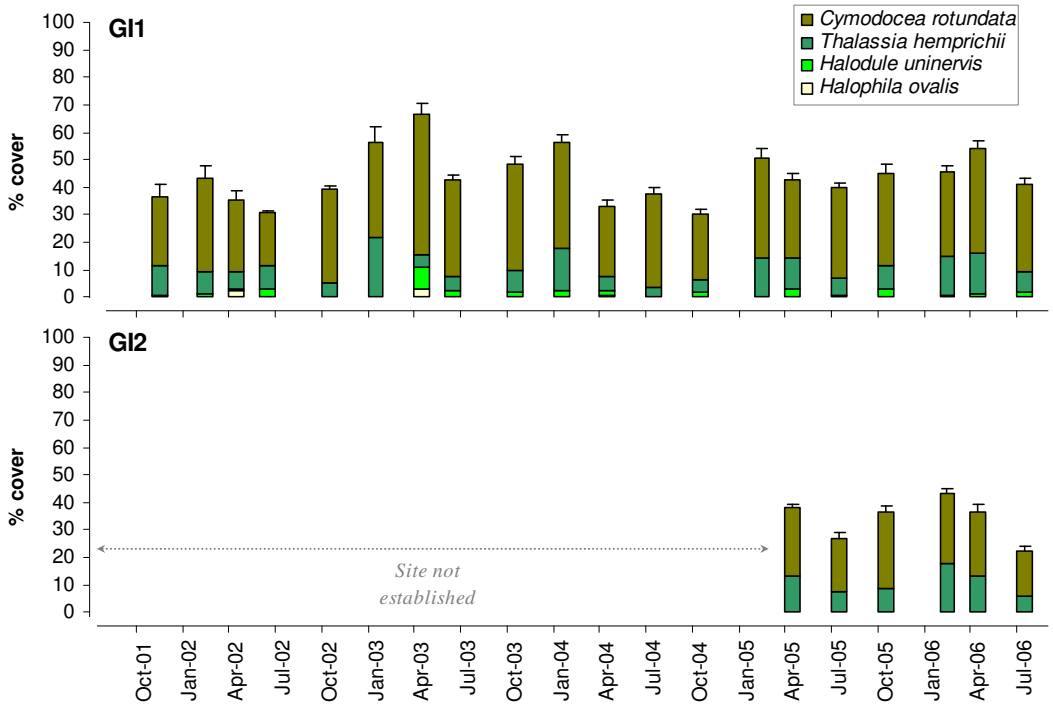


Figure 6.4. Mean percentage cover for each seagrass species at Green Island Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

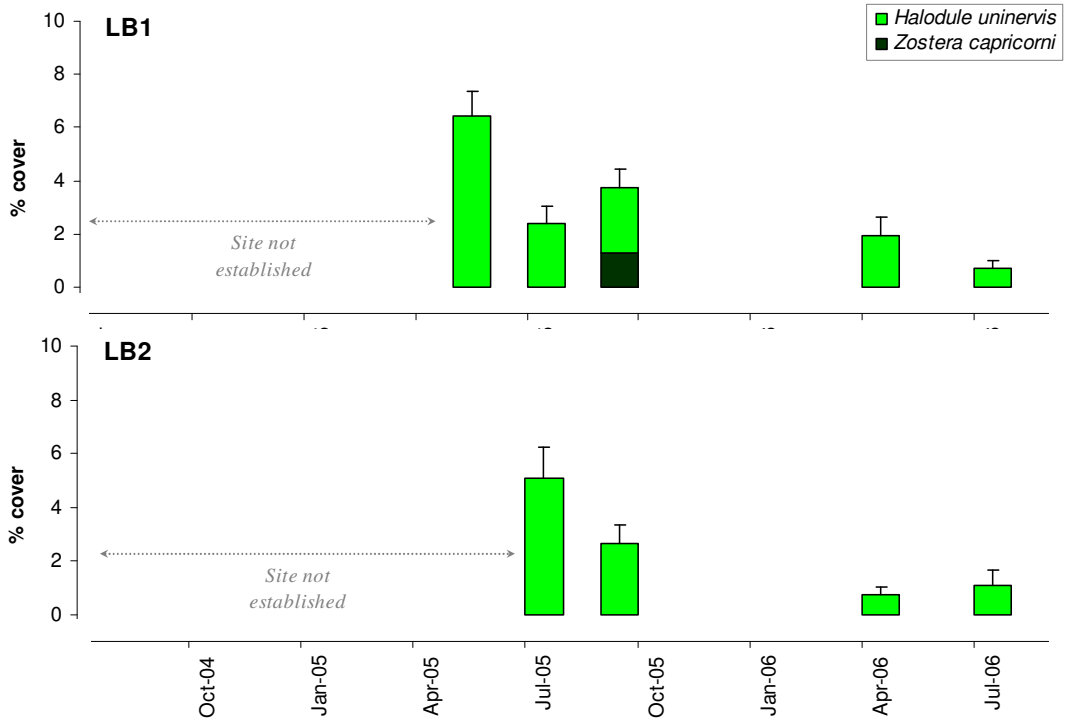


Figure 6.5. Mean percentage cover for each seagrass species at Mission Beach Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Cape York

Wet Tropics

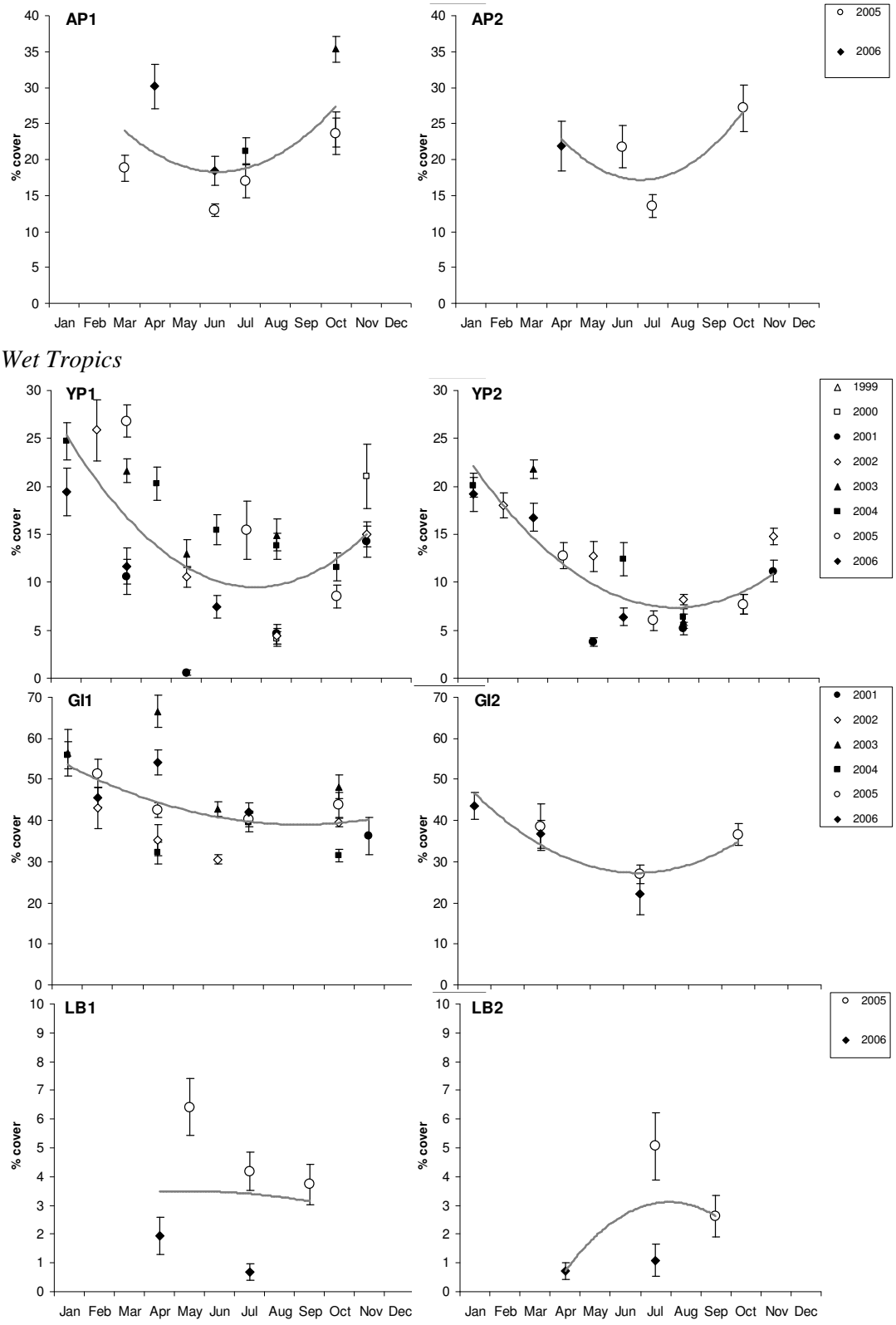


Figure 6.6. Mean percentage seagrass cover (all species pooled) (± Standard Error) for Cape York and Wet Tropics Seagrass-Watch long-term monitoring sites at time of year. NB: Polynomial trendline for all years pooled.

Burdekin

The sites selected for monitoring in Townsville were located on the southern shores of Halifax Bay. The most northern site is at Bushland Beach, while the replicate site is located at Shelley Beach (Cape Pallarenda). Both sites were dominated by *Halodule uninervis* with varying amounts of *Halophila ovalis*. Seagrass cover appears to have increased at Bushland Beach over the past 12-24 months, although cover at Shelley Beach remained relatively similar to past years (Figure 6.7). There were not any detected changes in species composition and both sites showed a seasonal pattern in seagrass cover, high in summer and low in winter (Figure 6.11).

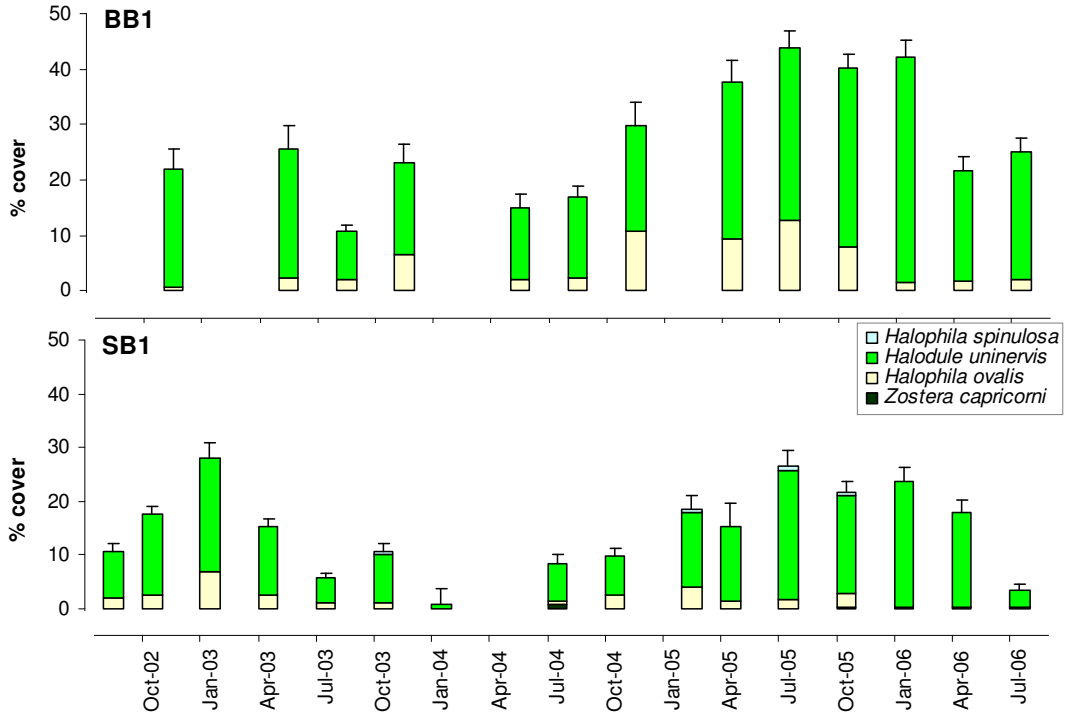


Figure 6.7. Mean percentage cover for each seagrass species at Townsville Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Two new sites were established on Magnetic Island. The site at Picnic Bay (MI1) was dominated by *Halodule uninervis* with *Halophila ovalis*. It was difficult to locate a replicate site in Picnic Bay, so a site was established in the adjacent Cockle Bay. The Cockle Bay site was dominated by *Halophila ovalis* with *Cymodocea serrulata/ Thalassia hemprichii/ Halodule uninervis (*Figure 6.8). A very small patch of *Zostera capricorni* recruited to MI1 in the past 6-12 months. Both sites were located on fringing reef flats. Selection of sites was also facilitated by available historical data on seagrass growth and nutrients from previous research at these locations (Mellors 2003). Due to the paucity of data, it is difficult to describe a seasonal pattern in seagrass cover with sufficient certainty (Figure 6.11).

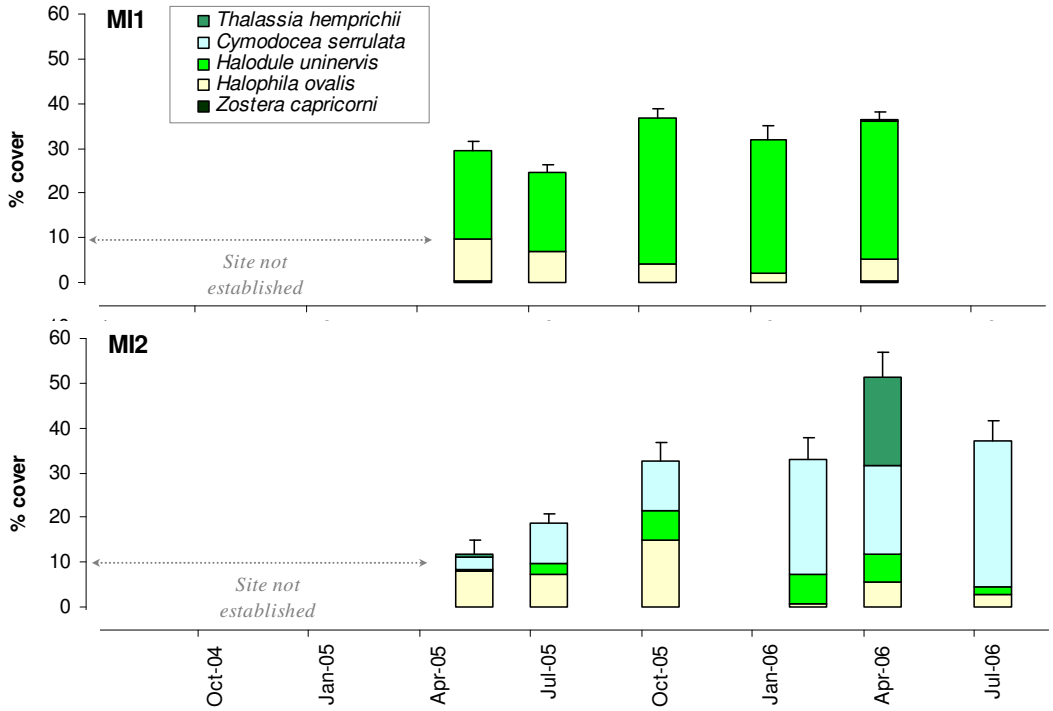


Figure 6.8. Mean percentage cover for each seagrass species at Magnetic Island Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Mackay Whitsunday

Existing sites in Pioneer Bay were selected for monitoring in the Whitsunday region. The sites were located on intertidal sand/mud flats adjacent to Cannonvale in southern Pioneer Bay. The meadows cover approximately 60ha and were dominated by *Halodule uninervis* and *Halophila ovalis* mixed with low amounts of *Zostera capricorni.* Species composition remained stable over the monitoring period and indicated natural seasonal patterns (Figure 6.9, 6.11). 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 these sites with the highest feeding activity (evidenced by trails) recorded in March and September.

New sites were established at Sarina Inlet south of Mackay in 2005 as no easily accessible and significantly sized intertidal seagrass meadows were located nearer to the mouth of the Pioneer River. The sites were located on an intertidal mud/sand bank of the inlet. The meadow was dominated by *Zostera capricorni* with some *Halophila ovalis*. Seagrass cover in April 2006 was significantly lower that that recorded in September/October 2005 but was similar to cover recorded in April 2005 (Figure 6.10). As the dataset for this location is limited to 12 months, it is not possible to determine if this is a natural/seasonal fluctuation in seagrass abundance (Figure 6.11).

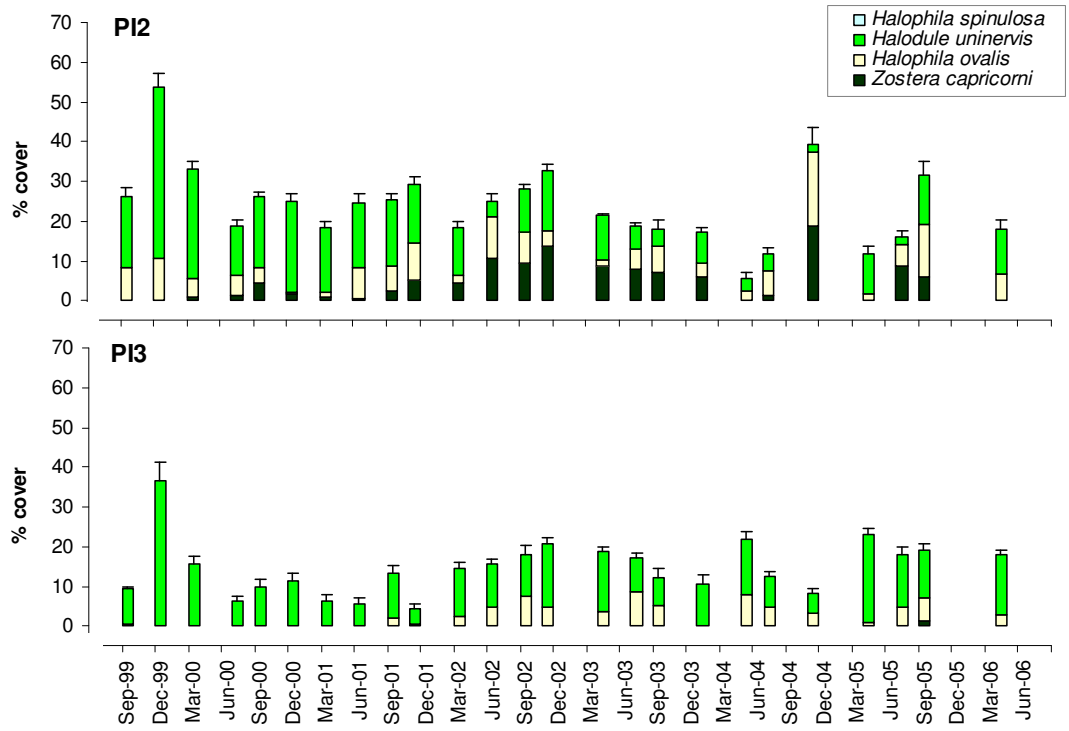


Figure 6.9. Mean percentage cover for each seagrass species at Whitsunday Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

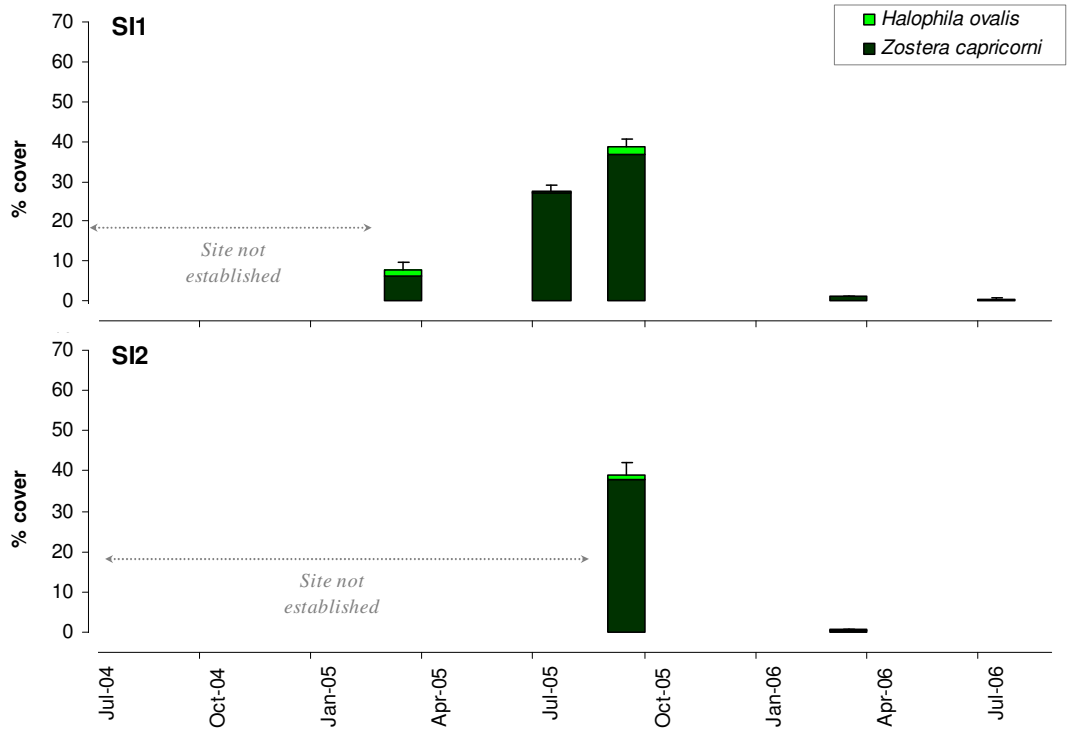


Figure 6.10. Mean percentage cover for each seagrass species at Mackay Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

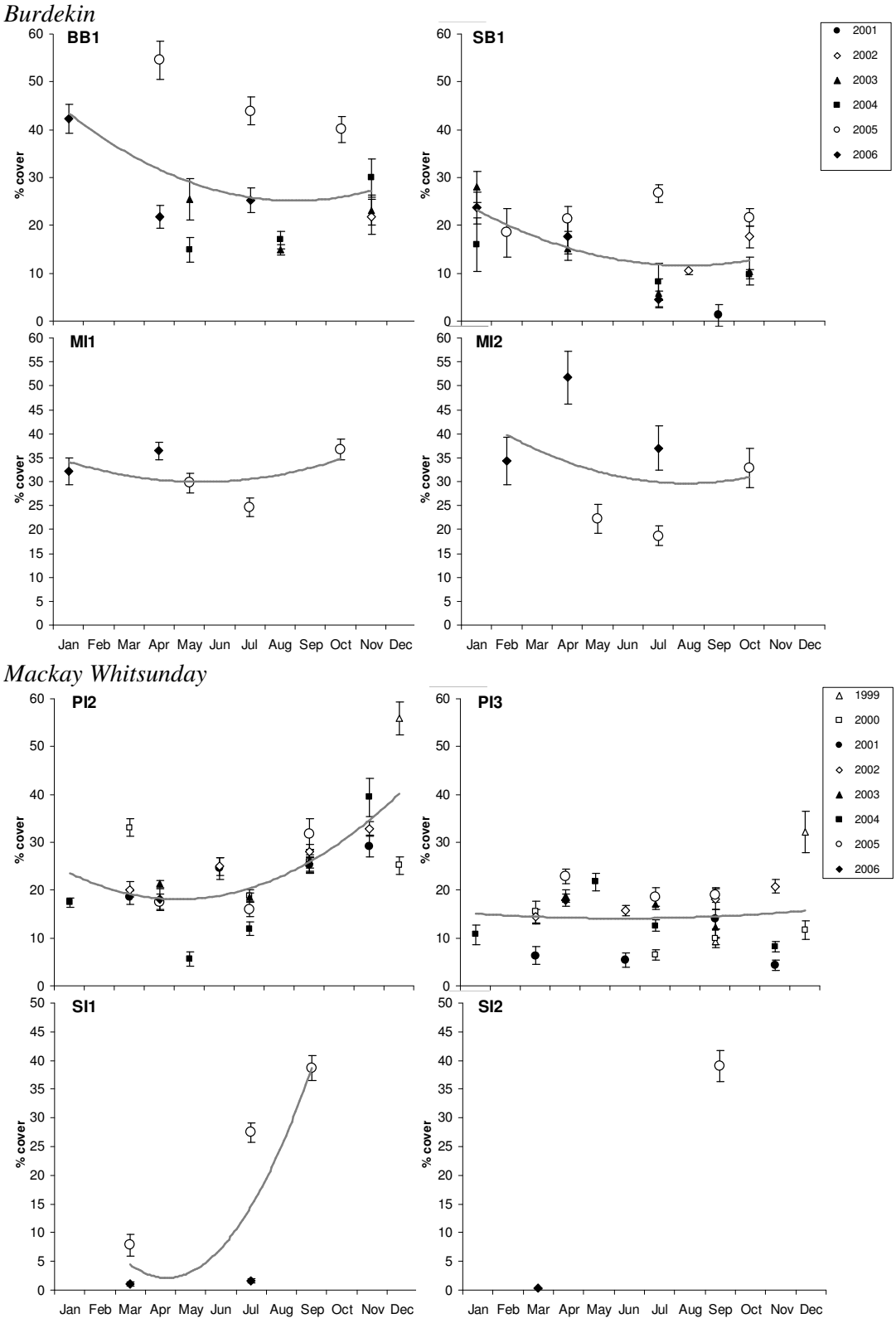


Figure 6.11. Mean percentage seagrass cover (all species pooled) (± Standard Error) for Burdekin and Mackay Whitsunday Seagrass-Watch long-term monitoring sites at time of year. NB: Polynomial trendline for all years pooled.

Fitzroy

In Shoalwater Bay, existing Seagrass-Watch sites were selected for monitoring as no easily accessible and significantly sized seagrass meadows were located nearer to the mouth of the Fitzroy River. The Ross Creek (RC1) and Wheelan's Hut (WH1) sites were north of Sabina Point on the north western shores of Shoalwater Bay. Average seagrass cover ranged from ~15% to 45% (Figure 6.12). Species compositions at each site appear fairly similar between years with *Zostera capricorni* dominating, and no apparent seasonal patterns (Figure 6.15).

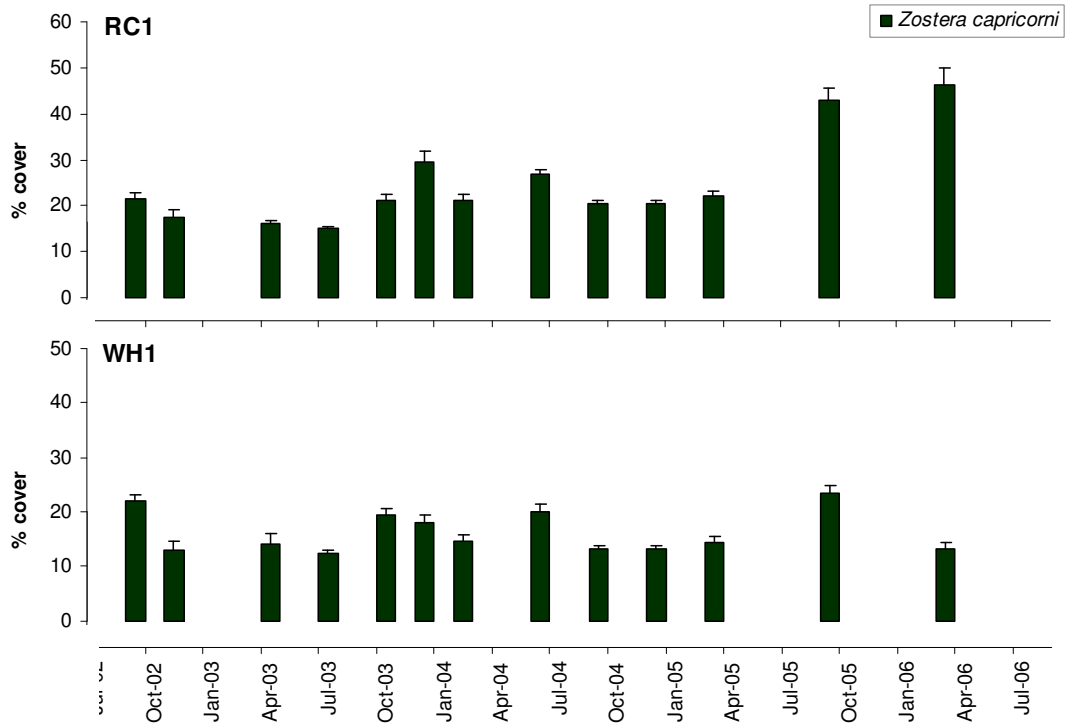


Figure 6.12. Mean percentage cover for each seagrass species at Shoalwater Bay Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: if no sampling conducted then x-axis is clear.

Burnett Mary

Gladstone Harbour sites were located on a large *Zostera capricorni* dominated meadow on the extensive intertidal Pelican Banks south of Curtis Island, as no easily accessible and significantly sized seagrass meadows were located nearer to the mouth of the Burnett River. This meadow is also part of the annual Port of Gladstone environmental monitoring program. A recent (February 2006) reconnaissance survey indicated that seagrass cover has decreased throughout the region. This was confirmed in April 2006 when the monitoring sites GH1 and GH2 were examined (Figure 6.13).

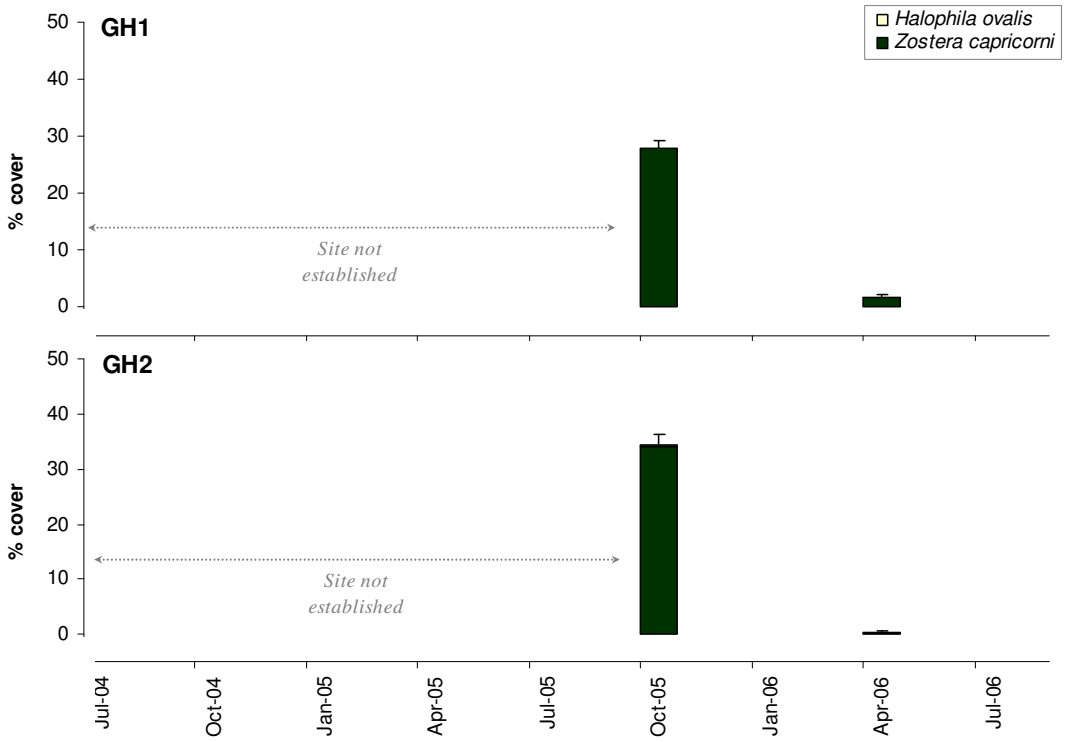


Figure 6.13. Mean percentage cover for each seagrass species at Gladstone Harbour Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: Zero values appear as a dash on the x-axis, if no sampling conducted then x-axis is clear.

Hervey Bay sites were located adjacent to the Urangan marina and close to the Mary River mouth. The sites were dominated by *Zostera capricorni* with minor components of *Halophila ovalis* and some *Halodule uninervis*. Following a major flood in August 1999, seagrass was absent (0% cover) until May 2000. In July 2000 seedlings of *Zostera capricorni* appeared. Since then cover has increased significantly. A decline was recorded in early 2004, however cover increased in late 2004 (Figure 6.14). A similar decline occurred in April 2006, and may be part of a seasonal pattern developing at the location (Figure 6.15).

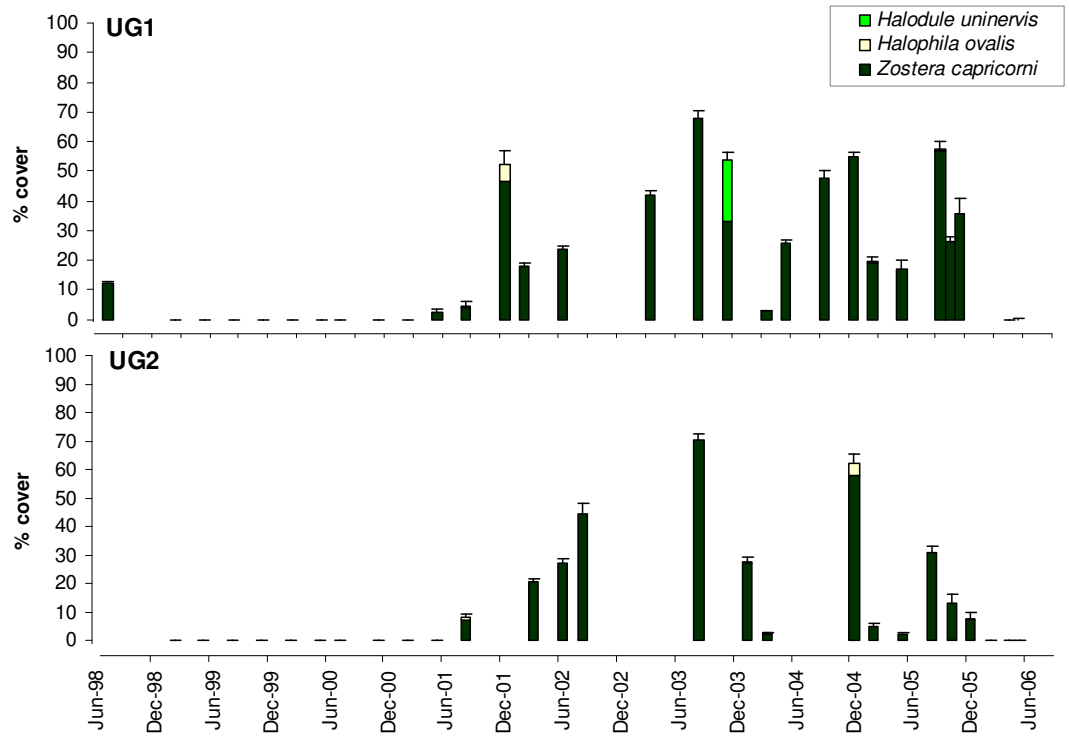
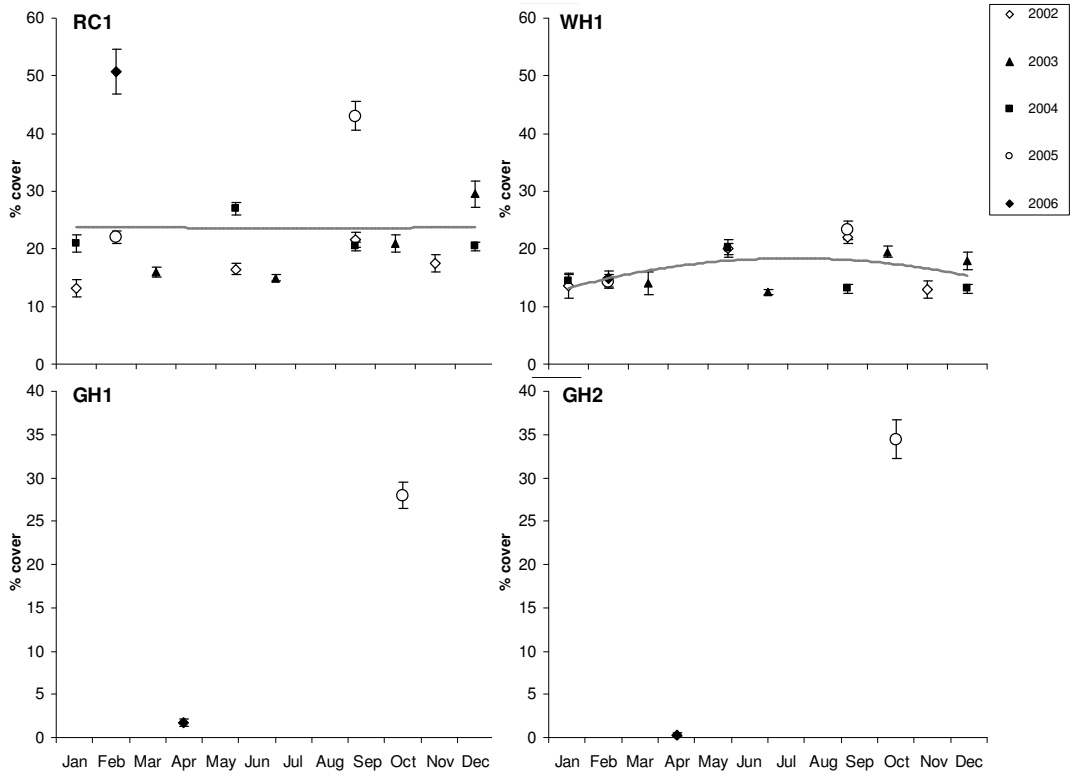


Figure 6.14. Mean percentage cover for each seagrass species at Hervey Bay Seagrass-Watch long-term monitoring sites (+ Standard Error). NB: Zero values appear as a dash on the x-axis, if no sampling conducted then x-axis is clear.

Fitzroy



Burnett Mary

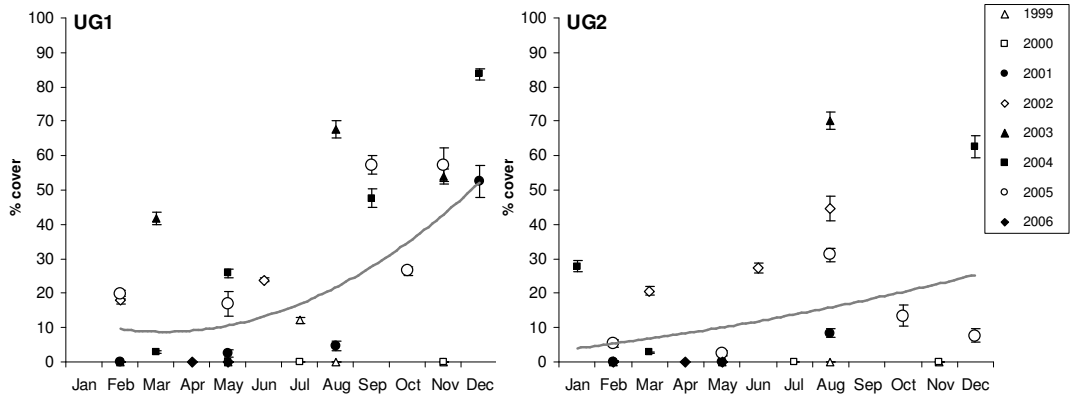


Figure 6.15. Mean percentage seagrass cover (all species pooled) (± Standard Error) for Fitzroy and Burnett Mary Seagrass-Watch long-term monitoring sites at time of year. NB: Polynomial trendline for all years pooled.

Edge mapping

Edge mapping was conducted within a 100m radius of all Seagrass-Watch monitoring sites in September/October 2005 and March/April 2006 (see Table 6.4, Appendix 6.1). Distribution of seagrass within the 100m radius of monitoring sites was significantly lower in April 2006 than in October 2005 (all sites pooled, Two sample T-Test, T=-2.35, d.f.=42, p=0.0236). Large losses at individual sites influenced this outcome. In most cases the difference from October 2005 to April 2006 fit the typical pattern of seagrass distribution in tropical northern Australian seagrasses with maximum distribution usually occurring in spring/summer and minima in winter (McKenzie *et al.* 1998).

There were no detectable differences in the edge mapping data of the seagrass meadows at Green Island, Bushland Beach and Shoalwater Bay between 2005 and 2006 (Appendix 6.1, Maps 3, 5 and 9). However, the area of seagrass declined more than 20% within the mapping boundaries in seven of the remaining 16 sites and seagrass was absent from four sites in April 2006 (Table 6.4).

Some meadows changed species or edges within the mapping area, but were outside the 50m x 50m monitoring sites. Other meadows, however, changed significantly, resulting in loss of seagrass within the 50m x 50m monitoring sites in April 2006 (Table 6.4). For example, at Magnetic Island and Pioneer Bay, seagrass within the monitoring sites remained similar; however the edges of the meadows and the presence of sparse *Halophila ovalis* differed slightly in April 2006 compared to October 2005 (Appendix 6.1, Maps 6 and 7).

At Archer Point (Cooktown), the edge of the main meadow decreased seaward into AP2 monitoring site, and resulted in loss of seagrass in one of the three transects monitored (Appendix 6.1, Map 1). Site AP1 however, remained within the main meadow and there were no noticeable edge effects.

At Yule Point, although the overall distribution of seagrass increased within the mapping area, a drainage channel bisected YP1 in March 2006 (Appendix 6.1, Map 2). This only appears to have had limited impact on the seagrass measures, as only 2 quadrats fell within the channel. Seagrass distribution increased at YP2.

At Lugger Bay, the distribution of the seagrass meadow changed, causing the edge of the meadow to fall within the main monitoring site (Appendix 6.1, Map 4). The meadow is relatively narrow (approximately 60-70m wide) and distributed along the seaward edge of a mobile sand bank. In April the landward edge of the meadow was impacted by sand movement and associated drainage channels. This moved the edge of the meadow to the boundaries of both of the monitoring sites.

Similarly, at Sarina Inlet, the meadow edges changed significantly between September 2005 and March 2006. At both sites, the landward and seaward edges of the meadows encroached into the monitoring sites, significantly reducing the presence of seagrass in one transect in SI1 and two transects in SI2

Seagrass cover was extensive in September/October 2005, however seagrass was absent from sites in Gladstone and Urangan in April 2006 (Appendix 6.1, Map 10 and 11).

Table 6.4 Area (ha) of seagrass meadow being monitored within 100m radius of site. Shading indicates decrease in meadow area

Sector	Region	Catchment	Monitoring location		October 2005	April 2006	% Difference	Change
Far Northern	Cape York	Endeavour	Cooktown	AP1	3.667	3.330	-9.2	Decrease seaward
				AP2	3.710	3.139	-15.4	Decrease seaward
Northern	Wet Tropics	Daintree	*NA*					
		Russell / Mulgrave Johnstone	Green Island	GI1	5.257	5.319	1.2	Increase landward
				GI2	4.632	4.647	0.3	negligible
			Cairns	YP1	1.326	1.789	34.9	Increase landward
				YP2	3.596	4.120	14.6	Increase landward
		Tully	Mission Beach	LB1	1.675	1.085	-35.2	Decrease landward
				LB2	1.801	1.448	-19.6	Decrease landward
Central	Burdekin	Herbert	*NA*					
		Burdekin	Magnetic Island	MI1	2.933	3.398	15.9	Increase landward
				MI2	4.104	4.342	5.8	Increase landward
			Townsville	SB1	4.303	3.485	-19.0	Decrease seaward
				BB1	5.312	5.312	0	No change
	Mackay Whitsunday	Proserpine	Whitsundays	PI2	3.432	3.534	3.0	Increase landward
				PI3	2.432	2.026	-16.7	Decrease landward
		Pioneer	Mackay	SI1	3.374	1.726	-48.8	Decrease seaward
				SI2	3.747	2.460	-34.3	Decrease landward
Southern	Fitzroy	Fitzroy	Shoalwater Bay	RC	5.380	5.380	0	No change
				WH	5.392	5.392	0	No change
	Burnett Mary	Burnett	Gladstone	GH1	5.394	0	-100	Meadow absent
				GH2	5.174	0	-100	Meadow absent
		Mary	Hervey Bay	UG1	5.266	0	-100	Meadow absent
				UG2	5.326	0	-100	Meadow absent

Sediment herbicide

Of the three herbicides analysed only Diuron was found above detectable limits. It was detected at four of 15 sites in 2005 and 15 of 22 sites in 2006. Sites above detectable limits were at the following locations: Green Island, Mission Beach, Magnetic Island, Townsville, Pioneer Bay, Sarina Inlet, Gladstone and Hervey Bay (Figure 6.16, Appendix 6.2). The highest concentration found was 0.42 ±0.13 µg/kg DW at Mission Beach, adjacent to the Tully River and Murray River sugar cane districts. The April 2005 results for Irgarol need to be treated with caution as the analytical method for this analysis had not been validated at the time of analysis.

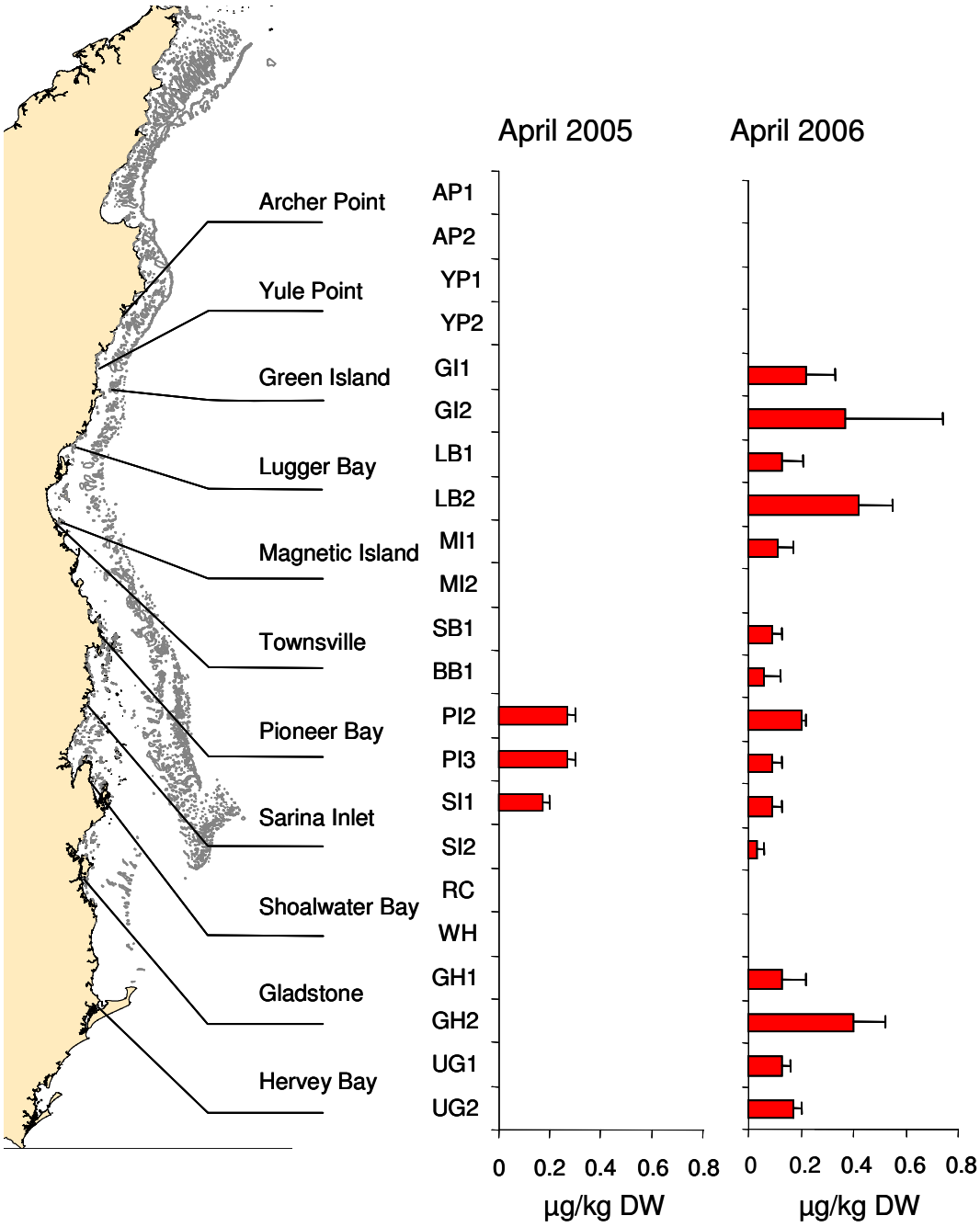


Figure 6.16. Concentration (µg/kg DW) of Diuron in sediments for each long-term monitoring site in April 2005 and April 2006 (+ SE).

Sediment physical characteristics

Both particle size density and porosity were measured to characterise the sediments of the sampled locations. Measure of these parameters gives an approximation of the mineralogy of the sediments, which in turn can give an indication of nutrient behaviour at these locations. Measurement of particle size density is required for the calculation of porosity.

The particle size density (PSD) is a measure of the mass of a sediment sample in a given volume of particles. The density of the particles is a result of the chemical composition and structure of the minerals in the sediment. Hence measurement of PSD can be deduced by comparing the sample's PSD with the known densities of other minerals.

Porosity is the measure of the volume of void space (pores occupied by air and water) of the sediment and is dependent on particle size (Folk 1974; Friedman & Sanders 1978). Typically porosity decreases as particle size increases. This is true for sediments that are well sorted.

Particle size density ranged from 1.99 (Urangan) to 2.70 (Green Island) with highest variability at Green Island (Figure 6.17). This indicates that sediments at all sites were quite sandy, with a variation in the quantity of finer (Urangan) to coarser sediment (Green Island) also being present in the seagrass plant rhizosphere. Whilst the range of PSD between sites did not appear great, there were significant differences between Locations (ANOVA d.f.= 10, 11 $p < 0.001$) with five distinct groups (Table 6.5). Both Urangan and Green Island were significantly different from each other, although Urangan showed similarities with Pioneer Bay. Green Island had similar sediments to Gladstone, Townsville, Yule Point and Lugger Bay.

Porosity measures ranged from a low of 0.44 (Urangan, Lugger Bay) to a high 0.52 (Green Island) (Figure 6.17). This range suggests that the sediment at these sites are quite mixed and that the entire sample is a mixture of coarse sand, sand, fine sand, silt and clay. Green Island had the coarsest sediment but also very high clay content and other minerals that have large surface areas. There were significant differences between locations (ANOVA d.f = 10, 11, $p = 0.003$). Green Island, Pioneer Bay, Sarina Inlet, Shoalwater Bay and Gladstone Harbour had greater proportions of silts and clays than Yule Point, Magnetic Island, Lugger Bay and Urangan (Table 6.5).

Table 6.5. Post priori groupings (a, b, c, d, e) for seagrass monitoring Location according to sediment characteristic and nutrient variable (PSD= particle size density). Locations with the same letter are not significantly different at the p=0.05 level.

PSD	PSD rank	Porosity	Porosity rank	PO_4^{3-}	PO_4^{3-} rank	NH_4^+	NH_4^+ rank
Green Island	a	Green Island.	a	Green Island.	a	Urangan	a
Gladstone	ab	Pioneer Bay	ab	Pioneer Bay	a	Green Island	b
Townsville	ab	Sarina Inlet	ab	Magnetic Island	b	Magnetic Island	b
Yule Pt	ab	Shoalwater	ab	Shoalwater	bc	Townsville	bc
Lugger Bay	ab	Gladstone	ab	Urangan	bc	Pioneer Bay	bc
Shoalwater	b	Townsville	bc	Townsville	c	Shoalwater	bc
Sarina Inlet	bc	Archer Pt	cd	Sarina Inlet	cd	Gladstone	bc
Magnetic Island.	bc	Yule Pt	cd	Gladstone	de	Sarina Inlet	bcd
Archer Pt	cd	Magnetic Is.	cd	Yule Pt	ef	Archer Pt	cd
Pioneer Bay	de	Lugger Bay	d	Lugger Bay	f	Yule Pt	d
Urangan	e	Urangan	d	Archer Pt	g	Lugger Bay	d

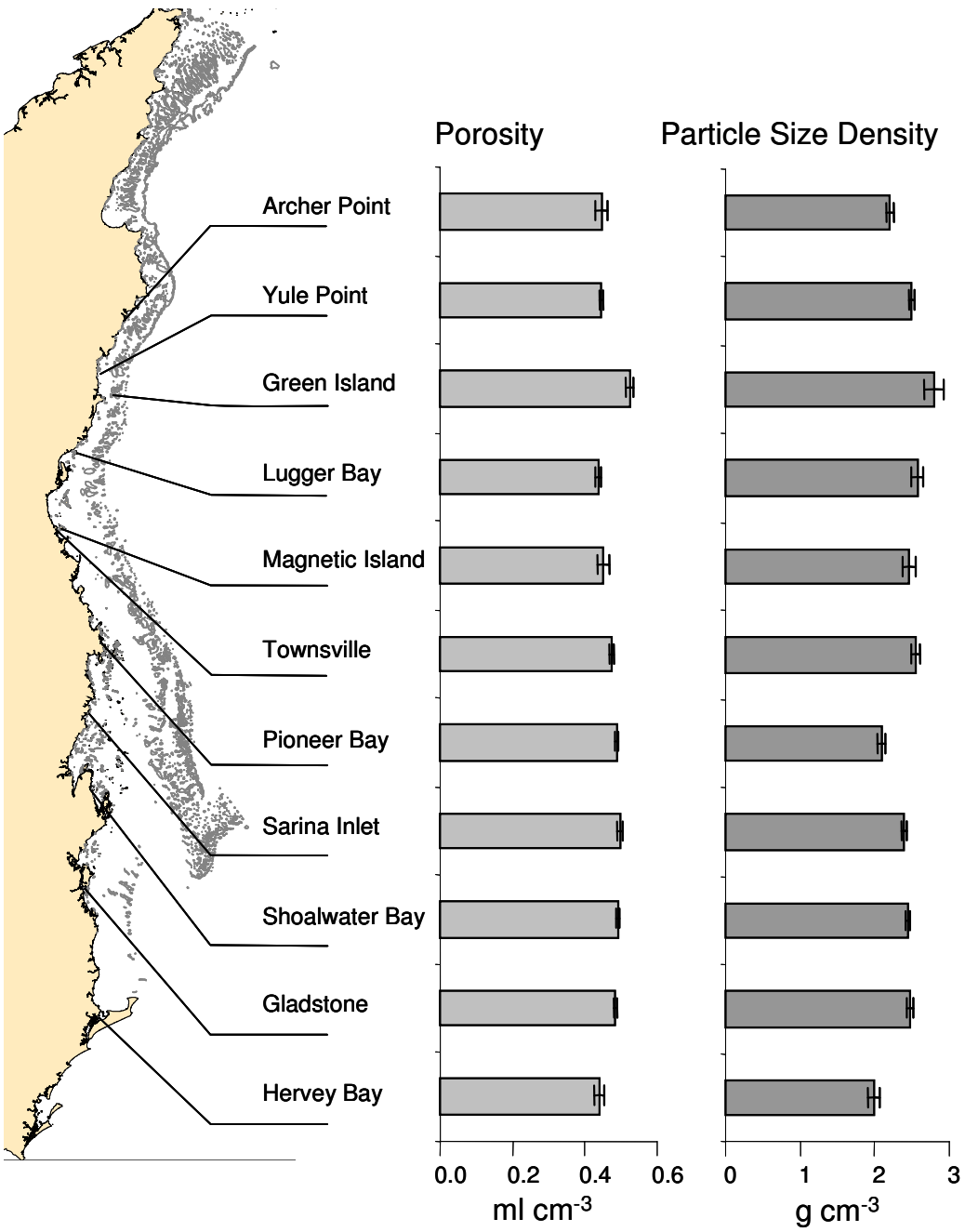


Figure 6.17. Sediment porosity (ml pore water per cm^3 wet sediment) and particle size density (mass of wet sediment/volume of wet sediment) for each long-term monitoring location (+ 95% CI).

Sediment nutrients

Units of nutrient concentrations are reported differently for nitrogen oxides than that for ammonium and phosphate, as a large number of nitrogen oxide measures were below anlyatical detection. To have reported this in µmols L^{-1} would have required a extensive explanation of varying levels of undetectable nitrogen oxides at each location as porosity (a multiplier in the calculation of µmols L^{-1}) differs between locations

Nitrogen oxides (NO_2^- & NO_3^-)

Of the 110 samples analysed for nitrogen oxides adsorbed to sediments, only 53 samples had detectable amounts (≥0.1mg/kg) present. Detectable limits ranged from 0.1 mg/kg to 0.3 mg/kg. Locations that had a concentration of 0.1 mg/kg nitrate and nitrite in more than one sample were: Townville, Magnetic Island, Sarina Inlet, Shoalwater Bay, Gladstone Harbour and Hervey Bay. The Townsville region recorded the highest nitrogen oxide levels along the coast: Cockle Bay (Magnetic Island) (0.2 mg/kg) and Shelley beach (0.2 – 0.3 mg/kg). In view of the paucity of data points no statistical analyses could be performed.

Ammonium (NH_4^+)

Levels of ammonium ranged from 802.78 µmols L_{sed}^{-1} (Hervey Bay) to 107.48 µmols L_{sed}^{-1} (Yule Point) (Figure 6.18). Differences between locations were significant (ANOVA d.f.=10,11, p = 0.011). Post priori testing of Least Significant Differences detected several groupings of Location on the basis of their levels of ammonium. Hervey Bay had significantly higher levels of NH_4^+ than any other location (Figure 6.18, Table 6.5). Magnetic Island and Green Island also recorded high levels of NH_4^+ however these levels overlapped with locations recording intermediate levels (Table 6.5, Figure 6.18). Intermediate levels of NH_4^+ were recorded from Shoalwater Bay, Pigeon Island, Gladstone and Townsville. Lugger Bay and Yule Point recorded the lowest levels of NH_4^+ but there was overlap with Archer Point and Sarina Inlet (Table 6.5). All four of these locations recorded levels lower than 200 µmols L_{sed}^{-1}.

Phosphate (PO_4^{3-})

Levels of PO_4^{3-} ranged from 808.13 µmols L_{sed}^{-1} (Green Island) to 156.23 µmols L_{sed}^{-1} (Archer Point) (Figure 6.18). Differences between locations were significant (ANOVA d.f.=10,11, p <0.001). Several groupings of locations were recognised from *post priori* testing of least significant differences. Green Island and Pioneer Bay grouped together due to their high levels of PO_4^{3-} . Carbonate sediments (Green Island) and those with fine clays (Pigeon Island) are well recognized as sequestering phosphate, making it less available for seagrass uptake. Archer Point had significantly lower levels of phosphate than any other locations. Lugger Bay was also significantly lower than other locations but significantly higher than Archer Point. The other locations were intermediate in levels,

In terms of the relative nutrient pools of NH_4^+ and PO_4^{3-} for this region , the majority of locations show N:P ratios <1, indicating that the PO_4^3 pool is larger than the NH_4^+ pool (Figure 6.18). Analysis of variance detected significant differences between locations (d.f.=10,11, p= 0.011) in N:P with Hervey Bay having significantly larger N:P than Pioneer Bay. Interpretation of the specific site values will be best done in a comparative way over time. At present the high of 800 µmols L_{sed}^{-1} at Hervey Bay resulting in an N:P of 1.7, represents a site of concern as it is so much higher than any

of the other sites sampled. An assessment of the health of this meadow will be forthcoming with continued monitoring.

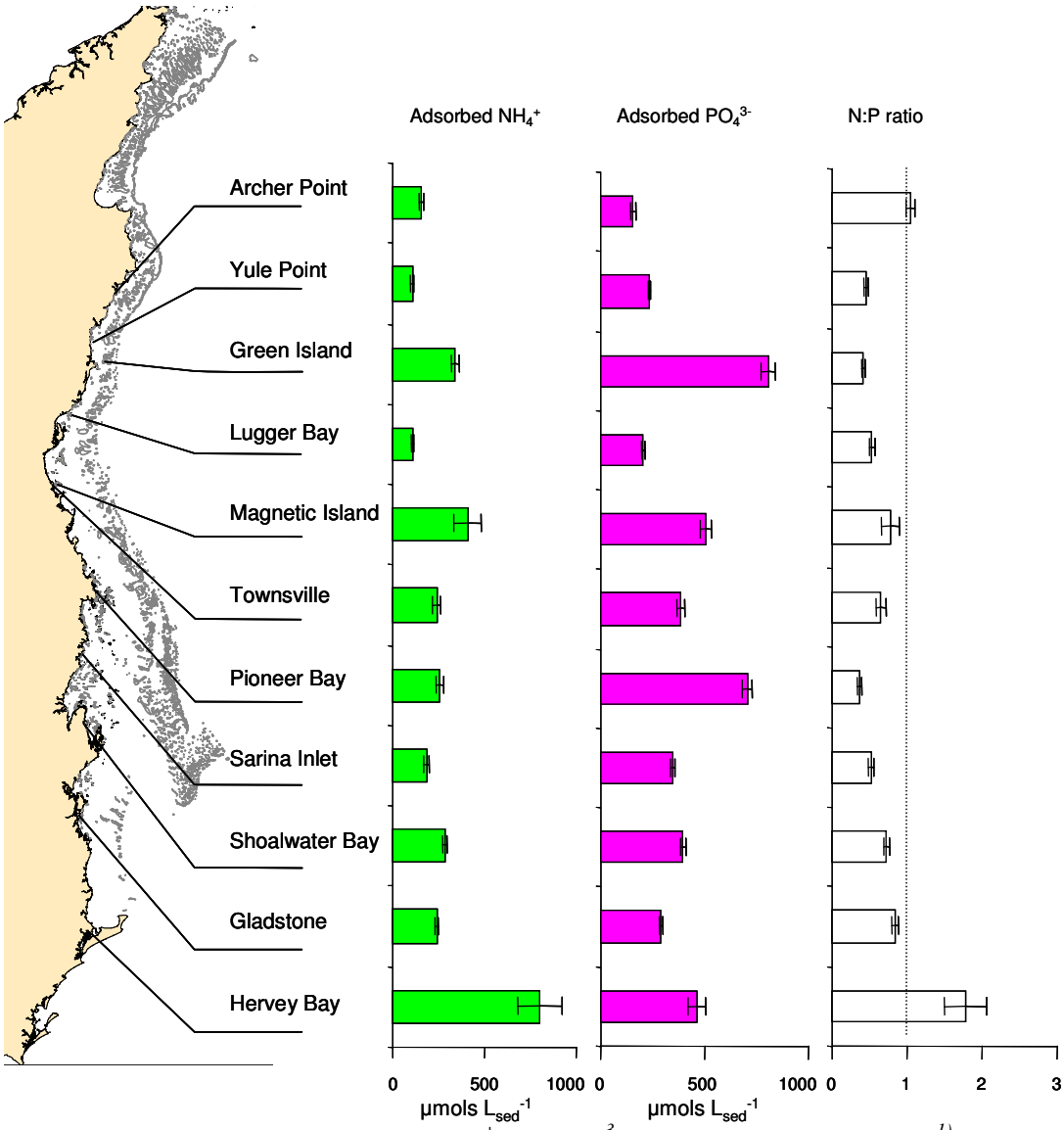


Figure 6.15. Sediment adsorbed NH_4^+ and PO_4^{3-} concentrations (µmol $L_{sed}^{-1)}$ for each long-term monitoring site, and ratio of pools (±95% CI).

Seagrass tissue nutrients

Due to contamination that occurred during the grinding phase of the seagrass processing, some tissue samples could not be analysed for %C, %N and %P. Samples that were not affected have been reported on. The ratio of these nutrients for each sample however, still had integrity and could be statistically analysed with confidence.

Comparing deviations in the ratios of carbon, nitrogen and phosphorus (C:N:P) retained within the plant tissue has been used extensively as an alternative means of evaluating the nutrient status of coastal waters (Duarte 1990, Johnson *et al.* 2006).

C: P ratio

Although the C: P ratios for *Halophila ovalis* leaf tissue were not significantly different between Locations (ANOVA p = 0.159), the data was extremely variable between samples within Locations, as well as between Locations. C:P ratios of *Halophila ovalis* were highly variable at Yule Point YP1, Gladstone Harbour GH1 and also between sites within these locations (Figure 6.19). Re-analysis without these anomalous data points did not change the non-significant outcome.

The C:P ratios for *Halodule uninervis* were significantly higher at Shoalwater Bay and Archer Point than Pioneer Bay, Townsville, Yule Point and Lugger Bay (ANOVA p=0.012). Ratios from Gladstone, Green Island and Magnetic Island overlapped with both of these groups (Table 6.6, Figure 6.19)

Zostera capricorni leaf tissue C:P ratios were significantly lower (mean=373:1) at Pioneer Bay sites than the other four sites where *Zostera capricorni* was present (Table 6.6, Figure 6.19), even when two large outliers were included or removed from the analysis(ANOVA p=0.041 and p=0.046, respectively).

N: P ratio

Leaf tissue N:P ratios for *Halophila ovalis* were highly variable and no significant differences between Locations could be detected (ANOVA p=0.103). The highest variability in leaf tissue N:P ratios occurred at Gladstone Harbour and Green Island (Figure 6.20). The majority of sites had *Halophila ovalis* leaf tissue N:P ratios between 7:1 and 15:1, indicting that the P content of the leaf is relatively large to the N content.

No significant differences in leaf tissue N:P ratios could be detected between Locations for *Halodule uninervis* (ANOVA p= 0.586). Although three outliers were detected, re-analysis did not alter the non- significant outcome when these points were removed.

Leaf tissue N:P ratios for *Zostera capricorni* were significantly higher at Urangan (mean=26:1) than Sarina Inlet, Pioneer Bay and Shoalwater Bay. Leaf tissue N:P ratios from Shoalwater Bay was significantly lower than Sarina Inlet and Gladstone (Table 6.6, Figure 6.20).

C: N ratio

Halophila ovalis leaf tissue C:N ratios were significantly higher at Yule Point than at any other location (ANOVA p<0.001) (Figure 6.21, Table 6.6). However, an aberrant data point was overly influencing the ANOVA outcome, as re-analysis without this outlier was non-significant (ANOVA p = 0.079).

Halodule uninervis leaf tissue C:N ratios were significantly higher at Shoalwater Bay than Archer Point, Magnetic Island, Pigeon Island , Townsville, Lugger Bay and Yule Point (ANOVA p= 0.015) (Figure 6.21, Table 6.6). Initial analysis of *Halodule uninervis* leaf tissue C: N ratios were not-significantly different between Locations

(ANOVA p= 0.298), as an outlier (a data point double that of any other Location and four times greater than any other C:N ratio within the Location) was influencing the data and was subsequently removed. Gladstone Harbour sites recorded values that included the range of values along the coast.

Zostera capricorni leaf tissue C: N ratios were not significantly different between locations (ANOVA p= 0.090). While outliers were identified they did not influence the ANOVA outcome after re-analysis.

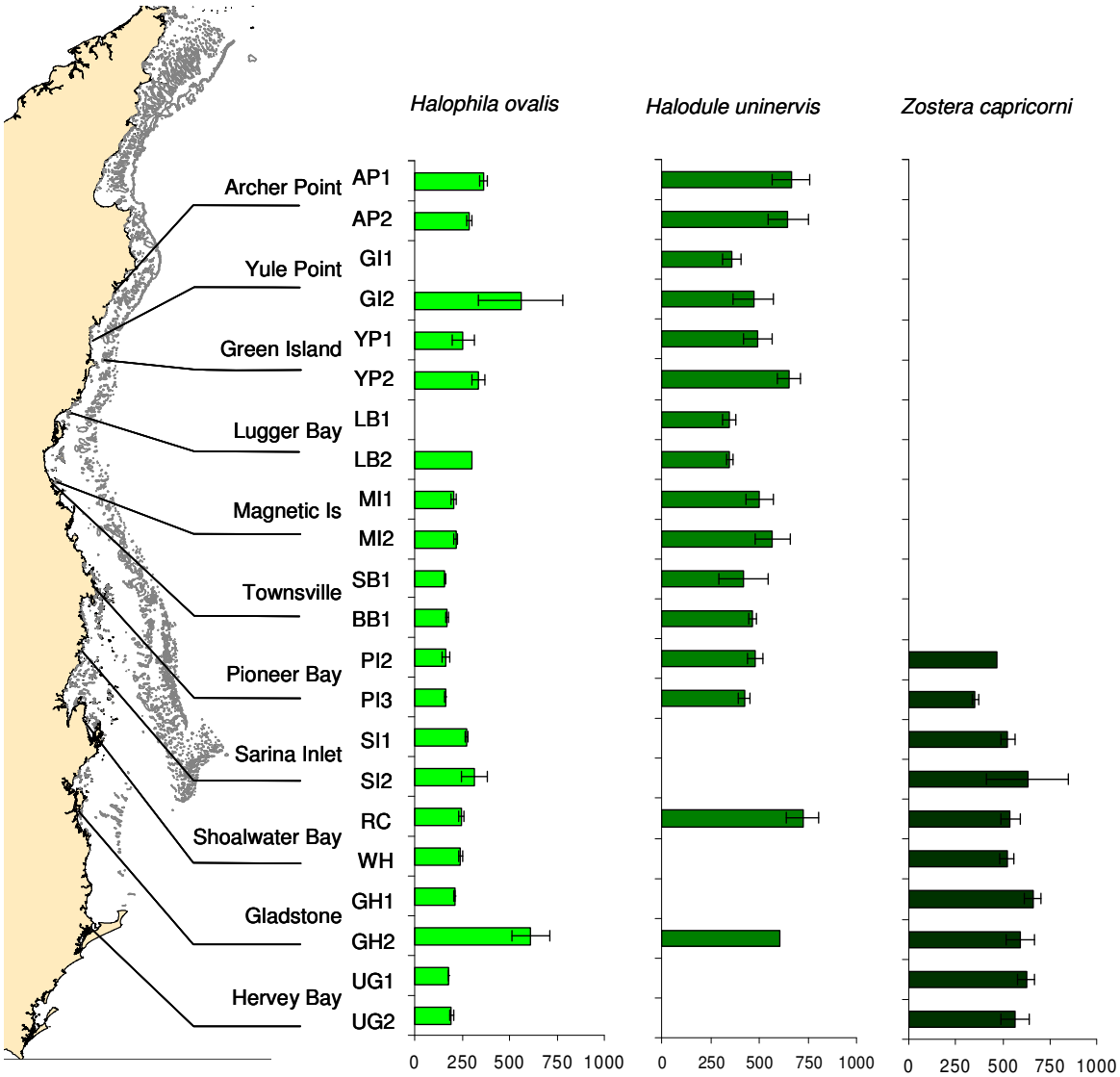


Figure 6.19. Atomic ratio of leaf tissue C: P (±95% CI).

Table 6.6. Locations grouped according to least significant differences for each tissue variable for each species.

C:P	C:P rank*	N:P	N:P rank	C:N	C:N rank
Halophila ovalis					
				Yule Point	a
				Shoalwater Bay	b
				Sarina Inlet	b
				Gladstone	bc
				Hervey Bay	bcd
				Lugger Bay	bcd
				Green Island	bcd
				Pioneer Bay	cd
				Magnetic Island	d
				Archer Point	d
				Townsville	d
Halodule uninervis					
Shoalwater Bay	a			Shoalwater Bay	a
Archer Point	a			Gladstone	abcde
Gladstone	abcd			Green Island	ab
Green Island	ab			Archer Point	bc
Magnetic Island	abc			Magnetic Island	bcd
Pioneer Bay	bcd			Pioneer Bay	cde
Townsville	bcd			Townsville	de
Yule Point	cd			Lugger Bay	e
Lugger Bay	d			Yule Point	e
Zostera capricorni					
		Hervey Bay	a		
		Gladstone	ab		
		Sarina Inlet	bc		
		Pioneer Bay	cd		
		Shoalwater Bay	d		

*Ranking: Locations with same letter are not significantly different at the p = 0.05 level

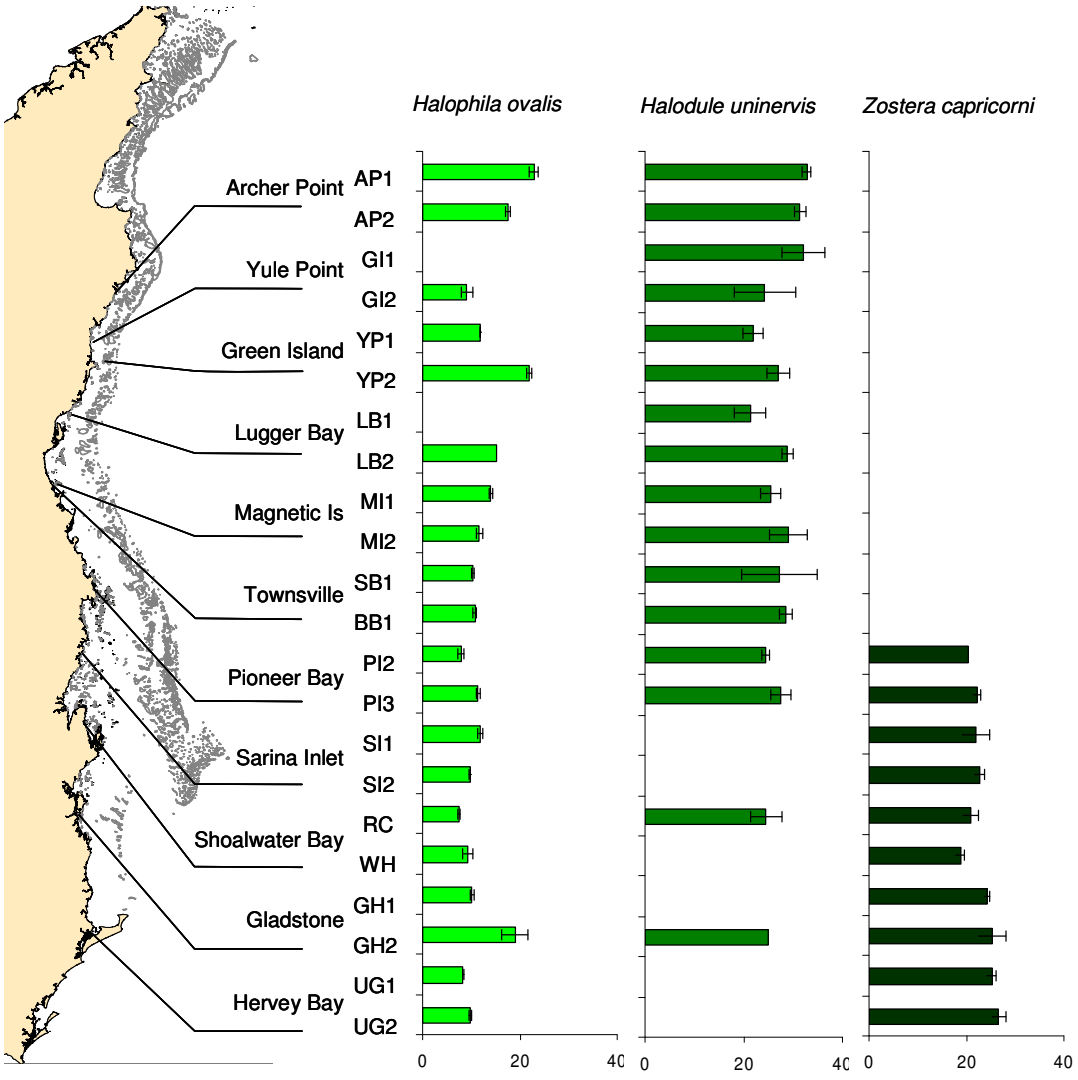


Figure 6.20. Atomic ratio of leaf tissue N: P (±95% CI).

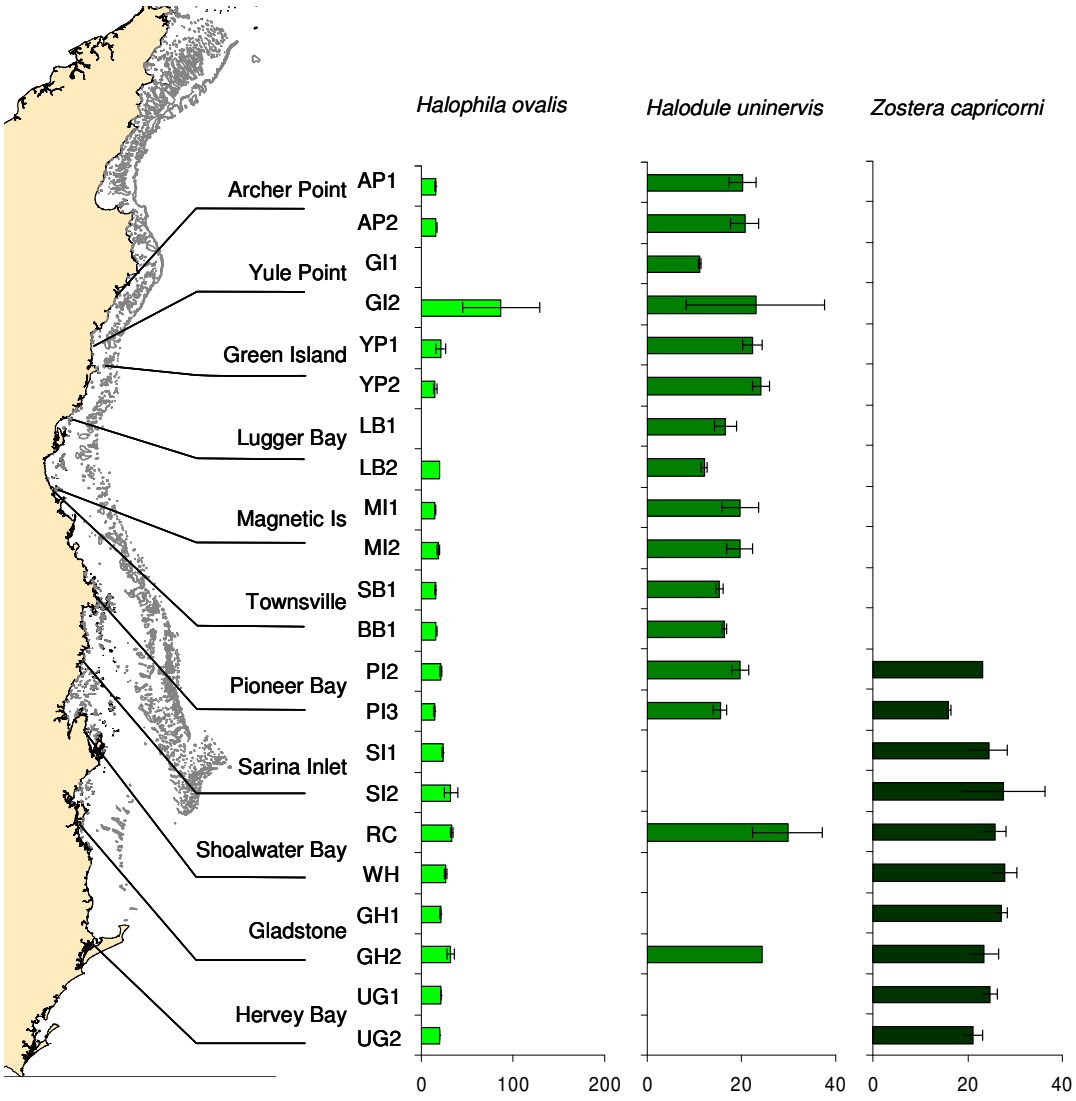


Figure 6.21. Atomic ratio of leaf tissue C: N (±95% CI) .

Seagrass reproductive health

Sexual reproduction was evident either as flowers or as a seed bank at all sites except one Green Island site (this includes flowering and seed counts). No clear indication of a relationship between sediment nutrients and reproductive effort was observed. Reproductive output in *Halophila ovalis* was highly correlated with tissue nitrogen concentrations.

Shoot production as counted by the number of nodes per core varied across sites (Figure 6.22) and reflects the differences in meadows. The number of shoots per core was often lower in April 2006 (Wet Season) although this was not consistent at all sites. For example there were more nodes per core in the Cooktown sites (AP1 and AP2) in April as opposed to October 2006.

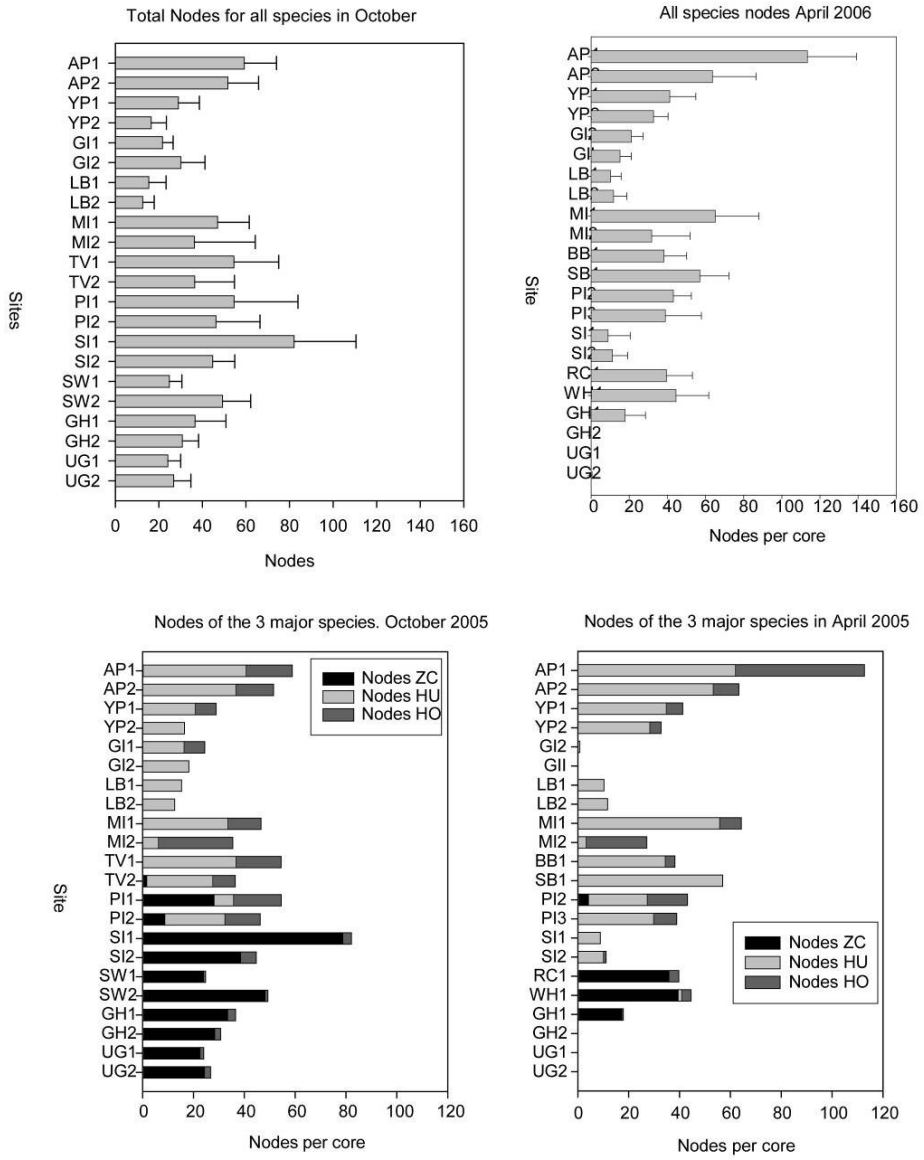


Figure 6.22. The number of nodes per core of the 3 species in October 2005 and April 2006. Top graphs are means (+ S.D.). The lower graphs show each species.

The number of reproductive structures per core (flowers and or fruits) was observed to be higher with the presence of *Zostera capricorni* at sites (Figure 6.23) in October 2005. Few reproductive structures were found in April 2006, the major flowering and fruiting period being completed (Figure 6.24). Some sites showed no evidence of flowering or fruiting, Yule Point, Green Island and Lugger Bay (Figure 6.23). All these sites however had evidence of a seed bank except Green Island (Figure 6.25). Total reproductive effort across all species is again dominated by the presence of *Zostera capricorni*. *Halodule uninervis* and *Halophila ovalis* contribute to the totals and are particularly important in the northern regions (Figure 6.26). The presence of a seed bank was strongly correlated with the presence of flowers or fruits in cores during October 2005 (Figure 6.27).

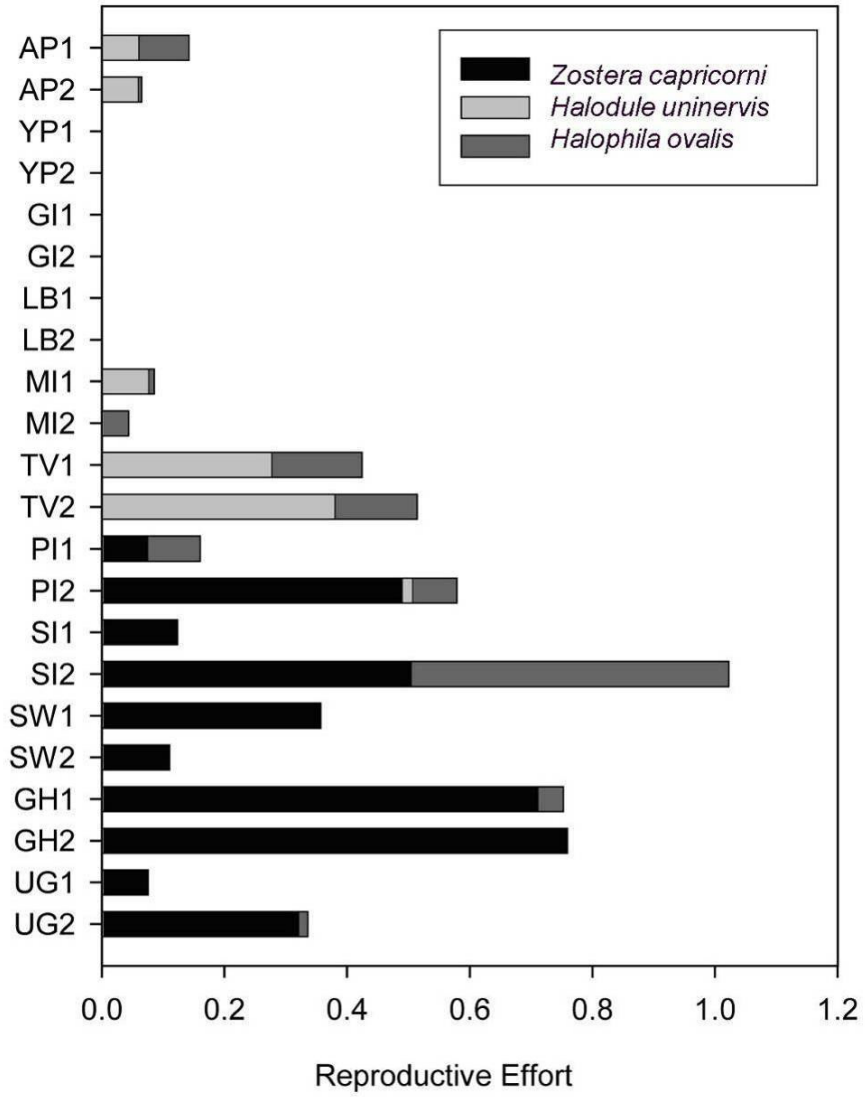


Figure 6.23. Total reproductive effort of the three study species in October 2005 (reproductive effort = the number of reproductive structures per core).

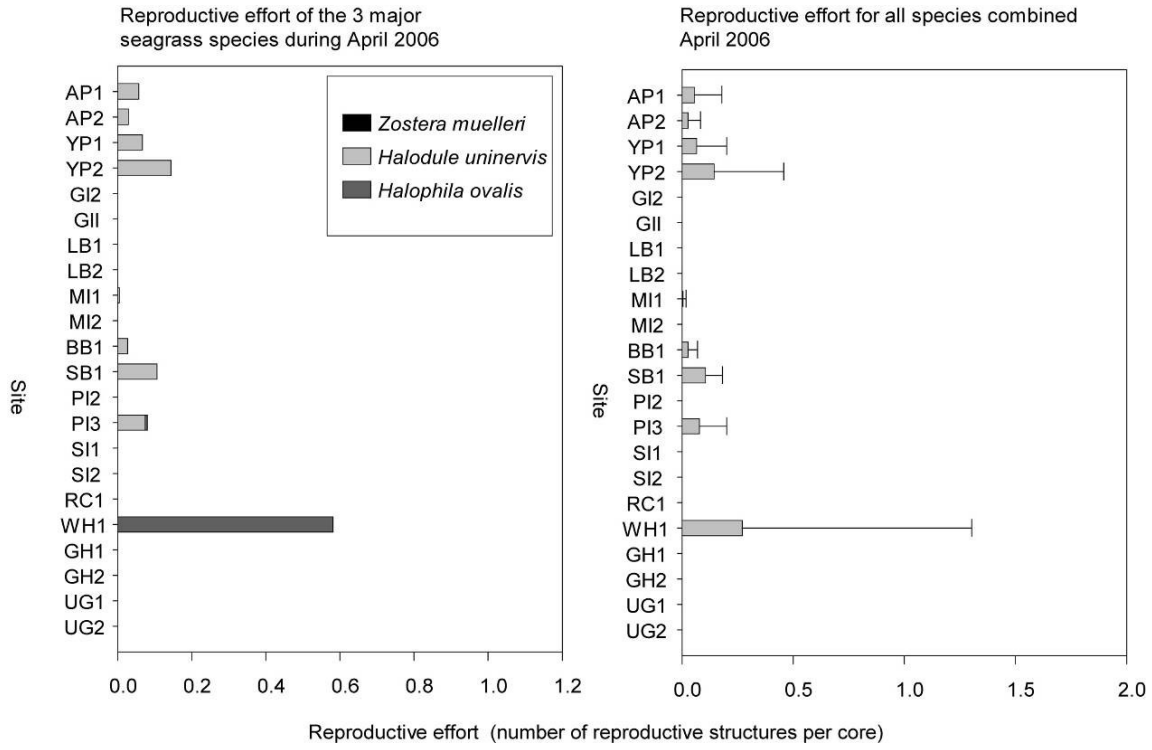


Figure 6.24. Reproductive effort of the three study species in April 2006 (reproductive effort = the number of reproductive structures per core).

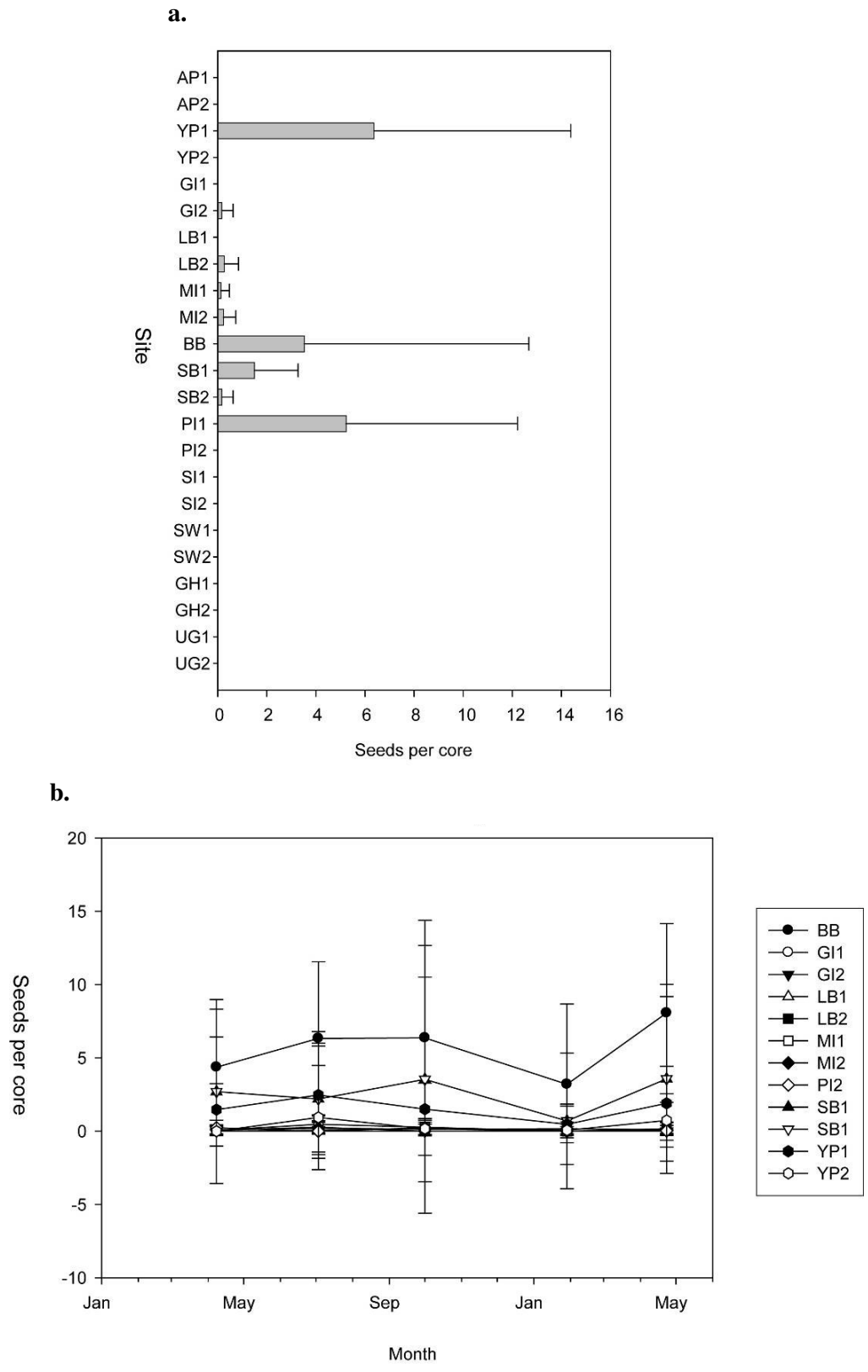


Figure 6.25. Presence of seeds in cores collected during October 2005 (a) and over the preceding five sampling periods during 2005-2006 (b).

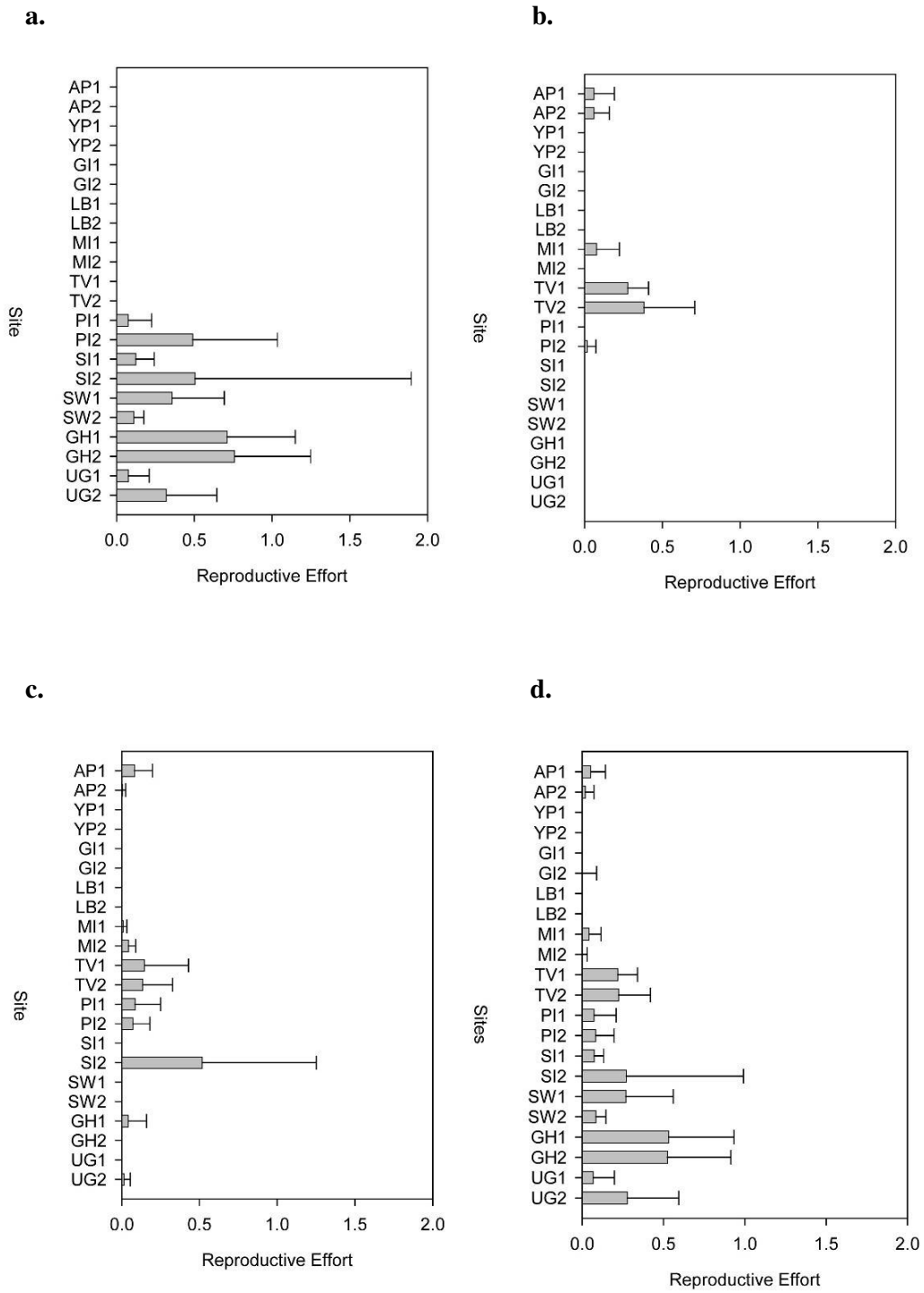


Figure 6.26. Reproductive effort calculated by core for a. Zostera capricorni, b. Halodule uninervis, c. Halophila ovalis, and d. all species pooled, in October 2005.

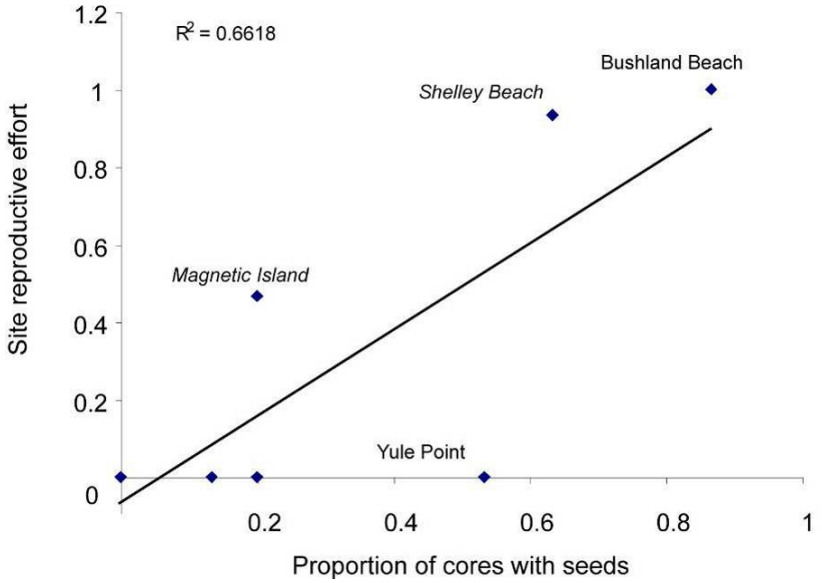


Figure 6.27. Correlation of the presence of a seed bank and the presence of flowers or fruits in cores.

All locations showed some evidence of reproductive effort, either as flowers, fruits or seeds in sediments. All meadows, with the exception of Lugger Bay, showed a capacity to recover from short-term disturbance via seed banks and thus represent healthy seagrass meadows.

Reproductive effort and nutrients

The production of flowers and fruit were analysed with respect to the sediment nutrient loads and tissue nutrient concentrations. No strong correlation was found with sediment nutrient concentration and flowering. The presence of higher tissue nitrogen concentrations (Table 6.7) were positively correlated with reproductive effort in *Halophila ovalis* (Table 6.7) suggesting nutrient limitation being responsible for flowering capacity. Given the generally moderate levels of tissue nutrients the evidence gathered here suggests seagrasses are primarily nutrient limited in their capacity to reproduce. Small increases in nutrients locally may in fact improve sexual reproduction. However the combined impact of increasing nutrients and light availability locally on flowering and fruiting remains untested.

Table 6.7. Correlations between reproductive effort and tissue nutrients in October 2005. Tissue nutrients and Reproductive effort – Pearson correlation co-efficient (r). SPSS v. 10, bold indicates p < 0.05 (one-tailed). (Note regression figures are R2 and 2-tailed based on transformed data, + 1 Log 10).

		C:P	N:P	C:N
All species (n=22)	Number nodes	0.15	-0.13	0.14
Includes Cn, Cr, Th	Reproductive effort w fruits	0.12	-0.38	0.33
	Reproductive effort w fruits (cat)	0.08	-0.49	0.33
	Flowering	0.05	-0.46	0.33
Zostera (n=10)	Number nodes	0.01	-0.40	0.71
	Flowering	0.23	-0.14	0.46
	Reproductive effort w fruits	0.15	0.01	0.03
	Reproductive effort w fruits (cat)	0.11	-0.01	0.08
Halodule (n=14)	Number nodes	0.24	0.12	0.25
	Flowering	-0.16	-0.25	-0.06
	Reproductive effort w fruits	-0.31	-0.34	-0.18
	Reproductive effort w fruits (cat)	-0.01	-0.15	0.05
Halophila (n=18)	Number nodes	-0.49	0.03	-0.70
	Reproductive effort w fruits	0.04	0.23	-0.15
	Reproductive effort w fruits (cat)	-0.20	0.36	-0.67
	Flowering	-0.04	0.40	-0.43

*Categories (cat) for reproductive effort:

All. 0, <0.1, <0.2, <0.4
Zostera. 0, <0.1, <0.5, >0.5
Halodule. 0, <0.05, <0.1, >0.1
Halophila. 0, <0.05, <0.1, >0.1

Discussion

Seagrasses form critical ecosystems in the north eastern Australian coastal waters and deserve similar attention from management agencies, researchers and the public as do the better known coral and fish populations. Their role in fisheries production and in sediment accumulation and stabilisation is well known but their role is much more diverse, spanning from directly providing food and filtering nutrients from the water, through to a role in carbon sequestration (Spalding *et al.,* 2003).

The intertidal seagrass monitoring (Seagrass–Watch program) has demonstrated that despite some temporary losses, intertidal seagrass in Queensland remain in relatively good condition (www.seagrasswatch.org). There are exceptions along the southern east-coast, including Great Sandy Strait, Hervey Bay, Gladstone, and some sites in the Mackay Whitsunday region.

Seagrass cover and distribution

The Reef Plan MMP used some existing Seagrass-Watch sites and included additional sites which have now been sampled in 2005 and 2006. The distribution of seagrass species monitored in this programme is representative of the intertidal meadows in GBRWHA (Lee Long *et al.,* 1993). While *Zostera* communities are found all along

the coast of Queensland this species predominates in southern intertidal meadows. Information on herbicides, flowering, plant morphology and seagrass area in the immediate region of the sampling sites were also collected at the Reef Plan MMP locations in addition to the suite of information collected in the broader Seagrass-Watch program. Sampling frequency at each site differed depending on the history of the site, the program funding collection of the data, and or the arrangements with the community at the site.

Between 2004 and 2005, seagrasses significantly declined in cover and distribution at four locations monitored as part of the Reef Plan MMP. The southern locations (Sarina Inlet, Gladstone Harbour and Hervey Bay) were dominated by *Zostera capricorni* communities, while the northern location (Mission Beach) was dominated by *Halodule uninervis*. The declines observed between October 2005 and April 2006 generally fit the accepted model of seasonal variation. The un-expected severity of the declines in intertidal *Zostera* meadows in Gladstone and other southern/central Queensland locations however may be related to atypical variations of climate such as rainfall, wind and water temperature occurring in the region between October 2005 and April 2006.

Studies of tropical and subtropical seagrass communities have found distinct seasonal patterns with maximum cover usually occurring in spring/summer and minima in winter (McKenzie, 1994; Lanyon and Marsh, 1995). This seasonal pattern is likely to be driven by a combination of climatic and environmental parameters, particularly rainfall, water and air temperature, and solar irradiance (Mellors *et al*., 1993; McKenzie, 1994).

The decline of seagrass in the Mission Beach area in the Wet Tropics appears a consequence of severe TC Larry, which crossed the coast 50km north of Mission Beach on 20 March 2006. Although TC Larry crossed the coast during a neap tide, there was a significant storm surge and waves that caused the sea level to exceed highest astronomical tide levels. For example, at Clump Point, sea levels exceeded predicted tides by 1.75m. Mission Beach Seagrass-Watch sites have been examined twice since TC Larry (April and July 2006). Immediately following the cyclone in late April, the abundance and distribution of the meadow was significantly lower than the same time in 2005. In July 2006, the meadow showed little sign of recovery and abundances were lower than April. The sites however occur on a naturally dynamic intertidal sand bank, which is often exposed to regular periods of disturbance from wave action and consequent sediment movement. It is unknown how long the meadow will take to recover, as there is no seed bank (no seeds have ever been found on the sites) and recovery will be limited to vegetative reproduction.

The decline of seagrasses at Urangan, near the mouth of the Mary River, appears to follow a pattern of loss and recovery which we would expect to continue. Other Seagrass-Watch monitoring sites adjacent to Urangan and the mouth of the Mary River also report patterns of loss and recovery (www.seagrasswatch.org). Intertidal *Zostera* dominated meadows in the greater region (within Great Sandy Strait), corroborate these findings.

The decline in intertidal seagrass across the Gladstone region between 2005 and 2006 is corroborated by the findings of a long term monitoring program conducted by

DPI&F and Central Queensland Ports Authority in the Port of Gladstone. In the Port of Gladstone, 13 intertidal seagrass meadows have been mapped and monitored in October-December since a detailed baseline survey conducted in 2002 (Rasheed *et al.* 2003; 2005; 2006). In October 2005, the state of seagrass meadows indicated that the marine environment in Port Curtis was relatively healthy (Rasheed *et al.,* 2006) and the Pelican Banks meadow (which includes the Reef Plan MMP sites) had increased 51% in biomass and 4% in area since the previous monitoring event (Table 6.6). However, an assessment conducted in February 2006, indicated that there was a large decline in abundance and distribution of intertidal *Zostera capricorni* meadows throughout the port area and at nearby Rodds Bay (Taylor *et al.,* 2006). The declines were most likely attributable to natural seasonal variation, coupled with a combination of other climatic factors and anthropogenic impacts, rather than the impacts of the oil spill from the bulk carrier "*Global Peace*" on 24 January 2006. Declines occurred in meadows outside of the oil spill area, including the meadow on Pelican Banks, within which the Seagrass-Watch monitoring sites are located. The Pelican Banks meadow decreased by 79% in biomass and 19% in area, since October 2005 (Table 6.8). Unlike the reported declines in seagrasses in Hervey Bay, there is not a reliable long term understanding of the natural background range of changes likely to be expected in Gladstone and a longer data set is needed for interpretation.

Table 6.8. Abundance and area (ha) of seagrass on Pelican Banks. From Taylor et al., (2006) and Rasheed et al., (2006). R= Index of Reliability.

	Nov/Dec 2002	Oct/Nov 2004	Oct 2005	Feb 2006
Mean biomass (g DW m-2)	20.8 ± 3.1	18.71 ± 2.13 *(-10%)*	28.3 ± 3.3 *(+51%)*	5.86 ± 0.89 *(-79%)*
Area ± R (ha)	624.8 ± 12.3	592.8 ± *(-5%)*	614.55 ± 11.9 *(+4%)*	499 ± 16.3 *(-19%)*

Herbicides

Water quality and ecological integrity of some coastal waters of the GBRWHA are affected by material originating in adjacent catchments as a result of human activity, including primary industries and urban and industrial development. Delivery of sediments and nutrients to rivers discharging into Great Barrier Reef waters is estimated to have increased four times since 1850 (Schaffelke *et al.,* 2005).

Coastal and estuarine seagrasses are exposed to water flows from catchments at least during flood seasons and periods of high water flow. Seagrasses absorb nutrients and trap sediment and may be exposed to chemical contaminants contained in the water or absorbed in sediment particles. Of particular concern are herbicides; chemicals designed as plant biocides that are surface applied to soil to kill unwanted vegetation in farm and agricultural settings. These chemicals may be washed into rivers and downstream and eventually concentrate to detectable levels in coastal sediments and marine plant tissues. Accumulations of the common herbicides is more likely close to the source and is likely to be in lower concentrations in offshore sediments (Haynes *et al*., 2000), however, there is a strong case to be made that any detectable amount is undesirable in what should be a pristine World Heritage Area.

Herbicides have an uncertain half life in marine sediments as they have been developed purely for terrestrial application. Atrazine has a short half life of three - 30 days, Irgarol 100 days and diuron 120 days, but toxic breakdown products may extend the time these chemicals can cause damage (Ralf *et al*., 2006; Haynes *et al*., 2000). It is likely that they remain in the water column and adsorb poorly into sediments and biota although that may not be the case if their origin is from soil erosion rather than from water. Thomas *et al*., (2001) notes that despite poor uptake, concentrations of herbicides in sediments may provide the best history of previous exposure.

There are few examples of a definite causal link between seagrass loss and herbicides, and none in Queensland. Based on laboratory aquaria studies, it is estimated that diuron concentrations of ~10 µg kg^{-1} in sediments inhibit seagrass photosynthesis (Haynes *et al.,* 2000b). The most detailed work in Queensland is from Haynes *et al,*. (2000b) who demonstrated concentrations of diuron in nearshore environments along the Queensland coast of 1-10 µg kg^{-1}. Although there have been post flood losses of seagrass in Queensland, these are more likely to be the result of light loss than from chemical contaminants (Preen *et al*., 1995).

As herbicides in Queensland are applied to prevent weed growth in agriculture the most likely time of detection in coastal waters would be late spring and summer months and the early rainfall season of November through to March. In the present study, sediment herbicide concentrations were measured in April 2005 and April 2006 (Appendix 1- 6.2). With its short half life, atrazine would be unlikely to be found in April and unlikely in sediments; no samples in either year were above detectable levels. Irgarol was also not detected in either year. Rodgers (1996) notes this chemical would be expected to remain in the water column and generally not enter sediments or biota.

Low but detectable concentrations of diuron were recorded across a number of locations sampled, suggesting wide spread contamination possibly from diffuse sources. The highest concentrations were in Mission Beach and Gladstone, 0.40 ±0.12 and 0.42 ±0.13 µg/kg respectively. These figures are well below the maximum concentrations recorded by Haynes *et al.,* (2000) (1.7 µg/kg in intertidal sediments at Cardwell) although their highest figures are outliers against otherwise low concentrations. Duke (2001) recorded concentrations of between 0.2 and 4.0 µg/kg in mangrove sediments at Mackay. Those figures are around 300 to 1300 times less than the application rates used by sugar cane farmers.

Herbicides also originate from other agricultural activities, urban weed control and possibly from sources such as antifouling paints. The key influence on the presence of herbicides in coastal sediments is likely to be significant rainfall shortly after application. This leads to a variable timing for entry into the marine system with poor predictability. To effectively assess herbicide concentrations in seagrass, samples would be best taken monthly from June to April at least for one year.

The record of diuron on the southern reef platform at Green Island requires further investigation. The island is 30km or more from potential sources of mainland contamination and detectable concentrations at this site indicate the potential for significant concentrations inshore. However, the levels found were below those that would be expected to kill seagrass. In the long term there is likely to be a detrimental

effect, such as a small reduction in the productivity of marine plants. There is a need for future reliable science relevant to *in situ* long term impacts of herbicides on marine plants and the associated biota that form the basis of food chains supporting fisheries in the GBRWHA. Little work has been conducted on the effect of chronic herbicide exposure, either in the water column or in the sediments, on phytoplankton. This lower end of the primary producer food chain is likely to be more susceptible to the presence of herbicides than the macrophytes. For example, phytoplankton are food for larval stages of many fish species. Haynes *et al.* (2000) identified this risk to the GBWHA six years ago but to the best of our knowledge no follow up work has been completed.

A longer term data set on herbicide concentrations is necessary to resolve variability in recorded sample concentrations; it is recommended that monthly sampling is conducted, initially at least for one year. Seagrass tissue and biota concentrations would add valuable information. Experimental work with phytoplankton would assist with an assessment of the potential risk to biodiversity in the GBRWHA. Given that atrazine has a very short half life and it is unlikely to be found in detectable levels in seagrass meadows, it is suggested that the funds allocated to atrazine measurements could be usefully reassigned to facilitate this work.

Sediment nutrients

In general, adsorbed nutrient levels were within the range recorded from a previous study (Mellors *et al.,* 2005). Adsorbed ammonium levels tended to be slightly lower than those recorded at equivalent sites, while adsorbed phosphate levels tended to be higher (*cf.* Mellors *et al.,* 2005). This can be accounted for by the different timing of sampling and the preferential acquisition of ammonium over phosphate by seagrass when actively growing (Touchette and Burkholder, 2000). N:P ratios indicated that, with the exception of Archer Point and Urangan, the pool of phosphate within the sediments is larger than the nitrogen pool.

The elevated levels of ammonium at Hervey Bay and the high levels of phosphate in the Whitsunday region require further investigation into the possible nutrient inputs into these meadows and geochemical nature of the receiving sediments. Correlation with water quality data or run-off data would be useful. Previous experience with correlating run-off data with seagrass sediment nutrient state has been unsuccessful (Mellors 2005). Evaluating meadow nutrient state with on site water quality measures may prove useful however the collection of on-site water quality parameters of intertidal seagrass meadows is beyond the scope of the current study. The elevated levels of phosphate at Green Island can be attributed to the calcium carbonate sediments present within the meadow rhizosphere.

Seagrass tissue nutrients

Leaf tissue contents, particularly of the macro-nutrients N and P, are indicators of the nutrient status of seagrasses as determined by nutrient availability (Hemminga and Duarte *et al.,* 2000). This indicator was used to examine areas for difference in nutrient richness. The ability to forecast from this indicator however, is poor (Borum *et al.* 2004) and the technique is expensive. Nevertheless, it is a repeatable and feasible technique to assess seagrass nutrient status and to compare approximate nutrient richness of sampling sites. As a tool it is acceptable provided it is recognised

that tissue nutrient contents are highly dependent on seagrass nutrient requirements for growth as determined by individual species, the nutrient history of the location and the age or stage of development of the meadow (see Mellors 2003, McMahon 2005). Little is known of the nutrient requirements for growth of the species present in this study with the exception of *Halophila ovalis* (McMahon 2005) as most work in this field has been done on structurally large species.

A comparison of absolute tissue nutrient contents found in this study (Table 6.9) show that %N and %P were marginally lower than the most recent inventory of tissue nutrients (Mellors *et al.,* 2005; Udy *et al.,* 1999) yet higher than the earliest recordings of tissue nutrients in this region (Birch, 1975; Lanyon, 1991). This may be due to sampling at different times of the year. The current sampling regime was dictated by timing of other biological monitoring (October). Temperate, northern hemisphere monitoring programmes have suggested that the best time to collect is during the growing season. However this is for species that are slower growing than the species regularly encountered in this region. Mellors *et al.,* (2005) and Udy *et al.,* (1999) reported values during winter months when growth is at its minimum and tissue contents are not diluted by the rapid addition of biomass, thereby truly reflecting ambient nutrient conditions. The higher recordings of tissue nutrients of the current study with those recorded by Birch (1975) and Lanyon (1991), who also collected in the winter months, may reflect an overall increase in bio-available nutrients at those locations as previously reported by Mellors *et al.,* (2005). This implies that seagrass tissue nutrients are similar in values to those recorded 10 years ago (Mellors et al 2005, Udy et al., 1999) but reflect an increase in plant tissue nutrients recorded in studies up to 30 years ago (Lanyon 1991, Brich 1975).

Insufficient data exists to assign critical levels for any seagrass species present in this study, therefore a comparison between locations and that to the global seagrass Redfield ratio for seagrasses (550:30:1) is presented. It has been shown that seagrasses growing in eutrophic conditions have C: N: P that reflect elevated nitrogen and phosphorus levels (Duarte, 1990). There were inherent species differences in relation to elemental ratios according to species specific requirement for nutrients and light. Contrasting spatial aspects of these elemental ratios provides insight into the relative contribution of the nutrient sources. In general *Halophila ovalis* had lower C:P ratios than *Zostera capricorni* and *Halodule uninervis* which is expected given the plant architectural differences between these species. *Zostera capricorni* and *Halodule uninervis* ratios were variable and differed between locations. With the exception of Yule Point, all C:N ratios for *Halophila ovalis* were below the "Redfield ratio". Sites where *Halodule uninervis* and *Zostera capricorni* ratios were below the "Redfield ratio" were Yule Point, Lugger Bay, Townsville and Pigeon Island. These ratios for *Halodule uninervis* and *Zostera capricorni* may be attributed to insufficient light for photosynthesis, as carbon concentrations have been shown to decline with light limitation (Neckles, 1993). Correlation with actual light data could help confirm this statement for seagrasses in this region. Unfortunately, this year of monitoring was constrained by finances and time. We have proposed to include light as one of the monitored variables for future monitoring when sufficient funds are available.

From a nutrient perspective, plants in nutrient-poor conditions show significantly higher C: N and/or C: P than those from nutrient rich conditions (Atkinson and Smith, 1983). The median C: N ratio for this study is 21.5, therefore locations with C: N

ratios below that value could be considered to be in nutrient rich conditions while those with ratios above this value to be in comparatively nutrient poor surroundings. This would identify Gladstone Harbour, Green Island, Lugger Bay, Magnetic Island, Pigeon Island, Townsville and Yule Point as comparatively nutrient rich environments against Shoalwater, Archer Point, Sarina Inlet and Urangan.

The N: P ratio may be more useful than C: N and C: P because they are not so reliant on structural carbon, thus reducing inter-plant variability (Johnson *et al.,* 2006). N: P in excess of 30 is considered to be evidence of P limitation and ratios less than 30-25 are considered to show N limitation (Duarte, 1990). Accordingly *Halophila ovalis* at every location showed N limitation, as did *Halodule uninervis* at Green Island and *Zostera capricorni* at Pigeon Island, Sarina Inlet and Shoalwater. In contrast, *Halodule uninervis* at Pigeon Island and Shoalwater were not nitrogen limited, illustrating the differences between nutrient requirements for individual species. Sites where neither *Halodule uninervis* or *Zostera capricorni* were nitrogen limited were Archer Point, Yule Point, Lugger Bay, Magnetic Island, Townsville and Urangan.

Halophila ovalis N: P ratio showed extreme levels of nitrogen limitation reflecting the sediment nutrient pools that in general showed sediment P to be greater than sediment N. The concurrence of *Zostera capricorni* N: P to sediment N:P ratios was quite close. No other patterns were obvious between plant tissue nutrients and sediment nutrients. This may reflect the species specific uptake mechanisms for nutrients and the location specific release of nutrients.

The variability of C: N: P plant ratios for different species and locations, their relationship with sediment nutrients, sediment type and the delivery of these nutrients suggest that there is further knowledge required about the interactions between nutrients and seagrasses in this region.

Reproductive health

The current tissue nutrient status of the sites surveyed suggests that plants are generally nutrient limited and evidence of sexual reproduction across the range of sampling indicates relatively healthy seagrass meadows.

Higher nutrient availability may be related to increasing reproductive effort however experimental work will be needed to further refine this relationship and to take into account light limitation due to water turbidity which will also limit reproductive effort. The level of sexual reproduction reflects the capacity of seagrass meadows to recover from disturbance provided that local impacts are mitigated in the short term.

Conclusions

The seagrass monitoring program has been successful to date in monitoring seagrass condition at a variety of locations, trialling the usefulness of seagrass tissue nutrients and sediment nutrients as an indicator of the relationship between seagrass health and water quality, detecting herbicides and assessing reproductive health. However, further research is required to understand results of these parameters, particularly the 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 seagrass. A number of recommendations are

made throughout the text that would facilitate this understanding including revised sampling periods to better reflect seagrass ecology, review of the herbicides that are analysed and more intensive studies of herbicides to relate variations to periods where land application is prevalent.

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Table 6.9. A comparison of plant tissue nutrients and their molar ratios for seagrass species investigated in this study with literature values characterised by plant component. Where values have been included from fertilization experiments, ambient values only are displayed. = no value quoted in the literature. [+] MI = Magnetic Island, * winter collections

Species	Location	%N	%P	N:P atomic	Source	This Study (October 2005			
						%C	%N	%P	N:P
Halophila ovalis	global average N = 2	0.7	0.18	9:1	from Duarte 1990				
Halophila ovalis	Cockle Bay, MI,[+] NQld, Aust	0.72	0.16	10:1	Birch 1975 (combined data)*	25	2.15	0.34	14:1
Halophila ovalis	Shelley Beach,, NQld, Aust	1.57	-	-	Lanyon 1991*	35.78	2.68	0.20	30:1
Halophila ovalis	Picnic Bay, MI, NQld, Aust	2.36	0.36	14:1	Mellors *et al* 2005*	30.4	2.20	0.40	12:1
Halophila ovalis	Archer Point					26.60	1.80	0.16	25:1
Halophila ovalis	Gladstone					32.5	1.60	0.41	9:1
Halophila ovalis	Pigeon Island					27.05	2.35	0.46	11:1
Zostera capricorni	global average N= 7	1.5	0.26	13:1	from Duarte 1990				
Zostera capricorni	Moreton Bay, SEQld, Aust	1.6	0.2	18:1	Udy and Dennison 1997b				
Zostera capricorni	Cairns, FNQld, Aust	1.73	0.18	22:1	Mellors *et al* 2005*				
Zostera capricorni	Cape Upstart, NQd, Aust	1.8	0.18	24:1	Mellors *et al* 2005*				
Zostera capricorni	Urangan, Hervey Bay	29.1	1.7	24.7	McMahon, 2005 (avg summer+ winter)	26.10	1.44	0.12	26:1
Zostera capricorni	Gladstone					30.05	1.5	0.13	26:1
Zostera capricorni	Pigeon Island					24.54	1.8	0.18	22:1
Zostera capricorni	Sarina Inlet					32.16	1.63	0.18	20:1
Zostera capricorni	Shoalwater					25.6	1.2	0.13	20:1
Halodule uninervis	global average N = 15	2.4	0.19	27:1	from Duarte 1990				
Halodule uninervis	Green Island, FNQld, Aust	2.4	0.26	20:1	Udy *et al.* 1999	35.26	1.75	0.15	26:1
Halodule uninervis	Moreton Bay, SEQld, Aust	2.4	0.24	22:1	Udy and Dennison 1997b				
Halodule uninervis	Cockle Bay, MI, NQld, Aust	0.92	0.15	14:1	Birch 1975	36.35	1.95	0.18	24:1
Halodule uninervis	Sth Mission Beach, FNQld, Aust	4.64	0.28	37:1	Mellors *et al* 2005*		1.30	0.10	29:1
Halodule uninervis	Cardwell, NQld, Aust	6.30	0.45	32:1	Mellors *et al* 2005*				
Halodule uninervis	Shelley Beach, NQld, Aust	3.32	0.32	23:1	Mellors *et al* 2005*	35.78	2.68	0.20	30:1
Halodule uninervis	Geoffrey Bay, MI, NQld, Aust	4.45	0.28	35:1	Mellors *et al* 2005*				
Halodule uninervis	Horseshoe Bay, MI NQld, Aust	4.19	0.21	44:1	Mellors *et al* 2005*				
Halodule uninervis	Cape Cleveland, NQld, Aust	3.47	0.23	33:1	Mellors *et al* 2005*				
Halodule uninervis	Bowen, NQld, Aust	3.02	0.18	38:1	Mellors *et al* 2005*				
Halodule uninervis	Bushland Beach					28.33	2.03	0.16	28:1
Halodule uninervis	Picnic Bay					31.22	1.9	0.15	28:1
Halodule uninervis	Yule Point						1.9	0.14	30:1

*Mellors *et al.* 2005 – winter collections

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7. Mud Crab Bioaccumulation Monitoring

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Introduction

Pesticide, petrochemical and trace metal contamination of the GBR

Organic and metal contamination of the Great Barrier Reef (GBR) is primarily due to effluent discharge, urban stormwater and agricultural and industrial runoff (Haynes and Johnson, 2000). Widespread use of now de-registered organochlorine (OC) insecticides such as DDT and dieldrin has resulted in global contamination of marine environments (Zitko, 2003). These persistent contaminants and their breakdown products have been detected at low concentrations in sub-tidal sediments of the GBR (Haynes et al., 2000a). Pesticides currently used in GBR catchments include the herbicides diuron and atrazine and the insecticides chlorpyrifos, and endosulfan (Hamilton and Haydon, 1996). These contemporary pesticides are far less persistent in the environment than the OCs such as DDT and dieldrin (van Emden and Service, 2004). While the herbicides diuron and atrazine have been detected in water samples (White et al., 2002; Shaw and Müller, 2005) and subtidal sediments (Haynes et al., 2000a) from the GBR, insecticides such as endosulfan and chlorpyrifos remain undetected (Shaw and Müller, 2005).

Polychlorinated biphenyls (PCBs), like the OC pesticides, are persistent in the environment and were once widely used in hydraulic fluids and electrical components (Breivik et al., 2004). PCBs have been detected in marine organisms such as dugongs (see below) but have not been reported in sediment or water samples of the GBR. Polyaromatic hydrocarbons (PAHs) originating from petrochemical sources are common at low concentrations in GBR habitats influenced by boating and urban pressures (Smith et al., 1985; Smith et al., 1987). Unlike pesticides, metals and metalloids are present naturally in all seawater, sediment and biota. Therefore, identifying elevated metal concentrations from anthropogenic sources in environmental samples is problematic (Fowler, 1990). However, several likely cases of sediment contamination have been identified on the GBR, including elevated copper near marinas (Brady et al., 1994) and elevated nickel (Ni), chromium (Cr), iron (Fe) and Zinc (Zn) near a nickel loading facility (Reichelt and Jones, 1994).

Effects of contaminants on keystone GBR organisms

Pesticides and elevated metals have the potential to adversely affect marine organisms. Recent research has demonstrated that herbicides currently applied on the GBR catchments can affect several key marine organisms 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). Fish (Humphrey et al., 2004) and corals (Markey et al., 2006) of the GBR are also susceptible to contemporary insecticides such as chlorpyrifos. Furthermore, metals, including copper, can affect sensitive life history transitions in corals such as fertilisation and settlement at relatively low concentrations (Reichelt-Brushett and Harrison, 2000; Negri and Heyward, 2001). Although laboratory experiments have revealed the toxic thresholds of corals, seagrass and mangroves to some contaminants,

the wider exposure of organisms on the GBR is practically unknown (Haynes and Johnson, 2000).

Bioaccumulation of pesticides and metals in marine organisms
Measuring contaminants in organisms is important to demonstrate exposure and uptake in natural environments. Bioaccumulated concentrations of some contaminants can provide environmental managers with a proxy estimate for environmental contamination as these may be difficult to measure in the water or sediments (Phillips and Rainbow, 1993). Measuring contaminants in marine biota such as macroinvertebrates may also alert management to otherwise unrecognised contaminants that are able to accumulate in marine food webs. Most insecticides, PAHs and PCBs are hydrophobic and tend to accumulate in the fatty tissues of organisms (Olsen et al., 1982). Many metals and metalloids are essential trace elements and are unavoidably present in all marine organisms. Their concentrations within organisms are, to some extent, regulated by biological detoxification processes such as metallothionein (Rainbow, 1995).

Bioaccumulation of metals and pesticides in marine organisms of the GBR was comprehensively reviewed by Haynes and Johnson (2000). Since then several relevant reports of contemporary herbicides and metals in GBR biota have been published. 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 GBR (Scarlett et al., 1999). The agricultural herbicide diuron was also detected in seagrass at nearshore 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). A recent 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) The filter-feeding barnacle species *Balanus amphitrite* was used successfully to monitor for cadmium within Ross Creek, which flows into the GBR (da Silva et al., 2004). Two species of oysters were transplanted along sites in the GBR between catchments of the Herbert and Burdekin Rivers and used to monitor zinc and cadmium (Olivier et al., 2002). Webster et al. (2001) demonstrated that a filter-feeding GBR sponge species is capable of accumulating high levels of copper in laboratory trials and may be a useful alternative species for bioaccumulation monitoring.

Mud crabs, a suitable bioindicator species for pesticide and metal contamination in the GBR
Crustaceans, including crabs, are widely recognised as useful species for biomonitoring (Phillips and Rainbow, 1993). Most of the data collected on bioaccumulation in crustacea are metal concentrations in barnacles, which are sessile but small, providing scant material for analysis of organic pollutants (Phillips and Rainbow, 1993). The mud crab *Scylla serrata* was proposed as a biomonitor species for the current RWQPP Marine Monitoring Program because of its capacity to bioaccumulate a range of contaminants, and its significance as a target species for subsistence, commercial and recreational fisheries. *Scylla serrata* males have limited territorial ranges (Ryan, 2003) and are large enough to provide ample tissue for chemical analysis. This species is also is resilient to moderately elevated insecticide (Rao and Kannupandi, 1990), PAH (Elumalai and Balasubramanian, 1997, 1999) and trace metal (Nagabhushanam et al., 1986; Dhavale, 1990; Reddy and Rao, 1990)

concentrations and should therefore be present in relatively polluted estuaries. Mortimer (2000) reported metal, metalloid and OC concentrations from *S. serrata* collected from the Brisbane River to Port Curtis (Southern GBR). In that study *S. serrata* accumulated a range of deregistered OCs and their metabolites (dieldrin, heptachlor epoxide, DDT, DDD, DDE and chlordane) and contemporary insecticides (chlorpyrifos and endosulfan) (see Discussion) as well as elevated metals Pb and Sn, particularly near urban areas. Another study of pesticide and metal concentrations in *S. serrata* from the Maroochy River and its tributaries near Brisbane also revealed the presence of de-registered OC pesticides, but no detectable concentrations of pesticides in current use (Mortimer and Cox, 1999). A small number of *S. serrata* were sampled from both the Daintree and Johnstone Rivers by Department of Primary Industries, with some found to contain trace concentrations of insecticides (dieldrin and DDT) and herbicides (atrazine and 2,4D) (Russell et al., 1996). Mud crabs were able to accumulate up to an order of magnitude more DDTs than other organisms such as bivalves and fish from the Johnstone R (Russell and Hales, 1993). Trace metals were examined in *S. serrata* by Andersen and Norton (2001), who found elevated levels of Cu and Zn in samples from Port Curtis compared with the Burdekin and Fitzroy Rivers that flow into the GBR lagoon.

Objective

Although rural runoff presents a potentially significant source of pesticides and trace metals to the GBR, no comprehensive survey of bioaccumulation within GBR catchments has been previously performed. The objective of this task was to identify spatial and temporal patterns of pesticides, PCBs, PAHs, metals and metalloids in mud crabs collected following the wet seasons (2004/2005 and 2005/2006) from 10 north Queensland rivers that flow into the Great Barrier Reef lagoon.

Methods

[Note: detailed documentation of methods was provided to GBRMPA in a separate report in October 2005: *Water Quality and Ecosystem Monitoring Programs - Reef Water Quality Protection Plan: Methods and Quality Assurance/Quality Control Procedures*.]

Sampling locations

Mud crabs (*S. serrata*) were collected from the river mouths of the 10 priority rivers and estuaries identified in the Reef Plan MMP contract. These rivers included the: Normanby, Barron, North Johnstone, Tully, Herbert, Burdekin, O'Connell, Pioneer, Fitzroy and Burnett. An additional urban site was sampled at Gordon Creek (near the main Townsville sewage outfall).

Sample collection and transport

Sampling was performed from March to July in both 2005 and 2006 by commercial fishers following consultation with the Director and regional delegates of the Queensland Seafood Industry Association. Up to 16 mature male crabs between 15 – 18 cm carapace width were collected using crab pots. Crabs of this size correspond to between 18 and 24 months of age (Mortimer *pers. obs.*). The crabs were immediately transported live to the Australian Institute of Marine Science (AIMS) by road or air for dissection.

Sample preparation

Upon arrival at AIMS, the crabs were placed in a seawater/ice slurry for 5 minutes. The mass and carapace widths were recorded and urine and hemolymph samples taken using 21 gauge syringes. The animals were sacrificed and hepatopancreas, muscle and gill samples taken. Most of the hepatopancreas was frozen at –20^{o}C for pesticide and trace metal analysis. Hepatopancreas isolated from 12 individual crabs was initially pooled into 4 hepatopancreas samples (each consisting of sub-samples of 3 randomly selected individuals) and these were extracted and analysed for pesticides to determine which contaminants were detectable in crabs at each site (Table 7.1). When pesticides were detected in the pooled samples, individuals from that pool were re-analysed. Sub-samples (~0.5 g) of hepatopancreas (x 3), muscle (x 2), gill (x 2), urine (x 2) and hemolymph (x 2) were snap frozen in liquid N_2 and stored at AIMS at –80^{o}C for future bioindicator assays.

Pesticide analyses by Queensland Health and Scientific Services (QHSS)

The analyte range included organochlorine (OC) pesticides, organophosphate pesticides, PCBs as selected congeners and PAHs. Analytes included organochlorines previously detected in mud crabs from Queensland (Mortimer, 2000) as well as a suite of herbicides and insecticides commonly used in catchments which drain into the GBR lagoon (Hamilton and Haydon, 1996). A comprehensive list of analytes and their limits of reporting are provided in Appendix 2 (Table A2-7.1). Sample receiving, handling, chemical analyses and data reporting at QHSS were based on NATA accredited methods (note that details about the accreditation can be downloaded from the NATA website http://www.nata.asn.au/). QHSS developed appropriate analytical techniques including QA/QC procedures and recovery studies for the analytes of interest in biota. These procedures include recoveries and surrogates for analytes of interest, blanks and duplicate (where possible) and internal standards for quantitation. Hepatopancreas tissue (pools = 4 g per sample wet or 12 g per pool, individuals = 4 – 6 g per sample) was solvent extracted on the accelerated solvent extraction system (ASE300) with dichloromethane and acetone. The extract initially underwent cleanup by gel permeation chromatography (QHSS Method 16621) followed by minicolumn solid phase cleanup for organochlorine pesticides, organophosphate pesticides, PCBs as selected congeners (QHSS Method 22281) and quantitation by gas chromatography-mass spectrometry (QHSS Method 15601) and gas chromatography-electron capture detection (QHSS Method 21317). Diuron and atrazine were analysed by liquid chromatography mass spectrometry (Negri et al., 2005).

Metals and metalloids

Sub-samples of hepatopancreas (~ 5 g wet weight) from 12 replicate crabs per site were freeze dried and allowed to equilibrate to constant moisture on the bench for 24 hours. The dry hepatopancreas was digested using HNO_3/$HClO_4$ (Thompson and Walsh, 1989). A Varian Liberty 220 inductively coupled plasma-atomic emission spectrometer (ICP-AES) was used to measure total Ca, Mg, Cu, Fe, Sr, Zn, Mn, Cd, Be, Co, Ni, Ba, Mo, Al, Cr, Pb, Sn, V and Se concentrations. Arsenic (As) was analysed on the Liberty 220 ICP-AES by hydride generation and mercury (Hg) by cold vapour generation following the method of Adair and Cobb (1999). Recoveries were assessed against certified reference materials from the National Research

Council of Canada (TORT2 Lobster hepatopancreas) and fell within the certified range for all elements analysed.

Statistical analyses

Although pesticide, PAH and PCB concentrations were regularly detected at more than half the sites, there were many individual crabs (~ 50%) which did not contain detectable organic contamination. This type of data was not amenable to statistical analyses, so individual concentrations are provided in summary graphs that highlight trends. Individual metal and metalloid concentrations were log-transformed to improve homogeneity and two-way analyses of variance (ANOVA) (Zar, 1996) were performed to test whether concentrations in crabs differed between sites and between sampling years. Significant differences between treatment means were assigned at $p < 0.05$. A principal components analysis (PCA) was performed on metal/metaloid concentrations. Metal concentrations used in PCA were transformed by dividing individual values by the overall mean (all sites, all replicates) for that metal. Statistical analyses were performed using Statistica 6.0, StatSoft Inc., Tulsa USA.

Results

The full pesticide and metal analysis datasets are presented in Appendix 1, Tables A1-7.1 and A1-7.2.

Crab acquisition

The collection of crab samples by commercial fishers was successful, with between 15-16 crabs collected from all but one of the study sites in 2005 (Table 7.1). The Pioneer R. is not fished commercially due to low crab numbers and theft of fishing gear. Despite exhaustive efforts by both a wholesaler and local commercial fishers, no legal male crabs were caught in the Pioneer R. in 2005. All sites were samples successfully in 2006 with 12 – 16 crabs collected from each site (Table 7.1). Severe road damage by cyclones Larry and Ingrid caused a delay in obtaining samples from the Normanby River in 2006.

Table 7.1 Details of bioaccumulation monitoring sample collection in 2005 and 2006. N=number of crabs collected, P=number of pooled samples, I=number of individuals analysed from positive pools. - indicates no crabs collected.

NRM Region	River	Date 2005	N	P	I	Organic contaminants detected	Date 2006	N	P	I	Organic contaminants detected
Cape York	**Normanby**	29/04/05	16	4	3	Y	14/06/06	16	4	0	N
Wet Tropics	**Barron**	19/04/05	16	4	9	Y	15/03/06	16	4	6	Y
	North Johnstone	28/05/05	16	4	3	Y	15/05/06	16	4	3	Y
	Tully	27/05/05	16	4	0	N	29/05/06	14	4	0	N
	Herbert	08/05/05	16	4	0	N	19/05/06	16	4	0	N
Burdekin	**Gordon Cr.**	14/04/05	16	4	3	N	01/04/06	16	4	3	Y
	Burdekin	23/05/05	16	4	12	Y	03/04/06	16	4	0	Y
Mackay Whitsunday	**Pioneer**	-	-	-	-	-	27/03/06	12	4	12	Y
	O'Connell	03/05/05	16	4	3	Y	25/03/06	16	4	12	Y
Fitzroy	**Fitzroy**	06/05/05	16	4	12	Y	19/03/06	16	4	9	Y
Burnett Mary	**Burnett**	21/05/05	15	4	12	Y	17/03/06	16	4	12	Y

Organochlorine (OC) insecticides

Pooled samples
The most common organic contaminants detected were banned insecticides such as DDT (mostly as the breakdown product DDE) and dieldrin (Table 7.2). These were detected in crabs from seven of the eleven rivers in 2005 and/or 2006. The highest concentration of total DDTs and dieldrin was 237 µg kg^{-1} lipid and 98 µg kg^{-1} lipid in pooled 2006 Burnett R. samples. Other banned OC insecticides such as trans chlordane, trans nonachlor and the heptachlor (as heptachlor epoxide), were detected at very low concentrations in single pools of crab hepatopancreas from the Normanby, Barron, North Johnstone and Burdekin Rivers in 2005. None were detected in pooled 2006 samples. The Fitzroy and Burnett River crabs contained the highest incidence of OC insecticides with all pooled samples from 2005 and three of the four pools in 2006 each containing detectable concentrations. The Pioneer R. was only able to be sampled in 2006 and all four pooled samples contained both DDTs and DDE (Table 7.2).

Individual samples
DDT and its breakdown products (mostly DDE) were detected in 36 and 38 individual crabs collected in 2005 and 2006 respectively (Fig. 7.1, and Appendix 1, Tables A1-7.1 and A1-7.2). This represents 30% of crabs collected in 2005 and 29% of crabs collected in 2006. DDTs were identified most often in crabs from the Barron, Fitzroy and Burnett Rivers in both 2005 and 2006. Nine of the 12 crabs collected from the Pioneer R. in 2006 contained DDTs. The incidence of DDT detection in crabs decreased between 2005 and 2006 for Normanby, Barron, Burdekin, Fitzroy and Burnett Rivers and rose for the O'Connell R. A maximum concentration of 1000 µg DDE kg^{-1} lipid was detected in a crab collected in 2006 from the Burnett R. but the majority of crabs contained less than 50 µg kg^{-1} lipid.

Dieldrin was the next most common organic contaminant and was identified in 15 crabs in 2005 and 37 crabs in 2006 (Fig. 7.1, Appendix 1: Tables A1-7.1 and A1-7.2). This represents 13% of crabs collected in 2005 and 28% of crabs collected in 2006. Dieldrin was found most frequently in crabs from the Burnett R. in both 2005 and 2006 and also in 11 of 12 crabs analysed from the Pioneer R. No trends in frequency of detection between sampling years were apparent; however, nine crabs from the O'Connell R. contained dieldrin in 2006 where none was detectable in 2005 samples. Concentrations ranged between 6 – 130 µg kg^{-1} lipid.

Table 7.2 Summary of organic contaminant screening results. Four pooled hepatopancreas samples from 3 crabs were each analysed by GC-MS and LC-MS. Pooled samples with detectable concentrations are shaded. The number of pools with detectable concentrations of each pesticide or contaminant class is reported (N). The concentration or range of concentrations (min–max) is also reported (µg kg^{-1} hepatopancreas lipid). NA = range not applicable

	Total PCBs		Total PAHs		chlorpyrifos		endosulfan		total DDTs		dieldrin		trans chlordane		trans nonachlor		heptachlor epoxide	
	N	range	N	range	N	range	N	range	N	range	N	range	N	range	N	range	N	range
2005																		
Normanby	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	1	7	0	NA	0	NA
Barron	0	NA	0	NA	0	NA	0	NA	3	8-117	3	9-36	1	5	1	6	0	NA
North-Johnstone	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	1	6
Tully	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Herbert	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Gordon Cr.	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Burdekin	0	NA	0	NA	0	NA	0	NA	4	7-9	1	17	0	NA	0	NA	1	11
Pioneer	0	NA	0	NA	0	NA	0	NA	-	-	-	-	-	-	-	-	-	-
O'Connell	0	NA	0	NA	0	NA	0	NA	0	NA	1	5	0	NA	0	NA	0	NA
Fitzroy	0	NA	0	NA	0	NA	0	NA	4	10-17	4	5-14	0	NA	0	NA	0	NA
Burnett	0	NA	0	NA	0	NA	0	NA	4	16-33	4	18-50	0	NA	0	NA	0	NA
2006																		
Normanby	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Barron	1	23	0	NA	0	NA	1	34	2	7-7	2	6-7	0	NA	0	NA	0	NA
North Johnstone	0	NA	1	220	0	NA	0	NA	0	NA	1	6	0	NA	0	NA	0	NA
Tully	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Herbert	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Gordon Cr.	0	NA	0	NA	0	NA	1	15	0	NA	0	NA	0	NA	0	NA	0	NA
Burdekin	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA	0	NA
Pioneer	3	16-120	1	16	0	NA	0	NA	4	7-30	4	23-69	0	NA	0	NA	0	NA
O'Connell	3	81-220	0	NA	0	NA	0	NA	3	34-103	4	8-39	0	NA	0	NA	0	NA
Fitzroy	0	NA	0	NA	1	63	0	NA	3	6-7	3	6-12	0	NA	0	NA	0	NA
Burnett	3	33-84	0	NA	0	NA	0	NA	3	24-237	4	50-98	0	NA	0	NA	0	NA

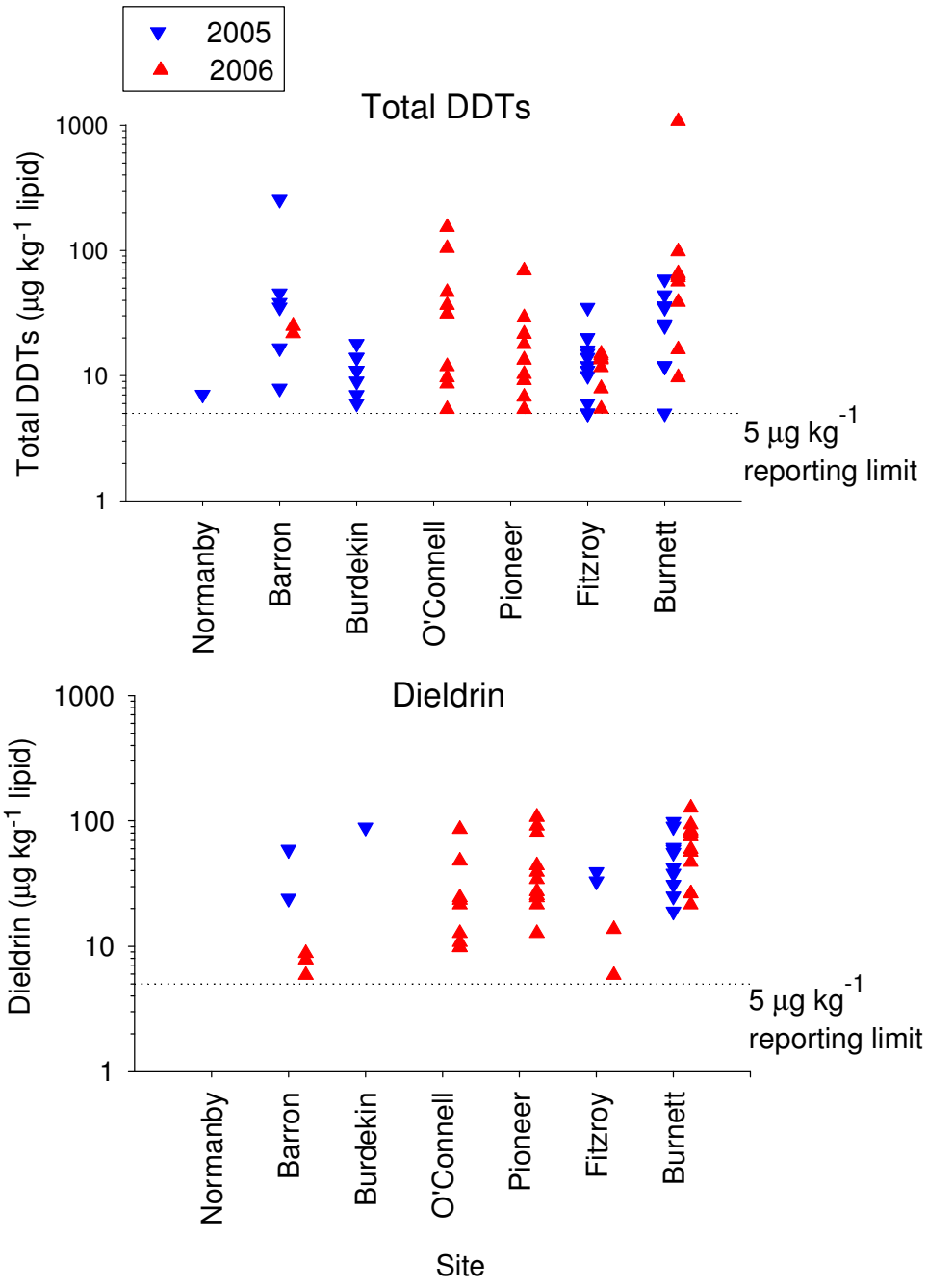

Figure 7.1 Total DDTs (DDT+DDE+DDD) and dieldrin concentrations in individual S. serrata *crabs (µg kg^{-1} hepatopancreas lipid).*

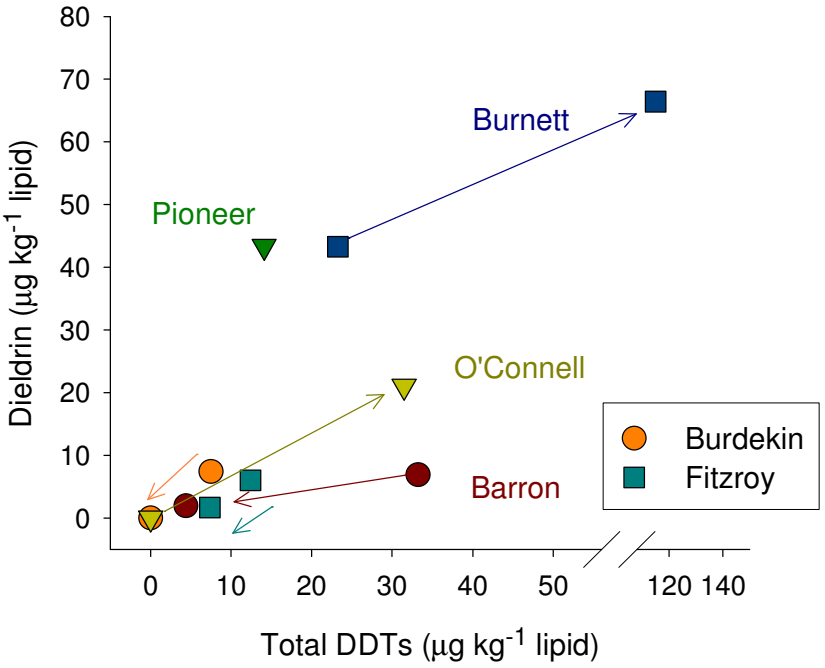

Figure 7.2 Mean DDTs (DDT+DDE+DDD) vs mean dieldrin concentration for individual crabs from each collection site and year, illustrating spatial differences and temporal trends in organochlorine (OC) concentrations. Means were calculated from a total of 12 crabs at each site and when OCs were not detected given a value of zero for that individual. Arrows point from 2005 to 2006 concentrations.

The mean DDT (DDT+DDE+DDD) concentration for individual crabs was plotted against mean dieldrin concentration for each collection site and year (Fig. 7.2). This plot revealed that when the mean DDTs increased from 2005 to 2006 (Burnett and O'Connell Rivers), this increase was accompanied by a parallel increase in mean dieldrin concentrations. Conversely a decrease in mean DDTs (Barron, Burdekin and Fitzroy Rivers) was matched by a decrease in dieldrin in crabs from this site.

The currently used insecticides endosulfan and chlorpyrifos were detected in pooled hepatopancreas samples from three rivers in 2006 (Table 7.2). Single pooled samples from the Barron R. and Gordon Ck. contained 34 and 15 µg kg^{-1} lipid endosulfan respectively. A pooled sample from the Fitzroy R. contained 63 µg kg^{-1} lipid chlorpyrifos. However, when single crabs from each of these pools were analysed individually, no chlorpyrifos or endosulfan was detectable. No herbicides such as diuron or atrazine were detected in the pooled samples.

PAHs and PCBs

Ten pooled samples collected in 2006, one from the Barron R. three each from the O'Connell, Pioneer and Burnett and Rivers, contained PCBs at concentrations up to 220 µg kg^{-1} lipid (Tables 7.2, Fig. 7.3 and Appendix 1: A1-7.3). When analysed individually, 11 *S. serrata* hepatopancreas samples contained between 12 and 562 µg kg^{-1} lipid (Appendix 1: Table A1-7.4). The highest concentrations were found in crabs from the O'Connell and Pioneer Rivers and the congener profile was relatively consistent between samples and usually dominated by IUPAC153 2,2',4,4',5,5'-hexachlorobiphenyl.

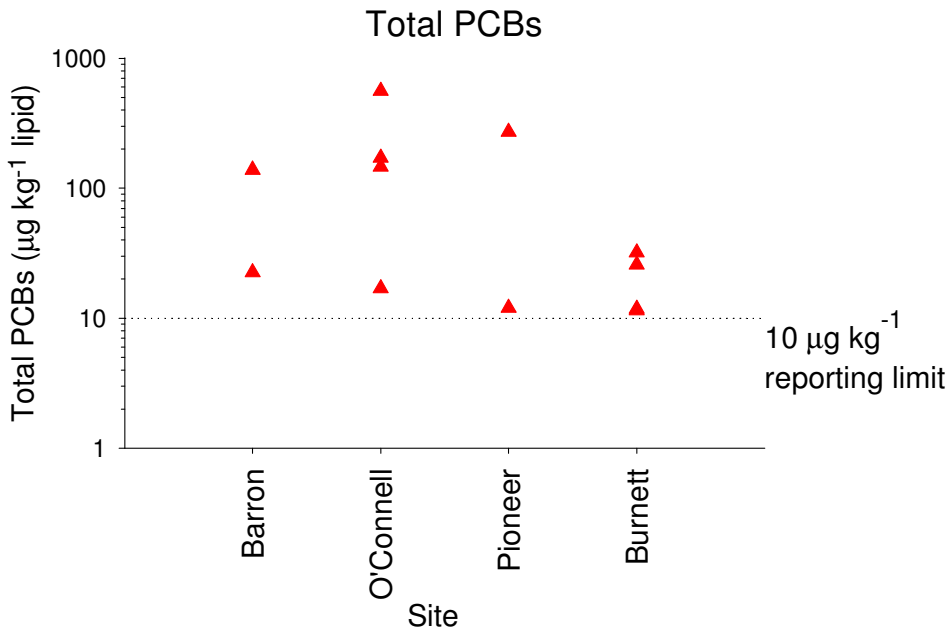

Figure 7.3 Total PCB concentrations in individual S. serrata *crabs (µg kg^{-1} hepatopancreas lipid.)*

Single pooled samples from the North Johnstone R. and Pioneer R. were found to contain 220 and 16 µg kg^{-1} lipid of PAHs respectively (Tables 7.2 and Appendix 1: A1-7.5). However, when analysed individually, none of the replicate crab samples were found to contain detectable PAHs (data not shown).

Metals

Crab hepatopancreas was analysed for a range of elements including As, Cd, Cu, Hg, Pb, Se, and Zn (Fig. 7.4) which are considered toxic at high concentrations and may originate from anthropogenic sources. Lead was only detected 5 samples and never at concentrations above 3 mg kg^{-1} dry weight hepatopancreas (Appendix 1: Tables A1-7.6 and A1-7.7). Chromium was also only detected at low concentrations, slightly higher than the reporting limit of 1 mg kg^{-1} dry weight hepatopancreas (Appendix 1: Tables A1-7.6 and A1-7.7).

A principal components analysis (PCA) of the hepatopancreas metal (As, Cd, Cu, Hg, Pb, Se, and Zn) concentrations was able to explain around 50% of metal profile variation in the first two factors and clearly demonstrated spatial and temporal patterns over the collection sites and periods (Fig. 7.5). Two-way ANOVAs also indicated significant differences in mean concentrations between sites and years for several of the metals and metalloids (Table 7.3).

Most crabs, apart from the North Johnstone R. samples exhibited similar metal profiles between years (Figs 7.4 and 7.5). Differences at that site were primarily due to much higher copper and zinc concentrations in 2005 compared with 2006. The only other cases where individual metals were significantly different ($p < 0.05$) between years were increased cadmium in the Barron R. and mercury in the Tully R. crabs in 2006. The Pioneer and North Johnstone River crabs exhibited distinctly different metal profiles from most other sites due to high zinc concentrations (Figs. 7.4 and 7.5). There was also some apparent site clustering of metal profiles within catchments (e.g. Tully, Herbert and Barron Rivers, Fig. 7.5).

Arsenic was higher in the Normanby R. crabs (100 and 101 mg kg^{-1} means in 2005 and 2006 respectively) than all other sites apart from the North Johnstone (2005) and Gordon Ck (2005 and 2006) (Fig. 7.4, Appendix 1: Tables A1-7.6 and A1-7.7). Cadmium was significantly higher in Fitzroy R. crabs (13.4 mg kg^{-1} in both 2005 and 2006) than in crabs from the Barron, O'Connell (2005), North Johnstone (2006), Tully and Herbert (2005 and 2006) Rivers. The Normanby R. crabs also contained high cadmium concentrations, especially in 2005 samples. Copper was the most variable metal in crab hepatopancreas and the most outstanding result was consistently low concentrations in North Johnstone R. crabs collected in 2005 (105 mg kg^{-1}). Mercury was significantly ($p < 0.05$) higher in crabs from the Tully R. (2.1 mg kg^{-1} in 2006) and Herbert R. (1.9 mg kg^{-1} in 2005) than all other crabs apart from those collected from the Herbert R. and Gordon Ck in 2006 (Fig. 7.4). Selenium concentrations were relatively consistent between sites and years apart from high concentrations in crabs from the O'Connell and Fitzroy Rivers in 2006 (14 and 15 mg kg^{-1} respectively). The highest concentrations of Zinc were detected in crabs from the North Johnstone R. in 2005 (334 mg kg^{-1}) and the Pioneer R. (314 mg kg^{-1}) in 2006.

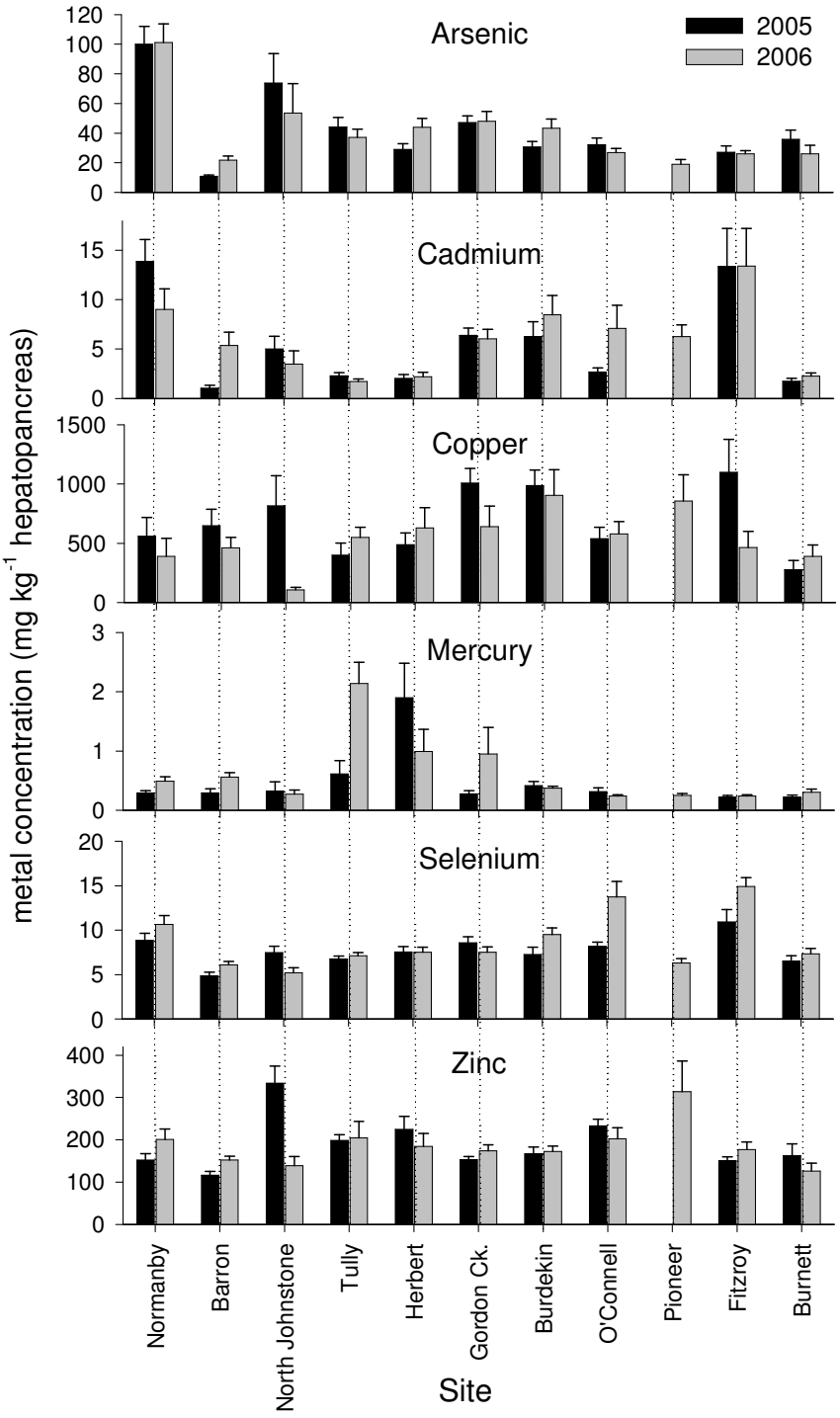

Figure 7.4 Mean (±SE) concentrations of metal and metalloids in S. serrata *collected in 2005 and 2006 (n = 12, mg kg^{-1} dry weight hepatopancreas). No samples were collected from the Pioneer River in 2005.*

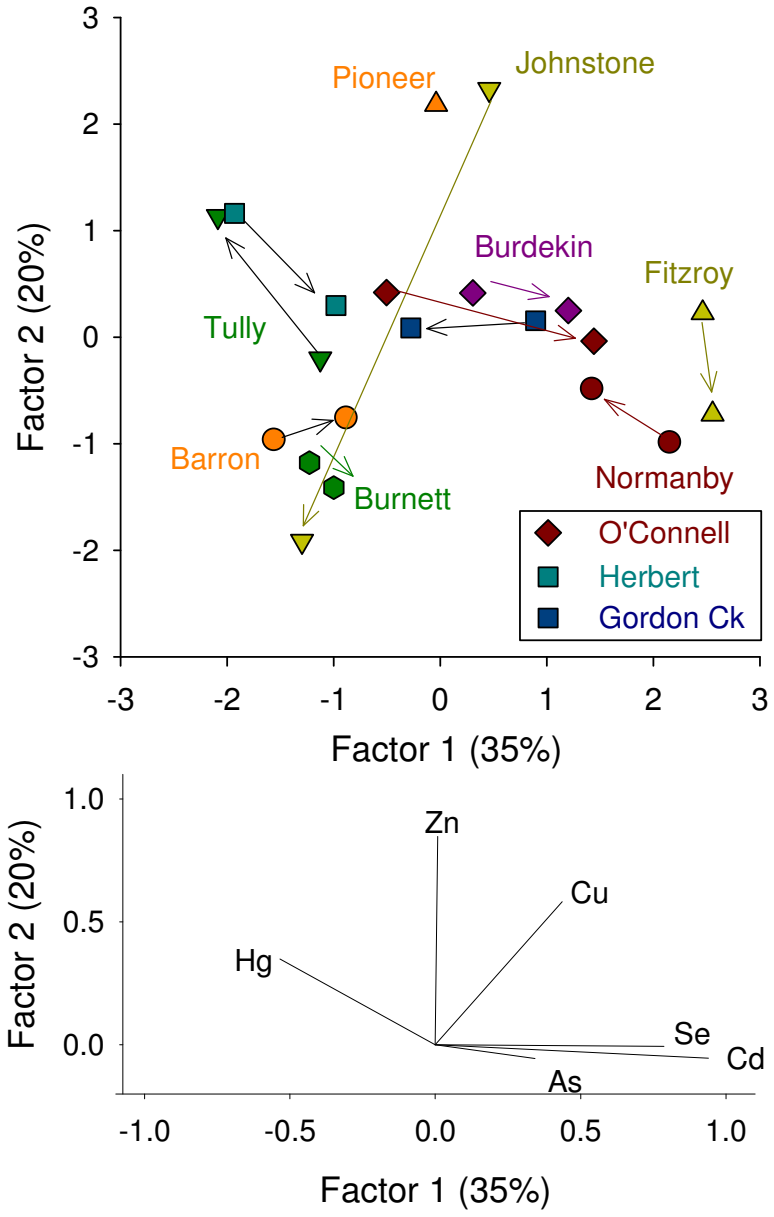

Figure 7.5 Principal component analyses (PCA) of 6 metal concentrations in crab hepatopancreas (means) from 11 sites over the two years of sampling. Arrows join data from the same site and point from 2005 to 2006 samples. Vectors represent the influence of each metal variable on the calculated factors.

Table 7.3 Two way analysis of variance (ANOVA) for metal and metalloid concentrations in response to site and year.

Metal		SS	df	F	p
Arsenic	Site	10.0	10	19.50	0.000
	Year	0.0602	1	1.17	0.281
	Site x Year	2.07	10	4.00	0.000
	Residuals		242		
Cadmium	Site	18.9	10	17.6	0.000
	Year	0.13	1	1.23	0.269
	Site x Year	4.43	10	4.13	0.000
	Residuals		242		
Copper	Site	7.80	10	4.51	0.000
	Year	1.64	1	9.50	0.002
	Site x Year	5.49	10	3.18	0.001
	Residuals		242		
Mercury	Site	9.02	10	9.80	0.000
	Year	0.718	1	7.80	0.006
	Site x Year	5.25	10	5.69	0.000
	Residuals		242		
Selenium	Site	2.56	10	15.6	0.000
	Year	0.0811	1	4.97	0.027
	Site x Year	0.813	10	5.01	0.000
	Residuals		242		
Zinc	Site	1.40	10	4.69	0.000
	Year	0.028	1	0.96	0.328
	Site x Year	1.35	10	4.55	0.000
	Residuals		242		

Discussion

Mud crabs from 7 of the 11 rivers samples (33% of all crabs sampled) contained persistent OC contaminants such as PCBs, dieldrin and the breakdown products of DDT. Contemporary pesticides such as chlorpyrifos and diuron were not detected in individual crabs, probably due to their comparatively short half-lives in the environment and higher polarity, which means they would be absorbed and accumulated less readily, as well as metabolised and excreted more rapidly than DDTs and dieldrin. Rivers with urban inputs such as the Burnett, Pioneer, Fitzroy and Barron contained the highest frequency and concentrations of OCs. Differences in metal and metalloid concentrations in crab hepatopancreas between rivers was also observed, although these differences may have been related to local geography. Mud crab biomonitoring proved complementary to passive sampler monitoring of the same rivers (Chapter 3) which accumulated a distinctly different suite of pesticides.

Deregistered organochlorines (OCs)

DDTs (mostly the breakdown products DDE) were detected in around one third of pooled crab samples collected from 6 of the 11 river mouths over the two sampling periods. The pooling of samples enabled a rapid assessment of each site and selection of individual

samples for detailed analyses. The absolute frequency and concentration of contaminants in individual crabs provided appropriate data for spatial and temporal comparisons. Of these pooled samples, 36 and 38 individual crabs contained detectable concentrations of DDE (in 2005 and 2006 respectively). The concentrations of DDTs were generally around 50 µg kg^{-1} lipid in individual crabs with a single crab sample from the Burnett R. reaching 1000 µg kg^{-1}. Dieldrin was the next most common OC and was detected in 15 individual crabs in 2005 and 37 crabs in 2006, with the maximum concentration reaching 255 µg kg^{-1} in a Barron R. crab. Other OC insecticides, including heptachlor epoxide, cis and trans chlordane, trans nonachlor and α-HCH were detected in 18 crabs over the two sampling periods but concentrations were very low, ranging from 5 – 30 µg kg^{-1}.

The results from the present study are directly comparable with those from a previous bioaccumulation monitoring study conducted on mud crabs from the Brisbane R., including a major tributary (Oxley Creek), two other rivers in SE Queensland (Maroochy and Pine Rivers), and Port Curtis between 1994 and 1998 (Mortimer, 2000). Both studies expressed organic contaminants on a per kg lipid basis since the lipophilic OCs are primarily accumulated in high-lipid tissues of crustacea (Phillips and Rainbow, 1993). Expression of OCs on the basis of lipid mass therefore reduces the variability in comparison with reporting on a whole tissue mass (wet or dry). Mortimer (2000) detected DDTs (mostly the metabolite DDE) and dieldrin in crabs from all sites sampled. The maximum mean concentration of DDTs was 2400 µg kg^{-1} lipid in crabs from the Brisbane R., while dieldrin reached a maximum mean concentration of 1400 µg kg^{-1} lipid in crabs from the urban/industrial Oxley Creek. In general, the concentrations reported in the Mortimer (2000) study were around an order of magnitude greater than those from the present study. DDTs and dieldrin were detected in less than 20% of mud crabs collected from the North Johnstone R. at concentrations as high as 290 and 2 µg kg^{-1} wet muscle mass respectively (Russell and Hales, 1993). The small intertidal burrowing crab *Australoplax tridentate* was also examined for OCs in several GBR rivers (Mortimer, 2000). The following concentrations of OCs (DDTs, dieldrin and heptachlor epoxide µg kg^{-1}) were reported: Trinity Inlet (30, 43, 43), Fitzroy R. (<50, 840, 48), North Johnstone (53, 280, 210), Ross Creek (2200, 4000, 730) and Ross R. (610, 90, <50). In that study the OC concentrations were once more an order of magnitude higher than the concentrations detected in the present study. It is plausible that the low concentrations of OCs in the present study are due to breakdown and loss of OCs from the environment during the decade between the Mortimer (2000) study collections (1996 – 1998) and those of the present study.

Sources of OCs

Persistent OC insecticides such as DDT, dieldrin, HCH, heptachlor and chlordane are distributed throughout almost all ecosystems and biota globally (Loganathan and Kannan, 1991). These insecticides were originally applied to control mosquitoes as well as crop pests and termites but most have been banned in Australia since the mid 1980s. Dieldrin and HCH were applied to sugarcane farms to control cane grubs from the late 1940s until they were banned from this application in 1987 (Cavanagh and Brunskill, 2003). The total amount of HCH applied to the Herbert and Burdekin regions between 1949 and 1998 was estimated at over 7000 tonnes (Cavanagh and Brunskill, 2003) and it was estimated that less than one tonne of dieldrin was applied in agriculture over the same period. The historical use of OC pesticides across other sugarcane growing areas in North QLD was thought to be similar to that of the Herbert and Burdekin (Agnew, 1997). The environmental half-life of dieldrin was estimated at around 400 days, indicating that a significant proportion of the insecticides applied should have diminished in the environment by time of the mud crab sampling in 2005.

Other OCs such as trans-chlordane, heptechlor epoxide and α-HCH detected at trace concentrations in this study have similarly long environmental persistence (Cavanagh and Brunskill, 2003). The Australian Pesticides and Veterinary Medicines Authority has advised that lindane (γ-HCH) is still registered for use to protect pineapple crops. Chlordane was until recently restricted for use in termite control; however there are no current registered products containing chlordane in Australia (APVMA, 2006). DDT is also extremely persistent and was used primarily for mosquito control and on crops such as cotton but less frequently on sugarcane. Heptachlor and chlordane have been used historically for termite control (Connell et al., 1999). The relatively low application levels of DDTs and dieldrin on crops in north QLD (Connell et al., 1999; Cavanagh and Brunskill, 2003) indicates that sources of OCs in the present study are likely to be urban and industrial rather than from agricultural applications (see Regional Comparison section below). It is also possible that OCs detected in biota may originate from more recent applications from stockpiles that survived since bans were imposed (McGuffog et al., 1996).

PCBs

Originally used in lubricants and electronic equipment, PCBs have been banned for use in Australia for two decades. PCBs were detected in crabs collected from 4 rivers in 2006. Although several of the samples contained concentrations just above the detection limit, five samples from the Barron, O'Connell and Pioneer Rivers contained more than 100 $\mu g\ kg^{-1}$ lipid. PCBs were not detected in sediments or seagrass in an extensive survey of GBR subtidal sites (Haynes et al., 2000a) and were not detected in mud crabs collected in the Brisbane and Gladstone areas or in the intertidal crab *A. tridentata* from 19 locations from Brisbane to Cairns (Mortimer, 2000). Low concentrations of PCBs have been reported in GBR biota such as sharks (Kannan et al., 1995), dolphins (Vetter et al., 2001), dugongs (Smillie and Waid, 1985; Vetter et al., 2001) and starfish (McCloskey and Deubert, 1972). The identification of PCBs in crab hepatopancreas here is intriguing. Mud crabs were found to contain highly chlorinated (5 – 7 chlorine residue) PCB congeners, which is consistent with previous accumulation studies. These congeners (such as 153, 138, 180) are more hydrophobic and readily bioaccumulate (Porte and Albaiges, 1994). The congener pattern may not reflect that of the original contamination due to this selective bioaccumulation and retention. It is possible that some of the crabs may have been collected from the vicinity of unknown and unreported illegal dump sites. PCBs were not detected in passive samplers (Chapter 3) and future monitoring with mud crabs may help identify the locations of point sources such as illegal dumps.

PAHs

PAHs were only detected in two pooled crab samples from the North Johnstone and Pioneer Rivers but these results were not confirmed upon thorough analysis of individual crab hepatopancreas samples from those pools, indicating that minor contamination of these two samples may have contributed to initial result. PAHs originate from a variety of sources, including fuel and lubricant spills or leaks, semi-combusted fuel, and combustion products from fires and from biogenic sources. PAHs are sometimes present in low concentrations around marinas of the GBR but were not detected in biota such as clams (Smith et al., 1987). The GBR has low concentrations of PAHs compared with other environments (Haynes and Johnson, 2000) and the present study indicates that future routine biomonitoring for PAHs may not be warranted.

Contemporary pesticides (chlorpyrifos, endosulfan, diuron, atrazine).

Chlorpyrifos was detected at low concentrations in one pooled sample from the Fitzroy R. and endosulfan in two pooled samples, one each from the Barron R. and Gordon Ck. near Townsville (Table 7.2). However, when individual crab hepatopancreas samples were re-analysed these insecticides were not detected. This inconsistency may have been due to non-homogeneity of the hepatopancreas during preparation or slight degradation may have occurred in the frozen samples in the months between analysing the pooled and individual samples, dropping the concentrations to below the detection limit. Whatever the reason for this discrepancy, is it clear that the absence of detectable chlorpyrifos and endosulfan from 96% of the pooled and 100% of the individual crabs indicates that mud crabs are not useful in monitoring these contemporary insecticides. Current herbicides such as diuron, atrazine, simazine etc. were also below detection limits in all crab samples.

Marine organisms are able to accumulate chlorpyrifos; however, the rate of elimination is rapid. This was clearly demonstrated in a two-level food chain experiment where *Artemia* spp. contaminated with chlorpyrifos were fed to a small fish species *Aphanius iberus* (Varo et al., 2002). Chlorpyrifos concentrations in the fish were measurable but dropped during exposure to the contaminated *Artemia*, probably due to rapid elimination, which may have been accelerated by adaptation of the fish metabolism. Chlorpyrifos was virtually undetectable in the fish after a day of depuration. In another experiment, the clam *Katalysia opima* was exposed to very high concentrations of chlorpyrifos (Kale et al., 2002). Depuration experiments revealed that chlorpyrifos had a half-life of approximately one day in these clams. Endosulfan also has a very short persistence in crustacea. Experimental exposures of the crayfish *Procambarus clarkii* revealed that of the 200 $\mu g\ kg^{-1}$ endosulfan that accumulated after 8 weeks, less than 2% remained in the crayfish after 4 weeks of depuration (Naqvi and Newton, 1990). No similar experiments have been performed for herbicides such as diuron or atrazine, but higher polarities mean that these herbicides are also likely to be eliminated very rapidly from mud crabs. Despite the low potential for bioaccumulation of most contemporary insecticides and herbicides, endosulfan has been detected in *S. serrata* hepatopancreas from the SE Queensland urban site of Oxley R. previously, but at extremely low concentrations (max 0.85 $\mu g\ kg^{-1}$ lipid) (Mortimer, 2000). Another study that included mud crab monitoring reported atrazine and 2,4D at 20 and 10 $\mu g\ kg^{-1}$ wet muscle tissue in crab samples collected from the Daintree R. (Russell et al., 1996). The herbicides 2,4D and 2,4,5T were not analysed in the present study since they rapidly degrade (Spain and Van Veld, 1983) and are rapidly eliminated from marine organisms (Wang et al., 1994). Although these chemicals are non-persistent in biota, this does not mean that they don't have significant impacts, and alternative monitoring techniques such as passive sampling (see comparison with passive samplers below) should be used to track their presence.

Spatial (regional and catchment) and temporal comparisons

The greatest incidence of mud crab contamination by OCs such as DDTs and dieldrin occurred in the Burnett, Fitzroy and Barron Rivers over both sampling years (Fig. 7.6). All 12 crabs analysed from the Pioneer R. in 2006 also contained DDTs and/or dieldrin. The high incidence of OCs correlated well with urban influences (Fig. 7.1 and Table 7.4). The Barron, Pioneer, Fitzroy and Burnett Rivers are in close proximity to Cairns, Rockhampton, Mackay and Bundaberg respectively and boast the four highest catchment populations. The incidence of these OCs was lowest in the low population catchments of the Normanby, North Johnstone, Tully, Herbert and Gordon Creek. Crabs from the O'Connell R. contained low concentrations of OCs and moderate PCB concentrations in 2006 only. Any relationship between agriculture and the incidence and concentration of OC insecticides and PCBs was not obvious. For

example, intensive sugarcane farming occurs across the Wet Tropics but it was the Barron R., which has the lowest areas of cropping and grazing (Table 7.5) that produced the only crabs in the region that contained detectable OCs (Fig. 7.6). Crab contamination by PCBs was observed at four sites, three of which are in catchments of high population (Barron, Pioneer, Burnett), but contamination of the O'Connell R. crabs did not fit this expected pattern and may result from illegal dumping (*Mortimer pers. obs.*).

The concentrations of OC insecticides and PCBs was low in crabs, often just above the limits of reporting and it is likely that many of the apparently uncontaminated crabs did contain OC concentrations below the level of detection. Frequent low concentrations contributed to patchy detection at some sites and made a definitive comparison between years difficult. However, both decreases (Barron, Burdekin and Fitzroy River crabs) and unexpected increases (Burnett and O'Connell River crabs) in mean OC insecticide concentrations and detection frequencies were observed (Figs. 7.2 and 7.6). Increased OC concentrations in O'Connell R. crabs was consistent with higher freshwater discharge before the 2006 collection but this was not the case for increased OCs in Burnett R. crabs (Table 2.4). These contrasting trends may be explained on the one hand by slow degradation and loss from the environment (as previously discussed), or the other hand by sampling of slightly different crab cohorts that had a different (potentially higher) exposure to the contaminants detected. This second scenario is supported by the fact that total DDT pollution increases in both Burnett and O'Connell River crabs from 2005 to 2006 were accompanied by increases in dieldrin as well as the detection of PCBs for the first time in 2006 (Fig. 7.6). However, in the medium to long term, the slow degradation of OC such as DDTs and dieldrin is likely to result in an overall drop in the body burdens of crabs in the future and these OCs may be difficult to detect 10 years from now.

*Table 7.4 Major influences on catchment water quality (GBRMPA, 2001a, 2001b). *Cropping includes sugarcane, cotton, fruit, bananas, horticulture*

NRM Region	River	Catchment population	Urban/Industrial	Agriculture (km^2)	
				Grazing	Cropping*
Cape York	Normanby	Very low	None, very low catchment population, historic gold mining.	18,500	35 (horticulture)
Wet Tropics	Barron	23,800	Close proximity to Cairns (population 120,000).	227	193 (sugar, fruit)
	North Johnstone	13,400	Close proximity to Innisfail (population 8,000)	493	438 (sugar)
	Tully	5,600	Close proximity to Tully (population 2,800)	316	273 (sugar)
	Herbert	8,800	Close proximity to Ingham (population 5,000)	7,330	626 (sugar)
Burdekin	Gordon Cr.	Very low	Overflow from Townsville sewage (population 135,000)	Low	No
	Burdekin	17,497	Near Ayr (population 8,700)	128,640	197 (sugar)
Mackay Whitsunday	Pioneer	44,159	Close proximity to Mackay (population 75,000)	1,166	296 (sugar)
	O'Connell	5,082	Near Proserpine (population 3,500)	1,904	264 (Sugar)
Fitzroy	Fitzroy	114,356	Close proximity to Rockhampton (population 60,000). Coal mining.	124,732	2790 (cotton)
Burnett Mary	Burnett	59,284	Close proximity to Bundaberg (population 45,000)	27,944	306 (sugar)

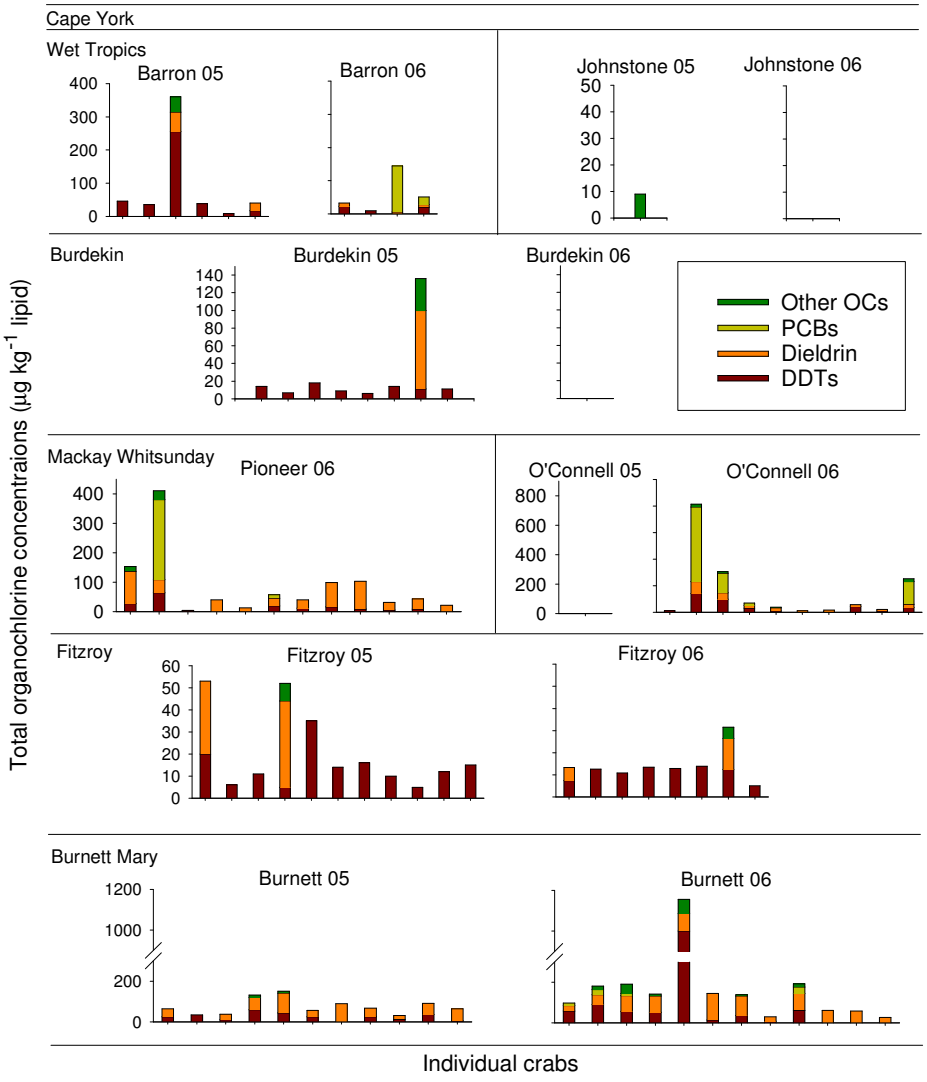

Figure 7.6 Total organochlorine (OC) concentrations of individual crabs from each river for each NRM region. Concentration scales vary between rivers. The Pioneer River was only sampled in 2006 and the Normanby River from the Cape York NRM contained only trace OC concentrations in a single crab (graph not shown). No OCs were detected from the Herbert and Tully Rivers of the Wet Tropics region or from Gordon Creek near Townsville.

Comparison between Bioaccumulation Monitoring and Passive Sampling

Passive samplers were deployed at river mouths close to where mud crabs were collected in the Barron, North Johnstone, Herbert, O'Connell, Pioneer, Fitzroy and Burnett Rivers (Chapter 3). Male mud crabs are thought have home range limited to several km (Ryan, 2003), and the crabs and passive samplers were in close proximity during these parallel studies. Passive samplers (SPMDs) detected the insecticides chlorpyrifos and diazanon in Barron, North Johnstone, Herbert, Pioneer, Fitzroy and Burnett Rivers (see Table 3.6) as well as other rivers not sampled in the Bioaccumulation Monitoring program. Endosulfan was not detected by passive sampling. DDE was detected in low concentrations (up to 0.02 ng l^{-1}) in the Barron and Fitzroy Rivers. Mud crabs in contrast, commonly accumulated and retained low to moderate concentrations of DDTs (mostly DDE), as well as dieldrin in samples from 7 of the 11 rivers. Crabs from the Barron, Pioneer, Fitzroy and Burnett Rivers were more highly contaminated with these OCs than crabs from most other rivers. However, as discussed above, little evidence of chlorpyrifos (or endosulfan) accumulation was evident in the crabs. The more polar herbicides (atrazine, simazine, diuron, hexazinone, ametryn, and tebuthiron) were each detected in most of the rivers monitored by passive samplers (EDs) (see Table 3.6). High time-averaged concentrations of diuron (up to 1400 ng l^{-1}) and atrazine (up to1500 ng l^{-1}) were detected in the Pioneer R. Again, these herbicides did not accumulate to detectable concentrations in mud crabs at any of the sites in the present study.

It is clear from these results that mud crabs and passive samplers have almost entirely distinct and different accumulation potentials. This is due to both the nature of passive samplers in comparison to living organisms and the availability of pesticides to the samplers and crabs. Passive samplers can accumulate organics, across a wide range of polarities, in a relatively predictable manner. Mud crabs on the other hand, readily metabolise the more polar contaminants such as the herbicides (e.g. diuron) and insecticides (e.g. chlorpyrifos), retaining only the most persistent OCs (e.g. DDE and dieldrin) and PCBs. The protracted elimination of these OCs, which may have biological half lives of over 12 months (Subramanian et al., 1987), mean that mud crabs are more likely to accumulate detectable concentrations of persistent OCs than most types of passive samplers. Mud crabs are also able to bioaccumulate OCs from a variety of sources, including the food web and sediments and the water column. Passive samplers accumulate only soluble contaminants. In this respect mud crab bioaccumulation monitoring complements passive sampling, which is superior at accumulating most current pesticides. The presence of more polar pesticides in passive samplers, and absence in representative biota (crabs), indicates ecosystem exposure but non-bioaccumulation.

Metals and metalloids

Metal and metalloid concentrations differed between sites but remained relatively consistent between the two sampling periods (Fig. 7.5). The PCA analysis which compared contaminant profiles (As, Cd, Cu, Hg, Se and Zn) from all sites and both years indicated that the greatest difference in metal profiles between years was for crabs from the North Johnstone R., which contained significantly more zinc and copper in 2005 compared with 2006. The PCA analysis also highlighted differences in metal profiles between sites. For instance, crabs from the four rivers in the Wet Tropics region (Barron, Tully, Herbert in both years and the North Johnstone in 2006) exhibited very similar metal profiles. The Fitzroy R. in the Fitzroy Basin and the Normanby R. in the Cape York catchment, on the other hand, grouped separate to the Wet Tropics Rivers in both years, primarily due to higher concentrations of cadmium (Fig. 7.5). Water flows do not seem to have consistently affected metal concentrations in crabs.

The Tully and Herbert Rivers both experienced more than twice the freshwater discharge before the 2006 collection (Table 2.4) compared with the previous year; however, mercury concentrations were higher in 2006 Tully crabs and lower in 2006 Herbert R. crabs (Fig. 7.4).

Data from the current monitoring were compared to results from a previous mud crab bioaccumulation monitoring program in southern QLD and Gladstone (summarised in Table 7.5) (Mortimer, 2000). Arsenic was higher in the present study generally by a factor of around 2, with only crabs from Ayr in the Mortimer (2000) study fitting within the range found in the present study. Mercury in the 2005 Herbert R. crabs was around an order of magnitude higher than detected in the southern Queensland crabs. The range of concentrations of cadmium, copper, selenium and zinc were similar between northern and southern collections. Another extensive mud crab monitoring program was undertaken at Port Curtis (Gladstone), the Fitzroy River, and the Burdekin River (Ayr) but these were reported on a wet weight basis and were therefore not directly comparable with results from the present study (Andersen and Norton, 2001). In this program mud crab hepatopancreas collected from Port Curtis contained significantly higher levels of total metals, in particular copper and zinc, than those collected from the control site in the Burdekin R. Input from industry in Port Curtis is a potential source of the elevated metal contamination. The control site near Ayr in the Burdekin region on the other hand is primarily rural (Mortimer, 2000). Relatively low levels of copper and zinc 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).

The concentrations of naturally occurring elements is often related to local geology (Phillips and Rainbow, 1993) and it is not clear what contribution anthropogenic sources may make to the metal concentrations within crabs in the present study. The most obvious case of potential anthropogenic contamination is arsenic in crabs from the Normanby R. This river flows through the most pristine catchment in the present study; however, crabs from this site contained high concentrations of arsenic compared with crabs from other sites. Historic gold mining has impacted this catchment (de Keyser and Lucas, 1968; Anon, 1993) and may contribute to elevated arsenic concentrations, as gold is often associated with arsenopyrite which can be mobilised into mine tailings and potentially into receiving rivers and estuaries. Between 500 and 1000 kg of the fungicide methoxyethylmercuric chloride (MEMC) was applied each year for 40 years in a single catchment that flows into the GBR (Brodie et al., 1984; Johnson and Ebert, 2000). Mercury concentrations in sediment cores taken from the GBR identified concentrations of up to 100 $\mu g\ kg^{-1}$, an order of magnitude higher than background concentrations (Walker and Brunskill, 1997). These concentrations were attributed to the contemporary application of mercury-based fungicides, such as MEMC, on sugar cane farms and may contribute to the higher concentrations identified in crabs from the Tully and Herbert Rivers. Gordon Creek crabs sampled in 2006 also contained higher mercury concentrations than some other sites in 2006. Potential sources at this site include both treated and untreated sewage outflow from the nearby Cleveland Bay Wastewater Plant. Mortimer (2000) recorded the highest concentrations of Pb, Se and Sn in crabs from urban and industrial sites. However, unlike OCs in this study there was no apparent relationship between catchment population (urbanisation) and metal concentrations in crabs.

Table 7.5 Mean metal concentrations in S. serrata *hepatopancreas (mg kg^{-1} dry weight) compared with values taken from Mortimer (2000)*. – signifies not collected or measured.*

Site	As		Cd		Cu		Hg		Se		Zn	
Ayr*	42		1.8		296		0.12		-		102	
Brisbane R.*	13		9.8		300		0.29		6.4		151	
Gladstone*	17		3.1		637		0.09		11.4		208	
Maroochy R.*	10		1.7		43		0.07		0.5		193	
Oxley Ck.*	4		16.8		334		BDL		1.4		162	
Pine R.*	8		4.1		67		BDL		6.2		86	
	2005	*2006*	*2005*	*2006*	*2005*	*2006*	*2005*	*2006*	*2005*	*2006*	*2005*	*2006*
Normanby	100	101	13.9	9.0	561	390	0.30	0.49	8.9	10.6	153	201
Barron	11	22	1.0	5.4	648	464	0.29	0.56	4.8	6.1	117	153
North Johnstone	74	54	5.0	3.5	817	105	0.33	0.27	7.5	5.2	334	139
Tully	44	37	2.3	1.7	399	550	0.61	2.14	6.8	7.1	199	205
Herbert	29	44	2.0	2.2	490	631	1.90	0.99	7.6	7.5	225	184
Gordon Ck	47	48	6.4	6.0	1009	641	0.28	0.95	8.6	7.5	153	174
Burdekin	31	43	6.3	8.5	986	904	0.42	0.37	7.3	9.5	166	173
O'Connell	32	27	2.7	7.1	540	579	0.32	0.24	8.2	13.7	233	203
Pioneer	-	19	-	6.3	-	857	-	0.25	-	6.3	-	314
Fitzroy	27	26	13.4	13.4	1100	464	0.23	0.24	10.9	14.9	151	177
Burnett	36	26	1.7	2.2	278	390	0.22	0.30	6.5	7.3	163	126

Copper and zinc are essential trace metals in crabs but may be toxic at high concentrations (Phillips and Rainbow, 1993). The haemolymph of crabs in particular contains high concentrations of copper which is involved in oxygen transport. These essential trace metals (copper and zinc) are highly regulated within crabs over a wide range of bio-availabilities, while non-essential metals such as cadmium are usually accumulated in proportion to availability. Mud crabs may therefore be useful biomonitors of non-essential metals such as arsenic, mercury, cadmium and selenium but more research needs to be done in combination with direct water and/or sediment samples to confirm this.

Conclusions

Obtaining replicate mud crabs within a narrow size-age class was straightforward with the help of commercial fishers. The mud crab *S. serrata* provided a large sample size for extraction, which enabled very low reporting concentrations for OCs and other pesticides. Only deregistered OC pesticides were detected in individual crabs. The inability of mud crabs to accumulate measurable concentrations of contemporary pesticides was; however, due to the chemical nature (relatively higher polarity and lower environmental persistence) of most current pesticides. For the same reasons, monitoring alternative species such as bivalves, barnacles or fish is unlikely to improve the detection of herbicides like diuron or insecticides like chlorpyrifos in estuarine organisms.

Despite the apparent inability of mud crabs to accumulate significant concentrations of current pesticides, estuarine and near-shore marine organisms such as crabs remain critical components in the assessment of water quality. Firstly, they provide important candidates for toxicity threshold comparisons. These thresholds are essential to the development of relevant and robust water quality targets and guidelines for pesticides. In addition, sensitive species such as microalgae (to herbicides) may be directly incorporated in biosensors to rapidly assess water quality (Bengtson Nash et al., 2005). Organisms such as the mud crabs used in this monitoring program would also provide tissue samples for bioindicator assays that can indirectly indicate exposure to current pesticides that are not accumulated to measurable concentrations.

The Reef Plan MMP included both mud crab bioaccumulation and passive sampler methods in the same rivers. Current pesticides were only detected in extracts from the passive samplers; however, mud crabs provided better evidence of residual OC contamination and the data here provide an extensive baseline along the GBR for future comparison. The presence of OCs such as DDT and dieldrin was clearly correlated to urban influences, with the greatest incidence and concentrations of OCs detected in crabs from the most populated catchments: Barron, Pioneer, Fitzroy and Burnett Rivers. OC concentrations in mud crabs were relatively low and, without further inputs, these concentrations are likely to fall below reporting limits within 5 to 10 years. Mud crab bioaccumulation monitoring should be repeated in 5 or 10 years if the assessment of this predicted reduction in deregistered OCs was of management interest.

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8. Overall Conclusions

There is well-documented evidence that benthic communities on inshore coral reefs of the Great Barrier Reef vary along measured or presumed gradients of terrestrial influence (Fabricius, 2005). Observed changes include variations in the cover, taxonomic composition and relative abundance of macroalgae, hard corals and soft corals, the recruitment of young hard corals and the abundance of coral bio-eroders (Fabricius *et al.,* 2005). However, on a reef-wide scale the temporal and spatial dynamics of important ecosystems such as coral reefs and seagrass meadows and their relationships with water quality and terrestrial run-off are still not well understood. 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. To assess the effects of terrestrial runoff on Great Barrier Reef lagoon water quality and ecosystems, it is necessary to understand the assimilative capacity and the critical thresholds of ecosystems in response to chronic and cumulative inputs of nutrients, sediments and pollutants. In the long-term, the Reef Plan MMP will provide necessary large-scale datasets to understand these ecological processes.

This one year project provides evidence that parameters currently monitored under the Reef Plan MMP are effectively providing suitable data to assess the effectiveness of the Reef Plan MMP, however, data collected also shows further avenues for enhancing the monitoring conducted to improve our capacity to understand and effectively manage observable changes in water quality.

River inputs into the Great Barrier Reef lagoon are naturally highly variable due to the monsoonal climate. In the reported monitoring period from 2004 to 2006 discharges from the ten priority rivers were below long-term averages, except for the five northern rivers in 2006 due to the effect of cyclone Larry. The associated loads of sediments and nutrients were also lower; concentrations of these variables are also influenced by discharge rates (there is generally a positive relationship, e.g. De'ath, 2005). While differences in total loads of terrestrial materials are important for the marine environment they are less conducive to inform land management options under Reef Plan as a result of their linkage to discharge rates. We suggest that discharge-normalised export loads are the most important performance measure for the reduction of catchment inputs to the Great Barrier Reef lagoon, together with more immediate measurements of concentrations of water quality parameters, at the sub-catchment or even paddock level.

However, currently there are only a few discharge-weighed estimates of Reef catchment river nutrient and sediment exports available. Effective sampling regimes to measure loads requires high intensity sampling of high-flow events and low intensity sampling during ambient flow, as was implemented in the current Reef Plan MMP. Due to the high natural climate variability of the region, river mouth export-monitoring should be sustained in the long-term. This, together with advanced modelling approaches, is likely to give us the ability to evaluate long-term temporal trends to assess the success of the Reef Plan.

The lagoon water quality sampling showed no distinct spatial patterns during the sampling period, apart from samples that were affected by a local flood event. This implies that, at least under low river flow conditions, the water of the coastal zone is generally well mixed and that it may be difficult to trace inputs from particular catchments outside distinct flood events. The situation may differ under higher river flow conditions. With time, inputs from individual

catchments are likely to be widely distributed along the coast. The more frequent chlorophyll sampling, which has been maintained for more than a decade, confirmed the distinct and persistent spatial pattern of higher concentration close to the coast (except adjacent to the less developed coast north of the Wet Tropics region), indicating higher nutrient availability.

Dissolved nutrient concentrations are inherently highly variable and concentrations are mostly close to detection limits. Because the dissolved nutrients are rapidly assimilated into phytoplankton biomass and rapidly recycled, plant pigments such as chlorophyll a and particulate nutrients are, therefore, more useful proxy indicators of the quantity of nutrients that are circulating within the Great Barrier Reef ecosystem. This is indicated by the long-term time series north of Cairns, which showed an increase in particulate phosphorus and suspended solids (and total dissolved phosphorus). It remains to be seen if the twice yearly sampling of the newly established water quality monitoring sites under Reef Plan will show similar trends in the long term. At this time, due to the limited time series of information available, the sampling can only serve as a rough indicator of the water quality surrounding the surveyed reefs. Sediment parameters, such as organic carbon and nitrogen content, have been discussed as useful water quality proxies (e.g. Schaffelke *et al.* 2003) and should be considered in future adaptations of the Reef Plan MMP. New marine indicators for water quality, as currently developed under the CRC Reef 'Catchment to Reef' Program, are likely to be available for field testing in the near future (Fabricius, Uthicke, Cooper and Klueter, in preparation) and it is assumed they will be integrated into the Reef Plan MMP after validation (Haynes *et al.*, in prep.).

A promising avenue is also the application of autonomous instruments to measure local environmental parameters to relate to changes in biological communities such as coral reefs or seagrass meadows and the application of remote sensing to obtain more frequent data for chlorophyll and suspended solids on a whole GBR scale.

This one year project was a proof-of-concept stage for water quality monitoring by remote sensing. We successfully provided satellite-based spatial and temporal information about near-surface concentrations of chlorophyll, suspended solids and vertical light attenuation in Reef lagoonal and coastal waters.

To develop a more cost-effective water quality monitoring framework we suggest to focus on improving the remote sensing methods and products for those parameters measurable by remote sensing (e.g., chlorophyll, SS, CDOM, K_d), improve the use of automated instrumentation for local high-frequency monitoring, and to focus *in situ* monitoring activities on i) those variables that cannot be measured by these techniques (e.g., nutrients and pesticides) and ii) on the provision of suitable validation data for the remote sensing and instrumentation approaches.

The first year of monitoring under Reef Plan MMP confirmed that pesticides including herbicides and selected insecticides and related degradation products are reaching the Great Barrier Reef Marine Park. Pesticides can affect Reef ecosystems in many ways and the impact of these chemicals is yet to be completely understood. However the results from this first deployment season clearly highlight that herbicides such as diuron and atrazine are likely to pose the highest risk to various non-target organisms such as algae, mangroves and/or corals and their symbionts.

The data from this study indicate that the wet season concentrations of herbicides present in some Great Barrier Reef rivers are high enough to cause direct phytotoxicity. It is therefore likely that these herbicides also have the potential to impact the integrity of nearshore ecosystems as they are transported into the marine environment. However, while herbicides (in particularly diuron) were routinely detectable at many of the inshore reef sites, their concentrations detected to date were below those that have been associated with acute toxicity in laboratory experiments (e.g. Negri *et al.,* 2005). Passive samplers give monthly average concentrations, however, it may be expected that short-term higher concentrations occur during flooding events when the marine ecosystems are also affected e.g. by nutrient and sediment inputs, high turbidity and low salinity.

Insecticides and PAHs were also detectable at river mouths and at selected inshore reef sites using passive sampling techniques, however, generally at much lower concentrations than suggested to affect coral settlement (Markey *et al.*, in press). Hence, there is little evidence that suggests risks to inshore reefs related to the observed concentrations of insecticides or PAHs.

Most laboratory based studies on the toxicity of pesticides to Reef organisms were based on acute toxicity on well-described endpoints. There is currently no evaluation of chronic toxicity of continuous, chronic, exposure to herbicides, insecticides and mixtures of these chemicals. This is a critical gap, as most exposure to pesticides in the Marine Park will be chronic rather than acute.

Complementing passive samplers, mud crabs (*Scylla serrata*) were useful biomonitoring species for persistent organochlorine contaminants such as PCBs, dieldrin and the breakdown products of DDT. While mud crabs preferentially accumulated non-polar persistent organochlorine pesticides, passive samplers better accumulate polar organic compounds.

The bioaccumulation monitoring program revealed that deregistered persistent organochlorines such as dieldrin and breakdown products of DDT were the most common organic contaminants to accumulate in mud crabs from the estuaries sampled, albeit at low concentrations. The most consistently contaminated crabs were sampled from the Barron, Pioneer, Fitzroy and Burnett Rivers and were correlated with urban influences. The Herbert, Tully, Johnstone and Normanby River crabs were the least contaminated. Contemporary pesticides such as chlorpyrifos and diuron were not detected in individual crabs, probably due to their comparatively short half-lives in the environment and higher polarity.

Passive samplers will be useful for ongoing monitoring of contemporary pesticides, while the mud crab monitoring will be useful for a periodic assessment of persistent organochlorine contaminants in Great Barrier Reef rivers.

The first year of coral monitoring at fixed locations at 35 inshore reef locations should be considered as a baseline assessment. While all variables (percent cover of hard corals, soft corals and macroalgae, the densities of juvenile corals and the numbers of genera present) varied greatly between the NRM regions and between reefs within the regions there were some indications of patterns that could be related to variation in water quality. Reefs close to river mouths had higher cover of macroalgae and lower densities and biodiversity of recruit-sized corals. More frequent local water quality monitoring would improve the identification of water quality-related patterns and should provide more precise local characterisation of the averages and ranges of environmental conditions experienced by reefs.

It is equally important to obtain a comprehensive record of the timing and intensity of other influences on these communities (e.g. cyclones, climate change, crown-of-thorns starfish outbreaks, coral disease) and an understanding of the processes and time-scales of recovery. The monitoring of reefs that were impacted by TC Larry (and have pre-cyclone community data from the first Reef Plan MMP surveys) will provide a unique opportunity for measuring the effects of catastrophic disturbances and the systems' resilience to such. The ability to recover from disturbances is fundamental to the long-term resilience of coral reef communities. We assume that sites with higher rates of coral larvae settlement will in general support greater future numbers of juvenile corals and subsequently of adult colonies. These assumptions need to be substantiated by careful analysis of future monitoring results, and should be complemented by studies of the ecological processes that effect resilience.

The intertidal seagrass monitoring indicated distinct local variability of seagrass meadows, similar to the finding of the coral monitoring. In the seagrass monitoring, site-specific environmental parameter were obtained, such as sediment and seagrass nutrients and sediment herbicides. Because of the high degree of site-variability and species-specific nutrient responses it will be essential to maintain as many sample sites for as long a period as possible to provide data for a reliable assessment of the condition and trend of seagrasses in the GBRWHA. This needs to be complemented by process studies that will examine the responses of seagrasses to terrestrial inputs, and to understand the effects of other influences (climate change, cyclones, major floods) on the resilience of intertidal seagrass meadows.

The present program has identified detectable concentrations of water column and sediment herbicides at Magnetic Island, Green Island and Low Isles, all of which have substantial subtidal seagrass meadows. However, the health of subtidal seagrasses is currently not being assessed under the Reef Plan MMP. While intertidal seagrass meadows are logistically easier to monitor, subtidal seagrasses may be of similar or higher ecological value and regular monitoring of these meadows should be considered in future adaptations of the Reef Plan MMP.

In conclusion, the first complete year of Reef Plan MMP activities has overcome a number of the limitations of previous water quality and ecosystem health monitoring activities in the Great Barrier Reef. Sampling locations for various activities have been co-located and the measurement of some environmental parameters has been included in the two biological monitoring tasks. The integration of the subtasks, however, should be further improved, by adaptation of the Reef Plan MMP design to better accommodate co-analysis of a water quality and biological variables.

Whilst the data collected has improved out our understanding of the We consider it premature to draw general conclusions about the health of Reef ecosystems based on the data obtained during the reporting period due to the limited period of sampling, which has not captured the full range of natural conditions inherent in these systems. However, some notable results were achieved: while eight of ten priority rivers exceeded the Queensland Water Quality Guideline values for most water quality variables, these variables were generally low in the nearshore lagoon. The nearshore lagoon is well mixed along the coast, confirming that river inputs are likely to be widely distributed along shore. Elevated levels of nutrients and suspended sediments are more localised and weather-dependent (land run-off after flooding rain and re-suspension by storm events). The herbicides atrazine and diuron are typically found at detectable levels at river mouths, inshore reef and intertidal seagrass locations, mostly with

elevated concentrations during the wet season, however, the ecological consequences of low level chronic exposure are uncertain. Only some coral reef health parameters could be linked to available water quality variables. At present, hard coral cover and species number and seagrass cover was high at most locations, and both coral reefs and intertidal seagrass meadows showed capacity for recovery from short-term disturbances, which are important in shaping these ecological communities.

This first year of monitoring has strengthened our view that processes shaping biological communities are complex and may be based on local interactions of various factors, such as water quality, climate change and disturbance. More frequent and local monitoring of environmental parameters will be necessary as well as understanding and documenting of the disturbance history of the sites. Long-term monitoring under Reef Plan MMP, as well as complimentary process-oriented research of the environmental implications of water quality on Great Barrier Reef ecosystems, will improve our understanding of changes in the Great Barrier Reef that may be attributable to the performance of the Reef Plan.

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