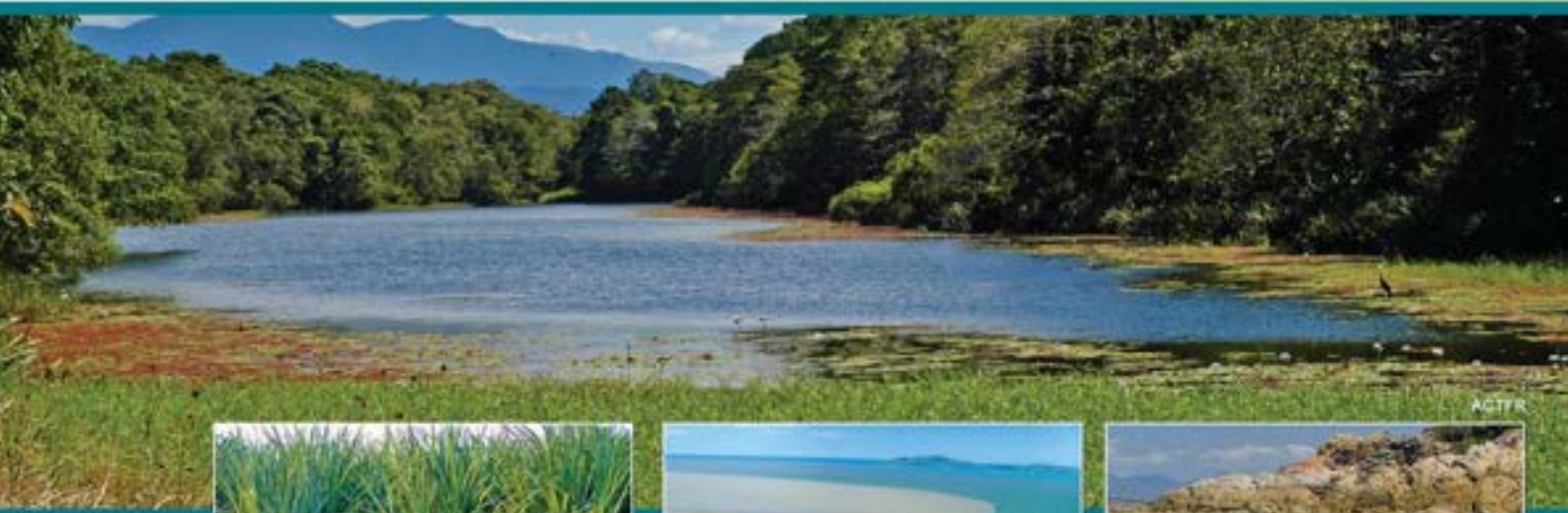




Improved understanding of biophysical and socio-economic connections between catchment and reef ecosystems: Wet and Dry Tropics case studies



Compiled by Michelle Devlin and Jane Waterhouse



Australian Government
Department of the Environment,
Water, Heritage and the Arts



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Australian Government

**Department of Sustainability, Environment,
Water, Population and Communities**

Supported by the Australian Government's
Marine and Tropical Sciences Research Facility

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ISBN 978-1-921359-58-3

This report should be cited as:

Devlin, M. J. and Waterhouse, J. (Compilers) (2010) *Improved understanding of biophysical and socio-economic connections between catchment and reef ecosystems: Wet and Dry Tropics case studies*. Synthesis Report prepared for the Marine and Tropical Sciences Research Facility (MTSRF) with contributions from K. Fabricius, M. Waycott, J. Wallace, F. Karim, R. Pearson, A. Arthington, J. Brodie, S. Lewis, Z. Bainbridge, P. Kuhnert, M. van Grieken and their research teams. Published by the Reef & Rainforest Research Centre Ltd, Cairns (145pp.).

Published by the Reef and Rainforest Research Centre on behalf of the Australian Government's Marine and Tropical Sciences Research Facility.

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December 2010

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Acronyms and Abbreviations

ACTFR	Australian Centre for Tropical Freshwater Research
APSIM	Agricultural Production Systems Simulator
ASTER	Advanced Spaceborne Thermal Emission and Reflection Radiometer
AVHRR	Advanced Very High Resolution Radiometer
Ba	Barium
BRIA	Burdekin River Irrigation Area
Ca	Calcium
CDOM	Coloured Dissolved Organic Matter
COTS	Crown-of-thorns starfish
CRC	Cooperative Research Centre
DEWHA	Commonwealth Department of the Environment, Water, Heritage and the Arts (now Sustainability, Environment, Water, Population and Communities)
DIN	Dissolved Inorganic Nitrogen
DOC	Dissolved Organic Carbon
DON	Dissolved Organic Nitrogen
DOP	Dissolved Organic Phosphorus
DPC	Queensland Department of the Premier and Cabinet
GAM	Generalised Additive Model
GBR	Great Barrier Reef
GBRMPA	Great Barrier Reef Marine Park Authority
GPS	Geographical Positioning System
LRE	Loads Regression Estimator
MERIS	Medium Resolution Imaging Spectrometer
MMP	Reef Rescue Marine Monitoring Program
MODIS	Moderate Resolution Imaging Spectroradiometer
MTSRF	Marine and Tropical Sciences Research Facility
NO_x-N	Nitrogen oxides nitrogen trace gas emission ratio
NRM	Natural Resource Management
P	Phosphorus
PN	Particulate Nitrogen
PP	Particulate Phosphorus
ppt	Parts per thousand
PSU	Practical Salinity Units
PS-II	Photosystem II
RRRC	Reef & Rainforest Research Centre Ltd.
SeaWIFS	Sea-viewing Wide Field-of-view Sensor

SS..... Suspended Sediment
TN..... Total Nitrogen
TSM..... Total Suspended Matter
TSS Total Suspended Sediment
WQIP..... Water Quality Improvement Program
WWF World Wide Fund for Nature

About this Report

This report provides an overview of the key findings of research conducted through the Marine and Tropical Sciences Research Facility (MTSRF) designed to improve our understanding of the linkages between catchment and reef processes, and how the quality of water from paddock, sub-catchment, catchment and marine systems can directly and indirectly influence the ecological functioning of the Great Barrier Reef (GBR). The research aimed to inform and facilitate management action and remediation to reduce, restore and increase resilience of the inshore GBR ecosystems. The research findings are also applicable elsewhere, particularly in tropical ecosystems, but many outcomes can be translated for broader application in catchment and marine ecosystem management.

A key achievement of the MTSRF has been the strong cooperation and collaboration between research institutions in project development and implementation. Many of the findings presented in this report were derived from large collaborative projects funded from several sources, including the MTSRF, and the research institutions have also contributed significant in-kind resources. It should be noted that supporting information external to the MTSRF is included in this report to provide context or to complete the discussion. Publications specifically generated through research funded by the MTSRF are identified in the reference list.

The report is one in a series of information products that summarise MTSRF research findings relevant to managing water quality in the GBR. Other synthesis products – companion reports to this synthesis report – related to water quality include:

- A summary of MTSRF Water Quality Program highlights (Waterhouse and Devlin, 2010);
- ‘Optimising water quality and impact monitoring, evaluation and reporting programs’ (Waterhouse, 2010);
- ‘Identification of priority pollutants and priority areas in Great Barrier Reef catchments’ (Waterhouse and Brodie, 2010); and
- A synthesis of water quality and climate change interactions, and socio-economic influences on water quality management in the Great Barrier Reef (in prep.).

Acknowledgements

The compilers gratefully acknowledge the combined authorship of the researchers who contributed to this report. Outstanding work by Katharina Fabricius, Michelle Waycott, Jim Wallace, Fazlul Karim, Richard Pearson, Angela Arthington, Jon Brodie, Stephen Lewis, Zoe Bainbridge, Petra Kuhnert, Martijn van Grieken and their research teams has been instrumental in advancing our knowledge of the catchment to reef processes.

Thanks are extended to the then Department of the Environment, Water, Heritage and the Arts (DEWHA) (now Department of Sustainability, Environment, Water, Population and Communities) for funding the research through the MTSRF, and the North Queensland based Reef & Rainforest Research Centre (RRRC) for supporting this program over the past four years. The involvement of primary end users in this research, including the Great Barrier Reef Marine Park Authority (GBRMPA), DEWHA, WWF-Australia, the Queensland Department of the Premier and Cabinet (DPC) and Regional Natural Resource Management (NRM) groups, has focused the research outcomes to ensure that the findings can be (and have been) directly applied to the management of the Great Barrier Reef.

Executive Summary

Over the past thirty years an increasing amount of research and monitoring effort has been devoted to documenting and understanding the nature and importance of water quality issues for the Great Barrier Reef (GBR). Attention has become focused on land-based runoff as a primary source of pollutants into the GBR. This report reviews, synthesises and analyses the work carried out over the course of the Marine and Tropical Science Research Facility (MTSRF) in relation to our current understanding of the relationships between catchment processes, pollutant loads delivered to instream environments (including wetlands and estuaries) and the marine environment, and the impacts on instream environments and the near shore environment. Key sources of information for this synthesis of catchment to reef water quality research are largely from MTSRF Theme 3 'Halting and Reversing the Decline of Water Quality'¹. Case studies from MTSRF research conducted in the Dry Tropics (Burdekin) and Wet Tropics (Tully) of North Queensland are presented to demonstrate a more detailed understanding of catchment to reef relationships, and to highlight the advances made in our understanding of the broader relationships. The highlights of these case studies are summarised in Figure i and ii respectively.

This report highlights research results that have changed our understanding since the commencement of the MTSRF Research Program in 2006 and hence may be critical in revising the aims or priorities of water quality policy frameworks such as the Reef Water Quality Protection Plan², given the new understanding. The breadth and diversity of MTSRF funded research is presented in this report as well as other companion reports (Waterhouse, 2010; Waterhouse and Brodie, 2010). The success of the MTSRF model stems primarily from the research crossing over many science disciplines, including the social and economic sectors. The combination of these multiple strands of information has allowed a comprehensive approach to studying catchment-to-reef processes.

A number of key findings of the MTSRF in relation to the processes that connect the whole catchment to reef system are highlighted below:

- Conceptual biophysical models have been developed to identify appropriate health indicators of freshwater ecosystems, including stream, floodplain lagoon and wetland health, while probable thresholds of concern, in terms of contaminant concentrations, ecological processes and biodiversity have also been investigated for these ecosystems. Indicators of freshwater ecosystems have been developed and are related to pressures that include patterns and types of land use, general water quality and contaminants, hydrological regime, channel and habitat structure, riparian vegetation condition and alien species of plants and fish. Measurement of spatial and temporal variability of biophysical indicators in floodplain wetlands of the Tully-Murray catchment have been correlated with those pressures.
- Connectivity between freshwater ecosystems is important for maintaining ecosystem health and has been studied using hydrological modelling in the Tully-Murray floodplain area. The degree of connectivity of different wetlands, ranging from those wetlands that are more permanently connected with streams and drains to those that are connected only when there are large overbank floods, varies with wetland location and flood magnitude. These results have important implications for (i) the movement and recruitment patterns of aquatic biota during and after flood events, (ii) wetland habitat characteristics and water quality, (iii) the biodiversity of individual wetlands over time, and (iv) the potential for wetland processes to influence the quality of water flowing to the

¹ http://www.rrrc.org.au/mtsr/theme_3/index.html

² <http://www.reefplan.qld.gov.au/>

GBR lagoon. As the hydrodynamic model is driven by daily rainfall it should also be possible to quantify the potential impacts of climate change on wetland connectivity, if the future changes in rainfall can be specified.

- Sediments, nutrients and pesticides are the priority pollutants for management of water quality in the GBR. Studies within the MTSRF have informed the refinement of knowledge of priority areas for pollutant generation and, hence, management in GBR catchments.
- In the Burdekin River catchment, sediment load is dependent on the catchment characteristics and size of flow event. However, regardless of flood event size in the other catchments, the upper Burdekin basin is always likely to be the dominant source (83-97%) of suspended sediment into the Burdekin Falls Dam. Total suspended sediment (TSS) load delivered over the dam spillway makes up a smaller proportion (20-50%) of the total load exported from the Burdekin River than the below dam catchment area, and it is estimated that 50-80% of the suspended sediment export ('bulk' suspended sediment) to the GBR lagoon has been sourced from the catchment area below the Burdekin Falls Dam. Thus, management efforts should be primarily focused on these lower catchments which make up only a small percentage of the overall Burdekin catchment area.
- Studies in the Burdekin catchment show that there are different delivery pathways between the bulk (heavier) sediment and the finer sediment. There is little deposition of the finer clay fraction as it is transported within the catchment compared to coarser size fractions (such as silts and sand) which are preferentially being deposited within the dam or during other opportunities for deposition. Size distribution shows the movement of the finer sediment from the upper catchments, through the dam and into the marine environment. These results are also relevant to other Dry Tropics catchments in the GBR. Further studies show that the finer fraction (<38 μm component) of the sediment is present in the turbid primary plume which is generally constrained closer to the coast but was not seen in the larger secondary plume as inorganic matter. These latest particle size results indicate that the finer clay fractions are being transported not only throughout the catchment, with little opportunity for deposition, but also within the marine environment via resultant flood plumes. It is this finer fraction which has been linked to the degradation of coral reef ecosystems and therefore may pose the greatest risk to receiving marine ecosystems.
- Building on this knowledge, receiving water models can be used to develop sediment budgets for areas within the GBR. For example, a hydrodynamic model has been developed for Cleveland Bay (receiving waters of the Burdekin River) which shows that the amount of riverine sediments settling on the bay may exceed the amount of sediment exported from the bay by 50-75%. Sediment is thus accumulating in the bay on an annual basis, with potentially negative effects on coral reefs. A net sediment outflow from the bay may only occur during years that experience a tropical cyclone. Thus the majority of the sediment accumulates in areas where it is frequently re-suspended by waves under trade winds, thus increasing the turbidity of the bay.
- In the Tully-Murray River catchments, estimates of nutrient loads being delivered during flood events to the GBR lagoon have been significantly underestimated in the past. Through MTSRF research, the flood contributions were found to increase the mean annual loads of phosphorus and nitrogen loads by 30-50% above previous river based estimates. These results indicate that there is therefore a clear need to obtain estimates of the contribution that floods make to marine loads in other GBR catchments.
- Comprehensive research on the impact of sediments and nutrients on the GBR ecosystems has been undertaken as part of the MTSRF, and the preceding CRC Reef Research Centre and Rainforest CRC joint 'Catchment to Reef' Program, and can be represented in a series of conceptual models. This work has led to the development of 'thresholds of concern' for several water quality variables and ecosystem components, which in turn have been used in the development of Water Quality Guidelines for the

Great Barrier Reef Marine Park (GBRMPA, 2009)³. The research has also demonstrated a link between elevated concentrations of nutrients and the location and frequency of COTS outbreaks.

- Studies on the effects of herbicides on GBR ecosystems have shown that herbicides are being detected in many locations in the GBR, especially following rain events, and that increased exposure can potentially threaten ecosystems within the GBR. The herbicides most commonly detected in the GBR lagoon are designed to inhibit photosystem II in plants and so the risk of these herbicides should be considered additively. Previous studies have examined the risk of individual herbicides in isolation; recent monitoring studies show that 80% of the time when herbicides are detected, two or more herbicides are present in the GBR lagoon following wet season river discharge and, consequently, the area at risk to pesticide exposure increases when the additive risk is considered.
- Coral cores have been used to track change in material delivery to the GBR over long time periods. Coral Ba/Ca ratios in both short and long term coral core records display an increasing trend over time, particularly post European settlement (c. 1880) and in the last ~30 years, although peak values do not always coincide with river floods. In addition, the geochemical results from coral cores collected along a water quality gradient through the Whitsunday Islands have been useful in establishing local and regional patterns of terrestrial influence factors. These patterns correlate with an increased chronic terrestrial influence in the Whitsunday Islands. However, coral Y/Ca ratios typically lack long-term trends although peaks do generally relate to river discharge. Ba/Ca records from a long-lived coral (>100 years old) show a close correspondence with the generally annual river discharge peaks, providing further evidence that this approach provides a good proxy for changes in terrestrial inputs in the Wet and Dry Tropics.
- Recent publications presented for the Tully region (Kroon, 2009) showcased MTSRF supported research as a key component in the detailing of this ecosystem approach within the Tully catchment and marine region. In summary, this work included the estimate of the contribution of overbank (flood) flows to total pollutant loads, previously not taken into account in load estimates to the GBR (Wallace *et al.* 2009a,b). Maughan and Brodie (2009) provide a spatial model to visualise GBR exposure to land-sourced pollutants under current and changed land use regimes. Devlin and Schaffelke (2009) identified the transport and extent of pollutants in Tully flood plumes, and identified areas of high exposure and ultimately at high risk from the impacts of altered land use activities. This was reported as the number of marine biological systems that were frequently inundated by higher concentrations of sediment, nutrients and pesticides. The challenge to produce target estimates from catchment models with known levels of uncertainty, but robust enough for management purposes, was examined by Brodie *et al.* (2009a). The outcomes of these inter-related studies have contributed significantly to our capacity to understand and predict direct and indirect relationships between land use and management, impacts on water quality and flow on effects on marine biodiversity.

³ http://www.gbrmpa.gov.au/corp_site/key_issues/water_quality/water_quality_guidelines

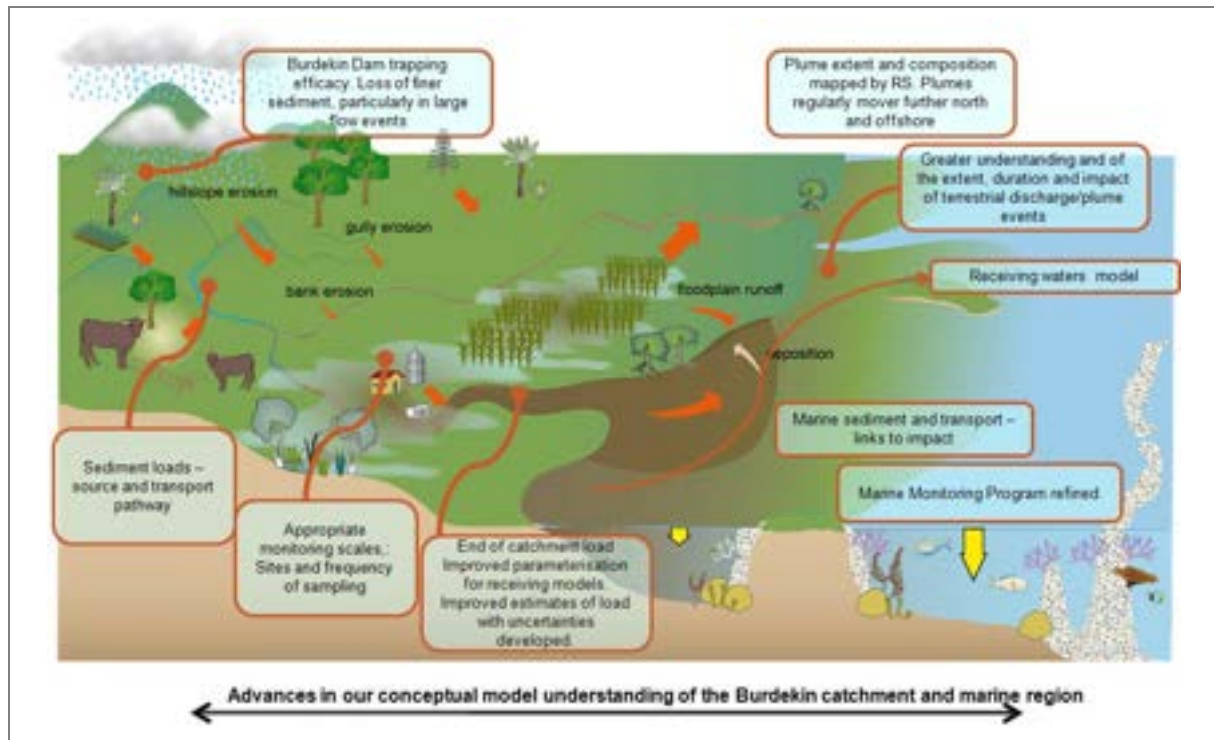


Figure i: Advances in conceptual understanding of water quality in the Burdekin catchment and marine region (base model developed by Prange, 2007).

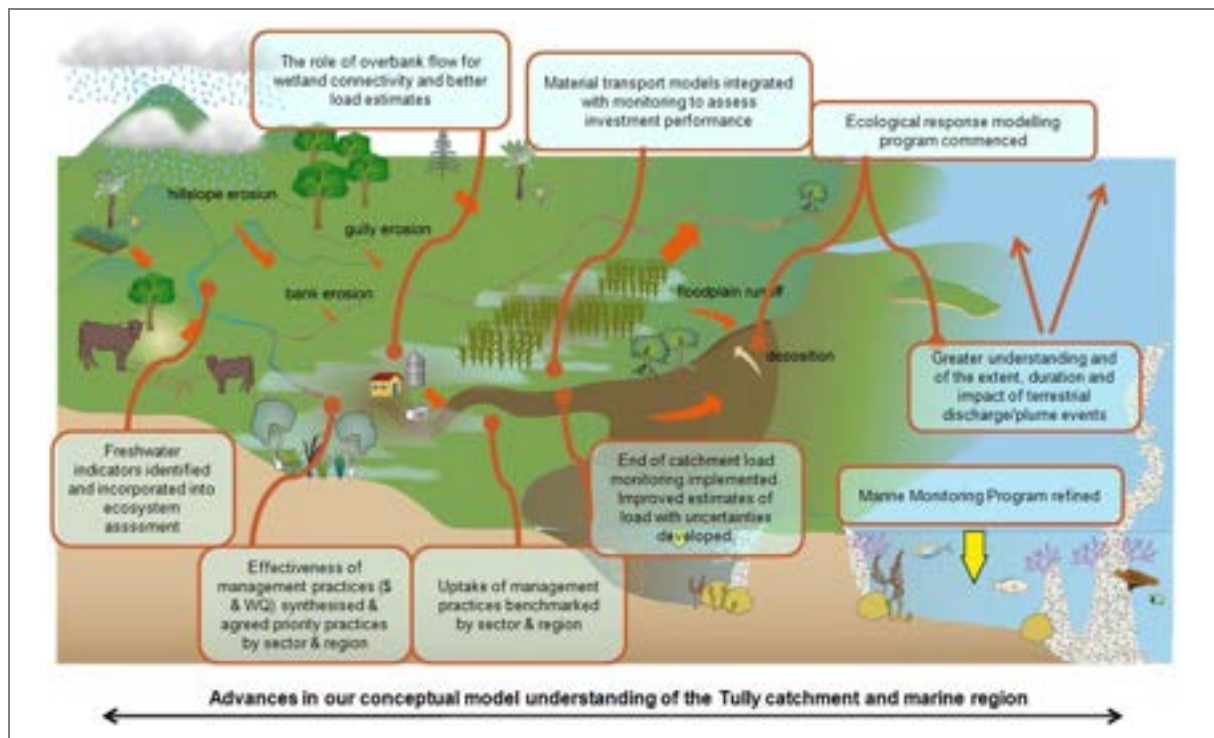


Figure ii: Advances in conceptual understanding of water quality in the Tully catchment and marine region (base model derived from Prange, 2007).

1. Introduction

1.1 Research timeline

Over the last thirty years, an increasing amount of research and monitoring effort has been devoted to documenting and understanding the nature and importance of water quality issues for the Great Barrier Reef (GBR). Attention has focused on land-based runoff as a primary source of pollutants to the GBR. Over the past decade, a number of conferences, reports, workshops and publications have consolidated our understanding of water quality issues for the GBR and thus provided the background for, and were instrumental in, the development of the Reef Water Quality Protection Plan (2003) ('Reef Plan') (Queensland Department of the Premier and Cabinet, 2003) and subsequent review in 2009 (Queensland Department of the Premier and Cabinet, 2009).

In the period since the release of the Reef Plan a number of large-scale research, monitoring and management programs with some emphasis on GBR water quality have been implemented. Further details on these programs can be found in the companion report by Waterhouse (2010). The establishment of the Australian Government's Marine and Tropical Sciences Research Facility (MTSRF) has pulled together much of this activity and, throughout the four years, integrated the research outputs for successful uptake in the management and protection of the GBR, demonstrating a greater awareness and understanding of GBR water quality issues.

The purpose of this synthesis and summary report is to review, synthesise and analyse the work carried out over the course of the MTSRF with respect to our current understanding of the relationships between catchment processes, pollutant loads delivered to instream environments (including wetlands and estuaries) and the marine environment, and the impacts on instream environments and the near shore environment. Key sources of information for this synthesis of catchment to reef water quality research are largely from MTSRF Theme 3 'Halting and Reversing the Decline of Water Quality'. Case studies from research programs in the Dry Tropics (Burdekin) and Wet Tropics (Tully) are presented to demonstrate more detailed understanding of catchment to reef relationships. We will highlight research results that have changed our understanding since the commencement of the MTSRF Research Program in 2006 and hence may be critical in revising the aims or priorities of water quality policy frameworks, including the Reef Plan, given the new understanding.

1.2 The MTSRF Water Quality Program (2006-2010)

The conceptual understanding of the catchment to reef processes in the GBR was well established in some locations prior to the commencement of the MTSRF water quality research program and was summarised in a synthesis report of current knowledge in 2006 by Brodie and others (2008a). However some significant gaps existed in the overall understanding of the connectivity between systems. Since then, major government funded initiatives have been put in place to restore, rehabilitate and protect catchment habitat through the adoption of best management practices and prioritisation of catchment activities that would reduce sediment, nutrient and pesticide runoff (Queensland Department of the Premier and Cabinet, 2009). These efforts have been informed by outcomes of research supported by the MTSRF Research Program in conjunction with other collaborative research programs such as the CSIRO Water for a Healthy Country Flagship⁴ GBR Program, the

⁴ <http://www.csiro.au/org/WfHC.html#2>

Australian Government Coastal Catchments Initiative⁵, the Reef Water Quality Partnership⁶ and various Queensland Government research programs.

Advances in our understanding of the connectivity between catchment scale processes and the GBR allow us to monitor and evaluate management targets defined in the updated Reef Plan (Queensland Department of the Premier and Cabinet, 2009) and Reef Rescue program (Australian Government, 2007). A large component of MTSRF funding over the past four years has supported work to identify water quality indicators that can be linked to land-based changes. The development of these water quality indicators and changes in our monitoring and evaluation techniques are covered in the companion report by Waterhouse (2010). These efforts provide the basis to the scientific framework for the Reef Rescue Marine Monitoring Program⁷ ('Reef Rescue MMP') and, more broadly, the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program⁸ (Paddock to Reef Program, Queensland Department of the Premier and Cabinet, 2010).

Research into the role of the catchment and, in particular, wetlands, in whole-of-system understanding for the GBR was introduced in 2006 as part of a need to achieve broader marine connectivity understanding and to further define the connections between the physical and biological components of both freshwater and marine ecosystems. As part of the MTSRF Research Program freshwater systems, riparian zones and wetlands have been studied as important ecological systems in their own right. MTSRF research also continued to investigate the impact and delivery of pollutants in the GBR, in particular, the impact of sediment from the Burdekin catchment and the impact of nutrients from Wet Tropics catchments. The role of wetlands, small tributaries and overbank flow have been investigated as part of our understanding of how these estuarine environments and processes affect the contaminant flow into marine waters. New load estimates have been calculated based on new statistical approaches and better understanding of where the water goes, and how much actually moves into offshore environments. A new approach to risk assessment has been considered based on higher data frequency and more accurate model parameterisation. Finally, in collaboration with the Reef Rescue MMP, a more accurate assessment of exposure to terrestrial discharge at a regional level has been made based on our knowledge of plume extent and concentrations and the frequency of inundation to biological systems. The past four years has seen a comprehensive integration of marine ecosystem status data with field data, high frequency logger data and the use of appropriate remote sensing techniques to give a better understanding of the changes in water quality and potential impacts from contaminant inputs on the GBR.

⁵ <http://www.environment.gov.au/water/publications/action/case-studies/nwqms-cci.html#cci>

⁶ http://www.regionalnrm.qld.gov.au/policies_plans_legislation/policies_strategies/gbr_water_quality/reef_partnership.html

⁷ <http://www.rrrc.org.au/mmp/index.html>

⁸ <http://www.reefplan.qld.gov.au/publications/paddock-to-reef.shtm>

2. Connections between catchment and reef ecosystems: a GBR wide perspective

Freshwater and marine systems have traditionally been identified as separate systems, where research on understanding components specific to the structure and functioning of these ecosystems has been undertaken. This section outlines the broad understanding of catchment to reef relationships and showcases the work that has contributed to this understanding, largely obtained through the efforts of the MTSRF.

2.1 GBR catchments

The GBR is a very large area to manage effectively, and to ensure appropriate information at the appropriate scale it is separated into large regionally defined areas. These areas are specific to catchments and are divided into six Natural Resource Management (NRM) regions (Figure 2.1), each with different land use, biophysical and socio-economic characteristics. The Cape York region is largely undeveloped and is considered to have the least impact on GBR ecosystems from existing land based activities. In contrast, the Wet Tropics, Burdekin Dry Tropics, Mackay-Whitsunday, Fitzroy and Burnett-Mary regions are characterised by agricultural land uses including sugarcane, grazing, bananas and other horticulture, cropping such as grains and cotton, mining and urban development, and contribute varying amounts of land-based contaminants to the GBR throughout the wet season. Current knowledge about the sources of contaminants from specific land uses, and priority source areas of contaminants is summarised in the companion report by Waterhouse and Brodie (2010). The research incorporated in that report builds on a large data source of past and current research and monitoring information, including advances in our understanding of GBR connectivity through research outputs of the MTSRF.

2.1.1 Management practices for the GBR

Managers need to know what agricultural activity is driving water quality change and how best to effect positive implementation of change within the agricultural activities. MTSRF funded research (van Grieken *et al.* 2010a) has identified the most important agricultural production systems in the GBR catchment from a profit point of view, more specifically their gross direct economic value (Access Economics, 2005) as well as from a water quality point of view (Queensland Department of the Premier and Cabinet, 2009). Researchers have also identified priority management actions for each identified production system and have prioritised them according to water quality improvement potential (van Grieken *et al.* 2010a).

There are three main production systems in the GBR catchment – sugarcane, grazing and horticulture (Table 2.1). Sugarcane is found on the coastal fringes of the Wet Tropics, Burdekin Dry Tropics and Mackay-Whitsundays. Grazing is predominantly found in the rangelands of the Burdekin and Fitzroy, while horticulture, a smaller system but still significant, is found throughout the Wet Tropics. Management actions recommended for each production system include nutrient, pesticide and soil management and actions to reduce soil erosion (Table 2.1). Identification of farming practice type in relation to water quality improvement potential is important for monitoring and modelling of improvement in agricultural practices and this has been completed through collaborations involving stakeholders and MTSRF funded researchers. As summarised in Table 2.2, there is a system of practices, structured from ‘Best’ (B) practices, which hold the highest potential for improving water quality, to ‘Dated’ (D) practices, having the lowest improvement potential. ‘Aspirational’ (A) management practices may further improve water quality but are currently under research and not yet commercially proven, and therefore are difficult to model accurately because improvement data is not yet available.



Figure 2.1: Great Barrier Reef catchments and Natural Resource Management (NRM) regions.
Source: GBRMPA.

Table 2.1: Key industries and priority management actions for each identified industry to address the issue of water pollution by nutrients, pesticides and sedimentation. Source: van Grieken *et al.* (2010a).

	Production System		
	Sugarcane	Grazing	Horticulture (bananas)
Dominant region(s)	Wet Tropics Burdekin Dry Tropics Mackay-Whitsundays	Rangelands (Burdekin Dry Tropics) Wet coastal grazing (Wet Tropics)	Wet Tropics
Priority actions	Nutrient, pesticide and soil management	Pasture (reduced stocking rates), riparian (frontage) and gully management	Nutrient, soil, insect/disease and irrigation management

Table 2.2: Description of practices that identify the type of management practices taken by the farmer. Source: van Grieken *et al.* (2010a).

Practice class		Description
A	Aspirational / Commercial viability, not yet proven	Aspirational / proof of concept, practice / farming system under research / scientifically sound but unproven
B	Best practice	Best practice / farming system currently available
C	Common practice	Current code of practice level of farming system
D	Dated practice	Dated and likely degrading practice / farming system

2.2 Water quality and catchment health

MTSRF funded research on freshwater systems (Pearson *et al.* 2010a, 2010b; Wallace *et al.* 2010a, 2010b; Karim *et al.* 2010a, 2010b, 2010c; Godfrey, 2009; Godfrey *et al.* 2010; Arthington and Pearson, 2007; Mackay *et al.* 2010; McJannet, 2007) has advanced our understanding of the ecological processes, linkages and interdependencies that govern the biodiversity, physical condition, ecological health, temporal trends and resilience of rivers and floodplain wetlands within GBR catchments. Water quality in GBR freshwater systems is a complex issue. Variables do not act alone and the actual impact of a contaminant is sometimes hard to predict. For example, hypoxia resulting from eutrophication may not occur if water is also very turbid due to the limitation of photosynthesis (Arthington and Pearson, 2007). Contamination of water can be very short-lived as a result of short-term flood events that may carry substantial loads of contaminants to the sea but have little long-term impact on streams. On the other hand, smaller but chronic inputs of contaminants have the greatest effect on ambient conditions – the conditions under which freshwater plants and animals spend most of their lives. Moreover, the biological environment is not only affected by water quality, it also may be the major determinant of water quality, especially in the warm waters of the tropics. Thus, hypoxia, which is a predominant water quality factor in some tropical waterways (Pearson *et al.* 2003), results from respiration by blooms of algae, macrophytes and microbes, which in turn are enhanced by high levels of nutrients, organic inputs, temperature and light. Thus, assessments of water quality need to take into account not only the physical and chemical nature of a water body, but also its biological state and dynamics.

The development of conceptual biophysical models to identify (i) appropriate indicators of waterway ecosystem health, and (ii) probable thresholds of concern, in terms of stressors (such as contaminant concentrations, instream habitat, riparian zone condition, hydrological change) and their effects on biodiversity and ecological processes has been carried out within MTSRF funded projects. In parallel, and closely linked to the assessment of floodplains and wetlands, is a model which focuses on the floodplain hydrological regime. These models are referred to below and included where appropriate.

The concept of river health considers not only the structural integrity of stream ecosystems, but also functional aspects such as the resilience of the system, that is, its capacity to resist or overcome disturbances (Rapport *et al.* 1998). However, prior to the MTSRF and CRC funded 'Catchment to Reef' programs, our understanding of the functional aspects of streams and how they are altered by land-use disturbance was largely conceptual, with broad-scale understanding of links between the physical, hydrological, biogeochemical and ecological processes. This is represented in the conceptual model of a freshwater system and the links between the different drivers and processes in Figure 2.2. Gross changes, such as the clearing of riparian vegetation and allochthonous production being replaced by autochthonous production, are relatively straightforward (Bunn *et al.* 1998; Pusey and Arthington, 2003), but measuring more subtle changes in ecosystem function is more difficult. As a result, studies of impact need to be able to detect structural changes in the biotic assemblages present and infer functional changes through shifts in functional guilds and food-web structure (Pearson and Penridge, 1987; Bunn 1995; Bunn *et al.* 1998; Rayner *et al.* 2010). Recent MTSRF funded research carried out in the Wet Tropics explicitly demonstrates the biotic responses to particular contaminants (Pearson and Connolly, 2000; Pearson *et al.* 2003; Connolly *et al.* 2004, 2007a, 2007b, 2007c; Connolly and Pearson, 2007) and identifies indicators and monitoring protocols to best identify river health and allow the detection of changes in biota and food-web structures (see Pearson *et al.* 2010a).

The complexity of water quality at a catchment scale is illustrated in Figure 2.3, which summarises some major variables and interactions affecting ambient water quality in Australian tropical fresh waters. The diagram is not exhaustive in its coverage, but serves to demonstrate the inter-related nature of water quality processes and measures and their influence on the ecosystem. It also indicates that these processes occur as part of a continuum from the catchment to the sea, determined by the flow regime of the system. In perennial streams and rivers, the inputs and outputs of surface waters are continuous, whereas in floodplain lagoons and riverine waterholes in the Dry Tropics, inputs and downstream outputs may be local or non-existent.

The short-term nature of large flow events in GBR catchments means that water quality during the event is less of an issue in inland waters. However, the power of floods in removing plants and animals and the connectivity they create between waterways and wetlands are the dominant considerations. The long-term exposure of biota to water quality characteristics during low flow means that water quality, at a local scale and during low flow conditions becomes a predominant issue in Dry Tropics streams and wetlands, and in some of the smaller and/or more peripheral streams of the Wet Tropics (Arthington and Pearson, 2007).

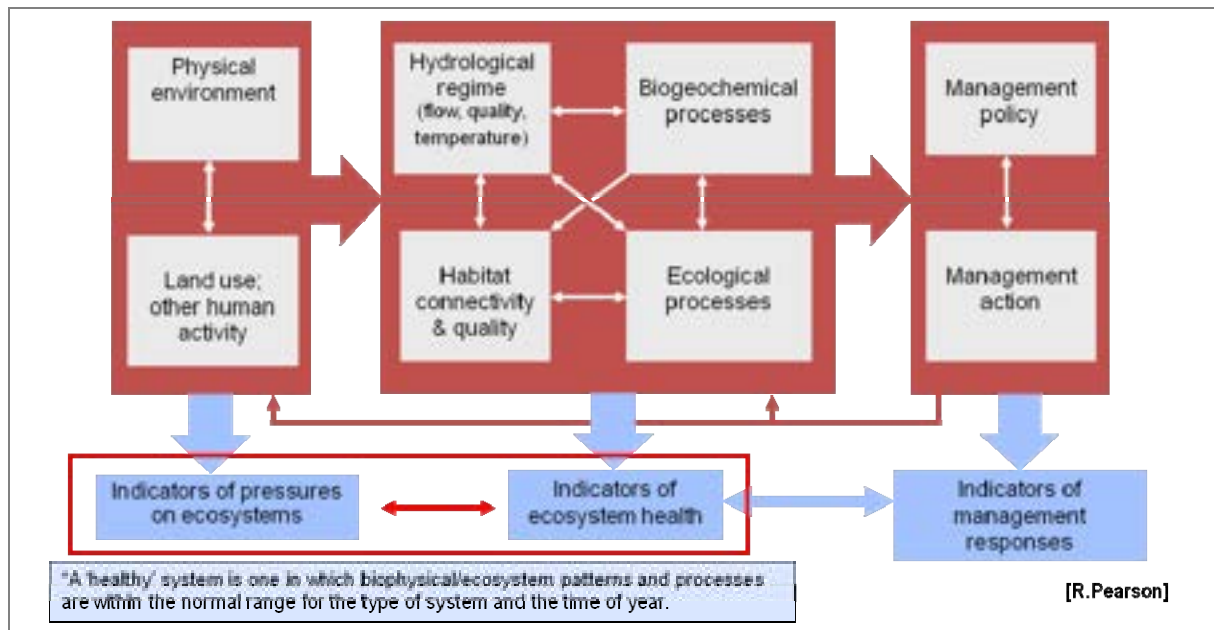


Figure 2.2: Conceptual model of a freshwater system and the links between the different drivers and processes. The model describes the cumulative impacts of changes in the physical environment, the hydrological regime, biogeochemical processes and habitat on ecology and subsequent management action. Different stages of the model may be monitored by the appropriate indicators. Source: Professor R. Pearson.

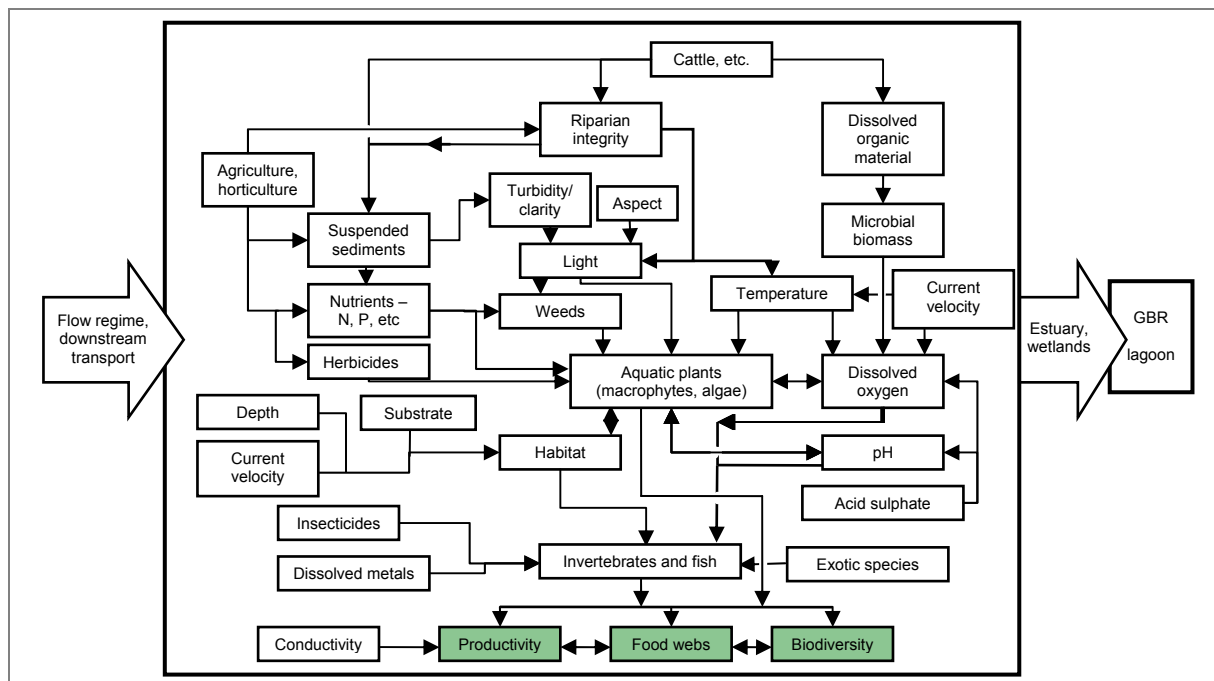


Figure 2.3: Biophysical interactions affecting ambient water quality in tropical agricultural landscapes. The large box (centre) represents typical processes and interactions in a stream reach, a discrete waterhole or a habitat within them. Large arrows represent flow-related connectivity. Shaded boxes represent different types of ecological response. Connectivity with the terrestrial landscape is implicit in some of the smaller boxes. Not all factors or interactions can be shown – for example, riparian integrity has a number of influences on ecological responses that are not indicated here; and there is no indication of the influences of urban infrastructure, mines, etc. Source: Professor R. Pearson.

Floodplains and wetlands are important physical and biological components and links in the aquatic continuum, providing unique and essential habitat and connectivity for specialist and wide-ranging biota (Wallace *et al.* 2010a). Yet very little is known about the hydrological dynamics of these systems, and how hydrology and physical connectivity influence aquatic habitats, water quality, biological diversity and ecosystem processes. The basic conceptual model of the exchange between freshwater and estuarine reaches with the adjacent floodplain is illustrated in Figure 2.4. These systems provide access and vital habitat for iconic species such as Barramundi, but they are typically poorly managed, highly impacted and, in the case of freshwater wetlands, severely depleted (~75% of such wetlands in GBR catchments having been lost to agricultural and other development) (Pearson *et al.* 2010b). Proper management will depend on understanding the biophysical relationships and connectivity in these systems. It is particularly important to develop a better and more predictive capacity to quantitatively link changes in land use, land management, water management or climate change to freshwater and marine ecosystem health.

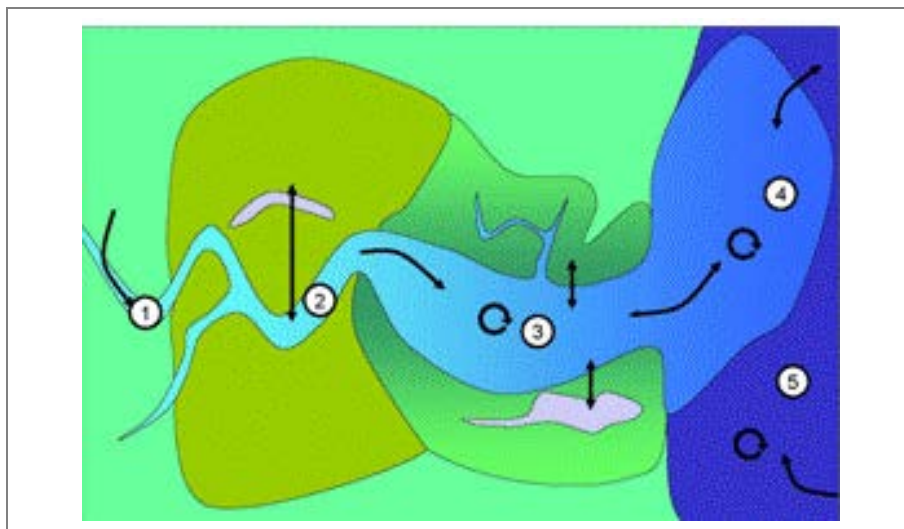


Figure 2.4: Schematic diagram of river flowing to the GBR lagoon, illustrating lateral exchange between freshwater and estuarine reaches with the adjacent floodplain. (1) Montane and slope river reaches; (2) Lowland reaches; (3) Estuarine reaches; (4) River plume extending seaward; (5) Coastal waters. Source: Wallace *et al.* 2007, adapted from Gehrke and Sheaves (2006).

Other advances in knowledge of freshwater ecosystems developed through MTSRF funded research are summarised below.

- The most serious factors affecting health in Wet Tropics streams and wetlands are changes to habitats (flow modification, loss of riparian vegetation, weed invasion, infrastructure). Water quality effects of agriculture are evident in streams and lagoons, particularly in systems that are not flushed throughout the year. In perennial streams (e.g. Russell-Mulgrave) and lagoons (e.g. Tully-Murray), dilution ameliorates impacts.
- The diverse aquatic invertebrate assemblages of Wet Tropics waterways are sensitive to habitat and water quality changes; they are good indicators of local ecological conditions.
- Wet Tropics waterways provide essential habitat, including nursery habitat, for unique fish assemblages, which are very good indicators of the physical characteristics, hydrological connectivity and ecological condition of sub-catchments and floodplain lagoons.

- The presence of alien fishes is a strong indicator of disturbance in the broader landscape, and an early warning indicator of the potential for further disturbance from increasing numbers of individuals and species (e.g. tilapia).
- Most native fish species avoid the alien ponded-pasture grass *Hymenachne amplexicaulis* and other alien plants that now occupy Wet Tropics waterways.
- Surrogate measures such as the Cassowary Coast Regional Council (previously Cardwell Shire) Floodplain Project Values and Threats scores can provide rapid assessment of waterway ecosystem health.
- Connectivity pathways of streams and wetlands to estuaries, coasts and the GBR are an essential feature of Wet Tropics waterways, and can be compromised by infrastructure, flow regulation, weeds and water quality barriers.
- Ecological condition of floodplain lagoons may serve as a powerful indicator of climate change, because these lagoons are vulnerable to sea-level rise and hydrologic alterations, especially loss of flood pulses, dry-season base flow and connectivity between rivers and wetlands.
- Hydrological connectivity between individual lagoons and the stream network is vital for maintenance of fish assemblages and normal waterway function.
- The special nature of the Tully-Murray wetlands as a unique assemblage of Wet Tropics habitats, with functional links to the GBR lagoon, needs to be specially recognised.

2.3 GBR pollutants: sources and pathways

2.3.1 Priority pollutants

A number of different pollutants are of concern in the GBR, where they are discharged from catchments. Each pollutant has different sources, pathways and impacts on GBR ecosystems. The three main pollutant categories of concern include nutrients, particularly dissolved inorganic nitrogen and phosphorus, sediments, with more concern on the finer, more mobile sediments which may potentially be influencing the long-term turbidity of inshore systems, and pesticides.

Ongoing research has allowed us to more clearly identify and quantify losses of suspended sediments, nutrients and pesticides from different land uses and land management practices, showing strong regional differences (as reported in Brodie and Waterhouse, 2009). There is a large difference in the pollutants of concern between the wet and dry catchments within the GBR catchment area. Due to the wetter climates and presence of intensive agricultural land uses (sugarcane and horticulture) and their associated fertiliser and pesticide usage, the Wet Tropics and Mackay-Whitsunday areas have been identified as regions of high nutrient and pesticide runoff concern (Furnas, 2003; Fabricius *et al.* 2005; Devantier *et al.* 2006; Brodie and Waterhouse, 2009), whilst the significantly larger Fitzroy and Burdekin River catchments (each ~135,000 km²), dominated by unimproved savannah/woodland rangeland grazing, are identified as considerable contributors of suspended sediment to the GBR lagoon (Mitchell and Furnas, 2001; Furnas, 2003; O'Reagain *et al.* 2005; Bainbridge *et al.* 2006a, 2006b; Packett, 2007; Packett *et al.* 2009; Waters and Packett, 2007; Brodie and Waterhouse, 2009). Our understanding of the transport and trapping of contaminants as they move through the GBR catchments from paddock to river mouth has also improved greatly through MTSRF funded research. For example, assessment of the timeframes likely to be involved in measuring signals of change in the priority pollutants for the GBR has been reported by Bainbridge and others (Bainbridge *et al.* 2009a; Brodie *et al.* 2007a) and is summarised in Table 2.3.

Considerable time lags in response to changed practices exist within the catchment, limiting the usefulness of monitoring activities in detecting changes in pollutant loads or concentrations at the end-of-catchment scale in the short term (<5 years) which may result from on-ground incentive programs. Due to lag times in response to changed management practices and a noisy water quality signal associated with inter-annual flow variability, it would take more than a decade to detect reductions in pollutant loads, which is outside the current targets of the Reef Rescue timeframe. In addition, the level of uncertainty in the calculation of pollutant loads can equal or exceed the proposed resource condition or pollutant load targets. Hence, the only way to assess the effectiveness of management actions on water quality in the short term in such a system is to utilise modelling tools (e.g. SedNet or WaterCAST models) to predict material transport and delivery and management scenario forecasting. Receiving water models, such as ChloroSim are also required to relate these end-of-catchment pollutant loads to ecosystem response (Wooldridge *et al.* 2006). Water quality guidelines (trigger values) for the GBR can be then used within these receiving water models to revise end-of-river targets (Brodie *et al.* 2009a).

Comprehensive studies undertaken in the Burdekin and Tully catchments through the MTSRF are described in detail in Sections 4 and 5. These improvements in knowledge of pollutant sources and transport processes have led to significant advancements in our ability to estimate pollutant loads with greater confidence across the GBR. The most recent evidence related to specific pollutants, the main sources and delivery pathways is summarised below and is also discussed in the companion report by Waterhouse and Brodie (2010).

Table 2.3: Timeframes for water quality trends/signals to be detected for three parameter examples at spatial scales from paddock to reef as a result of management actions implemented. Source: Bainbridge *et al.* (2009a). Note: The Burdekin Rangelands (SS), lower Burdekin (DIN) and Tully floodplain (herbicides) have been used as examples to demonstrate varying scales.

Management Actions / Remedial Activity	Water Quality Parameter		
	Suspended Sediment	Dissolved Nitrogen	Herbicides
	Erosion control mechanisms for grazing lands <i>e.g. riparian fencing and wet season spelling</i>	Reduction of fertiliser use in cropping lands <i>e.g. implement The Six Easy Steps Approach⁹</i>	Minimise/optimize pesticide use through new technologies <i>e.g. shielded sprayers, control traffic</i>
Timeframe of water quality trends/signals being detected at different spatial scales			
Paddock/ Plot Scale	Change likely to be detected after two to three wet seasons <i>e.g. Virginia Park Station</i>	Months to three years, dependent on the nitrogen stored in the system (e.g. soil, organic matter) <i>e.g. Burdekin River Irrigation Area paddock</i>	Months to one year, dependent on previous usage and residuals in the system <i>e.g. Tully paddock</i>
Local Scale <i>e.g. immediate drainage line/ local waterway</i>	Likely to be detected within five to ten years depending on system noise <i>e.g. Weany Creek</i>	Likely to be detected within one to three years, depending on rate of adoption within local area and system noise <i>e.g. local cane drain</i>	Likely to be detected within one year due to relatively short half life (i.e. diuron half life in soil is 90 days, and likely complete life less than two years) <i>e.g. local cane drain</i>
Sub-catchment Scale	Greater than ten years, even for major scale land management interventions across the sub catchment <i>e.g. Fanning River</i>	If sugarcane is dominant land use in catchment and management change is widely adopted then could expect to measure change <10 years, particularly if detailed pre-monitoring data is available <i>e.g. Upper Barratta Creek</i>	If sugarcane is dominant land use in catchment and management change is widely adopted then could expect to measure change within two years, particularly if detailed pre-monitoring data is available <i>e.g. Davidson Creek</i>
End-of-catchment Scale	Dilution of signal as only small percentage of total catchment area under improved management at any one time, and hydrological variability or noise is high. Likely more than fifty years (major erosion control management intervention across the Burdekin) <i>e.g. Burdekin River (Inkerman)</i>	If sugarcane is dominant land use in catchment and management change is widely adopted then could expect to measure change < 10 years, particularly if detailed pre-monitoring data is available <i>e.g. Barratta Creek (Bruce Highway)</i>	Change detected less than two years, however may be dilution effect depending on amount of cane in catchment, and proportion of uptake by the industry within this catchment <i>e.g. Tully River (Euramo)</i>
Estuarine and Marine Scale <i>e.g. coastal waters within adjacent bay</i>	Limited likelihood of detecting signal from this management action due to size of catchment. Likely more than fifty years before change in turbidity <i>e.g. Upstart Bay</i>	Likely to detect change in chlorophyll from this management action (major nitrogen fertiliser reduction across the lower Burdekin sugar lands) less than twenty years, with variability due to other sources of nutrients (e.g. Burdekin plume), seasonal variations in nitrogen cycling and sea water mixing <i>e.g. Bowling Green Bay</i>	Changes likely to be detected within two years in the flood plume, however signal may be difficult to detect if the coastal waters are also influenced by larger river flood plumes (e.g. Herbert or Murray Rivers) <i>e.g. Dunk Island and Family Islands Group</i>

⁹ BSES Ltd Best-Practice Nutrient Management: The Six Easy Steps Approach (<http://www.bses.org.au/InfoSheets/2008/IS08006.pdf>)

Nutrients

- Contemporary estimates of nitrogen loads suggest that the total N discharge to the GBR has increased from 14,000 tonnes per year in pre-development times (prior to 1850) to a current discharge of 58,000 tonnes per year, a four-fold increase (Kroon *et al.* 2010; Brodie *et al.* 2009b; Furnas, 2003; McKergow *et al.* 2005a).
- The predominant form of nitrogen delivered to the GBR has also changed. Discharge of nitrogen from natural landscapes is predominantly in the form of dissolved organic nitrogen (DON) (Harris, 2001; Brodie and Mitchell, 2005). This is still the case in undisturbed forest stream runoff in the GBR catchment area (Brodie and Mitchell, 2006), however, nitrogen discharge from agricultural and urban lands is now dominated by dissolved inorganic nitrogen (DIN = nitrate + nitrite + ammonium) derived from fertiliser and sewage wastes, and particulate nitrogen (PN) derived from soil erosion (Brodie and Mitchell, 2005). The shift from a predominantly DON discharge in pre-1850 times to a predominantly (bioavailable) PN and DIN discharge in modern times has important consequences for the effects of discharged nitrogen (Fabricius, 2005). Nitrate has a higher risk as it is 'bioavailable' for in-stream uptake (e.g. weed growth) and for downstream uptake (e.g. promoting algae on inshore coral reefs).
- Change in land use has also led to the increase in nitrate discharge in some individual catchments being much larger relative to the increase in total nitrogen discharge (e.g. estimated to be six times in the Johnstone River (Hunter and Walton, 2008) and ten times in the Tully River (Armour *et al.* 2009). The larger increases in the inorganic nitrogen fraction are associated with intensive fertiliser use on sugarcane and banana crops in these catchments.
- A strong relationship exists between the areas of nitrogen-fertilised land use in a catchment and the mean nitrate (assumed here as $\text{NO}_2 + \text{NO}_3 = \text{NO}_x$) concentration during high flow conditions, implicating fertiliser residues as the primary source of nitrate (Mitchell *et al.* 2006; Faithful *et al.* 2005, 2006; Mitchell *et al.* 2009; Kroon and Brodie, 2009; Armour *et al.* 2009; Bainbridge *et al.* 2009b). Elevated stream concentrations of nitrate indicate fertiliser application above plant requirements in sugarcane and bananas. Nitrate has only a small natural occurrence in north Queensland from pristine sources (Brodie and Mitchell, 2006), slightly more elevated levels from the lightly grazed Normanby River catchment (Furnas *et al.* 2006) but much higher concentrations from cropping or horticulture activities.
- Anthropogenic loads of contaminants can also be estimated from modelled results of pre-European estimates and current loads defined using monitoring and modelling. Based on the most recent estimates of DIN loads, it is estimated that the total current DIN load to the GBR is approximately 13,500 tonnes per year and the total anthropogenic load to the GBR is approximately 6,900 tonnes per year (approximately 52% of the total load). Of the 6,900 tonnes anthropogenic DIN load to the GBR, approximately 6,150 tonnes is estimated to be derived from sugarcane (approximately 89% of the anthropogenic load), 730 tonnes from horticulture (11% of the anthropogenic load) and 60 tonnes (approximately 1% of the anthropogenic load) other land uses including urban and other crops (Kroon *et al.* 2010).
- Elevated concentrations of dissolved inorganic phosphorus are also related to fertiliser application above plant requirements in intensive cropping and to locally specific soil characteristics (Brodie *et al.* 2008b, 2008c).
- Analysis of data on fertiliser use, loss potential and transport has ranked fertilised agricultural areas of the coastal Wet Tropics and Mackay-Whitsunday as the hot-spot areas for nutrients (mainly nitrogen) that pose the greatest risk to the GBR (Brodie *et al.* 2009c).

Sediment

- Changes in loads and concentrations of suspended sediment loads to the GBR since European settlement have been estimated using models such as SedNet and ANNEX (e.g. Brodie *et al.* 2003; Cogle *et al.* 2006) and other models (e.g. Furnas, 2003) at the catchment and sub-catchment scale. Results from such modelling studies indicate that in many rivers suspended sediment loads (and hence mean concentrations) may have increased by a factor of between five and ten since European settlement.
- Most sediment originates from grazing lands of the Dry and sub-tropics (Brodie *et al.* 2009c). The influence of land use on sediment loads is now well known at a regional scale but more work is required to identify sources at finer scales, due to variability associated with hillslope, streambank and gully erosion within individual catchments (Brodie *et al.* 2009c).
- In the Dry Tropics, high suspended sediment concentrations in streams are associated with rangeland grazing and locally specific catchment characteristics, whereas sediment fluxes are relatively low from cropping land uses due to improvements in management practices over the last twenty years (Dight, 2009). In the Wet Tropics, sediment fluxes are comparatively lower due to high vegetation cover maintained throughout the year from high and year-round rainfall and different land management practices (Brodie and Waterhouse, 2009) from Dry Tropics regions within industries such as beef grazing.
- Losses of suspended sediment from sugarcane cultivation have been shown to be relatively low, reflecting some fifteen years of improved soil conservation measures including green cane harvesting, trash blanketing and reduced tillage (Rayment, 2003; McJannet *et al.* 2005; Bainbridge *et al.* 2006a, 2006b, 2007, 2008; Rohde *et al.* 2006; Faithful *et al.* 2006; Brodie and Waterhouse, 2009). However, there is still some evidence of elevated erosion in sugarcane cultivation areas compared to forested areas as shown from the results of Hateley (2007), who identified the sources of sediments collected from waterways draining different land uses within the Tully River catchment. In contrast, rangeland beef grazing lands lose large quantities of suspended sediment through erosion associated with low vegetation cover (e.g. Brodie *et al.* 2003; McKergow *et al.* 2005b; O'Reagain *et al.* 2005; Bainbridge *et al.* 2006a, 2006b; Bartley *et al.* 2006, 2007; Dougall *et al.* 2006; Fentie *et al.* 2006; Hateley *et al.* 2006).
- The fine sediment fraction is considered to be the most important fraction because fine particles have many times greater collective surface area than coarser particles to carry pollutants (e.g. adsorbed pesticides, trace metals and nutrients), fine particles may be carried considerable distances in the marine environment, and fine particles are capable of producing higher turbidity levels in the freshwater and marine environment (Wolanski *et al.* 2008; Bainbridge *et al.* 2009a).
- A second-order effect of human development is the loss of natural trapping from land clearing and grazing, reduced vegetation complexity (e.g. forest to crops), loss of freshwater lagoons and loss of riparian vegetation. This can be considered to be an exacerbating mechanism that decreases local uptake of nutrients and sequestration of sediments, thereby further increasing the runoff of these materials from the land (Arthington and Pearson, 2007; Pearson *et al.* 2010a, 2010b).

Pesticides

- The presence of pesticide residues, especially herbicides, is widespread in waterbodies of the GBR region, including streams, wetlands, estuaries, coastal and reef waters (e.g. Packett *et al.* 2005; Rohde *et al.* 2006, 2008; Lewis *et al.* 2007a, 2009b). Residues commonly detected include atrazine, diuron, ametryn, hexazinone and tebuthiuron.

Although most of the concentrations are very low, these substances would not have been present at all before agricultural development of the catchments.

- Recent work on pesticide monitoring in paddocks, rivers and the marine environment has progressed our understanding of the extent and persistence of pesticides in freshwater and marine areas (Lewis *et al.* 2009b; Bainbridge *et al.* 2009b; Shaw *et al.* 2010).
- The loss of herbicide residues – particularly diuron, atrazine, hexazinone and ametryn – from sugarcane cultivation has been firmly established (Packett *et al.* 2005; Rohde *et al.* 2006; Lewis *et al.* 2007a; Stork *et al.* 2007) in the dominant sugarcane regions from Bundaberg to Tully.
- Concentrations of pesticides in rivers and streams are highest in areas of intensive agricultural activity including sugarcane but also from grazing lands (tebuthiuron) (Lewis *et al.* 2009b; Brodie and Waterhouse, 2009). Highest concentrations of pesticides in marine waters are likely to be highest off these rivers with high intensive agriculture. This is supported by the data collected within the Reef Rescue MMP (e.g. Paxman *et al.* 2009; Johnson *et al.* 2010).
- Concentrations of pesticides are variable over time and space and thus it can be difficult to define the full impact of pesticide concentrations at any given sampling point. The complex transport mechanisms and variability within receiving waters can make it difficult to define the overall risk area within GBR waters. Work on the definition and areal extent of risk and impact from pesticides is ongoing.

2.3.2 Pollutant load estimations

Pollutant load estimations for the GBR have been completed in several studies since the late 1990s although, in many cases, these estimates have been highly uncertain due to an inadequacy of sampling sites in some locations and application of inconsistent methods over time and between programs (Brodie *et al.* 2009a). As the estimate of pollutant loads and the detection of an appropriate reduction are a major component of the Reef Plan and Reef Rescue targets, it has been an important outcome of the MTSRF to deliver advancements in our ability to estimate contaminant loading into the GBR. For example, hydrological modelling by Wallace, Karim and others (e.g. Wallace *et al.* 2008a, 2008b, 2009a, 2009b, 2010a; Karim *et al.* 2010a, 2010b, 2010c) has identified the importance of over-bank flow in pollutant delivery to the GBR. Assessment of the trapping efficiency of the Burdekin dam by Lewis and others (e.g. Lewis *et al.* 2009a) has highlighted the need to understand the impact of manmade structures such as dams on the resultant load calculations. Statistical and modelling work by Kuhnert and others (e.g. Kuhnert *et al.* 2007, 2008, 2009, 2010; Kuhnert and Henderson, 2010) has shown the importance of including variability of the system in the calculation of pollutant load estimates. Further work by Brodie and others (e.g. Brodie *et al.* 2009b, 2009c; Kroon *et al.* 2010) has incorporated a number of these advancements to present 'best estimates' of pollutant loads for each of the GBR catchments. Integration of these sources, plus a number of regionally based measurements and literature has allowed the ranking of catchment risk relative to anthropogenic load, potential exposure and ecosystem status (Brodie and Waterhouse, 2009). This information has fed directly into the implementation of the *Great Barrier Reef Protection Amendment Act* (2009)¹⁰ introduced by the Queensland Government in 2009. A comprehensive overview of the outcomes of these studies is provided in the companion report by Waterhouse and Brodie (2010); the key points relevant to this report are summarised below.

¹⁰ <http://www.legislation.qld.gov.au/LEGISLTN/ACTS/2009/09AC042.pdf>

Brodie and others (2009b) collated available load data for the 33 major river basins discharging to the GBR and calculated the best estimate of the current end-of-catchment loads for the following parameters: suspended sediment, dissolved inorganic nitrogen, dissolved organic nitrogen, particulate nitrogen, dissolved inorganic phosphorus (filterable reactive phosphorus), dissolved organic phosphorus and herbicides. A summary of the report results for current best estimates of pollutant loads discharged to the GBR and the modelled 'natural' loads is provided in Table 2.4. These results have subsequently been improved through the efforts of Brodie, Lewis (pesticides) and most recently, Kroon and others (Brodie and Waterhouse, 2009; Brodie *et al.* 2009c; Kroon *et al.* 2010).

MTSRF research has also delivered improved techniques of pollutant load estimations. For example, Kuhnert *et al.* (2009) successfully modelled the factors that drive load variability over temporal and spatial scales. This approach offers a statistically robust approach to load estimates (Table 2.5). Kuhnert *et al.* (2009) identify a number of different stages of flow and show that the characteristics of the flow event are important in the estimation of load measurements. The generalised rating curve approach is novel as it seeks to represent a number of important system processes for GBR catchments to account for expected or implied system behaviours:

1. **First Flush**, the first significant channelised flow in a water year accompanied by high concentrations (represented as a percentile of flow and used in the calculation of other system processes);
2. **Rising/Falling Limb**, which allows higher or lower concentrations on the rising limb when runoff energies are higher and sediment supply may also be high. This is usually represented at shorter time scales than exhaustion, which is parameterised for between-event variations. This covariate is based on the first flush defined for that period;
3. **Exhaustion**, representing the limited supply of sediments and nutrients due to previous events, represented by a discounted flow term;
4. **Hysteresis**, representing complex interactions between flow and concentration with strong historical effects and dependence captured by non-linear terms for flow and incorporating hydrological processes; and
5. **Overbank Flow**, described as flow that goes overbank in flood events, which is not well recorded by standard river gauges. This is work currently investigated by Wallace and others (2010a, 2009a, 2009b) and has not yet been included in the statistical model.

In high-flow events, there can be a significant error in load estimates based on the volume of overbank flow that was not accounted for in traditional load calculations. For example, the Tully and Murray Rivers break their banks and the floodwaters merge and flow to the ocean as a large sheet of water many kilometres wide. MTSRF funded research by Wallace and others (2009b) showed that during the thirteen floods between 2006 and 2008, the Tully River gauge at Euramo recorded only 36-88% of the flood discharge, while the Upper Murray gauge recorded only 11-27% of the flood discharge. Furthermore, current ocean sediment and nutrient loads are based on concentrations measured within the rivers, yet until the MTSRF funded project was initiated, the sediment and nutrient concentrations in overbank flood waters were not known. Wallace and others (2010a, 2009a, 2009b) have presented new estimates of flood discharge that include overbank flows combined with direct measurements of sediment and nutrient concentrations in flood waters to calculate the loads of sediment and nutrient delivered to the ocean.

This combined work on load estimation has advanced our reporting of end-of-catchment loads to the Reef Rescue MMP, particularly for the Burdekin and Tully catchments (Kuhnert and Henderson, 2010). Revised load estimates have recently been prepared in a collaborative project between the CSIRO, Australian Centre for Tropical Freshwater Research (ACTFR) (supported by MTSRF) and the Queensland Department of the Premier and Cabinet for the Paddock to Reef Program (Kroon *et al.* 2010).

Table 2.4: Comparison of the natural and current loads estimated for 33 major river basins draining into GBR waters. Source: Brodie *et al.* (2009b).

ID	Name	DIN natural	DIN current	DON natural	DON current	PN natural	PN current	DIP natural	DIP current	PP natural	PP current	DOP natural	DOP current	TSS natural	TSS current	Total PSII nat	Total PSII cur
	Unit	tonnes per year												'000 tonnes per year		kg per year	
1	Jacky Jacky Creek	133	250	149	200	10	400	12	10	14	60	14	10	17	100	0	0
2	Olive-Pascoe Rivers	236	500	268	400	27	1,000	10	10	38	200	25	20	57	330	0	0
3	Lockhart River	85	200	167	200	8	300	5	6	14	70	13	10	28	150	0	0
4	Stewart River	69	150	85	120	7	200	3	3	10	40	8	10	22	110	0	0
5	Normanby River	517	900	544	800	85	2,000	10	20	72	400	53	50	184	1,100	0	0
6	Jeannie River	122	250	140	150	23	500	5	6	19	70	13	10	66	240	0	0
7	Endeavour River	119	250	186	250	18	500	3	5	23	100	16	10	70	370	0	0
8	Daintree River	123	200	426	250	22	300	9	10	25	80	29	15	45	240	0	20
9	Mossman River	17	50	46	50	4	100	1	2	3	20	3	5	7	80	0	70
10	Barron River	59	200	152	150	13	150	3	5	12	40	11	25	25	100	0	180
11	Mulgrave-Russell Rivers	166	1,000	566	500	23	700	14	20	32	250	38	50	41	210	0	260
12	Johnstone River	243	850	825	400	26	1,000	26	20	38	300	56	60	41	260	0	440
13	Tully River	172	600	523	250	14	450	17	20	25	90	36	15	24	120	0	250
14	Murray River (QLD)	72	300	158	150	4	170	8	10	8	40	12	8	9	50	0	170
16	Herbert River	257	700	442	500	47	700	10	20	47	200	36	30	107	540	0	450
17	Black River	26	60	43	50	8	300	1	5	6	70	4	3	30	180	0	10
18	Ross River	16	70	17	50	6	200	0	8	3	50	2	4	20	190	0	0
19	Haughton River	42	150	42	100	7	400	2	15	6	120	4	7	29	200	0	400
20	Burdekin River	979	1,300	1,020	1,800	165	9,000	16	180	165	1,800	100	50	478	4,600	0	100

Connections between catchment and reef ecosystems: Wet and Dry Tropics case studies

ID	Name	DIN natural	DIN current	DON natural	DON current	PN natural	PN current	DIP natural	DIP current	PP natural	PP current	DOP natural	DOP current	TSS natural	TSS current	Total PSII nat	Total PSII cur
	Unit	tonnes per year												'000 tonnes per year		kg per year	
21	Don River	33	60	33	100	9	500	1	10	6	150	3	6	39	590	0	0
22	Proserpine River	83	450	111	160	9	200	4	50	19	50	10	10	45	50	0	1,000
39	O'Connell River	125	700	152	250	18	400	4	50	31	150	14	15	99	150	0	250
40	Pioneer River	84	600	111	120	10	800	2	100	22	300	10	50	50	280	0	1,500
24	Plane Creek	95	250	103	200	11	200	4	50	20	100	10	15	54	60	0	800
25	Styx River	23	50	24	150	5	600	1	20	5	200	2	10	25	250	0	0
41	Shoalwater Creek	41	50	41	150	5	500	1	20	7	150	4	10	22	100	0	0
26	Water Park Creek	41	80	41	130	4	400	3	20	6	100	4	10	10	100	0	0
27	Fitzroy River (QLD)	607	1,500	634	2,500	70	8,000	7	300	75	3,000	62	70	275	3,400	0	1,200
29	Calliope River	20	50	21	50	5	300	0	5	4	100	2	3	20	200	0	0
30	Boyne River	38	70	39	30	13	120	1	5	9	30	4	3	41	50	0	0
31	Baffle Creek	96	150	97	150	10	500	4	10	14	120	10	8	35	300	0	20
32	Kolan River	48	100	49	50	7	200	1	10	8	40	5	4	22	100	0	100
33	Burnett River	175	400	191	300	30	1,000	3	35	29	300	18	50	99	200	0	300
34	Burrum River	61	150	69	100	3	250	6	10	7	70	6	6	9	100	0	400
35	Mary River (Qld)	280	450	314	400	33	1,000	9	50	53	300	30	25	98	1,200	0	150
Total		5,303	13,090	7,829	11,210	759	33,340	206	1,120	875	9,160	667	687	2,243	16,300	0	8,070

Table 2.5: Steps in the estimates of loads using the statistical approach outlined by Kuhnert *et al.* (2009).

Step	Process
1. Estimation steps for flow	Prediction of flow at regular time intervals using a time series model such that the model captures all of the peak flows. The predicted flow is then matched to concentration sampling times and used only when flow was not collected at that specific time interval.
2. Estimation steps for concentration	Prediction of concentration using a generalised additive model (GAM) that incorporates all important covariates in an attempt to capture the underlying hydrological processes concerned with the flow and transportation of sediment and nutrient loads.
3. Estimation of the load	An estimate of the load in the third step using the predicted concentration and predicted flow and incorporating a unit-conversion constant for time interval used.
4. Calculation of the standard error of the load	Standard errors are computed incorporating both measurement error and errors due to the spatial location of sampling sites.

2.4 Water quality and marine health

2.4.1 Impacts on marine ecosystems

Changes in the physical, chemical or biological state of the environment determine the quality of ecosystems and the scale of impact on the overall 'health' of a system. In other words, changes in the state may have environmental or economic 'impacts' on the functioning of ecosystems, their life supporting abilities, and ultimately on human health and the economic and social performance of society. Over the last decade, targeted research and monitoring have identified a number of impacts and changes in the GBR which could have substantive impacts on the long-term health of GBR waters and their biota.

There is unequivocal and growing evidence that parts of the GBR are now facing eutrophic conditions, with nutrient enrichment (Fabricius, 2007; Johnson *et al.* 2010; Brodie *et al.* in press), high phytoplankton biomass (Furnas *et al.* 2005), potential changes in the phytoplankton food web structure, increased competition at the coral reef level (Thompson *et al.* 2010; Devantier *et al.* 2006), shifts in trophic food webs observed in the proliferation of COTS in areas which are regularly inundated by anthropogenic loads (Brodie *et al.* 2005) and increases in long-term turbidity related to the export of finer sediment out of the large Dry Tropic regions (Wolanski *et al.* 2008). Our understanding of the water quality drivers is outlined by a simple conceptual diagram (Figure 2.5) which outlines how changes in the key components of water quality, i.e. sedimentation, turbidity and light attenuation, and nutrients, influence biotic structure.

In addition, there is growing consensus in the literature that the largest (continuing) impact of nutrient enriched runoff events is associated with the eventual fate of this nutrient source (Furnas *et al.* 2005, in press; Fabricius, 2005). After initially being diluted and dispersed in the water column, the nutrients are rapidly taken up by phytoplankton and, eventually, after being recycled (once or many times) through pelagic food webs, are converted into other forms of organic matter (DOC, DON, DOP, detritus) (Alongi and McKinnon, 2005; Furnas *et al.* 2005). This organic matter is ultimately transformed into forms (e.g. marine snow) that may be deposited on benthic communities, such as coral reefs, and influence their structure, productivity, and health (see for example, Anthony and Fabricius, 2000; Fabricius and Wolanski, 2000).

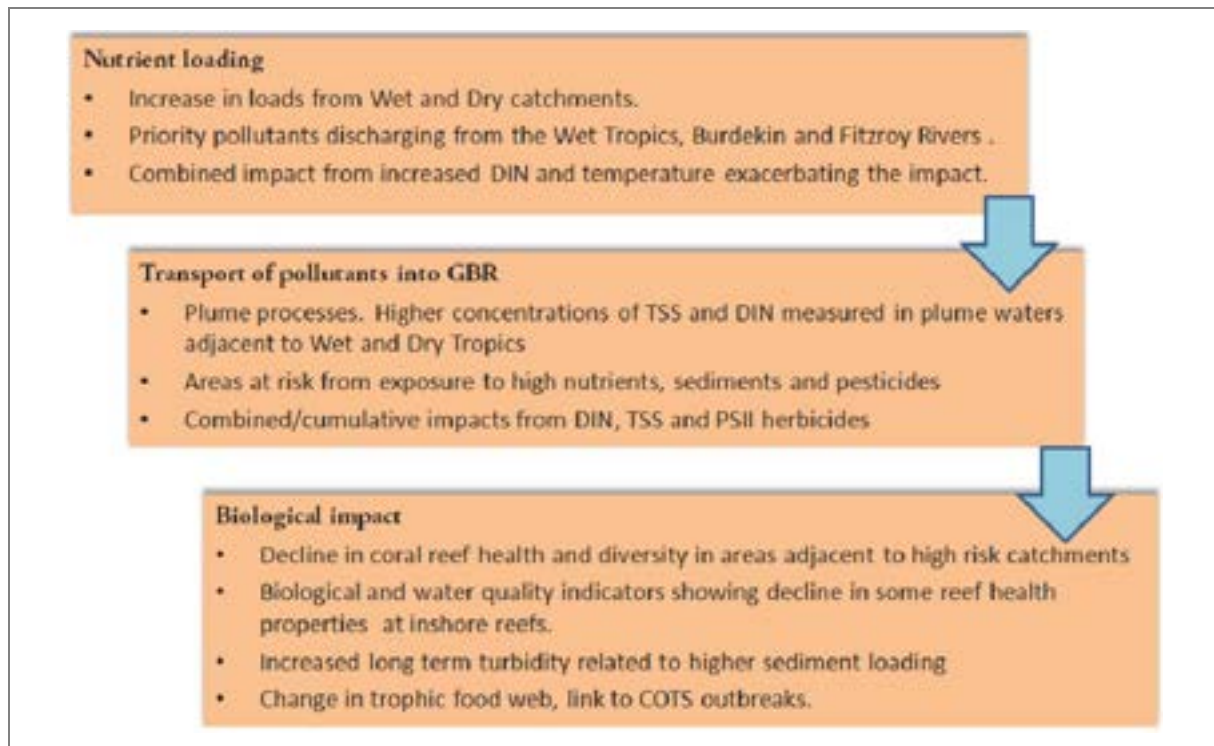


Figure 2.5: Burden of proof linking nutrient enrichment to negative biological impact.
Source: Devlin *et al.* (2010).

2.4.2 Impact on GBR ecosystems: coral reefs

Our understanding of the effects of land-sourced contaminants on GBR species and ecosystems has been expanded enormously since 2003. Strong links between coral reef health and water quality conditions have been shown (Fabricius, 2005; Fabricius *et al.* 2005), including sedimentation stress (Philipp and Fabricius, 2003; Weber *et al.* 2006) and the effects of muddy marine snow (Fabricius *et al.* 2003; Wolanski *et al.* 2003a). Devantier *et al.* (2006) highlighted significant spatial differences in the patterns of coral cover and diversity across the GBR, identifying the areas adjacent to the Wet Tropics having significantly lower reef diversity than areas to the north or south. It is most likely that adjacent catchment pressures were contributing to this decline in reef health. Changes in water quality drivers can start to be directly related to known and cited biological impacts measured in the GBR, as shown in the conceptual model developed by Fabricius and others (see Figure 2.6).

Environmental conditions clearly influence the benthic communities found on coastal and inshore coral reefs of the GBR. These reefs differ markedly from those found in clearer, offshore waters (e.g. Done, 1992; Wismer *et al.* 2009). Within the inshore zone there appears to be a threshold beyond which environmental conditions are not suitable for coral reef development, indicated for example by the historical lack of corals on hard substrates in some areas. The processes shaping biological communities, however, are complex and variable depending on spatial and temporal scales and are likely to include local interactions of various factors such as water quality, climate change and physical disturbance. This complexity may obscure the relationships between coral communities and specific environmental conditions and has hampered the quantification of anthropogenic impacts on inshore coral communities. However, environmental conditions such as water quality can explain some of the considerable variation in coral community composition (Fabricius *et al.* 2004; van Woesik *et al.* 1999; Cooper *et al.* 2009; Thompson *et al.* 2010) and most likely reflect species-specific environmental tolerances (e.g. Anthony and Connolly, 2004; Anthony and Fabricius, 2000).

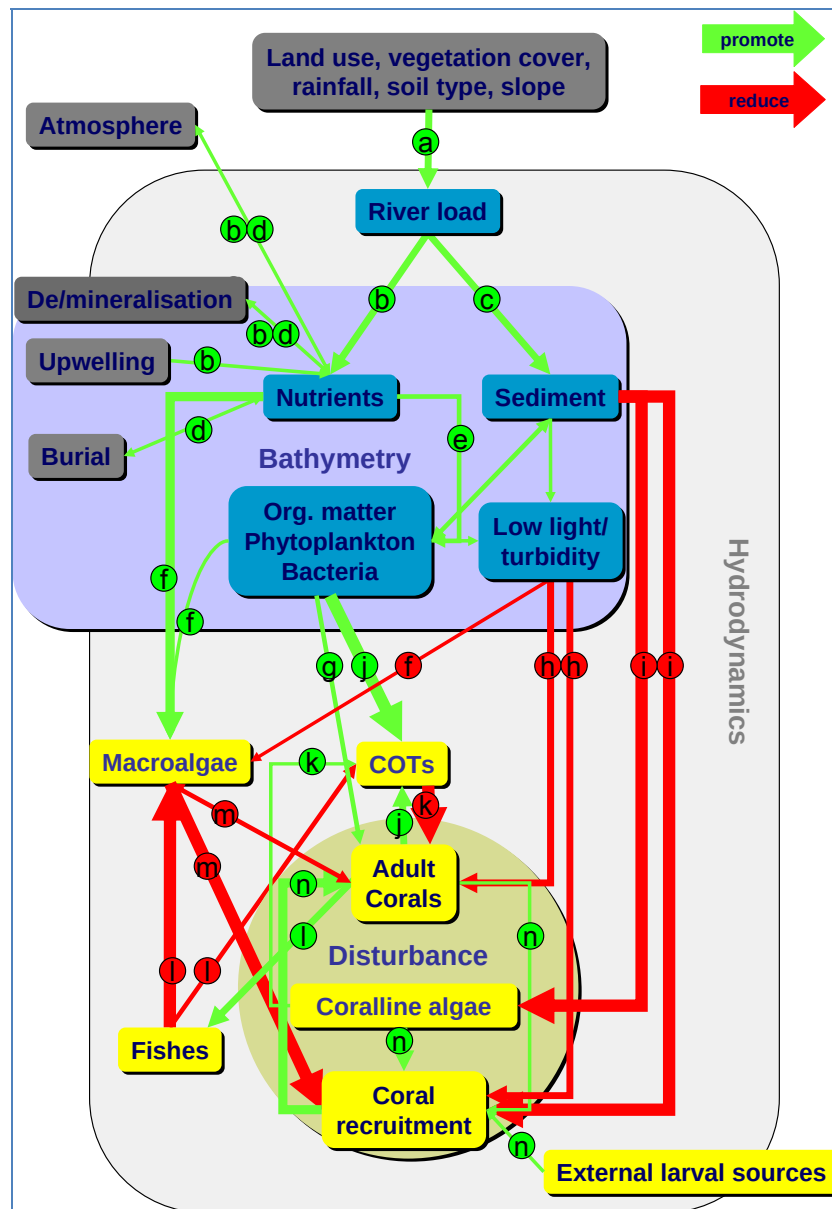


Figure 2.6: Conceptual model of the relationships between the four main water quality constituents (blue) and biotic responses (yellow). External and physical factors (grey boxes) further shape the relationships. Source: Fabricius (2007).

For example, a water quality gradient exists through the Whitsunday Island group (van Woesik *et al.* 1999) and a clear correlation between water quality parameters and reef condition is evident throughout such gradients (Fabricius and De'ath, 2004; Fabricius *et al.* 2005; Cooper *et al.* 2007, 2009). Recent work by Fabricius (2011a) and Brodie *et al.* (in press) has identified negative changes at indicator and ecosystem levels which are most likely due to the higher nutrient conditions found along the central and southern GBR areas.

Causal links between increased sediment and nutrient loads within GBR waters and state/impacts of corals (Figure 2.7) have been robustly discussed in recent scientific literature (e.g. Fabricius *et al.* 2005; Devantier *et al.* 2006; Wolanski *et al.* 2008; De'ath and Fabricius, 2008, 2011a; Delean and De'ath, 2008; Cooper *et al.* 2008, 2009; Thompson *et al.* 2010; Fabricius, 2011a, 2011b; Brodie *et al.* in press). Most studies show that high levels of

dissolved inorganic nitrogen and phosphorus can cause significant physiological changes in corals, but do not kill or greatly harm individual coral colonies (reviewed in Fabricius, 2005). However, exposure to dissolved inorganic nitrogen can lead to declining calcification, higher concentrations of photo-pigments (affecting the energy and nutrient transfer between zooxanthellae and host; Marubini and Davies, 1996), and potentially higher rates of coral diseases (Bruno *et al.* 2003). Recent studies conclude that the main way in which dissolved inorganic nutrients affect corals appears to be by enriching organic matter in plankton and sediments (Fabricius, 2011a). Furthermore, in areas of nutrient upwelling or in heavily polluted locations, chronically elevated levels of dissolved inorganic nutrients may so alter the coral physiology and calcification as to cause noticeable changes in coral communities (Birkeland, 1997).

Macroalgae and heterotrophic filter-feeders benefit more from dissolved inorganic and particulate organic nutrients than do corals. As a result, corals that can grow at extremely low food concentrations may be out-competed by macroalgae and/or more heterotrophic communities that grow best in high nutrient environments (Fabricius, 2011a). Densities of benthic filter feeders – such as sponges, bryozoans, bivalves, barnacles and ascidians – increase in response to nutrient enrichment (Costa Jr *et al.* 2000). In high densities some filter feeders, such as internal macro-bioeroders, can substantially weaken the structure of coral reefs and increase their susceptibility to storm damage.

In addition, links between nutrient enrichment and COTS population outbreaks are now well supported in both anthropogenically enriched systems such as the GBR (Brodie *et al.* 2005) and naturally enriched systems such as the northern Pacific (Houk *et al.* 2007). After primary COTS outbreaks have formed in a region with high phytoplankton concentrations, many of their numerous larvae may be transported by currents to remote regions, hence secondary COTS outbreaks may form far away from areas of eutrophication.

Other specific indicators of environmental stress on coral reef communities have also been used to document stress-related changes in benthic community composition. For example, reef sediments in the Mackay-Whitsunday region have consistently high levels of fine-grained particles, compared to other regions, and these values have increased since 2005. Densities of juvenile corals in the Mackay-Whitsunday region have declined at the same time as these observed changes in sediment composition. The increase in fine-grain sediment particles is related to changes in river flows of the nearest rivers (Proserpine, O'Connell and Pioneer Rivers); flows were below long-term medians for several years prior to 2005 and since 2006 have been substantially higher than median flow levels. Fluctuating sediment loads from the catchment lead to local changes in marine sediment composition. As turbidity is largely a function of wave and tidal re-suspension (Larcombe *et al.* 1995), changes in sediment composition toward finer grained particles would logically lead to increased levels of turbidity. Both turbidity and sedimentation have the potential to stress corals by reducing light availability for photosynthesis, with sedimentation also incurring an energy cost to corals when active removal is required. Juvenile corals are most susceptible to turbidity and sedimentation (Fabricius, 2005).

The impacts of increased sedimentation and turbidity on coral communities are also well known (see Fabricius, 2011b). Benthic irradiance is a crucial factor for reef corals. Light limitation, such as that from increased turbidity, reduces photosynthesis, leading to slower calcification and thinner tissues (Anthony and Hoegh-Guldberg, 2003; Allemand, in press). Sedimentation reduces coral recruitment rates and coral biodiversity, with many sensitive species being under-represented or absent in sediment-exposed communities. High sedimentation rates are related to low abundances of corallines in coral reefs (Kendrick, 1991; Fabricius and De'ath, 2001).

While adult corals can tolerate prolonged periods of low light, competition with macroalgae and moderate levels of sedimentation, the settlement of coral larvae and the survival of newly settled young and small colonies are extremely sensitive (Fabricius, 2005). Very little settlement occurs on sediment-covered surfaces, and the tolerance of coral recruits to sediment is at least one order of magnitude lower than that of adult corals. Settlement of coral larvae is also controlled by light intensity and spectral composition; reduced light reduces the depth at which larvae settle. Octocorals (soft corals and sea fans), are passive suspension feeders and species richness declines by up to sixty percent along a gradient of increasing turbidity, due to the disappearance of zooxanthellate octocorals (Fabricius and De'ath, 2004).

Coral records have provided tangible evidence of increased sediment and nutrient loads to the GBR lagoon since European settlement around 1860 (McCulloch *et al.* 2003; Sinclair and McCulloch, 2004; Jupiter *et al.* 2007; Lewis *et al.* 2007b; Marion, 2007; Lewis *et al.* 2010; Mallela *et al.* 2010). The use of coral cores to show the presence of a 'terrestrial signal' in the GBR, and hence changes in the delivery of materials from the land to the GBR with catchment development are now well established. Ba/Ca ratios as an indicator of suspended sediment delivery from the Burdekin have been established (McCulloch *et al.* 2003; Lewis *et al.* 2007b), as well as other metals that indicate specific changes in grazing practices (Lewis *et al.* 2007b). Additional coral proxies of water quality are currently being developed (Alibert *et al.* 2003; Wyndham *et al.* 2004; Sinclair, 2005; Marion *et al.* 2005). Changes in the delivery of water due to vegetation change or loss in catchments have also been investigated using coral cores and some dispute currently exists over the interpretation of this record (McCulloch, 2004; Lough, 2007). Changes in nitrogen delivery to reefs associated with increasing fertiliser use for sugarcane cultivation have been demonstrated from coral cores off Mackay (Jupiter *et al.* 2007; Marion, 2007). Cooper and Fabricius (2007) have used physical coral indicators to investigate the productivity of reefs through the water quality gradient in the Whitsunday Island group. The species composition of foraminifera also shows a relationship with water quality conditions along this gradient (Uthicke and Nobes, 2007). This tool has previously been applied to coral reefs in the Caribbean (Uthicke *et al.* 2006). Preliminary findings (Mallela *et al.*, unpublished data) from Dunk Island coral cores suggest that the coral skeletons may record long-term phosphorus (P/Ca) records that mimic particulate phosphorus (PP) loads from the Tully River, which in turn reflect agricultural activities on the adjacent catchments. Until now the ability to monitor, detect, and quantify nutrient inputs from agricultural runoff over long time scales has proved difficult and controversial.

MTSRF funded research has also indicated that the impacts from climate change will be exacerbated in areas which regularly experience high concentrations of dissolved inorganic nutrients (Wooldridge and Done, 2009). Impacts from water quality and herbicides can also be cumulative with increased impacts on corals where the stressors are combined (Negri *et al.* 2009). While the majority of the GBR is considered to be in reasonable health (Miller and Sweatman, 2004), some cornerstone publications have examined the endangerment of GBR organisms and described the apparent biodiversity loss of inshore reefs adjacent to catchments in which intensive agriculture is carried out (for example, Fabricius *et al.* 2005; DeVantier *et al.* 2006; Bruno and Selig, 2007). In addition, several studies have demonstrated the risks of increased nutrients, for example, increased frequency of COTS outbreaks (Brodie *et al.* 2005); change in nitrogen isotope composition of corals due to fertiliser runoff (Marion, 2007) and laboratory studies have shown the high toxicity of several commonly used pesticides in the GBR catchments on marine organisms (Negri *et al.* 2009). These findings are particularly valuable to set water quality targets for terrestrial pollutant export to the GBR lagoon, as discussed in Brodie *et al.* (2009a).

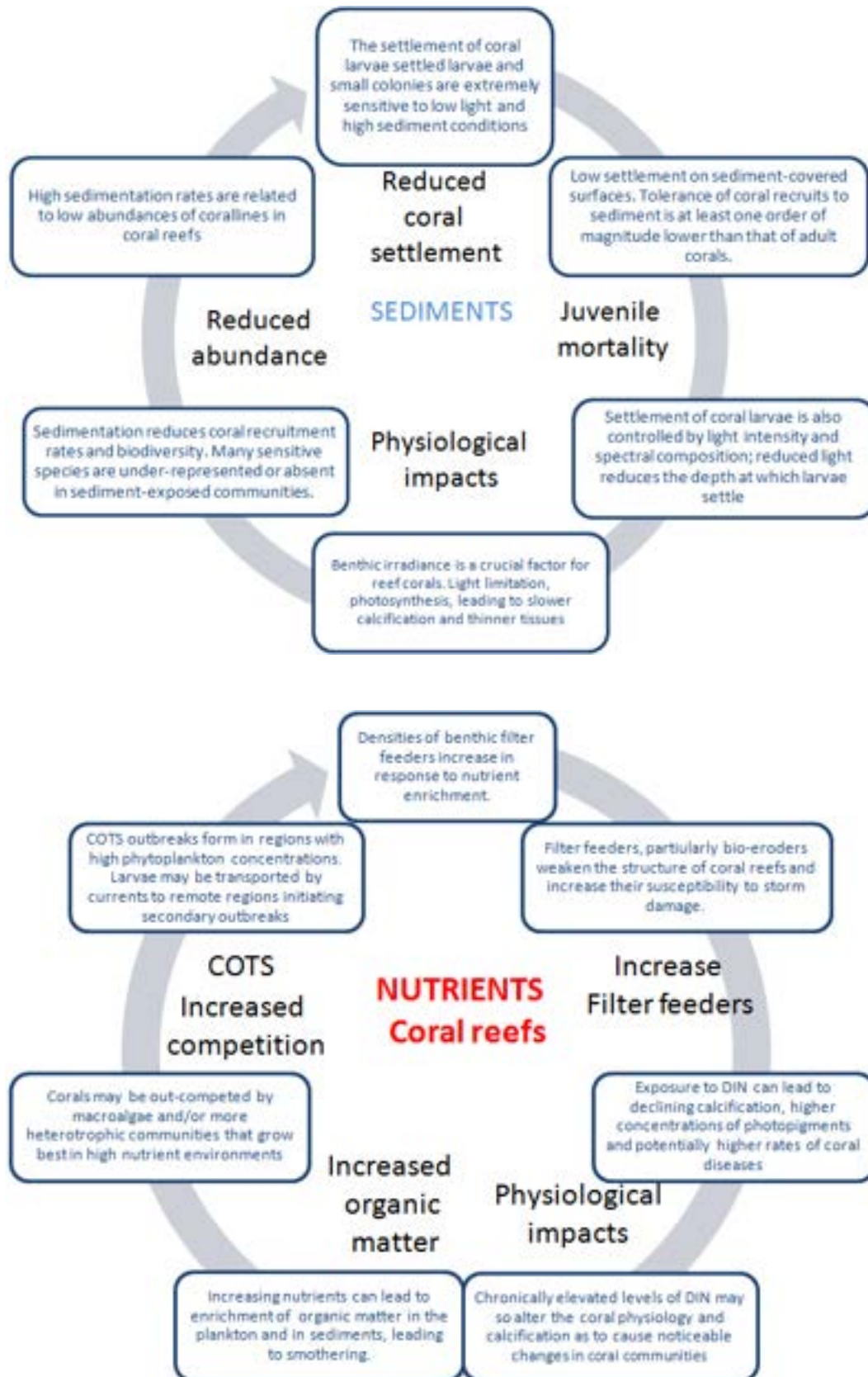


Figure 2.7: Potential and risk impacts of increasing nutrients and sediments on coral reefs.

2.4.3 Impact on GBR ecosystems: seagrass beds

Seagrasses face an array of pressures in GBR waters as human populations increase and the potential effects of global warming, such as increased storm activity, come into play (Waycott and McKenzie, 2010; Grech, 2010; Grech *et al.* 2010). Seagrasses have been exposed to climate change in the past, but only over time scales of millions of years. The changes in coastal waters currently being experienced may exceed the capacity of seagrasses to adapt. The potential consequences of degraded water quality reported at some seagrass locations within GBR waters are shown in Figure 2.8 (Waycott and McKenzie, 2010). Although little is known about the physiological mechanisms that control seagrass responses to nutrient enrichment, increased growth is generally expected until light interactions result in seagrass decline (Touchette and Burkholder, 2000).

The distribution and growth of seagrasses is dependent on a variety of factors such as temperature, salinity, nutrient availability, substratum characteristics and underwater light availability (turbidity) (Waycott and McKenzie, 2010; Collier and Waycott, 2010). Terrigenous runoff, physical disturbance, low light and low nutrients, respectively, are the main drivers of each of the four seagrass habitat types found in Queensland and changes to any or all of these factors may cause seagrass decline (Waycott and McKenzie, 2010). 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) (Waycott *et al.* 2005), 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 seagrass loss (Mellors *et al.* 2005).

Shifts in seagrass dominance as a consequence of nutrient enrichment have been reported in tropical seagrasses, where species with higher elemental requirements have a competitive advantage (Fourqurean *et al.* 1997). Elevated nutrient content of plants can also increase rates of herbivory. For example, Boyer *et al.* (2004) reported that nutrient enrichment increased consumption by up to thirty percent. Grazing by macroherbivores (such as dugong and green sea turtle) has a significant impact on the structure of seagrass communities in northern Australia (Carruthers *et al.* 2002). Elevated tissue nutrient concentrations in the leaves of seagrasses are indicators of excessive nutrient loads (Dennison *et al.* 1993). The ratio of the major nutrients in seagrass tissues is indicative of the status of plant utilisation of available nutrients; when in excess, plants are saturated and a tendency for the ecosystem to have excessive algal growth occurs (summarised in Waycott and McKenzie, 2010).

Seagrasses also respond to light limitation at the plant (e.g. pigment content, leaf morphology) and meadow scales (e.g. distribution and species composition) (Collier and Waycott, 2010). As minimum light requirements for seagrasses are generally species specific, species better adapted to low light would be competitively advantaged by lower light environments. However, seagrasses will only persist until light conditions are insufficient to maintain a positive carbon balance, leading to a decline in seagrass growth and distribution (Ralph *et al.* 2007).

Multiple lines of evidence strongly suggest that investment in improved land management practices that reduce inshore turbidity and chlorophyll levels will help correct multiple ecological imbalances that have arisen from poor water quality management (De'ath and Fabricius, 2010).

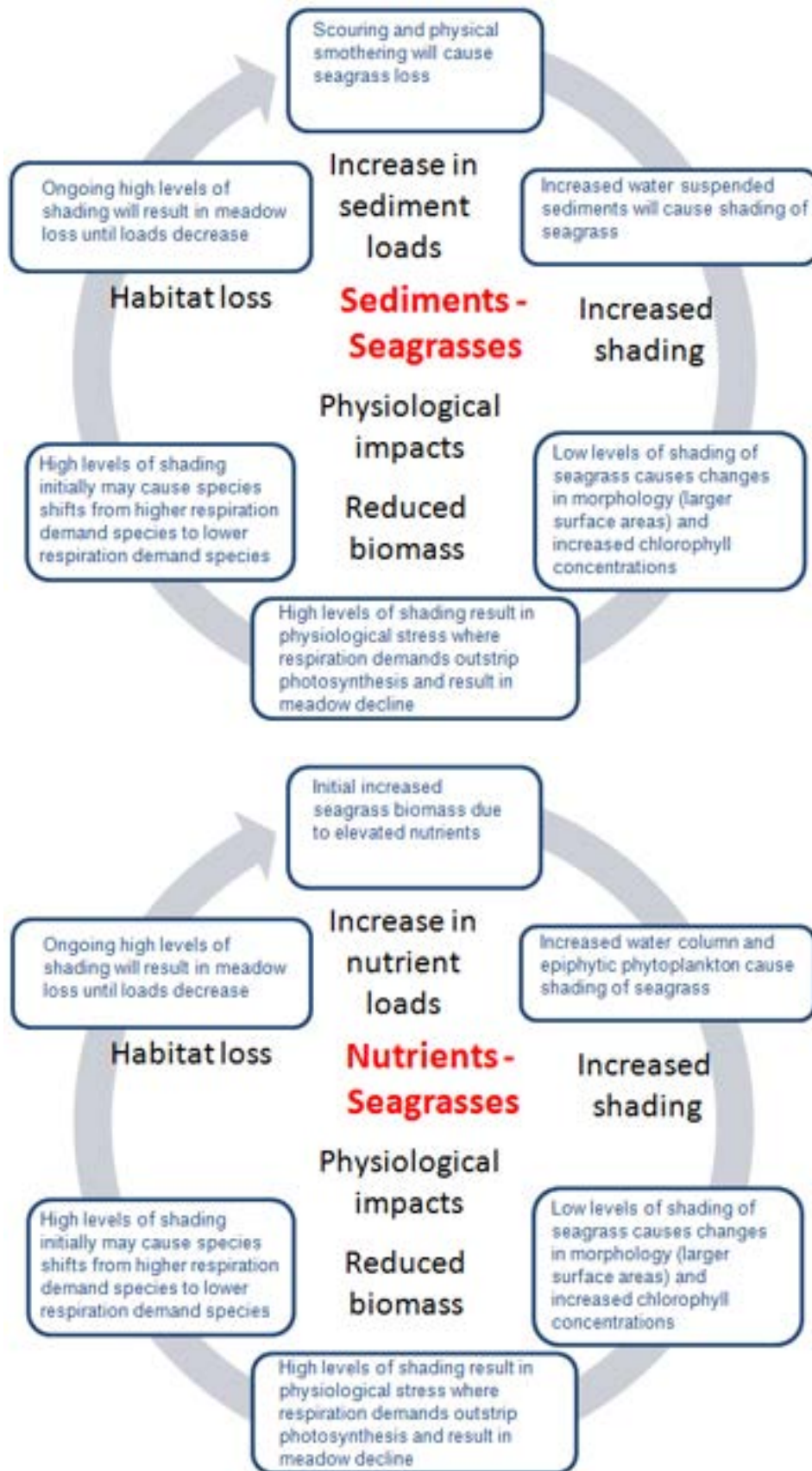


Figure 2.8: Potential and known risk impacts of increasing nutrients and sediments on seagrass communities.

2.4.4 Pesticides: source and potential impacts

A comprehensive review of the current understanding of pesticides on the GBR has been carried out as an output of the MTSRF Research Program. This section provides a summary of this synthesis.

Measurements taken in flood plume waters show that herbicide residues in parts of inshore waters of the GBR lagoon are, at times, at concentrations known to have negative effects on marine plants. While previous studies have examined the risk of individual herbicides in isolation, recent monitoring studies (Lewis *et al.* 2009b; in press) show that eighty percent of the time when herbicides are detected, two or more herbicides are present in the GBR lagoon following wet season river discharge. The herbicides most commonly detected in the GBR lagoon are designed to inhibit photosystem II (PS-II) in plants and so the risk of these herbicides should be considered additively. An additive PS-II risk using a normalisation index has been used to better assess herbicide risk in the GBR. An area of concern is the waters adjacent to the Mackay-Whitsunday region, where the risk of herbicides increases when the additive risk is considered (Lewis *et al.* 2009b). This research also identifies that the area of risk for most regions is greatly increased under the proposed additive PS-II inhibition guideline which highlights the current challenges facing ecotoxicologists to resolve the validity of pulse amplitude modulation for toxicity assessments. Lewis *et al.* (2009b) show that inshore areas of the GBR lagoon may be negatively affected from additive herbicide exposure, which could reduce the resilience of this important ecosystem.

Pesticide sampling from 2005 to 2009 showed the concentrations of herbicides adjacent to the four main geographical areas with a high proportion of agricultural activity in the adjacent catchments. Data presented in Lewis *et al.* (2009b) showed diuron and atrazine (96/184) were the most commonly measured herbicide residues in the GBR lagoon and were detected adjacent to all four geographic regions. Hexazinone residues were detected adjacent to the Tully-Murray, Burdekin-Townsville and Mackay-Whitsunday regions while tebuthiuron residues were detected adjacent to the Burdekin-Townsville, Mackay-Whitsunday and Fitzroy regions. Simazine residues were detected in the GBR lagoon adjacent to the Tully-Murray, Mackay-Whitsunday and Fitzroy regions while ametryn was only detected adjacent to the Mackay-Whitsunday region. Other non PS-II herbicides detected in the GBR lagoon included bromacil (Tully-Murray) and metolachlor (Fitzroy) while imidacloprid residues (insecticide) were detected adjacent to the Tully-Murray region. The majority of these detections are below ecosystem health guidelines however there are areas with a high percentage of individual pesticides above the guidelines in certain regions (i.e. tebuthiuron in the Fitzroy), however when normalised as an additive index, the guidelines are not exceeded.

Pesticides are all designed to negatively affect physiological processes through the disruption of biochemical pathways (Negri *et al.* 2005). A given mode of action is typically not specific to a target organism, and hence non-target organisms may be affected by the presence of pesticides. For example, herbicides such as diuron, atrazine, hexazinone and tebuthiuron bind to the D1 protein in PS-II with immediate effects including:

1. Reduction in primary production or photosynthetic efficiency [$\Delta F/F_m$ or light adapted yields measured using pulse amplitude modulation (PAM) fluorometry]; and
2. Secondary damage to PS-II reaction centres [F_v/F_m] leading to chronic photoinhibition (Jones *et al.* 2003).

Low-level, chronic exposure to herbicides may exert subtle selective pressure on lower trophic levels due to their mode of action and species specific differences in sensitivity, with potentially negative consequences for the resilience of the reef ecosystem. While the PS-II herbicides are capable of affecting marine plants directly, they may also impact upon animals

such as corals by inhibiting photosynthesis in symbiotic microalgae (Jones and Kerswell, 2003). PS-II herbicides have relatively long half-lives compared to other herbicides such as 2,4-D and glyphosate.

However, it has been difficult to ascribe measured marine impacts or changes to the increased concentrations of pesticides in GBR waters as the causal links between delivery and impact have not been identified with the marine area. However, a great deal of recent laboratory work has been able to show impacts from a cellular to system change within coral communities. Coral bleaching has been demonstrated to occur following exposure to diuron concentrations of $10 \mu\text{g L}^{-1}$ and above (Jones *et al.* 2003; Negri *et al.* 2005; Cantin *et al.* 2007). Extended laboratory exposures (two to three months) to diuron concentrations as low as $1 \mu\text{g L}^{-1}$ resulted in reduced photosynthetic efficiency and chronic photoinhibition and, consequently, a decrease in energy acquisition in the coral *Acropora valida* (Cantin *et al.* 2007). This drop in energy was strongly linked to diminished reproductive output in three coral species at exposure to $10 \mu\text{g L}^{-1}$ of diuron, and resulted in the mortality of *A. valida* colonies (Cantin *et al.* 2007). Acute exposure to high concentrations of diuron ($10\text{--}100 \mu\text{g L}^{-1}$) resulted in irreversible secondary damage to PS-II reaction centres and partial colony mortality (Jones *et al.* 2003; Cantin *et al.* 2007).

Despite evidence of serious chronic impacts on coral colonies following exposure to $1\text{--}10 \mu\text{g L}^{-1}$ diuron, the photosynthetic efficiency of their symbionts recovered to control levels within three to five days of returning the corals to uncontaminated water (Cantin *et al.* 2007). Similarly, reductions in photosynthetic efficiency of coral symbionts following exposure to $0.3 \mu\text{g L}^{-1}$ diuron were also temporary (Jones and Kerswell, 2003; Jones *et al.* 2003; Negri *et al.* 2005), thus demonstrating the reversibility of herbicide impact on symbiotic organisms. However, differences in recovery have been observed between taxa. For example, initial recovery of $\Delta F/F_m$ in seagrasses following exposure to $0.1 \mu\text{g L}^{-1}$ of diuron is not sustained, with depression of growth persisting for five days after plants were returned to seawater (Haynes *et al.* 2000). The apparent sensitivity of seagrass may be due to exposure of the roots to herbicides with a high sorption potential, in addition to exposure of the blades to herbicides dissolved in the water column. More work is required in this area to define the level of impact between species, the ability to recover and to identify impacts in the marine environment.

2.4.5 Water quality guidelines

Comprehensive management of water quality issues has been taken through the implementation of the Reef Plan, which targeted catchment rehabilitation in areas with the most likely impact on the GBR. More targeted management approaches can be assisted by the use of water quality thresholds or trigger values which help identify areas that have exceeded acceptable limits for water quality conditions. The Great Barrier Reef Marine Park Authority has prepared Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009) based on the available scientific evidence of direct biological effects from exposure to particular contaminants. Trigger levels have been identified, where managers need to take action, when these are exceeded (GBRMPA, 2009).

De'ath and Fabricius (2008, 2010) presented ecological trigger values for secchi depth, TSS and chlorophyll. Levels in excess of these values have been well correlated with a decrease in ecological functioning. Ecological condition of the GBR is significantly higher if mean annual water clarity does not drop below 10 m secchi depth and mean annual chlorophyll concentration remains below $0.45 \mu\text{g L}^{-1}$. These values have now become the guideline triggers for water quality management. These guideline values benefit photosynthetic organisms such as hard corals and phototrophic octocorals, and do not adversely impact on other important reef organisms such as heterotrophic octocorals. Chlorophyll values are ~40% higher in summer and ~30% lower in winter than mean annual values. Seasonal

chlorophyll guideline triggers should be adjusted accordingly, to $0.63 \mu\text{g L}^{-1}$ in summer and $0.32 \mu\text{g L}^{-1}$ in winter. Seasonal adjustments for secchi depths are presently not available due to lack of seasonal data. For TSS the maximum annual means as trigger values: 2.0 mg L^{-1} SS, $1.5 \mu\text{mol L}^{-1}$ PN, and $0.09 \mu\text{mol L}^{-1}$ PP.

Using this information, the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009) define trigger values that will be used to:

- Support setting targets for water quality leaving catchments;
- Prompt management actions where trigger levels are exceeded;
- Encourage strategies to minimise release of contaminants;
- Identify further research into impacts of contaminants in the Marine Park;
- Assess cumulative impacts on the GBR ecosystems at local and regional levels; and
- Provide an information source for NRM bodies, industry, government and communities.

The guidelines are now used to assess the results of the water quality monitoring components of the Reef Rescue MMP, and have been trialed for application to the remote sensing data for chlorophyll and turbidity. However, further work is required to determine suitable application of the guidelines during the wet season when concentrations are elevated through event discharges and frequent cloud cover limits the number of valid remote sensing days.

2.5 Pollutant movement in the GBR

Knowledge of the transport of terrestrially-sourced dissolved and particulate materials to the marine environment during high flow events has considerably increased in the last five years through MTSRF research and several collaborative projects. Improvements in satellite technology for tracing of freshwater plumes, sediments and chlorophyll *a* (Brodie *et al.* 2006, 2007a, 2010; Brando *et al.* 2010; Schroeder *et al.* 2009; Devlin *et al.* 2009; Devlin and Schaffelke, 2009), coral records (both physical and geochemical records to assess the exposure of coral reefs to freshwater plumes and to quantify changes in sediment and nutrient loads) (Lewis *et al.* 2007b; Lough, 2007; Marion, 2007; Mallella *et al.* 2010; Lewis *et al.* 2010) and increased laboratory sensitivity and instruments to measure pollutants, in particular passive samplers and the direct measurement of pesticides and nutrients (Rohde *et al.* 2006; Shaw and Müller, 2005; Devlin and Brodie, 2005; Devlin and Schaffelke, 2009; Schroeder *et al.* 2009; Schaffelke *et al.* 2009; Brando *et al.* 2010; Paxman *et al.* 2009; Johnson *et al.* 2010) have all provided an enhanced understanding of pollutant risk for the marine ecosystems of the GBR.

In major river flows many materials show their highest concentrations at the rising stage of the flow hydrograph and during the first major flow of the wet season (the 'first flush'). This condition is due to the hydraulic power of the water flow removing most of the easily available materials which have built up in the system since the last wet season (e.g. see Wallace *et al.* 2009a, 2009b). This is particularly observable for suspended sediment and associated particulate nutrients. Late in the hydrograph, concentrations are normally much lower, as materials have been exhausted during the rising and peak stages of the event; dilution may also play a part in this phenomenon (Devlin and Brodie, 2005; Devlin and Schaffelke, 2009). This large difference in concentration between the rising and falling stages of the hydrograph is called hysteresis. This phenomenon may only occur if subsequent events do not generate larger flows than the first flush, as very high energy floods generate the highest sediment concentrations regardless of antecedent conditions.

The majority of suspended sediments are initially deposited very close to the river mouth before being remobilised by prevailing southeasterly winds and by storm events and then becoming trapped in estuaries, north-facing embayments and mangroves (Lambrechts *et al.* 2010; Bainbridge *et al.* 2009a; Lewis *et al.* 2006; Devlin and Brodie, 2005). The clay sediment fraction may travel larger distances in the marine environment and become deposited on the mid-shelf below re-suspension depths (Wolanski *et al.* 2008).

Characteristics of flood plumes being measured and monitored in GBR waters as part of the Reef Rescue MMP include:

- Dissolved materials (including nutrients and pesticides) travel much further in the marine environment and are elevated several orders of magnitude above ambient concentrations in the freshwater plume. After turbidity decreases in the plume, nutrients are rapidly consumed by phytoplankton and algal blooms become evident in the marine waters (Devlin *et al.* 2009; Johnson *et al.* 2010).
- Satellite images and plume sampling reveal that algal blooms develop when weather conditions clear and full sunlight becomes available for increased photosynthesis (usually two to five days after the peak discharge occurs) (Brodie *et al.* 2010).

The large increase in the availability of new satellite remote sensing platforms (e.g. MODIS, MERIS, ASTER, SPOT-5, QUICKBIRD, IKONOS, SeaWiFS) added to existing platforms (LANDSAT, AVHRR) has allowed the possibility of tracking flood plume dispersal in the GBR lagoon on a daily basis. The use of such images combined with traditional concurrent surface vessel sampling and image analysis for parameters such as suspended sediments and chlorophyll *a* (Devlin and Schaffelke, 2009) has allowed the assessment of the degree of exposure of GBR reefs (and other ecosystems) to be quantified (Brodie *et al.* 2006; Rohde *et al.* 2006; Brodie *et al.* 2007b). In addition, analysis of the datasets collected in long-term studies of GBR water quality have allowed fine-scale assessment of water quality in the GBR lagoon (Furnas, 2003; De'ath, 2005, 2006; Devlin and Brodie, 2005; Furnas *et al.* 2005; Brodie *et al.* 2007b). Use of these combined datasets (flood plume and long-term water quality) has allowed modelling to link river discharge of contaminants to reef ecosystem exposure and to assess possible effects (Devlin *et al.* 2003; Wolanski and De'ath, 2005; Wooldridge *et al.* 2006). Ultimately, linking end-of-catchment loads with marine trigger values requires a receiving water model such as the ChloroSim model (Wooldridge *et al.* 2006) to relate pollutant loads to ecosystem response. Currently this type of model is only available for some priority pollutants and GBR targets (e.g. nitrate end-of-catchment loads and chlorophyll concentrations in the GBR lagoon from ChloroSim) and it is a priority research area to develop these relationships for all pollutants.

From these studies and advances in the integration of remote sensing with *in situ* water quality monitoring, we now know that river water plumes reach further offshore in the GBR than originally thought (Devlin and Schaffelke, 2009; Schroeder *et al.* 2008; Brodie *et al.* 2010). Mapping of spatial differences in water quality concentrations (De'ath and Fabricius, 2008) demonstrates very clearly a strong gradient of change north to south and inshore to offshore. Elevated nutrient concentrations (considerably above ambient levels) have also been detected several kilometres (up to hundreds of kilometres) from the mouths of major GBR catchments (Devlin and Brodie, 2005; Brodie *et al.* 2006; Rohde *et al.* 2006; Schaffelke and Slivkoff, 2007; Devlin *et al.* 2010; Schroeder *et al.* 2009). In addition, coral records are able to quantify the increased sediment and nutrient loads exported from the waterways of the GBR catchments (McCulloch *et al.* 2003; Lewis *et al.* 2007b; Marion, 2007; Jupiter *et al.* 2008). Recently, coral proxies (Ba, Y, Mn) have provided evidence of increased sediment export to the GBR lagoon from the Burdekin River catchment (McCulloch *et al.* 2003; Lewis *et al.* 2007b) while nitrogen isotopes in the (insoluble) organic component of the coral skeleton have been used to quantify increases in nutrient loads from the Pioneer River

(Marion, 2007; Jupiter *et al.* 2007, 2008). Changes in the nitrogen isotopic signature and coral nitrogen concentrations were correlated with increased fertiliser application in the Pioneer River catchment although additional studies from other parts of the GBR as well as understanding N isotope dynamics in river water plumes are required to validate these findings.

2.6 Assessing the risk and exposure of GBR ecosystems to declining water quality

The areas of freshwater, plume exposure, and places at risk of impact from altered water quality have been identified by a number of researchers (King *et al.* 2002; De'ath, 2006; Delean and De'ath, 2008; Maughan and Brodie, 2009; Devlin and Schaffelke, 2009; Brodie and Waterhouse, 2009; Brodie *et al.* 2010; Devlin *et al.* 2010) to link cause and effect and provide information on priority management areas. This use of spatial information using a combination of aerial imagery, remote sensing imagery, water quality concentrations and long-term monitoring data has helped to identify the areas 'at risk from impacted water quality in GBR waters. All of the spatial outputs from the mapping of risk and exposure highlight an affected area south of Cooktown and north of the Mackay-Whitsundays, concurrent with extensive agricultural activities on the adjacent catchment. Delean and De'ath (2008) used a system of potential indicators of reef ecosystem health as a way to map spatial patterns of relative reef ecosystem health. These analyses show similar spatial patterns of relative reef ecosystem health across both individual and composite indicators, with reefs in the central inner-shelf regions being in relatively poor health. The patterns of relative health were also related to measures of water quality based on water clarity and concentrations of chlorophyll, with poorer health coinciding with low levels of water quality (low clarity and high chlorophyll). A recent paper by De'ath and Fabricius (2010) modelled a strong relationship between water clarity, chlorophyll and measures of reef status. Water clarity and chlorophyll showed strong spatial patterns, with water clarity increasing more than threefold from inshore to offshore waters and chlorophyll decreasing approximately twofold from inshore to offshore and approximately twofold from south to north. Richness of hard corals and phototrophic octocorals declined with increasing turbidity and chlorophyll, whereas macroalgae and the richness of heterotrophic octocorals increased.

The reefs at greatest relative risk for both benthos and fishes are located in the inner central third of the GBR, with a greater cross-shelf extent for benthos. In terms of the NRM regions, this corresponds to the inshore areas from the Wet Tropics and Burdekin to the upper Mackay-Whitsunday region. De'ath and Fabricius (2008, 2010) mapped large-scale concentrations of water quality information to show that the inshore areas of the Great Barrier Reef with highest values of chlorophyll and lowest values of secchi depth are those areas adjacent to the Wet Tropics. The mapping of risk and areas of high impact/change have supported the recently released Consensus statement which identifies the central GBR as a high impact area. The ongoing MTSRF funded research continues to build up our understanding of the linkages between catchment activities and GBR ecological health. What is certain is that the contaminant load is significantly higher from areas with fertilised agriculture, driving altered loads of nutrients, particularly DIN and PS-II herbicides.

Linking the movement of flood plumes and affected water quality to reef exposure has been useful in identifying the reefs at risk from contaminants (Devlin *et al.* 2003). This work led to a Reef Exposure model, where the exposure criterion would factor parameters such as the proximity of the reef to the source of the contaminant, the likelihood and frequency of exposure of the reef to river plumes, and the amount of contaminant within the plumes at a range of distances (Maughan *et al.* 2008; Maughan and Brodie, 2009). The model provides a relatively simple way of combining contaminant load estimates, river flow and variability

characteristics with plume and contaminant behaviour, and the distance of every reef to each river mouth to give an estimated reef exposure class. Historical flood plume extent data (modelled) was used to quantify the typical spatial extent of the summer runoff and seawater mixing zone of the GBR lagoon. This spatial analysis again demonstrated the existence of a discernible north-south gradient along the length of the GBR. The undisturbed catchments of the northern GBR showed much lower levels of nutrient enrichment with a strong correlation between this north-south enrichment gradient and the flood concentration of dissolved inorganic nitrogen (DIN) entrained by the various river systems (Wooldridge *et al.* 2006). Recent work (Delean and De'ath, 2008; De'ath and Fabricius, 2008, 2010; Devlin *et al.* 2010) identifies strong gradients of water quality with biological measurements, showing very clearly that water quality influences changes in biological responses.

Increased understanding of the movement, extent and duration of flood plumes established through MTSRF funded research and the Reef Rescue MMP has enabled an estimate of the number of inshore marine ecosystems (coral reefs, seagrass meadows and seabed) in areas of high exposure to flood plume waters carrying high concentrations of pollutants (DIN, TSS and PS-II herbicides) (Devlin *et al.* 2010). First, using pollutant load estimations from Brodie *et al.* (2009b), regional areas were ranked according to the volume of pollutant loading for dissolved nutrients (DIN), total suspended sediments and PS-II herbicides (Table 2.6).

Table 2.6: NRM catchments ranked by pollutant load for DIN, TSS and PS-II herbicides (where 1 represents the lowest pollutant load and 6 the highest). Data derived from Brodie *et al.* (2009b).

NRM Region	Ranking of pollutant load		
	TSS	Dissolved Nutrients	Pesticides
Cape York	1	1	1
Wet Tropics	3	5	5
Burdekin	6	6	2
Mackay-Whitsundays	2	4	6
Fitzroy	5	3	4
Mary-Burnett	4	2	3

The frequency of exposure to selected contaminants (DIN, TSS and PS-II herbicides) was then identified by combining information from the ranked catchment loads and the frequency and areal extent of flood plumes. A major factor in the scale of exposure is the distance and direction of the ecosystems from the catchments of concern (Maughan and Brodie, 2009). For example, it is the inshore region of Mackay-Whitsunday where elevated pollutants in flood plumes can affect a significant number of reefs and seagrasses due to the high number of ecosystems and their close proximity to frequent exposure from flood plumes.

This exposure mapping allows the identification of the number of seagrass and coral reef systems which are located in the high to moderate exposure areas. High exposure is identified in areas where plume waters occur at least two to five times per year from land use activities specific to that pollutant (i.e. grazing/TSS in the Burdekin). Moderate exposure is identified in areas where plume waters occur at least once or twice per year. Plume periods can vary between catchments, but in general Wet Tropics systems experience high flows from a period of days to weeks, intermittently. Dry Tropics systems are usually associated with much longer flow periods, and recent events in both the Burdekin and Fitzroy catchments have sustained high flow for periods of four to six weeks. Table 2.7 compares

the number of ecosystems within the high to moderate category for each pollutant. The actual number of reefs and seagrass beds located within each exposure category depends on the proximity of the ecological systems to the riverine influence gradient. Mackay-Whitsundays have a large number of reefs within the high exposure category due to the close proximity of the reefs to the Whitsunday rivers, and the large extent measured from flood plume imagery taken from Mackay-Whitsundays.

Table 2.7: Number and area (km²) of seagrass, coral reef and seabed ecosystems located within areas of high to medium exposure to specific pollutants in flood plume waters, calculated by spatial mapping of plume extent and catchment loads. Source: Devlin *et al.* (2010).

Frequency of exposure to elevated DIN concentrations					
Region	Seagrasses		Reefs		Seabed
	Number	Area (km ²)	Number	Area (km ²)	Area (km ²)
Wet Tropics	70	185	99	186	8,358
Burdekin	72	584	66	54.8	8,958
Mackay-Whitsunday	97	198	331	208	9,138
Fitzroy	98	223	148	138	6,583
Frequency of exposure to elevated TSS concentrations					
Region	Seagrasses		Reefs		Seabed
	Number	Area (km ²)	Number	Area (km ²)	Area (km ²)
Wet Tropics	65	185	64	83	12,331
Burdekin	72	584	66	54.8	8,958
Mackay-Whitsunday	90	178	229	106	8,297
Fitzroy	104	225	173	159	7,904
Frequency of exposure to elevated PSII herbicide concentrations					
Region	Seagrasses		Reefs		Seabed
	Number	Area (km ²)	Number	Area (km ²)	Area (km ²)
Wet Tropics	70	185	99	107	8,359
Burdekin	66	487	36	20	3,759
Mackay-Whitsunday	95	198	328	208	9,138
Fitzroy	104	225	173	159	7,904

The exposure mapping focuses on the movement, extent and frequency of individual pollutants. However, during high flow periods, these pollutants move together and would have combined exposure pressures. The actual movement, dispersion and uptake of the individual pollutants would vary depending on the mixing properties; however, the areas regularly exposed to plume waters would see high exposures to all three pollutants. Figure 2.9 identifies the spatial extent of combined exposure for up to three pollutants. The rules identify 'high' exposure as a combination of TSS/DIN, TSS/PSII or DIN/PS-II scoring high and the other pollutant scoring medium. This exposure score then identifies different combinations of exposure rankings down to the three pollutants all being 'low' exposure.

Exposure does not indicate certainty of an ecological effect on the plants and animals present within the plume. The probability of actually exceeding the guidelines is limited to a smaller area contained within the high to moderate exposure plume area. The areas identified as high to moderate exposure will see plume waters which contain elevated concentrations of pollutants, (the pollutant dependent on the adjacent landscape) and may potentially impact on the ecology. In particular the potential PS-II exposure shows PS-II herbicide exposure at detectable concentrations sometimes at levels that can cause measurable effects on marine organisms. Despite elevated concentrations being measured across these exposure areas in periods of high flow, it is not sufficient to ascribe certainty that water quality values will exceed thresholds based on water quality guidelines and/or be linked to a measurable ecological impact. These exposure areas represent areas which could be identified as potential areas for impact relative to terrestrial discharge. Continuing research, monitoring and mapping could be used to resolve the extent of probable impact over these exposure areas.

Regionally specific spatial models, created by the combination of remote sensing imagery and *in situ* water quality data are now being investigated with exposure models for the Tully and Burdekin (Devlin and Schaffelke, 2009; Devlin *et al.* 2009, 2010) identifying the actual number and location of reefs and seagrass beds at risk from high TSS and/or nutrient waters. Further work on the application of remote sensing algorithms is underway with CDOM measurements being used to define the actual full extent of plume (riverine) waters for each year, and the total area of freshwater influence (Schroeder *et al.* 2008). Finally, remote sensing algorithms and high frequency data loggers are being sourced for use in water quality compliance monitoring over the regional areas (Brando *et al.* 2010; Devlin *et al.* 2009).

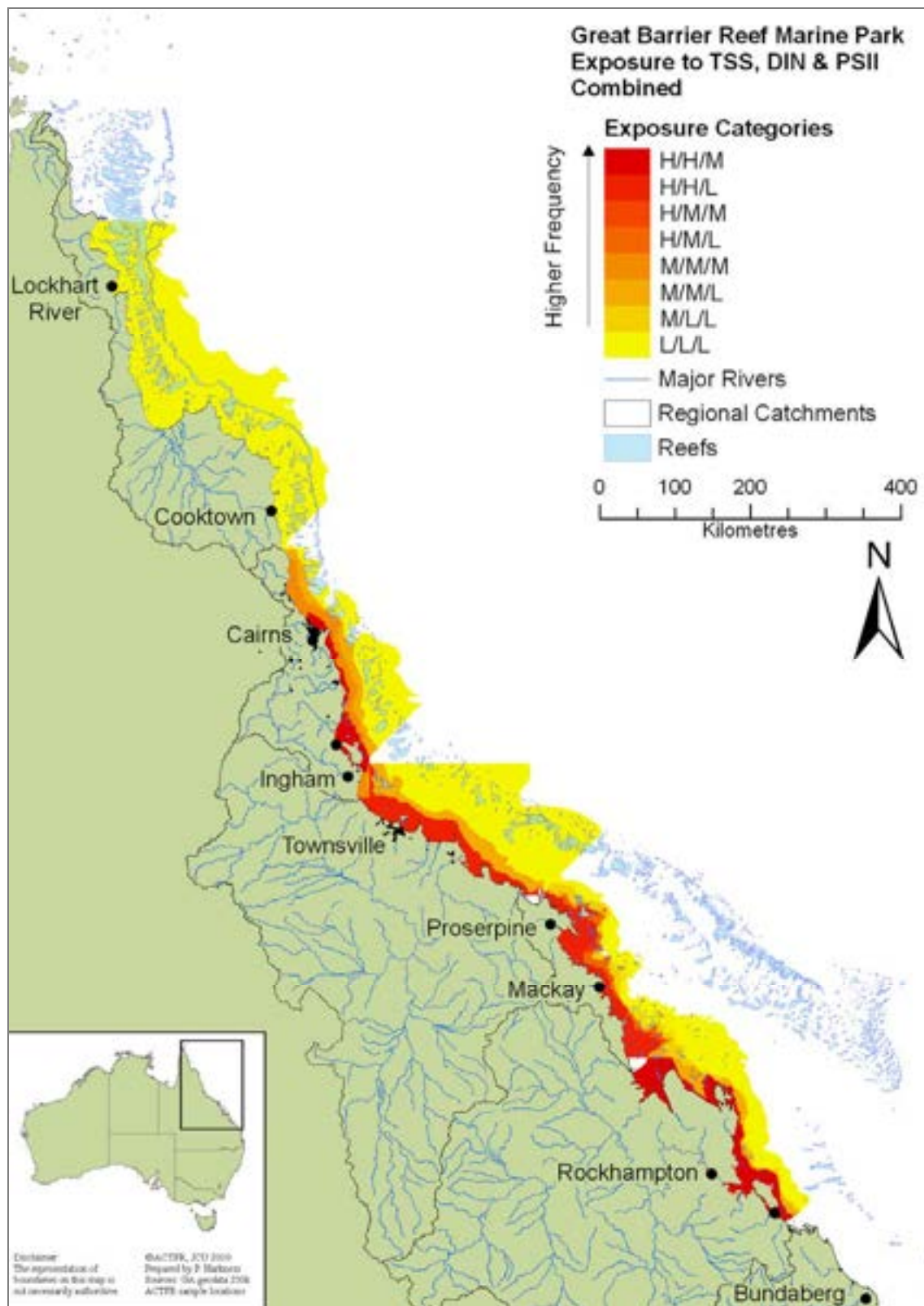


Figure 2.9: Spatial extent of the exposure categories for TSS, DIN and PS-II herbicides. Exposure categories are based on the combination of the exposure rankings for the three pollutants. Highest exposure is identified by two out of the three pollutants being at high exposure. Source: Devlin *et al.* (2010).

3. Overview of case studies

The following sections contain outcomes of research at a catchment level to highlight the advances that have been made in our conceptual understanding of freshwater and marine processes that drive the connections between catchment and reef ecosystems. A case study of the Tully catchment and marine area is presented to illustrate Wet Tropics system dynamics, while the Burdekin case study represents catchment to reef processes in a large, Dry Tropics system which has had exceptional flows in the last two years (Devlin *et al.* 2009).

The Tully catchment and marine area is located within the Wet Tropics region between Townsville and Cairns (Figure 3.1). The Burdekin catchment and marine area is located just south of the Herbert River and extends several hundred kilometres inland (Figure 3.1). There are important differences within and between these case study areas and many of the research outputs and management activities are specific to the catchment. However, many of the principles of the research findings can be applied broadly to these catchment types and are discussed in the summaries provided in Section 4 and 5 for each case study.

3.1 Characteristics of Wet and Dry Tropics catchments

Wet Tropics catchments are characterised by wetter climates and presence of intensive agricultural land uses (sugarcane and horticulture) and their associated fertiliser and pesticide usage, with the Wet Tropics being identified as a region of high nutrient and pesticide runoff concern (Fabricius *et al.* 2005; Brodie *et al.* 2009c).

Dry Tropics catchments are characterised by drier, more variable climates and being much larger in size than the Wet Tropics catchments (each ~135,000 km²). Dry Tropics catchments are dominated by unimproved savannah/woodland rangeland grazing, and are identified as considerable contributors of suspended sediment to the GBR lagoon (Bainbridge *et al.* 2006a, 2006b; Brodie *et al.* 2009c).

The Tully and Burdekin catchment areas differ substantially in size, hydrology, agricultural activities and priority management practices. The past two years have seen exceptional river flows into the GBR catchment. In 2007/2008, both the dry tropical Burdekin and Fitzroy Rivers experienced extensive flooding, and this unusual situation was repeated for the Burdekin River in the 2008/2009 wet season. During the same two-year period, the Wet Tropics (except for the Mulgrave River) and Mackay-Whitsunday regions experienced above average flow (Table 3.1). Freshwater discharge from the GBR catchment in 2008/2009 was 2.2 times the long-term annual median flow, with the Burdekin River experiencing more than five times the annual median flow. This highlights the differences between the case study areas and the similarities between these catchments and others in the GBR.

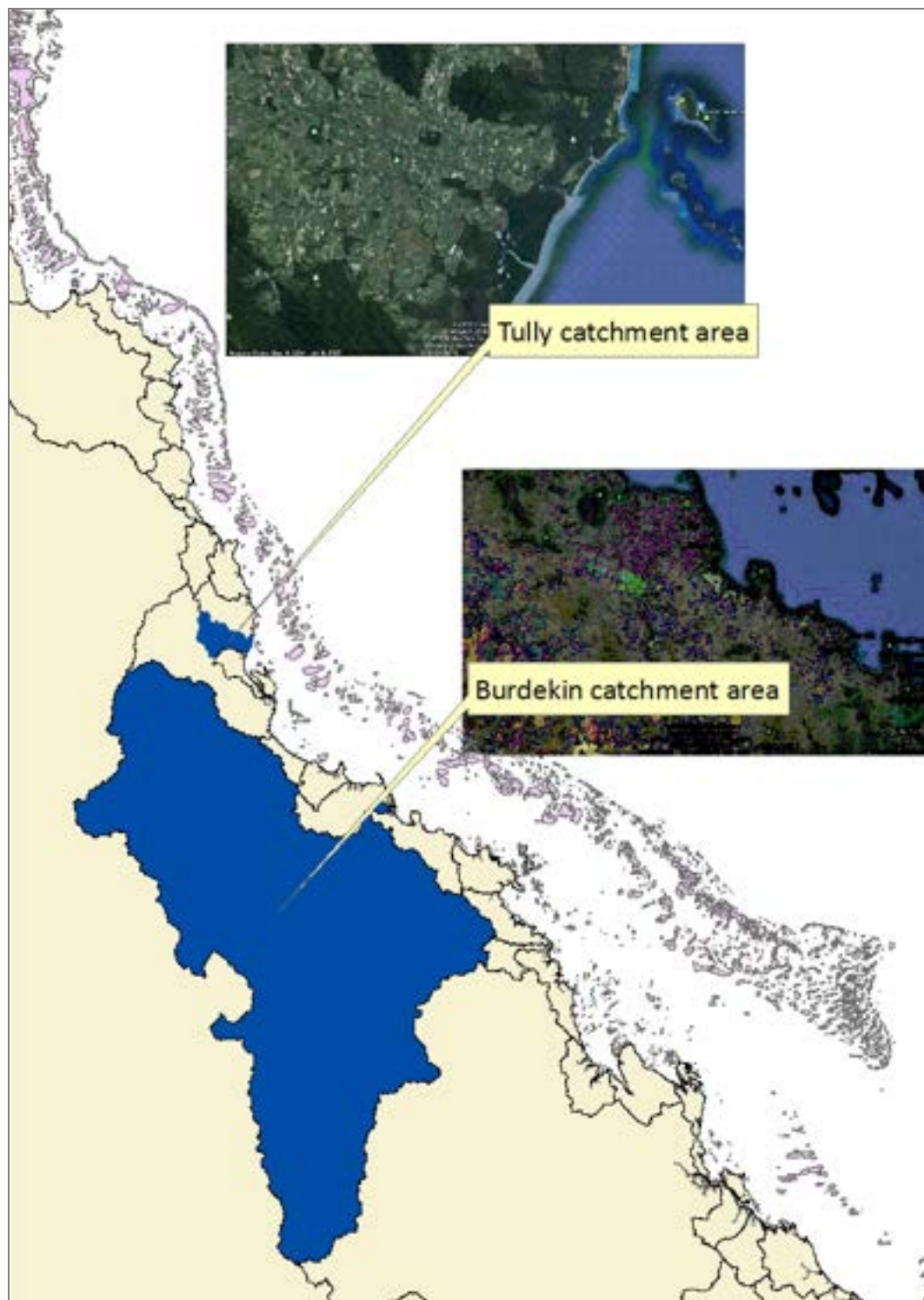


Figure 3.1: Location of the two case study catchments within the Great Barrier Reef catchment area. Source: ACTFR.

Table 3.1: Annual freshwater discharge (ML) for major GBR catchment rivers in 2008/2009. Median and mean annual flow is estimated from available long-term time series for each river. Data supplied by the Queensland Department of Environment and Resource Management. Source: Devlin *et al.* (2009).

Region	River	Long-term river discharge median (ML)	Long-term river discharge mean (ML)	Total year discharge 2008/2009 (ML)	Difference between 2008/2009 flow and long-term median (ML)	Relative difference between 2008/2009 flow and long-term median
Cape York	Normanby	3,550,421	3,707,007	2,338,784	-1,211,637	0.66
Wet Tropics	Barron	692,447	795,275	779,456	87,009	1.13
	Mulgrave	719,625	743,399	688,515	-31,110	0.96
	Russell	1,049,894	1,051,743	1,212,230	162,337	1.16
	North Johnstone	1,845,338	1,797,648	1,986,776	141,438	1.08
	South Johnstone	810,025	801,454	1,043,893	233,868	1.29
	Tully	3,128,458	3,175,298	3,759,051	630,593	1.20
	Herbert	3,122,768	3,492,135	9,606,409	6,483,641	3.08
Burdekin	Burdekin	5,957,450	9,575,660	30,110,062	24,152,612	5.05
Mackay-Whitsunday	Proserpine	35,736	70,568	63,263	27,527	1.77
	O'Connell	148,376	201,478	167,586	19,211	1.13
	Pioneer	731,441	648,238	931,808	200,367	1.27
	Plane	112,790	154,092	188,195	75,405	1.67
Fitzroy	Fitzroy	2,708,440	4,461,132	2,193,040	-515,400	0.81
Burnett	Burnett	147,814	217,511	12,079	-135,735	0.08
Total		24,761,023	30,892,638	55,081,147	30,320,124	2.22

3.2 Research focus

The main aim of the catchment-to-reef research on both the Tully and Burdekin catchment and marine areas was to extend our understanding of the range of processes that control and shape water quality sources, pathways and impacts. The research has been targeted within each catchment to identify the particular issues or problems that are unique to that catchment, though in many cases, this work can be extended into other catchments and can be considered to be typical of wet or dry catchments. Table 3.2 summarises the research areas and key publications resulting from this work as part of the MTSRF Research Program. In the Burdekin catchment and marine area, MTSRF funded research has targeted:

- Relationships between land use and pollutant loads delivered to the GBR;
- Prioritisation of catchment land use relative to marine load and associated management practices;
- Material transport pathways through sub-catchments to the end of the catchment;
- Transport, uptake and impact of sediment in the catchment;
- The fate of pollutants in the marine environment and impacts, with a focus on movement and impact of the finer sediment load (related to ecological harm); and
- The role of flood size and frequency on marine system exposure.

In the Tully catchment and marine area, MTSRF funded research has targeted:

- Relationships between land use and pollutant loads delivered to the GBR;
- Wetland condition and connectivity;
- Transport, uptake and impact of dissolved nutrients, from paddock scale to end of catchment, to the GBR;
- Estimation of the role of the floodplain in pollutant delivery to the GBR;
- Fate of the high nutrient delivery from the catchment to the GBR;
- Detection and monitoring of pesticides;
- Further work on the knowledge of the impacts of degraded water quality on instream environments and the nearshore marine environment;
- Fate of fine sediments in inner coastal marine areas; and
- The role of flood size and frequency on marine system exposure.

Table 3.2: Summary, description and peer reviewed papers associated with each MTSRF funded water quality project (2006-2010). The reference list is not exhaustive. For a full list of references associated with MTSRF funded research, refer to Section 8.

MTSRF Project and Research Focus	Area(s) of Research	Lead Researcher and Institution	References
Project 3.7.1 Marine	GBR water quality impact on reef ecosystems, indicators for monitoring and evaluation	Katharina Fabricius, AIMS	- Fabricius, 2011a, 2011b - Fabricius <i>et al.</i> 2010a, 2010b - Cooper <i>et al.</i> 2008, 2009 - De'ath and Fabricius, 2008, 2010
Project 3.7.1 Tully, Burdekin	Fine suspended sediment transport and fate in Tully and Burdekin marine areas	Eric Wolanski, AIMS Katharina Fabricius, AIMS	- Wolanski <i>et al.</i> 2007, 2008, 2009 - Lambrechts <i>et al.</i> 2010
Project 3.7.2 Burdekin	- Burdekin catchment to reef tracing study - Sediment movement - Dam trapping efficiency - Pesticides	Jon Brodie, ACTFR Stephen Lewis, ACTFR Zoe Bainbridge, ACTFR	- Lewis <i>et al.</i> 2007c, 2008, 2009a, 2009b - Bainbridge <i>et al.</i> 2009a, 2009b - Brodie <i>et al.</i> 2009b - Brodie and Bainbridge, 2008
Project 3.7.2 GBR wide	- Load estimates - Identification of catchment risk	Jon Brodie, ACTFR Jane Waterhouse, ACTFR Stephen Lewis, ACTFR Zoe Bainbridge, ACTFR	- Brodie <i>et al.</i> 2009b, 2009c - Brodie and Waterhouse, 2009 - Bainbridge <i>et al.</i> 2007, 2008, 2009b - Lewis <i>et al.</i> 2007c
Project 3.7.2 Tully Burdekin Mackay-Whitsundays	Long term records of land use change	Stephen Lewis, ACTFR Malcolm McCulloch, ANU	- Lewis <i>et al.</i> 2007b, 2010 - Mallella <i>et al.</i> 2010
Project 3.7.3 Tully	- Freshwater impacts - Wetland connectivity - Fish indicators	Richard Pearson, JCU Angela Arthington, GU Jim Wallace, CSIRO	- Arthington <i>et al.</i> 2006, 2010 - Arthington and Pearson, 2007 - Connolly and Pearson, 2007 - Pearson and Stork, 2008 - Pearson <i>et al.</i> 2010a, 2010b - Wallace <i>et al.</i> 2008a, 2008b, 2009a, 2009b, 2010a, 2010b, 2010c - Karim 2008a, 2008b, 2009a, 2009b, 2009c, 2010a, 2010b, 2010c, 2010d
Project 3.7.4 Tully	- Catchment processes, wetlands - Duration of floodplain connectivity - Hydrodynamic model	Jim Wallace, CSIRO Fazlul Karim, CSIRO	- Wallace <i>et al.</i> 2007, 2008a, 2008b, 2009a, 2009b, 2010a - Karim <i>et al.</i> 2008a, 2008b, 2009a, 2009b, 2009c, 2010a, 2010b, 2010c, 2010d
Project 3.7.5 Tully Burdekin	Socio economic modeling in wet and dry tropics catchments	Martijn van Grieken, CSIRO Peter Roebeling, CSIRO	- Roebeling <i>et al.</i> 2007a, 2007b, 2009 - Roebeling and van Grieken, 2007 - Roebeling and Webster, 2007 - van Grieken <i>et al.</i> 2009, 2010a, 2010b

MTSRF Project and Research Focus	Area(s) of Research	Lead Researcher and Institution	References
Project 3.7.7 GBR wide	Improved estimates of pollutant loads	Petra Kuhnert, CSIRO	- Kuhnert <i>et al.</i> 2009 - Kuhnert and Henderson, 2010
Reef Rescue MMP / Project 3.7.2 Tully, Burdekin	- Plume extent and movement - Marine risk assessment	Michelle Devlin, ACTFR Jane Waterhouse, ACTFR Stephen Lewis, ACTFR Zoe Bainbridge, ACTFR	- Devlin and Schaffelke, 2009 - Maughan and Brodie, 2009 - Devlin <i>et al.</i> 2009, 2010, in press-a - Brodie <i>et al.</i> 2010 - Schroeder <i>et al.</i> 2009

4. Case Study 1: Burdekin catchment

4.1 Introduction

The Burdekin region includes the Black, Burdekin, Don, Haughton and Ross River catchments as well as several smaller coastal catchments, all of which discharge into the GBR lagoon. Because of its geographical location, rainfall in the region is lower than other regions in tropical Queensland, although there is considerable year-to-year variation, with 75% of the annual rainfall received between December and March. River discharge, particularly from the Burdekin River, can be quite high due to the size of the catchment. For example, the annual river flow of the Burdekin River in 2008/2009 (October 2008 to October 2009) exceeded the long-term median flow by more than five times, with peak flows recorded in February 2009, but accounted for only 73% of the highest flow on record in 1991.

The large Burdekin River catchment (130,035 km²) has been identified as a significant contributor of suspended sediments and nutrients to the GBR lagoon (Brodie *et al.* 2009c; Kroon *et al.* 2010; Mitchell *et al.* 2006; Furnas, 2003; Belperio, 1979). Research has focused on the end-of-river with comparatively little study on the sources of these terrestrial materials from within the catchment. This limitation has been addressed by recent modelling and monitoring efforts at the sub catchment scale, some of which have been undertaken as collaborative activities as part of the MTSRF Research Program (e.g. Bainbridge *et al.* 2008, 2006a, 2006b). This research has identified the high variability of load contributions from the major sub-catchments of the Burdekin which reflects the highly variable climate, geology, vegetation and topography of this large semi-arid catchment (Bainbridge *et al.* 2006a, 2006b).

Knowledge of the load contributions has been instrumental in identifying priority catchment areas for on-ground remedial works in the Burdekin catchment. Recent work supported by the MTSRF incorporates a suite of developments, ranging from the correlation of water quality concentrations with land use characteristics, acquisition of *in situ* data to test the current models, and refinement of model outputs through a better understanding of catchment processes.

The Burdekin River catchment is dominated by very high levels of fine suspended sediment, deriving from grazing-related erosion over this huge catchment area, while the dissolved nutrients (N or P) are at relatively low levels (as measured at the Burdekin Inkerman Bridge). By contrast, the water quality of the Lower Burdekin Area, which includes the Burdekin River Irrigation Area (BRIA) and the Burdekin Delta, is greatly affected by its fertiliser-additive land use, mostly irrigated sugarcane. Despite the relatively small size of cropping in the Lower Burdekin area (6.5%; Table 4.1), its exports contain high levels of dissolved nutrients and significant levels of herbicide residues. Cropping is also carried out in two small areas of the Burdekin River catchment. Cereals are grown in the southern section of the Suttor sub-catchment, overlapping into the Belyando sub-catchment, while sugarcane is grown at the bottom end of the Burdekin River in the small delta area near Dalbeg, which drains into the Burdekin River. These fertiliser additive land uses add up to just 0.52% of the total catchment area.

Table 4.1: Mean percentage land uses for the major sub-catchments of the Burdekin Water Quality Improvement Program (WQIP) program. Source: Brodie and Bainbridge (2008).

Major Sub-catchment	Land use area (mean percentage)			Catchment Area (km ²)
	Grazing	Cropping	Urban	
Upper Burdekin	87	0	1	60,371
Cape	90	0	0	20,864
Suttor	81	14	0	16,291
Belyando	94	3	0	62,881
Bowen Bogie	63	1	1	14,525
East Burdekin	95	3	0	2,734
Lower Burdekin/ Haughton River/ Don River	71	25	1	5,978

4.2 Pollutant sources and loads

The Burdekin River has a large upstream watershed area (115,000 km²) within the Dry Tropics, and due to the large spatial area and the dominance of grazing it is seen as a priority watershed to measure, model and manage sediment loads to the GBR. Extensive research and monitoring programs in the Burdekin catchment over the last five years have shown that different landscapes in the catchment are producing vastly different amounts of suspended sediments, most likely dependent on geology and/or soil type, elevation, climate and vegetation as well as grazing management styles (Brodie *et al.* 2003; O'Reagain *et al.* 2005; Bainbridge *et al.* 2006a; 2006b; 2009a). This has confirmed modelling results that predicted large spatial differences in erosion and sediment delivery (Brodie *et al.* 2003; McKergow *et al.* 2005b; Fentie *et al.* 2006). Modelling of sediment and nutrient movement has been applied extensively throughout the Burdekin catchment to advance our understanding of the pathways and processes for sediment and nutrients. Comparisons between modelled predictions (using SedNet and ANNEX) and measured suspended sediment and nutrient species concentrations in the Burdekin show fair correlation for suspended sediment, good correlation for dissolved nitrogen and phosphorus species but poor correlations for particulate nutrient species (Mitchell *et al.* 2007; Sherman *et al.* 2007; Bainbridge *et al.* 2007).

A combination of modelling outputs and adequate field sampling has been recommended as the preferred approach for the estimation of loads in the Burdekin catchment (Bainbridge *et al.* 2009a). Models are required to estimate sediment and nutrient inputs coupled with additional monitoring data at different spatial and temporal scales to further test the accuracy of these models. Bainbridge *et al.* (2007) comment on the overestimation of particulate nitrogen exports due to the lack of information on the highly weathered, nutrient-poor landscapes. Lewis *et al.* (2009a) further comment on the overestimation of the sediment trapping efficiency of the Burdekin Falls Dam by the SedNet model. There is continual improvement and understanding of load models and ongoing refinements to existing models for the better estimate of loads, particularly in large dry catchments such as the Burdekin.

4.2.1 Current load estimates for the Burdekin catchment

Various efforts have been implemented to improve load estimation techniques in the Burdekin River, which is seen as a priority river for management of pollutant runoff, and as such, appropriate and accurate load estimates are required under the continual evolution of the Reef Rescue package. Improvements in the measurement of load estimates in the Burdekin catchment derived from MTSRF funded research are summarised below.

- A combination of monitoring activities and modelling tools link on-ground management at the paddock or plot scale to end-of-catchment resource condition loads, and finally to marine trigger values required for ecosystem protection. This combined approach overcomes many of the uncertainties associated with monitoring at shorter time scales as well as the shortfalls of the models available through data input and calibration (Bainbridge *et al.* 2009a; Brodie *et al.* 2009a).
- Load estimates for the Burdekin River have now been improved through the sediment trapping efficiency work by Lewis and others (2009a). This work had suggested that the classic Brune algorithm used to model trapping efficiency is not suitable for highly variable event flows that are characteristic of the Dry Tropics. Minor modifications to the Brune model have been suggested to better account for the event driven nature of reservoirs which receive highly variable inflow from tropical rivers.
- Accurate load estimates are able to be calculated from a field sampling program if adequate sampling occurs over the flow hydrograph. Lewis *et al.* (2007c) recommend six samples evenly spaced over the flow hydrograph (2-3 samples on rise, 1 on peak and 2-3 on falling limb) will provide reliable load estimates (within 10% of best estimate). Daily sampling is also recommended in catchments, with more frequent sampling for sub catchments with high TSS on the rising limb. However, a sample collected every two days is an adequate sampling frequency for load calculations of larger catchments such as the Burdekin River.
- Updated figures for the sources of DIN to the regional load were published by Brodie and Bainbridge (2008). Previous estimates reported by Brodie *et al.* (2003) did not include accurate modelling of the lower Burdekin sugar area. The updated method estimates surface and sub-surface annual losses of 3,000 tonnes (2,000 tonnes loss to GBR waters by surface pathway and 1,000 tonnes loss to GBR waters by groundwater pathway). Therefore, the overall total annual load estimate of DIN delivery to the GBR for the region is estimated to be 4,480 tonnes (Brodie and Waterhouse, 2009).
- Using the information above, estimates of the current and anthropogenic loads for the sub catchments in the Burdekin region have been developed (Brodie *et al.* 2009b; Brodie and Waterhouse, 2009). The majority of suspended sediment loads delivered to the GBR from the Burdekin catchment are derived from grazing land uses in the Burdekin basin (Figure 4.1), and grazing is the dominant source of sediments throughout the Burdekin catchment. Figure 4.2 illustrates the relative contributions for TSS, DIN and PS-II herbicides for each sub catchment. The Bowen-Bogie sub-catchment has the highest loads for TSS, while the Lower Burdekin has substantially higher loads of anthropogenic DIN than any other Burdekin sub catchment. This elevated DIN load is a consequence of the dominance of sugarcane in the Lower Burdekin.

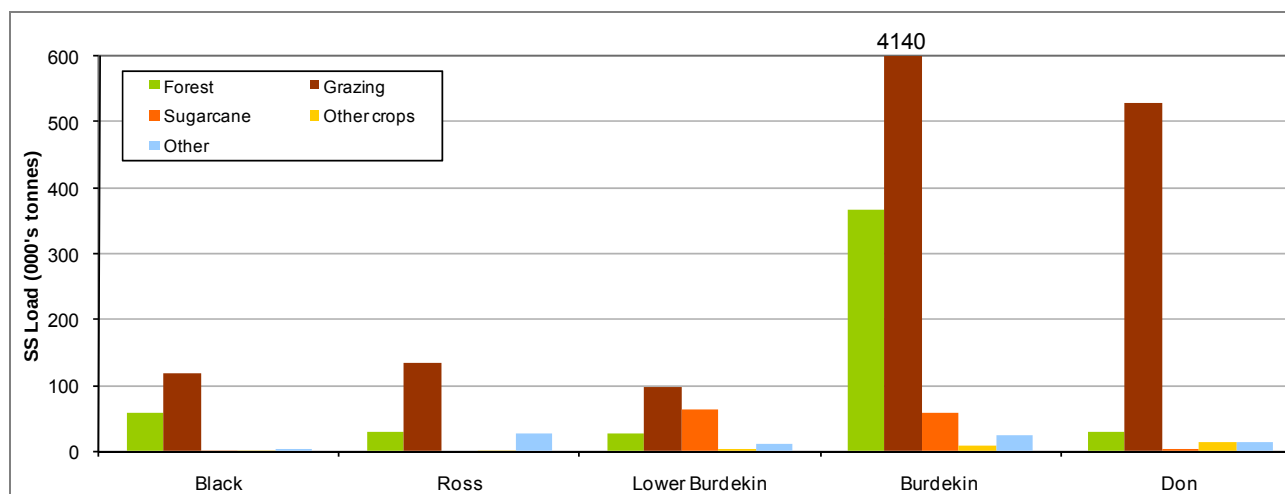


Figure 4.1: Estimates of SS load for the primary land uses in the Burdekin catchment. Source: Brodie *et al.* (2009b).

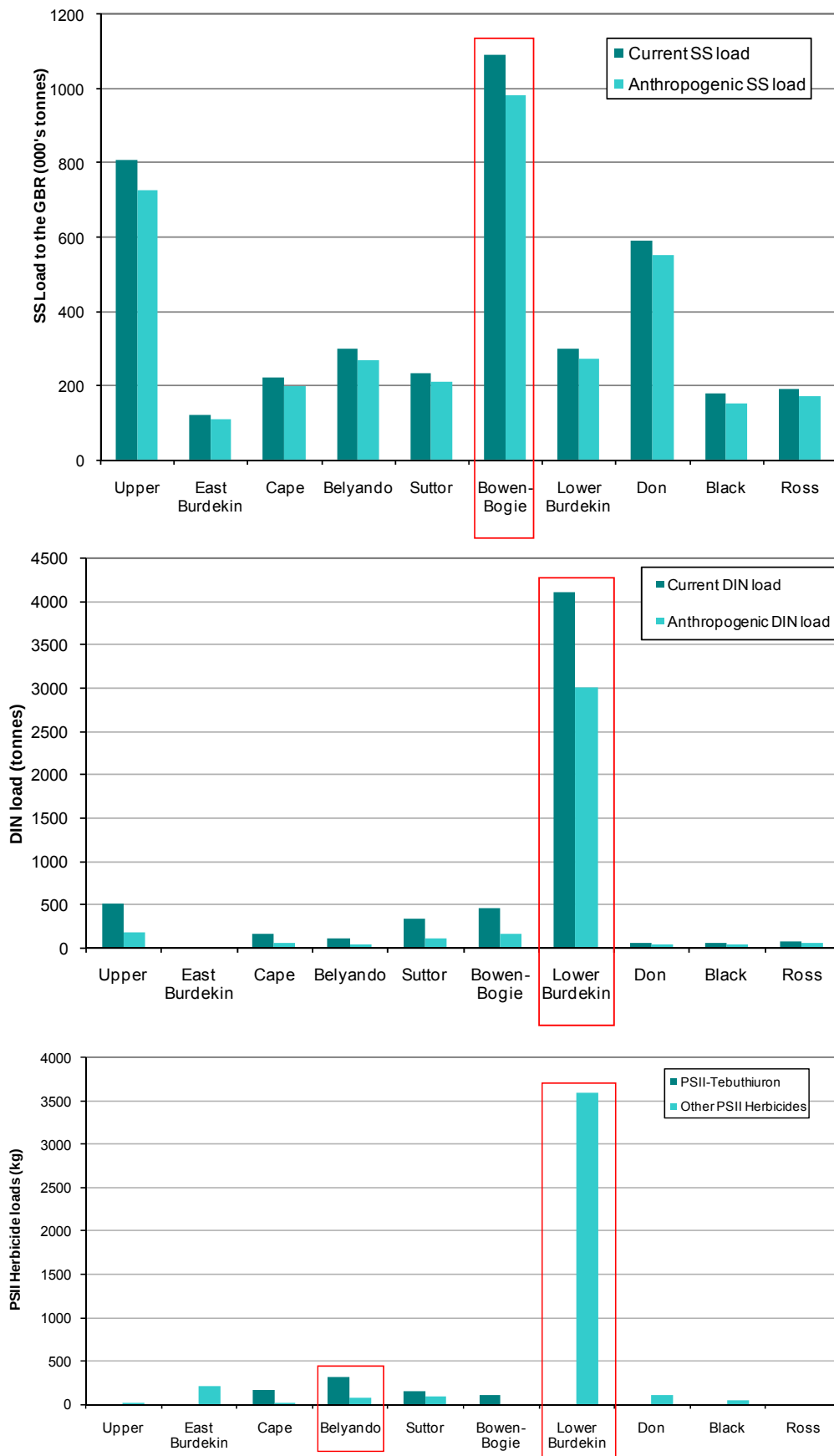


Figure 4.2: Load estimates for the Burdekin sub-catchments with the relative contribution of TSS, DIN and PS-II herbicides for each sub-catchment.

4.2.2 Improvements in load estimation techniques

Through the MTSRF, Kuhnert and others have developed improved methods for calculating pollutant loads in the GBR, using the Burdekin as one of the case study areas, representative of a dry catchment. The Loads Regression Estimator (LRE) package in the *R* programming language (Kuhnert *et al.* 2009; Kuhnert and Henderson, 2010) has been developed to estimate suspended sediment loads from the Burdekin catchment. This approach deals with the uncertainty inherent in the large episodic flow events that are typical of the Burdekin catchment. The use of flow characteristics (first flush, rising, falling limbs) captures key hydrological process to reduce knowledge uncertainty about the river system.

Sediment load estimates were calculated for the end of river system site, Inkerman Bridge using data that spanned 36 water years (Figure 4.3). The results (Kuhnert and Henderson, 2010) are summarised below.

- Annual loads and mean annual concentrations were estimated for the Inkerman Bridge site for 36 water years using the LRE methodology. Summaries of the data indicated considerable bias in the concentration sampling with no bias in flow samples due to the regular sampling intervals (hourly).
- A model was fit to 824 concentration samples where linear and quadratic terms for flow, a seasonal term and smooth terms for the discounted flow and trend were fitted. Results showed a reasonable fit with 69.9% of the variance explained. A seasonal term fit in the model showed increases in TSS concentration during the wet months (October to April) and decreases during the drier months of the year (May to September).
- Average mean concentrations were higher in some years than others. Further investigation revealed that where cyclones had passed through the Bowen sub catchment of the Burdekin, TSS concentrations were considerably higher. Inclusion of terms that reflect these events in the model may help explain the variability in concentration and load for this catchment.

Figure 4.4a displays load estimates and corresponding 80% confidence intervals for each water year in millions of tonnes, accompanied by the total volume of flow in megalitres. In this figure, the TSS loads are higher when the flow is large. To obtain a more accurate picture of loads through time, Kuhnert and Henderson (2010) standardised by flow and produced Figure 4.4b. This shows the average mean concentration in TSS across water years. Significant features of this plot are the large concentrations in 1985/86, 1987/88-1988/89 and later in 1996/97-1997/98. This is the period where cyclones passed through the Bowen sub-catchment of the Burdekin. Inclusion of terms that reflect these events in the model may help explain increases in concentration for this catchment. Furthermore, inclusion of a term that captured the Burdekin dam construction may also explain changes in concentration from 1987 onwards.

The use of multiple years and long-term weather conditions has helped build history and create a time series of flow and concentration characteristics that can be used to predict across time. Longer time frames may show a simple linear model which may not be apparent across shorter time frames. The datasets show that different terms are applicable to different responses (i.e. different water quality parameters). In the Burdekin River (Inkerman Bridge dataset), a seasonal term was more appropriate to estimate NO_x whereas for TSS, the rising/falling limb term was identified as a more appropriate fit (Kuhnert *et al.* 2008, 2009).

Nutrients were also investigated for this site in an alternative project examining baseline loads for the GBR (see Kroon *et al.* 2010).

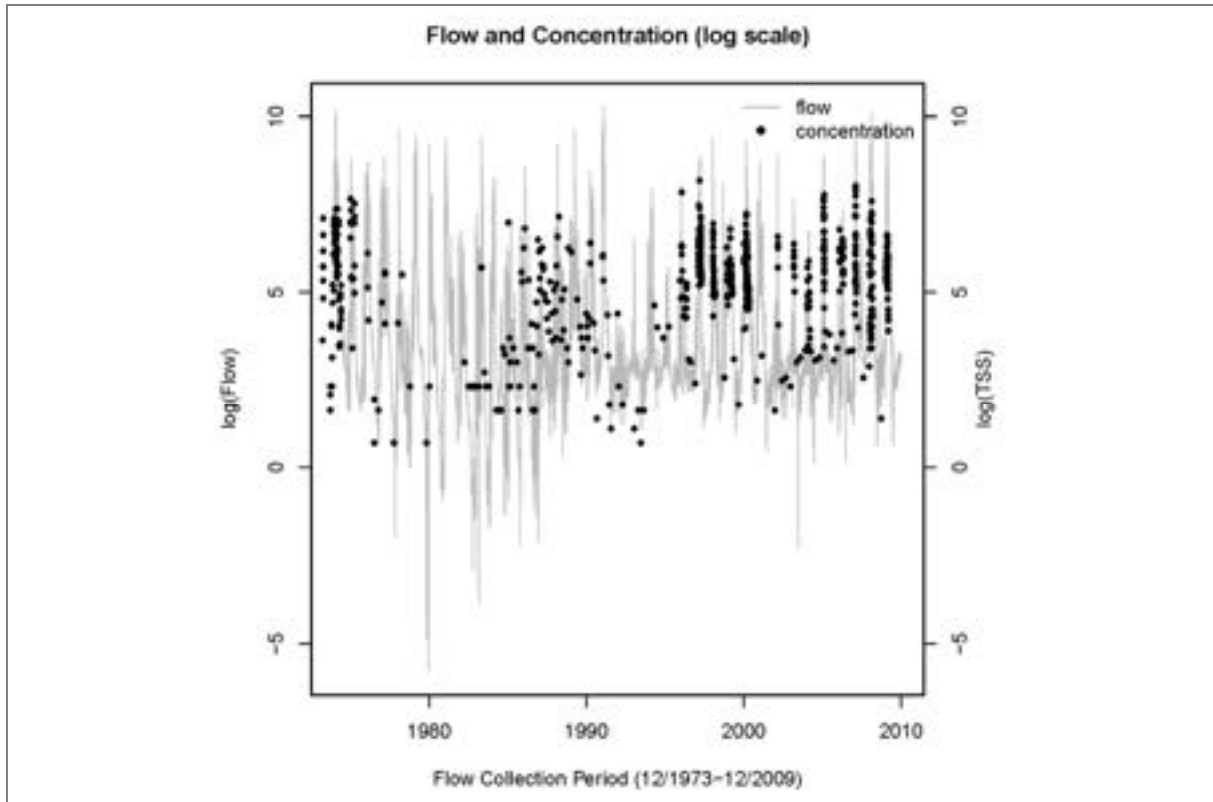


Figure 4.3: Flow (solid line) and TSS concentration (points) shown on the log scale and sampled from the Burdekin River site at Inkerman Bridge between December 1973 and December 2009. Source: Kuhnert and Henderson (2010).

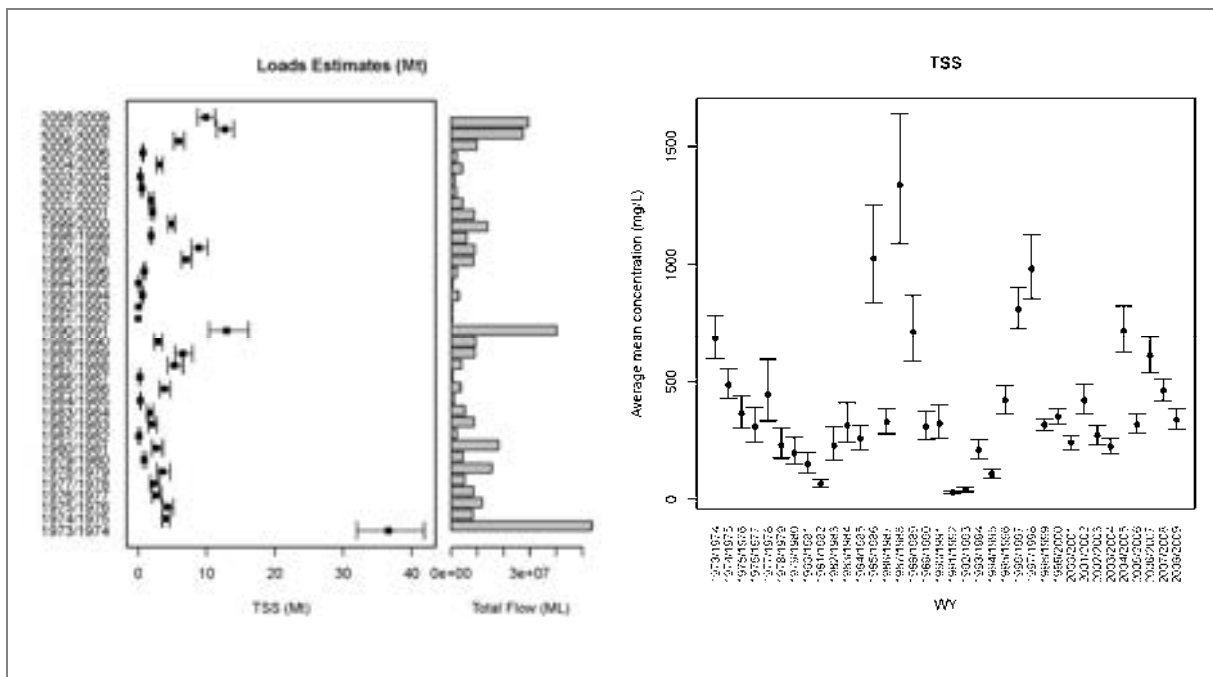


Figure 4.4: Plots for the Burdekin site showing (a) the estimated TSS load (Mt) and 80% confidence intervals for each water year accompanied by the total volume of flow (ML), and (b) the average mean concentration (mg/L) for each water year. Source: Kuhnert and Henderson (2010).

4.2.3 Burdekin Dam trapping

MTSRF funded research on the role of the Burdekin Dam in trapping materials from the upper catchment highlights the influence of the dam on sediment export and has established improved load estimations to determine priority areas for remedial works (Lewis *et al.* 2009a).

Current SedNet and ANNEX modelling of the Burdekin catchment has suggested that the Burdekin Falls Dam is a very efficient trap for sediment and particulate matter (Fentie *et al.* 2006; Post *et al.* 2006a, 2006b). More recent models have estimated that the Dam traps 77-82% of suspended sediment from the upper catchment and 79% of particulate nitrogen and phosphorus, with negligible trapping of dissolved materials (Fentie *et al.* 2006; Post *et al.* 2006b). However, MTSRF funded research using sediment traps, water column/bottom profiling and water sampling within the dam reservoir during flow events did not support this high trapping efficiency (Lewis *et al.* 2009a; in press).

Lewis and others have shown that higher trapping efficiency is a product of relatively small catchment flows and is driven by a lower dam level prior to the onset of the wet season (Lewis *et al.* 2009a). However, sediment trapping in the dam is typically 60-70% during moderate to very large flow events (Lewis *et al.* 2009a). A consistent trapping efficiency estimate of ~60% ($\pm 10\%$) is probably more reflective of an 'average' trapping estimate. This suggests that SedNet models have overestimated the trapping efficiency of the Burdekin Falls Dam.

The sediment trapping algorithm within the SedNet model is based on a well-established relationship between trapping efficiency and the ratio of reservoir capacity to annual inflow for 'normal ponded reservoirs', which receive runoff that is more evenly distributed throughout the year than is the case for the Burdekin River. This algorithm is not relevant for the Burdekin Falls Dam, which experiences strong thermal stratification and highly episodic flows and therefore shorter residence. Incorporation of a 60% dam trapping efficiency factor in current SedNet models would result in a recalculation of a greater value for an annual export of 3.5 million tonnes (Kinsey-Henderson and Sherman, 2007), which is close to the estimate of Furnas (2003) of 3.8 million tonnes and also to the flow-normalised loads (using the discharge records specified by SedNet) calculated over nine years of monitoring data of 4.6 million tonnes (Bainbridge *et al.* 2008; Kuhnert and Henderson, 2010).

The efficiency of the dam in trapping sediments will have important implications for land use management and prioritisation of remedial works. If limited sediments are being transported past the dam then remedial works above the dam will have a negligible effect on the amount of sediment and particulate matter being delivered to the mouth of the river and to the GBR lagoon. As much of the remedial work is targeted at reducing bulk sediment loads to the GBR, works above the dam would not be undertaken for this purpose. Lewis *et al.* (2009a, in press) show that in high flow events the Burdekin Dam reservoir would act more like a river than a dam. This finding supports the results of Cooper *et al.* (2006) who, using trace element and isotopic tracing methods, found that the bottom sediments within Lake Dalrymple were from the upper Burdekin River. Therefore, management to reduce bulk suspended sediment delivery to the dam should focus on the upper Burdekin River catchment area.

In large flows, the majority (80%) of the total suspended sediment load exported from the Burdekin River into the marine system is sourced from the catchment area below the Dam. This area below the dam comprises only ten percent of the total Burdekin catchment area. Thus, through this work, it is now recommended that remedial works to reduce the 'bulk' suspended sediment load exported from the Burdekin River should focus on the catchment area below the dam. However this assertion only relates to the management of the 'bulk'

suspended sediment supply and not to specific sediments which may travel further in the marine environment (i.e. dispersive clays) and thus may be more ecologically important.

4.3 Assessment of risk from the Burdekin catchment

In a collaborative project between the MTSRF and DERM, Brodie and Waterhouse (2009) provided a relative risk assessment of pollutants for regional NRM regions in the GBR catchments, incorporating the Burdekin region. To identify the relative risk for each region, they calculated a relative risk score using the formula shown below. This study is described in further detail in the companion report by Waterhouse and Brodie (2010).

Relative Risk = Anthropogenic Load Score The sum of scores for TSS, DIN, and PS-II herbicide anthropogenic loads (Source: Brodie <i>et al.</i> 2009a) + Reef Condition Score The sum of scores for Coastal and inner shelf macroalgal cover, Coastal and inner shelf hard coral richness, Coastal and inner shelf secchi depth, and Coastal and inner shelf chlorophyll (Source: De'ath and Fabricius, 2010). + Reef Exposure Score The sum of scores for TSS exposure, DIN exposure, and PSII herbicide exposure (Source: Maughan <i>et al.</i> 2008)

The Anthropogenic Load Score is calculated as the sum of categorised scores of anthropogenic sediment, nutrient and pesticide loads derived from Brodie *et al.* (2009b). The Reef Condition Score is calculated using the data analysed by De'ath and Fabricius (2008) to support the development of the Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). The Reef Exposure Score is calculated from the number of reefs that are located within the areas identified as high or very high risk of exposure by Maughan *et al.* (2008). The comparison of relative risk scores across the NRM regions is shown in Figure 4.5. From these results it is possible to develop a risk ranking between the NRM regions, indicating highest management priority to the Wet Tropics and Mackay-Whitsunday regions. While the Burdekin had the highest Anthropogenic Load Score, the Reef Exposure Score was relatively low, which is partly due to the large distance of the bulk of the reefs from the coast.

In the overall assessment, Burdekin sugarcane is a top priority in management of nutrient and pesticide loss and Burdekin grazing is a top priority activity in the management of sediment loss to the GBR. In general, improvements in PS-II Herbicide loads are expected to be the quickest to detect, followed by improvements in DIN due to fertiliser management, with considerable times to reduce suspended sediment due to erosion management. Thus, recommendations for management for Burdekin catchment could begin with herbicide and fertiliser management in sugarcane as reduction of loadings through application of best management practices, e.g. The Six Easy Steps¹¹, will see reductions and in the shortest timeframes. To achieve the load reductions with respect to sediment over the longer term, grazing land management should be implemented in the region as a priority.

¹¹ BSES Ltd Best-Practice Nutrient Management: The Six Easy Steps Approach
(<http://www.bses.org.au/InfoSheets/2008/IS08006.pdf>)

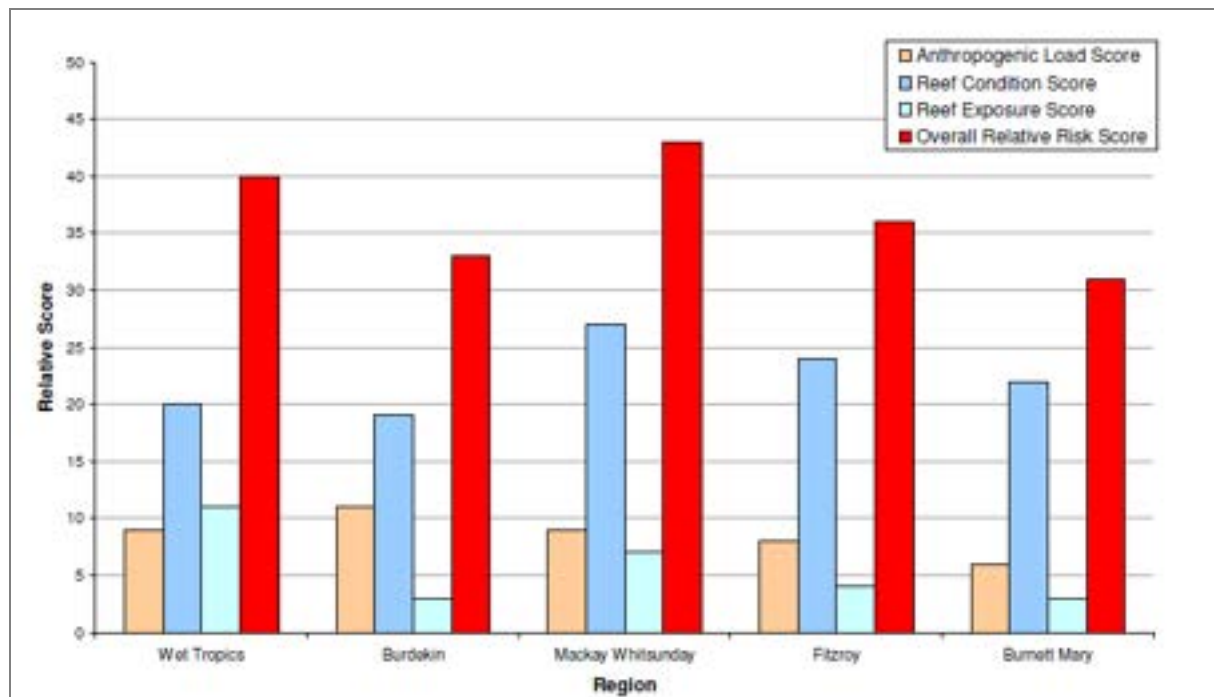


Figure 4.5: Comparison of the relative risk scores for the GBR catchments.
Source: Brodie and Waterhouse (2009).

4.4 Socio-economic influences in the Burdekin catchment

Socio-economic research into the cost effectiveness of the recommended Best Management Practices (BMPs) for the major industries (sugarcane, horticulture, grazing and forestry) within the GBR catchment area has been conducted across the entire region (Roebeling, 2006; Roebeling *et al.* 2007b, 2009; van Grieken *et al.* 2010a), and specifically for the Tully-Murray and Burdekin Water Quality Improvement Plan (WQIP) processes (Roebeling *et al.* 2004; 2007b; Bohnet *et al.* 2006, 2007; Roebeling and van Grieken, 2007; Roebeling and Webster, 2007; van Grieken *et al.* 2009, 2010a, 2010b, 2010c). This approach also incorporates production system simulation models (such as APSIM) and the SedNet and ANNEX models (Roebeling *et al.* 2007b). Socio-economic research has also been conducted by Lankester and Greiner (2007) for the Burdekin WQIP, focusing specifically on the grazing industry within this region, which found that a suite of incentive measures would be required to encourage the adoption of BMPs within this region. Henderson and Kroon (2009) present a comprehensive summary of the economic and environmental effects of changes in land use and land management for water quality improvement. This report highlights the advantages of a coupled environmental-economic approach to sustainable water quality management when identifying management action targets.

In grazing lands, one of the most important management influences on the amounts of sediments entering waterways is pasture management and maintaining a good ground cover of perennial tussock grasses. Where pasture cover is reduced through poor grazing management, the soil is exposed more strongly to the effects of erosive tropical and subtropical rainfall. The accompanying loss of soil is not only an issue for sediment loads and their impacts downstream, but can also permanently reduce the productive capacity of pastures (creating a vicious cycle where stocking rates need to be further reduced if continued overgrazing and pasture deterioration are to be avoided). One of the primary management influences to reduce sediment flows off grazing lands is therefore improved pasture/grazing management to maintain and enhance the protective cover of perennial

grasses. This would include practices such as setting stocking rates that are safely matched to the productive capacity of the land, wet season spelling to assist recovery of degraded pastures, and strategies for dealing with climate variability, such as conservative stocking rates or adjusting stocking rates based on seasonal forecasts (soils are particularly susceptible to erosion when good rains return after droughts if grazing management has not maintained protective ground cover during the drought). Sugarcane in the Burdekin is still a significant activity that has important repercussions for downstream water quality. For sugarcane, the economic analysis to determine implementation costs entails a few components. For each farming system (van Grieken *et al.* 2010a) the long-term productivity costs and benefits (gross margins) are assessed along with the investment costs to facilitate change to improved farming systems (van Grieken *et al.* 2010b; 2010c) (Table 4.2 and 4.3).

Table 4.2: Financial costs for each farming system in the Burdekin Delta. A = Aspirational / commercial viability not yet proven; B = Best practices; C = Common practice; D = Dated practice. Source: van Grieken *et al.* (2010a).

Farming system	Average gross margin (\$/ha/yr)	Investment costs (\$)
A	4,015	83,000
B	4,182	53,250
C	3,530	-
D	3,231	-

Table 4.3: Financial costs for each farming system in the Burdekin River Irrigation Area. Source: van Grieken *et al.* (2010b, 2010c).

Farming system	Average gross margin (\$/ha/yr)	Investment costs (\$)
A	2,119	86,000
B	2,199	43,250
C	1,769	35,000
D	1,378	-

There are both costs and benefits involved in moving to 'improved' farming systems (grouping of management practices) in the Burdekin basin (represented by the Delta region) (Figure 4.6). In other words, farming systems that have the potential to reduce available dissolved inorganic nitrogen (DIN) (from runoff and deep drainage leaching) to enter the waterways from the paddock in the corresponding regions.

For each farming system (steady state D, C, B and A, not in transition) the corresponding information is given on DIN (kg/ha/yr) available to enter the (hydrological) system, farm gross margins (AU\$/ha/yr) and the investments (e.g. machinery) required to move from one farming system to the other. It must be noted that transaction costs (or hidden costs of change) are not incorporated; hence total costs of change are likely to be underestimated.

For example, in the Burdekin Dry Tropics, moving from a C farming system to a B farming system will require (for a 120 ha sugarcane farm operating in the Delta area on a relatively coarser texture soil and where groundwater is used for irrigation) the investment of approximately AU\$45,000 (e.g. the purchase and modification of machinery). It will potentially reduce DIN pollution from the paddock while increasing (steady state) farm gross

margins. A few examples of the changes that farmers face to move from a C to a B farming system are the use of GPS for planting, a reduction of tillage operations, fertiliser application rates based on soil tests, the use of legume crops in half of the fallow area, the development of a soil management plan, improved record keeping and the use of climate and weather forecasts.

In conclusion, for the Burdekin catchment, change in (dominant) agricultural practices can be summarised as below:

Sugarcane: Improved practices may lead to increased productivity benefits but show significant investment costs. It must be noted that costs and benefits associated with a transition will be different for each individual grower and therefore each circumstance requires careful consideration before making a change in management practice.

Grazing: In general, improved practices (reduced stocking rates) come at a productivity loss. However, in some cases, reducing stocking rates may lead to benefits (increased gross margins).

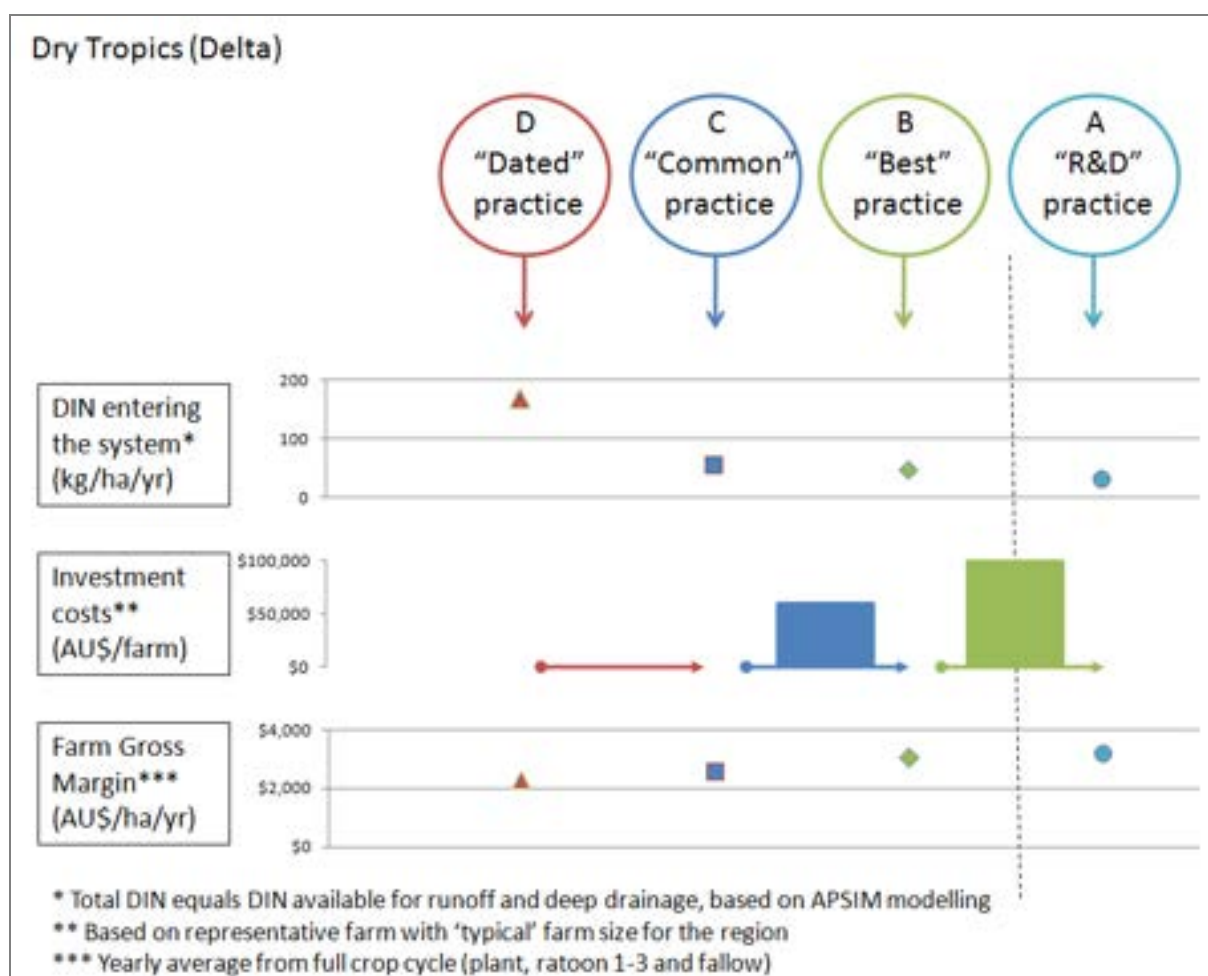


Figure 4.6: Investment costs from moving from one agricultural system to another for sugarcane activity in the Burdekin Delta. Source: M. van Grieken.

4.5 Transport of Burdekin pollutants to the marine environment

The Burdekin River is a large system with an average annual discharge of 8.5 million ML compared to other rivers/creeks in the region, such as the Haughton River (average annual discharge of 0.37 million ML) and Barratta Creek (0.28 million ML). The marine extent of these smaller, localised streams is probably limited to Bowling Green Bay (Lambrechts *et al.* 2010) and any potential environmental impacts from these streams would be restricted to this area. However, it should be noted that Bowling Green Bay contains a number of important marine and estuarine environments, including mangroves and seagrass beds. Bowling Green Bay is also a well-known recruitment area for juvenile marlin and sailfish. The export of nitrate and herbicide residue from these lower Burdekin streams may therefore be a concern to these environments.

Status of Burdekin marine areas

In the Burdekin region, water quality, flood plumes, seagrass habitats, and coral reefs are all monitored as part of the Reef Rescue MMP, including a number of sites in the Burdekin marine area. Overviews of the results are presented in Johnson *et al.* (2010), Prange *et al.* (2007, 2009) and Haynes *et al.* (2005). A few key points included below from an assessment of data from 2006 to 2009 included in Johnson *et al.* (2010) provide an indication of the status of water quality and ecosystems health in this area.

- Guideline values were exceeded at all sites for wet and dry season means of chlorophyll and suspended solids. Geoffrey Bay (Magnetic Island) generally had the highest seasonal means of all sites in this region, and the means of all variables, except for particulate nitrogen in the wet season, exceeded guideline values.
- Spatial representation of the values exceeding the guideline for chlorophyll and suspended solids identifies the area between Magnetic Island and the Palm Islands as being at highest risk.
- Seagrass abundance declined in the latter part of the 2008-2009 sampling period at coastal habitats and was variable at reef habitats. Seagrass reproductive health status over the period 2006 to 2009 was assessed to be 'good' at the Townsville coastal site, and 'variable' at Magnetic Island. Seagrass tissue nutrients indicate a potentially low light environment at all sites. Decreasing C:N ratios at coastal sites since 2006 indicate decreasing light availability.
- Coastal and reef habitats were found to be nutrient rich (large phosphorus pool), with nutrient availability to the plant phosphorus limited at the coastal site and replete at the reef site. However, epiphytes declined at the Townsville coastal site and were variable at Magnetic Island, following similar patterns as seagrass abundance.
- Coral community status had a negative score, with overall status lower than the Wet Tropics region to the north and the Mackay-Whitsunday region to the south because of the high occurrence of disturbance events in the region, including significant bleaching events in 1998 and 2002, and several major flood events since 1990.
- Coral recruitment had a negative score which could also be related to disturbances and large flood events in the last two years.

Materials in the Burdekin River discharge may be transported in two phases: dissolved or particulate forms. For example, nutrients may be transported in either the dissolved form (dissolved inorganic or organic) or adsorbed onto sediment particles (particulate). Similarly, pesticide residues may also be transported in either form depending on their chemical properties. The fate/transport of these two phases/forms in the marine environment is often considerably different: the dissolved forms are commonly transported long distances with the

freshwater plume where they become diluted with seawater mixing or consumed by biological activity, whereas the particulates are typically deposited in close proximity (~15 km) to the river mouth before being reworked and redeposited in low energy, north-facing embayments.

Aerial mapping (Devlin *et al.* 2001), direct water measurements (Wolanski and Jones, 1981; Devlin *et al.* 2001; Devlin and Brodie, 2005) and satellite imagery (Brodie and Bainbridge, 2008; Devlin *et al.* 2010) show that most sediment is deposited within ten kilometres of the Burdekin delta or within the 10 PSU (practical salinity units) salinity zone. Suspended solid concentrations rapidly fall from ~500-1000 mg/L to <50 mg/L around the 5-10 PSU salinity gradient (around 5-10 km from the river mouth) from biological and physio-chemical processes that cause fine sediment to flocculate and fall out of suspension (Brodie *et al.* 2004; Devlin and Brodie, 2005). Suspended sediment concentrations in the 10-35 PSU salinity zone of the Burdekin River plumes are commonly between 1-10 mg/L which suggests that extremely fine-grained sediment and colloidal particles may travel large distances with the freshwater plume in the marine environment (Wolanski *et al.* 2008; Lambrechts *et al.* 2008).

4.5.1 Burdekin sediment characteristics

The type and composition of the sediment that flows out to the GBR has different impacts on the marine environment. Large, heavier particulate matter tends to fall out quite quickly once it enters the marine zone. This process can be seen in plume water quality data (Devlin and Brodie, 2005) where there is a rapid fall in the high suspended sediment concentrations as the plume waters move into the marine zone. However, the finer sediments can and do move much further offshore and can have significant impacts on marine ecosystems. As part of the MTSRF Research Program, Bainbridge and others have demonstrated that the particle size in Burdekin flow waters is highly variable over any single flow events across all major river arms and also the Burdekin Falls Dam overflow. This result suggests that different sources of suspended sediment are being transported from different catchment areas during flow events. All four major river arms upstream of the dam drain considerable catchment areas and also contain several different rock and soil types (Lewis *et al.* 2009a). Generally, the dominant particle size fraction measured at all Burdekin sites was in the fine to medium silt range.

The particle size distribution data (Bainbridge *et al.* 2008) indicate that wash load (mud fraction = clays and silt size particles) suspended sediment is being sampled exclusively in the surface samples collected from the four major river arms upstream of the Burdekin Falls Dam. Since no particles in the four upstream river arms were coarser than those measured in the dam overflow waters, all particles have the potential to remain in the wash load and pass over the dam spillway. Previously it was thought that most of the 'fine-grained' particles were derived from the southern Belyando and Suttor River arms of the Burdekin (Faithful and Griffiths, 2000), however, data show that similarly fine particles can also be derived from the upper Burdekin and Cape Rivers.

4.5.2 Sediment transport

The majority of fine sediments exported to the GBR lagoon are reworked from the Burdekin Delta and transported northwards along the coast by re-suspension from southeasterly trade winds, where the sediments are deposited back onto the coast or retained in sheltered, low energy north facing embayments (Woolfe and Larcombe, 1999). Most of the fine sediment particles (~80-90%) exported from the Burdekin River are trapped in Bowling Green Bay (Orpin *et al.* 2004; Lewis *et al.* 2006). Sediment accumulation on the inner shelf is also relatively low (<0.2 mm per year: Belperio, 1983). There is a small but significant percentage of the extremely fine particles (clays and colloids) transported large distances in the Burdekin

plume and the final fate of these materials in the marine environment was largely unknown. Recent work has shown that a component ($<38\ \mu\text{m}$) of the clay fraction is transported in the adjacent marine environment past the initial turbid plume (Bainbridge *et al.* 2009a, 2009b). This turbid inshore plume is characterised by large particulate material that moves into the marine lagoon and deposits quickly through the processes of physical and biological flocculation. It is the movement of the finer fraction which has been linked to the degradation of coral reef ecosystems and therefore may pose the greatest risk to receiving marine ecosystems.

Cleveland Bay modelling of the movement of sediment (Lambrechts *et al.* 2010) has shown that sediments were redeposited in Cleveland Bay from the combined processes of wave mixing and shear induced erosion (prevailing during cyclonic conditions). Application of the sediment model indicated that for present land use conditions, the amount of riverine sediments settling on the bay may exceed by 50-75% the amount of sediment exported from the bay. Thus the increased sediment runoff from the Burdekin catchment will cause the accumulation of sediment in the bay on an annual basis which, in turn, may degrade the fringing coral reefs. For those years when the bay was affected by a tropical cyclone there may be a net sediment outflow from the bay. During the dry, tradewind season, fine sediment was progressively winnowed out of the shallow reef waters. This has implications for land-based remediation measures. Land-based management that reduces the amount of riverine fine sediment inflow into Cleveland Bay would reduce the length of time when high turbidity prevails over seagrass and corals. Order of magnitude estimates suggest that if land management policies were implemented in the catchments of the Ross and Burdekin Rivers to reduce by a factor of four the fine sediment discharge, the turbidity in Cleveland Bay would be halved after 170 days following cessation of river floods, which, in turn, would promote seagrass and reef growth.

4.6 Extent and potential exposure of Burdekin plume waters

Monitoring of flood plumes is an important component of the Reef Rescue MMP, where the spatial extent of the riverine waters is monitored and assessed in conjunction with other biological and chemical parameters (see Devlin *et al.* 2009, 2010; Johnson *et al.* 2010). In a collaborative effort between the Reef Rescue MMP, James Cook University and the CSIRO, a combination of field and satellite image mapping is now used to map the extent and concentration associated with GBR flood plume (Devlin *et al.* 2009, 2010). Remote sensing is more cost-effective and more informative for a variety of detection, monitoring and process understanding tasks. True colour imagery and application of appropriate algorithms have been used to develop a better understanding of the extent of plume waters in relation to weather and flow conditions (Figure 4.7). Advanced algorithms have been applied to plume imagery to calculate concentrations of TSS, chlorophyll and CDOM during and after a significant flow event to trace the extent of water quality parameters at peak concentrations (e.g. Brando *et al.* 2010). The extent and concentrations of plume waters, coupled with extensive *in situ* water quality sampling has been used to estimate the risk of plume exposure in inshore biological systems within GBR waters, with several monitoring campaigns in the Burdekin region (Devlin *et al.* 2009).

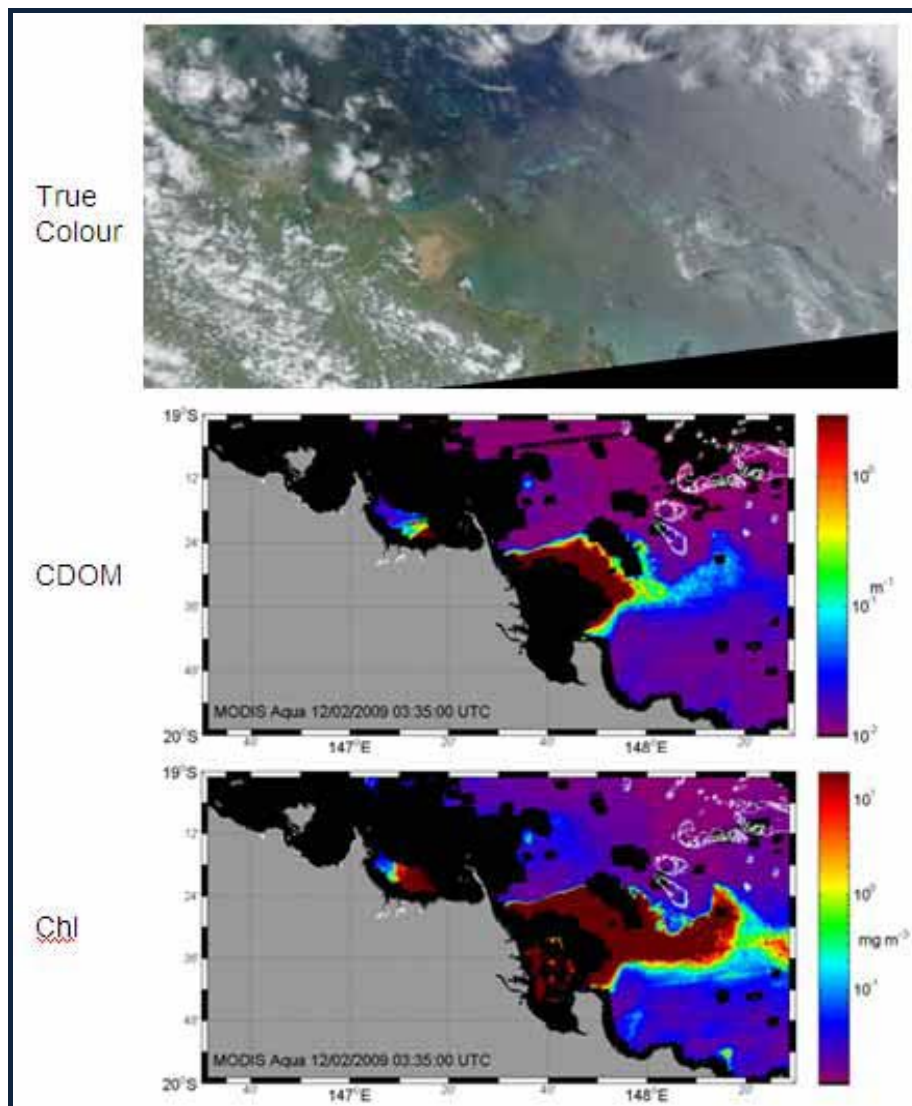


Figure 4.7: Example of a Burdekin plume image, reworked as a true colour image, with CDOM and chlorophyll algorithms also applied to the image. Source: ACTFR, JCU.

The optical complexity and variability of GBR coastal waters is illustrated by a MODIS true colour (RGB) composite acquired on 10 February 2007 covering the catchment of the Burdekin River and Repulse Bay in the Mackay-Whitsunday region (Figure 4.8). Intense wet season rainfall caused rivers in this region to produce large discharges to the GBR lagoon. The image captures the full variation of colour or, more precisely, spectral reflectance, ranging from deep blue open ocean waters to more green and brownish coastal waters. This satellite image illustrates as well the influence of land use on the composition of the flood waters. In the north, the Burdekin River discharges high loads of inorganic sediments into the lagoon, while further south Repulse Bay, with regional land use dominated by sugarcane cultivation and beef grazing, receives high loads of dissolved organic matter (Brodie *et al.* 2010).

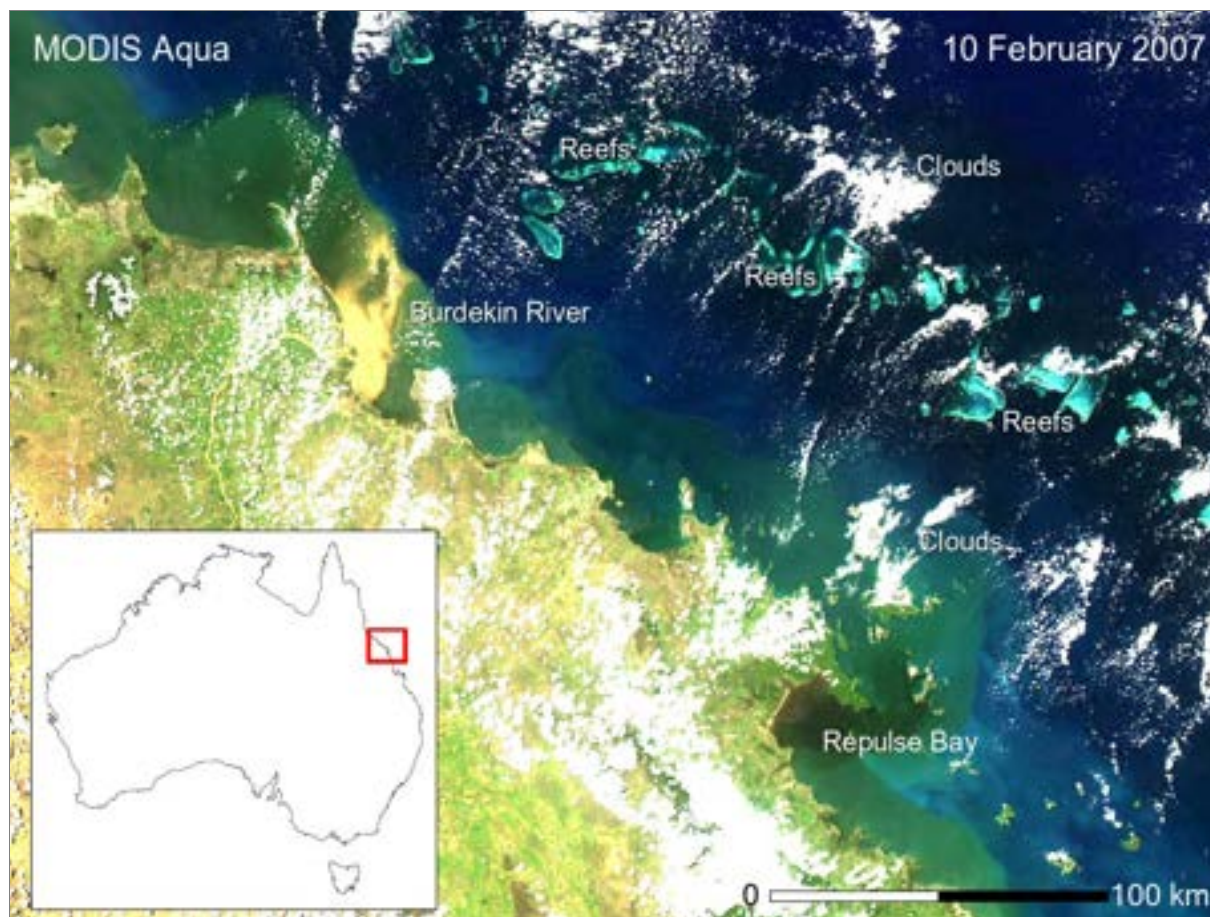


Figure 4.8: MODIS Aqua imagery acquired 10 February 2007, showing a sediment dominated flood plume of the Burdekin River, and a plume dominated by dissolved inorganic matter in Repulse Bay. Source: CSIRO.

Devlin and others have calculated a plume exposure map (Figure 4.9) for the Burdekin region from the intersection of the plume image and type from both aerial surveys (1996-1999) and remotely sensed images (2002-2009) from the Burdekin marine area (Devlin and Schaffelke, 2009; Devlin *et al.* 2009). Water characteristics were identified in each image, with the primary water type, characterised by high turbidity, high sediment plume discharging relatively close to the river mouth. Plumes characterised by lower turbidity (low values of TSM) and higher production (identified by elevated chlorophyll values) are usually measured in the middle salinity ranges (5-25 ppt). Turbidity (measured as TSM) may change through these secondary waters as a result of the offshore transport of the finer particulate material and desorption processes. Plume water types moving further offshore may be characterised by elevated CDOM and/or chlorophyll *a* values. The area can be mapped much further offshore and north of the river mouth than the visually evident primary and secondary waters.

The number of reefs and seagrasses exposed to the plume waters varies from year to year, and is dependent on the type of plume. Attempts to quantify this exposure have recently been undertaken for the GBRMPA and are described below (Devlin *et al.* 2010). The primary extent of the Burdekin plume, defined by higher sediment carrying waters, is shown to regularly move past Cape Upstart. High exposure areas are identified between Cape Upstart and Magnetic Island. Plume waters moving beyond Townsville are most likely to be influenced by the smaller rivers between the Burdekin and Cleveland Bay. Medium to high exposure areas are identified offshore of the Burdekin rivers moving past Magnetic Island.

Medium to low exposure areas are identified all the way past the Palm Island group and moving towards the offshore reefs (Figure 4.9).

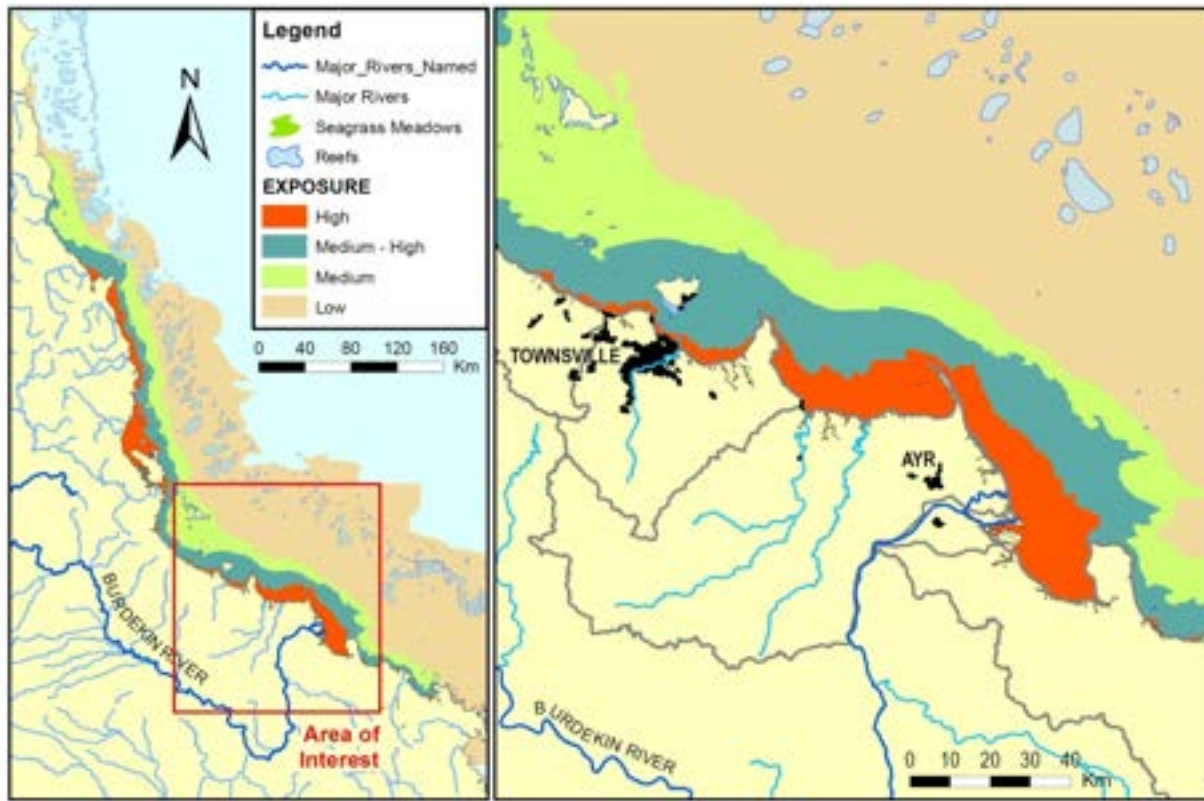


Figure 4.9: Exposure map for the Burdekin marine area. Image constructed from GIS imagery of plume extents from 1994 to 2008. Source: Devlin *et al.* (2009).

Figure 4.10(a) shows the occurrence of water types common in plume waters, including a transitional phase of fresh water to primary, secondary and tertiary water types. Further analysis of the plume types and their frequency has been coupled with a catchment ranking based on pollutant risk. Pollutants included nutrients (DIN), TSS and pesticides. This risk ranking coupled with plume exposure allows us to create a preliminary map identifying the area most likely to exceed water quality guidelines for chlorophyll (Figure 4.10b) and TSS (Figure 4.10c). The number of excess TSS levels for the Burdekin is high, even through the secondary waters, and as such forms the most potentially damaging pollutant currently discharging from the Burdekin catchment. Current Reef Rescue management practices for erosion reduction in the Burdekin catchment will be instrumental in dealing with this issue. The persistence of these elevated concentrations of chlorophyll and suspended solids has yet to be shown. However, cases of guideline value exceedence have been identified in flood plume water quality data and used to extrapolate plume behaviour in correlation with river flow and remote sensing images (Johnson *et al.* 2010; Devlin *et al.* 2009, 2010). Further work to obtain integrated time series data throughout high flow events, including more extensive sampling of depth profiles and continuous *in situ* logger data in combination with *in situ* surveys of coral reefs, will assist in improving the correlation of flood monitoring data with the long-term changes in pollutant concentrations and ecological impacts.

Another new approach to mapping the extent of freshwater discharge to the GBR is currently under development, and uses only the regional parameterised CDOM product applied to MODIS data as a surrogate for low-salinity waters. A maximum CDOM absorption map is

generated from January to March of each year through aggregation of daily CDOM imagery for the Burdekin region. By applying a CDOM cut-off threshold, previously defined from linear regression of *in situ* CDOM and salinity measurements, fresh water extent can be mapped, as illustrated in Figure 4.11. Ongoing work on the relationship between CDOM and salinity will be useful in the further validation of this mapping method.

Mapping of plume extent through gradients of water quality variables, qualitative assessment of true colour and aerial imagery, application of appropriate algorithms and catchment information all identify the Burdekin as a 'hot spot' area, with high concentrations of sediments and nutrients discharging during high flow events. Wooldridge and others (2006) modelled plume exposure with DIN and salinity ratios, and identified the area from the Tully to south of the Burdekin as having very high probability of runoff enrichment. Coupled with this is the Burdekin hydrology, where high flow events usually discharge over a period of weeks rather than days. Thus, based on this information and the exposure mapping by Devlin *et al.* (2010), the marine environment adjacent and north of the Burdekin River is considered to be at risk from the impacts of high levels of sediment and nutrients and potentially high levels of pesticides.

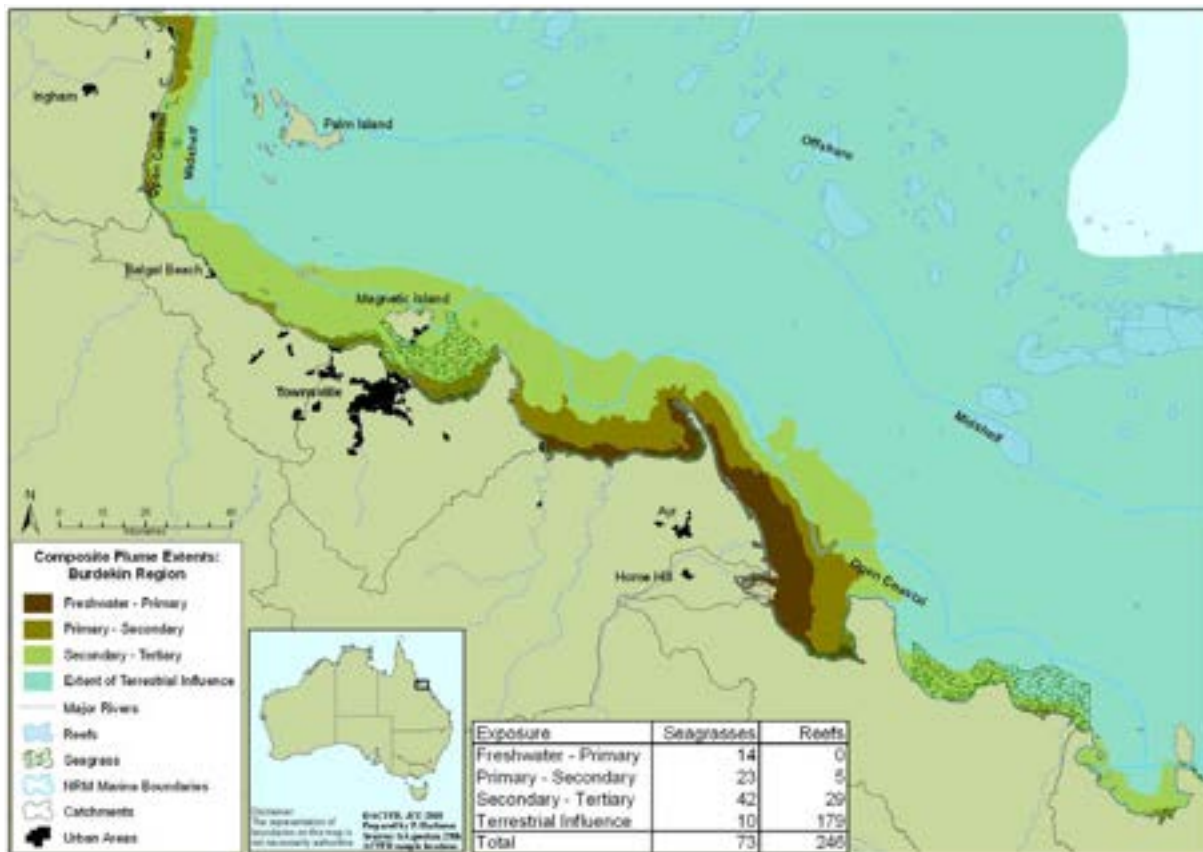


Figure 4.10(a): Extent of plume types in the Burdekin marine area. Source: Devlin *et al.* (2010).

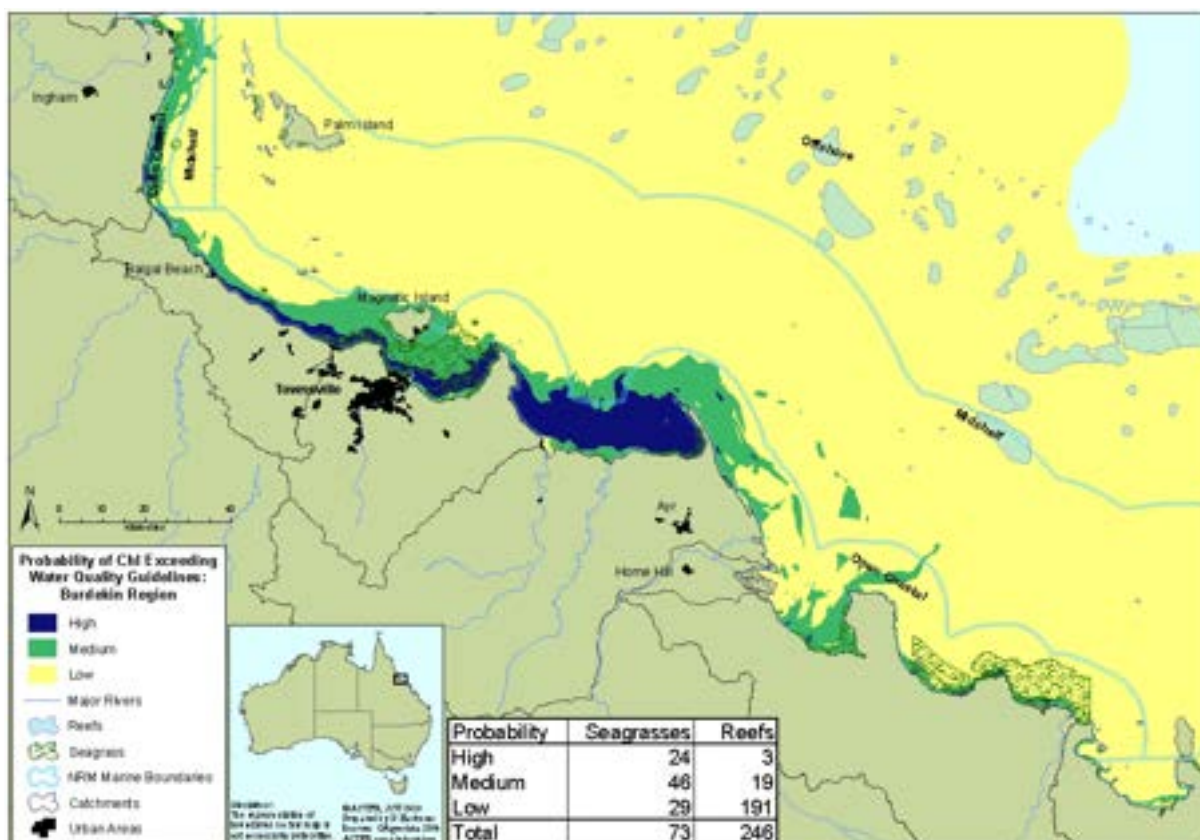


Figure 4.10(b): Identification of areas most likely to exceed water quality guidelines for chlorophyll in the Burdekin marine area. Source: Devlin *et al.* (2010).

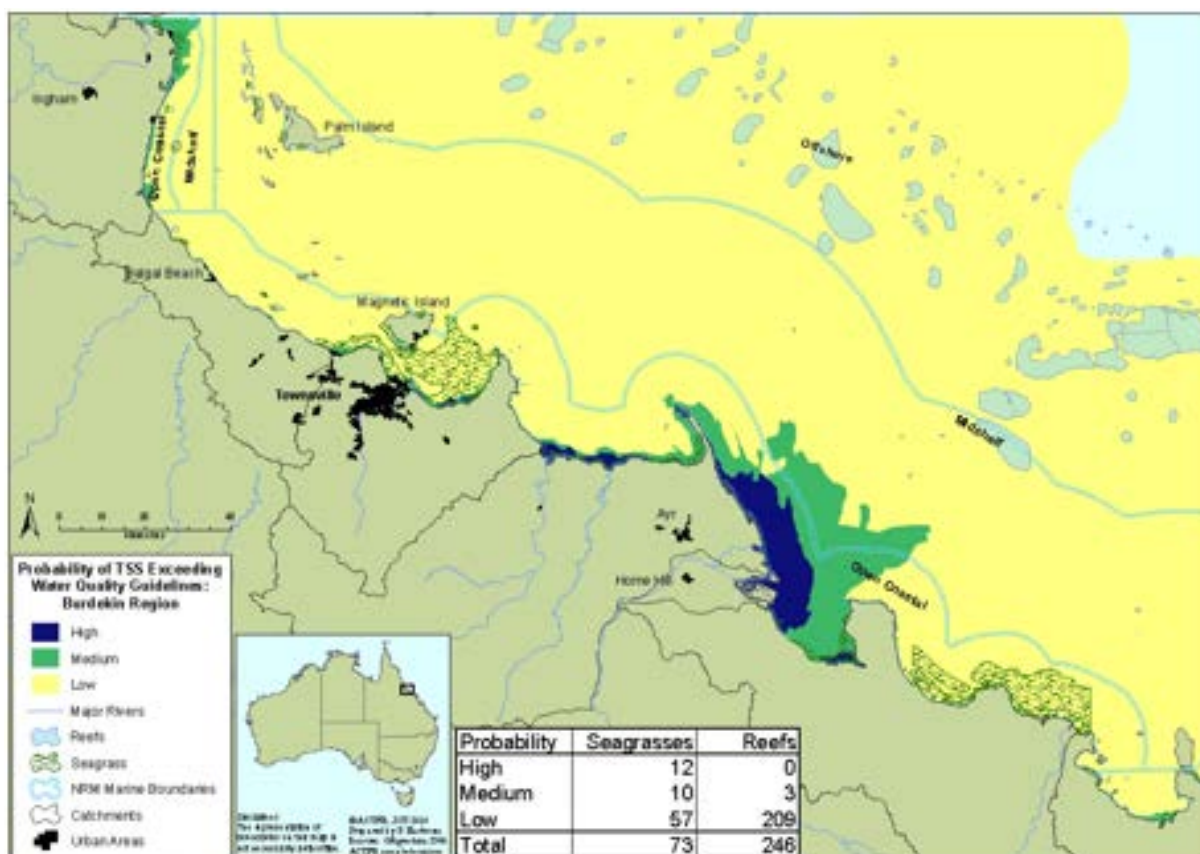


Figure 4.10(c): Identification of areas most likely to exceed water quality guidelines for TSS in the Burdekin marine area. Source: Devlin *et al.* (2010).

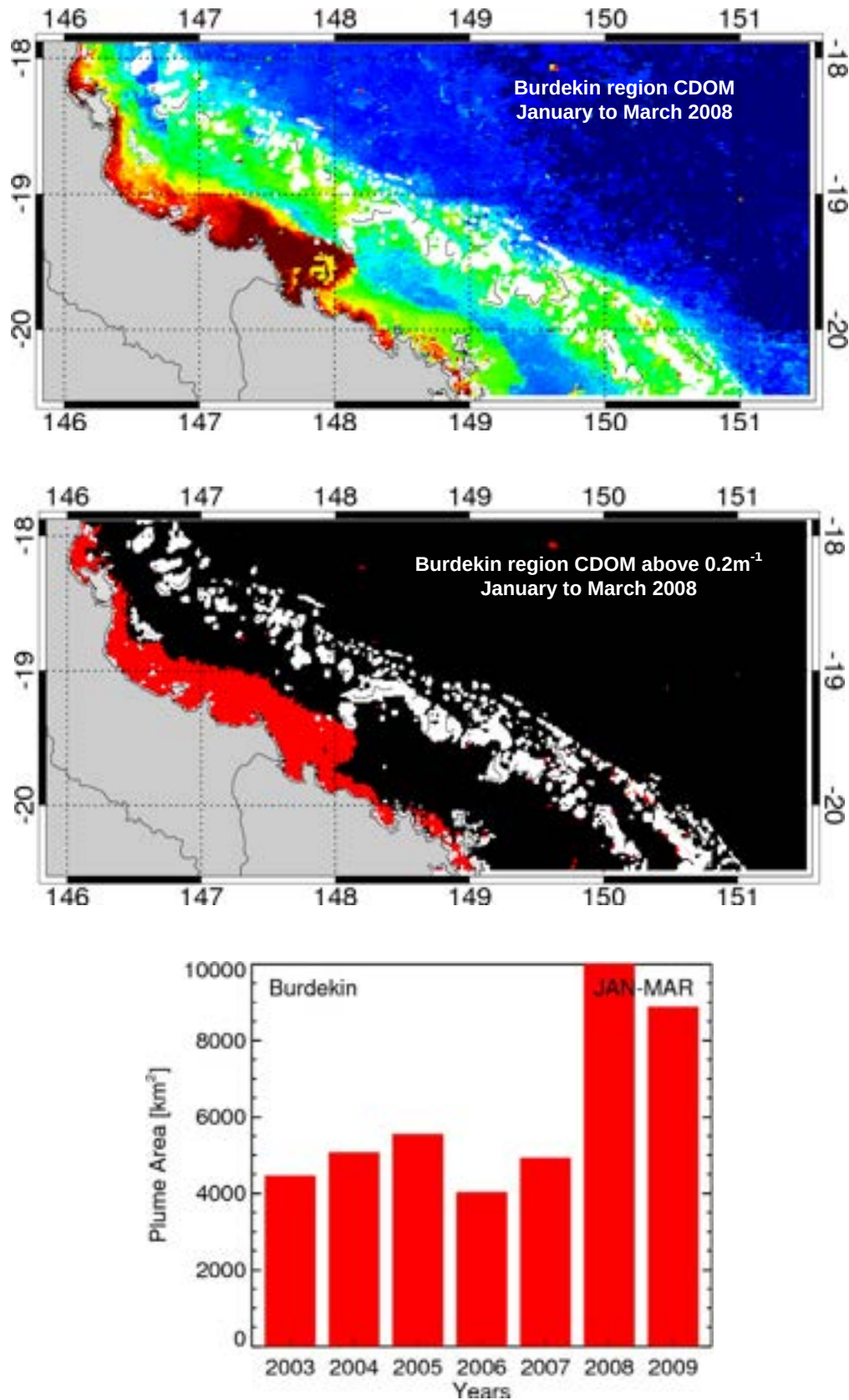


Figure 4.11: Maximum CDOM absorption from regional parameterised ocean colour algorithm mapped for the period January to March 2008 for the Burdekin region. Fresh water plume extent is mapped by applying a CDOM threshold derived from linear regression of *in situ* CDOM and salinity measurements. Source: Brando *et al.* (2010).

4.7 Improved understanding of the conceptual model for the Burdekin

A number of conceptual models have been produced for the GBR (Haynes *et al.* 2007) which outline our current understanding of the connectivity between catchment and reef, and movement from small-scale paddock sites, through wetlands, rivers and the coastal environment. MTSRF funded research presented in this summary and in companion reports to this summary (Waterhouse, 2010; Waterhouse and Brodie, 2010) have improved our understanding of this conceptual model, both for the whole GBR and also specifically for regional areas such as the Burdekin. Recent work focused on the Burdekin Falls Dam and trapping efficiency, the factors that supply, affect and influence sediment loads in the Burdekin catchment and marine environment, use of appropriate monitoring frequency and site selection and better statistical tools have all increased our understanding of pollutant loads, particularly sediment, moving out from the Burdekin catchment. Work on the extent, frequency of flood plumes, the mode and method of pollutant transport, better understanding of where impacts may occur and appropriate indicators which allow us to identify the area of impact have also increased our understanding of how the Burdekin catchment can affect the adjacent marine environment. Regional summaries are essential for the greater understanding of the long-term, chronic impacts that land based changes can have on GBR ecosystems. Figure 4.12 summarises the advances of understanding in a recent conceptual model from the Burdekin catchment (Prange, 2007).

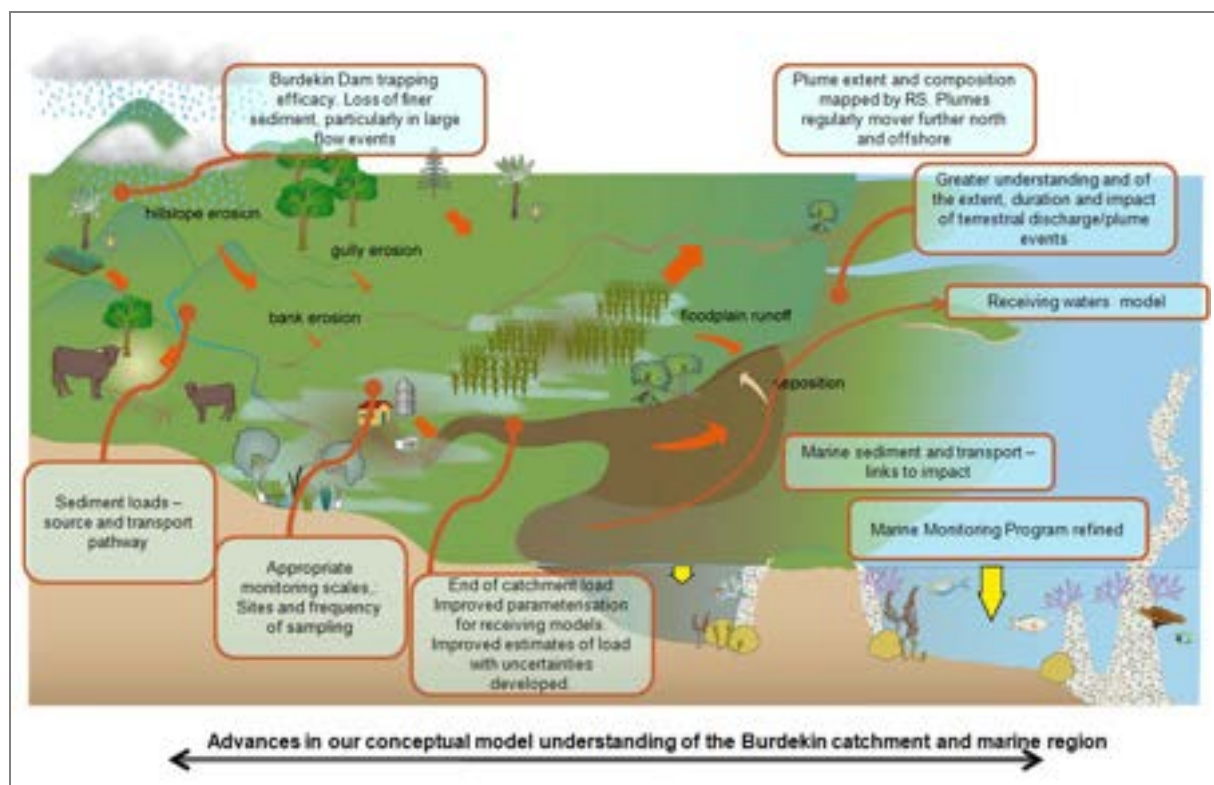


Figure 4.12: Changes in our understanding of the current Burdekin catchment conceptual model. Base model developed by Prange (2007).

MTSRF funded research has focused on the continuum between catchment and reef and that is evident with the findings that have been presented here for the Burdekin catchment and adjacent marine environment.

The detection and reporting of end-of-catchment loads is a key parameter in the Australian Government's Reef Rescue package to reduce the runoff of sediments, nutrients and pesticides into the GBR lagoon. End-of-catchment pollutant load targets have been set for a selection of priority GBR catchments to determine the effectiveness of catchment management actions over time. However, the ability to detect changes in water quality at the catchment's end and to assess this against the set targets over short time frames (i.e. a few years) is limited (Bainbridge *et al.* 2009a; Brodie *et al.* 2009a). This is particularly so for a large dry tropical catchment such as the Burdekin River, which has high inter and intra annual flow variability, and where considerable time lags exist before water quality improvement may occur at the end of catchment.

Research on load estimates, through better understanding of the variability inherent in load calculations from these large dry catchments, accounting for large flow events will allow a more comparable estimate on load reductions and potentially reduce the amount of time that would have been required for the detection of change.

MTSRF funded research has identified sediment sources throughout the Burdekin catchment, highlighting the areas where sediment erosion was highest. Research on the Burdekin Falls Dam has led to improvements in model parameters as well as a clearer understanding of the processes that are occurring as water moves through the dam system and over the dam wall. This is very pertinent as the model adjustments are being applied to other systems where the dam is influenced by short, episodic, high rainfall events. It also leads to a more common understanding of the potential impacts of the building of a dam in a North Queensland catchment. Catchment based research has been useful in identifying the sources and types of sediment as they move off the sub-catchments, but is also now beginning to identify the signals associated with finer sediment, allowing the tracing of the finer, mobile sediment which can impact substantially on inshore systems. Ongoing work on the estimates of loads, accounting for the very specific states of a North Queensland flood event gives a far greater confidence when estimating long-term flow rates. Recommendations for appropriate and frequent monitoring also leads to a greater confidence in our load estimation. Load measurements are the key factor in the Paddock to Reef Program and, as such, we must be confident in our ability to estimate loads accurately and be able to identify a change in loads within two to five years. Again, this work on load estimates and statistical rigor can and has been applied in other large dry catchments including the Fitzroy.

5. Case Study 2: Tully catchment

5.1 Introduction

The Tully-Murray basin in Far North Queensland is one of 35 basins discharging into the GBR (Figure 3.1). It is a small, faster flowing tropical river that extends approximately 130 km before discharging into Rockingham Bay. The Tully catchment itself is located in the southern part of the Wet Tropics region in Queensland, covering an area of 2,790 km² when combined with the Murray catchment (Furnas, 2003). Topography of the catchment varies from steep mountainous areas in the west to low relief floodplain in the east (Wallace *et al.* 2009a, 2009b; Karim *et al.* 2008a, 2008b). Flow discharge within each year is highly variable, peaking between January and April. As the topography is flat and the downstream areas of the Tully and Murray rivers are located close to each other, floodwaters have a tendency to merge during flood events causing the export of phosphorus and nitrogen to be much higher when compared to the annual average riverine load (Wallace *et al.* 2009a, 2009b).

Under the Reef Plan, the Tully basin was identified as a high risk catchment (Queensland Department of the Premier and Cabinet, 2003), and has been the focus of extensive collaborative research programs including the MTSRF and CSIRO Water for a Healthy Country Program in the GBR. Kroon (2009) synthesised the outcomes of a series of papers from this work published as a special issue in the peer-reviewed journal *Marine and Freshwater Research*, and a full synthesis was prepared by Henderson and Kroon (2009). These syntheses use the outcomes of the different strands of research and monitoring to encapsulate an integrated approach addressing the deteriorating water quality from land-based runoff in the Tully basin.

High levels of rainfall, combined with near-coastal steep topography and extensive fertilised land use on the floodplain provide the potential for erosion and pollutant transport to receiving waters (Kroon, 2008). Moreover, increased runoff rates and amount, due to removal of wetlands and floodplain vegetation and the installation of land drainage systems in coastal floodplains, mean that higher sediment and nutrient loads may now reach receiving waters (Kroon, 2008; Kroon *et al.* 2010). These loads are likely to have an impact on protected and important habitats in estuarine and marine receiving waters due to their close proximity to the coast. The coastal and inshore areas adjacent to the Tully catchment are regularly exposed to flood waters from the Tully River, and to a lesser extent from the Herbert River via the Hinchinbrook Channel. The Tully catchment area encompasses a range of different land uses (Table 5.1) which may affect water quality, as well as a number of point sources from industrial activities. These include sugarcane cultivation (15.8%), beef grazing (Wet Tropics type, 7.2%), horticulture (principally bananas with some papaya, 3.1%), urban (Tully, Mission Beach, Cardwell, 1%), forestry (pine, hardwoods, 1.7%), natural/forest (including rainforest, wet sclerophyll forest, defence lands, 69.8%), aquaculture (prawn, barramundi ponds, barramundi cage culture), sugar mills, sewage treatment plants and quarries. Each of these different land uses has different water quality concerns, so that key pollutants from the Tully catchment will vary between land uses.

The Tully River is among Australia's least variable rivers, representing the generally wet tropical climate of the region. It floods regularly, three or four times per year (Wallace *et al.* 2009a, 2009b) with riverine discharge extending into adjacent marine waters. The marine environment adjacent to the Tully catchment contains several continental islands with well-developed fringing reefs, which are of public and economic importance for the tourism industry and recreational activities such as camping and fishing. The coastal and inshore zone also supports intertidal and subtidal seagrass beds. The area encompasses several inshore Marine Park reserves ('no-take' zones, which allow non-extractive recreational use)

and a large Conservation Park zone (very limited extraction of marine resources permitted) around the greater Dunk Island area.

In the last two to three decades, the Tully-Murray coastal floodplain has undergone extensive modification largely due to the expansion of the sugarcane and horticulture (banana and pawpaw) industries (Mitchell *et al.* 2006). Tourism forms another key industry within the region; the township of Mission Beach is the access point to nearby islands and their associated fringing reefs (e.g. Dunk and Bedarra Islands). These reefs, as well as other inshore coral reefs of the Wet Tropics region, are degraded due to poor water quality (and other stressors), which has been linked to the runoff of terrestrial materials from agricultural lands (Fabricius *et al.* 2005; DeVantier *et al.* 2006; Brodie *et al.* 2007b; Wolanski *et al.* 2008).

Table 5.1: Percentage land uses for the major sub-catchments of the Tully and Murray catchments (see Brodie *et al.* 2009a).

Catchment	Land Use Type (%)								Water (%)
	Natural/Forest	Forestry	Grazing	Sugar Cropping	Bananas Horticulture	Other Horticulture	Animal Prod.	Urban Human	
Upper, Dam	84.36	10.11	0.04					0.01	5.46
Jarra Creek	89.85		3.89	1.75	4.29			0.10	0.00
Echo Creek	74.63		11.08	11.05	2.96			0.17	0.00
Davidson Creek									
Banyan Creek	57.99		2.59	33.18	1.10	0.02		5.03	0.04
Main-channel	66.55	0.01	8.33	17.99	4.45	0.02	0.64	0.50	0.00
All Tully River	73.25	1.87	6.09	12.97	3.08	0.01	0.28	0.74	1.01
Hull River	76.18	0.03	2.37	12.81	3.48	0.04		3.07	0.00
Murray River	57.04	1.80	12.17	25.36	2.38	0.77		0.34	0.00
Aggregate of Tully, Murray and Hull Rivers	69.76	1.73	7.23	15.78	2.94	0.18	0.20	0.80	0.71

5.2 Pollutant sources and loads

5.2.1 Land use contributions

Dissolved inorganic nitrogen runoff associated with nitrogen fertiliser loss has been identified as a major water quality issue in the Tully-Murray basin (Mitchell *et al.* 2001, 2006, 2009).

The dissolved inorganic nitrogen (nitrate) is supplied by the high proportion of fertilised agriculture activity in the catchment. For the Tully River catchment, Mitchell and others (2006, 2009) demonstrated a relatively linear relationship between the proportion of fertiliser-additive land use above a sampled site and the average stream nitrate concentration at that site (Figure 5.1). The increase in nitrate draining from 100% pristine land use to 100% fertiliser-additive land use was estimated to be 35 times in the Tully. When this multiplier and the large areas of sugarcane and horticulture in the Tully are taken into account with the bioavailable impact of this nutrient form, nitrate is therefore considered to be the key nutrient pollutant in the Tully Water Quality Improvement Plan (WQIP) region (Brodie *et al.* 2009b; Brodie and Waterhouse, 2009; Bainbridge *et al.* 2009b).

Other nitrogen forms are considered less important in runoff. Ammonia (NH₃), the other dissolved inorganic nitrogen form, is also bioavailable and is usually present in lower concentrations, but not always. Typically, it seems that ammonia is either preferentially taken up by riparian vegetation or converted to nitrate further downstream from the farms.

It has previously been considered that DON is not directly bioavailable and is turned over slowly in the marine environment, compared to dissolved inorganic forms. However, recent work cited by Wallace *et al.* (2007, 2009a, 2009b) has suggested that macrophytes, bacteria and phytoplankton in rivers and estuaries could be capable of directly assimilating DON at rates as high as 20-30% of those at which they assimilate DIN. Wallace argues that since DON is present in floodwaters at about four to five times the concentration of DIN, its biologically available load on freshwater, estuarine and marine biological systems may be as large as for DIN. Nevertheless, the biological impact of runoff derived DON in the downstream tropical seas is still somewhat uncertain. DON always forms by far the largest pool of nitrogen present in the marine waters of the GBR (two orders of magnitude higher concentrations than DIN) (Furnas, 2003), though the immediate origin of most DON (and other forms of nitrogen) is via recycling and transformation in the plankton communities of the shelf (Furnas *et al.* 2005). Particulate nitrogen (PN) is also considerably elevated, draining directly from cropping lands as shown at the Tully sugar and banana farms (Brodie *et al.* 2009a) compared to pristine areas.

Of the phosphorus (P) nutrients, the dissolved inorganic, bioavailable form is potentially the most polluting, though the concentrations of this form are usually low. Although some phosphate fertiliser is used on sugarcane, high phosphate concentrations are not found in most streams of the northern tropics (Brodie *et al.* 2009a). However, much higher levels are seen in the runoff from banana farms, where considerable phosphate fertiliser is used. Increases in the availability of dissolved inorganic phosphorus could have a significant impact on the nutrient ratio in the marine environment where phosphorus is the limiting nutrient. Non limiting Redfield ratios could potentially drive further increases in the primary production and phytoplankton community.

In a collaborative project between Terrain NRM, the Australian Centre for Tropical Freshwater Research (ACTFR) and the MTSRF, Bainbridge and others (2009b) identified the key pollutant sources and characterised the water quality signals of the different land uses within the Tully-Murray basin. Catchment disturbances have resulted in only minor increases in sediment loss in the Tully-Murray basin, with median SS concentrations only slightly elevated for the urban and sugarcane land use categories compared with forest land use

(Figure 5.1). The introduction of green cane trash blanketing and minimum tillage practices over the past two decades by the sugarcane industry has resulted in considerable reductions in soil erosion from this land use component (Rayment, 2003). In contrast, suspended sediment (SS) concentrations are much higher in the Burdekin River catchment (Bainbridge *et al.* 2007) which is dominated by rangeland beef grazing in a dry tropical rainfall regime, where ground cover is considerably lower, leaving soils more exposed and erosion prone. Runoff of SS is seen as a relatively low concern in the Tully-Murray basin. However, a component of the SS load discharged from the Tully and Murray rivers is transported to the adjacent inshore fringing reefs (Devlin and Brodie, 2005) and may therefore influence turbidity on these reefs (Wolanski *et al.* 2008).

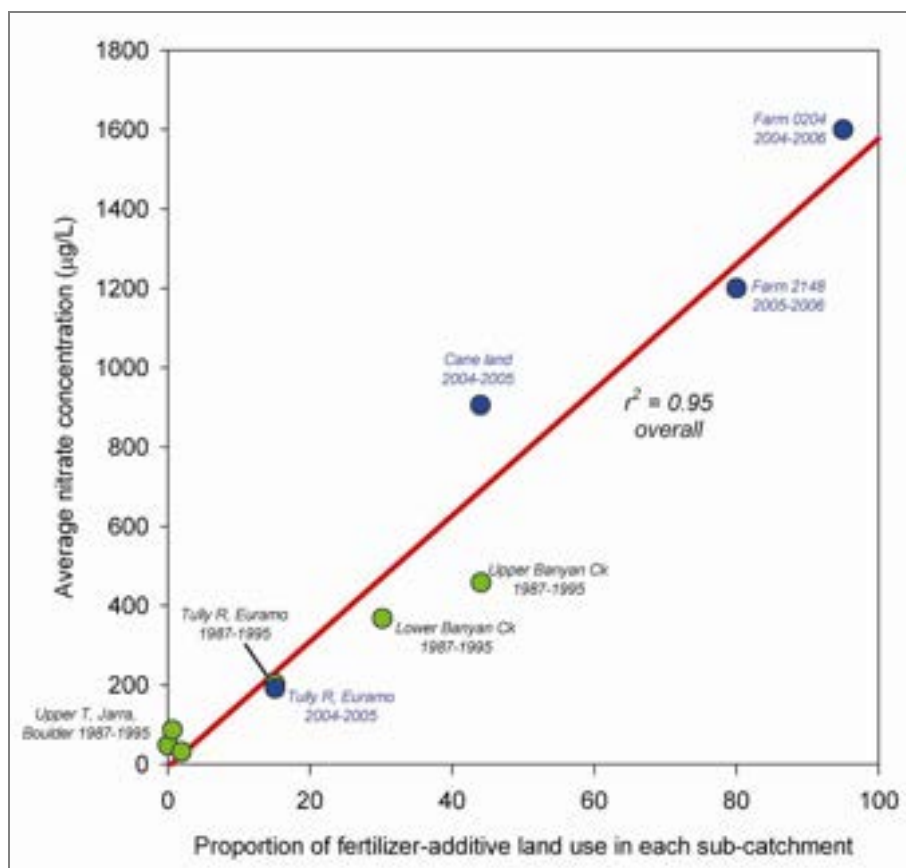


Figure 5.1: Export from the Tully WQIP region is dominated by very high levels of dissolved inorganic nitrogen (DIN), mostly nitrate, derived from fertilised cropping while the suspended sediments are at relatively low levels compared to other GBR catchment streams, especially those draining the Dry Tropics region. Source: Mitchell *et al.* (2006).

The runoff of $\text{NO}_x\text{-N}$ was particularly elevated in the sugarcane land use category, which indicates the runoff of fertilisers applied by this industry (Figure 5.2). In both the sugarcane and banana land uses, $\text{NO}_x\text{-N}$ contributed a high proportion (~70%) of TN compared with other land uses (Figure 5.2, Armour *et al.* 2009). The elevated concentrations of PN and DON in the sugarcane, grazing and urban land uses (Bainbridge *et al.* 2009b) likely reflect the location of these land uses on rich lowland soil types that also have a long history of nutrient enrichment through fertiliser use and mill mud application.

The concentrations of all forms of phosphorus species were low across all land uses in the Tully-Murray basin and reflect the low soil erosion in the region, as phosphorus may be strongly bound to the soils (i.e. in the particulate form).

Herbicides are also important pollutants in the area and the herbicides diuron, atrazine (including its degradation product desethyl atrazine) and hexazinone were frequently detected in waterways draining the sugarcane land use (2006-2007). Similarly to the relationship between nitrate concentrations and fertilised land (Figure 5.1) a linear relationship exists ($r^2 = 0.71$) between mean diuron concentrations and the proportion of sugarcane land use in the upstream catchment area (Figure 5.3). This implies a fairly uniform application rate of these herbicides across the sugarcane industry within the Tully-Murray basin (Bainbridge *et al.* 2009b).

Figure 5.2: Average proportion of nitrogen and phosphorus fractions for each land use category. Fractions include particulate nitrogen (PN), particulate phosphorus (PP), dissolved organic nitrogen (DON), dissolved organic phosphorus (DOP), oxidised nitrogen ($\text{NO}_x\text{-N}$), ammonia and filterable reactive phosphorus (FRP). In this instance, 'n' indicates the number of sites representing each land use category. Source: Bainbridge *et al.* (2009b).

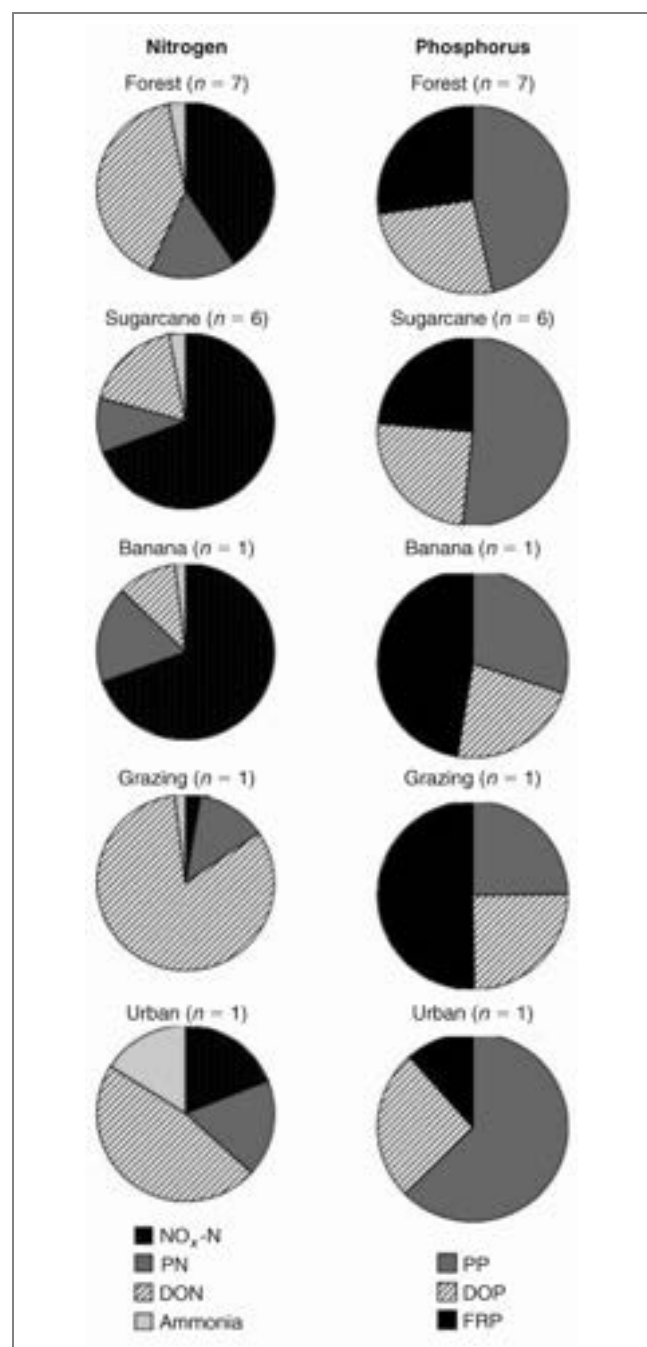
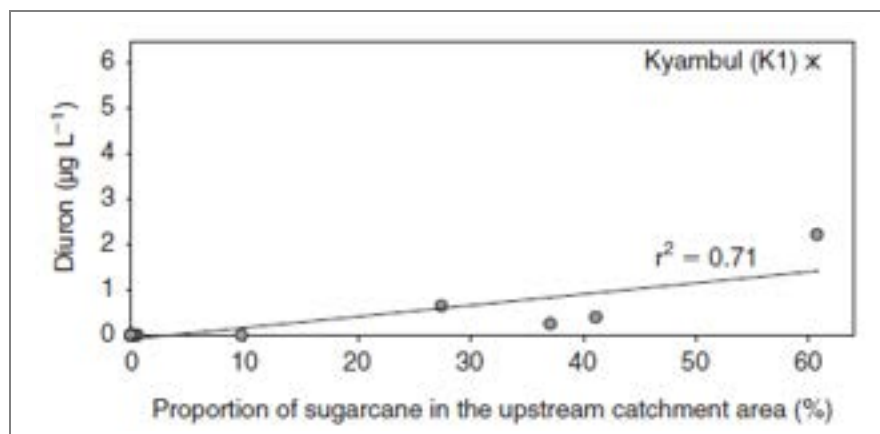


Figure 5.3: Relationship between the mean diuron concentration ($\mu\text{g/L}$) for each sub-catchment site (2006-2007) and the area of sugarcane land use in the upstream catchment. Source: Bainbridge *et al.* (2009b).



5.2.2 Current load estimates for the Tully catchment

The Tully catchment forms part of the Wet Tropics region, which contains eight rivers, located from south of Townsville (Herbert River) to north of Port Douglas (Daintree and Mossman Rivers), all of which discharge into the GBR lagoon. All of these rivers tend to flood on an annual basis over a number of high flow events. The Tully River is seen as a priority river for management of pollutant runoff, and as such appropriate and accurate load estimates are required for management. Improvements in the measurement of load estimates in the Tully catchment derived from MTSRF funded research are summarised below:

- A combination of monitoring activities and modelling tools link on-ground management at the paddock or plot scale to end-of-catchment resource condition loads, and finally to marine trigger values required for ecosystem protection. This combined approach overcomes many of the uncertainties associated with monitoring at shorter time scales as well as the shortfalls of the models available through data input and calibration (Bainbridge *et al.* 2009a; Brodie *et al.* 2009a).
- Load estimates for the Tully River have now been improved through the overbank flow work by Wallace and others (e.g. Wallace *et al.* 2010a, 2010b, 2010c). Flood contributions were found to increase the mean annual loads of phosphorus and nitrogen by thirty to fifty percent above previous river based estimates. Overbank floods can make a large contribution to the marine load of sediment and nutrients, despite the relatively low concentrations of these materials in flood waters. Overbank load is not well recorded by standard river gauges. This is described further in Section 5.2.3.
- Estimates for the proportion of DIN delivered by sugarcane have been modified by Brodie and others to take into account more recent studies for the region including those by Armour *et al.* (2007) and Hunter and Walton (2008). Using these estimates, and based on knowledge of land use areas in the Wet Tropics Region (and relevant to the Tully), it is estimated that the sources of DIN in the region are approximately 75% sugar and 5% bananas, 12% grazing and forest, and 8% other crops/dairy and urban.
- Using this information, estimates of the current and anthropogenic loads for the Tully catchment have been developed (Brodie *et al.* 2009b; Brodie and Waterhouse, 2009). The majority of DIN loads delivered to the GBR from the Tully catchment are derived from sugarcane (Figure 5.4). Figure 5.5(a-c) illustrates the relative contributions for TSS, DIN and PS-II herbicides for each of the Wet Tropics catchments. The Herbert catchment has the highest loads for TSS, with the Tully measuring the fifth highest load for TSS. However, the Tully catchment has the third highest proportion and loads of anthropogenic DIN. This elevated DIN load is a consequence of the dominance of sugarcane in Wet

Tropics catchments. PS-II herbicide loads also correlate with the high DIN loads in a number of catchments, with the Tully catchment showing the fourth highest with an annual load of over 1,000 kg.

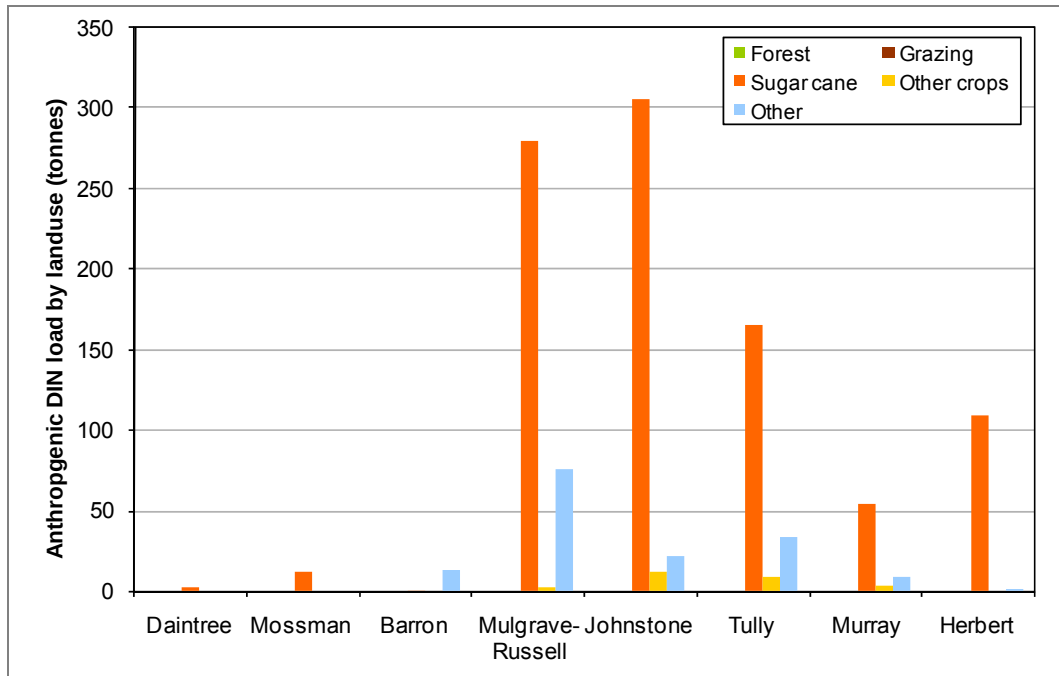


Figure 5.4: Estimated anthropogenic DIN load by land use for the Wet Tropics catchments. Source: Brodie *et al.* (2009b).

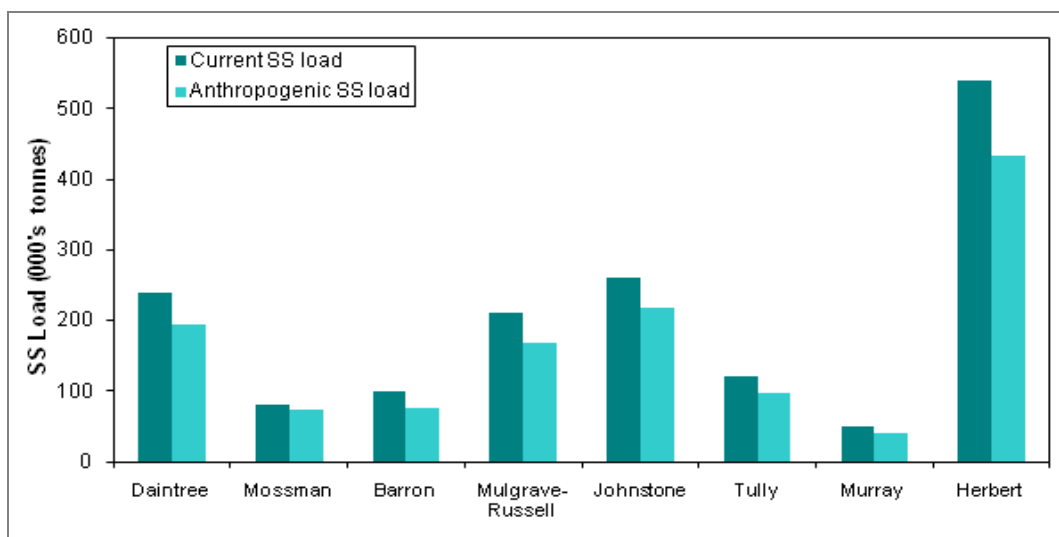


Figure 5.5(a): Load estimates for the Wet Tropics catchments with the relative contribution of TSS for each catchment. Source: Brodie *et al.* (2009b).

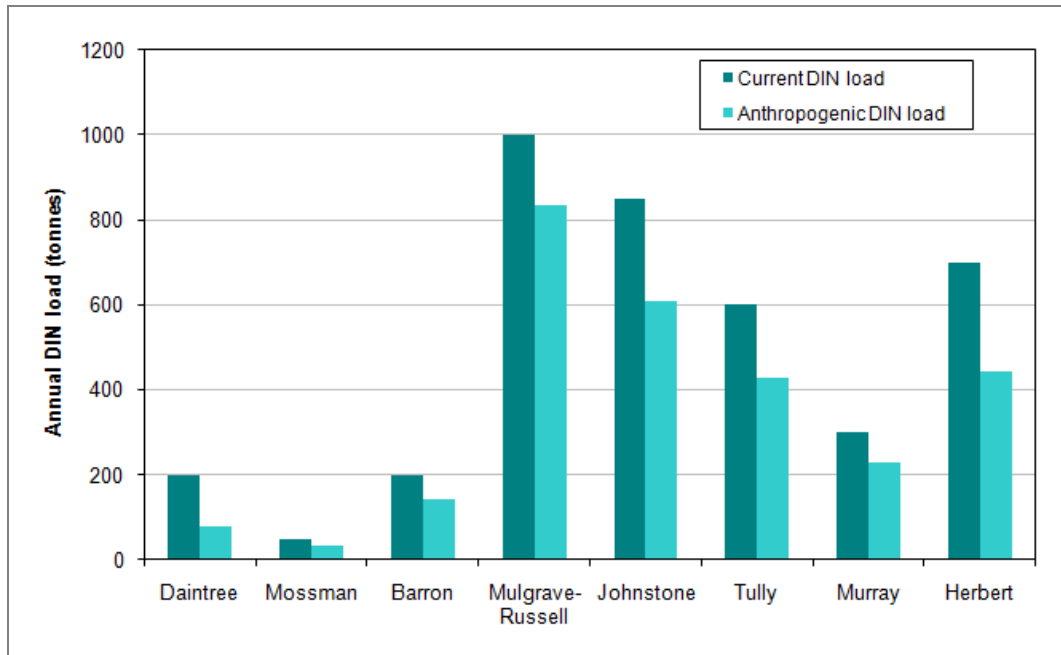


Figure 5.5(b): Load estimates for the Wet Tropics catchments with relative contribution of DIN for each catchment. Source: Brodie *et al.* (2009b).

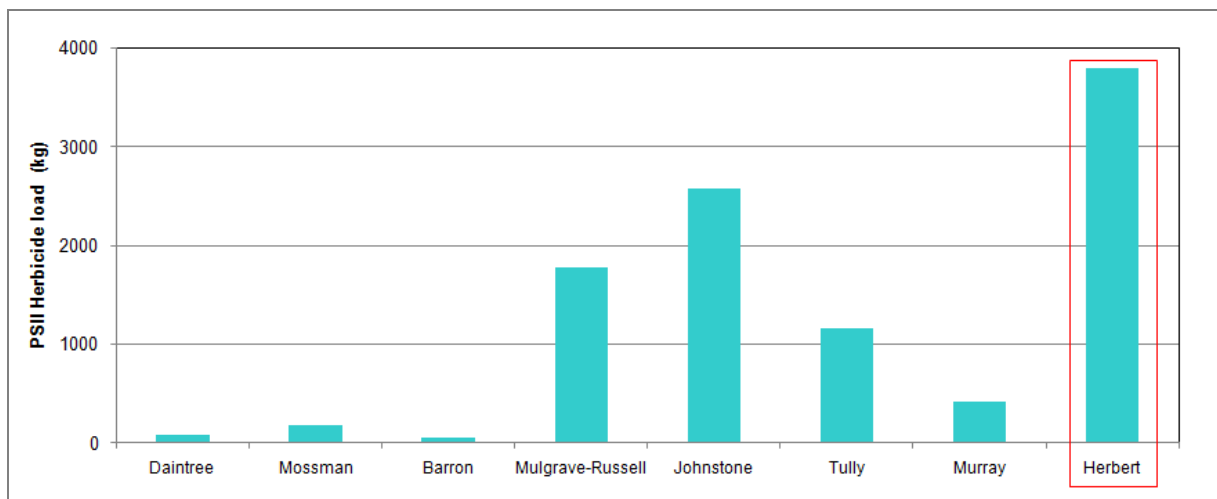


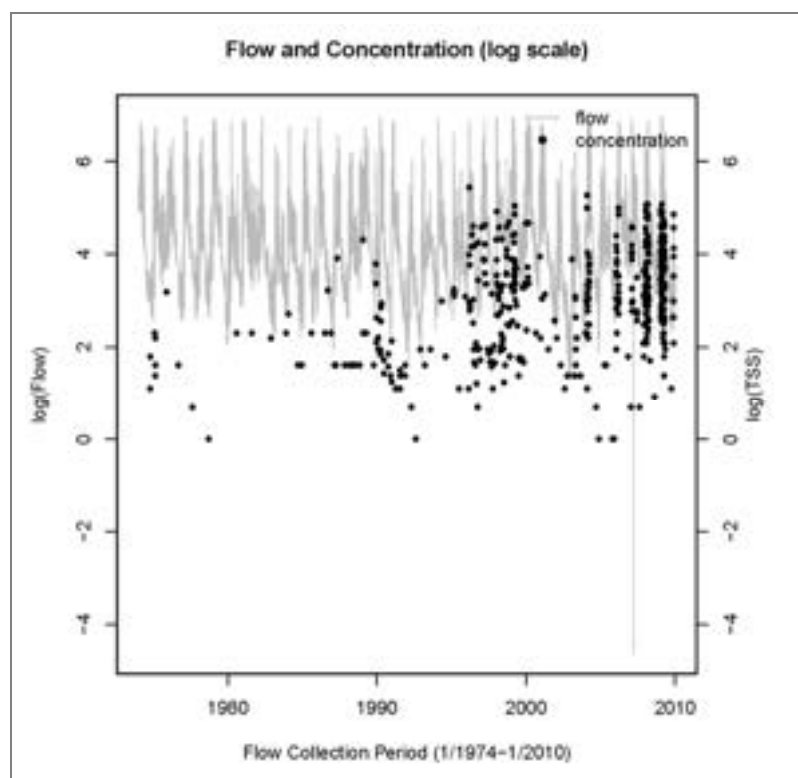
Figure 5.5(c): Load estimates for the Wet Tropics catchments with relative contribution of PS-II herbicides for each catchment. Source: Brodie *et al.* (2009b).

5.2.3 Improvements in load estimation techniques

As discussed in Section 4.2.2, Kuhnert and others have developed improved methods for calculating pollutant loads in the GBR, using the Tully catchment as one of the case study areas, representative of a wet catchment. The Loads Regression Estimator (LRE) package in the *R* programming language (Kuhnert *et al.* 2008, 2009; Kuhnert and Henderson, 2010) has been developed to estimate suspended sediment loads from the Tully catchment.

Flow records from the Tully River at Euramo were provided by DERM, collected between January 1974 and January 2009, spanning approximately eight years and consisting of 51,866 observations. Flow data were collected at irregular time intervals ranging from 0 to 43.91 days with a mean of 1.015 hours and a median of 2.24 days. For some periods, flow measurements were taken at intervals of a few days, but for much of the year the flow measurements are only approximately monthly. The total suspended solid concentration data were collected at sporadic time intervals, usually corresponding to an event. Figure 5.6 shows a plot of flow and concentration records for the 35 years of data collected in the Tully River. Note the increase in concentrations recorded during the later years.

Figure 5.6: Flow (solid line) and TSS concentration (points) shown on the log scale and sampled from the Tully River at Euramo between January 1974 and January 2010. Note, the large negative flow value corresponds to a zero observed flow. Source: Kuhnert and Henderson (2010).



The LRE package was used to create a modelling dataset consisting of 489 observations. Linear and GAM models have been considered which incorporate a quadratic term for flow, a rising/falling limb, a seasonal term and a discounted flow term. The strongest model included all of these terms with the exception of the term incorporating the rising/falling limb. The final model explained 74.2% of the variation in the data, which was much higher than a model which only incorporates flow (49.9% explained). The models show that as flow increases, TSS increases. Furthermore, as large events occur more frequently, TSS tends to decrease, indicating possible exhaustion of the system and dilution of material occurring. Estimates of TSS loads are shown in Figure 5.7. Annual loads calculated for each water year are shown with 80% confidence intervals alongside a bar plot showing the volume of flow recorded (Figure 5.7a). Average mean concentrations are shown in Figure 5.7(b) for each water year.

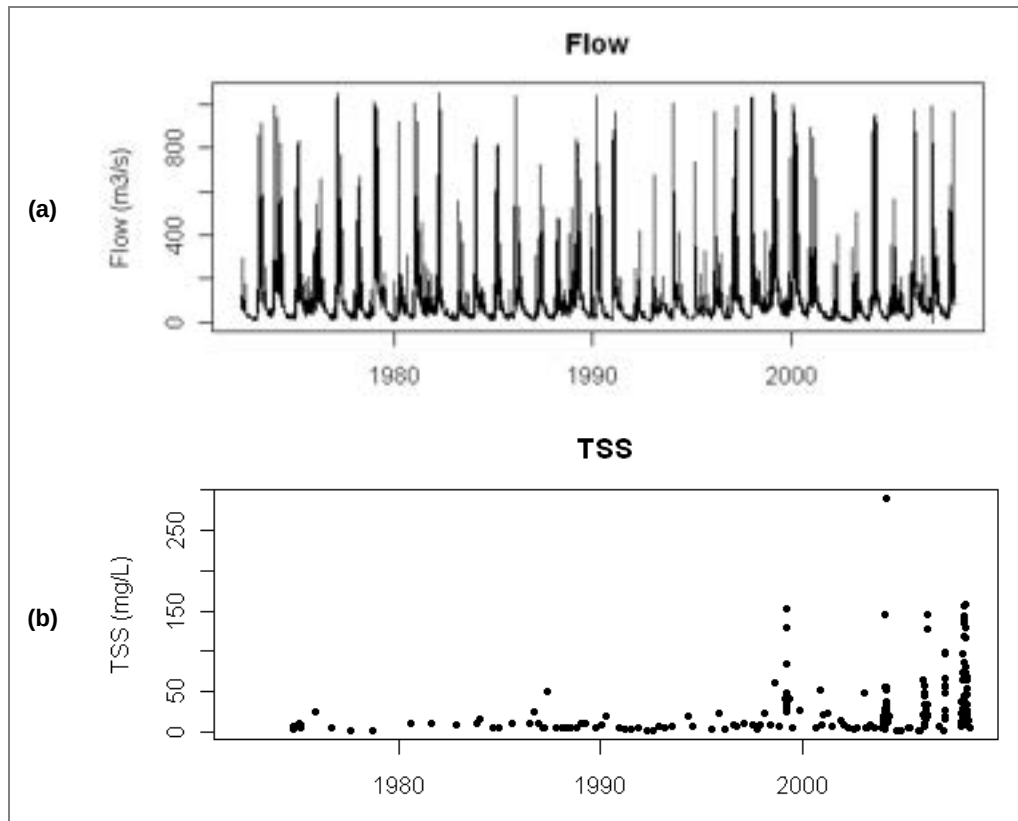


Figure 5.7: Flow (solid line) plot showing (a) flow, and (b) TSS concentration (points) captured at the log scale and sampled along the Tully River at Euramo between January 1974 and January 2010. Note, the large negative flow value corresponds to a zero observed flow. Source: Kuhnert and Henderson (2010).

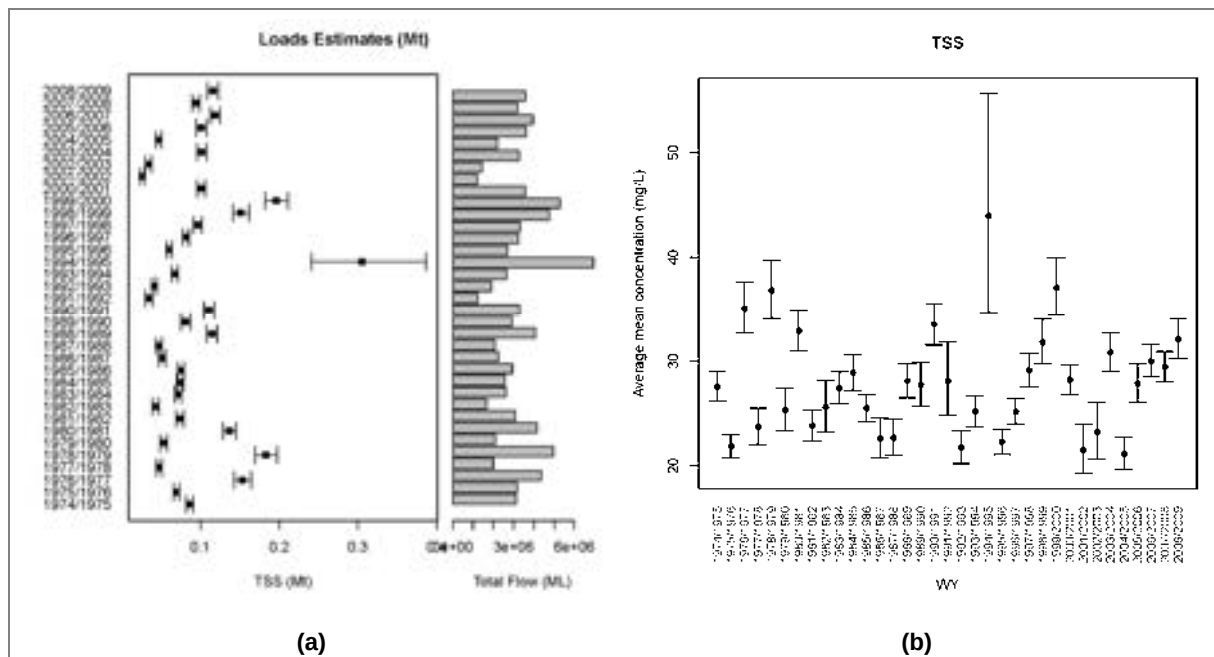


Figure 5.8: Plots for the Tully River sampling site showing (a) the estimated TSS load (Mt) and 80% confidence intervals for each year accompanied by the total volume of flow (ML) and (b) the average mean concentration (mg/L) for each water year. Source: Kuhnert and Henderson (2010).

5.2.4 Incorporating overbank flow in load estimations

In the past, Tully River loads have been estimated via monitoring at the gauging station in the river channel at Euramo (approximately 15 km upstream from the coast). However, in high-flow events, the Tully and Murray rivers break their banks and the floodwaters merge and flow to the ocean as a large sheet of water many kilometres wide. During these conditions the river gauges do not record the total catchment discharge very well. For example, Wallace *et al.* (2009b) showed that during the 13 flood events between 2006 and 2008, the Tully River gauge at Euramo recorded only 36-88% of the flood discharge and the Upper Murray gauge only recorded 11-27% of the flood discharge. Furthermore, current ocean sediment and nutrient loads are based on concentrations measured within the rivers, yet until this particular MTSRF project was initiated, the sediment and nutrient concentrations in overbank flood waters were not known.

Research by Wallace, Karim and others as part of the MTSRF water quality program has revealed that estimates of nutrient loads being delivered during flood events from the Tully-Murray catchments to the GBR lagoon have been significantly underestimated (Wallace *et al.* 2010a, 2009a, 2009b; Karim *et al.* 2010a, 2010b, 2010c). Flood contributions were found to increase the mean annual loads of phosphorus and nitrogen by 30-50% above previous river based estimates. Overbank floods can make a large contribution to the marine load of sediment and nutrients, despite the relatively low concentrations of these materials in flood waters. Current marine load estimates of material fluxes (based on gauged flows, measured river concentrations and modelling) from Australian wet tropical catchments with frequent flooding are probably too low, by quite significant amounts, depending on estimation method and constituent. For example, current estimates of annual average loads of phosphorus and nitrogen from the Tully and Murray catchments are 30-50% too low.

It was also found that sediment loads do not increase when flooding is taken into account and this may be because this material is source limited, whereas nutrient fluxes are transport limited. Annual marine loads will be very dependent on the number and size of overbank flood events in any year. This will make the monitoring of any trends in ocean loads difficult as the trend may be small in relation to natural inter-annual variability. Further analysis is required to quantify how large a change in load needs to be over a given period before it can be detected within the inter-annual variability. Despite the relatively high number of water quality samples in the Tully-Murray catchment, load estimates by Wallace *et al.* 2009a, 2009b still have a high uncertainty, e.g. up to $\pm 69\%$ for DIN. This means that monitoring of marine loads also requires a significant number of samples, preferably of both river and flood flows – otherwise there will be very large uncertainties in mean ocean loads, making it difficult (or even impossible) to detect any load reductions due to land use or management changes.

Given the above, in catchments that experience overbank flow in flood periods, monitoring of marine loads will require a significant number of samples of both river and flood flows (in time and space), otherwise the large uncertainties in mean loads may be misleading and it may be difficult to detect any load reduction trends (Wallace *et al.* 2010a). An assessment of which GBR rivers may contain significant overbank flood loads is now being made by Wallace and others (2010c).

Table 5.2: Long-term (1972-2008) annual average sediment and nutrient loads leaving the Tully and Murray catchments. Total loads are separated into those occurring while flow is in-bank and while flow is overbank (i.e. during flooding). For comparison, the annual average loads from all of the published studies in the Tully and Murray catchments are also shown. Source: Wallace *et al.* (2009b).

	All studies			Wallace <i>et al.</i> (2009b)		
	TN (tonnes)	TP (tonnes)	TSS (tonnes)	TN (tonnes)	TP (tonnes)	TSS (tonnes)
In-bank flow	1,129	114	76,097	1,129	114	76,097
Overbank flood	543	55	36,626	1,322	109	38,842
Total	1,672	169	112,723	2,451	223	114,939

Another important finding of the research was the composition of the flood waters compared to river waters, showing that concentrations of dissolved organic nitrogen are higher than dissolved inorganic nitrogen in flood waters, which is the opposite of river water (Wallace *et al.* 2010a, 2009a, 2009b). This has implications for load based estimations in this catchment and those with similar overbank conditions and the type of management practices that may be adopted to reduce nutrient delivery to the GBR. DON loads to the ocean may be around twice those previously estimated from riverine data. Reducing DON loads will require different interventions to those used in agriculture to reduce DIN, for example, measures that slow down and reduce drainage, such as the introduction and/or rehabilitation of riparian zones and wetlands.

The implications of the flood water quality studies in the Tully and Murray catchments, and potentially for other GBR catchments, are as follows:

1. Overbank floods can make a large contribution to the marine load of sediment and nutrients and much of this load may not be recorded by standard river gauges.
2. In GBR catchments where floods are a significant proportion of the annual flow, current marine load estimates of sediment and nutrients (based on gauged flows, measured river concentrations and modelling) are probably too low, by significant amounts, depending on estimation method and constituent.
3. The size of this underestimate in any year will depend on the number and size of overbank flood events in that year. This will make the monitoring of any underlying trends in ocean loads difficult unless it is possible to remove inter-annual variability.
4. Monitoring of marine loads will take a significant number of samples of both river and flood flows (in time and space) – otherwise the large uncertainties in mean loads may be misleading and it may be difficult to detect any load reduction trends.
5. The cause of the above underestimate in loads is mainly due to the poor recording of flood (overbank) discharges by river gauges, but also to differences in flood water and river water quality concentrations.
6. Flood waters can carry more dissolved organic nitrogen (DON) than dissolved inorganic nitrogen (DIN) and this is the opposite of their concentrations in river water. Consequently, DON loads to the ocean may be much higher than those previously estimated from riverine data.
7. Water Quality Improvement Plan (WQIP) actions that focus on farm interventions in agriculture will potentially reduce DIN loads.
8. Reductions in DON (and sediment) loads that arise outside the floodplain require different interventions to those used in agriculture to reduce DIN, e.g. measures that slow and reduce drainage and the introduction and/or rehabilitation of riparian zones and wetlands.

5.3 Pollutant transport and trapping

5.3.1 Wetland condition and function in the Tully catchment

The loss of wetlands and lagoon systems suggests that a major deterioration of ecosystem health has occurred in the Tully-Murray waterways. The floodplains and the lagoon ecosystems of the Tully-Murray floodplain have suffered from the increase in intensive agriculture (Pearson *et al.* 2010a) through clearing, modification and drainage. There has been severe habitat impact, at least peripherally, through loss of riparian vegetation and moderate weed infestation. Loss of riparian vegetation reduces perching and basking opportunities for amphibians, reptiles and birds, removes bird breeding sites, reduces habitat for terrestrial stages of aquatic insects, reduces shallow water habitat complexity through loss of snags, decreases input of natural detritus, and leads to higher light levels, promoting weed growth (see Pearson *et al.* 2010a).

Clearing within sub-catchments has promoted changes to natural drainage pathways and connectivity. Clearing changes hydrological signatures by allowing more rapid runoff, as does the introduction of artificial drains and straightening of natural channels. Removal of riparian shade allows weed infestation of drainage channels, reducing connectivity for fish.

Impaired water quality at a paddock, sub-catchment and catchment level has been driven by the excess of nutrients, organic material, sediments and pesticides draining from agricultural land. Nutrients can affect freshwater health in a number of ways, but especially through promotion of plant growth. This can create plankton blooms, leading to oxygen deficiencies and fish kills, and to toxic effects on fish and other animals. Nutrients encourage macrophytes to grow, and can be a particular problem where riparian shade is removed and exotic weeds establish, crowding out native plants and clogging waterways, reducing available habitat. Nutrients also promote microbial decay processes, again potentially increasing occurrence of oxygen deficit (hypoxia), leading to fish asphyxiation. Organic material derived from leached sugars and decaying trash on the land is broken down by microbes in waterways, frequently leading to hypoxia and fish kills. This has been a major problem in the Herbert and Burdekin floodplains, but may be less severe in the Tully-Murray because of the higher rainfall and regular flushing.

Suspended sediments reduce light penetration and normal photosynthesis, although paradoxically this could be beneficial where there are nutrient-induced plankton blooms. Excess and chronic turbidity may cause fish problems in feeding and respiration. Sediments typically have nutrients and organic material attached, so exacerbate problems linked with nutrient enrichment.

The most prevalent pesticides in use in the Tully-Murray region are various herbicides which may have limited direct effect on macroinvertebrates, fish and other animals (we do not know what their effects are) but are likely to have substantial indirect effects by modifying vegetation in the waterways. It is likely that some plant species are more tolerant of herbicides than others, so that there will be a differential response within the plant community (including exotic plants); this is likely to have an effect on habitats of animals. So far we are unable to identify any such effects.

5.3.2 Assessment of freshwater health in the Tully-Murray wetlands

This would appear to be the case for natural streams, which have lost habitat values. However, despite there being clear gradients in our ecosystem health indicators across the floodplain, the integrity of fish and macroinvertebrate communities appeared to be surprisingly good. If it is the case, in contrast with Herbert and Burdekin wetlands, it is probably because of the higher level of flushing in this very high rainfall area, protecting the lagoons from the worst of the potential impacts.

Several taxa/assemblages could be useful surrogates or proxies of ecosystem health in the lagoons – that is, suitable indicators (see Pearson *et al.* 2007, 2010a; Arthington and Pearson, 2007). Descriptions of potential ecosystem indicators for the Tully-Murray freshwater ecosystems developed through the MTSRF are summarised in Table 5.3 (page 79). It also appears that, despite lack of active management of the waterways and their surrounds for improved environmental outcomes, there is substantial resilience to impacts, largely because of the regular flows in the system.

5.3.3 Assessment of freshwater health in the Tully-Murray wetlands

Habitat quality and the ecological integrity of floodplain wetlands depends on many factors, but a key determinant is how the wetland is hydrologically connected to the main river channel over time (Junk *et al.* 1989; Paterson and Whitfield, 2000; Tockner *et al.* 2000; Bunn and Arthington, 2002; Frazier and Page, 2006). In a wet tropical region, permanent flows often provide continuous in-stream connectivity; however, off-stream wetlands may be isolated for significant periods when low flows are constrained to the main stream channels. Flood flows provide the opportunities for these off-stream wetlands to be connected with the main streams. During floods there is an exchange of water, sediments, chemicals and biota between the main channels and floodplain wetlands. The importance of overbank flow connection for the productivity and exchanges of major aquatic biota in river-floodplain systems has been emphasised in many studies (e.g. Junk *et al.* 1989; Heiler *et al.* 1995; Middleton, 2002). The single most important factor for the persistence of the fish assemblage in an isolated wetland is the flow connection between the wetland and a main stream (Arthington *et al.* 2006; Lasne *et al.* 2007).

Wetland condition also needs to be reported to demonstrate that there has not been further degradation of wetlands in the GBR catchments, and further work is required to fulfil this requirement of the Paddock to Reef Program. Significant progress in defining how wetland ecological condition is affected by flood regimes has been made in MTSRF funded research conducted by Pearson, Wallace and others in the Tully-Murray catchments (Pearson *et al.* 2010a, 2010b; Wallace *et al.* 2010b; Karim *et al.* 2010a, 2010b, 2010c; Godfrey, 2009).

MTSRF funded research has produced the first quantitative estimates of wetland connectivity during and after flooding using floodplain hydrodynamic models, which have shown that during flood events the duration of connectivity of individual wetlands varied (from one to twelve days) depending on flood magnitude and location in the floodplain, with some wetlands only connected during large floods (Karim *et al.* 2010a). Figure 5.9 shows that all of the wetlands studied were connected to the Tully River for shorter periods than they were to the Murray River, due to their proximity to the Murray River and the higher bank heights and levees on the Tully River. These variations in wetland connectivity could affect the movement of aquatic biota during flood events and the variability of habitat and biodiversity of individual wetlands. Flood pulses produce an initial connection between the wetlands, but after the inundation recedes, this is followed by a period of connectivity via the natural streams and the manmade drains on the floodplain. Figure 5.10 shows an example of additional hydrodynamic modelling of the post flood period (Karim *et al.* 2010b) when wetlands that are located near the rivers, and/or have good network connection, maintain longer connection

times with the rivers. Drainage network connectivity to both rivers varied from 30 to 365 days, and was much greater than flood inundation connectivity for the same wetlands (Figure 5.9). The connectivity of artificial wetlands varied greatly, from ten to one hundred percent of the year, according to the type of network connection they have; a result that has important implications for the location of these types of wetland.

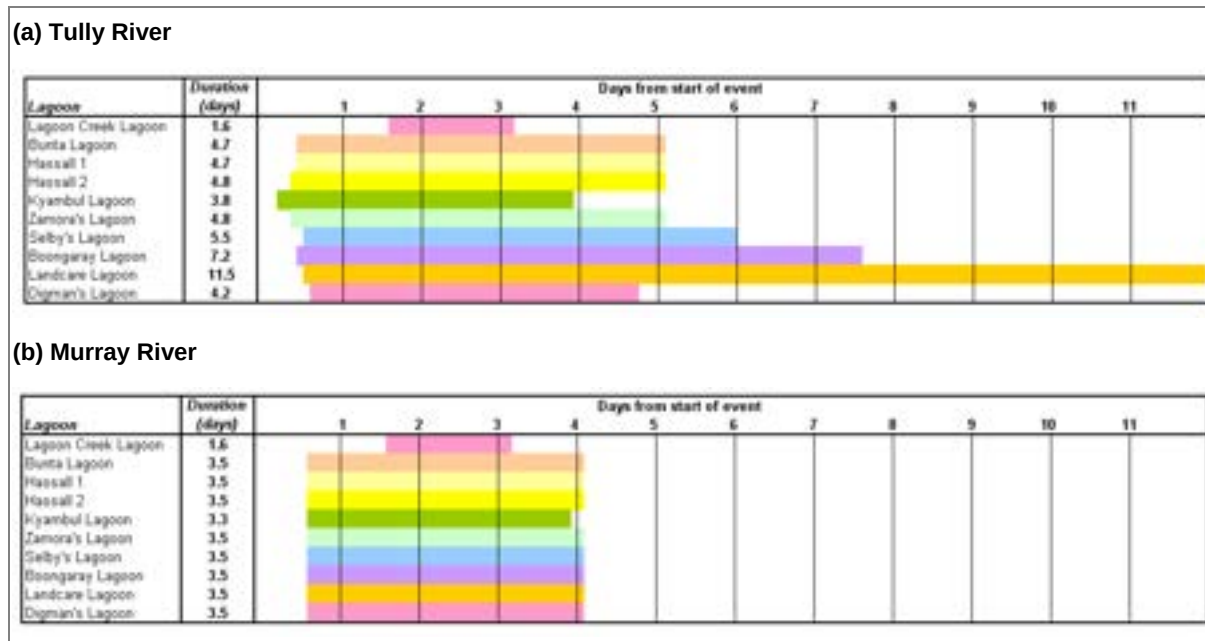


Figure 5.9: Summary of the timing and duration of connectivity of ten floodplain wetlands to the (a) Tully and (b) Murray Rivers during floods with an annual return period of twenty years. Source: CSIRO.



Figure 5.10: Timing and duration of connectivity of individual wetlands via the stream and drainage network to the (a) Tully and (b) Murray Rivers during 2008 for the threshold of water depth of 10 cm. Source: CSIRO.

Table 5.3: Description of potential ecosystem indicators tested in the Tully-Murray catchment/ freshwater systems. Scoring is based on the applicability of the indicator with scoring ranked from 1 to 5, with 5 being the highest. Derived from Pearson *et al.* (2010a).

Biological element	Characteristics	Indicator efficacy	Indicator score for Tully catchment
Macrophytes	• Provide habitat and refuge from predators • Driver of macroinvertebrate community response	• Useful as a determinate in the ratio of introduced to native species • Description and sampling is arbitrary when consider overall ecosystem health (Mackay <i>et al.</i> 2010) • Presence and absence can explain differences in macro invertebrate communities	4
Riparian condition	• Absence may be stimulus for weed growth (para grass) • Lack of shade/high nutrients affect habitat and influence fauna	• Presence or absence correlated with macroinvertebrates	3
Water quality	• Strong influence of salinity, pH, Dissolved Oxygen and pesticides. Difficult to identify precise cause and effect	• Water quality parameters may link to distance from coast/agricultural source. • Can also provide snapshot assessment of current status and source (high sediments, nutrients or pesticides) information	3
Zooplankton	• Easy to sample and count, but does require skill to identify beyond higher taxonomic categories	• Promising indicator. Abundance showed correlation with ecosystem gradients, even at coarse taxonomic scale	2
Macro-invertebrates	• Very efficient indicator, representing very diverse taxa with a diversity of responses to stressors. Easy to sample and sedentary thus reflect local conditions. Integrate conditions over time. Have short life cycles so show rapid population response to changed conditions.	• Very good discriminators of subtle changes in condition in Wet Tropics streams (Connolly <i>et al.</i> 2007a). • If dominant effect of habitat is removed, macroinvertebrates show distinct gradients of response to conditions in the wetlands and therefore have the potential to be useful health indicators.	4
Fish	• Longer lived and more mobile so less useful at site level. Do respond to changing conditions across sites and thus good indicators at catchment. • More sampling effort but less processing time (easier to identify)	• Represent condition gradients in Wet Tropics streams • Fish assemblages provide good indications of environmental gradients, including general habitat quality and proximity to rivers and river mouths	4
Birds	• They are diverse, with diverse relationship with wetlands, and are easy to identify and count remotely	• can provide a very good indication of wetland ecosystem health • Low population numbers in Tully-Murray system. Absence may reflect poor riparian habitat (limited perching and nesting sites), avoidance of deep waters because of crocodile threats, or other habitat/water quality issues	1

This MTSRF funded project has also shown how connectivity modelling can be used to identify when water levels in a drainage network fall below critical thresholds for fish movement, using readily available river gauge data. These types of relationship are central to the concept of setting environmentally acceptable flows. Quantitative connectivity modelling will also be useful to help explain the variation in habitat structure, aquatic biota composition and water quality of individual wetlands over time.

5.3.4 Improvements in the hydrological model

Floodplain hydro-dynamic models have been used to quantify two important aspects of hydro-ecological functioning: (i) sources, sinks and transport of sediments and nutrients across floodplains (Wallace *et al.* 2009a, 2009b), and (ii) connectivity of wetland systems within floodplains (Wallace, 2101a; Karim *et al.* 2010a, 2010b, 2010c). The twin development of conceptual models of the ecological dynamics of these systems (Pearson *et al.* 2010a; Godfrey, 2009) and how these interact with the hydrological processes is strategically designed to improve our capability to predict the impacts of changes in land use, land and water management and climate on the flow and water quality regimes and ecological dynamics in the wetlands and floodplains of catchments adjacent to the GBR (Karim *et al.* 2009a).

In parallel to the floodplain hydro-dynamic models above, there has also been progress in a process based estuarine eco-hydrology model which has three interacting components: (i) hydrology, (ii) biota/biology and (iii) ecosystem health (Webster *et al.* 2008). The physical sub-model takes freshwater, sediment, nutrient and plankton from the river and adjacent wetlands feeding the estuary. These elements are then mixed and transformed within the tidal limit and then transported towards the ocean. The biology sub-model uses the sediment and nutrient concentrations determined in the physical model to affect the interactions between phytoplankton, zooplankton, bivalve biota and fish. Key components of this model are the carbon and detritus fluxes within the estuary, as these ultimately determine the biomass of the different trophic levels in the food web and hence the overall ecosystem health. Successful applications of this estuarine modelling system were given for the Guadiana estuary in Portugal as well as the GBR. This suggests that, in both temperate and tropical wetland fringed estuaries, the main drivers and key processes are correctly incorporated in the model. These were summarised as: (i) river inflow of nutrient, (ii) residence time (which is driven by both the river and the ocean), (iii) wetland sediment trapping, outwelling of particulate matter and juvenile refuge, (iv) ocean physical (tidal stirring) and biological (larval supply) influences and (v) the role of sediment in the estuarine food web. Further development of this model will assist in linking end-of-catchment loads to hydrological and ecological models to predict the response of improved management practices to GBR ecosystem health.

5.4 Assessment of risk from the Tully catchment

Analysis of data on fertiliser use, loss potential and transport has ranked fertilised agricultural areas of the coastal Wet Tropics and Mackay-Whitsunday as the hot-spot areas for nutrients (mainly nitrogen) that pose the greatest risk to GBR reefs (Brodie, 2007). In the Wet Tropics, sediment fluxes are comparatively lower due to high vegetation cover maintained throughout the year from high and year-round rainfall and different land management practices (Kroon, 2008) from Dry Tropics regions within industries such as beef grazing. Urban development sites can be local high impact sources of suspended sediment. Of the herbicide residues most commonly found in surface waters in the GBR region, diuron, atrazine, ametryn and hexazinone derive largely from areas of sugarcane cultivation, while tebuthiuron is derived from rangeland beef grazing areas (Lewis *et al.* 2009b).

As discussed in Section 4.3, Brodie and Waterhouse (2009) provided a relative risk assessment of pollutants for the regional NRM regions in the GBR catchments, incorporating the Wet Tropics region. This study is described in further detail in the companion report by Waterhouse and Brodie (2010), with results shown in Figure 4.5. From these results the highest management priorities are in the Wet Tropics region, incorporating the Tully catchment, and the Mackay-Whitsunday region.

In the assessment, coastal grazing in the wetter catchments is differentiated from other (extensive) dry tropics grazing by continuous high pasture cover and hence low erosion potential. The dairy referenced in this report is intensive and varies considerably from extensive grazing but is a relatively limited source of diffuse runoff (by both area and loss per hectare) in the GBR catchments, hence not a high priority in any of the regions. Wet Tropics sugarcane and bananas come out as the highest priority in the risk ranking, with coastal grazing identified in the risk ranking but as one of the third priorities.

Within these rankings, subdivisions can be made based on (1) the likely speed at which improvements can be achieved from applied management in the different land uses, and (2) the quantity of the contaminant load. In general, improvements in PS-II herbicide loads will be the quickest to eventuate, followed by improvements in DIN due to fertiliser management, with considerable times to reduce suspended sediment due to erosion management. Wet Tropics bananas are significant despite being only 5% of total load; however it is Wet Tropics sugarcane (75% of total load) which is a more important priority for management. Coastal grazing and intensive dairy generally contribute small contaminant loads, partly due to relatively low land use area. Thus management for the Tully catchment could begin with sugarcane herbicide and fertiliser management. Reduction of loadings through application of best management practices, e.g. The Six Easy Steps Approach, will have effect in the shortest timeframes. This is also the case for horticultural industries such as bananas, where proven practices to manage contaminants are available.

5.5 Socio-economic influences in the Tully catchment

Socio-economic research into the cost effectiveness of recommended BMPs for the major industries (sugarcane, horticulture, grazing and forestry) within the GBR catchment area has been conducted as a collaboration between the CSIRO and MTSRF across the entire region (Roebeling, 2006; Roebeling *et al.* 2005; 2007a), and specifically for the Tully-Murray region (Roebeling *et al.* 2004, 2007b; Bohnet *et al.* 2006, 2007; Roebeling and van Grieken, 2007; Roebeling and Webster, 2007; van Grieken *et al.* 2009, 2010a, 2010b, 2010c). This approach also incorporates production system simulation models (such as APSIM) and the SedNet and ANNEX models (Roebeling *et al.* 2007a). For sugarcane, the economic analysis approach is the same as implemented in the Burdekin Dry Tropics. For each farming system (U, B, C and D; see van Grieken *et al.* 2010a) the long-term productivity costs and benefits (gross margins) and the investment costs to facilitate change to improved farming systems (van Grieken *et al.* 2010b; 2010c) are assessed (Table 5.4).

Table 5.4: Financial costs for each farming system in the Wet Tropics related to sugarcane (A = Aspirational / commercial viability not yet proven, B = Best practices, C = Common practice, D = Dated practice). Source: van Grieken *et al.* (2010b).

Farming system	Average gross margin (\$/ha/yr)	Investment costs (\$)
A	869	88,000
B	870	43,250
C	724	-
D	593	-

There are costs and benefits involved in moving to 'improved' farming systems (grouping of management practices) in the Wet Tropics as represented by the Tully region (Figure 5.11). In other words, the type of improved cost benefits associated with farming systems reducing the dissolved inorganic nitrogen (DIN) available, from runoff and deep drainage leaching, from entering the waterways from the paddock in the corresponding regions. D refers to 'dated' (or degrading) farming system, C to 'common' (or current), B to 'best' (or industry recommended) and A to 'Aspirational' or commercial viability not yet proven (or proof of concept, research and development).

This work is calculated per farming system (steady state D, C, B and A, not in transition) with the corresponding information given in DIN (kg/ha/yr) available to enter the (hydrological) system, farm gross margins (AU\$/ha/yr) and the investments (e.g. machinery) required to move from one farming system to the other. It must be noted that transaction costs (or hidden costs of change) are not incorporated hence total cost of change is likely to be underestimated.

In the Tully catchment, moving from a C farming system to a B farming system will require the investment of approximately AU\$43,000 (e.g. the purchase and modification of machinery) (Figure 5.11). It will potentially reduce DIN pollution from the paddock by more than 10% but with increasing (steady state) farm gross margins greater than 25%. A few examples of the changes that farmers face to move from a C to a B farming system are the use of GPS for planting, a reduction of tillage operations, fertiliser application rates based on soil tests, the use of legume crops in half of the fallow area, the development of a soil management plan, improved record keeping and the use of climate and weather forecasts.

In conclusion, for the Tully catchment, change in (dominant) agricultural practices can be summarised as below:

Sugarcane: Improved practices may lead to increased productivity benefits but show significant investment costs. It must be noted that costs and benefits associated with a transition will be different for each individual grower and therefore each circumstance needs to be carefully considered before making a change in management practice.

Bananas: Overall, there are expected to be benefits to growers in the Tully region through transitions towards water quality improvement management practices, although these will vary for each individual grower depending on their starting point and their individual property scenario. Further education regarding the expected benefits of transition to improved cane management practices may encourage some growers in the region to begin the transition. However, each circumstance needs to be carefully considered before making a change in management practice.

Grazing: In wet coastal grazing, matching stocking rates to pasture carrying capacity leads to a small reduction in sediment delivery as well as an increase in profitability. In general, gross margins are lower for pastures with a lower carrying capacity, i.e. in particular for pastures on soils where pasture growth is affected by waterlogging.

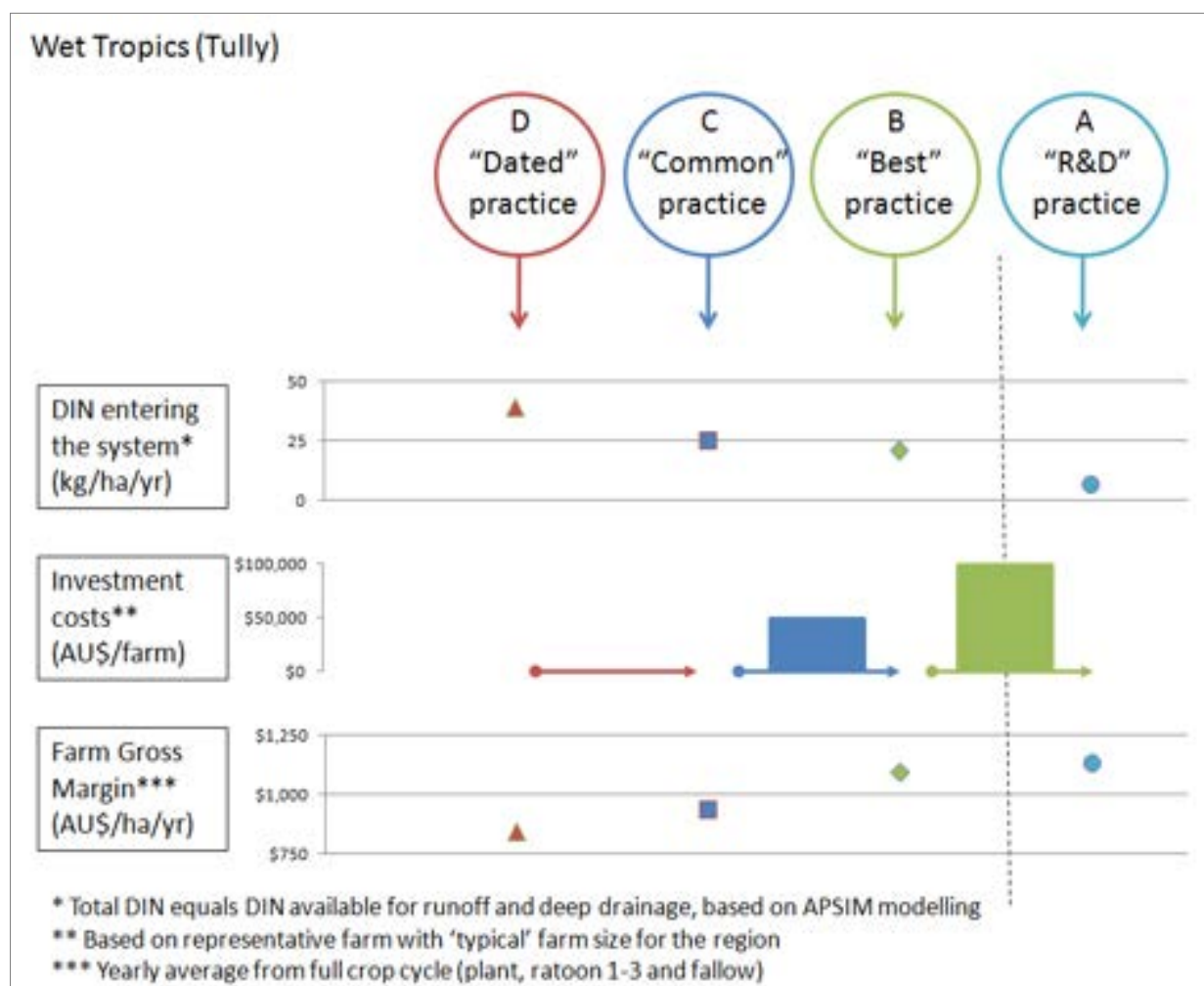


Figure 5.11: Investment costs of moving from one agricultural system to another for sugarcane activity in the Tully catchment. Source: M. van Grieken.

Another important aspect of understanding and managing water quality issues in the GBR is related to management approaches and understanding what influences water quality planning and governance arrangements. Robinson and others have considered the social and institutional aspects of water quality management in the Tully catchment as part of their MTSRF project, with considerable effort towards improving water quality planning for management (e.g. Robinson *et al.* 2009a, 2009b, 2010; Robinson and Taylor, 2008; Taylor and Robinson, 2010). Using the Tully-Murray WQIP as a case study, a 'SMART' partnership framework was developed that could be used to assess the capacity of governance arrangements to facilitate specific and spatially explicit actions to improve water quality. Measures of management and partnership performance were also negotiated to enable judgments of management effectiveness, efficiency and appropriateness to be shared. The SMART partnership framework was used to inform the design partner activities needed to deliver agreed water quality targets in Far North Queensland based on several criteria:

- i. **Specific delivery roles and responsibilities** are often difficult but critical to negotiate spatially and between sectors;
- ii. **Measures of partnership success** are often sector specific but essential to negotiate to ensure sustained delivery of water quality outcomes;
- iii. **Achievable partnerships** are those that embrace a wide range of actions that align with government, industry and community sector capacity and motivations to engage in NRM issues;
- iv. **Realistic partnerships** support actions that can already be delivered under existing institutional arrangements and relationships; and
- v. **Timely partnerships** reflect strategic and successional planning that ensures the most effective organisational links are implemented through the target setting, implementation and evaluation phases of water quality planning and management.

These criteria were used to analyse three key functional components of adaptive approaches to the (re)design of environmental governance arrangements: (i) partnership scoping and design, (ii) partnerships to explore and appraise delivery options, and (iii) evaluation and reporting of partnership effectiveness. During the first year of research the Tully catchment the focus was predominantly on the first of these applications. An overview of the outcomes is presented in Table 5.5. Using this information, the research team was then able to identify the actions required to deliver the management targets with a focus on stakeholder negotiations.

Following further investigation of the frameworks for monitoring and evaluation of management partnerships across the GBR (see for example Robinson and Taylor, 2008), a framework was established to evaluate collaborative governance performance (Robinson *et al.* 2009, 2010; Taylor and Robinson, 2010). Table 5.6 presents the framework developed that uses knowledge indicators to evaluate collaborative governance performance tested in the Wet Tropics region for Reef Rescue grants. Results from testing this framework are presented in Robinson *et al.* (2009b). The approach enabled new insights, gained through application of the framework, to be incorporated into existing collaborative structures and decision-making processes. In addition, the timely feedback process helped to strengthen partnerships through providing a collectively developed agenda to guide deliberation and apply improvements to local arrangements. Specific improvements implemented by the regional body were focused around: regular communications with partners; enhancing partner engagement in planning and priority setting; and further developing tools, processes, and procedures to support efficient and effective program delivery. This approach could be applied across all regional NRM groups in the GBR catchment, or elsewhere in national or international settings.

Table 5.5: An overview of the SMART criteria used to assess Tully WQIP nitrate management delivery purposes. Source: Robinson *et al.* (2009a).

Partnership Assessment Criteria	Explanation	Assessment of partnership needs for proposed Tully WQIP nitrate management delivery purposes
Specific	Partnership supports specific place and purpose of activities	- Partnerships need to be flexible to support specific practices that nominally provide a balanced benefit to water quality and landholder needs. - Partnerships that support BMP practice uptake need to address spatial extent of intervention. This can be variable between Tully's landholders and agricultural production communities.
Measurable	Comparable measures of delivery and partnership success identified	WQIP monitoring and evaluation frameworks need to integrate scientific and management expertise to judge the delivery of progress that incorporates economic and social criteria, and determine appropriate thresholds for management practice change.
Achievable	- Thesholds for commitment to delivery identified - Partnerships and activities are witin existing institutional boundary conditions	There is still a need for local, regional and policy arrangements to clarify partner roles to align effort and address concerns that agencies are doing their 'fair share' to support WQIP delivery.
Relevant	Partnerships and activities are understood and applicable to partners	WQIP partnerships need to build on existing co-investment and activities that support local agendas, plans and initiatives.
Timed	Temporal dimensions of partnerships and activity functions considered	- Partnerships established to support BMP incentives need to be responsive to the sequencing of actions and temporal variability (e.g. seasons, markets) affecting practice delivery. - Partnership commitments need to reflect and support the sustained effort required to meet reef targets.

Table 5.6: Knowledge attributes for collaborative water governance: an assessment framework applied to Reef Rescue grants delivery in the Wet Tropics NRM Region.
Source: Robinson *et al.* (2010).

Knowledge Sharing Functions	Attributes	Application Context
1. Integration	Diversity – multiple types of knowledge (local scientific, policy relevant) are identified and recognised Deliberation – institutions support debate between knowledge holders to frame problem and build understanding Inclusiveness – knowledge sharing and problem framing processes accessible and inclusive	Scoping and problem / task framing stage. For policy development / resource allocation decisions
2. Translation	Credibility – knowledge used to inform priorities and actions is credible in terms of trustworthiness and adequacy Legitimacy – decisions and supporting knowledge legitimised through appropriate representation Salience – the provision of knowledge and subsequent decision making is timely and the type of knowledge is appropriate to problem context	Design and implementation stage: To design policy implementation / resource prioritisation strategies
3. Adaptation	Relevance – measure of success or thresholds are cooperatively developed and are relevant to partners' views on 'good' implementation Roles – to monitor and evaluate impact in their respective domains agreed amongst partners Responsibilities – for sharing results from implementation, (i) between partners, and (ii) between scales of delivery, e.g. local-regional, are articulated Capacity – partners have the capacity to incorporate insights from review or monitoring into their own institutional behaviours	Feedbacks for learning / assessing program effectiveness
4. Impact	Outputs and outcomes – monitors and reports on efficacy of partnerships to achieve water quality (and other negotiated) goals. May have short- and long-term components and deliver social (i.e. building institutional capacity) and biophysical (i.e. improved water quality) benefits	Determines progress towards intended outcomes and positive/negative consequences

5.6 Transport of Tully pollutants to the marine environment

5.6.1 Nutrient transport

Dissolved nutrients are transported from the Tully catchment into the marine environment, predominantly over short-term high flow events, and over the wet season. DIN concentrations decrease along an increasing salinity gradient controlled by conservative (dilution) and non-conservative (biogeochemical uptake) processes. Concentrations are reduced from the freshwater end, however the DIN concentrations in the higher salinities (above 30) are elevated (1-5 μM) in comparison to non-flood levels (Furnas, 2003; De'ath and Fabricius, 2008). Concentrations at the freshwater end varied between events, with initial concentrations exceeding 15 μM in 1995 and 2007, in comparison to all other years where the initial concentrations ranged from 5 to just under 10 μM (Devlin and Waterhouse, 2010). Sampling in both 1995 and 2007 captured the 'first flush' events carrying high concentrations of newly mobilised DIN from the fertilised agricultural lands on the adjacent catchment (Bainbridge *et al.* 2009b). DIP increased from the lower to middle salinity ranges, reflecting desorption of dissolved inorganic phosphorus from suspended particles and dilution in higher salinities.

5.6.2 Sediment transport

Wolanski and others have continued to investigate sediment delivery and fate in the Tully marine area through the MTSRF water quality research program. River floods can deposit terrigenous mud directly on to coral reefs (Wolanski *et al.* 2003a). The terrigenous mud originates from river floods and storm-induced suspension and degrades coral reefs by shading and smothering benthic organism (Wolanski *et al.* 2005). Algal mats can further retain the mud and prevent the recruitment of coral larvae. Coral reefs located on a gradient away from the Tully River mouth are thought to be at risk from the increased riverine mud but uncertainty exists as to how much of this mud reaches and impacts on coral reefs.

The bulk of the fine sediment reaching coral reefs from flood plumes comes in the rising stage of the first flush of the river flood. Tully River plumes deposited mud directly within the inner shelf of the GBR including its coral reefs, during the early rising stage of the river flood, as a result of the first flush of soil from the river catchment (Wolanski *et al.* 2008). This mud was later re-suspended by wind-driven waves and redistributed by advection and diffusion due to strong longshore and cross-shore currents to a distance of at least twenty kilometres offshore. The mud deposited below the re-suspension depth (twelve metres in open waters and three metres in sheltered waters; Wolanski *et al.* 2005), may remain on the bottom except if shifted by cyclonic waves. Strong wind events resuspend the mud above this depth and maintain high turbidity over coral reefs until it is advected away or deposited in deeper waters.

The fate of mud in the Tully marine environment is similar to that reported by Wolanski and others (2003b), who showed that the processes are similar in both systems and the impact on coral reefs is similar. In both systems, the riverine mud is initially deposited inshore by river plumes and then re-distributed over a longer time period by waves and currents (Figure 5.12). The mud is initially trapped in sheltered zones behind islands and reefs. In mangrove swamps the freshly deposited sediment is reworked by 'bioturbation' and redistributed by waves and currents while in the GBR it is reworked by waves and redistributed northward by longshore currents and nephroid layers (Orpin *et al.* 2004; Wolanski and Spagnol, 2000; Francis *et al.* 2007). The inshore corals are generally surrounded by a shallow seafloor and waves repeatedly resuspend the material and prolong the exposure of the benthos to suspended sediment until it is moved out of the GBR (Francis *et al.* 2007). The key question for inshore coral reefs of the GBR is whether the yearly gain of riverine mud exceeds the

yearly export of mud by oceanographic processes (Richmond *et al.* 2007). A net sediment budget for the inshore GBR is needed to answer this question.

Previous work (Wolanski *et al.* 2005) had shown that the first flush of sediment associated with a plume event moves a large amount of mud up to ten kilometres offshore, where it does deposit in calm weather. However, in re-suspension events, the mud can be redistributed up to twenty kilometres offshore, extinguishing irradiance over coral reefs at four metres' depth for ten days during the study. Wolanski *et al.* (2008) report on oceanographic processes that enable river plumes to spread at least twenty kilometres offshore, twice as far as predicted by numerical models (King *et al.* 2002).

In terms of risk, it is the continuing supply of the finer sediment made available during the re-suspension of mud over the whole year that can potentially expose corals to longer turbidity events. This sedimentation occurs during the wet season, which is also when juvenile corals have just settled on the substratum, and these juveniles are particularly susceptible to nutrient-rich sediments (Fabricius *et al.* 2003). The implication is that the GBR coral reef ecosystem extends into adjacent watersheds, and their conservation in Marine Protected Areas will fail without a decrease of soil and nutrient erosion in the adjoining river catchment coupled with the creation of terrestrial protected areas to act as buffer zones (Richmond *et al.* 2007).

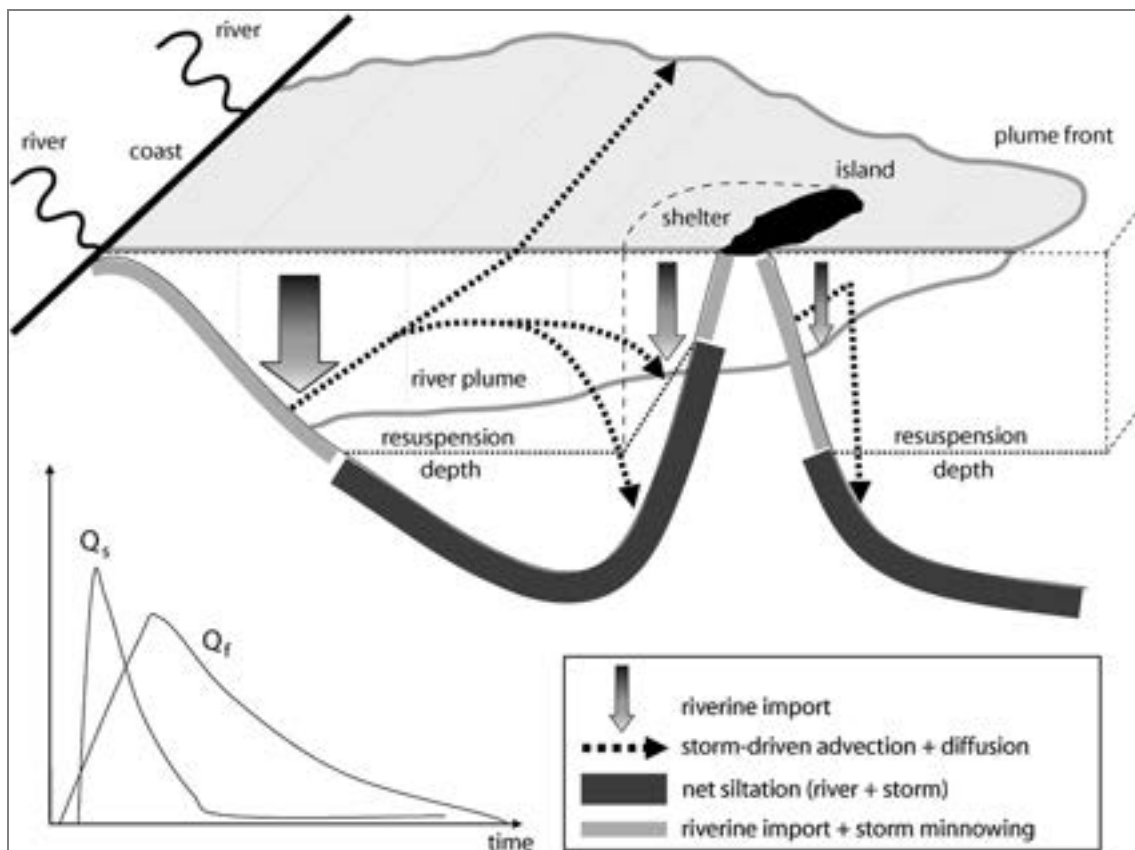


Figure 5.12: Sketch of the dynamics of river and fine sediment in the transient river plume, highlighting mud deposition during the first flush of eroded soil from the catchment during the rise in river flood waters, the spread of river plumes, the re-suspension and minnowing of that mud during storms and its spread in a wide area of the inner shelf, the preferential settling of that sediment below the resuspended depth, and the accumulation of sediment higher up in the water column in the shelter zone in the lee of islands and reefs. Source: Wolanski *et al.* (2005).

5.6.3 Extent and exposure of Tully plume waters

Status of Tully marine areas

In the Wet Tropics region, water quality, flood plumes, seagrass habitats and coral reefs are all monitored as part of the Reef Rescue MMP, including a number of sites in the Tully marine area. Overviews of monitoring results are found in Johnson *et al.* (2010), Prange *et al.* (2007, 2009) and Haynes *et al.* (2005). However, a few key points included below from an assessment of data collected between 2006 and 2009 (included in Johnson *et al.* 2010) provide an indication of the status of water quality and ecosystem health in this area.

- Annual and seasonal suspended solids means have exceeded the Water Quality Guidelines for the Great Barrier Reef Marine Park for suspended solids, likely due to flood events and re-suspension at the Dunk Island sites.
- Seasonal means of chlorophyll and suspended solids for the four years of monitoring exceeded guideline values at Dunk Island, which generally had the highest seasonal means of all sites in this region.
- Inshore reefs in the Wet Tropics are regularly exposed to detectable concentrations of pesticides during flood plumes (Lewis *et al.* 2009b) and concentrations at inshore reefs are measurable during both the wet and dry seasons (see also Prange *et al.* 2009). The consequences of this chronic exposure are currently unknown and further investigation is required. However, chronic stress due to poor water quality is likely to manifest as either an increase in the susceptibility of corals to disturbance events such as thermal bleaching (Wooldridge, 2009) or inhibition of their recovery following a disturbance. Either or both of these outcomes would result in a change in community composition. Such shifts are likely to occur after disturbance events as species suited to the changed environmental conditions will predominantly re-colonise available substratum. This differs from non-disturbed communities where gradual shifts in environmental conditions may be masked by physiological (Anthony and Fabricius, 2000) and morphological (Anthony *et al.* 2004) plasticity of corals that allows existing colonies to persist in conditions they would not be able to recruit into, forming relic communities.
- Seagrass cover in the region, although seasonal, has generally increased or stabilised over the past year and is naturally lower at coastal compared to reef habitats. Seagrass reproductive health status over the period 2006 to 2009 was poor at Green Island (reef) and Lugga Bay (coastal), and variable at other sites. In 2008 and 2009, seed counts were generally lower than in previous years. A continued absence of flowering and fruiting at these sites will result in poor capacity to recover from disturbance. Inter-annual differences in sexual reproduction are evident and these differences principally relate to the decline of meadows.
- Devlin and Schaffelke (2009) reported that approximately 93% of seagrass meadows within the Tully subregion of the Wet Tropics were inundated every year by primary flood plumes, exposing the seagrass to intermittently high sediment and high nutrient concentrations and potentially high loads of total suspended sediment.
- Lugga Bay and Dunk Island are also located within the modelled diuron (0.1-0.9 ng/L) first flush plume zone for the Tully and Murray Rivers (Lewis *et al.* 2009b).
- Although no pesticides were present in seagrass sediments during the 2008 and 2009 monitoring period, they have been reported previously from Lugga Bay in April 2006 (McKenzie and Unsworth, 2009). Pesticides have never been reported from seagrass sediments on Dunk Island, however monitoring was not established at this location until late in the 2006 dry season.

- Seagrass epiphytes and tissue nutrients at coastal habitats suggest nutrient saturated conditions, with potentially low light availability. Seagrasses in reef habitats are growing in clearer waters (higher light environments) and are nitrogen limited.
- Coral community status scores were negative for reefs in the Herbert/Tully sub-region. On average, reefs in these locations had relatively high cover of macroalgae and moderate to low coral cover with no signs of recovery from past disturbances (e.g. Tropical Cyclone *Larry*). This may be an indication of local environmental conditions hindering recruitment. However, more surveys over time are required to detect any consistent trend.

Flood plume exposure and extent

Flood plume monitoring is undertaken as part of the Reef Rescue MMP with the Tully marine area being one of the focus areas for study of plume characteristics and plume dynamics. A single example of the information that can be derived from plume movement out of the Tully and Herbert Rivers has been identified using remotely sensed images derived on 14 January 2009 (Figure 5.13). The top image illustrates the primary and secondary plumes associated with the Herbert and Tully River floods, and shows the high chlorophyll levels associated with these primary and secondary plume waters. A very turbid inshore plume can be seen south of the Herbert River and extending north of Dunk Island. The lower images show the calculated CDOM and chlorophyll for 14 January 2009. These images indicate that the influence of terrestrial discharge may extend a considerable distance beyond the outer reefs as tertiary water types.

The flood plume exposure map for the Tully River (Figure 5.14) was calculated from the intersection of aerial and plume images taken from both aerial surveys (1995-2000) and remote sensing images (2003-2009) for the Tully sub-region (Devlin and Schaffelke, 2009). Thirty-seven reefs and fourteen seagrass beds in the Tully region were exposed to some degree to riverine plume waters during eleven flood events from the period 1994 to 2007. Over the eleven years, a minimum of eleven reefs (30%) and a maximum of 37 reefs (100%) were inundated by either a primary or secondary plume, indicating that it is likely that at least a third of the reefs are exposed to plume waters every year. In years with data to validate plume type (1998, 2003-2008), it is estimated that 6-15 reefs were inundated by primary plume waters carrying high sediment loads, which is up to 41% of the inshore reefs in the Tully sub-region and 5-16 reefs (43%) were inundated by secondary plumes with elevated nutrient and chlorophyll concentrations. These exposure rates have significant implications for the transport of sediments and nutrients onto inshore reefs, and on the settlement and survival of corals, and growth of macroalgae.

Recent mapping work (Devlin *et al.* 2010) has now identified water types common to the Tully marine area and which are likely to occur during the wet season (Figure 5.15a). The mapping of the most common plume water types shows a transitional phase of freshwater to primary waters, secondary and tertiary water types. The water types represent particular characteristics including high TSS in the primary plume, high chlorophyll *a* and CDOM in the secondary, less turbid plume and elevated CDOM as an indication of freshwater extent in the tertiary plume. The water quality values associated with each water type help identify which areas are likely to exceed current water quality guidelines for chlorophyll *a* (Figure 5.15b) and TSS (Figure 5.15c) (GBRMPA, 2009).

A maximum CDOM absorption map has been generated from January to March of each year through aggregation of daily CDOM imagery for the Wet Tropics Region (Figure 5.16). The definition of plume extent through the mapping of CDOM concentration identifies the full extent of the flood plume and allows a year to year calculation of the plume extent (measured in kilometres), which may be useful as a monitoring tool.

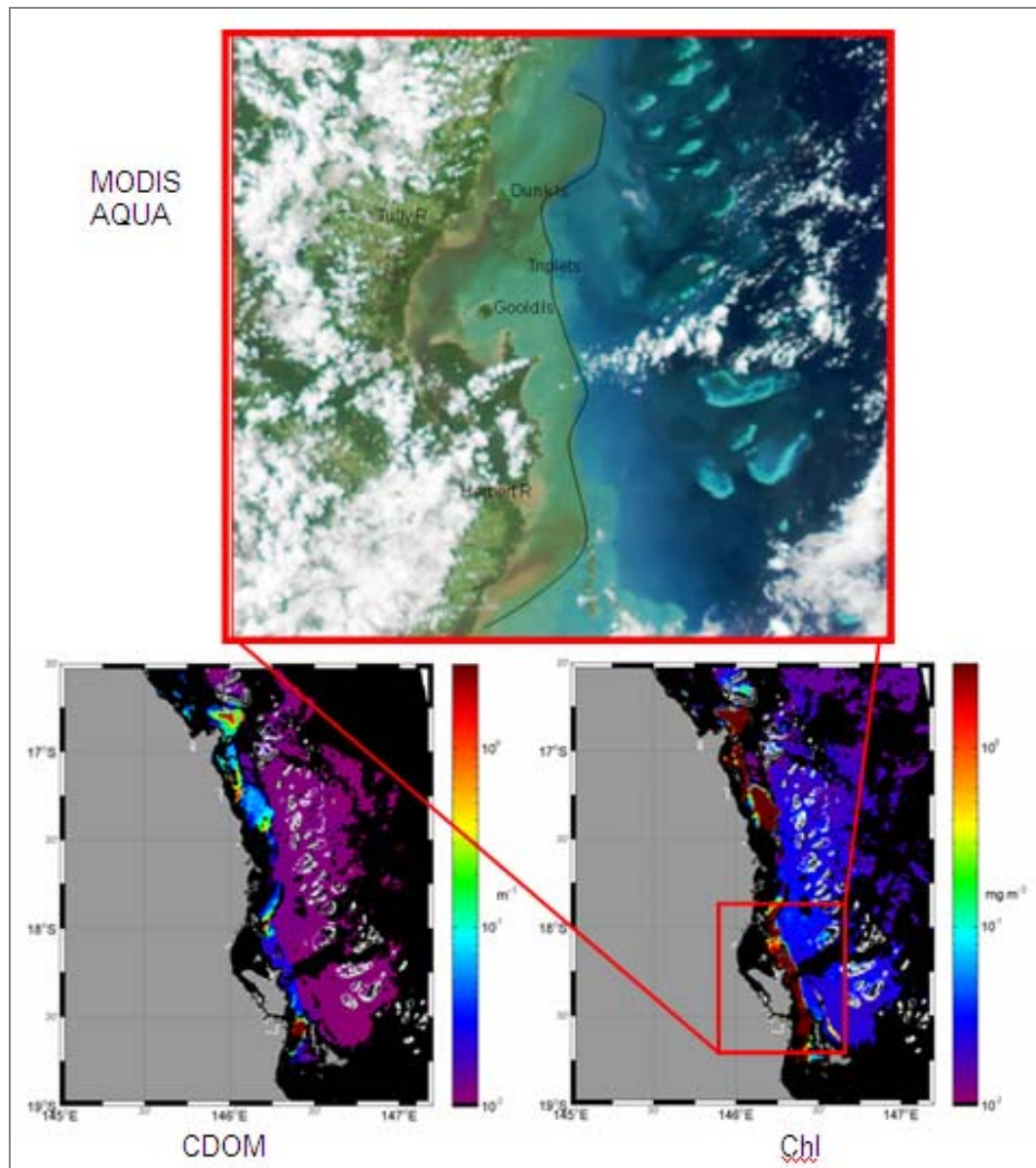


Figure 5.13: Remotely sensed (comparable to true colour) NASA images, CDOM and chlorophyll, 14 January 2009. Source: James Cook University.

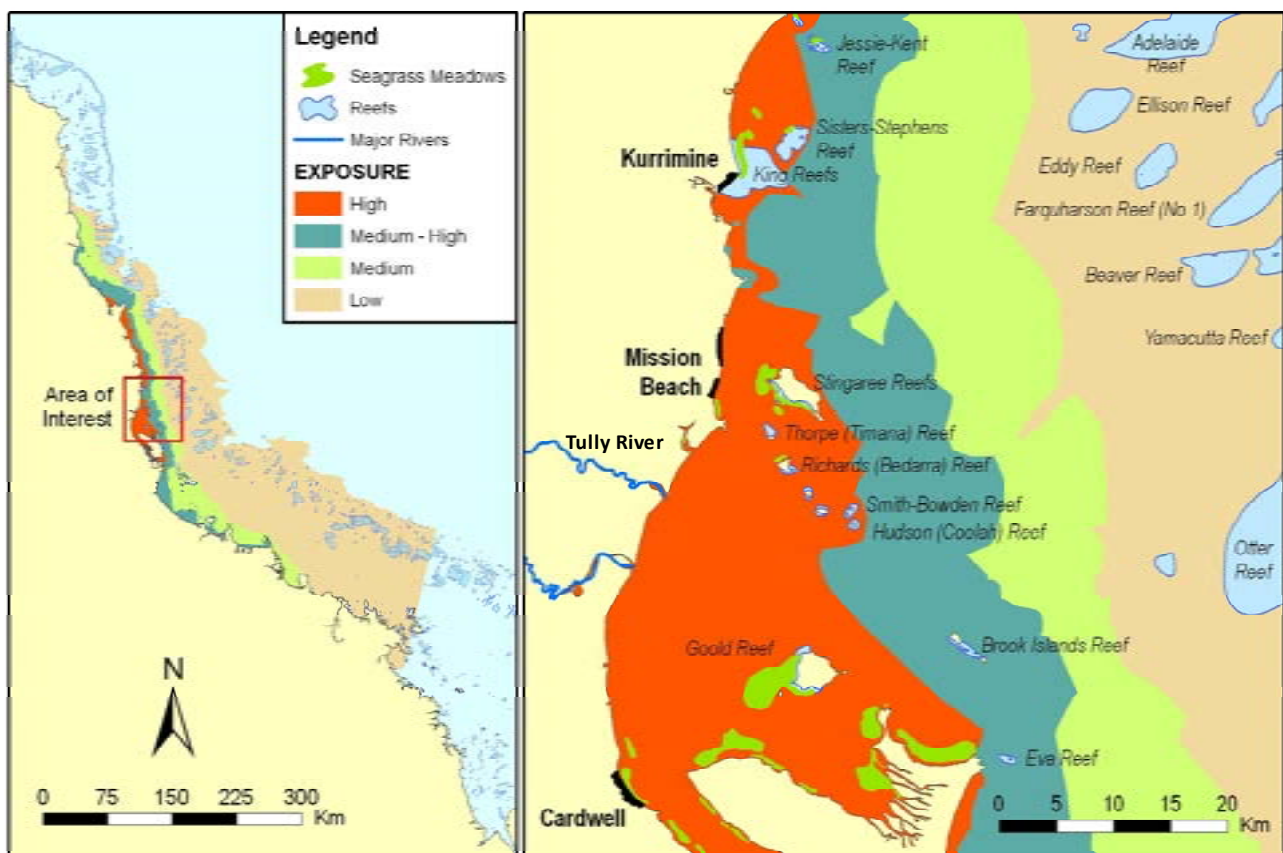


Figure 5.14: Exposure map for the Tully marine area. Image constructed from GIS imagery of plume extents from 1994 to 2008. Source: Devlin and Schaffelke (2009).

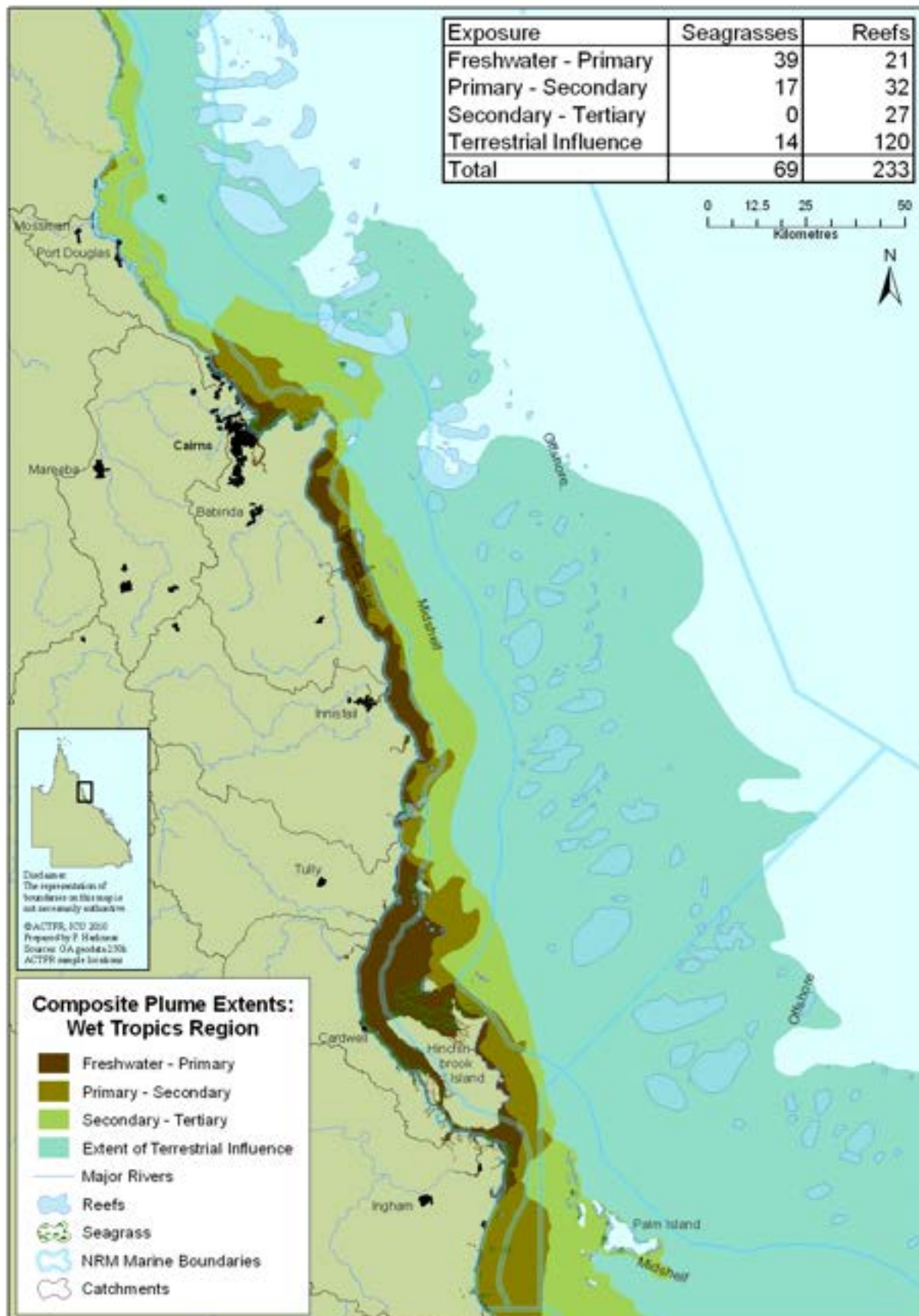


Figure 5.15(a): Extent of plume types in the Wet Tropics region. Source: Devlin *et al.* (2010).

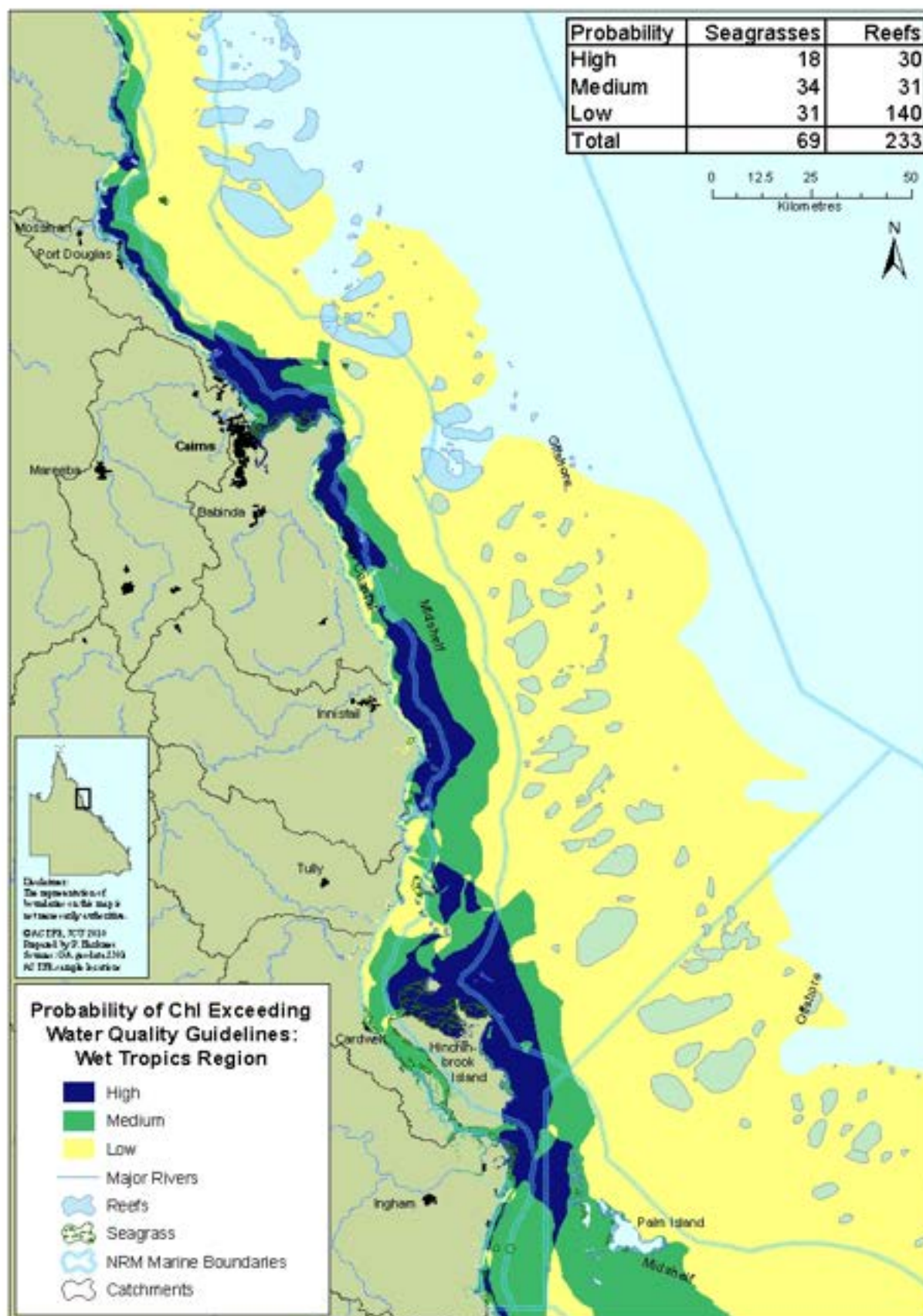


Figure 5.15(b): Identification of areas most likely to exceed water quality guidelines for chlorophyll a in the Wet Tropics. Source: Devlin *et al.* (2010).

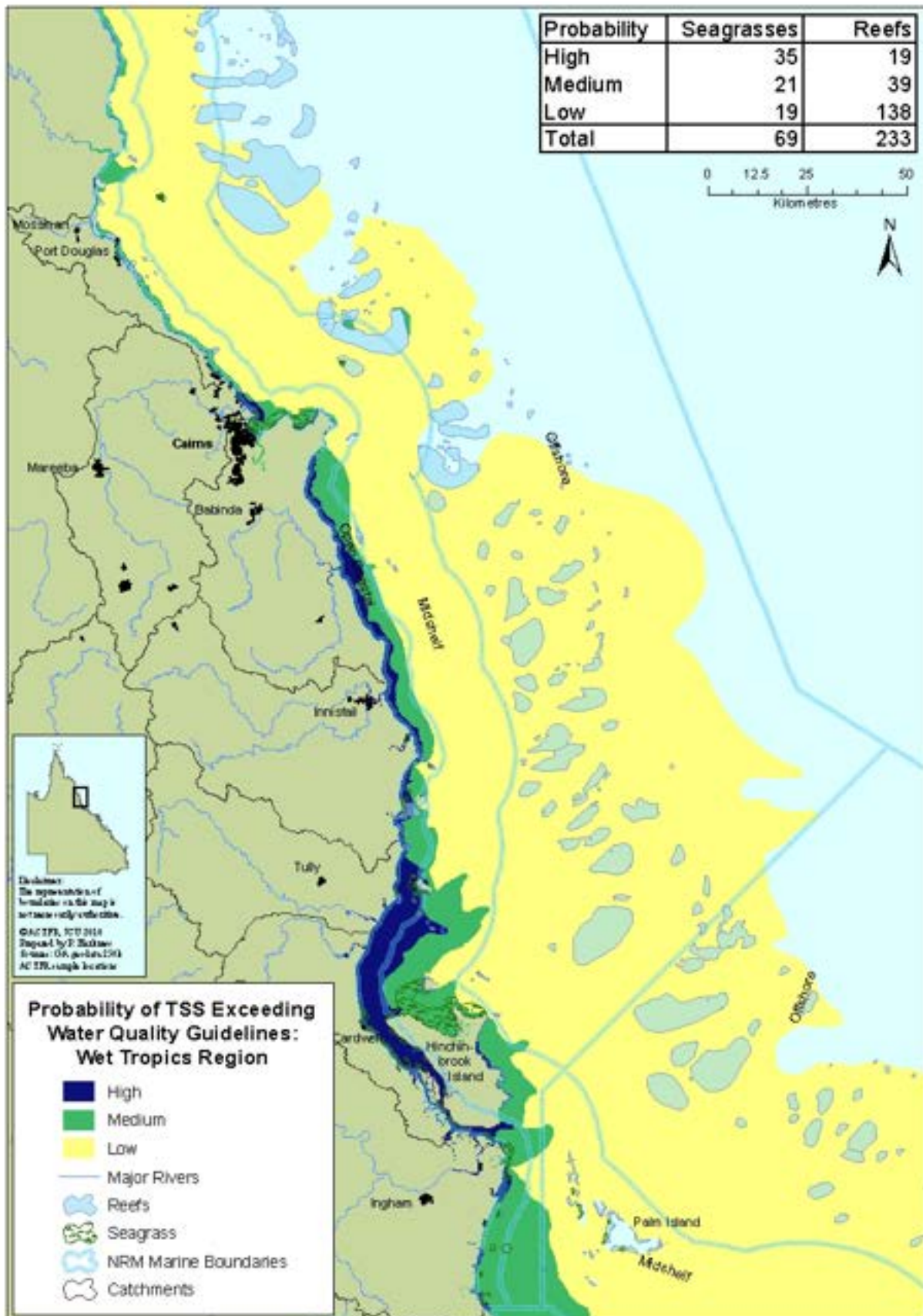


Figure 5.15(c): Identification of areas most likely to exceed water quality guidelines for TSS in the Wet Tropics. Source: Devlin *et al.* (2010).

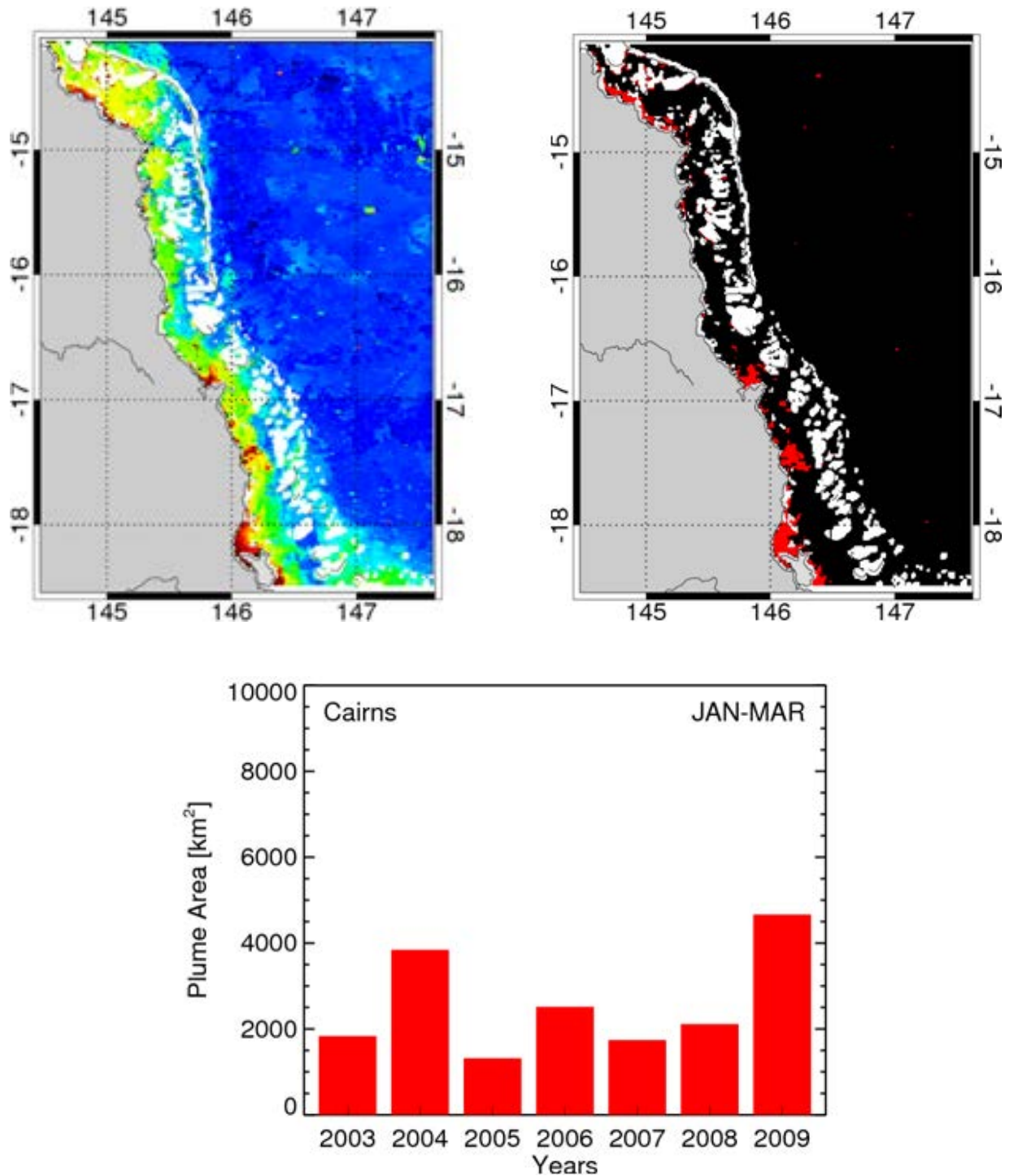


Figure 5.16: Maximum CDOM absorption from regional parameterised ocean colour algorithm mapped for the period January to March 2008 for the Wet Tropics region. Freshwater plume extent is mapped by applying a CDOM threshold derived from linear regression of *in situ* CDOM and salinity measurements. Source: Devlin *et al.* (in press-b), Schroeder *et al.* (in press).

Direct water sampling in 2008 and 2009 indicated that most water quality variables at Dunk Island (Wet Tropics region), Magnetic Island (Burdekin region) and Pelican Island (Fitzroy region) did not comply with guideline values. The inshore coral reefs and seagrass beds adjacent to the Tully catchment are likely to be affected by these elevated concentrations at least during the weeks of exposure. The longer-term impacts of flood plumes are currently not well understood but are subject to ongoing research. These include, for example, recurrent re-suspension of settled material leading to periodically elevated TSS concentrations over long time periods or ongoing high nutrient availability from foodweb cycling.

However, single exceedances of the guideline values have been identified from flood plume water quality data and used to extrapolate plume behaviour in correlation with river flow and remote sensing images. Further work to obtain integrated time series data throughout high flow events, including more extensive sampling of depth profiles and continuous *in situ* logger data in combination with *in situ* surveys of coral reefs, will assist in improving the correlation of flood monitoring data with the long-term changes in pollutant concentrations and ecological impacts.

5.7 Improved understanding of the conceptual model for the Tully region

A number of conceptual models have been produced for the GBR (Haynes *et al.* 2007) which outlines our current understanding of the connectivity between catchment and reef, and movement from small scale paddock sites, through wetlands, rivers and the coastal environment. MTSRF research presented in this summary and in companion reports to this summary (Waterhouse, 2010; Waterhouse and Brodie, 2010) has progressed our understanding of this conceptual model, both for our whole GBR understanding and also specific to regional areas, such as the Tully. Figure 5.17 summarises this progress.

Recent MTSRF research has focused on the understanding of the link between agricultural activity and pollutant load, particularly in respect to the movement of dissolved inorganic nutrients from fertilised agriculture. Research findings in the Tully have focused on improved understanding of freshwater indicators and how the response of key ecosystem metrics can be used to assess a much larger freshwater area. The freshwater research has identified the importance of the Tully wetlands and their unique position as ecologically important functioning wetlands. The continuum between the freshwater ecosystems and the marine environment has been further investigated by the role of overbank flow, and the connectivity between wetlands in small to large flow events. The Tully wetlands form disparate water systems that connect for days to a week over short-term flow events. The lengths of these connections have important consequences for the freshwater and estuarine biota that move through these wetlands. Better understanding of the Tully wetlands and freshwater systems will have important implications for (i) the movement and recruitment patterns of aquatic biota during and after flood events, (ii) wetland habitat characteristics and water quality, (iii) the biodiversity of individual wetlands over time, and (iv) the potential for wetland processes to influence the quality of water flowing to the GBR lagoon (Kroon, 2009).

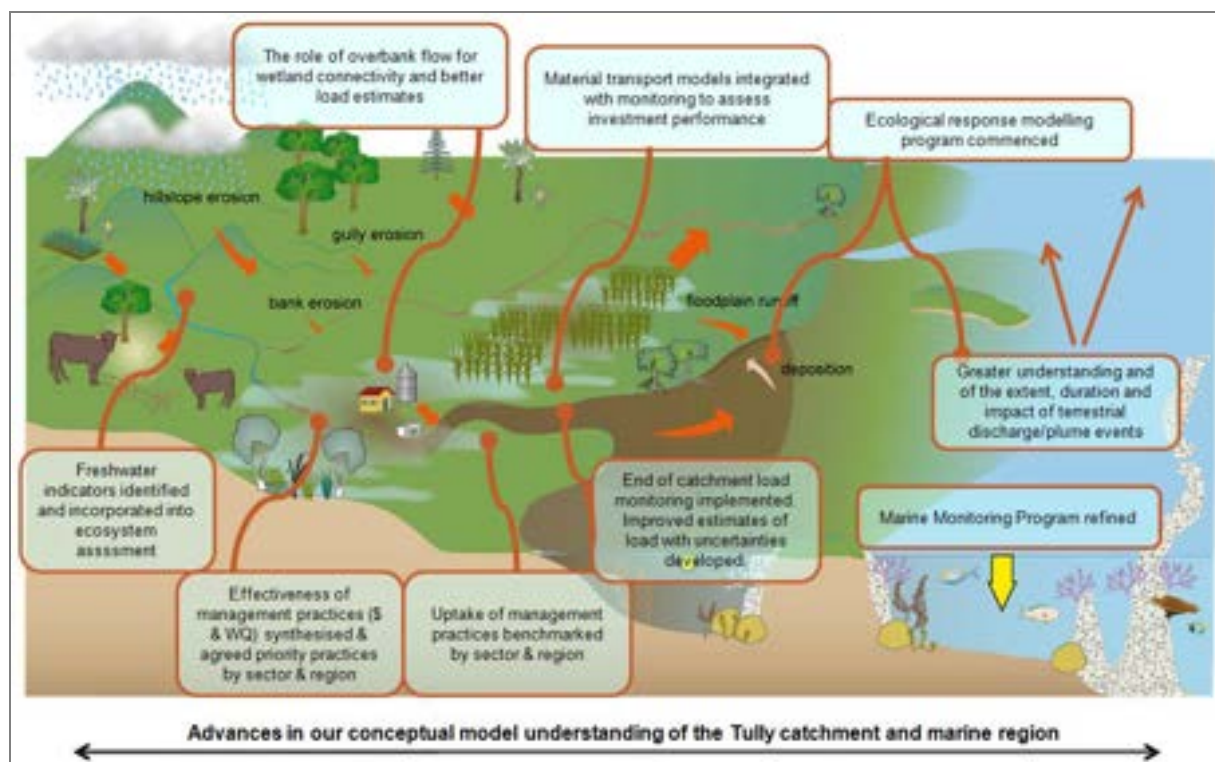


Figure 5.17: Changes in our understanding of the current Tully conceptual model (base model derived from Prange, 2007).

Better understanding of overbank flow (not measured by the river gauging system) has given a better estimate of nutrient and sediment loads moving into GBR waters. Research on load estimates, through better understanding of the variability inherent in load calculations from the episodic Wet Tropics rivers, accounts for the importance of the first flush and subsequent, even multiple, events. More appropriate and accurate statistical analysis allows a more comparable estimate on load reductions and potentially reduces the amount of time that would have been required for the detection of change.

6. Questions answered and future directions

The development of this progressive approach to catchment management has been somewhat forward of our ability to accurately measure and monitor subsequent physical and chemical response in the marine environment. The bulk of research within the catchment-to-reef framework has been to build further knowledge of relationships between catchment and reef ecosystems in order to develop more appropriate assessment systems for catchment and GBR ecosystems and to design and implement suitable management responses. Research has focused on freshwater status, connectivity between upstream and downstream environments and on the movement, delivery and impact of pollutants. This work has built on the efforts of the joint Rainforest CRC and CRC Reef 'Catchment to Reef' Program (1993-2006). A summary of gaps in knowledge prior to the commencement of the MTSRF water quality research program are listed in Table 6.1, with an indication of the success of MTSRF funded research in filling those gaps. Water quality research has been carried out over the last four years to address these questions and to facilitate understanding of the catchment-to-reef connectivity processes. The areas which are still poorly understood or where understanding is limited are also identified here and should form the basis of continuing research programs.

Table 6.1: Identification of research gaps as recognised prior to the commencement of MTSRF funded projects, and identification of the success of each item on completion of the MTSRF Research Program in 2010.

Gap	Details	Success
Research Area: Catchment Health		
New monitoring tools	New tools required for water quality assessment against benchmarks. Water quality monitoring has frequently been inadequate due to limitations of full temporal and spatial sampling requirements. Previous focus on major events and end of river. Monitoring frameworks are required to address the issues of why, what, when and how to monitor water quality and system health.	Whole-of-system, multi-scale monitoring and evaluation framework developed and adopted in the GBR.
Geographic extent of knowledge	Expansion of understanding of Wet Tropics streams and rivers programs to include the Dry Tropics and standing water bodies.	Major progress in Wet Tropics, more work required in Dry Tropics.
Holistic assessment of system health	Holistic assessment of system health is required because water quality <i>per se</i> is only one of several important issues such as habitat degradation, riparian condition, invasive species and normal variation in natural processes.	Holistic assessment methods available for streams and floodplain lagoons.
Sub-lethal stress	Need for tools to detect sub-lethal stress in aquatic organisms exposed to impacts that can shortcut the often slow traditional methods of detecting environmental impacts. There is also a need for more sensitive and unambiguous indicators of environmental quality, such as measurements of the physiological stress that develops in organisms long before conditions become so bad that populations or communities change substantially or crash.	Some progress on fish and invertebrate responses to hypoxia, nutrients and sediments.
Quantifying connectivity	Understanding connectivity between catchments and the GBR, including further understanding of biological links between sea and fresh water and catchment waterways, and quantitative importance of connections for freshwater and marine species.	Models of freshwater connectivity achieved, more work required on links between sea and freshwater systems.

Gap	Details	Success
Ecosystem function models	More explicit and quantitative models of stream, river, estuary and wetland function.	Improved understanding achieved but model development still required.
Role of the riparian zone	Quantify its roles to facilitate management activities aimed at controlling water quality while simultaneously sustaining processes vital for river ecosystem health. There is a clear need for tools to quantify the filtering role of the riparian ribbon and the effects of different land uses.	This work was proposed but not funded through the MTSRF Research Program.
Role of wetlands	Quantification of processes within wetlands, including the support of biological communities, the storage of fresh water, arresting flows to the sea and enhancing prospects for perennial aquatic habitat.	Biological communities are much better understood in GBR wetlands.
Role of groundwater	Lack of knowledge about groundwater-fed systems.	Limited improvement in knowledge. Further work required.
Identifying and managing new impacts	Need for good models of how different climate change scenarios might directly affect pristine ecosystems.	Connectivity models developed through MTSRF will assist with this.
Role of Best Management Practices (BMPs)	Lack of quantitative evidence linking BMPS with water quality benefits to downstream waterbodies.	Future research priority.
Research Area: Catchment Water Quality		
Gully erosion	Main causes of gully erosion in the landscape and best remedial management practices.	Some progress in Normanby catchment and others outside the MTSRF geographic extent. Future research priority.
Role of riparian vegetation	The role of riparian vegetation in stabilising stream bank erosion on different sized streams.	Improved understanding in the Wet Tropics, further work required in Dry Tropics.
Agricultural best practices	What grazing land and/or riparian zone management practices will best maintain productivity and biodiversity and material trapping capacity of these systems?	Some progress outside of the MTSRF. Future research priority.
Variability in TSS	What are the major drivers for this large variability in suspended sediments in terms of natural and/or anthropogenic factors?	Improved understanding in the Burdekin and Normanby catchments, further work required across the GBR.
Source and extent of TSS	Origin of fine grained, non settling suspended sediments, specifically in the small but high risk component that is transported out to sea.	Successful study in Burdekin catchment.
Residence times of suspended sediments	Residence time of varying particle size fractions of suspended sediments being transported through the catchment.	Major progress made but further work required.

Gap	Details	Success
Impact of suspended sediments loads to GBR turbidity	Do increased suspended sediments loads due to increased erosion from agricultural and urban development in major rivers lead to increased regional turbidity generated by re-suspension in inshore areas of the GBR lagoon?	Major progress made but further work required across the GBR.
Burdekin dam trapping efficiency	Is the Burdekin Falls Dam a highly efficient suspended sediments trap as predicted by SedNet? Implications of land use management above and below the dam.	Successful – the Burdekin Falls Dam traps 60% of sediment on average.
Research Area: Marine Water Quality		
Importance of DON	DON is a large component of the nitrogen load in many rivers and the degree of its bioavailability will be a critical factor in assessing the risk to both fresh and marine ecosystems from nitrogen driven eutrophication.	Export of DON estimated for GBR. Further work on source and impact required.
Revised risk assessment	Relative risks to GBR ecosystems of the individual terrestrial-sourced pollutants. Weighting of risk relative to pollutant. Identifying the degree of deviation from natural plus potential consequences.	High risk areas identified. Further work on the links with impact required.
Marine health	Status of GBR ecosystems and identification of causal links to known cases of disturbance and degradation	Comprehensive monitoring program in place. Ongoing work is identifying long-term status.
Evidence of water quality gradient	Is it possible to detect a water quality gradient, based on biological measurements? Can we correlate poor water quality with changes in the biota?	Successful studies in Whitsundays. Model and statistical outputs.
Marine indicators	Identification and uptake of marine indicators that can be utilised in the Reef Rescue MMP	Successful and ongoing.

7. Conclusions

This report has presented a summary of recent research in the GBR catchments, with particular emphasis on the Burdekin and Tully catchments. Current monitoring information is also presented that has benefitted from the advances outlined in this work. However, the impact of the pollutant has been presented as a single pressure or stressor and is usually discussed as a single driver. The proposed synergy between nutrient loads and susceptibility of corals to thermal bleaching (Wooldridge, 2009) should be highlighted as a concern to the ongoing resilience and health of GBR waters. Increased sea temperatures have globally increased the frequency of broad scale and severe mortality events of coral reefs (Hoegh-Guldberg, 1999, 2004; Wilkinson, 2000). The poor status of coral reef communities (Thompson *et al.* 2010) in the Burdekin and Tully regions is likely to be the result of coral mortality during the mass bleaching events in the summer of 1998 and 2009 (Berkelmans *et al.* 2004; Sweatman *et al.* 2007) and subsequent limited recovery. Susceptibility to thermal stress can be heightened by poor water quality and recovery hindered (Hoegh-Guldberg *et al.* 2007). The negligible increase in coral cover in the Burdekin region may be due to a lack of larval supply and low survival, indicated by regionally low settlement of spat and low density of hard coral juveniles (Thompson and Dolman, 2010). With the frequency and severity of disturbance events projected to increase in response to continuing rises in greenhouse gases (Steffen, 2009) any increase in coral susceptibility to thermal stress as a result of local anthropogenic nutrient loads will have significant consequences for GBR inshore reef communities. Interactions between water quality and climate change are poorly understood and require further practical investigation.

This report provides a summary of relevant MTSRF funded work and publications to highlight advances in our understanding of the links between the GBR catchment and long-term health of the GBR ecosystem. The work presented ranges across many scientific disciplines, allowing a cross analysis of the current status of water quality inputs, status and biological impacts. Using a selection of the research outputs, we can identify how the current work has built on the state of knowledge at the onset of the MTSRF research period. This showcases the breadth and diversity of the MTSRF research presented in this report and companion reports by Waterhouse (2010) and Waterhouse and Brodie (2010). The success of the MTSRF model comes primarily from research crossing over many scientific disciplines, including socio-economic and economic work. The combinations of this multi strand of information have allowed a comprehensive approach to studying the catchment-to-reef process. Paddock-to-catchment-to-reef is a myriad and complex process that requires this cross-boundary approach. All parts of the process are linked and so it must be studied as an integrated system.

To conclude, the key findings of the MTSRF water quality research program in relation to the processes that connect the whole catchment to reef system are summarised below.

- Conceptual biophysical models have been developed to identify appropriate indicators of freshwater ecosystems, including stream, floodplain lagoon and wetland health. Probable thresholds of concern, in terms of contaminant concentrations, ecological processes and biodiversity have been investigated for these ecosystems. Indicators of freshwater ecosystems have been developed and are related to pressures that include patterns and types of land use, general water quality and contaminants, hydrological regime, channel and habitat structure, riparian vegetation condition and alien species of plants and fish. Measurements of spatial and temporal variability of biophysical indicators in floodplain wetlands of the Tully-Murray catchment have been correlated with those pressures.
- Connectivity between freshwater ecosystems is important for maintaining ecosystem health and has been studied using hydrological modelling in the Tully-Murray floodplain

area. The degree of connectivity of different wetlands, ranging from those wetlands that are more permanently connected with streams and drains to those that are connected only when there are large overbank floods, varies with wetland location and flood magnitude. These results have important implications for (i) the movement and recruitment patterns of aquatic biota during and after flood events, (ii) wetland habitat characteristics and water quality, (iii) the biodiversity of individual wetlands over time, and (iv) the potential for wetland processes to influence the quality of water flowing to the GBR lagoon. As the hydrodynamic model is driven by daily rainfall it should also be possible to quantify the potential impacts of climate change on wetland connectivity if the future changes in rainfall can be specified.

- Sediments, nutrients and pesticides are the priority pollutants for management of water quality in the GBR. MTSRF funded studies have informed the refinement of knowledge of priority areas for pollutant generation and, hence, management in the GBR catchments.
- In the Burdekin River catchment, sediment load is dependent on the catchment characteristics and size of flow event. However, regardless of flood event size in the other catchments, the upper Burdekin basin is always likely to be the dominant source (83-97%) of suspended sediment into the Burdekin Falls Dam. TSS load delivered over the dam spillway makes up a smaller proportion (20-50%) of the total load exported from the Burdekin River than the below dam catchment area, and it is estimated that 50-80% of the suspended sediment export ('bulk' suspended sediment) to the GBR lagoon has been sourced from the catchment area below the Burdekin Falls Dam. Thus, management efforts should primarily focus on these lower catchments which make up only a smaller percentage of the overall Burdekin catchment area.
- Studies in the Burdekin catchment show that there are different delivery pathways between the bulk (heavier) sediment and the finer sediment. There is little deposition of the finer clay fraction as it is transported within the catchment compared to coarser size fractions (such as silts and sand) which are preferentially being deposited within the dam or during other opportunities for deposition. Size distribution shows the movement of the finer sediment from the upper catchments, through the dam and into the marine environment. These results are also relevant to other Dry Tropics catchments in the GBR. Further studies show that the finer fraction (<38 µm component) of the sediment is present in the turbid primary plume which is generally constrained closer to the coast but was not seen in the larger secondary plume as inorganic matter. These latest particle size results indicate that the finer clay fractions are being transported not only throughout the catchment with little opportunity for deposition, but also within the marine environment via resultant flood plumes. It is this finer fraction which has been linked to the degradation of coral reef ecosystems and therefore may pose the greatest risk to receiving marine ecosystems.
- Building on this knowledge, receiving water models can be used to develop sediment budgets for areas within the GBR. For example, a hydrodynamic model has been developed for Cleveland Bay (receiving waters of the Burdekin River) which shows that the amount of riverine sediments settling on the bay may exceed the amount of sediment exported from the bay by 50-75%. Sediment is thus accumulating in the bay on an annual basis, with potentially negative effects on coral reefs. A net sediment outflow from the bay may only occur during years with a tropical cyclone. Thus the majority of the sediment accumulates in areas where it is frequently resuspended by waves under trade winds, thus increasing the turbidity of the bay.
- In the Tully-Murray River catchments, estimates of nutrient loads being delivered during flood events to the GBR lagoon have been significantly underestimated in the past. MTSRF research has shown that flood contributions increase the mean annual loads of phosphorus and nitrogen loads by 30-50% above previous river based estimates. These results indicate that there is therefore a clear need to obtain estimates of the contribution that floods make to marine loads in other GBR catchments.

- Comprehensive research on the impact of sediments and nutrients on the GBR ecosystems has been undertaken as part of the MTSRF and the preceding joint 'Catchment to Reef' Program of the CRC Reef and Rainforest CRC, and can be represented in a series of conceptual models. This work has led to the development of 'thresholds of concern' for several water quality variables and ecosystem components, which in turn have been used in the development of Water Quality Guidelines for the Great Barrier Reef Marine Park (GBRMPA, 2009). The research has also demonstrated a link between elevated concentrations of nutrients and the location and frequency of COTS outbreaks.
- Studies on the effects of herbicides on GBR ecosystems have shown that herbicides are being detected in many locations in the GBR, especially following rain events, and that increased exposure can potentially threaten ecosystems within the GBR. The herbicides most commonly detected in the GBR lagoon are designed to inhibit PS-II in plants and so the risk of these herbicides should be considered additively. Previous studies have examined the risk of individual herbicides in isolation; recent monitoring studies show that 80% of the time when herbicides are detected, two or more herbicides are present in the GBR lagoon following wet season river discharge and, consequently, the area at risk to pesticide exposure increases when the additive risk is considered.
- Coral cores have been used to track change in material delivery to the GBR over long time periods. Coral Ba/Ca ratios in both the short and long-term coral core records display an increasing trend over time, particularly after European settlement (c. 1880) and in the last thirty years, although peak values do not always coincide with river floods. In addition, the geochemical results from coral cores collected along a water quality gradient through the Whitsunday Islands have been useful in establishing local and regional patterns of terrestrial influence factors. These patterns correlate with an increased chronic terrestrial influence in the Whitsunday Islands. However, coral Y/Ca ratios typically lack long-term trends, although peaks do generally relate to river discharge. Ba/Ca records from a long-lived coral (>100 years) show a close correspondence with the generally annual river discharge peaks, providing further evidence that this approach provides a good proxy for changes in terrestrial inputs in the Wet and Dry Tropics.
- Recent publications presented for the Tully (Kroon, 2009) showcased MTSRF supported research as a key component in the detailing of this ecosystem approach within the Tully catchment and marine region. In summary, this work included the estimate of the contribution of overbank (flood) flows to total pollutant loads, previously not taken into account in load estimates to the GBR (Wallace *et al.* 2009a, 2009b). Maughan and Brodie (2009) provide a spatial model to visualise GBR exposure to land-sourced pollutants under current and changed land-use regimes. Devlin and Schaffelke (2009) identified the transport and extent of pollutants in Tully flood plumes, and identified areas of high exposure and ultimately at high risk from the impacts of altered land-use activities. This was reported as the number of marine biological systems that were frequently inundated by higher concentrations of sediment, nutrients and pesticides. The challenge to produce target estimates from catchment models with known levels of uncertainty, but robust enough for management purposes, was examined by Brodie *et al.* (2009a). The outcomes of these inter-related studies have contributed significantly to our capacity to understand and predict direct and indirect relationships between land use and management, impacts on water quality and flow-on effects on marine biodiversity.

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Note: All references generated through MTSRF Program research are indicated by an asterisk (*).

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