Assessment of the relative risk of degraded water quality to ecosystems of the Fitzroy NRM Region, Great Barrier Reef

October 2015

Prepared by TropWATER James Cook University for the Fitzroy Basin Association

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A Report for the Fitzroy Basin Association

October 2015

Jane Waterhouse, Dieter Tracey, Jon Brodie, Stephen Lewis, Eduardo da Silva, Michelle Devlin, Amelia Wenger, Dominique O'Brien, Johanna Johnson, Jeffrey Maynard, Scott Heron, Caroline Petus

Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER)

James Cook University

Townsville

Phone : (07) 4781 4262

Email: TropWATER@jcu.edu.au

Web: www.jcu.edu.au/tropwater/

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For further information contact:

Jane Waterhouse
Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER)
James Cook University
Email: Jane.Waterhouse@jcu.edu.au
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Executive Summary

A risk assessment method was developed and applied to the Fitzroy Natural Resource Management (NRM) region in the Great Barrier Reef (GBR) to provide robust and scientifically defensible information for catchment managers on the key land-based pollutants of greatest risk to the health of the two main GBR ecosystems (coral reefs and seagrass beds) in the region.

The main water quality pollutants of concern for the whole GBR are enhanced levels of suspended sediments, excess nutrients and pesticides (predominantly photosystem II-inhibiting herbicides) added to the GBR lagoon from the adjacent catchments. Until recently, there has been insufficient knowledge about the relative exposure to and effects of these pollutants to guide effective prioritisation of the management of their sources. This assessment has attempted to utilise the best available information to assess the differences between the basins in the Fitzroy NRM region in influencing GBR ecosystems.

The relative risk of degraded water quality among the basins in the Fitzroy region was determined by combining information on the estimated ecological risk of water quality to coral reefs and seagrass meadows in the region with end-of-catchment pollutant loads. The framework was developed from the approach used in the GBR-wide relative risk assessment conducted by Brodie et al. (2013a) to inform Reef Plan 2013 priorities, and modified where necessary to reflect issues and data availability in the Fitzroy region. There are also several improvements to the input data in this assessment including: definition of zones of influence for each basin in an attempt to attribute marine risk back to individual basins; incorporation of additional pollutants in the assessment of end-of-catchment loads; and revision of the input data for the Marine Risk Index to reduce the uncertainty in the analysis.

Ecological risk is generally defined as the product of the *likelihood* of an effect occurring and the *consequences* if that effect was to occur. However, in this assessment there is some inconsistency in our capacity across the variables to produce a true likelihood or true consequence estimate as mostly we have no or limited ability to produce these estimates right now. Therefore, ecological risk in the GBR is expressed simply as the area of coral reefs and seagrass meadows within a range of assessment classes (very low to very high relative risk) for several water quality variables in river zone of influence in the GBR lagoon. Our method for calculating risk essentially assesses the likelihood of exceedance of a selected threshold. This likelihood was set as 1 for a parameter and location if observations or modelled data indicate that the threshold was exceeded. Conversely, the likelihood was set as 0 if observations or modelled data indicate that the threshold was not exceeded. As consequences are mostly unknown at a regional or species level, potential impact was calculated as the area of coral reef, seagrass meadows and area of GBR lagoon waters (in km^2) within the highest assessment classes of the water quality variables (reflecting the highest severity of influence). The effects of multiplying the habitat area by 1 or 0 for the likelihood mean that the final assessment of risk in this assessment is only an indication of potential impact — the area of coral reef and seagrass meadows in which exceedance of an agreed threshold was modelled or observed. This becomes an assessment of 'relative risk' by comparing the

areas of each habitat affected by the highest assessment classes of the variables among river 'zones of influence' in the Fitzroy region, and was used to generate a 'Marine Risk Index' for coral reefs and seagrass meadows. The zones of influence represent the area covered by the largest river plumes in a wet season.

For assessment of the marine risk, a suite of water quality variables was chosen that represent the pollutants of greatest concern with regards to land-sourced pollutants and potential impacts on coral reef and seagrass ecosystems. These include exceedance of ecologically relevant thresholds for concentrations of total suspended solids (TSS) from remote sensing data, chlorophyll *a* obtained from long-term in situ monitoring data, and the distribution of key pollutants including TSS, dissolved inorganic nitrogen (DIN), particulate nitrogen (PN) and photosystem II-inhibiting herbicides (PSII herbicides) in the marine environment during flood conditions (based on end-of-catchment loads and plume loading estimates). A factor that represents the direct influence of crown-of-thorns starfish (COTS) on coral reefs in the COTS Initiation Zone was included in previous assessments but has been excluded here as it is not considered to be relevant to the Fitzroy NRM region. Modelled end-of-catchment pollutant loads (generated from the Source Catchments Model framework for the Paddock to Reef Program) were obtained for each basin for key pollutants (TSS, DIN, PSII herbicides, PN, Dissolved Inorganic Phosphorus (DIP) and Particulate Phosphorus (PP)), and only the anthropogenic portions of regional total pollutant loads were considered in relating the relative risk to the basins. The anthropogenic load is calculated as the difference between the long-term average annual load, and the estimated pre-European annual load. A factor representing the differential influence of river discharges on the COTS Initiation Zone was also considered in other assessments but not factored in here.

It should be noted that the previous GBR risk assessments also incorporated parameters to represent assessment of the exceedance of ecologically relevant thresholds for concentrations of chlorophyll *a* (Chl-a) and TSS. However, this data was obtained from remote sensing analysis and a recent study undertaken as part of the improvements to the risk assessment method has indicated that there is low confidence in the results in some locations (refer to Maynard et al. 2015 and Petus et al. 2015). More detailed analysis of the relationship between Chl-a satellite results and in situ data in the coastal zone has revealed significant uncertainties in some locations. Until these aspects are resolved further, the Chl-a data has been excluded from this analysis. Instead, results from in situ chlorophyll monitoring have been included as a measure of long-term water quality conditions (sourced from De'ath and Fabricius 2008). The TSS sourced from remote sensing analysis is still included as we did not have the resources to fully investigate the reliability of this dataset as well. However, additional qualitative assessment against the photic depth data will be included for additional interpretation.

The information was then considered by technical experts to make conclusions about the relative risk of degraded water quality to coral reefs and seagrass meadows among the basins in the Fitzroy region. The marine assessment for each basin was constrained to the 'zones of influence' defined for the main rivers in the Fitzroy region. These zones of influence for rivers were defined using outputs from the AIMS hydrodynamic model for the 2010–11 wet season (December to April inclusive). The zones represent the areas of an average distribution of wet season river plumes (described in Waterhouse et al. 2014); however, the entire 'zone' is weighted equally and therefore does not factor in a water quality gradient of distance from the river mouth. This gradient is, however, represented in the actual water quality conditions that comprise the Marine Index described below.

The key results are summarised below.

Marine risk

When all water quality variables are combined into the Marine Risk Index, the water quality influence in the region is generally constrained to the inshore areas, with hotspot areas in Shoalwater Bay and Keppel Bay for sediments, and Keppel Bay for nutrients. However, as noted above, the sediment influence in Shoalwater Bay is not believed to be linked to river discharge (Logan et al. 2014, in press), and the area is naturally turbid due to shallow and large tidal variation. The influence of PSII herbicides does not appear to extend in the marine environment to any significant extent, supported by monitoring data where tebuthiuron was the only pesticide that exceeded the Water Quality Guidelines at a North Keppel Island routine monitoring site in 2012–13 and was below the Guidelines in 2013–14 (Gallen et al. 2014).

The combined assessment of the relative risk of marine water quality variables (Section 3.3) highlights that the areas in the Very High relative risk class were located in Keppel Bay, extending out to the Keppel Island group. Analysis of the zones of influence modelling indicates that the Fitzroy Basin has the greatest influence on this area, appearing to occur annually. This modelling also suggests that Water Park Creek and the Calliope River also influence the Keppel Island Group in larger flow events; however, these rivers only contribute 1–2% of the relative combined anthropogenic loads of the Fitzroy Basin. Nevertheless, when considering combined and cumulative impacts, it is still important to ensure that the water quality from these basins does not decline to exert additional pressures on these receiving environments.

The areas around Port Curtis and extending up to Curtis Island are in the High and Moderate relative risk classes, and in this assessment, these areas were in the receiving areas of the Zones of Influence of the Calliope and Boyne rivers each year. While the influence of these rivers is small in comparison to the Fitzroy River in the context of the whole region, the Calliope and Boyne basins are important to consider in terms of localised impacts on these receiving environments and as above, need to be managed to prevent increasing pressure from these basins in the future.

The proportion of surveyed seagrass area in the Very High and High assessment classes for each variable is greater than 66% and up to 100% for all sediment and nutrient variables. A large proportion of this seagrass is located in Shoalwater Bay. The proportion of deepwater modelled seagrass in the Very High and High assessment classes is less than 5% for all variables. While the areas of coral reef within the highest assessment classes for individual variables and the Marine Risk Index are relatively small, they often include highly valued tourism and recreation sites of the GBR. Examples include the Keppel Island Group and Curtis Island.

Results for important habitat features in the Very High to Low relative risk areas of the Fitzroy region are summarised in Table i. The areas in the Very Low relative areas are not considered here.

Table i. Results of the relative risk assessment for important habitat features in the Very High to Low areas of the Fitzroy region.

Habitat Feature	Description	Relative risk results	Likely rivers of influence
Northumberland Island group (northern inshore areas)	Contains two main island groups, fringing coral reefs and shoals.	Moderate	Fitzroy
Percy Islands	Island group contains fringing coral reefs.	Low	Fitzroy
Broad Sound	Limited coral reefs and seagrass beds and is naturally highly turbid due to large tidal ranges and is relatively shallow.	Moderate to Low	Fitzroy (limited)
Shoalwater Bay	Extensive intertidal seagrass beds, Ramsar wetland, and is protected by the Shoalwater Dugong Protected Area.	Very High in the innermost areas, with a gradient to Very Low risk in the outer part of the bay. However, as described above, the water quality conditions are unlikely to be driven by anthropogenic influences.	Fitzroy (limited)
Keppel Island group	Fringing (inshore) coral reefs, intertidal seagrass beds and island habitats.	High	Fitzroy Water Park (predominantly constrained to North Keppels) Calliope (predominantly constrained to southern-outer areas)
Keppel Bay (coastal areas)	Balaclava Island listed on Register of National Estate, naturally high turbidity with limited coral reefs and seagrass beds but contains important	Very High in the coastal areas, shifting to High and then Moderate in the outer limits of the bay.	Fitzroy

Habitat Feature	Description	Relative risk results	Likely rivers of influence
	coastal wetlands.		
Curtis Island	Fringing coral reefs on south eastern coast, surveyed seagrass at southern end, wetland areas.	Very High and High.	Calliope Boyne Fitzroy (predominantly northern areas only)
Capricornia Cays Group	Mid-shelf coral reefs and deepwater modelled seagrass.	Low for reefs located closest to the coast including Rock Cod Shoal, Irving Reef, Polmaise Reef and Mast Head Island and reefs. Very Low elsewhere.	Fitzroy Potentially Burnett Mary River in flood events eg. 2010–11
Rodds Bay Dugong Protection Area (across the southern boundary)	Extensive intertidal seagrass beds and fringing coral reefs on the eastern coastal of Facing Island.	This area is influenced by Gladstone Harbour and Calliope and Boyne River mouths.	Calliope Boyne Burnett (outside of this region) Fitzroy (limited)

It is important to recognise that the input variables represent longer term time series, and in most cases, represent average conditions. The response of coral reef and seagrass ecosystems to conditions in individual flood events, and the influence of repeated years of flood conditions, is also important. These aspects are discussed further in Section 4.

End-of-catchment loads

An assessment of end-of-catchment loads provides a link between the marine risk and land-based pollutant delivery. The anthropogenic load was incorporated as a proportion of the total regional load, as it is only the anthropogenic portion that is assumed to be the ‘manageable’ component of pollutant loads. In the assessment of end-of-catchment pollutant loads (Section 3.4) the greatest relative contributions of combined end-of-basin loads to the Fitzroy region is dominantly from the Fitzroy Basin, contributing at least 87% of the total regional load for each constituent. Approximately 85% of the Fitzroy Basin is used for grazing (see Section 4.1). Within the Fitzroy Basin, Dougall et al. (2014) identified that the Dawson catchment generates the largest proportion of total sediment to the GBR, followed by the Isaac and Lower Fitzroy. The differences between the Styx, Shoalwater, Water Park, Calliope and Boyne basins are relatively small. However, of these basins, the Styx Basin is the highest contributor to all constituents (80% grazing land use). Grazing is the dominant land use across the region delivering sediment loads to the GBR.

Combined assessment of the relative risk of degraded water quality in the Fitzroy region to guide management priorities

Using the information obtained through the above analyses for the marine water quality variables and end-of-basin pollutant loads, a quantitative combined assessment was completed to inform water quality management priorities among the basins in the Fitzroy region. However, the value of this level of assessment given the dominance of the Fitzroy River and the limitations of the data question the relevance of this additional analysis. Accordingly, this information should only be used to guide management decisions in conjunction with additional qualitative information, some of which is presented in Section 4 of this report.

The results show that the Fitzroy River dominates the greatest risk to each habitat in terms of the potential water quality impact from all of the assessment variables in the Fitzroy region and end-of-catchment anthropogenic loads of TSS, DIN, PSII herbicides, PN, DIP and PP. The Water Park Creek, Boyne and Calliope rivers each pose less than 6% of the relative risk posed by the Fitzroy River. The influence of the Styx and Shoalwater basins cannot be assessed as zones of influence are not available for these basins.

From these findings, it can be concluded that ***the greatest risk posed to coral reefs and seagrass from degraded water quality in the Fitzroy region is from the Fitzroy Basin***. The areas that appear to be exposed to the greatest land-based influence are Keppel Bay and the Keppel Island Group, and Port Curtis. The Water Park Creek, Boyne and Calliope basins each pose less than 6% of the relative risk posed by the Fitzroy River to coral reef and seagrass ecosystems in the region, and specific assessments have not been conducted for Shoalwater or Styx basins. Nevertheless, when considering the combined and cumulative impacts on receiving environments, it is still important to ensure that the water quality from these basins does not decline to exert additional pressures on these receiving environments. There are limited apparent differences between the coastal basins (excluding the Fitzroy), although modelled results for the Styx River indicate a higher sediment load than the other coastal basins.

Given the dominance and large area of the Fitzroy Basin, further prioritisation between sub-catchments within the basin is required. This has been undertaken by Star et al. (2015) where several sub-catchments are identified as the highest priorities for meeting the sediment reduction targets in the region in the most cost-effective manner.

It is recognised that there are many uncertainties associated with the input datasets and method for combining these indexes at a basin-scale at this time (see Section 5); further discussion is recommended prior to making any management decisions based on these results.

Other factors

While this assessment has been limited to the influence of end-of-catchment pollutant loads on coral reefs and seagrass, consideration of other influences including urban and port influences has also been taken into account. With the addition of these influences, direct management of port development areas and associated activities become important in the Calliope Basin. However, it is not within the scope of this study to compare the relative impact of these activities with run-off from catchment land uses.

The high frequency of extreme events in the period 2008 to 2013 has also had a significant impact on the condition and risks posed to ecosystems in the Fitzroy region. Scenarios that assess the possible implications of these events continuing in the future are currently being assessed but were not available at the time of this report (Wenger et al. in prep.).

It should be noted that the confidence in the results at this time is low to moderate due to limitations in some of the input data related to river flows, pollutant loads and water quality concentrations for some variables in the assessment. Accordingly, it is suggested that the results for these basins are likely to be an underestimate of the relative risk of degraded water quality in the region; however, the results do correlate with current status reported in Johnson et al. (2015). This first attempt of assigning relative risk in the marine environment to individual basins by defining zones of influence for individual basins in the region (where possible) demonstrates how this method could be applied for future assessments; however, further refinement of the definition of these zones is recommended if more definitive results are required to differentiate between the basins with greater confidence.

Table ii. Summary of the outcomes of the overall assessment of the relative risk of water quality in the Fitzroy region.

Basin	Basin area (km²)/ % region area	Annual Average River Flow (ML)[1]	Zone of influence (km²)	Marine Risk Index (based on marine assessment only)		Basin Anthropogenic Load as a proportion of the Total Regional Load (%)						Loads Index	Relative Risk Index	Dominant pollutant sources (% land use area)	Overall Rating of Relative Risk
				Coral Reef	Seagrass (survey)	TSS	DIN	PSII Herb	PN	DIP	PP				
Styx	3,013 (2%)	272,000	n/a	n/a	n/a	2.1	0.0	0.03	3.0	**0.0**	3.0	0.03	n/a	Grazing (80%)	**VERY LOW**
Shoalwater	3,601 (2%)	387,000	n/a	n/a	n/a	1.3	0.0	6.23	1.4	0.0	1.4	0.01	n/a	Limited (60% conservation)	**VERY LOW**
Water Park Creek	1,836 (1%)	392,000	2,279	0.09	0.02	0.3	0.0	0.02	0.8	0.0	0.8	0.01	0.05	Limited (63% conservation) Urban (Yeppoon)	**VERY LOW**
Fitzroy	142,552 (93%)	4,650,000	35,409	1.00	1.00	66.7	3.8	93.6	67.9	7.2	67.9	1.00	1.00	Grazing (85%) Dryland cropping (5%)	**VERY HIGH**
Calliope	2,241 1%	117,000	1,802	0.05	0.07	1.4	0.0	0.02	1.9	0.0	1.9	0.02	0.06	Grazing (82%) Port	**VERY LOW**
Boyne	2,496 2%	40,000	1,824	0.03	0.07	0.4	0.0	0.06	0.7	0.0	0.7	0.01	0.05	Grazing (74%)	**VERY LOW**

[1] Dougall et al. (2014).

Shading represents the following relative classes: Red = Very High (0.8-1.0); Dark orange = High (0.6-0.8); Orange = Moderate (0.4-0.6); Yellow = Low (0.2-0.4); No colour = Very Low (0-0.2)

1 Introduction

Exposure to land-sourced pollution has been identified as an important factor in the world-wide decline in coral reef condition (Pandolfi et al. 2003; Burke et al. 2011). Different parts of the Great Barrier Reef World Heritage Area (GBRWHA) are exposed to different degrees of influence from land-sourced pollutants. The degree of exposure is a function of factors such as distance from the coast and river mouths, the magnitude of river discharges, wind and current directions, the mobility of different pollutant types, and the different land uses in the Great Barrier Reef (GBR) catchment (Brodie et al. 2012). This differential exposure to land-sourced pollutants results in varying levels of direct and indirect threats to coastal and marine ecosystems in the GBR including coral reefs and seagrass. Understanding these differences is important for prioritising investment between management areas.

The Fitzroy Natural Resource Management (NRM) region is one of six NRM regions in the GBR catchment (see inset Figure 1.1). The region is part of the Great Barrier Reef World Heritage Area and Great Barrier Reef Marine Park. The NRM region has an approximate catchment area of 156,000 km^2 and is approximately 37% of the total GBR catchment area (423,122 km^2) (Dougall et al. 2014). There are six Australian Water Resources Council Basins that make up the region (ANRA 2002). From north to south they are Styx, Shoalwater, Water Park Creek, Fitzroy, Calliope and Boyne basins (Figure 1.1). The Fitzroy Basin dominates in terms of area (93%), while the smaller basins make up the remainder (7%). Due to the size of the Fitzroy Basin, it is commonly discussed in terms of its catchments, which include the Callide, Comet, Connors, Fitzroy (lower), Lower Dawson, Lower Isaac, McKenzie, Theresa, Upper Dawson and Upper Isaac (Figure 1.1). This scale of information is presented in Star et al. (2015).

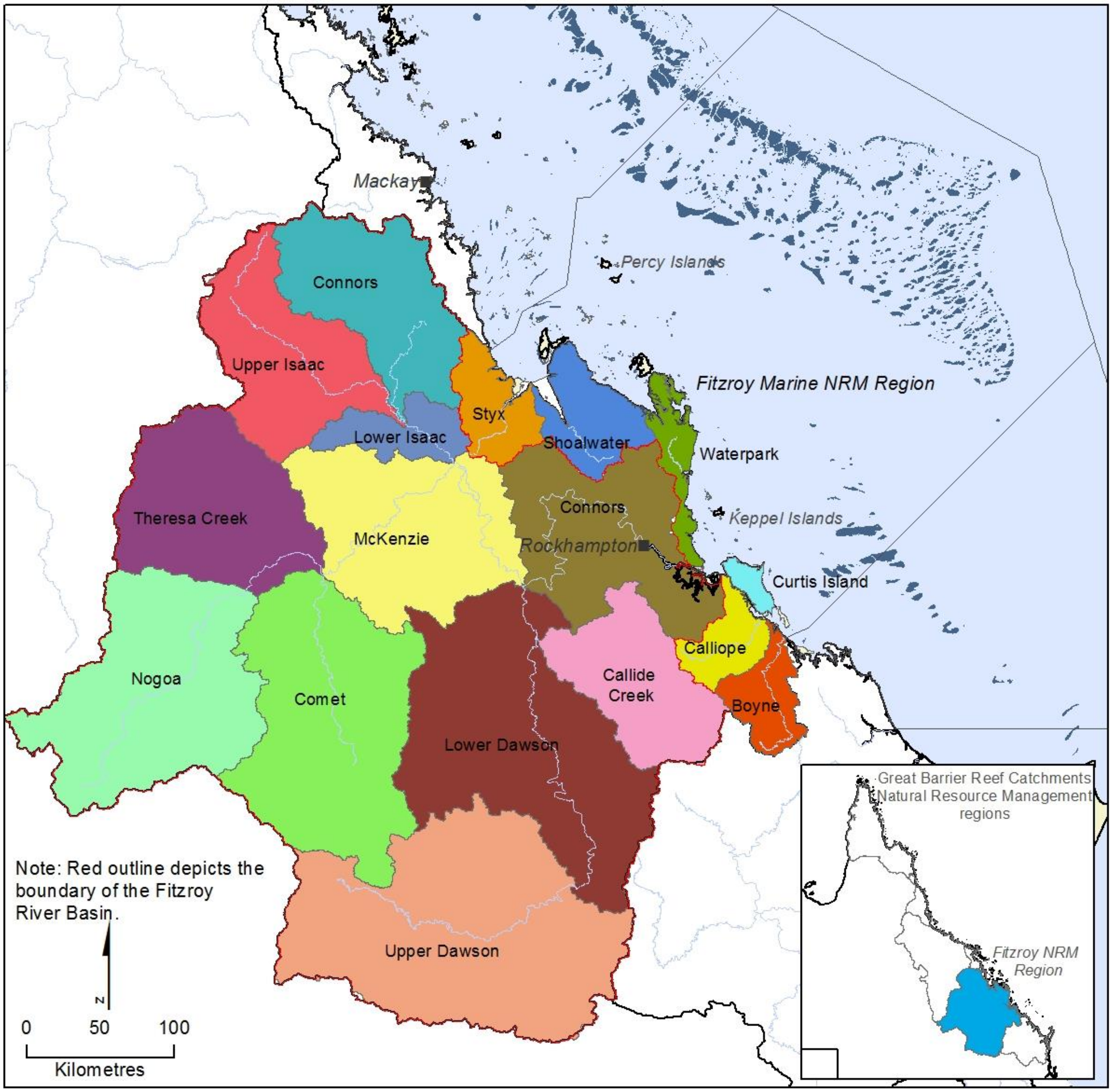

Figure 1.1. Fitzroy Natural Resource Management region, and the major basins.
Inset shows the six GBR NRM regions, and highlights the Fitzroy NRM region.

The Fitzroy region is recognised for its diverse and unique marine and coastal environments including coral reefs, seagrass meadows, tidal wetlands, estuaries, continental islands and the species they support. Some of these species are listed as threatened or vulnerable, and have significant cultural values. Marine and coastal ecosystems also support important tourism and fisheries industries that depend on the healthy natural resources of the region. The region is characterised by subtropical climate with intermittent periods of high river flow driven by the EL Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) (Lough et al. 2015). Current major land uses are grazing (~78%), nature conservation (~8%), forestry (~6%), dryland cropping (~5%) and other land uses including irrigated cropping (including cotton), urban, horticulture and sugarcane all less than 1% of the area (derived from QLUMP 2009). Rockhampton is the largest population centre (~75,000 residents), with other larger centres in Yeppoon and Gladstone. Many smaller towns are located along the coast and throughout the catchments.

Assessment of the current status of key marine and coastal assets in the Fitzroy NRM region has identified a number of assets that are in poor or very poor condition. These include inshore coral reefs, inshore and reef seagrass meadows, dugong, turtles, dolphins, low-lying islands, and species of climate-sensitive seabirds (Johnson et al. 2015).

The Fitzroy NRM region has been identified as a high risk region in terms of the influence of degraded water quality on GBR ecosystems (Brodie et al. 2013a, 2013b; Waterhouse et al. 2012). In the most recent relative risk assessment of degraded water quality on the GBR (Brodie et al. 2013a), the Fitzroy region was ranked as the second highest risk NRM region (equivalent to the Burdekin NRM region) compared to other regions for overall water quality risk to coral reefs and seagrass. These rankings were largely associated with the high loads of sediments that are delivered to the GBR from the catchments in the region.

Even though the nutrient-related variables of chlorophyll threshold exceedance and DIN plume loading were ranked highest in the Fitzroy region, there is insufficient knowledge of the sources of dissolved inorganic nitrogen in the region to make recommendations about management priorities for these. This is discussed further in the limitation of this report.

Previous assessments of the relative risk of degraded water quality on GBR ecosystems have largely been undertaken at a GBR-wide scale, with the assessment of relative risk between NRM regions (Brodie et al. 2013a; Waterhouse et al. 2012; Brodie & Waterhouse 2009; Cotsell et al. 2009; Greiner et al. 2005). The results of these assessments have been used to inform prioritisation across the NRM regions in terms of management effort (such as Reef Plan 2009 and 2013, the *Queensland Great Barrier Reef Protection Amendment Act 2009*) or investment including the Reef Rescue and Reef Programme initiatives. There have been two recent regionally specific assessments of relative risk from degraded water quality in the Fitzroy region:

1. Brodie et al. (2009) conducted an assessment that focused on pollutant loads at a basin scale, and identified the highest total loads and generation rates of anthropogenic suspended sediment, dissolved inorganic and PSII herbicide loads in each of the Wet Tropics, Fitzroy, Mackay-Whitsunday and Fitzroy NRM regions. The assessment was used to help direct

management activities to basins and pollutants of most concern under the Queensland Government Reef Protection Package in 2009; and
2. Australian Government (2014) (led by M. Barson) completed an assessment of pollutant loads from catchments flowing to the GBR lagoon and the impact of transported materials (sediments, nutrients and herbicides) on coral reefs and seagrasses, to support and inform discussion and decisions on funding priorities for investment, particularly through Reef Water Quality Grants in 2013 (part of the Australian Government Reef Programme).

Several improvements in catchment modelling (see Dougall et al. 2014 for the most recently published data) and availability of longer time series of monitoring data to support this modelling effort has resulted in greater confidence in the input data required for a regionally based water quality risk assessment. The capability to assess the relative risk of different pollutants and basins to marine ecosystems has also progressed (Brodie et al. 2013a). In the same period, the Australian Government has supported the revision and development of regionally based Water Quality Improvement Plans (WQIPs) in all of the GBR NRM regions. Continued investment towards a water quality grant program for the region has also occurred through the Australian Government Reef Water Quality Programme (formerly Reef Rescue). These initiatives have driven the need to undertake an updated relative risk assessment of water quality issues in the Fitzroy region.

This report presents the results of an updated assessment of the relative risk of the influence of sediments, nutrients and PSII herbicides on key GBR ecosystems in the Fitzroy region. The assessment considers the most relevant pollutants for GBR water quality in the GBR, i.e. sediments, nutrients and PSII herbicides — and is based on a methodology developed for the relative risk assessment undertaken for the whole GBR in 2013 (see Brodie et al. 2013a) that has been modified for regional application (Waterhouse et al. 2014a, 2014b). The full report prepared by Brodie et al. (2013a) can be downloaded for a full explanation of the assessment techniques used in that assessment.[1]

As an important note, this report refers to suspended (fine) sediments and nutrients (nitrogen, phosphorus) as 'pollutants'. Within this report, we explicitly mean enhanced concentrations of or exposures to these pollutants, which are derived from (directly or indirectly) human activities in the GBR ecosystem or adjoining systems (e.g. river catchments). Suspended sediments and nutrients naturally occur in the environment; indeed, all living things in ecosystems of the GBR require nutrients, and many have evolved to live in or on sediment. The natural concentrations of these materials in GBR waters and inflowing rivers can vary, at least episodically, over considerable ranges. Pesticides do not naturally occur in the environment. Pollution occurs when human activities raise ambient levels of these materials (time averages, or event-related) to concentrations that cause environmental harm and changes to the physical structure, biological communities and biological functions of the ecosystem.

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1 http://research.jcu.edu.au/research/tropwater/publications/copy4_of_1328Assessmentoftherelativeriskofdegradedwaterqualitytoecosystemsof theGreatBarrierReef.pdf

2 Methods

2.1 The water quality risk assessment framework

Ecological Risk Assessment (ERA) is a term used for a variety of methods to determine the risk posed by a stressor, for example a pollutant, to the health of an ecosystem. 'Risk' is usually defined as the probability that an adverse effect will occur as a result of ecosystem exposure to a certain concentration of the stressor. Risk is often quantified as the product of the *likelihood* of an event occurring (exposure) and the *consequences* (also measured as effects) of that event. Risk assessments are used as decision tools that rank risks to human values in order to prioritise management actions and investments (e.g. Burgman 2005; AS/NZS 2004). A number of methodologies are available to carry out the analysis with Bayesian techniques now often favoured by decision makers (e.g. Hart et al. 2005; Hart & Pollino 2008). Due to limitations in data availability and limitations with time and resources, a relatively simple methodology suitable for the existing datasets, resources and timeframes has been developed based on a modification of the typical ERA framework.

Ecological risk is assessed here using a relatively simple approach, following that developed for the GBR-wide relative risk assessment in 2013 (Brodie et al. 2013a). The *likelihood of exposure* of a species or habitat to an impact is typically a function of the intensity of the impact (the concentration or load of a pollutant) and the length of time it is exposed to the impact. For example, a seagrass meadow may be exposed to a high intensity impact for a short period of time (acute), or to lower intensities for longer periods (chronic). When quantifying exposure, it is important to determine the threshold concentrations that lead to an effect on species or habitats; that is, the concentration that potentially leads to damage or mortality within hours or days, as well as understanding long-term average concentrations and the duration of exposure. This complicates the description of exposure thresholds given their values may change by one to two orders of magnitude between days, seasons and years. Hence, some key water quality variables such as suspended sediments are divided into different thresholds based on ecological responses and periods of exposure. To reflect this, each threshold is classified into several assessment classes to represent the potential differences between the duration and severity of the influence (from lowest to highest).

The *consequences* are the measured effects of the water quality exposure. Current knowledge of the effects of degraded water quality on the health of the GBR are summarised in the 2013 Scientific Consensus Statement (Brodie et al. 2013b). The GBR Water Quality Guidelines reflect our knowledge of ecological thresholds for water quality variables for coral reefs in the GBR (GBRMPA 2010). However, only limited information is available to draw conclusions on the effects of the exposure of sediments, nutrients and PSII herbicides on seagrass health. Evidence shows that one of the greatest drivers of seagrass health is the availability of light, which is reduced by increased suspended sediment and the secondary effects of increased nutrients such as increased growth of epiphytes and phytoplankton

(Collier et al. 2012). However, in the absence of more regionally and species-specific knowledge of pollutant impacts on seagrass, the same threshold concentrations have been used for coral reefs and seagrass meadows in this assessment. It is also recognised that the consequence of the exposure of species or habitats to a range of water quality conditions is complicated by the influence of multiple pressures, and many external influences including weather conditions; however, it is difficult to factor these into the risk assessment in any quantitative way.

Given the above and recognising the inconsistencies in the spatial and temporal availability of the water quality data in the GBR, our capacity to produce a true likelihood or true consequence estimate for this assessment is limited. It was therefore necessary to develop an effective, simple and standard methodology for the risk assessment that could be implemented with the available data, in a way that could be easily communicated and discussed with decision-makers and stakeholders. For this reason, in this study ecological risk in the GBR is expressed simply as the area of coral reefs and seagrass meadows within a range of assessment classes (very low to very high relative risk) for several water quality variables in each NRM region in the GBR catchment. Our method for calculating risk essentially assesses the likelihood of exceedance of a selected threshold. This likelihood was set as 1 for a parameter and location if observations or modelled data indicate that the threshold was exceeded. Conversely, the likelihood was set as 0 if observations or modelled data indicate that the threshold was not exceeded. As consequences are mostly unknown at a regional or species level, potential impact was calculated as the area of coral reef, seagrass meadows and area of GBR lagoon waters (in km^2) within the highest assessment classes of the water quality variables (reflecting the highest severity of influence). The effects of multiplying the habitat area by 1 or 0 for the likelihood mean that the *final assessment of risk in this assessment is only an indication of potential impact* — the area of coral reef and seagrass meadows in which exceedance of an agreed threshold was modelled or observed. This becomes an assessment of 'relative risk' by comparing the areas of each habitat affected by the highest assessment classes of the variables among NRM regions, and was used to generate a 'Marine Risk Index'.

In the GBR-wide study conducted in 2013 (Brodie et al. 2013a; referred to herein as the 2013 risk assessment) the relative risk of degraded water quality to coral reefs and seagrass was assessed by combining information on end-of-catchment pollutant loads of sediments, nutrients and PSII herbicides with the estimated ecological risk of water quality to coral reefs and seagrass meadows for the GBR. Three primary indexes were developed in the original method (see Figure 2.1): 1) a Marine Risk Index that represents an estimate of ecological risk of water quality to coral reefs and seagrass; 2) a Loads Index that represents the contribution of pollutant loads from each basin; and 3) a crown-of-thorns starfish (COTS) Influence Index that represented the regional contribution of observed freshwater discharge (as a proxy for nutrient delivery) to the area where primary outbreaks of COTS are known to occur. The three indexes were combined to generate a Relative Risk Index for coral reefs and seagrass meadows for each NRM region. This index ultimately ranked the relative risk of degraded water quality to coral reefs and seagrass in the GBR among NRM regions. According to the modelled data (Brinkman et al. 2014), the COTS Influence Index is not relevant in this region and has therefore been excluded from the assessment.

To conduct a comparable assessment that just focused on the Fitzroy region, separate areas of influence for rivers discharging into the marine environment were estimated. As described in Section 2.3, eReefs hydrodynamic modelling, river flow, plume direction and imagery was used to derive the zone of influence for the Fitzroy River. The Water Park, Calliope and Boyne rivers are not included in the hydrodynamic model, and the zones of influence were estimated using a path-distance modelling approach. This enables the estimated relative risk in the marine environment to be attributed to each basin. Insufficient data is available to estimate zones of influence for rivers in the Shoalwater and Styx basins.

The main elements of the framework are shown in Figure 2.1. The combined index ultimately ranks the relative risk of degraded water quality to coral reefs and seagrass among Fitzroy basins where possible. It is important to note that the variables selected in the marine assessment represent average conditions over several years (it varies between datasets), and therefore additional discussion is included about the influence of years with high river flow and flooding in Section 4.

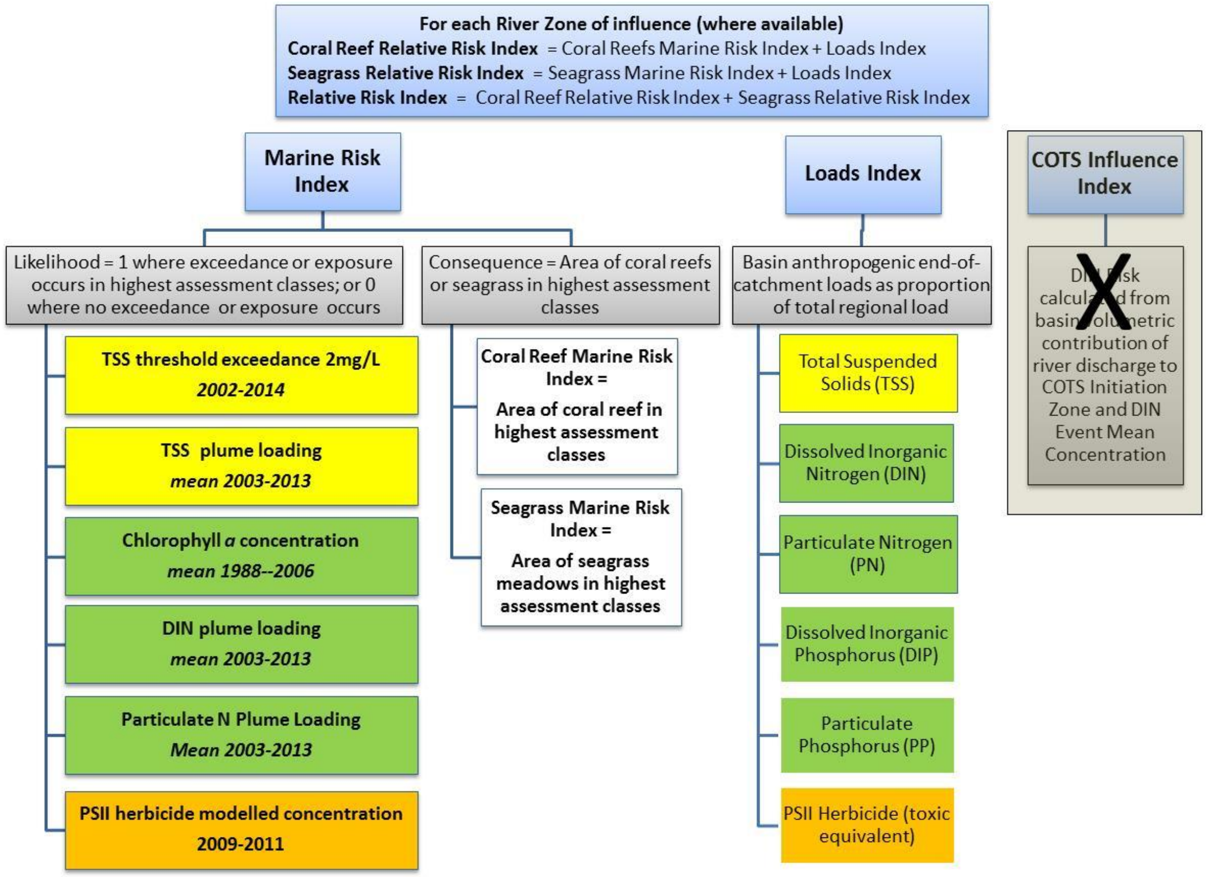


Figure 2.1. Risk assessment framework.

Figure 2.1 shows the components of the Marine Risk Index to represent marine water quality ecological risk to coral reefs and seagrass meadows, and a Loads Index to represent catchment influences on GBR water quality using end-of-catchment anthropogenic pollutant loads. The COTS Influence Index to factor in the importance of river discharges on the COTS Initiation Zone for coral reefs used in previous assessments was not considered relevant for this assessment. The colours represent groups of variables: yellow = sediment related variables, green = nutrient related variables and orange = PSII herbicide related variables.Habitats and boundaries

2.1.1 The Fitzroy marine NRM region

The marine NRM region (as defined by GBRMPA; see Figure 2.2) extends seawards from the northern and southern boundaries of the NRM region, to the outer edge of the Great Barrier Reef Marine Park, and has an area of approximately 85,500 km^2. However, this is an administrative boundary and does not necessarily reflect the extent of influence of the catchments on the marine environment in the region. Furthermore, catchments outside the Fitzroy NRM region may influence the marine ecosystems within the region (such as large flows from the Mary River to the south), and the Fitzroy River is known to influence the adjacent Mackay-Whitsunday NRM region, as illustrated in the example shown in Figure 2.3. In this study, reference to Keppel Bay includes the embayment between Emu Point (at Emu Park) and Cape Keppel at the northern end of Curtis Island.

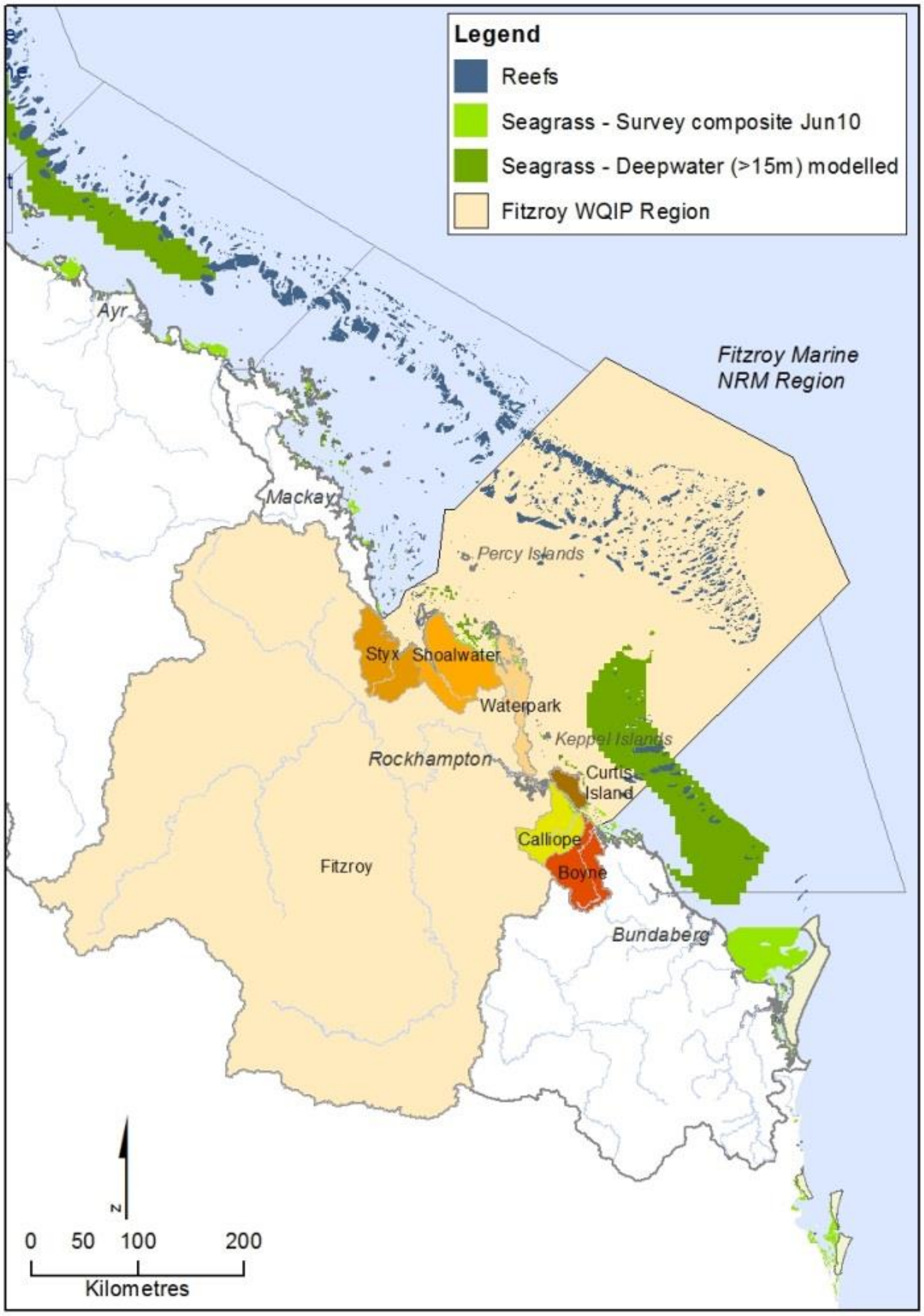
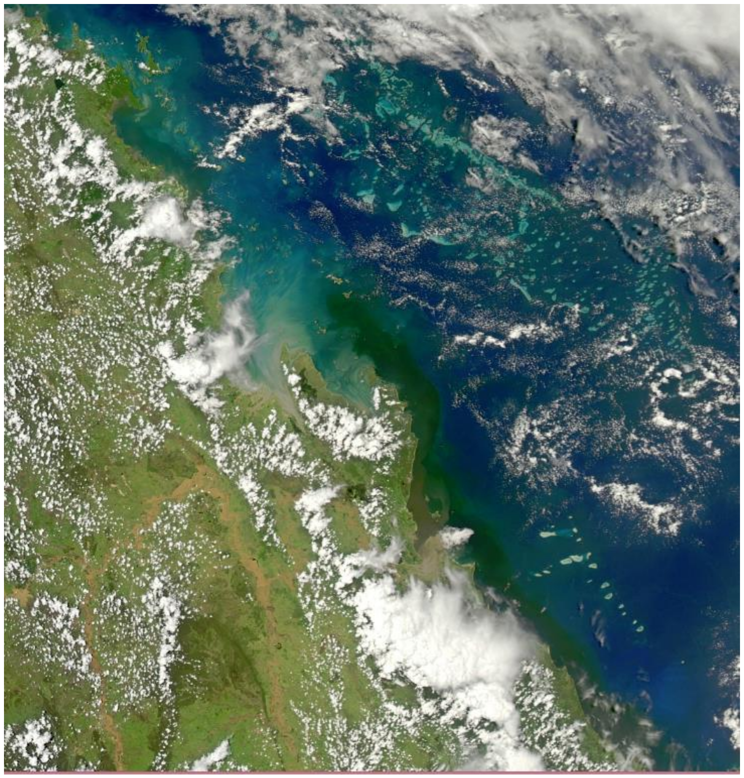


Figure 2.2. Assessment boundaries considered in this risk assessment
Includes the Fitzroy NRM region basins and marine NRM region.



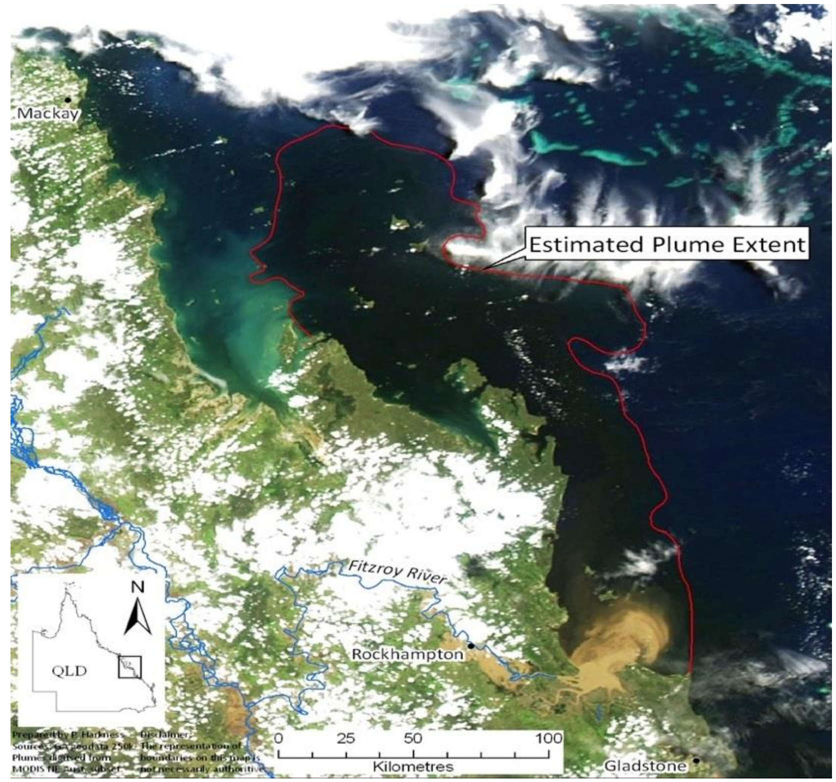


Figure 2.3. MODIS-Aqua image from (left) 4 January 2011 and (right) 11 January 2011 showing the extent of river plume influence from the Fitzroy region

Provided by TropWATER. These images show the contrast between the phytoplankton dominated plume (green — from the river) adjacent to sediment plumes in Shoalwater Bay generated through (tidal) re-suspension.

2.1.2 Defining basin 'zones of influence'

Fitzroy River

Zones of influence for rivers in the Fitzroy region were defined using a combination of river flow data, in situ salinity data, and output from a highly resolved hydrodynamic model (eReefs) for the 2008–09, 2010–11, 2011–12 and 2012–13 wet seasons (December to April inclusive) (note that the hydrodynamic model is not available for 2009–2010). The Fitzroy River is the only river that is modelled in the region. The area of river plume influence (tracer plume) was mapped using virtual tracers that are released from the river mouth, proportionally to the river discharge. Because tracers present conservative behavior, a tracer concentration threshold equivalent to salinity 36 ppt was used to define the edge of the river plume. The delimitation of the area of river plume influence has been developed by Nicholas Wolff (Wolff et al. 2014). The river basin zone of influence was defined as the area where over the wet season (c.a. from December to April, inclusive) tracer concentration was equivalent to salinity 36 ppt at least 5% of the time each year, combined over the four years. Tracer plumes were converted to smoothed shapefiles in a five step process: 1) interpolating into coastal areas; 2) resampling to the same grid used for all Risk Index layers (from 0.036 to 0.01 decimal degrees, using bilinear interpolation); 3) converting from raster to polygon; 4) applying a PAEK smoothing algorithm; and 5) erasing mainland and island areas.

Zones of influence for unmodelled rivers

River plumes for each of the unmodelled rivers can only be generated where flow data is available, which includes the Water Park Creek, Calliope and Boyne rivers. For these unmodelled rivers, zones of influence were derived using the ArcGIS path-distance tool, with plume extent constrained to a maximum distance from the river mouth predicted from river discharge. Zones of influence for the modelled rivers (tracer plumes) were used to derive this flow-distance relationship (Figure 2.4). Total wet season discharges for all rivers were obtained from the Department of Natural Resources and Mines[2], and they were calculated as the sum of the daily discharges for the period between the 1st of December to the 30th of April for each wet season in the years of interest.

The path-distance tool determines the minimum accumulative travel cost from a source to each cell location in a raster (ArcMap Spatial Analyst, ESRI 2010). Path-distance takes into account horizontal and vertical factors that affect this movement, making it useful in dispersion modelling, flow movement, and least-cost path analysis (ESRI 2010). Three main inputs were used in the path-distance tool to model individual plumes: (i) SOURCE — the point coordinates of the freshwater discharge; (ii) COST RASTER – a surface raster indicating the impedance for the plume movement, and (iii) HORIZONTAL RASTER – a surface raster indicating the main direction of plume propagation. To have a greater control on the modelled plume, the linear type horizontal factor was included to the path-distance function. This

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[2]Queensland, https://www.dnrm.qld.gov.au/water/water-monitoring-and-data/portal Accessed August 2015

horizontal factor was parameterised, and the cost raster selected based on the best match up (percent overlap) between tracer plumes and path-distance plumes when path-distance plumes were constrained to the same maximum linear extent as the tracer plumes. The cost raster surface used was the inverted frequency of occurrence of plumes during the wet season (i.e. the accumulative cost of travel was inversely proportional to the frequency of observed river plume). The main direction of plume propagation was set as 315º Azimuth to account for the prevailing wind (i.e. trade winds) and current direction in the wet season (Luick et al. 2007).

River plumes were generated using the path-distance tool for the same four years as the tracer plumes (2008–09, 2010–11, 2011–12, 2012–13). Plume extent was constrained using the predicted maximum distance between river mouth and the outer edge of plume (Figure 2.4). Plume areas for each year were converted to shapefiles and, as with the tracer plumes, zone of influence has been defined using the combined area of the four years.

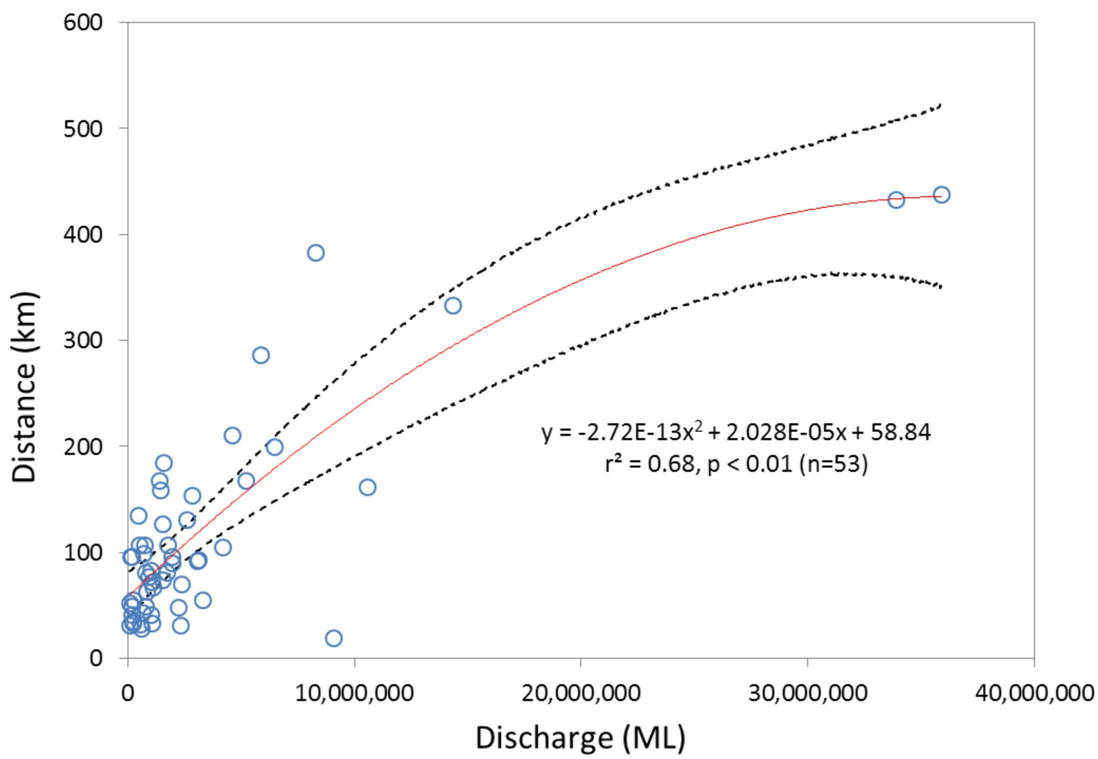

Figure 2.4. Relationship between river discharge and the distance between river mouth and the outer edge of tracer plume.

2.1.3 Habitat mapping

The habitats considered in the assessment were coral reefs and seagrass meadows, based on the best available information (Figure 2.2). For coral reefs, the spatial layer used is the GBRMPA Spatial Data Centre's coral reefs spatial data file (December 2012).

The seagrass habitat map used (supplied by TropWATER James Cook University 2013) is comprised of a composite of the survey data up to 2010 (observed habitat) and a statistical model of seagrass present in GBRWHA waters >15 metres in depth. In this model spatial distribution is a statistically modelled probability of seagrass presence (using generalised additive models with binomial error and smoothed terms in relative distance across and along the GBR), based on ground-truthed points (Coles et al. 2009). Locations with seagrass habitat probability >50% were included in the assessment.

The calculation of the areas of habitat are based on a pixel-based assessment and therefore vary slightly from official published figures (which were used in Brodie et al. 2013a).

2.2 Selecting and classifying variables for estimating ecological risk

A suite of water quality variables were selected to represent the pollutants of greatest concern with regards to land-sourced pollutants and potential impacts on GBR ecosystems. These are summarised in Table 2.1, and in the marine assessment include ecologically relevant thresholds for concentrations of Chl-a from in situ monitoring data, and the distribution of key pollutants including total suspended sediments (TSS), dissolved inorganic nitrogen (DIN), particulate nitrogen (PN) and photosystem II-inhibiting herbicides (PSII herbicides) in the marine environment during flood conditions (based on end-of-catchment loads and plume loading estimates). For each variable, thresholds above which impacts have been observed or predicted were defined and classified into three to five classes (from lowest to highest). We have used average conditions over several years (time period varies between datasets) for most variables, and the influence of years with high river flow and flooding is discussed as supporting information in Section 4.

More detailed information on pollutant impacts on GBR ecosystems is provided in the 2013 Scientific Consensus Statement *Chapter 1 Marine and coastal ecosystem impacts from degraded water quality* (Schaffelke et al. 2013). The selected variables and thresholds represent long-term conditions (chronic exposure) and wet season pollutant loadings in flood plumes (acute exposure).

2.2.1 Annual TSS and Chlorophyll

The 2013 risk assessment used long-term, remotely sensed data to define the areas where the Chl-a and TSS concentrations exceeded different ecologically relevant threshold values at different frequency intervals. The frequency of exceedance of the GBR Water Quality Guidelines was used for Chl-a

(0.45μg/L) and TSS (2 mg/L) concentrations, and an additional higher threshold was also applied to TSS concentrations (6.6mg/L or 5NTU) to factor in more severe effects on coral reefs, seagrass and fish. However, regionally specific application of this data in the Wet Tropics and Burnett-Mary regions (Waterhouse et al. 2014a, 2014b) highlighted a number of limitations with the data relevant to this application, explained below.

The data used in the 2013 water quality risk assessment were provided by CSIRO in November 2012. In the 18 months that followed, the data were re-processed and set to a different, higher-resolution grid. This is the ocean colour data that is now publically available for download through the eReefs Water Quality Dashboard[3], and used for the Burnett-Mary WQIP (Waterhouse et al. 2014b). In both cases, the thresholds used for relative risk were based on number of days exceedance (unit=days during ~10-year time series) of the GBR Water Quality Guidelines (GBRMPA 2010) for Chl-a (0.45μg/L) and TSS (2 mg/L), plus an additional exceedance for TSS of 6.6mg/L (or 5 NTU) to represent a concentration where greater severity of potential impacts is known to occur. For this assessment, remotely sensed Chl-a and TSS data were acquired from the eReefs Marine Water Quality dashboard for the period 01 November 2002 to 31 October 2014.

It is possible to record the number of valid observations in each pixel (1 km^2 grid) over any time period (see Figure 2.4). The number of valid observations is a result of the strict quality control criteria applied to the imagery: pixels with cloud or cloud shadow, low view and illumination angles (solar zenith and observer zenith higher than 60 degrees) are flagged and dismissed, as are pixels where the atmospheric correction failed. In the 2014 analysis, during the 12 years between 2001 and 2013 the maximum number of valid observations was 1682, which means at best, valid observations are made 38% of the time. For large areas of the GBRWHA, particularly in the wet season, valid observations are made less than 20% of the time. In the Fitzroy region, there is an area of low confidence in Shoalwater Bay, where there is less than 20% valid observations, and in the wet season, the proportion of valid observations is typically below 25%. For this reason, using absolute thresholds for number of days exceedances or even average concentrations can be very misleading as the frequency of valid observations varies greatly in space and time. There are many pixels where there are a low number of valid observations, particularly during the wet season where there is significant cloud cover; some of these pixels may have been given a lower risk classification than would be the case if exceedances were expressed as percentages of valid observations. Therefore for this risk assessment, we have calculated the percentage of the valid observations that exceed the water quality thresholds.

[3] http://www.bom.gov.au/marinewaterquality/ Accessed August 2015

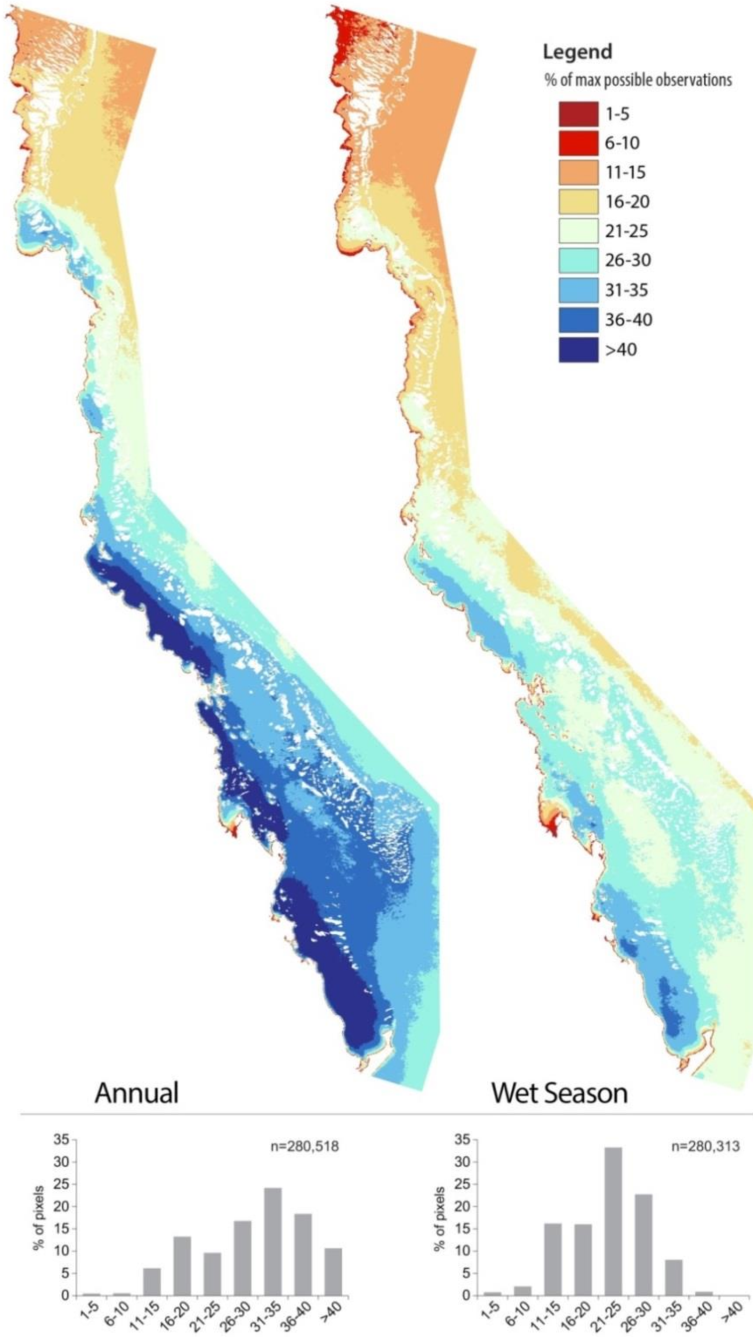

Figure 2.5. Percent of maximum possible observations during annual and wet season periods 2002–2014 that were valid out of a total maximum for Annual of 4383 and for Wet of 2175.

Maximum possible observations are calculated as follows: Annual – November 2002 to October 2014 is 12*365+3 leap days = 4383 days; Wet – November 2002 to October 2014 is 12*181+3 leap days = 2175 days.

The results for the original and revised methods are available for comparison in Maynard et al. (2015). Importantly, the results vary between the methods, particularly for Chl-a. In summary (across the GBR):

- For **Chl-a** between 40 and 70% of the pixels stayed within the same risk classification/category. However, between 10 and 50% of the pixels increased 1 or 2 classes, which is particularly relevant in the coastal areas. The area of each marine-NRM previously considered to be in the highest risk class for exceedance of Chl-a 0.45 μg/L is roughly half what these current results are suggesting.
- For **TSS 2mg/L** declines in the risk classifications were far more common than increases.
- **TSS 7mg/L** results are largely unchanged using this new method.

It is important to recognise that there are also other, potentially significant factors that influence the confidence of the use of these remotely sensed datasets in the assessment, including the reliability of remote sensing data in highly turbid waters. Petus et al. (2015) conducted a preliminary study of the latter aspect, which has highlighted that there is limited confidence in the remote sensing data under certain conditions in the GBR — particularly in shallow and turbid waters. The study concluded that:

- Assessing Chl-a concentrations with remotely sensed data is notoriously challenging in optically complex (case II) coastal waters such as the GBR lagoon and limitations of the remote sensing data must be understood in order to efficiently use these data as a monitoring tool.
- The analyses showed that the satellite Chl-a values were significantly higher than the in situ Marine Monitoring Program (MMP) measurements (wet season samples collected in plume waters).
- There was a strong variability at the regional NRM scale, and satellite Chl-a values were significantly higher than the in situ measurements in all regions, except the Mackay-Whitsunday region (but results were insignificant in this region).
- The maximum retrieval errors were calculated in the Cape York region (mean Error 506 ± 651%). The minimum retrieval errors were calculated in Mackay-Whitsunday NRM (mean Error 56 ± 61%) though the results were not significant. Retrieval errors in the Fitzroy were 401 ± 872%, which is also relatively high compared to other regions (Wet Tropics were 128 ± 274%; Burdekin were 108 ± 148%; not tested the Burnett-Mary region). It must be underlined that <u>the errors and bias reported in this study are performance statistics for the wet seasons and for flood plume waters only</u>. Validation of the remote sensing Chl-a retrievals based on observations performed mainly during the dry season have been presented in King et al. (2014) with stronger validation statistics i.e. Error = 89%.
- There was also a strong variability at the cross-shelf scale and the satellite Chl-a values were significantly higher than the in situ measurements in all cross-shelf regions, except the offshore region.

- A trend toward an increase of uncertainties in the satellite Chl-a concentration was observed when the TSS concentration increases and the bottom depth decreases; with thresholds values estimated around Non-Algal Particulates (proxy for TSS) 2 mg L^{-1} and depth less than 25 metres. Based on this assessment, Figure 2.5 shows a preliminary indication of a Chl-a satellite confidence map based on the 25 metre bathymetry contour. In the Fitzroy region, this is almost the entire area defined as the 'inshore' zone (as defined in the GBR Water Quality Guidelines), and also extending into the mid-shelf areas in the northern and southern sections of the marine NRM region.
- Overall at the GBR-scale, and at individual regional scales, correlation between Chl-a (satellite) and Chl-a (in situ measurements), using the data analysed here, was poor with $R^2 < 0.23$ in all cases except on Cape York (see above). However, for the Cape York results, although correlation was good (R^2=0.78) bias was high, giving a constant large over-estimation of the Chl-a concentrations.

Figure 2.6. Preliminary indication of a Chl-a satellite confidence map based on the 25 metre bathymetry contour.

The areas shown in orange are less than 25m depth and therefore, indicate 'High' uncertainty.

The full results of both of these studies are summarised in a separate report (collated by Waterhouse and Brodie 2015) and fully described in Maynard et al. (2015) and Petus et al. (2015).

As a result of the outcomes of these studies, we have concluded not to use the remote sensing datasets for Chl-a in this assessment, but rather, a long-term chlorophyll monitoring dataset (see below). However, based on current information we have decided to continue to use the TSS 2mg/L exceedance assessment as insufficient analysis of the confidence in this dataset has been undertaken to conclude otherwise. Turbidity is highly correlated with depth due to re-suspension of fine sediment due to currents and wind events, so the pattern is expected to be influenced by bathymetry anyway. We considered using photic depth as an alternative measure of sediment influence in the marine environment, based on the analysis conducted by Fabricius et al. (2014) and Logan et al. (2014, in press); however, possible limitations of these techniques in shallow waters also require further investigation. In addition, there are potential limits to the application of the GBR Water Quality Guidelines for Secchi depth (which can be applied to photic depth in shallow waters).

Chl-a method and assessment classes

The dataset used to represent Chl-a is surface water quality data collected by AIMS, GBRMPA and co-workers between 1988 and 2006. Niskin or bucket surface water samples and laboratory analyses of a suite of physical and chemical water quality data, including Chl-a, were undertaken typically at monthly intervals (Brodie et al. 2007; Furnas et al. 2005). The values used in this assessment are a representation of the mean concentration, extrapolated in De'ath and Fabricius (2008) to generate a map of long-term Chl-a conditions, based on the locations shown in Figure 2.6. The data was then classified against the GBR Water Quality Guidelines to provide a relate risk classification, shown in Table 2.4. There are several limitations in using this dataset including the variability in the collection of data across the region (leading to lower confidence in some locations; Figure 2.7), the collection period is now dated (1988 to 2006) and it does not provide any indication of the severity of the potential impact, i.e. the frequency of occurrence or exceedance of the thresholds. However, given the limitations with the remote sensing data in inshore areas, it is considered to provide the best available indicator of year-round nutrient conditions in the GBR for this assessment at this time (noting also time and resource limitations).

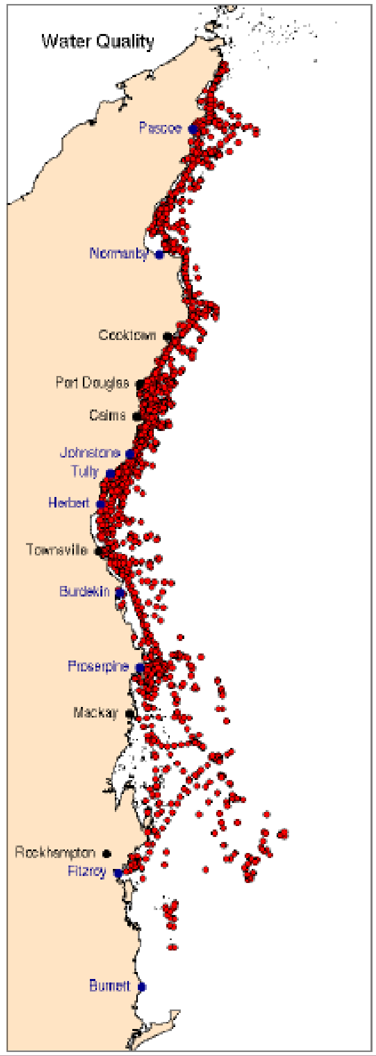

Figure 2.7. Location of the 4067 lagoon water quality stations including in the long-term chlorophyll monitoring program between 1988 and 2006.

Note that there is relatively low density of sampling south of Mackay.

Table 2.1. Long-term Chl-a mean concentration risk classifications.

Mean value µg/L	Score	Reason
<0.18	Very Low: 0	Limit of analysis and considered to be insignificant in terms of potential ecological impacts for coral reefs and seagrass
0.18-0.25	Low: 0.25	Preliminary threshold for seagrass ecological impacts
0.25-0.45	Medium: 0.5	GBR Water Quality Guideline
0.45-0.8	High: 0.75	Increased COTS larval survival
>0.8	Very High: 1	More severe ecosystem impacts and where 100% survival of COTS larvae is likely

TSS method and assessment classes

Maynard et al. (2015) calculated the percentage of the valid observations that exceed the water quality thresholds. Remotely sensed TSS data were acquired from the eReefs Marine Water Quality dashboard for the period 01 November 2002 to 31 October 2014. Exposure thresholds were based on the GBR Water Quality Guidelines for TSS: 2 mg/L. Analyses were undertaken for two time periods for each year: the wet season (01 November to 30 April) and annually (01 November to 31 October). Each time period was identified by the year in which it ended (i.e. April or October). The distribution in the percentage of exceedance data was used to define relative risk classifications from very low to very high, based on the annual aggregated results. These risk classifications were then applied to the percentage of exceedances from each year (as per the 2013 risk assessment), shown in Table 2.2.

Table 2.2. Classifications applied to the remote sensing datasets for the percentage of valid observations exceeding the thresholds.

Classification	Percentage of valid observations exceeding the threshold: TSS 2mg/L
Very Low	0
Low	1-10
Medium	11-20
High	21-50
Very High	51-100

Assessment of the use of photic depth data as an indicator of turbidity (acquired using remote sensing algorithms, Weeks et al. 2012) was also considered in this study. Photic depth can be used as an indicator of Secchi depth, and while the GBR Water Quality Guidelines contain values for Secchi depth (>10 metres except in enclosed coastal waters that are >1.5 metres), the guidelines require further definition in inshore shallow waters where the depth is <10 metres and ecosystems are conditioned to survive in more turbid waters (those with lower Secchi depth). It was therefore concluded that inclusion of the data as an indicator of TSS would be premature for this assessment.

2.2.2 TSS, DIN and PN loading

Ecological impacts of terrestrial run-off on coral reefs and seagrass meadows can be experienced as either acute, short-term changes associated with formation of high-nutrient, high-sediment, low salinity flood plumes or the more chronic impacts associated with long-term changes in water quality (Devlin et al. 2012). The ecological impact of terrestrial contaminants varies not only with the type of pollutant, the magnitude and extent of the riverine influence but also with the ecosystems being affected and the frequency and duration of plume occurrence (for example, see Devlin et al. 2012). River plume models can help to develop risk maps by defining areas that may experience acute or chronic high exposure to pollutants or stressors (Alvarez-Romero et al. 2013). Details of the pollutant movement and frequency of inundation can be key measurements in attributing water quality decline to ecosystem change. These contribute to the 'likelihood' component of the risk equation.

Plume loading maps have been updated since the 2013 risk assessment. These maps are available annually but we have selected a mean assessment across the full period of availability (2003 to 2013) to represent average conditions; however, the influences of large river discharge and flood events are discussed in Section 4.

TSS

The effects of elevated concentrations of suspended solids on GBR ecosystems including coral, seagrass and algal communities were reviewed in Brodie et al. (2013c). The greatest influence of increased turbidity caused by re-suspension of sediment in waters of depths less than 12 metres in depth is reduced light for benthic phototropic communities including coral reefs and seagrass (Larcombe et al. 1995; Anthony et al. 2004; Orpin et al. 2004; Alongi & McKinnon 2005). This re-suspension-driven turbidity persists for many months of the year in GBR coastal waters. Suspended solids in flood plumes also reduce light for benthic communities but the effects are only present for short periods, typically days to weeks. Hence, a long-term time series is most relevant in the assessment of chronic effects of elevated suspended solids and turbidity on habitats. However, typically the re-suspended sediment is that which was delivered as a sediment loading during the previous wet season and potentially earlier wet seasons as well. Hence, there is a strong connection between turbidity and river loadings of sediment (Fabricius et al. 2014). The TSS plume loading modelling (da Silva et al. in prep.) allows us to assess loadings and in a sense predict the likely conditions of suspended sediment in various areas of the GBR lagoon. When fine sediment is delivered to shallow areas less than 12 metres in depth, it is a good indicator for likely re-suspension later in the year. Therefore, both the concentration data and the modelled loading distribution are relevant to this assessment. The actual exposure of benthic organisms (for example in Cleveland Bay) to flood plume turbidity is more relevant for the assessment of acute effects (see Petus et al. 2014).

Nutrients

Land-sourced run-off containing elevated nutrient concentrations results in flood plumes in the GBR lagoon which may result in a range of impacts on coral communities (Fabricius et al. 2005; Fabricius 2011; Brodie et al. 2011). Dissolved inorganic and particulate forms of nutrients discharged into the GBR are both important in driving ecological effects but it is currently thought that increased nitrogen inputs are more important than phosphorus inputs (Furnas et al. 2013), although this is still uncertain. Dissolved inorganic forms of nitrogen and phosphorus are considered to be of greatest concern compared to dissolved organic and particulate forms of nutrients, as they are immediately and completely bioavailable for algal growth. Particulate forms mostly become bioavailable over longer timeframes, and dissolved organic forms typically have limited and delayed bioavailability (Furnas et al. 2013).

Most studies in GBR waters 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 & Davies 1996), and potentially higher rates of coral diseases (Bruno et al. 2003). 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 2011). 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. Critically, more recent research shows that direct interactions between nutrients species such as nitrate and enhanced coral bleaching susceptibility will be important as a clear example of direct synergy between climate change stress and nutrient enrichment stress (Wooldridge 2009; Wooldridge & Done 2009).

The impacts of nutrients on seagrass are less well known and there has been limited detailed exploration of nutrient dynamics and nutrient limitation in the GBR, with notable exceptions (Udy et al. 1999; Mellors 2003). Therefore, nutrients as an environmental driver has so far been difficult to elucidate because of other over-riding factors such as light limitation, which tends to be a primary driver (Collier & Waycott 2009). Nutrient enrichment can stimulate seagrass growth (Udy & Dennison 1997; Udy et al. 1999) if other factors, such as light availability, are not limiting (Collier 2013). Although a theoretical nutrient toxicity level does exist, nutrient over-enrichment tends to impact at ecosystem scales and follows a path of eutrophication with excessive production of organic matter. In addition, nutrients favour the growth of plankton, macroalgae and epiphytic algae, all of which attenuate light to seagrass leaves (Collier 2013). In the GBR some very high epiphyte loads occur on seagrass meadows of the GBR (McKenzie et al. 2012) and are likely to reduce light reaching seagrass leaves. However, to date, these have largely been seasonal blooms, and epiphyte cover has not correlated well with seagrass abundance (McKenzie et al. 2012). Although nutrient enrichment has been linked to high algal cover (Campbell et al. 2002), seagrass loss has rarely been attributed to nutrient over-enrichment. Further discussion of the impact of flood plumes and degraded water quality on seagrass ecosystems in the GBR is included in Petus et al. (2014).

It is important to note that particulate matter in plumes changes from ‘clay’ (or mineral)-based material in inshore regions, to organic matter (algal material) in offshore regions. These different types of particulate matter can have different effects on coral reefs and seagrass meadows as described in Brodie et al. (2013c).

Further discussion of the impacts of TSS and DIN on GBR ecosystems is provided in Schaffelke et al. (2013). Given the importance of flood plumes in delivering TSS, DIN and PN to the GBR, plume loadings have been included in the assessment.

Method

The frequency and extent of the influence of flood plumes containing differing concentrations of TSS, DIN and PN is used to provide an estimation of the extent of surface exposure of coral reefs and seagrass during wet season conditions. TSS, DIN and PN plume load maps were produced combining in situ data collected under the Marine Monitoring Program (GBRMPA), plume maps derived from Moderate Resolution Imaging Spectroradiometer (MODIS) imagery and monitored end-of catchment load for each parameter in each wet season (c.a. December to April, inclusive) from 2003 to 2013 (Brodie et al. 2015; da Silva et al. in prep.).

The river loads provide the amount of each constituent (TSS, DIN or PN) that has been delivered along the GBR. The in situ data provides the constituent mass variation as a function of the river plume movement away from the river mouth. The satellite imagery provides the direction and intensity that each constituent mass is transported over the GBR lagoon. As a result, this method produces maps of dispersion of each constituent in the GBR waters expressed in mass per area. Annual maps of each constituent were produced to describe differences in GBR exposure to these pollutants. Mass/area maps were converted to concentration maps by dividing it by the bathymetry of the GBR lagoon. Annual maps were averaged to represent the mean distribution of each constituent concentration over the GBR lagoon.

Assessment classes

The assessment classes are based on knowledge of in situ monitoring wet season and ambient conditions in flood events, and are shown in Table 2.3.

Table 2.3. Classifications applied to the remote sensing datasets for the percentage of valid observations exceeding the thresholds.

Parameter	Classification: Score				
	Very Low: 0	Low: 0.25	Medium: 0.5	High: 0.75	Very High: 1
TSS loading (mg/L)	<1	1-2	2-6.6	6.6-15	>15
PN loading (µg/L)	<10		10-20	>20	
DIN loading (µg/L)	<1	1-2	2-7	7-25	>25

2.2.3 Pesticide concentration mapping

Waters of the GBR lagoon are contaminated with a range of pesticides including herbicides, insecticides and fungicides. Pesticides, unlike nutrients, sediments and metals, have no natural sources and their concentrations have been positively correlated with low salinity associated with river run-off (Lewis et al. 2009; Kennedy et al. 2012). Therefore, the occurrence of pesticides in the GBR can be attributed with great confidence to agriculture in the catchments that result in river discharge into the GBR lagoon. Of the 34 pesticides that have been detected in catchments draining to the GBR, several persistent and mobile PSII herbicides dominate the pesticides identified in water samples and passive samplers in both near-shore and offshore sites on the GBR. Further information describing the relevance of PSII herbicides to this assessment is described in Lewis et al. (2013).

Multiple PSII herbicides are usually detected in water samples from the GBR (Lewis et al. 2012) and their combined effects on microalgae are additive (Shaw et al. 2009; Magnusson et al. 2010). This additive toxicity is not currently addressed in regulatory guidelines (King et al. 2013; Lewis et al. 2012) and is considered to be important in this assessment. The reduced photosynthesis in algae due to herbicide exposure causes reductions in the growth of these algae (Magnusson et al. 2008) and changes in species composition (Magnusson et al. 2012) but the effects of chronic exposures in near-shore environments remain largely unknown. This assessment incorporates an assessment of the acute exposure of PSII herbicides in the 2009–11 wet seasons.

Method

A full description of this method is provided in Lewis et al. (2013). A modelling approach based on the relationship between Colour Dissolved Organic Matter (CDOM) and sea surface salinity (Schroeder et al. 2012), was used with the results of in situ end of catchment and GBR lagoon pesticide concentration results for the 2009–2010 and 2010–2011 wet seasons.

Pesticide concentrations were assessed at the end-of-catchment monitoring sites in the 2009–2010 and 2010–2011 water years (Smith et al. 2012; Turner et al. 2012, 2013) to identify the periods where the higher concentrations coincided with elevated stream flows (based on the gauges of the Queensland Department of Natural Resources and Mines; QDNRM 2012). MODIS Level-0 data with 1 km^2 resolution were acquired from the NASA Ocean Colour website[4]. The most appropriate satellite image (i.e. the most free of cloud cover and sun glint) was selected for each NRM region within one week following the highest PSII concentration. MODIS images were processed with the SeaWiFS Data Analysis System (SeaDAS). The semi-analytical model developed by Garvel-Siegel-Maritorena (GSM, Maritorena et al. 2002) implemented in SeaDAS was used to retrieve the absorption coefficient for dissolved and detrital material (CDOM+D). Bio-optical algorithms often fail to retrieve correct information over reef bottom type. Pixels values corresponding to reef locations were thus masked out from the CDOM regional maps. The Cape York region was excluded from this process due to the lack of monitoring data and the limited use of pesticides in this region.

[4] http://oceancolor.gsfc.nasa.gov Accessed August 2015

CDOM was extracted from the satellite images and the relationship established by Schroeder et al. (2012) between measured salinity, and CDOM was used to estimate sea surface salinity in the flood plumes. All of the regional pesticide maps were imported into ArcGIS for post-processing. Missing information (related to atmospheric perturbations, cloud cover or reefs that were masked out) was interpolated in ArcGIS. Pesticide levels were classified into different level of risk and the areas of reef and seagrass meadows at risk for each NRM region were quantified.

Two different but complementary methods were used to determine the risk posed by mixtures of PSII herbicides. These were the Toxic Equivalence Quotient (TEQ) method (e.g. Kennedy et al. 2012; Smith et al. 2012) and the multiple substances potentially affected fraction (ms-PAF) method (Traas et al. 2002). Importantly both methods use the concentration addition model to determine the toxicity of mixtures of PSII herbicides. The maps shown in this assessment are from the TEQ method.

Assessment classes

The key PSII herbicides of concern (diuron, hexazinone, atrazine, tebuthiuron, ametryn and simazine) were normalised to a herbicide-equivalent concentration, which is based on the relative toxicity of diuron. The risk posed by PSII herbicides collectively could then be examined using the concentration addition model for joint toxicity (see Kennedy et al. 2012). The relative toxicities (EC_{50} and EC_{25}) of marine organisms including coral species (*Seriatopora hystrix* and *Acropora formosa*), diatoms (*Phaeodactylum tricornutum*) and green algae (*Chlorella vulgaris*) (Jones & Kerswell 2003; Bengtson Nash et al. 2005; Muller et al. 2008) to each PSII herbicide compared to diuron was determined and then averaged to produce the relative toxicity factors (RTFs) (Kennedy et al. 2012). The TEQ method was applied to the measured EC_{50} and EC_{25}s of PSII herbicides that inhibit the effective quantum yield (YII) in plants. Inhibition in YII by PSII herbicides is proportional to inhibition of photosynthesis and growth in tropical microalgae (Magnusson et al. 2008) as well as reduced energy acquisition by the host coral from its photosynthetic symbionts (Cantin et al. 2009).

Based on the toxicity of diuron calculated in several studies on coral and seagrass species we devised a set of threshold values that were considered to match the following risk classifications:

- **Very High**: >10 µg/L causes reduced growth and mortality in seagrass (Gao et al. 2011) and loss of symbionts (bleaching) in corals (Jones et al. 2003; Negri et al. 2005). The effect on health and survival of foundation species of the GBR can be catastrophic.
- **High:** 2.3–10 µg/L Photosynthesis is reduced by between 50% and 90% in corals (Jones & Kerswell 2003; Negri et al. 2011); seagrass (Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). A 50% reduction of growth and biomass of tropical microalgae was also reported in this concentration range (Magnusson et al. 2008). The community structure of tropical microalgae is significantly affected and this causes significant

changes in the tolerance of microbial communities to herbicides (Magnusson et al. 2012). The effect on primary production is major.

- **Medium:** 0.5–2.3 µg/L Photosynthesis is reduced by between 10% and 50% in corals (Negri et al. 2011); seagrass (Haynes et al. 2000; Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). The community structure of tropical microalgae can be affected by concentrations of diuron as low as 1.6 µg/L (Magnusson et al. 2012). The effect on primary production is moderate.
- **Low:** 0.1–0.5 µg/L Photosynthesis is reduced by up to 10% in corals (Negri et al. 2011); seagrass (Haynes et al. 2000; Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). The effect on primary production is minor.
- **Very Low:** 0.025–0.1 µg/L No observed effect on photosynthesis in corals (Negri et al. 2011); seagrass (Haynes et al. 2000; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010).
- **No Risk:** < 0.025 µg/L

The highest risk classification determined for any point in a flood plume from a catchment was adopted as the risk posed by that catchment.

Further explanations of the methods are provided in Lewis et al. (2013).

2.2.4 Other variables

Additional variables were considered that have not been included here due to the current lack of data showing their temporal and spatial patterns and ecological impacts. These include phosphorus exposure, chronic exposure to PSII herbicides and non-PSII herbicides, and time series of PSII herbicide concentration data.

However, it is possible to include more pollutants in the loads assessment. For the loads assessment, while we consider nitrogen to be a more important nutrient than phosphorus with respect to effects in the marine environment (Furnas et al. 2013), we have limited certainty around this assumption. In our assessment in ranking of end-of-catchment pollutant loads, we have considered PP and DIP to be equally relevant to PN and DIN given our current limitations in understanding.

Table 2.4. Summary of water quality variables, assessment classes and data sources included in the marine risk assessment.

Variables	Assessment Class					Data source/methodology
	Very Low 1	Low 2	Medium 3	High 4	Very High 5	
Sediments						
Total Suspended Solids (TSS) concentration (mg/L)						Based on daily satellite observations of TSS in the period 1 November 2002 to 30 April 2014. Data has been interpolated across reefs (which are masked during image processing) using Euclidean Allocation in ArcGIS. Classification of frequency of exceedance is based on the number of valid observations in the full observation period. Method for extraction described in Maynard et al. (2015).
Frequency of exceedance % 2 mg/L threshold	<1	1–10	10–20	20–50	50–100	Threshold correlates strongly with declines in ecosystem condition such as increased macroalgal growth and declining diversity. Average annual threshold for TSS in the Great Barrier Reef Water Quality Guidelines.
TSS Plume Loading (mg/L) (mean 2003-2013)	<1	1–2	2–6.6	6.6–15	>15	The frequency and extent of the influence of flood plumes containing differing concentrations of TSS is used to provide an estimation of the extent of surface exposure of coral reefs and seagrass during wet season conditions. TSS plume load maps were produced combining in situ data collected under the Marine Monitoring Program, plume maps derived from MODIS imagery and monitored end-of catchment TSS load in each wet season (c.a. December to April, inclusive) from 2003 to 2013 (Brodie et al. 2015; da Silva et al. in prep.). The river loads provide the amount of TSS that has been delivered along the GBR. The in situ data provides the TSS mass variation as a function of the river plume movement away from the river mouth. The satellite imagery provides the direction and intensity the TSS mass is transported over the GBR lagoon. This method produces maps of the TSS dispersion in the GBR waters expressed in mass per area. Annual maps of TSS were produced to describe differences in GBR exposure to this pollutant. Mass/area maps were converted to concentration maps by dividing it by the bathymetry of the GBR lagoon. Annual maps were averaged to represent the mean distribution of the TSS concentration over the GBR lagoon. The same method is applied to calculate DIN and PN loading.

Variables	Assessment Class					Data source/methodology
	Very Low 1	Low 2	Medium 3	High 4	Very High 5	
Nutrients						
Chlorophyll *a* (Chl-a) concentration (µg/L)						Assessment classes were based on long-term Chl-a mean concentration derived from in situ monitoring data (1988-2006) (De'ath & Fabricius 2008).
Mean concentration (µg/L)	<0.18	0.18–0.25	0.25–0.45	0.45–0.8	>0.8	Chl-a is an indicator of nutrient enrichment in marine waters. De'ath and Fabricius (2008) identified 0.45 µg/L as an important ecological threshold for macroalgal cover, hard coral species richness, octocoral species richness. Annual average threshold for chlorophyll in the Great Barrier Reef Water Quality Guidelines. Significant benefits for the ecological status of reefs in the region are likely if mean annual chlorophyll concentrations remain below this concentration. Other classes are described in Section 1.1.1.
Dissolved Inorganic Nitrogen (DIN) Plume Loading (µg/L) (mean 2003-2013)	<1	1–2	2–7	7–25	>25	Elevated dissolved inorganic nitrogen (DIN) is an indicator of nutrient enrichment. High concentrations of DIN can reduce coral recruitment (Babcock & Davies 1991; Loya et al. 2004), enhance coral bleaching susceptibility (Wooldridge & Done 2009) and change the relationship between coral and macroalgal abundance (De'ath & Fabricius 2010). Elevated concentrations can also be deleterious to seagrass by lowering ambient light levels via the proliferation of local light absorbing algae, thereby reducing the amount of photosynthesis in seagrass, particularly in deeper water (Collier 2013). Particulate nutrients are also important in driving ecological effects. Particulate forms mostly become bioavailable over longer time frames than dissolved inorganic nutrients (Furnas et al. 2013). The same method is applied to calculate TSS, DIN and PN loading. Refer above.

Variables	Assessment Class					Data source/methodology
	Very Low **1**	**Low** **2**	**Medium** **3**	**High** **4**	**Very High** **5**	
Particulate Nitrogen (PN) Plume Loading (µg/L) (mean 2003–2013)	<10		10–20		>20	The same method is applied to calculate TSS, DIN and PN loading. Refer above.
PSII Herbicides						
PSII Herbicide modelled concentration (µg/L)	0.025–0.1	0.1–0.5	0.5–2.3	2.3–10	>10	Based on an estimate of the relationship between CDOM and salinity, and then a modelled salinity to PSII herbicide concentration relationship in a flood plume event in one river in each NRM region in 2009–2011. Data has been interpolated across reefs (which are masked during image processing) using Euclidean Allocation in ArcGIS. Risk posed was determined using a number of methods — some only assessed acute toxic effects, others both acute and chronic. Described in Lewis et al. (2013). **No Risk: <0.025 µg/L**; **Very Low: >0.025–0.1 µg/L**: No observable effect; **Low: 0.1–0.5 µg/L**: Photosynthesis is reduced by up to 10% in corals (Negri et al. 2011); seagrass (Haynes et al. 2000; Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). The effect on primary production is minor. **Medium: 0.5–2.3 µg/L**: Photosynthesis is reduced by between 10% and 50% in corals (Negri et al. 2011); seagrass (Haynes et al. 2000; Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). The community structure of tropical microalgae can be affected by concentrations of diuron as low as 1.6 µg/L (Magnusson et al. 2012). The effect on primary production is moderate. **High: 2.3–10 µg/L** Photosynthesis is reduced by between 50% and 90% in corals (Jones & Kerswell 2003; Negri et al. 2011); seagrass (Chesworth et al. 2004; Gao et al. 2011; Flores et al. 2013) and microalgae (Magnusson et al. 2008, 2010). A 50%

Variables	Assessment Class					Data source/methodology
	Very Low 1	**Low 2**	**Medium 3**	**High 4**	**Very High 5**	
						reduction of growth and biomass of tropical microalgae was also reported in this concentration range (Magnusson et al. 2008). The community structure of tropical microalgae is significantly affected and this causes significant changes in the tolerance of microbial communities to herbicides (Magnusson et al. 2012). The effect on primary production is major. **Very High: > 10 µg/L**: reduced growth and mortality in seagrass (Gao et al. 2011) and loss of symbionts (bleaching) in corals (Jones et al. 2003; Negri et al. 2005).

2.3 Assessment Indexes

The variables described above and shown in Figure 2.1 have been combined into a number of indexes related to marine ecological risk and end-of-catchment pollutant loads.

2.3.1 Marine Risk Index

To estimate ecological risk in this assessment, we selected six water quality variables described above that represent key run-off-transported pollutants of greatest concern to two GBR ecosystems: seagrass meadows and coral reefs. To account for limitations in the available datasets (see Brodie et al. 2013a for further explanation), ecological risk was expressed as the area of these ecosystems within a range of spatially defined assessment classes (very low to very high relative risk) for several water quality variables for the marine NRM region, and then in each river's zone of influence where available (see explanation below). The variables included ecologically relevant thresholds for concentrations of Chl-a obtained from long-term in situ monitoring data and TSS from remote sensing observations, and the distribution of key pollutants including TSS, DIN, PN and PSII herbicides in the marine environment during flood conditions (based on end-of-catchment loads and plume loading estimates). A factor that represents the influence of crown-of-thorns Starfish (COTS) on coral reefs, and the differential influence of river discharges on the COTS initiation zone was included in previous assessments and has been excluded here as it is not considered to be relevant to the Fitzroy NRM region.

For each of the variables shown in Figure 2.1 and Table 2.1 a classified spatial data layer was prepared in ArcGIS. The classifications, scores and overall weightings for this assessment were customised using the expert opinions of the project team and are shown in Table 2.5. The assessment classes for each variable were allocated a score between 0 (lowest severity) and 1 (highest severity) at the 1 km^2 pixel scale. Pixels in the highest assessment class all received the maximum value of 1. For example, for the TSS threshold of 2 mg/L the scores for the frequency of exceedance classes would be Very Low (<1% exceedance) = 0; Low (1–10% exceedance) = 0.25; Medium (10–20% exceedance) = 0.5; High (20–50% exceedance) = 0.75; and Very High (50–100% exceedance) = 1.0. The areas of coral reefs and seagrass meadows were reported for each assessment class in for the marine NRM region and each available river zone of influence in ArcGIS.

Table 2.5. Summary of the classes for each variable and the weightings given to each assessment class for the combined relative risk assessment.

Variables	Overall weighting	Assessment Class				
		Very Low 1	Low 2	Medium 3	High 4	Very High 5
TSS threshold exceedance 2mg/L Frequency of exceedance (%)	1/6	<1	1-10	10-20	20-50	50-100
Score		*0*	*0.25*	*0.5*	*0.75*	*1.0*
TSS Plume Loading (mg/L) (mean 2003–2013)	1/6	<1	1-2	2-6.6	6.6-15	>15
Score		*0*	*0.25*	*0.5*	*0.75*	*1.0*
Chl long term concentration (µg/L) (mean 1988–2006)	1/6	<0.18	0.18-0.25	0.25-0.45	0.45-0.8	>0.8
Score		*0*	*0.25*	*0.5*	*0.75*	*1.0*
DIN Plume Loading (µg/L) (mean 2003–2013)	1/6	<1	1-2	2-7	7-25	>25
Score		*0*	*0.25*	*0.5*	*0.75*	*1.0*
PN Plume Loading (µg/L) ((mean 2003–2013)	1/6	<10		10-20	>20	
Score		*0*		*0.5*	*1*	
PSII Herbicide modelled concentration (µg/L) (2009–2011)	1/6	0.025-0.1	0.1-0.5	0.5-2.3	2.3-10	>10
Score		*0.25*	*0.5*	*0.75*	*1.0*	*No occurrence*

Note: The variables are described in Table 2.4.

Ideally, the classes for each variable would be scaled so that they are equivalent in terms of potential ecological impacts to provide comparable weightings between variables. However, our knowledge of ecosystem impacts is not sufficiently advanced to allow comparable scaling of variables. As temporal and spatial resolution of the input data increases and the knowledge of the impacts of sediments, nutrients and PSII herbicides on GBR ecosystems is advanced, this capability can be improved in future assessments. After testing several approaches to weighting the variables, it was agreed to weight each spatial layer equally and as additive factors. The data layers were then combined using the Union tool in ArcGIS and the values of each coincident pixel were summed, normalised and classified into five even break classes ranging from Very Low to Very High. An example of the process applied in ArcGIS is shown in Figure 2.8.

The area of coral reefs and seagrass meadows in each of the five assessment classes of the combined layer in the marine NRM region was calculated. Where the river zones of influence area are available, areas were calculated to allow comparison between basins. The Marine Risk Index for each basin was calculated by summing the areas of coral reefs and seagrass meadows only in the highest assessment classes of the combined layer. To allow relative comparison between the basins in the region, each result was anchored to the basin with the maximum area, which was given a score of 1.0. This enabled an assessment of the relative differences between basins in terms of combined water quality risk for coral reefs and seagrass meadows. The final output is a Coral Reef Marine Risk Index and a Seagrass Marine Risk Index.

As a final step, these Indexes were summed with the Loads Index for each basin to determine the overall relative risk of degraded water quality to coral reef and seagrass ecosystems for each basin in the Fitzroy region.

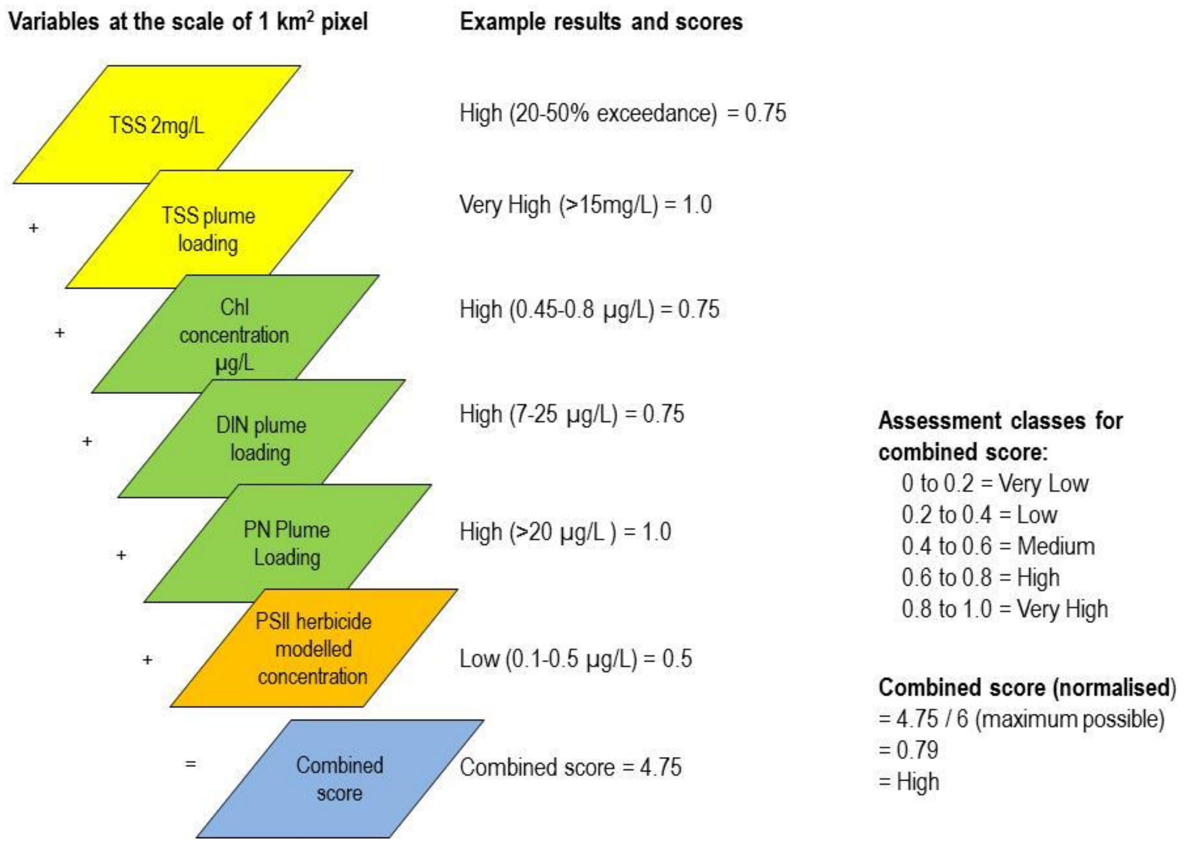

Figure 2.8. Example of the results in one pixel (1 km2) in ArcGIS.

The result for coincident cells from each layer is summed to give a combined score, normalised and classified into five assessment classes (Very Low to Very High). In this example the combined score gives the cell a score within the High assessment class in terms of relative risk of degraded water quality. The colours represent groups of variables: yellow = sediment related variables, green = nutrient related variables and orange = PSII herbicide related variables.

2.3.2 Loads Index

To inform management priorities that aim to address the risks identified in the Marine Risk Index, it is necessary to understand the influence of river discharge from each of the basins, as these discharges carry the majority of the pollutants into the GBR lagoon. Modelled end-of-catchment pollutant loads, generated from the Source Catchments model framework for the Paddock to Reef Program (Dougall et al. 2014) were obtained for each basin for key pollutants: TSS, DIN, PN, DIP, PP and PSII herbicides. First, the Source Catchments modelling framework was used as a synthesis tool that incorporates new information on paddock modelling of TSS, speciated nitrogen and phosphorous, and PSII herbicides, plus spatially and temporally remote sensed inputs (Dougall et al.

2014). This resulted in a consistent set of end-of-catchment pollutant loads for each of the basins in the Fitzroy region (Dougall et al. 2014), which is part of a larger project that models all of the 35 GBR catchments (Waters et al. 2014). Anthropogenic load is calculated as the difference between the long-term average annual load and the estimated pre-European annual loads. A fixed climate period was used (1986 to 2009) for all model runs to normalise for climate variability and provide a consistent representation of pre-development and anthropogenic generated catchment loads. This therefore represents an ‘average’ year rather than the extremes such as those recorded in the period 2008 to the wet season in 2013. In addition, functionality from the previous iteration of catchment modelling, SedNet/ANNEX (for example see Cogle et al. 2006), was incorporated into Source Catchments to represent hillslope, gully and streambank erosion and floodplain deposition processes.

For this assessment, the anthropogenic load was incorporated as a proportion of the total load, as it is only the anthropogenic portion that is assumed to be the ‘manageable’ component of pollutant loads. The anthropogenic load is calculated as the difference between the long-term average annual load (when 2008–2009 management inputs and distributions are assumed), and the estimated pre-European load. The proportional basin contributions were then anchored (to normalise to a standard scale) and summed to generate a combined **Loads Index** for TSS, speciated nitrogen and phosphorous, and PSII herbicides for each basin. This assumes that the relative importance of each load is equal, which may not be the case, although there is currently insufficient knowledge to weight the importance of the four pollutants relative to each other.

It is recognised that assessment of the input of PSII herbicides from each region can be expressed in a number of ways, and while loads allow comparison between basins, it is the toxicity and therefore concentration that is most relevant to the receiving environment. Therefore, toxic equivalent loads have been calculated, which reflects the ecological toxicity of the loads (Lewis et al. 2013). The ‘toxic loads’ were calculated using a three-step process:

(1) the calculation of herbicide load data for diuron, atrazine, hexazinone, ametryn and tebuthiuron for the individual sub-catchments of the GBR (see Lewis et al. 2011)

(2) the conversion of these data to a combined ‘toxic’ PSII load using the toxic equivalency factors of Smith et al. in prep.)

(3) the calculation of an annual mean concentration for the individual sub-catchments of the GBR.

The calculation of the herbicide load data involved re-analysis of the Lewis et al. (2011) model to include the monitored load data from the 2010–11 water year, from Turner et al. (2013). A combination of monitored load data and land use data were used to model herbicide loads across basins using the approach outlined in Lewis et al. (2011). The load data for the individual herbicides were then converted to a toxic PSII herbicide load (normalised relative to the toxicity of diuron), using the toxic equivalency factors calculated by Smith et al. (in prep.). Hence the PSII herbicide loads represent a normalised toxicity for each of the herbicides, recognising that some have greater PSII herbicide inhibition potential. Finally, the basin ‘toxic loads’ were divided by their respective mean annual flows to calculate an annual mean concentration. Since the toxicity of herbicides is related to concentration rather than load, this step is designed to help account for the influence of dilution on the herbicide toxicity between the different basins.

2.3.3 Relative Risk Index

To provide an overall relative ecological risk ranking between the Fitzroy basins (where possible), the Marine Risk Indexes for coral reefs and seagrass meadows were summed with the Loads Index, to generate a Coral Reef Relative Risk Index and a Seagrass Relative Risk Index. These final indexes for coral reefs and seagrass were then summed and normalised (0 to 1) to give an overall assessment of the relative risk of degraded water quality to coral reefs and seagrass meadows to generate a **Relative Risk Index** for each basin.

2.4 Recognising and assessing uncertainties in the data

Given the limited time and resources available for this study, differences in uncertainty and hence our confidence in the data can only be assessed highly subjectively and no specific quantitative estimates were considered. If such qualitative assessments of uncertainty in our methodologies and data were undertaken, uncertainty would be assessed as varying as much within as among basins. However, in an attempt to provide relative differences between datasets, a qualitative statement of data confidence is included (low, moderate or high) below and noted in the Results (Section 3) for each variable.

The zones of influence defined for the modelled rivers in the Fitzroy region are an estimate only and the method requires refinement. There are a number of limitations to the existing approach:

- For the hydrodynamic model, each river is modelled individually ('turned on' in the model one at a time) so there is no influence of the combined forcing of multiple river discharges. The general movement of river discharges in a northern direction will influence water movement and hence spatial extent in reality. This is relevant for the Mary River, which is south of the Fitzroy River and extends north into the region in large wet season events.
- The selection of the threshold requires further testing to optimise the representation of average wet season conditions. The tracer thresholds also need to be correlated with in situ and/or remote sensing data to show that the threshold level is physically, chemically and biologically relevant.
- The zones were defined using four years of data and should be extended to account for greater inter-annual variability.
- For the unmodelled rivers, the path-distance approach enabled the assessment to extend beyond the rivers that are covered by the hydrodynamic model (just the Fitzroy River in this region). However, the sources of error in the path-distance approach are two-fold: 1) the ability to predict plume maximum extent (i.e. the river mouth to plume edge distance) is only as good as the relationship between extent and flow; and 2) even if the mouth-to-edge distance could be accurately predicted, the plume shape cannot be accurately replicated. The path-distance function was parameterised based on the best overlap with tracer

plumes, but with the current technique, overlap is in the order of 50–70(+) %. Further work is underway to test this, for example, working on shorter timescales (e.g. weekly) may be beneficial.

The relative ranking of uncertainty in the input data for this study has been estimated from the literature and expert opinion. The results for this ranking are included in the description of each variable, and can be summarised as follows:

- Remote sensing TSS — low certainty
- Long-term chlorophyll — moderate to high certainty
- TSS plume loading — moderate certainty
- DIN Plume loading — moderate certainty
- PN Plume loading — moderate certainty
- PSII concentration model — low certainty
- River loads — moderate/high certainty
- Coral reef areas — high certainty
- Seagrass areas — monitored: low/moderate certainty; modelled: low/moderate certainty — applied 50% probability map
- River zones of influence — low certainty

Further discussion of the uncertainties and limitations of the assessment, in addition to improvements to the previous risk assessments, are presented in Section 5.

3 Results

3.1 River zones of influence and habitat areas

The marine area defined for the WQIP is shown in Figure 2.2 and includes the marine NRM region as defined by GBRMPA. This area is further delineated by defining 'zones of influence' (herein referred to as ZoI) for the Water Park, Fitzroy, Calliope and Boyne rivers, but not the Styx or Shoalwater basins (see Section 2.2.2 for Methods). It should be noted that there is a degree of uncertainty in these assessments associated with the modelling approach and the limited period of assessment, but they are still considered to be useful for comparative assessments between basins.

The main characteristics of the zones of influence are summarised below. Only the Fitzroy ZoI extends beyond the inshore areas and for all rivers, the outer boundary is typically within 60 kilometres of the coast (Figure 3.1). There is high variability within the ZoI for each river between years that are included in the assessment — 2008–09, 2010–11, 2011–12 and 2012–13 (as noted above, 2009–10 is not included in the model but this was also a relatively large discharge event for the region).

- The largest river ZoI in the region is by far from the Fitzroy River, with an estimated maximum extent area of 35,409 km^2 (Figure 3.1). This was associated with the 2010–2011 flood events, when the river discharge was almost 38,000,000 ML. The zone of influence extended as far north as the Whitsunday Islands during the 2010–11 and 2012–13 wet seasons (~8,500,000 ML), although the southern extent of these plumes were still within the southern boundary of the NRM region reflecting the northward movement of the plumes.
- The estimated maximum extent ZoI for the Water Park, Calliope and Boyne rivers are much smaller but each have similar areas, estimated around 2,000 km^2 (Figure 3.2, Table 3.1).
- The maximum extent of the Water Park Creek ZoI is estimated to be 2,279 km^2 extending in a northerly direction to Warginburra Peninsula (Shoalwater Bay is on the western side of this peninsula) (Figure 3.2a). In terms of southern extent, in the four years considered, the ZoI only intersected with the whole Keppel Island Group in 2009, and only to North Keppel Island in 2011, 2012 and 2013.
- The maximum extent of the Calliope River ZoI is estimated to be 1,824 km^2 (Figure 3.2c). The river mouth is located just north of Gladstone, and inside the area known as Port Curtis at the southern end of Curtis Island. The ZoI heads in a northerly direction to the north-eastern point of Curtis Island, and only extended to the southern end of the Keppel Island Group in 2013. The ZoI intersected Hummocky Island, several shoals and modelled deepwater seagrass areas in 2011, 2012 and 2013.
- The maximum extent of the Boyne River ZoI is estimated to be 1,802 km^2 (Figure 3.2c). The plumes move in a northerly direction in line with the northern end of Curtis Island, except in 2009 when the ZoI also extended south to Bustard Bay and 1770, but was steered outside of Hummock Hill Island and therefore appeared to go outside of Rodds Bay, which contains

extensive surveyed seagrass areas and is a Dugong Protection Area; however, this assessment would require further verification.

- As expected, the area of coral reefs and seagrass in the zones varies considerably between rivers and is not directly proportional to the area of the zone, due to spatial variability in habitat distribution in the region.

Data confidence: Low due to limitations associated with river flow (see Section 2.4), definition of the threshold, coverage of the spatial layer in coastal areas and limited consideration of the combined effect of river discharges. In addition, not all rivers in the region are modelled, so it is difficult to use as a region-wide assessment.

Table 3.1 shows the area of coral reef, seagrass and zone of influence for the rivers modelled in the Fitzroy region. The total area of coral reef in the GBR is estimated around 24,000 km^2. The total area of coral reefs calculated in the Fitzroy marine NRM region for this assessment is 4,910 km^2. This varies slightly (<2%) from the standard reported area of 4,855 km^2 (Johnson et al. 2015) due to the pixel-based analysis used in this assessment. From the mapping data used in this assessment, the ZoI for the Fitzroy River has the highest area of coral reef estimated at approximately 559 km^2.

Approximately 35,000 km^2 of potential seagrass habitat has been mapped in the coastal waters around Queensland and Torres Strait since the mid-1980s. Surveys and statistical modelling of seagrass in offshore waters deeper than 15 metres (using the 50% probability assessment) shows that 37,454 km^2 of the sea floor within the Great Barrier Reef World Heritage Area and Torres Strait has some seagrass present, making Queensland's seagrass resources globally significant. From the mapping data used in this assessment, the total area of potential seagrass habitat (surveyed) in the marine NRM region for the Fitzroy is 253 km^2 (compared to 241 km^2 reported in Johnson et al. 2015 due to the pixel-based assessment here) and the modelled deepwater (>15m) seagrass estimate is 5,668 km^2. The combined total of 5,921 km^2 (compared to 5,775 km^2 in Johnson et al. 2015; <3% difference due to the pixel-based assessment here) accounts for ~17% of the total area reported for the GBR. The area of surveyed seagrass in the Fitzroy River ZoI is greater than in the region, as it extends northwards into the inshore areas of the Mackay-Whitsunday region (419 km^2; see Figure 3.1).

Data confidence: High for coral reefs, and low/moderate for seagrass given spatial and temporal coverage of the monitoring. The potential extent of deepwater seagrass is modelled and we have used the 50% probability assessment.

Table 3.1. The calculated total area (km^2) of the zone of influence (maximum extent: 2009, 2011, 2012, 2013), mapped coral reef, and mapped and modelled seagrass for the modelled river basins in the Fitzroy region used for this assessment.

River	Mean Annual Discharge (ML)	Zone of Influence (km^2)	Reef (km^2)	Seagrass (km^2)		
				Survey composite	Deepwater (>15m) modelled	Total
Shoalwater	272,000	n/a				
Styx	387,000	n/a				
Water Park	392,000	2279	26	13	8	21
Fitzroy	4,650,000	35,409[1]	559	419	2,863	3,281
Calliope	117,000	1,802	19	80	45	125
Boyne	40,000	1,824	20	119	30	149
Fitzroy Marine NRM region			**4,910**	**253**	**5,668**	**5,921**

Note that this maximum extent was based on an exceptionally large flow in 2010–2011, estimated to be almost 35,000,000ML. River zones of influence are not available for the Shoalwater and Styx basins.

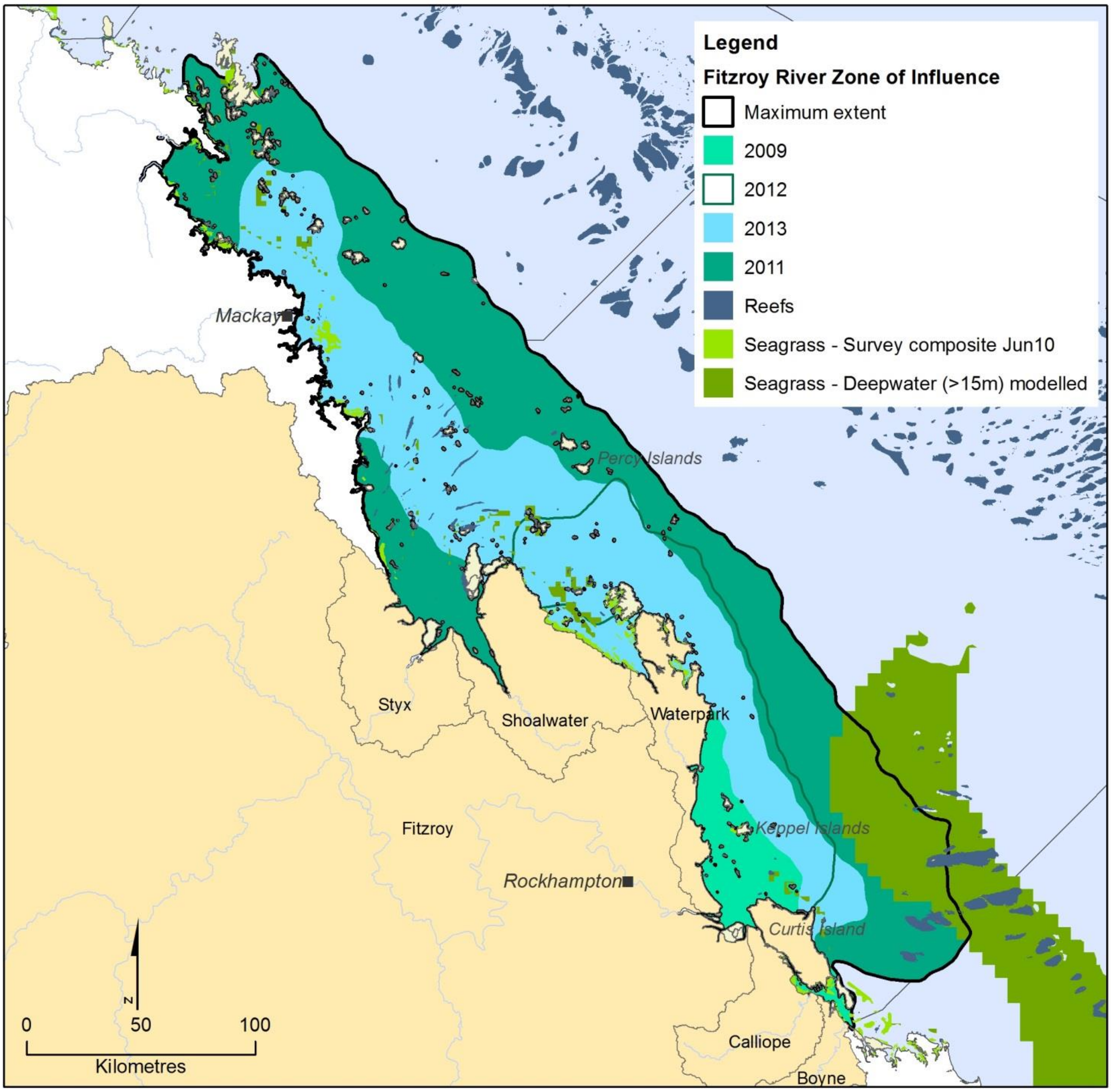

Figure 3.1. Zone of influence modelled for the Fitzroy River.

The method for deriving these zones is described in Section 2.2.2.

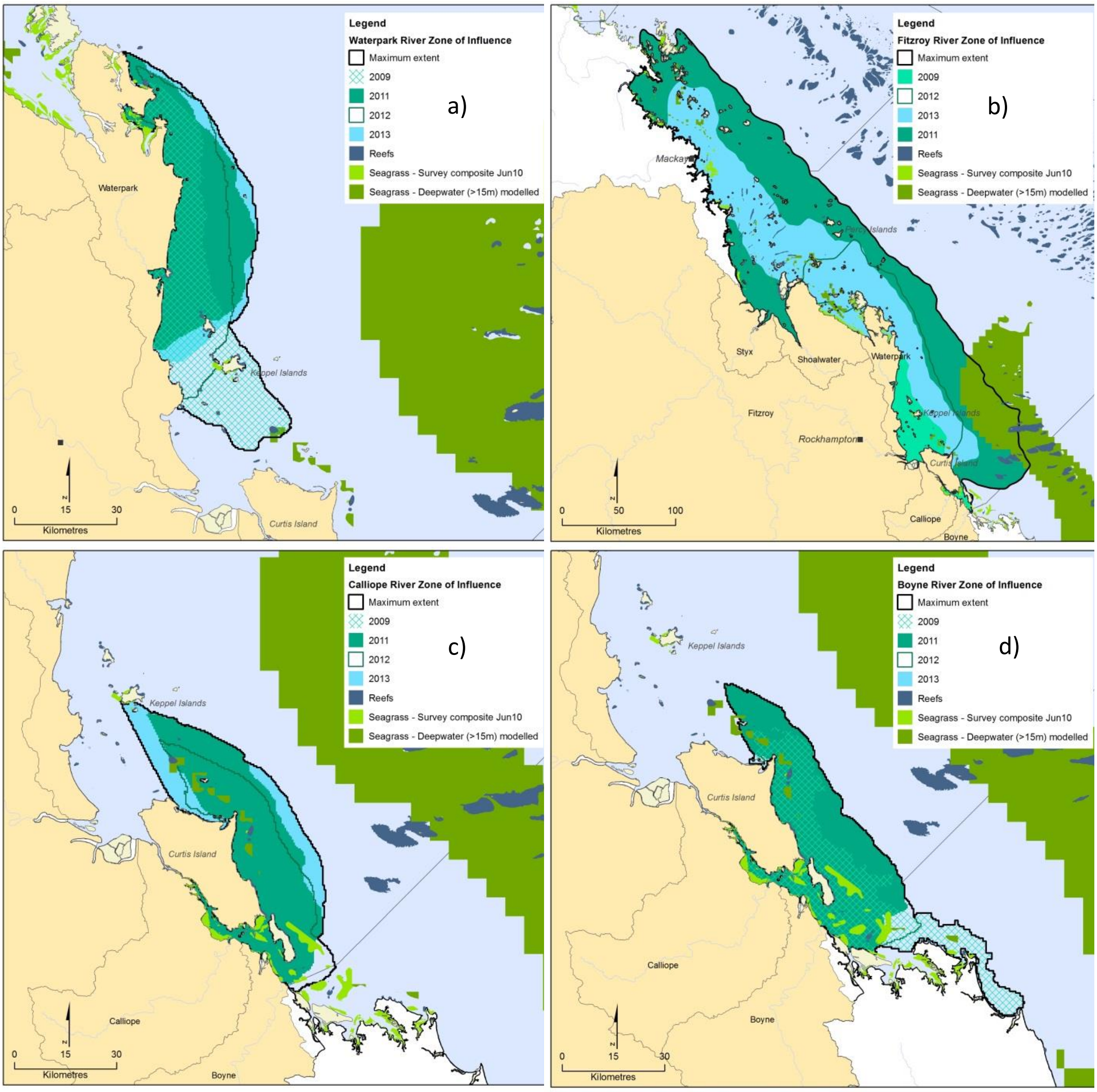

Figure 3.2. Zones of influence modelled for the (a) Water Park Creek, (b) Fitzroy River [note change in scale], (c) Calliope River and (d) Boyne River

Based on application of a threshold to the wet season mean of the tracer data that equates to a salinity of 36ppt, in the wet seasons of 2008–09, 2010–11, 2011–12 and 2012–13. The method for deriving these zones is described in Section 2.2.1.

3.2 Relative differences between marine water quality variables and basin influences

The following section presents the results of the individual variables considered in this assessment. This part of the risk assessment identifies the areas where each water quality variable is considered to pose the greatest relative risk to coral reefs and seagrass in the Fitzroy region. The output can be used to guide priorities for management of individual pollutants, but is not definitive and should only be used in conjunction with expert opinion.

We also applied this approach at a basin-scale, using the zones of influence for each basin as the assessment unit; however, it has been agreed by the project team that the data is not sufficiently reliable at this stage to take the assessment to this level of detail to draw specific conclusions to differentiate relative importance between pollutants. These results are available from the project team as a demonstration of the potential application of this approach but not presented in this report. The areas reported here are relevant to the entire Fitzroy marine NRM region.

The maps for each classified variable are presented below to give an indication of the spatial patterns of pollutant influence in the Fitzroy region. Area calculations are rounded to the nearest whole km^2 for ease of reporting but does include summed portions of some 1 km^2 pixels.

a) Sediments

Total suspended solids threshold exceedance, 2 mg/L

Five assessment classes were used for TSS 2 mg/L based on the percentage of valid observations that exceed the threshold in the period 2002 to 2014, ranging from Very Low to Very High. The results of the assessment are shown in Table 3.2 and Figure 3.2.

The areas of greatest exceedance are located along the coastal areas, with concentrated areas in Broad Sound, Shoalwater Bay and then heading south from the Water Park Creek mouth to Curtis Island, including Keppel Bay, and south beyond the NRM region boundary. There are 90 km^2 of coral reefs in the Very High assessment class (which is <2% of reefs in the region), and approximately 195 km^2 of surveyed seagrass in this area, which is almost 77% of the surveyed seagrass in the region. Only a small proportion of deepwater modelled seagrass (<1%) is within the Very High assessment class. These inshore areas are locations with some of the highest use and visitation rates; this is a result common to all individual variables and is reviewed in the discussion. A majority of the coral reefs and deepwater modelled seagrass are in the Very Low assessment class.

The elevated exceedance around Broad Sound and Shoalwater Bay are most likely associated with naturally high turbidity (it is a relatively shallow area) or uncertainties in the remote sensing results that have not been resolved. Further validation of the algorithm in this area is required to improve the confidence in this result. These conclusions are also supported by the analysis of photic depth and river discharge by Logan et al. (2014, in press).

Data confidence: Low due to limited validation of remote sensing data in near-shore coastal areas, particularly in the shallow and naturally turbid areas such as Shoalwater Bay.

Table 3.2. Area of coral reefs and seagrass meadows within the Very Low to Very High assessment classes for TSS 2 mg/L and the percent of the total habitat area in the region in each class.

TSS 2mg/L Exceedance	Very Low		Low		Moderate		High		Very High		Total
	<1%		1-10%		10-20%		20-50%		50-100%		
Proportion of valid observations	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	64282	74.0	15942	18.4	1278	1.5	1983	2.3	3375	3.9	86860
Coral Reefs	4639	94.5	133	2.7	8	0.2	41	0.8	90	1.8	4910
Seagrass											
Composite survey	1	0.5	9	3.7	21	8.2	28	11.0	195	76.7	255
Deepwater	4648	82.0	744	13.1	23	0.4	206	3.6	47	0.8	5667
Total Seagrass	4649	79	753	13	44	1	234	4	242	4	5922

Note: Results for the assessment are based on frequency of exceedance of TSS 2 mg/L using remote sensing data 2002–2014 (see methods in Section 2.3.1).

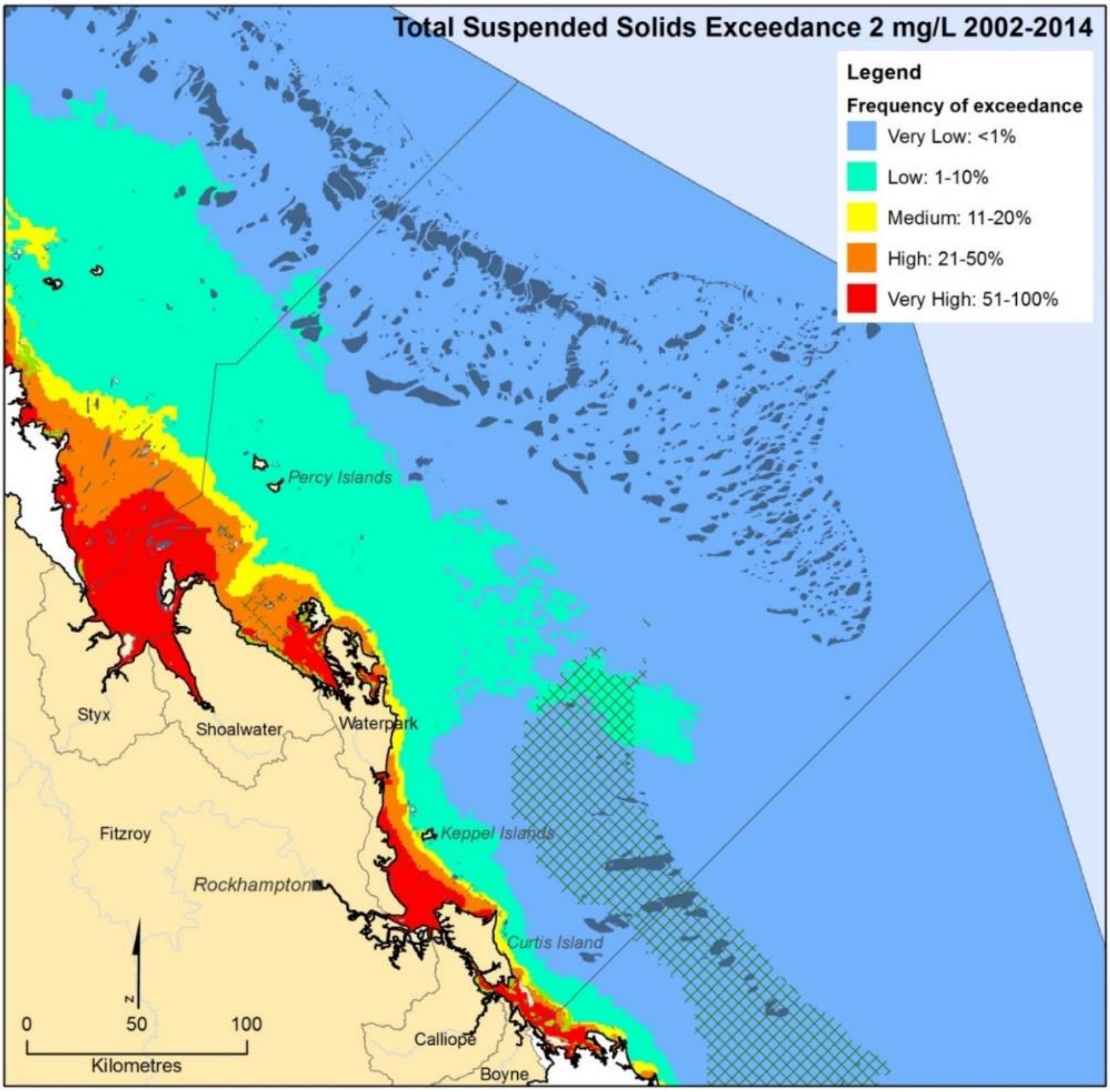

Figure 3.3. Results for the assessment of frequency of exceedance of TSS 2 mg/L using daily remote sensing data 2002–2014.

Results for the assessment are based on frequency of exceedance of TSS 2 mg/L (see methods in Table 2.1). Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

TSS plume loading (mean 2003–2013)

As shown in Table 2.1, five assessment classes (Very Low to Very High) were used for TSS plume loading based on an interpolated map derived from plume frequency information from remote sensing and scaled river load data. The results of the assessment are shown in Table 3.3 and Figure 3.3.

The TSS plume loading area of influence is relatively constrained and only the Low and Very Low assessment classes extend beyond the inshore area. The areas of greatest exceedance are located in Shoalwater Bay and the coastal areas of Keppel Bay. There is only 6 km^2 of coral reefs in the Very High assessment class (which is <1% of reefs in the region) with a majority of reef area in the Very Low assessment class (90%). Approximately 71 km^2 of surveyed seagrass (and no deepwater modelled seagrass) are in the Very High assessment class, which is 28% of the surveyed seagrass in the region. The largest proportion of surveyed seagrass is in the High assessment class (45%) and a majority of the deepwater modelled seagrass is in the Very Low or Low assessment class.

Data confidence: Low/moderate.

Table 3.3. Area of coral reefs and seagrass meadows within the Very Low to Very High assessment classes for TSS Plume loading and the percentage of the total habitat area in the region in each class.

TSS loading	Very Low		Low		Moderate		High		Very High		Total
mg/L	**<1**		**1 to 2**		**2 to 6.6**		**6.6 to 15**		**>15**		
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	65749	75.7	12467	14.35	6537	7.5	1281	1.5	825	1.0	86860
Coral Reefs	4419	90.0	339	6.91	109	2.2	36	0.7	6	0.1	4910
Seagrass											
Composite survey	0	0.0	3	1.4	65	25.7	114	45.0	71	28.0	253
Deepwater	1819	32.1	3420	60.3	422	7.4	6	0.1	0	0.0	5668
Total Seagrass	1819	31	3424	58	487	8	120	2	71	1	5922

The assessment classes are based on mean wet season TSS loading concentrations 2003 to 2013 (see methods in Section 2.3.2).

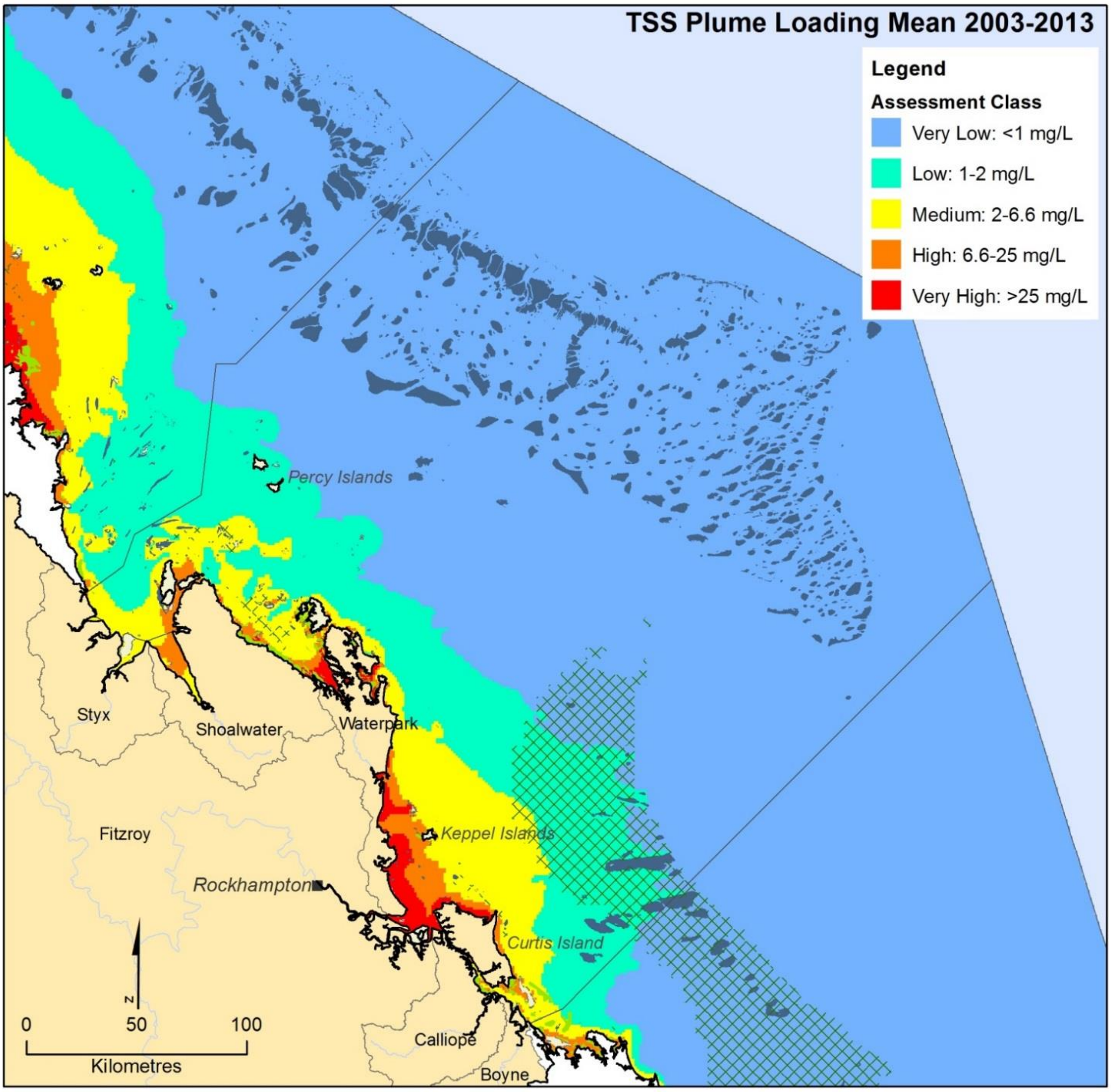

Figure 3.4. Results for the assessment of TSS plume loading (mean of annual assessments 2003 to 2013).

The assessment classes are based on concentration estimates derived from an interpolation of a multi-year analysis that combines scaled river loads data and flood plume frequency analysis from satellite imagery (see methods Table 2.1). Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

b) Nutrients

Chlorophyll-a (Chl-a) long-term monitored concentration

Chl-a concentrations are relevant year-round as an indication of nutrient enrichment in marine waters. As shown in Table 2.1, five assessment classes were used for long-term mean Chl-a concentrations based on in situ monitoring in the period 1988 to 2006, classified according to relative potential impact ranging from Very Low to Very High. In the analysis of this dataset for the development of the GBR Water Quality Guidelines by De'ath and Fabricius (2008), the exceedance of the Chl-a guideline of 0.45µg/L was assessed, including a level of confidence in the estimates; see Figure 3.5. The confidence map shows relatively low confidence in the areas around Shoalwater Bay and Broad Sound, and north to Mackay. This is due to relatively low frequency of sampling in these locations, illustrated in the maps shown in Figure 3.6.

The results of the assessment are shown in Table 3.4 and Figure 3.4. The areas of greatest exceedance are located within Shoalwater Bay and in a band approximately 30 kilometres wide along the coast from Townshend Island to the southern end of the NRM region boundary. This incorporates Keppel Bay and the Keppel Island Group. There are 51 km^2 of coral reefs in the Very High assessment class (which is 1% of reefs in the region) with the largest proportion of reef area in the Moderate assessment class (65%). All of the surveyed seagrass in the area is within the High (98 km^2 or 38%) or Very High assessment (157 km^2 or 62%) class. However, Shoalwater Bay is shown to have low data confidence (Figure 3.5); therefore the area of surveyed seagrass in the Very High assessment class may be an over-estimate of relative risk for Chl-a. There is only 45 km^2 of deepwater modelled seagrass in the Very High assessment classes, with the remainder in the Low (43%) and Moderate (56%) classes.

Data confidence: Low to moderate due to limited spatial coverage of monitored data.

Table 3.4. Area of coral reefs and seagrass meadows within the Very Low to Very High assessment classes for mean long-term chlorophyll concentration and the percentage of the total habitat area in the region in each class.

Chl-a long term mon concentration	Very Low		Low		Moderate		High		Very High		Total
µg/L	<0.18		0.18 to 0.25		0.25 to 0.45		0.45 to 0.8		>0.8		
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	8430	9.7	2	0.00	47070	54.2	25817	29.7	5541	6.4	86860
Coral Reefs	0	0.0	2	0.05	3201	65.2	1655	33.7	51	1.0	4910
Seagrass											
Composite survey	0	0.0	0	0.0	0	0.0	98	38.4	157	61.6	255
Deepwater	0	0.0	2458	43.4	3164	55.8	0	0.0	45	0.8	5667
Total Seagrass	0	0	2458	42	3164	53	98	2	202	3	5922

The assessment classes are based on mean concentrations 1988 to 2006 (see methods in Section 2.3.1).

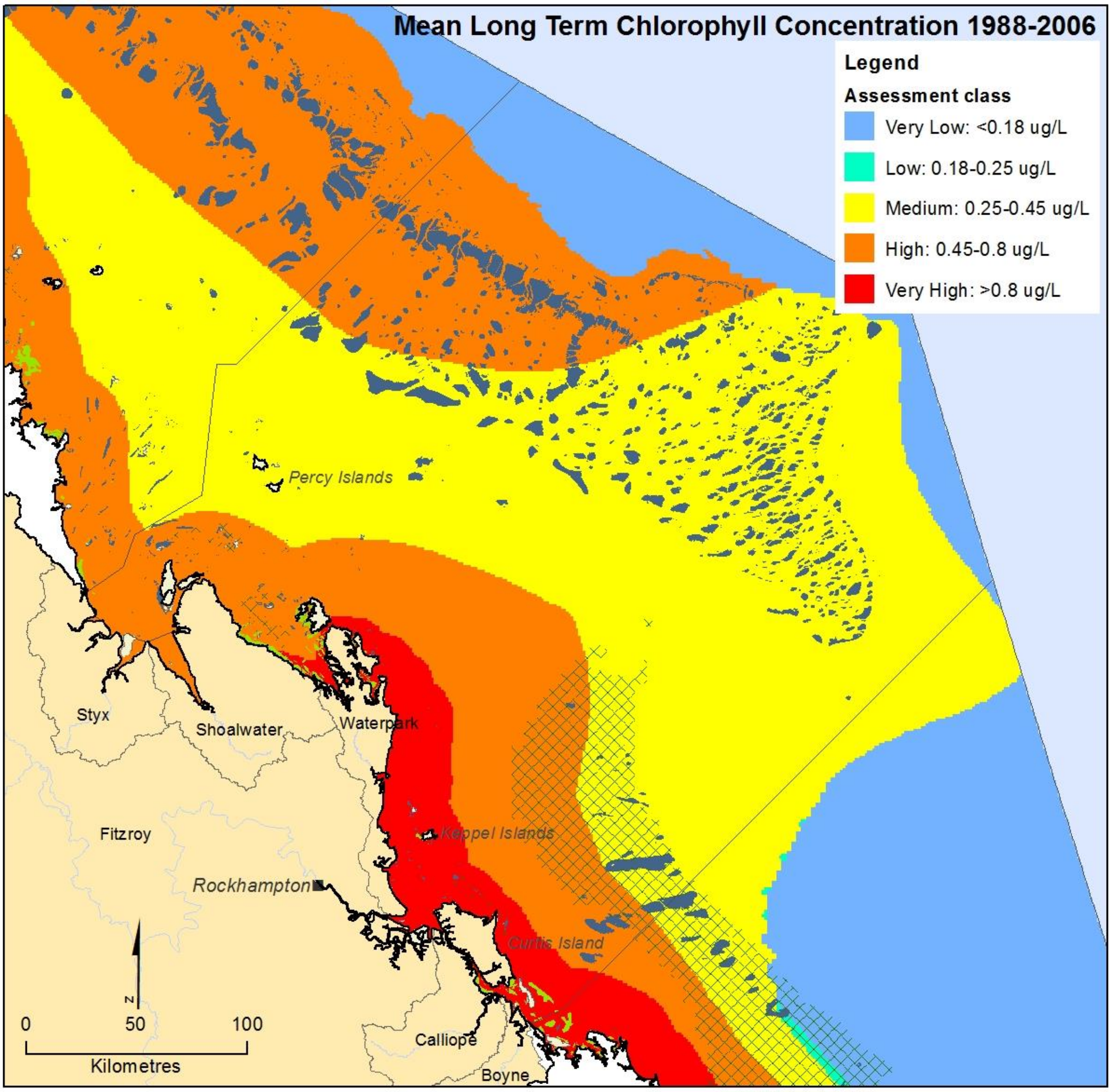

Figure 3.5. Results for the assessment of mean long-term chlorophyll monitoring data Chl a 0.45 µg/L 1998–2006.

Methods are described in Table 2.1. Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch. Refer to Figure 3.6 for a map showing variability in data confidence.

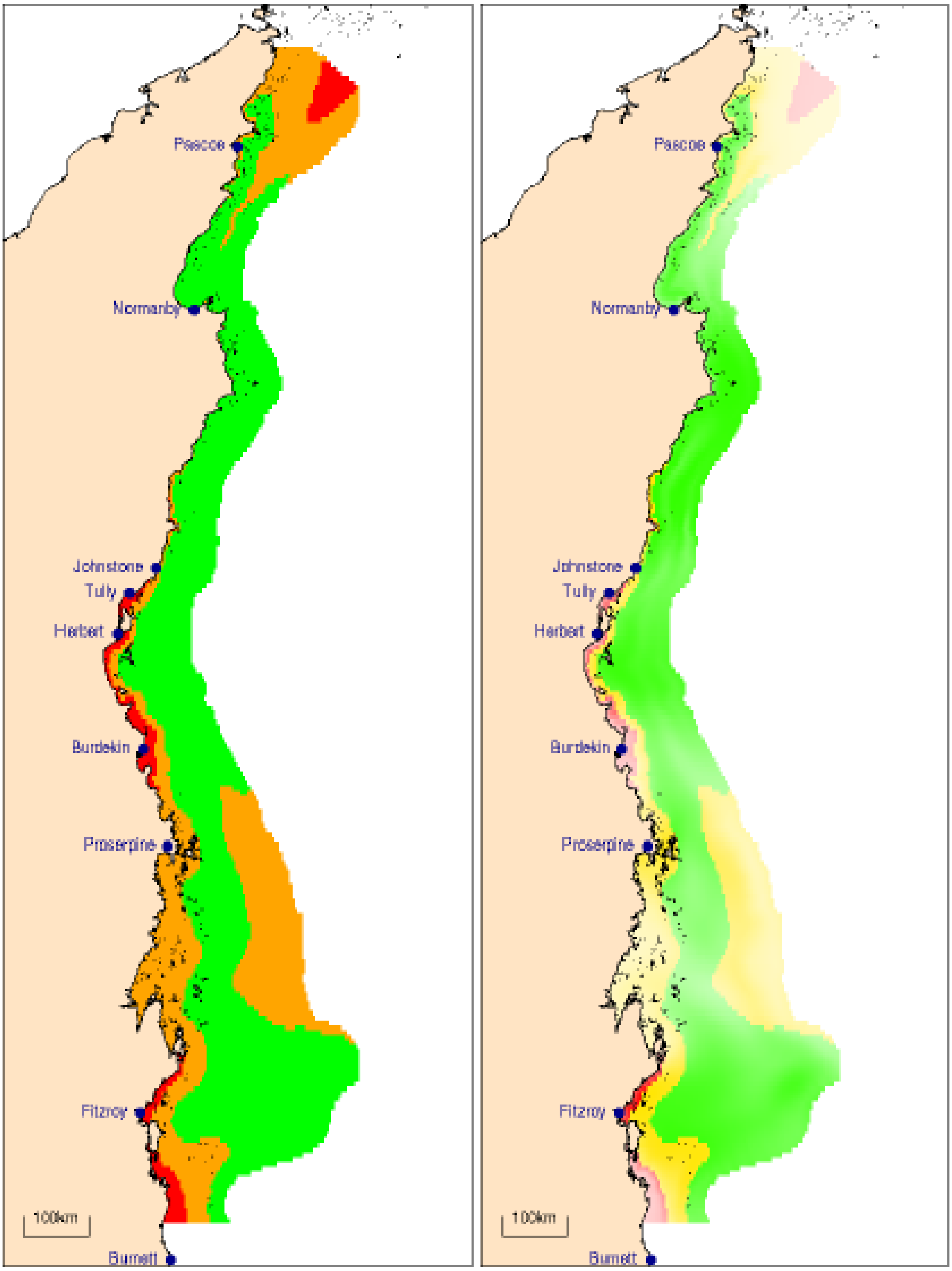

Figure 3.6. Annual mean chlorophyll-a concentrations (1988–2006 dataset.

Locations that are less than (green) or exceed (orange and red) the water quality guideline trigger value of a maximum annual mean of 0.45 µg/L chlorophyll. Orange zones show areas that exceed the guideline trigger values, having chlorophyll values of 0.45 – 0.8 µg/L. Red zones show areas of greatest concern with >0.8 µg/L chlorophyll. The level of fading (right panel) indicates the level of confidence in the estimates with faded areas being more uncertain. Source: De'ath & Fabricius (2008).

DIN plume loading (mean 2003–2013)

As shown in Table 2.1, five assessment classes (Very Low to Very High) were used for DIN plume loading based on an interpolated map derived from plume frequency information from remote sensing and scaled river load data (Brodie et al. 2015; da Silva et al. in prep.). The results of the assessment are shown in Table 3.5 and Figure 3.6.

The DIN plume loading area of influence is relatively constrained and only the Low and Very Low assessment classes extend beyond the inshore area. The majority of Keppel Bay, and the Keppel Island Group are in the High assessment class. There are no coral reefs or seagrass (surveyed or deepwater modelled) in the Very High assessment class; however, 66% of the surveyed seagrass is in the High assessment class, and these are mostly located in Shoalwater Bay. However, the results in Shoalwater Bay are likely to be influenced by shallow bathymetry in this area, which is to derive a concentration (calculated from load maps of mass per unit area and bathymetry). Approximately 98% of coral reefs and deepwater modelled seagrass are located in the Very Low or Low assessment classes.

Data confidence: Low/moderate. Very low confidence in Shoalwater Bay.

Table 3.5. Area of coral reefs and seagrass meadows within the Very Low to Very High assessment classes for DIN Plume loading and the percentage of the total habitat area in the region in each class.

DIN loading	Very Low		Low		Moderate		High		Very High		Total
μg/L	<1		1-2		2-7		7-25		>25		
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	71336	82.1	10201	11.74	3716	4.3	1480	1.7	127	0.1	86860
Coral Reefs	4506	91.8	323	6.58	50	1.0	30	0.6	0	0.0	4910
Seagrass											
Composite survey	0	0.0	21	8.2	65	25.6	169	66.2	0	0.0	255
Deepwater	2970	52.4	2613	46.1	84	1.5	0	0.0	0	0.0	5667
Total Seagrass	2970	50	2634	44	149	3	169	3	0	0	5922

The assessment classes are based on mean wet season DIN loading concentrations 2003 to 2013 (see methods in Section 2.3.2).

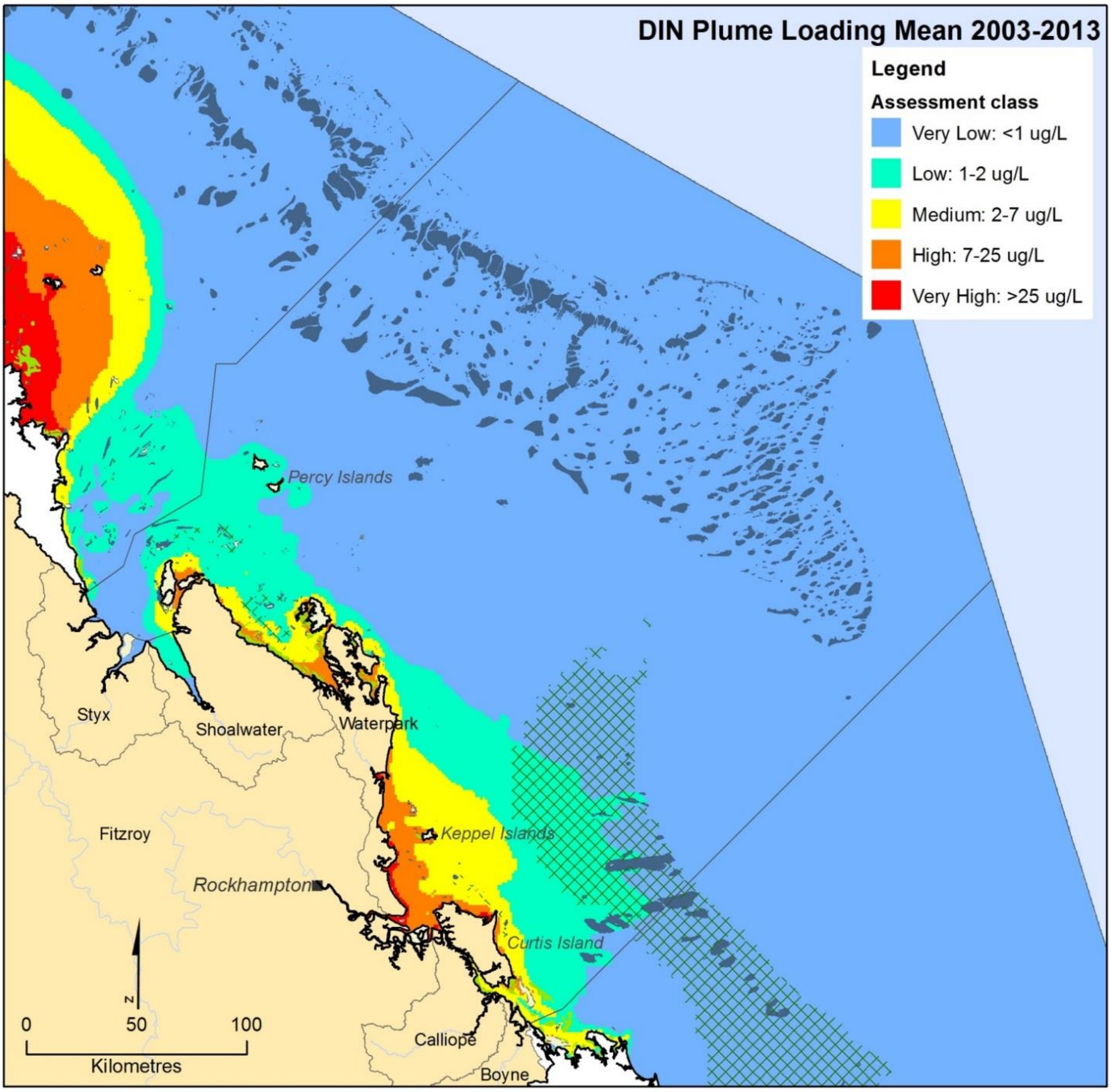

Figure 3.7. Results for the assessment of DIN plume loading (mean of annual assessments 2003 to 2013).

The assessment classes are concentration estimates derived from an interpolation of a multi-year analysis that combines scaled river loads data and flood plume frequency analysis from satellite imagery (see methods in Section 2.1.2). Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

PN plume loading (mean 2007–2011)

As shown in Table 2.1, three assessment classes (Low, Moderate, High) were used for PN plume loading based on an interpolated map derived from plume frequency information from remote sensing and scaled river load data (Brodie et al. 2015; da Silva et al. in prep.). The results of the assessment are shown in Table 3.6 and Figure 3.7.

The PN plume loading area of influence is relatively constrained to the coast and only the Low assessment class extends beyond the inshore area. The majority of Keppel Bay, including the Keppel Island Group, is in the High assessment class. Nearly all (99%) of coral reefs are located in the Low assessment class; however, there are 44 km^2 in the High assessment class. Almost 80% of the surveyed seagrass is in the High assessment class, with a large proportion of this within Shoalwater Bay. However, as noted for DIN loading, the results in Shoalwater Bay are likely to be influenced by shallow bathymetry in this area, which is to derive a concentration (calculated from load maps of mass per unit area and bathymetry). Nearly all (99%) of the deepwater modelled seagrass is in the Low assessment class.

Data confidence: Low/moderate. Very low in Shoalwater Bay.

Table 3.6. Area of coral reefs and seagrass meadows within the Low to High assessment classes for PN plume loading and the percentage of the total habitat area in the region in each class.

PN Plume Loading	Low		Moderate		High		Total
µg/L	<10		10-20		>20		
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	82205	94.6	2302	2.7	2353	2.7	86860
Coral Reefs	4843	98.7	22	0.4	44	0.9	4910
Seagrass							
Composite survey	33	12.8	20	7.8	202	79.5	255
Deepwater	5622	99.2	36	0.6	9	0.2	5668
Total Seagrass	5655	95	56	1	212	4	5922

The assessment classes are based on mean wet season PN loading concentrations 2003 to 2013 (see methods in Section 2.3.2).

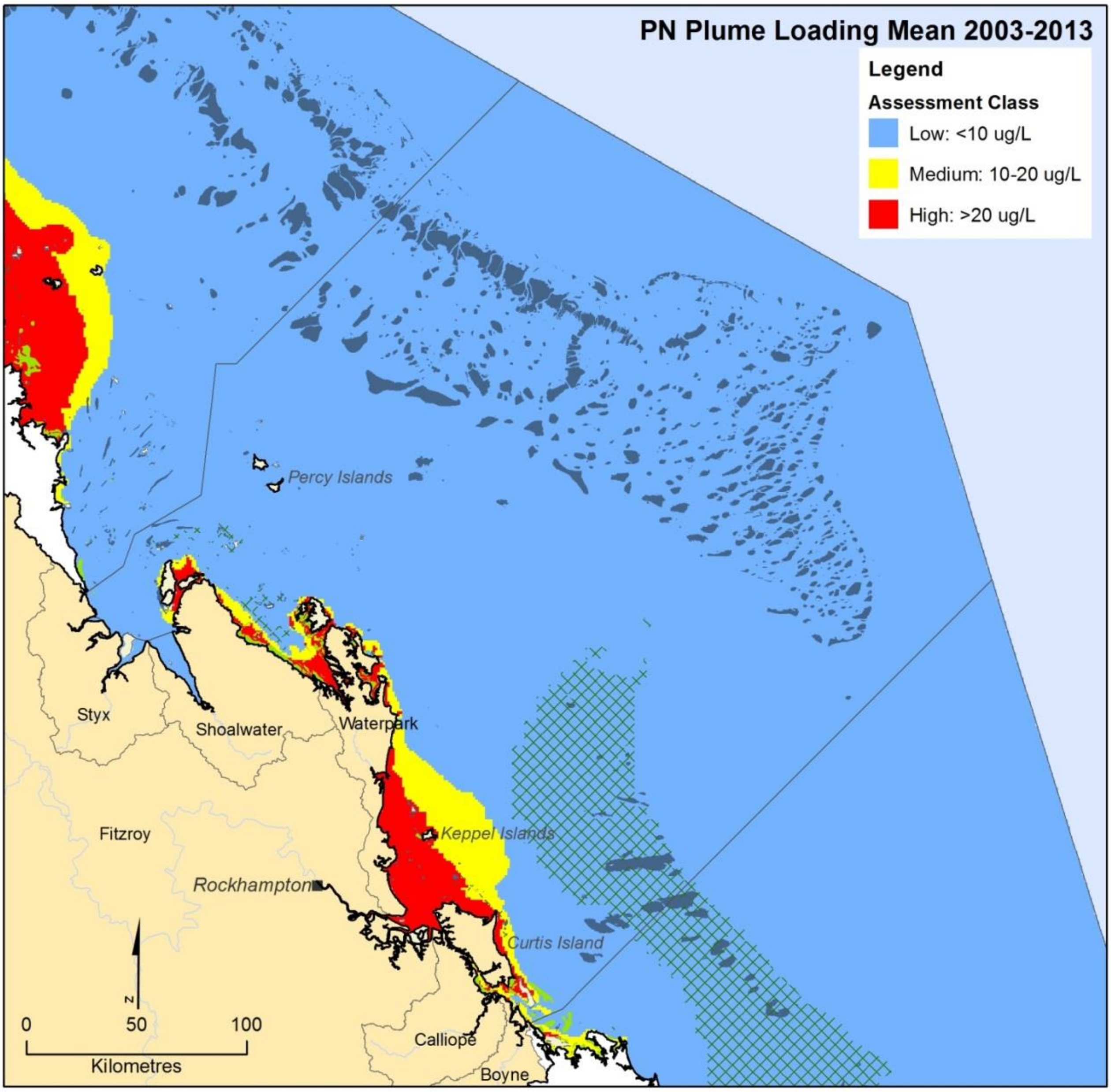

Figure 3.8. Results for the assessment of PN plume loading (mean of annual assessments 2003 to 2013).

The assessment classes concentration estimates derived from an interpolation of a multi-year analysis that combines scaled river loads data and flood plume frequency analysis from satellite imagery (see methods in Section 2.1.2). Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

c) PSII Herbicides

PSII Herbicide modelled concentration, 2009–2011

As shown in Table 2.1, six assessment classes were used for PSII herbicides based on the toxicity of diuron calculated in several studies on coral and seagrass species (see Lewis et al. 2013) ranging from No Risk to Very High risk. These were then used for assessing the results of an estimate of the relationship between additive PSII herbicide concentrations and CDOM (salinity proxy) in flood plume conditions; see Section 2.1.3). The results of the assessment are shown in Table 3.7 and Figure 3.8.

All of the marine areas in the Fitzroy region are in the Low, Very Low or No Risk assessment class. The areas within the Low assessment class extend from the Fitzroy River mouth into the southern areas of Keppel Bay but only includes ~6 km^2 of coral reefs. A majority of the surveyed and deepwater modelled seagrass are in the No risk assessment class.

Data confidence: Low/moderate due to limited availability of PSII herbicide concentration data in rivers and the marine environment during flood events.

Table 3.7. Area of coral reefs and seagrass within the No risk to Very High risk assessment classes for exposure to PSII herbicides in 2009–2011 and the percentage of the NRM region that the area represents.

PSII Herbicide modelled concentration	No Risk		Very Low		Low		Moderate		High		Total
µg/L	<0.025 µg/L		0.025-0.1 µg/L		0.1 - 0.5 µg/L		0.5 - 2.3 µg/L		2.3 - 10 µg/L		
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)
Total Area	84789	97.6	1318	1.52	753	0.9	0	0.0	0	0.0	86860
Coral Reefs	4824	99.2	32	0.66	6	0.1	0	0.0	0	0.0	4910
Seagrass											
Composite survey	248	97.3	7	2.7	0	0.0	0	0.0	0	0.0	255
Deepwater	5598	99.9	7	0.1	0	0.0	0	0.0	0	0.0	5605
Total Seagrass	5847	99	14	1	0	0	0	0	0	0	5922

Results for the assessment are based on the exposure assessment undertaken by Lewis et al. (2013) and described in Section 2.3.3.

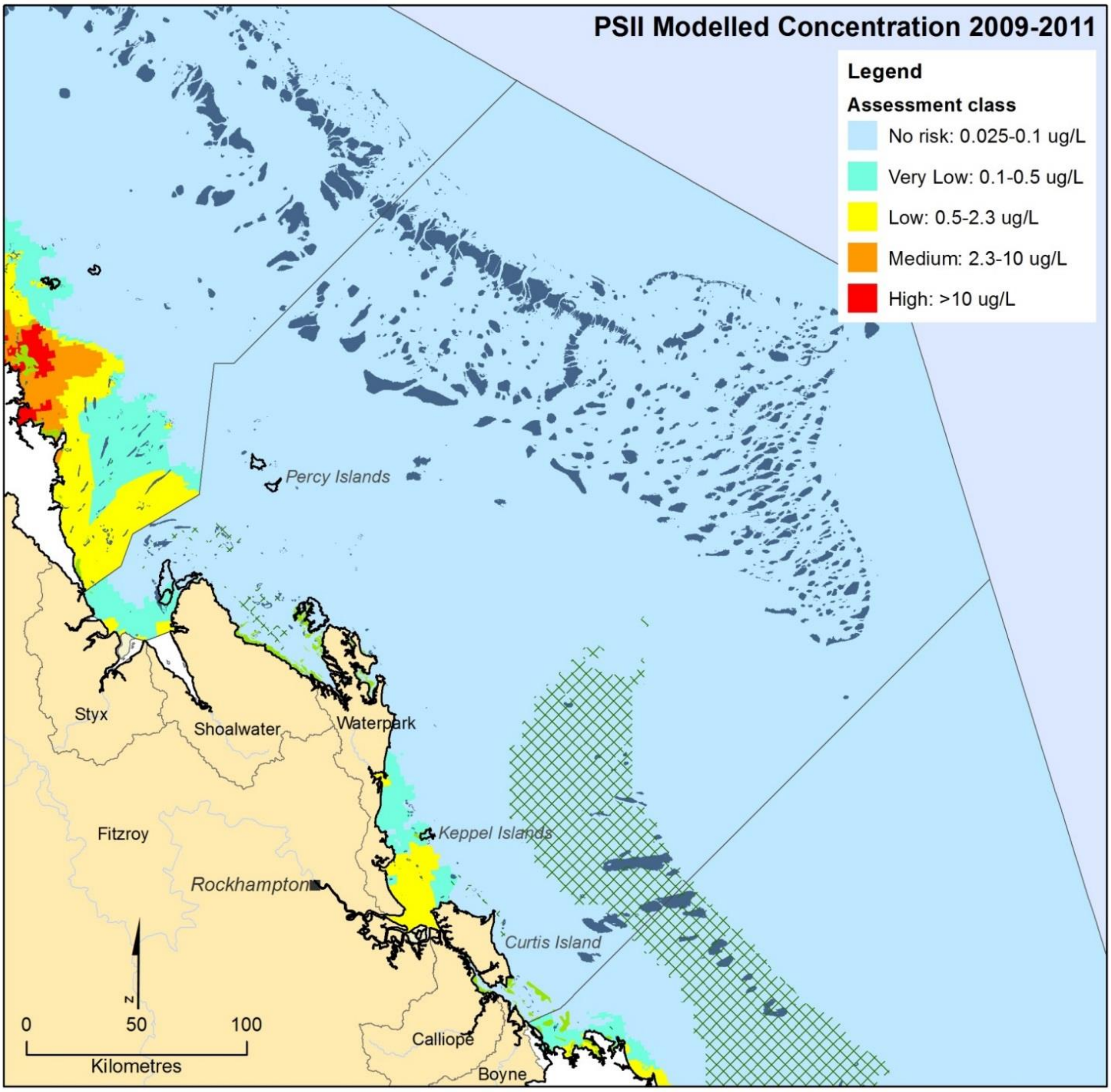

Figure 3.9. Results for the assessment of exposure to PSII herbicides.

Results for the assessment are based on the exposure assessment undertaken by Lewis et al. (2013) and shown in Table 2.1. Based on an estimate of the relationship between additive PSII herbicide concentrations (2010–11 and 2011–12 water years) and CDOM (salinity proxy) in flood plume conditions (2012–13 wet season). Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

d) Relative differences between pollutants

The assessment of individual variables presented above (a — sediment, b — nutrients and c — PSII herbicides) can be used to guide priorities for management of individual pollutants in the Fitzroy region to some extent, but should be used in conjunction with further expert opinion and local technical expertise due to the data limitations noted above for each variable.

Table 3.8 summarises the area of coral reefs and seagrass in the Very High and High assessment classes for each variable. In summary:

- The greatest proportion of coral reefs in the Very High and High assessment classes is for Chl-a; for the remaining variables less than 3% of the regional area of coral reefs is in these classes. However, the data for Chl-a is presented as a long-term average, and therefore does not factor in frequency of exceedance of the guideline value. When the Chl-a results are excluded from the assessment, the greatest area of the highest risk categories are for TSS 2mg/L exceedance, suggesting that TSS has a dominant influence in the region.
- The proportion of surveyed seagrass area in the Very High and High assessment classes is greater than 66% and up to 100% for all sediment and nutrient variables. A large proportion of this seagrass is located in Shoalwater Bay.
- The proportion of deepwater modelled seagrass in the Very High and High assessment classes is less than 5% for all variables.

Table 3.8. Area of coral reefs and seagrass within the Very High and High assessment classes for each variable and the percentage of the NRM region that the area represents.

	Sediment				Nutrients						Pesticides	
	TSS 2mg/L exceedance		**TSS loading**		**Chl-a mean concentration**		**DIN loading**		**PN loading**		**PSII herb modelled concentration**	
	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region	Area (km^2)	% of habitat in region
Total Area	5358	6%	2107	2%	31358	36%	1607	2%	2353	3%	0	0%
Coral Reefs	130	3%	42	1%	1707	35%	30	1%	44	1%	0	0%
Seagrass												
Composite survey	223	88%	185	73%	255	100%	169	66%	202	79%	0	0%
Deepwater	252	4%	6	0%	45	1%	0	0%	9	0%	0	0%
Total Seagrass	475	8%	191	3%	300	5%	169	3%	212	4%	0	0%

Table 3.9 summarises the 'hotspot' areas for each variable, and each map is also shown in a panel in Figure 3.9 for comparison. This highlights that the water quality influence in the region is generally constrained to the inshore areas, with hotspot areas in Shoalwater Bay and Keppel Bay for sediments, and Keppel Bay for nutrients. However, as noted above, the sediment influence in Shoalwater is not believed to be linked to river discharge, and the area is naturally turbid due to shallow and large tidal variation. The influence of PSII herbicides does not appear to extend in the marine environment to any significant extent, supported by monitoring data where tebuthiuron was the only pesticide that exceeded the Water Quality Guidelines at a North Keppel Island routine monitoring site in 2012–13 and was below the Guidelines in 2013–14 (Gallen et al. 2014).

Table 3.9. Summary of the potential hotspot areas for highest relative risk for each variable in the Fitzroy NRM region.

Variable	Hotspot areas
Sediment	
TSS threshold exceedance 2mg/L (% valid observations)	Extends along the coastal areas, with concentrated areas in Broad Sound, Shoalwater Bay and then heading south from the Water Park Creek mouth to Curtis Island, including Keppel Bay, and south beyond the NRM region boundary. The exceedances in Broad Sound and Shoalwater Bay are likely to be naturally occurring rather than driven by river discharge (Logan et al., in press).
TSS Plume Loading (mg/L) (mean 2003-2013)	Relatively constrained and only the Low and Very Low assessment classes extend beyond the inshore area. The areas of greatest influence are in Shoalwater Bay and the coastal areas of Keppel Bay.
Nutrients	
Chl long term concentration (µg/L) (mean 1988-2006)	Within Shoalwater Bay and in a band approximately 30 kilometres wide along the coast from Townshend Island to the southern end of the NRM region boundary. This incorporates Keppel Bay and the Keppel Island Group.
DIN Plume Loading (µg/L) (mean 2003-2013)	Relatively constrained and only the Low and Very Low assessment classes extend beyond the inshore area. Shoalwater Bay, Keppel Bay and the Keppel Island Group are in the High assessment class.
PN Plume Loading (µg/L) (mean 2003-2013)	Relatively constrained to the coast and only the Low assessment class extends beyond the inshore area. The majority of Keppel Bay, including the Keppel Island Group, is in the High assessment class.
Pesticides	
PSII Herbicide modelled concentration (µg/L) (2009-2011)	All of the marine areas in the Fitzroy region are in the Low, Very Low or No Risk assessment class. The areas within the Low assessment class extend from the Fitzroy River mouth into the southern areas of Keppel Bay, but only includes ~6 km^2 of coral reefs. A majority of the surveyed and deepwater modelled seagrass are in the No risk assessment class.

Sediment **Nutrient** **Particulate N (top) and PSII herbicides (bottom)**

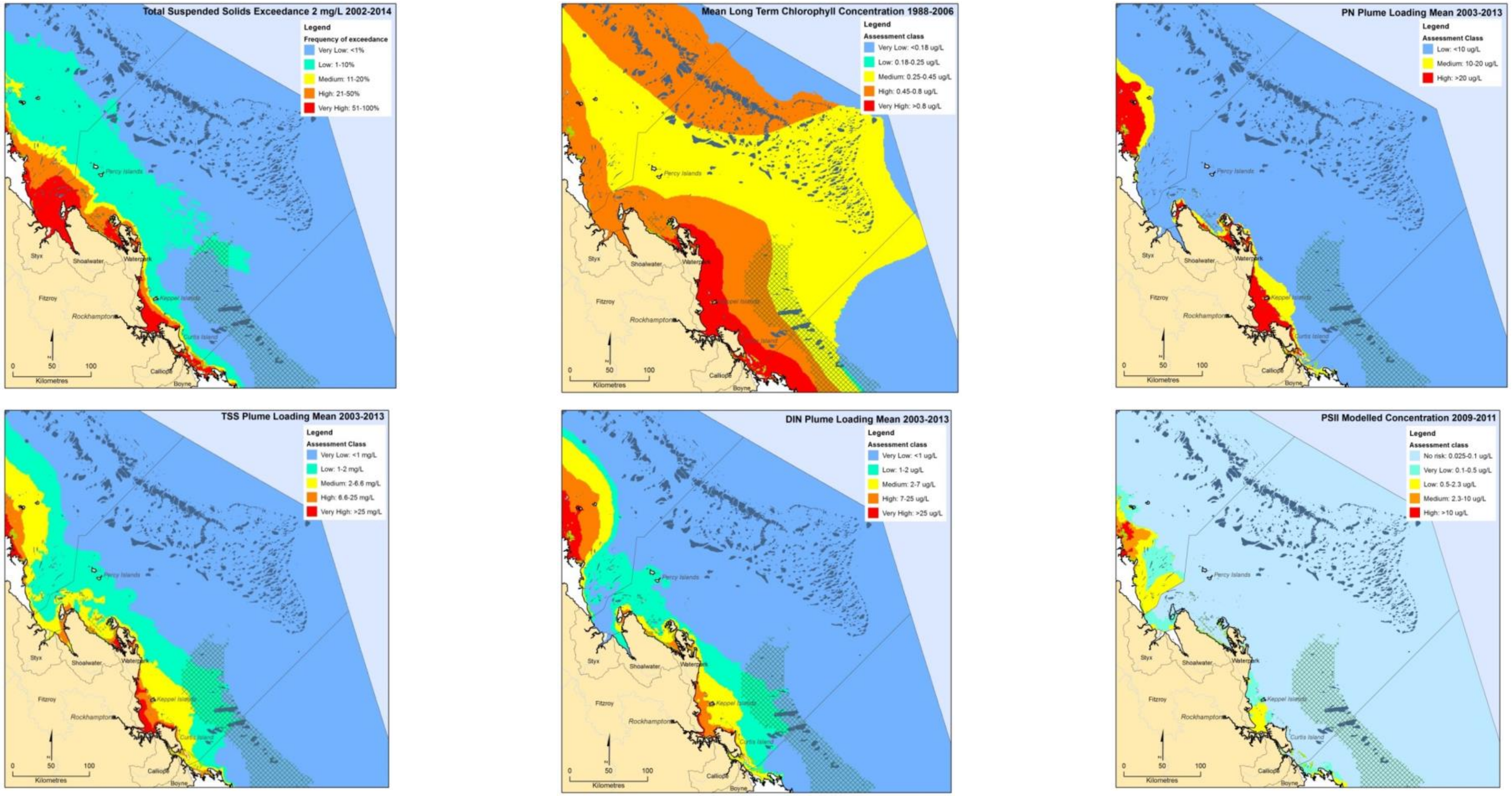

Figure 3.10. Results of all variables presented for comparison and identification of the areas of highest relative risk from individual variables in the Fitzroy NRM region.

3.3 Marine Risk Index: Combined risk of degraded water quality to coral reefs and seagrass

The combined assessment takes into account all assessment classes for each variable to identify the areas of highest relative risk to degraded water quality in the Fitzroy region, and hence where coral reefs and seagrass are most likely to be under pressure from degraded water quality.

As described in Section 3, five assessment classes were used for the combined assessment of relative risk ranging from Very Low to Very High. The results of the assessment are shown in Table 3.10 and Figure 3.10. The key findings are:

- Overall, the greatest area of influence from degraded water quality is in Keppel Bay.
- The areas in the Very High relative risk class are concentrated in the inner area of Shoalwater Bay and in Keppel Bay, from the coastal areas from Corio Bay (the mouth of Water Park Creek) to the northern end of Curtis Island. This area is locally influenced by the Fitzroy River and Water Park Creek. However, as described for individual variables, it is unlikely that the results in Shoalwater Bay are driven by anthropogenic influences and are more likely associated with natural tidal conditions and depth.
- The High relative risk class extends beyond the Very High area to include the Keppel Island Group, a greater area of Shoalwater Bay and the north-east part of Broad Sound. However, the results in Shoalwater Bay and Broad Sound are considered to be highly uncertain due to the shallow conditions and naturally highly turbid conditions.
- The combined area of the Very High and High relative risk classes (2,482 km^2) incorporates less than 1% (51 km^2) of the coral reefs in the region but 84% (213 km^2) of the surveyed seagrass in the region. There is no deepwater modelled seagrass in these areas (the area is relatively shallow).
- The Moderate relative risk class extends beyond the High area but is still contained within the inshore area and within approximately 30 kilometres of the coast at the most extensive point beyond the Keppel Islands (approximately 60 kilometres from the Fitzroy River mouth). The northern boundary of the Moderate relative risk class is most likely strongly influenced by the boundaries of the PSII herbicide assessment, which was conducted within individual marine NRM regions, thereby showing an unrealistic boundary of the Moderate risk areas in the northern part of the region.
- The Low relative risk class extends in a band approximately 40 to 70 kilometres from the coast, and incorporates the Percy Island Group. This area contains approximately 3% of the region's coral reefs, and 38% of the deepwater modelled seagrass. Only 5% of the surveyed seagrass is in this area.
- The Very Low relative risk class extends to the outer boundary of the GBR Marine Park and incorporates a majority (80%) of the area of coral reefs, 60% of the deepwater modelled seagrass, and none of the surveyed seagrass.

Table 3.10. Estimated area of coral reefs and seagrass meadows within the five relative risk classes in the Fitzroy NRM region.

Habitat	Very Low		Low		Moderate		High		Very High		High + Very High	
	Area (km^2)	% region	Area (km^2)	% region	Area (km^2)	% region	Area (km^2)	% region	Area (km^2)	% region	**Area (km^2)**	**% region**
Total Area	69306	79.8	11862	13.7	3210	3.7	1487	1.7	995	1.1	**2482**	**3%**
Coral Reefs	4632	94.3	152	3.1	74	1.5	42	0.9	9	0.2	**51**	**1%**
Seagrass												
Composite survey	0	0.0	13	5.0	28	11.0	163	64.2	50	19.7	**213**	**84%**
Deepwater	3422	60.4	2126	37.5	113	2.0	7	0.1	0	0.0	**7**	**0%**
Total Seagrass	3422	58	2139	36	141	2	170	3	50	1	**220**	**4%**

The % region is the proportion of the habitat in the region for each assessment class.

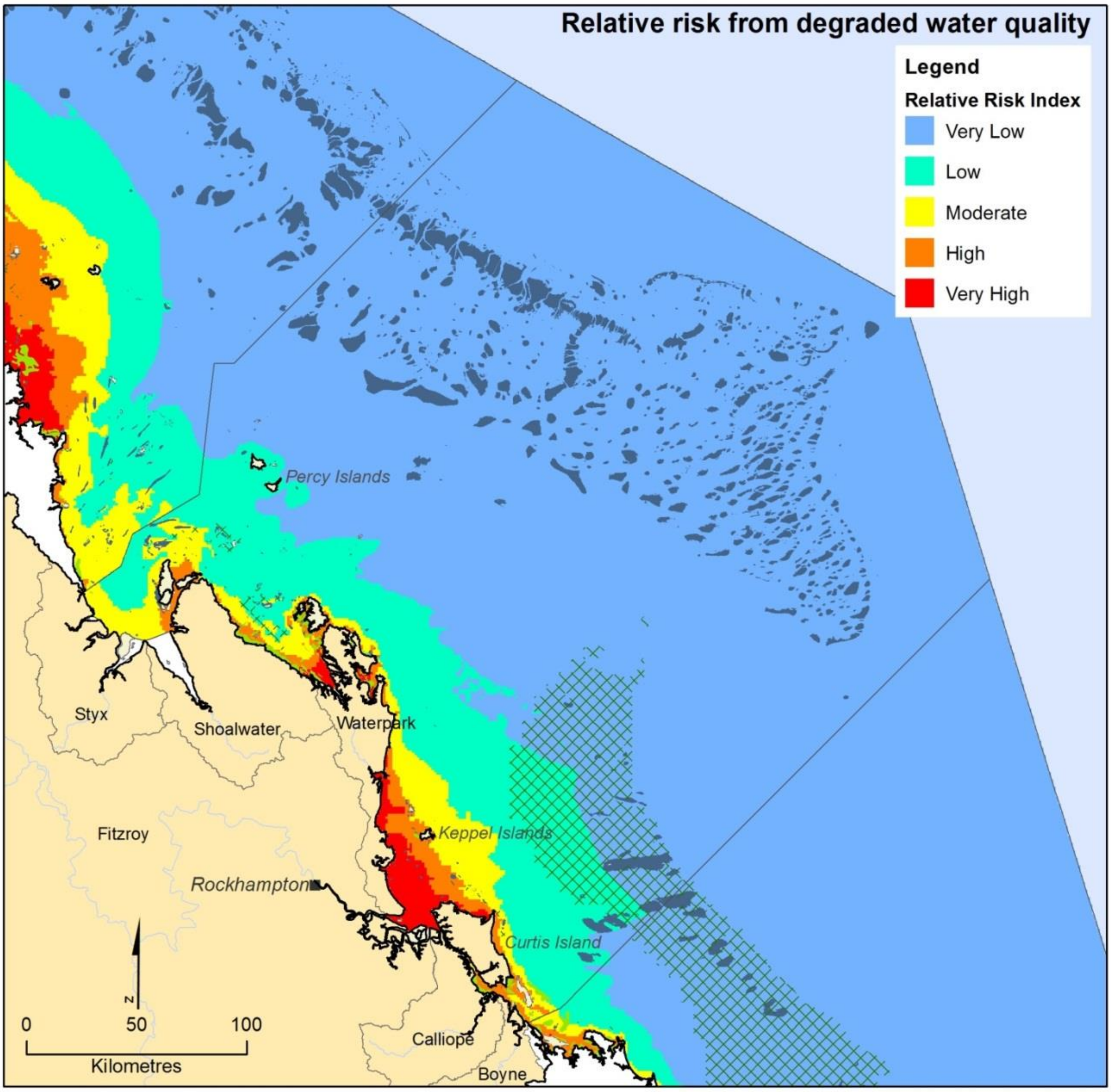

Figure 3.11. Combined assessment (1 km2 resolution) of the relative risk of water quality variables. The areas (in km2) of habitat types within each class are shown in Table 3.7. Reefs are shown in blue, surveyed seagrass (composite as at June 2010) shown in light green, deepwater modelled seagrass (>15m, 50% probability) shown in green hatch.

The assessment can also be conducted for each river zone of influence (Table 3.11). While the dominant influence of the Fitzroy River is important, this assessment is useful to identify whether there are relative differences between the Water Park, Calliope and Boyne rivers. The key findings are:

- The modelled zone of influence for the Fitzroy River extends along the coast of the whole region, approximately 50 to 70 kilometres outwards from the coast, and north to the Whitsunday Island Group. This area contains extensive potential seagrass habitat (419 km^2)

and coral reefs (559 km^2), and overlaps with influence from the rivers in the Mackay-Whitsunday region. It is estimated that 93% of the seagrass areas and 26% of the coral reefs in the ZoI are within the High and Very High relative risk classes.

- Using this method, the relative differences between the Water Park, Calliope and Boyne rivers are not significant, and given the uncertainties in the data, are considered to be similar for coral reefs. However, the results indicate that the Calliope and Boyne rivers may have a greater influence on potential seagrass habitat than Water Park Creek; however, anthropogenic influences are important to distinguish and are considered in the Loads Index, Section 3.4.
- It is also important to note the influence of the rivers in the Burnett-Mary region in the Fitzroy region during flood conditions, such as those documented in the 2010–2011 and 2012–2013 wet seasons (da Silva et al. 2013). In 2010–2011 when both the Fitzroy and Burnett-Mary region rivers were experiencing extensive flood conditions at the same time, the Mary River plume extended to the mid and outer shelf reefs in the Fitzroy region and analysis of plume dynamics showed that the Fitzroy River plume steered north and Mary River plume deflected to the mid and outer reef areas (da Silva et al. 2013; Devlin et al. 2013b). Figure 3.12 provides an example of the extension of these waters in a northerly direction, and further offshore.
- The zone of influence estimated for the Burnett River (path-distance model) extended to the northern point of Curtis Island in the 2011 and 2013 wet seasons and incorporated the area of Port Curtis (Figure 3.13).

Table 3.11. Area of coral reefs and seagrass meadows within the five relative risk classes in each river zone of influence.

Basins and habitat	**Area (km²)**							**Based on area VH+H**
Coral Reefs	**V Low**	**Low**	**Moderate**	**High**	**V High**	**Total**	**High & V High**	**Marine Risk Index**
Water Park	0	1	1	17	7	26	24	0.17
Fitzroy	61	225	127	101	44	559	146	1.00
Calliope	0	0	8	11	0	19	11	0.08
Boyne	0	0	11	8	0	20	8	0.06
Total	**61**	**226**	**148**	**138**	**51**	**624**	**189**	
						Max	*146*	
Seagrass (surveyed)	**V Low**	**Low**	**Moderate**	**High**	**V High**	**Total**	**High & V High**	**Marine Risk Index**
Water Park	0	0	1	8	3	13	12	0.03
Fitzroy	0	0	29	191	198	419	389	1.00
Calliope	0	13	22	44	2	80	46	0.12
Boyne	0	21	47	47	5	119	52	0.13
Total	**0**	**33**	**99**	**290**	**209**	**631**	**498**	
						Max	*389*	
Seagrass (deepwater modelled)	**V Low**	**Low**	**Moderate**	**High**	**V High**	***Total***	**High & V High**	**Marine Risk Index**
Water Park	0	0	3	5	0	8	5	0.03
Fitzroy	522	2037	127	154	22	2863	176	1.00
Calliope	0	0	38	7	0	45	7	0.04
Boyne	0	6	24	0	0	30	0	0.00
Total	**522**	**2043**	**192**	**165**	**22**	**2945**	**188**	
						Max	*176*	

The sum of the area within the High and Very High classes form the Risk Index, which compares all summed areas to the maximum area, which is given a score of 1. The highest scores are shaded in red.

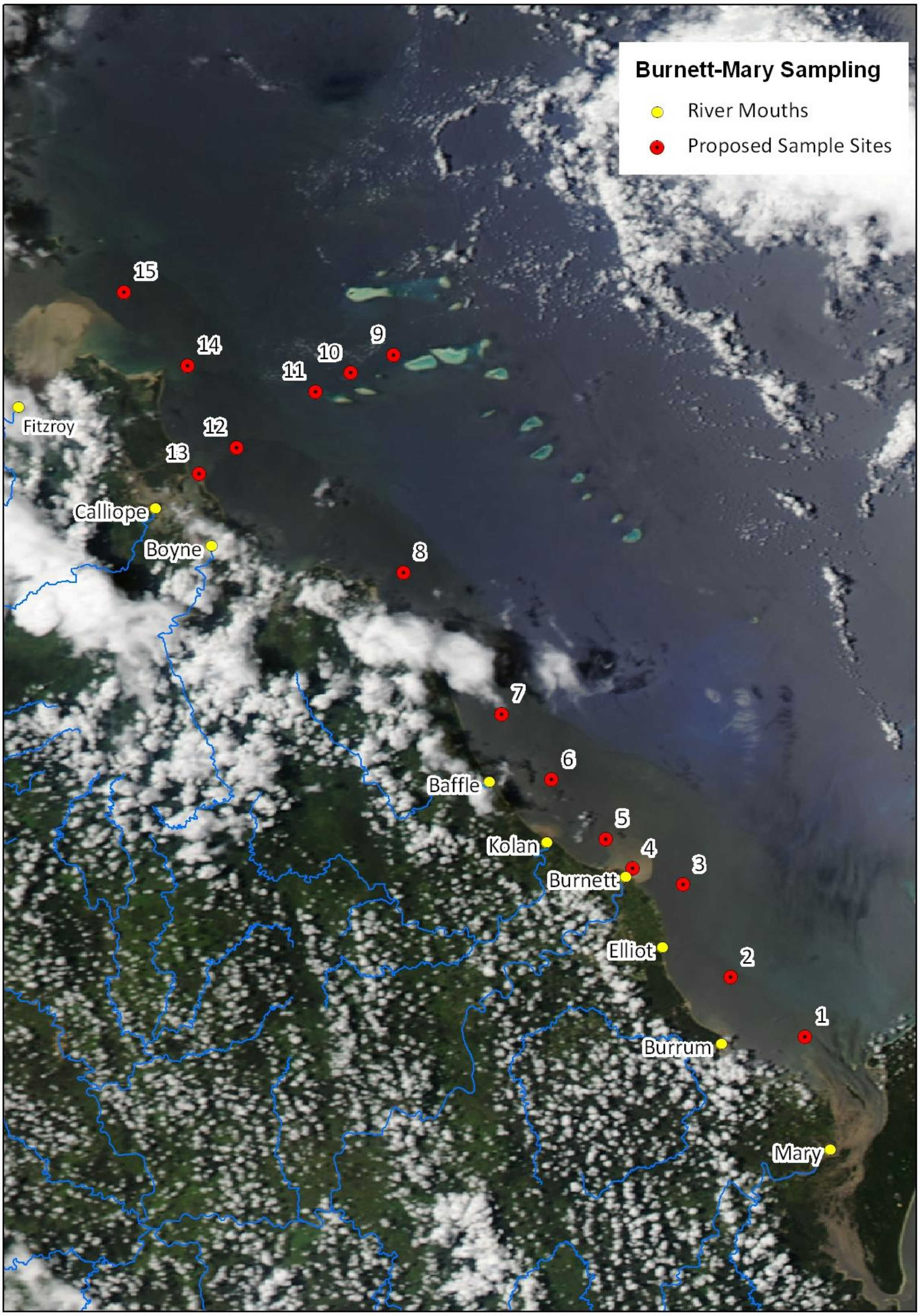

Figure 3.12. Burnett and Mary rivers' plumes in 2010–2011

Numbers show water quality sampling points showing extension of these plume waters north.

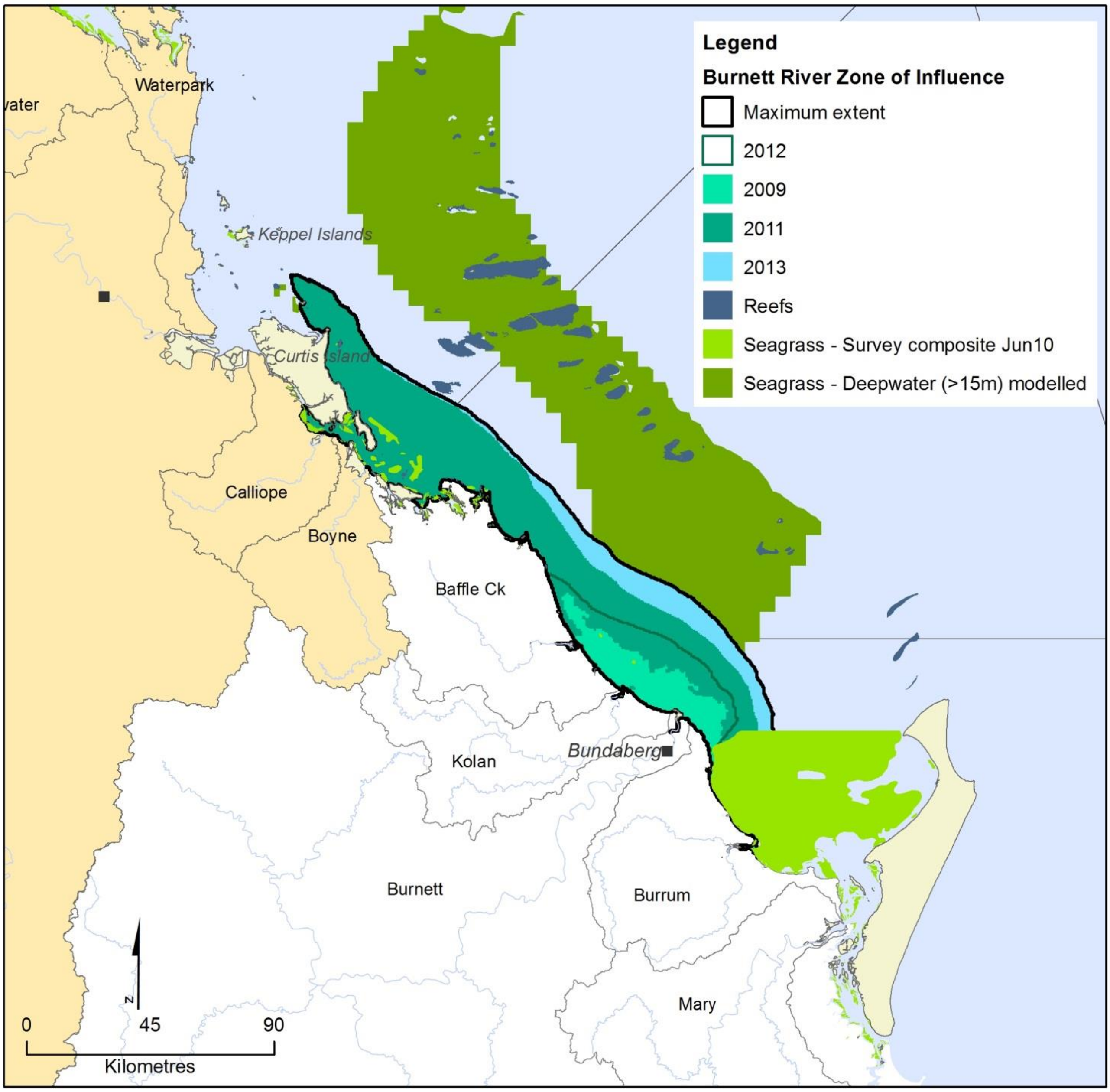

Figure 3.13. Modelled (path-distance) Zone of influence for the Burnett River zone of influence — 2009, 2011, 2012, 2013.

See Section 2.2.1 for the method.

Results for important habitat features in the Very High to Low relative risk areas of the Fitzroy region are summarised in Table 3.12. The areas in the Very Low relative areas are not considered here.

While the areas of coral reef and seagrass within the highest assessment classes for individual variables and the Marine Risk Index are relatively small, they often include highly valued tourism and recreation sites of the GBR. Examples include the Keppel Island Group and Curtis Island.

Table 3.12. Results of the relative risk assessment for important habitat features in the Very High to Low areas of the Fitzroy region.

Habitat Feature	Description	Relative risk results	Likely rivers of influence
Northumberland Island group (northern inshore areas)	Contains two main island groups, fringing coral reefs and shoals.	Moderate	Fitzroy
Percy Islands	Island group contains fringing coral reefs.	Low	Fitzroy
Broad Sound	Limited coral reefs and seagrass beds and is naturally highly turbid due to large tidal ranges and is relatively shallow.	Moderate to Low	Fitzroy (limited)
Shoalwater Bay	Extensive intertidal seagrass beds, Ramsar wetland, and is protected by the Shoalwater Dugong Protected Area.	Very High in the innermost areas, with a gradient to Very Low risk in the outer part of the bay. However, as described above, the water quality conditions are unlikely to be driven by anthropogenic influences.	Fitzroy (limited)
Keppel Island group	Fringing (inshore) coral reefs, intertidal seagrass beds and island habitats.	High	Fitzroy Water Park (predominantly constrained to North Keppels) Calliope (predominantly constrained to southern-outer areas)
Keppel Bay (coastal areas)	Balaclava Island listed on Register of National Estate, naturally high turbidity with limited coral reefs and seagrass beds but contains important coastal wetlands.	Very High in the coastal areas, shifting to High and then Moderate in the outer limits of the bay.	Fitzroy

Habitat Feature	Description	Relative risk results	Likely rivers of influence
Curtis Island	Fringing coral reefs on south eastern coast, surveyed seagrass at southern end, wetland areas.	Very High and High.	Calliope Boyne Fitzroy (predominantly northern areas only)
Capricorn Group	Mid-shelf coral reefs and deepwater modelled seagrass.	Low for reefs located closest to the coast including Rock Cod Shoal, Irving Reef, Polmaise Reef and Mast Head Island and reefs. Very Low elsewhere.	Fitzroy Potentially Burnett River in flood events eg. 2010–11
Rodds Bay Dugong Protection Area (across the southern boundary)	Extensive intertidal seagrass beds and fringing coral reefs on the eastern coastal of Facing Island.	This area is influenced by Gladstone Harbour and Calliope and Boyne River mouths.	Calliope Boyne Burnett (outside of this region) Fitzroy (limited)

This combined assessment of water quality variables can be used to guide overall management priorities for addressing the risks from degraded water quality to coral reefs and seagrass between Fitzroy basins in conjunction with information about catchment pollutant load delivery. This information is presented in the following section. However, it is important to recognise that the input variables represent longer term time series, and in most cases, represent average conditions. The response of coral reef and seagrass ecosystems to conditions in individual flood events, and the influence of repeated years of flood conditions are also important. These aspects are discussed further in Section 4.

3.4 Loads Index: Assessment of end-of-catchment pollutant loads

The pollutant load information allows managers to relate the Marine Risk results to management priorities among basins and land uses. Further analysis of basin pollutant loads can be undertaken for TSS, DIN, PN, DIP, PP and PSII herbicides including comparisons of the total and anthropogenic load contributions from each basin to the total regional loads. The data is derived from the report of the Source Catchments modelling for the Fitzroy region, prepared by Dougall et al. (2014), using the baseline (2008) annual average loads.

It is recognised that the concentrations of PSII herbicides are more ecologically relevant than loads for GBR ecosystems. The pollutant load helps to understand the ecological implications of a pollutant in the environment. Hydrodynamics, both the status of the water body receiving the load and the biogeochemical process play an important role in determining if that load will or will not be of any harm to the environment. For determining risk to aquatic biota from PSII herbicides, assessing toxic effects from concentration data normalised to represent 'additive' PSII herbicide toxicity is a more ecologically relevant method than an assessment of the PSII load transported to the marine environment generated through the Source Catchments model. The Source Catchments model is based on an annual average load of the total sum of the five common PSII herbicides (i.e. diuron, atrazine, ametryn, hexazinone and tebuthiuron) and does not consider the differences in toxicity between these herbicides.

However, the PSII modelled load does provide an indication of the contribution of PSII herbicides from each basin based on an 'average' year, i.e. a long-term average that adjusts for extreme weather conditions. While our risk analysis includes the toxic PSII concentration based on monitoring data from the Fitzroy River, we have limited concentration data that is comparable between the basins and so our risk scores are likely to be on the conservative side.

The estimated pollutant loads for the basins in the Fitzroy are shown in Table 3.13 and Table 3.14, and graphed in Figures 3.13 to 3.18.

Data confidence: Moderate due to considerable and ongoing improvements to the Source Catchments modelling. The available monitoring data in the Fitzroy region shows good correlation with the end-of-catchment monitoring data.

Table 3.13. Total and anthropogenic loads for TSS, DIN and PSII herbicides from Fitzroy basins, and as percentages of the total regional load and regional anthropogenic load.

TSS loads ($kt.y^{-1}$)

Basin Name	Pre-Development Load	Total Load (2008)	Anthropogenic load (2008)	Anthropogenic load % of Regional Total Load	Ranking
Styx	28	68	40	2.1	2
Shoalwater	27	53	26	1.3	4
Water Park	27	32	5	0.3	6
Fitzroy	440	1740	1300	66.7	1
Calliope	16	44	28	1.4	3
Boyne	3	11	8	0.4	5
Regional total	**542**	**1948**	**1407**	**72.2**	

DIN loads ($t.y^{-1}$)

Basin Name	Pre-Development	Total Load (2008)	Anthropogenic load (2008)	Anthropogenic load % of Regional Total Load	Ranking
Styx	38	38	0	0.0	2
Shoalwater	45	45	0	0.0	2
Water Park	54	54	0	0.0	2
Fitzroy	1057	1106	48	3.8	1
Calliope	23	23	0	0.0	2
Boyne	6	6	0	0.0	2
Regional total	**1223**	**1272**	**49**	**3.9**	

PSII toxic equivalent loads ($kg.y^{-1}$)

Basin Name	Pre-Development	Total Load (2008)	Anthropogenic load (2008)	Anthropogenic load % of Regional Total Load	Ranking
Styx	0	0.0	0.0	0.0	3
Shoalwater	0	0.0	0.0	0.0	3
Water Park	0	0.0	0.0	0.0	3
Fitzroy	0	119.4	119.4	99.92	1
Calliope	0	0.0	0.0	0.0	3
Boyne	0	0.1	0.1	0.08	2
Regional total	0	119.5	119.5		

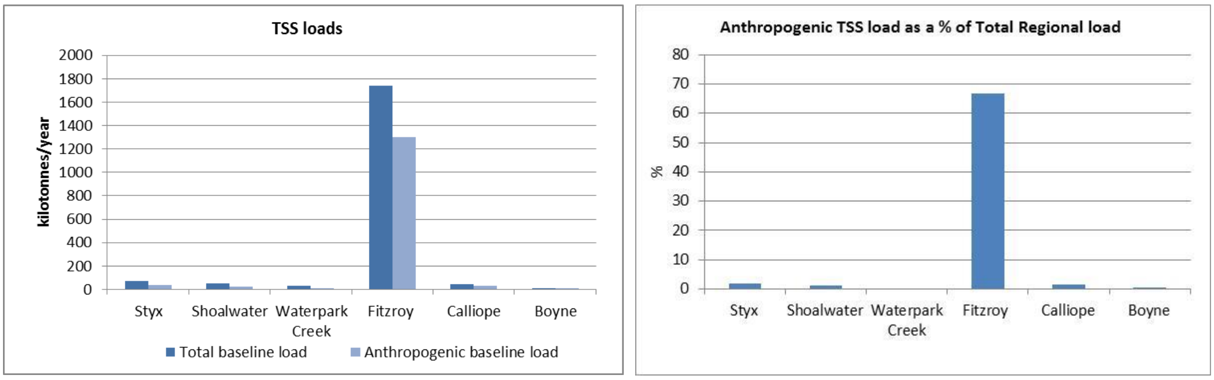

Figure 3.14. Annual load estimates for TSS from the basins in the Fitzroy region. The graphs show (a) Total (2008) and anthropogenic loads (2008) (kilotonnes), and (b) the proportion that the anthropogenic TSS from each basin contributes to the regional Total TSS Load.

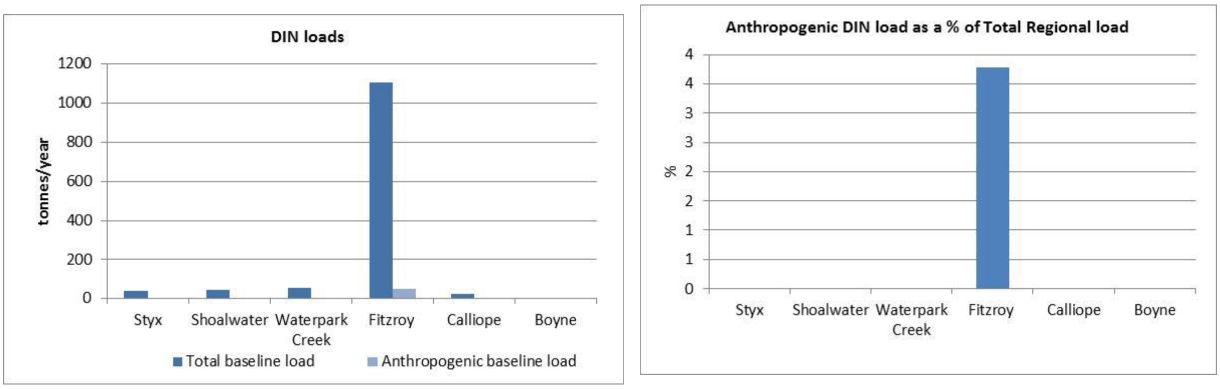

Figure 3.15. Annual load estimates for DIN from the basins in the Fitzroy region. The graphs show (a) Total (2008) and anthropogenic loads (2008) (tonnes), and (b) the proportion that the anthropogenic DIN from each basin contributes to the regional Total DIN Load.

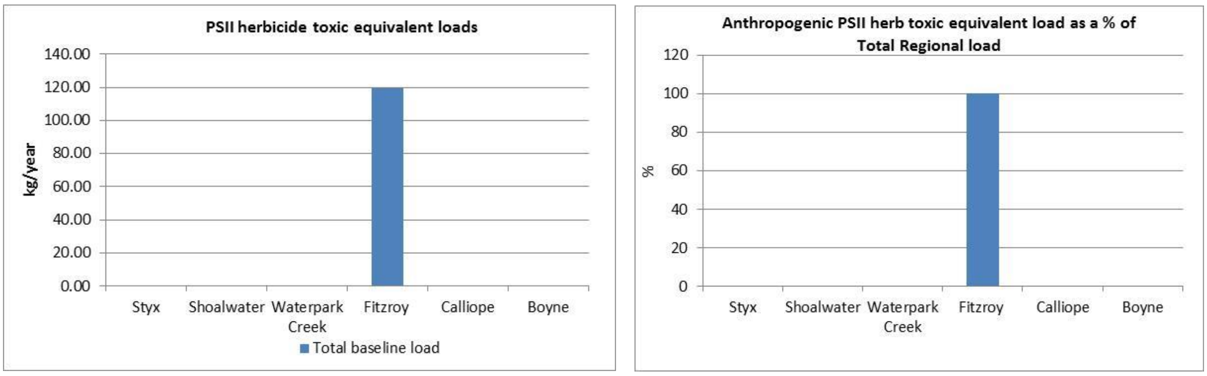

Figure 3.16. Annual load estimates for PSII herbicide toxic equivalent loads from the basins in the Fitzroy region. The graphs show (a) anthropogenic loads (2008) (kg), and (b) the proportion that these contributes to the regional total.

Table 3.14. Total and anthropogenic loads for PN, DIP and PP loads from Fitzroy basins, and as percentages of the total regional load and regional anthropogenic load.

PN loads ($t.y^{-1}$)					
Basin Name	**Pre-Development**	**Total Load (2008)**	**Anthropogenic load (2008)**	**Anthropogenic load % of Regional Total Load**	**Ranking**
Styx	25	60	35	3.0	2
Shoalwater	9	25	16	1.4	4
Water Park	8	18	10	0.8	5
Fitzroy	233	1035	802	67.9	1
Calliope	11	34	23	1.9	3
Boyne	2	10	8	0.7	6
Regional total	**288**	**1181**	**893**	**75.6**	

DIP loads ($t.y^{-1}$)					
Basin Name	**Pre-Development**	**Total Load (2008)**	**Anthropogenic load (2008)**	**Anthropogenic load % of Regional Total Load**	**Ranking**
Styx	7	8	0	0.0	2
Shoalwater	9	9	0	0.0	2
Water Park	10	10	0	0.0	2
Fitzroy	225	245	20	7.2	1
Calliope	4	4	0	0.0	2
Boyne	1	1	0	0.0	2
Regional total	257	278	21	**7.6**	

PP loads ($t.y^{-1}$)					
Basin Name	**Pre-Development**	**Total Load (2008)**	**Anthropogenic load (2008)**	**Anthropogenic load % of Regional Total load**	**Ranking**
Styx	12	29	17	2.2	2
Shoalwater	3	10	7	0.9	4
Water Park	4	6	2	0.3	6
Fitzroy	145	687	542	71.4	1
Calliope	8	21	13	**1.7**	3
Boyne	1	4	4	0.5	5
Regional total	**174**	**759**	**585**	**77.1**	

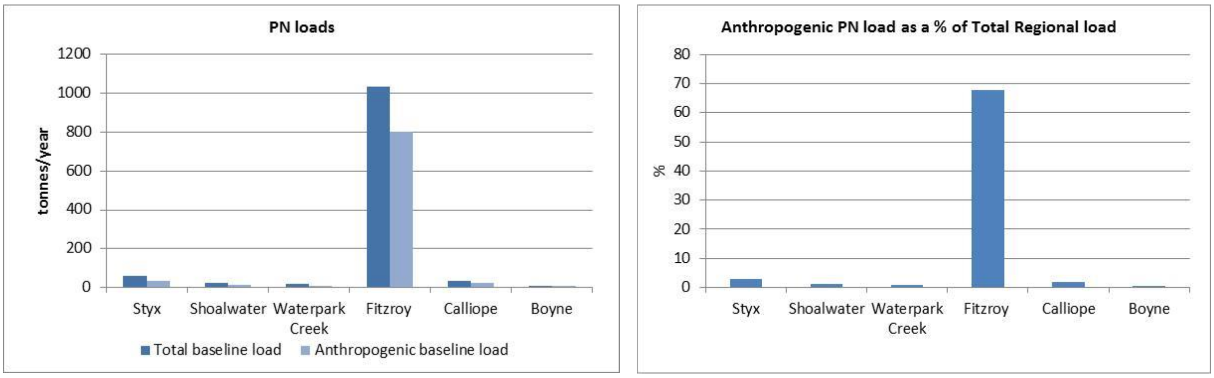

Figure 3.17. Annual load estimates for Particulate Nitrogen (PN) from the basins in the Fitzroy region. The graphs show (a) Total (2008) and anthropogenic loads (2008) (tonnes), and (b) the proportion that the anthropogenic PN from each basin contributes to the regional Total PN Load.

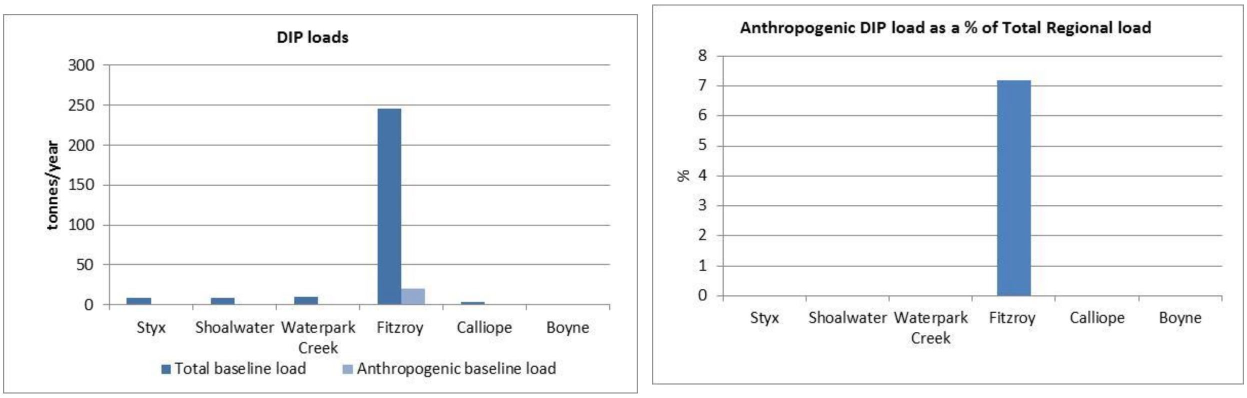

Figure 3.18. Annual load estimates for Dissolved Inorganic Phosphorus (DIP) from the basins in the Fitzroy region. The graphs show (a) Total (2008) and anthropogenic loads (2008) (tonnes), and (b) the proportion that the anthropogenic DIP from each basin contributes to the regional Total DIP Load.

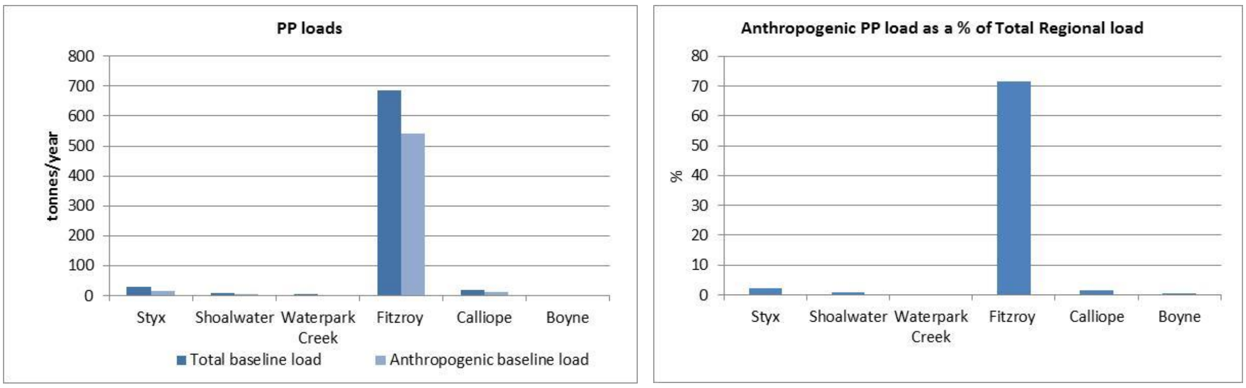

Figure 3.19. Annual load estimates for Particulate Phosphorus (PP) from the basins in the Fitzroy region. The graphs show (a) Total (2008) and anthropogenic loads (2008) (tonnes), and (b) the proportion that the anthropogenic PP from each basin contributes to the regional Total PP Load.

The 2013 Source Catchments modelling (2008 baseline) end-of-catchment pollutant load estimates for the Fitzroy region indicate that the Fitzroy Basin is the highest contributor for all constituents, contributing at least 87% of the total regional load for each constituent. Approximately 85% of the Fitzroy Basin is used for grazing (see Section 4.1). The differences between the Styx, Shoalwater, Water Park, Calliope and Boyne basins are relatively small. However, of these basins, the Styx Basin is the highest contributor to all constituents except for PSII herbicide toxic equivalent loads in the Boyne Basin, which is slightly higher; however, this result is based on limited monitoring results. The Styx Basin is 80% grazing land use. The key findings for each constituent are summarised below.

- **TSS:** The Fitzroy Basin contributes 89% of the total regional TSS load, and the greatest anthropogenic TSS load in the region, estimated at 1,300 kilotonnes per year. The anthropogenic contribution accounts for 67% of the total regional load. All other basins in the region each contribute less than 3% of the regional anthropogenic load (Figure 3.13), with the greatest contribution from the Styx Basin (2.8%). The lowest contributions are from the Water Park and Boyne basins. In comparison to all other GBR basins, the Fitzroy Basin is the second largest contributor of TSS to the total GBR TSS load (20%). In comparison to other NRM regions, the Fitzroy region contributes 23% of the total GBR TSS load.
- **DIN:** The Fitzroy Basin contributes 87% of the total regional DIN load, and the greatest anthropogenic DIN load in the region, estimated at 48 tonnes per year. However, the anthropogenic contribution accounts for only 4% of the total regional load. In the model, all other basins in the region contribute very small amounts of the regional anthropogenic load (reported as 0 tonnes in the model; Figure 3.14); however, this requires further investigation as there are cropping areas that are fertilised and sewage treatment plants in these basins that are likely to contribute some DIN load. In comparison to other NRM regions, the Fitzroy region has the third largest total DIN load (11% of total GBR load).
- **PSII herbicides (toxic equivalent load):** The Fitzroy Basin contributes 99% of the total regional PSII herbicide toxic equivalent load, estimated at 119 kg per year. In the model, all other basins in the region do not contribute detectable equivalent loads to the regional anthropogenic load (except for the Boyne reported as <1 kg for the) (Figure 3.15). In comparison to other NRM regions, the Fitzroy region contributes <2% of the total GBR PSII herbicide toxic equivalent load.
- **PN:** The Fitzroy Basin contributes 88% of the total regional PN load, and the greatest anthropogenic PN load in the region, estimated at 802 tonnes per year. The anthropogenic contribution accounts for 90% of the total regional load. All other basins in the region each contribute less than 4% of the regional anthropogenic load (Figure 3.16). The lowest contributions are from the Water Park and Boyne basins. In comparison to other NRM regions, the Fitzroy region has the third largest total PN load (10% of total GBR load).
- **DIP:** The Fitzroy Basin contributes 88% of the total regional DIP load, and the greatest anthropogenic DIP load in the region, estimated at 20 tonnes per year. The anthropogenic contribution accounts for 95% of the total regional load. All other basins in the region each contribute less than 4% of the regional anthropogenic load (Figure 3.17). In the model, all

other basins in the region contribute very small amounts of the regional anthropogenic load (reported as 0 tonnes in the model; Figure 3.17); however, as for DIN, there is low confidence in this result given our knowledge of land uses in these basins. In comparison to other NRM regions, the Fitzroy region has the second largest total DIP load (24% of total GBR load).

- **PP:** The Fitzroy Basin contributes 90% of the total regional PP load, and the greatest anthropogenic DIP load in the region, estimated at 542 tonnes per year. The anthropogenic contribution accounts for 93% of the total regional load. All other basins in the region each contribute less than 4% of the regional anthropogenic load (Figure 3.18). The lowest contributions are from the Water Park and Boyne basins. In comparison to other NRM regions, the Fitzroy region has the third largest total PP load (17% of total GBR load).

These pollutant load estimates were combined into a Loads Index, which is based on the anthropogenic proportion of the regional load for each basin and pollutant (described further in Section 2.3.2), shown in Table 3.15. This recognises that while the total load is important in affecting marine ecosystems, it is only the anthropogenic portion that is assumed to be the 'manageable' component. The proportional contributions for TSS, DIN, PSII herbicides, PN, DIP and PP are summed for each basin, and then normalised to the maximum to give a relative assessment.

The assessment shows the greatest relative contribution of combined end-of-basin loads to the Fitzroy region is from the Fitzroy Basin, with a significant dominance. In comparison, the relative contribution from each of the other basins is 1–3 % of the loads delivered by the Fitzroy Basin to the regions' anthropogenic load. Between these other basins, the Styx appears to generate the highest loads; however, there is limited data to validate these results (see Table 3.16 where the Fitzroy is excluded from the assessment).

Table 3.15. Loads Index for TSS, DIN, PSII herbicides (toxic equivalent load), PN, DIP and PP derived from the sum of the proportion of the basin anthropogenic load contributions to the total regional load.

	Basin anthropogenic load as % of Fitzroy regional total load								
Basin	**TSS**	**DIN**	**PSII**	**PN**	**DIP**	**PP**	**Sum**	**Loads Index**	**Loads Index Rank**
Styx	2.1	0.0	0.0	3.0	0.0	3.0	8.0	0.03	2
Shoalwater	1.3	0.0	0.0	1.4	0.0	1.4	4.0	0.01	4
Water Park	0.3	0.0	0.0	0.8	0.0	0.8	2.0	0.01	4
Fitzroy	66.7	3.8	99.92	67.9	7.2	67.9	313.4	1.00	1
Calliope	1.4	0.0	0.0	1.9	0.0	1.9	5.3	0.02	3
Boyne	0.4	0.0	0.08	0.7	0.0	0.7	1.8	0.01	4
						MAX	***313***		

The basin that had the largest summed contribution was given a score of 1; all other basins are expressed as a proportion. Loads data derived from Dougall et al. (2014).

Table 3.16. Loads Index for TSS, DIN, PSII herbicides (toxic load), PN, DIP and PP derived from the sum of the proportion of the basin anthropogenic load contributions to the total regional load, excluding the Fitzroy Basin.

	Basin anthropogenic load as % of Fitzroy regional total load								
Basin	**TSS**	**DIN**	**PSII**	**PN**	**DIP**	**PP**	**Sum**	**Loads Index**	**Loads Index Rank**
Styx	2.1	0.0	0.00	3.0	0.0	3.0	8.0	1.00	1
Shoalwater	1.3	0.0	0.00	1.4	0.0	1.4	4.0	0.51	3
Water Park	0.3	0.0	0.00	0.8	0.0	0.8	2.0	0.24	4
Calliope	1.4	0.0	0.00	1.9	0.0	1.9	5.3	0.67	2
Boyne	0.4	0.0	0.08	0.7	0.0	0.7	1.8	0.23	4
						MAX	***8.0***		

The basin that had the largest summed contribution was given a score of 1; all other basins are expressed as a proportion. Loads data derived from Dougall et al. (2014).

3.5 Combined assessment: Relative Risk Index

Using the information obtained through the above analyses for the marine water quality variables and end-of-basin pollutant loads, it is possible to make an assessment of the management priorities for minimising the risk of water quality impacts in the Fitzroy region. This section presents an option for a quantitative combined assessment to inform water quality management priorities among the basins in the Fitzroy region. However, the value of this level of assessment given the dominance of the Fitzroy River and the limitations of the data question the relevance of this additional analysis. Accordingly, this information should only be used to guide management decisions in conjunction with additional qualitative information, some of which is presented in Section 4 of this report.

As described in the methods (Section 2), to provide an overall ranking of relative risk between the basins the Loads Index and the Marine Risk Index for coral reefs and potential seagrass habitat were combined to generate a Coral Reef Relative Risk Index and Seagrass Relative Risk Index (Table 3.15). The final indexes for coral reefs and seagrass were then summed and anchored to provide an overall assessment of the relative risk of water quality to coral reefs and potential seagrass habitat — the Relative Risk Index (Table 3.16).

Table 3.17. Results of the overall risk assessment from summing the Loads Index and Marine Risk Index for coral reefs and seagrass.

Coral Reefs Risk Index	Risk Index for Reefs	Loads Index	Sum of Indexes	Final Index Reefs (Anchored)	Rank
Styx	n/a	0.03	n/a	n/a	
Shoalwater	n/a	0.01	n/a	n/a	
Water Park	0.17	0.01	0.17	0.09	2
Fitzroy	1.00	1.00	2.00	1.00	1
Calliope	0.08	0.02	0.10	0.05	3
Boyne	0.06	0.01	0.06	0.03	4
		Max	*2.00*		
Seagrass Risk Index (surveyed only)	Risk Index for Seagrass	Loads Index	Sum of Indexes	Final Index Seagrass (Anchored)	Rank
Styx	n/a	0.03	n/a	n/a	
Shoalwater	n/a	0.01	n/a	n/a	
Water Park	0.03	0.01	0.04	0.02	4
Fitzroy	1.00	1.00	2.00	1.00	1
Calliope	0.12	0.02	0.14	0.07	3
Boyne	0.13	0.01	0.14	0.07	2
		Max	*2.00*		

The basin that had the maximum value was given a score of 1; all other basins are expressed as a percentage based on the value in each basin relative to the area in the basin with the maximum value.

Table 3.18. Results of the overall risk assessment using a sum of the anchored indexes for coral reefs and seagrass.

Coral Reefs and Seagrass — FINAL INDEX	Final Index Reefs	Final Index Seagrass	Sum of Final Indexes	Final Score (Anchored)	Rank
Styx	n/a	n/a	n/a	n/a	
Shoalwater	n/a	n/a	n/a	n/a	
Water Park	0.09	0.02	0.10	0.05	3
Fitzroy	1.00	1.00	2.00	1.00	1
Calliope	0.05	0.07	0.12	0.06	2
Boyne	0.03	0.07	0.10	0.05	4
		Max	*2.00*		

The basin that had the largest sum of indexes was given a score of 1; all other basins are expressed as a percentage based on the sum of indexes in each basin relative to the sum in the basin with the maximum sum of indexes.

These results show that the Fitzroy River dominates the greatest risk to each habitat in terms of the potential water quality impact from all of the assessment variables in the Fitzroy region and end-of-catchment anthropogenic loads of TSS, DIN, PSII herbicides, PN, DIP and PP. The Water Park, Boyne and Calliope rivers each pose less than 6% of the relative risk posed by the Fitzroy River.

It is highlighted that there are many uncertainties associated with the input datasets and method for combining these indexes at a basin-scale at this time (see Section 6); further discussion is recommended prior to making any management decisions based on these results for the smaller coastal basins. In particular, ZoI are not available for the Styx and Shoalwater basins, and the input data for the Styx and Shoalwater basin loads are uncertain.

It is also noted that the value of using this type of final assessment where all values are combined into a single score can reduce the intrinsic value of each of the multiple datasets and stages of assessment used in this study and without sufficient explanation, may leave the final results subject to misinterpretation. However, it does provide an overall assessment of the relative risk of all water quality in the marine environment in the context of the end-of-catchment anthropogenic loads, which may be useful for managers in prioritising catchment-based investments in the Fitzroy region.

4 Linking Marine Risk to land-based pollutant loads

This section summarises the outcomes of the risk assessment using additional evidence from the supporting studies to draw conclusions about the relative risk of water quality to GBR ecosystems from the six river basins in the region. Further work by Star et al. (2015) assesses the management priorities at a sub-catchment scale (Neighbourhood Catchment Areas) to inform the implementation strategy for the WQIP.

Several limitations to the quantitative assessment are identified in Section 5; however, a number of these can be overcome by incorporation of new knowledge in a qualitative way to make conclusions about the relative risk of degraded water quality to the GBR.

Supplementary evidence that is important for the conclusions of our assessment are also included below.

4.1 Catchment land use

Land use characteristics of the Fitzroy region are shown in Table 4.1, and mapped in Figure 4.1. This information is all derived from Dougall et al. (2014). The dominant land uses by area are grazing (~78%), conservation (~8%), forestry (~6%) and dryland cropping (5%). Other land uses including urban, horticulture, irrigated cropping and sugarcane are all less than 1% of the regional land use area.

Grazing is the most common land use in the Fitzroy NRM region, with the majority of the region dedicated to cattle production (Figure 4.2 and Figure 4.3a). Large areas of dryland cropping occur in the western part of the basin, while irrigated cropping (including cotton) occurs around the townships of Emerald, Theodore and Biloela. There is also extensive coal mining occurring in the Bowen Basin, especially around the townships of Moranbah, Dysart, Blackwater and Moura. The coastal basins have a mix of land uses, dominated by grazing and conservation areas (Figure 4.3b).

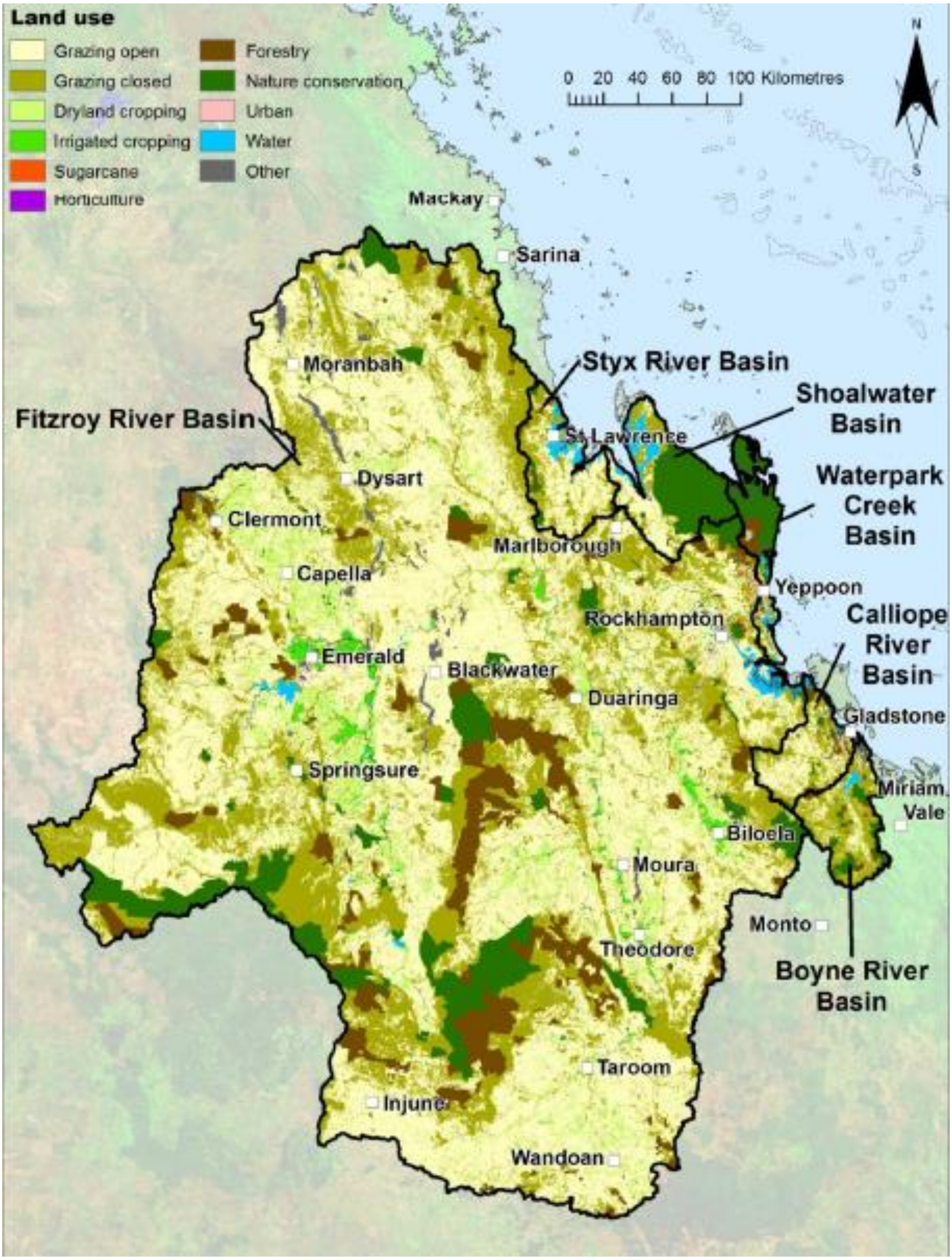

Figure 4.1. Land use map of the Fitzroy region.

Prepared using QLUMP 2009 data. Source: Dougall et al. (2014).

Table 4.1. Estimated land use by area (km²) in the Fitzroy region

Basin	Grazing Forested	Grazing Open	Dryland Cropping	Irrigated Cropping	Forestry	Horticulture	Sugarcane	Urban	Conservation	Water	Other	Total
Styx	1,075	1,339	5	0	76	<1	0	1	148	370	2	3,015
Shoalwater	672	792	0	0	12	0	0	<1	1,685	441	5	3,608
Water Park	179	114	<1	<1	191	8	0	73	1,157	115	7	1,845
Fitzroy	40,339	73,320	7,929	1,210	9,156	48	3	331	8,350	1,018	1,159	142,863
Calliope	842	989	<1	<1	133	3	0	43	107	93	36	2,244
Boyne	1,384	467	0	3	134	1	0	17	385	77	33	2,502
Total	**44,491**	**77,021**	**7,934**	**1,213**	**9,701**	**60**	**3**	**465**	**11,832**	**2,113**	**1,242**	**156,076**

Based on QLUMP data used in Source Catchments. Source: Dougall et al. (2014)

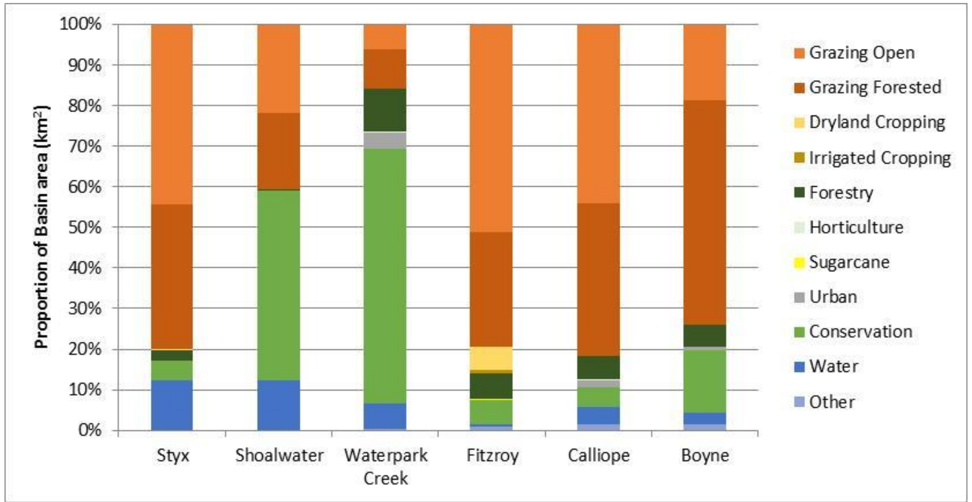

Figure 4.2. Land use characteristics in each basin, showing the proportion of the area of each basin in each land use.

Source: Derived from Dougall et al. (2014).

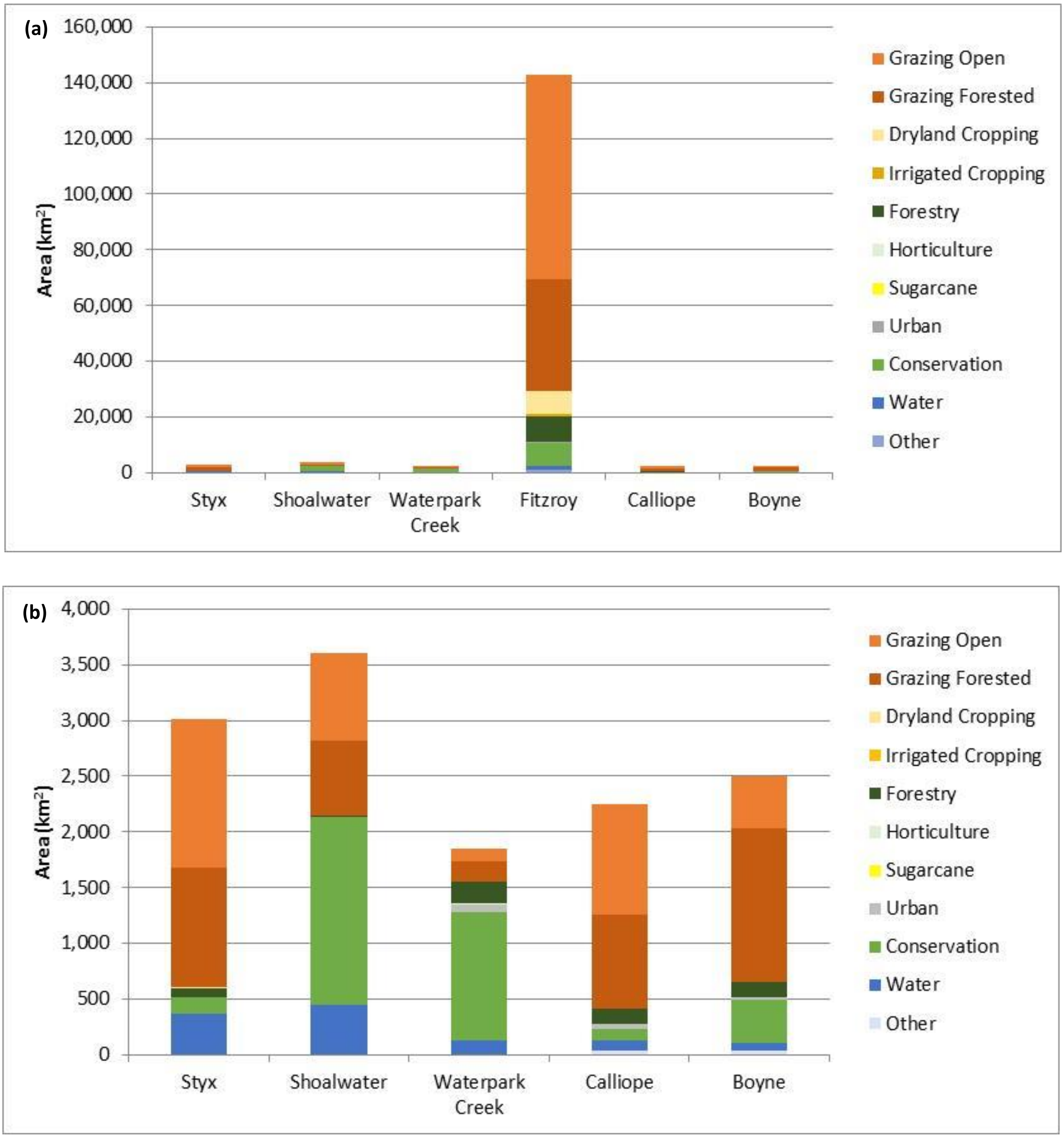

Figure 4.3. (a) Land use areas in each basin in the Fitzroy NRM region. The characteristics for the smaller coastal basins are shown at a larger scale in (b).

Source: Derived from Dougall et al. (2014).

The area and land use characteristics of the basins are varied. To summarise:

Styx: 80% of the basin is utilised for grazing of livestock (Beef cattle). The bulk of the remaining area within the basin forms waterways (~12%; i.e. river, wetlands, dam, lake or storage) and conservation and natural environments (~5%). Agriculture and plantations within the basin, both dryland and irrigated, cover less than 2.5% of the total basin area.

Shoalwater: Approximately 60% of land within the Shoalwater Basin is classified as water (marsh/wetland, river and reservoir/dam — 12%) and conservation and natural environments. The majority of the conservation and natural environment of the Shoalwater Basin is under the control of the Australian Department of Defence (Shoalwater Bay Training Area (SWBTA) and <1% of land is conserved under natural feature protection or classified as a national park. The remaining area (~40%) is mostly used for grazing.

Water Park: The majority of land within the Water Park Basin falls under conservation or natural environments (63%), grazing (16%) and forestry (10%). The remaining land within the catchment includes ~5% water (marsh/wetlands, river and reservoir/dam); <5% intensive uses (residential and associated services/industry); and <1% of production from dryland/irrigated agriculture and plantations.

Fitzroy: The majority of land is used for grazing (~85%). The remaining land use within the catchment includes approximately 5% dryland cropping, 6% of conservation and natural environments (nature conservation, minimal use including defence lands, and managed resource protection); and a small proportion of other land uses including intensive use (i.e. residential, industry, transport and utilities) and irrigated agriculture and plantations.

Calliope: The majority of land within the Calliope Basin is used for grazing (~82%) and production from forestry (~6%). The remaining land use within the basin includes approximately 5% of conservation and natural environments (nature conservation and minimal use); 7% of intensive use (i.e. residential, industry, transport and utilities) and 4% water (marshland/wetland, river, reservoir /dam).

Boyne: The majority of land within the Boyne Basin is used for grazing (~74%) and production from forestry (~5%). The remaining land use within the basin includes approximately 12% of conservation and natural environments (nature conservation and minimal use); 2.5% of intensive use (i.e. residential, industry, transport and utilities) and ~5% water (marshland/wetland, river, reservoir/dam).

The relative contribution of different land uses (from both hillslope and gully erosion) and streambank erosion in the Fitzroy region to constituent export is summarised in Dougall et al. (2014). The key findings are:

- Grazing (open and closed) are the greatest contributors of TSS load to export by land use comprising approximately 60% of the total TSS exported (see Table 4.2 and Figure 4.4).

- Streambank erosion was the second highest source at 23% with cropping less than 10% of the total fine sediment exported.
- Cropping (44%) and grazing (54%) are the two major sources of PSII herbicides exported. On a per unit area basis, however, cropping is the highest followed by grazing and horticulture respectively. PSII herbicides used in cropping lands and found in receiving waters are atrazine, ametryn, hexazinone and diuron (Packett et al. 2009). The herbicide tebuthiuron has also been detected in run-off originating from grazing lands in the Fitzroy (Packett et al. 2009).

Table 4.2. Land use contribution to total TSS loads for the Fitzroy Basins.

	Total TSS exported load, kilotonnes per year											
Basin	**Grazing Forested**	**Grazing Open**	**Dryland Cropping**	**Irrigated Cropping**	**Forestry**	**Horticulture**	**Sugarcane**	**Urban**	**Conservation**	**Stream**	**Other**	**Total**
Styx	23.8	29.2	0.2	0.0	1.8	0.0	0.0	0.0	2.9	10.1	0.0	68.1
Shoalwater	13.6	16.0	0.0	0.0	0.2	0.0	0.0	0.0	18.0	5.2	0.0	53.0
Water Park	3.8	3.6	0.0	0.2	4.9	0.1	0.0	0.8	18.0	0.6	0.1	32.2
Fitzroy	345.1	723.8	142.5	14.4	42.1	0.4	0.2	1.0	37.7	435.3	1.0	1743.4
Calliope	16.3	16.9	0.0	0.0	1.6	0.0	0.0	0.3	1.5	7.5	0.2	44.3
Boyne	5.1	1.8	0.0	0.0	0.3	0.0	0.0	0.1	1.0	2.5	0.1	10.9
Total	**407.7**	**791.4**	**142.7**	**14.6**	**50.9**	**0.5**	**0.2**	**2.1**	**79.2**	**461.1**	**1.4**	**1,951.9**

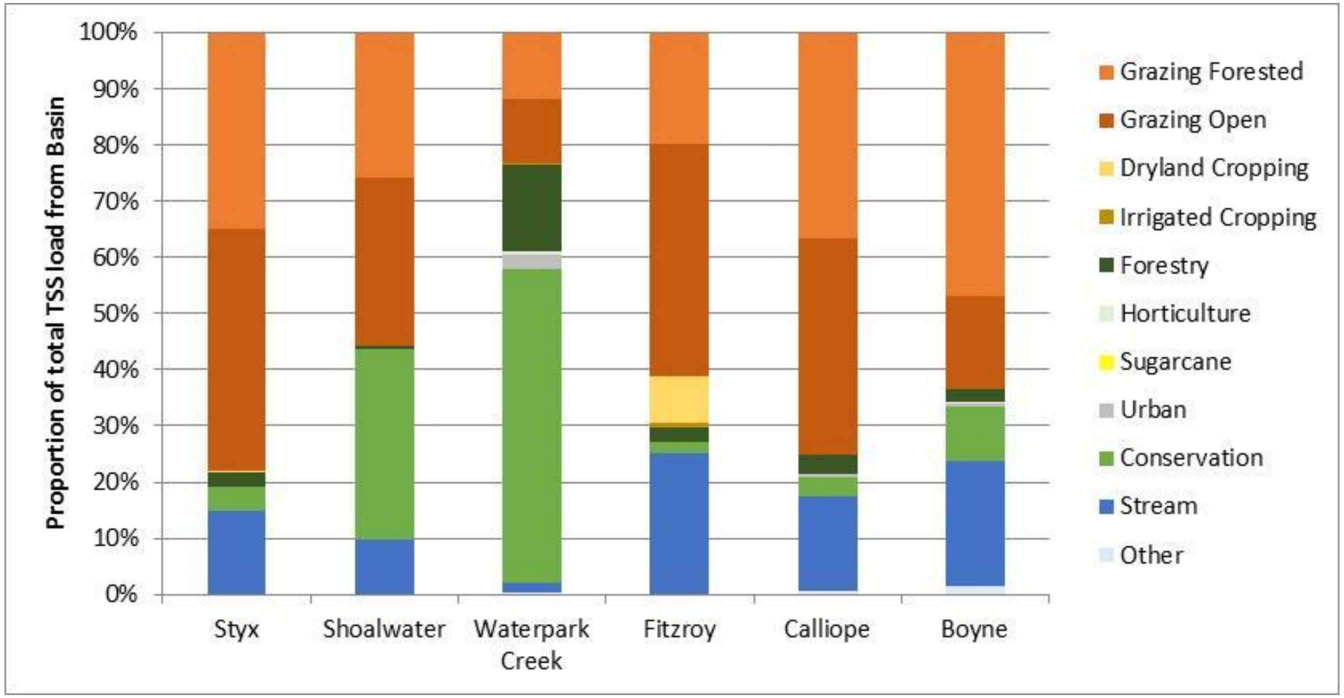

Figure 4.4. Total TSS loads by land use for each basin in the Fitzroy NRM region.

Source: Derived from Dougall et al. (2014).

Urban land uses contribute a large range of pollutants including TSS, nutrients, pesticides and other pollutants such as heavy metals, hydrocarbons and pharmaceuticals. Overall, urban land uses contribute less than 10% of the total regional load for all constituents. In the Fitzroy region urban expansion is occurring along the Capricorn Coast from Yeppoon to the south of Emu Park and around the main regional centres of Rockhampton and Gladstone.

Sewage discharges can be relevant at a local scale. There are several sewage treatment plants (STP) in the Fitzroy region that discharge into the GBRWHA or adjacent waterways. The loads for these treatment plants are estimated in Table 4.4, and are based on an assessment by Dougall et al. (2014) where it is estimated that ~79% of the Total Nitrogen and Total Phosphorus is in dissolved inorganic form.

Table 4.3. Major sewage treatment plants in the Fitzroy region.

Name of STP	Discharge point	Catchment	EP	DIN (kg/yr)	DON (kg/yr)	DIP (kg/yr)	DOP (kg/yr)
Yeppoon Sewage Treatment Plant	Corduroy Creek	Water Park	10,000-50,000	788	210		
North Rockhampton Sewage Treatment Plant	Fitzroy River	Fitzroy River	10,000-50,000	24,989	6,643	19,710	5,559
South Rockhampton Sewage Treatment Plant	Fitzroy River	Fitzroy River	10,000-50,000	29,801	7,922	10,960	3,091
West Rockhampton Sewage Treatment Plant	Fitzroy River	Fitzroy River	10,000-50,000	9,112	2,422	3,049	860
TOTAL				64,690	17,197	33,719	9,510

4.2 Ports

The Fitzroy region includes the Port of Gladstone, which is a significant coastal facility as one of the largest coal export ports in Australia and in 2011 approval was granted for development on Curtis Island of three liquefied natural gas (LNG) processing facilities, and the smaller Port of Rockhampton located in the Fitzroy River Delta at Port Alma. A variety of port and shipping activities, as well as portside industries, can influence water quality. These activities include capital and maintenance dredging, shipping movements and incidents, construction and maintenance of wharves and in-water structures, and emissions from industries.

4.3 Coastal wetlands and estuaries

The Fitzroy region contains large areas of important coastal wetland and mangrove communities that are vulnerable to changes in water quality. Wetlands have suffered degradation with only a few of these systems remaining in good condition (Packett et al. 2009). Johnson et al. (2015) summarised the threats to these ecosystems in the Fitzroy region and identified that these systems range considerably in status from very good to poor (wetland-dependent), with localised declines in wetland extent.

There are a number of important estuaries in the region, including the Fitzroy River Delta, which is the estuary of the largest seaward draining catchment on Australia's east coast, the Shoalwater and Corio Bay area, which is a Ramsar internationally important wetland, and the estuaries of Gladstone Harbour, a highly modified tidal wetland that includes the Port of Gladstone and significant industrial activities. These estuaries provide important breeding and nursery habitat for many species of fish and invertebrates (some commercially fished), and marine megafauna, including dugong and green turtles. The health of the Fitzroy River Delta was scored as 'B — good' in 2013–14 and 'C — fair' in 2012–13, based mostly on water quality indicators (Fitzroy Partnership for River Health Annual Report Card [5]). Gladstone Harbour was scored 'C — Satisfactory' for water quality (Gladstone Healthy Harbour Partnership's 2014 Pilot Report Card[6]).

4.4 Further understanding of sediment sources, delivery and fate

As supporting material for the WQIP, Lewis et al. (2015a) have prepared a review that synthesises the research related to understanding the sources, delivery and fate of sediments in the Fitzroy NRM region. Adapting a similar approach to the Bartley et al. (2014a; 2014b; see also Lewis et al. 2015b) synopsis on the Burdekin River, a a series of specific questions have been formulated from the marine environment back to the catchment area (and identifying specific catchment sources), presenting the latest process understanding.

The following questions were considered:

(1) Do Fitzroy River discharge and particulate constituents reach vulnerable marine ecosystems, what are the effects (if any) and what parts of the GBR are influenced by both direct exposure and secondary effects?

(2) How far are the particulate constituents transported in the GBR lagoon, what is their fate and when are they transported?

(3) Where do the particulate constituents come from in the Fitzroy catchment, what are their sources and are there priority areas for management?

[5] http://riverhealth.org.au/ Accessed June 2015
[6] http://rc.ghhp.org.au/report-cards#resultPanelEnvironmental Accessed June 2015

(4) What are the key erosion processes that release the priority sediment to the rivers and have erosion rates in the Fitzroy catchment changed over time?

This information highlights that, to the best of our current knowledge, sheet-wash erosion in the cultivated Tertiary basaltic soils, gully and scald erosion, and stream bank damage from cattle ramps and trials in grazing lands should be the priority for management of sediment erosion in the Fitzroy River catchment area. The latest modelling suggests that the Connors and Dawson catchments contribute the highest loads of fine sediment (i.e. 63 µm, although monitoring in the Connors suggest a large proportion is <10 µm). The geochemical tracing data suggest that basalt sources in the Comet and Nogoa catchments should also be a priority for management.

The synthesis also incorporates discussion of the influence of large-scale flood events on the marine ecosystems in the region, which is directly relevant to this risk assessment, especially given that most of the variables in the Marine Risk Index are represented as average conditions over several years. The following information directly relevant to this assessment is extracted from Lewis et al. (2015a).

Extreme events occurred in the Fitzroy River in 1991 and 2011 with documented adverse impacts on coastal and marine ecosystems. In summary, the 1991 event coupled with low winds caused a large freshwater plume to extend eastwards out to the mid-outer GBR to Heron Island. Salinities as low as 7–8 ppt were recorded near the Keppel Islands and major impacts associated with the freshwater plume included the widespread mortality of oyster and barnacle species in the Fitzroy River estuary and Keppel Bay (Coates 1992) and 'absolute mortality' of *Acroporid* and *Pocilloporid* corals to a depth of 1.3 m below low water datum in the reefs of the Keppel Islands (van Woesik et al. 1995). Recovery of the fringing reefs from the 1991 flooding was estimated at 10–15 years (Jones & Berkelmans 2014).

The major impacts from the 2011 extreme flooding from the Fitzroy River were also recorded in Keppel Bay with Jones and Berkelmans (2014) documenting 40–100% mortality of corals down to 8 m depth for many of the fringing reefs of the Keppel Islands (see also Tan et al. 2012). The coral mortality in the Keppel Islands also resulted in associated declines in coral reef fish abundance, diversity and fish assemblage structure (Williamson et al. 2014). While Jones and Berkelmans (2014) showed that the impact of freshwater in Keppel Bay far outweighed the 'pollutants' delivered from the Fitzroy River including suspended sediments, nitrogen, phosphorus and herbicides, continued monitoring in the region has shown the coral fringing reefs of the Keppels have continued to decline following subsequent exposures to low salinity and turbid waters in 2012 and 2013 (Wenger et al. 2015). This finding coupled with the knowledge that the reefs showed little decline prior to 2011 despite moderate events in 2008 and 2010 suggest that chronic exposure of turbidity (i.e. lowered photic depth) and hence the delivery of suspended sediment from the Fitzroy River during this period has contributed to this decline. Furthermore, Wenger et al. (2015) postulated that the fringing reefs of the Keppel Islands have reduced resistance to withstand repeated exposures of river flood plumes and associated constituents from the Fitzroy River. From a longer-term perspective, Rodriguez-Ramirez (2013) showed there was little variation in the living and dead coral assemblages of the Keppel Island fringing reefs. In fact, the coral death assemblages (aged using U-series dating) were all linked to disturbance events over the past three decades, suggesting that these reefs are well-adapted and resilient to periodic discharge from the Fitzroy River (Rodriguez-Ramirez 2013).

The extent of the repeated influence of repeated flood events in Keppel Bay is further supported by studies presented by Wenger et al. (2015) showing the extent of primary plume waters in the inshore areas of Keppel Bay from 2008 to 2013 (Figure 4.9). Flood plumes were mapped MODIS true colour satellite imagery (Álvarez-Romero et al. 2013). “Primary” and “secondary” water type classifications were applied to quantify exposure frequency of ecosystems to highly turbid water from flood plumes and subsequent re-suspension during the 2006–2013 wet seasons (December–April inclusive). The primary water type represents high turbidity (Devlin et al. 2012) with high Coloured Dissolved Organic Matter (CDOM) and Total Suspended Sediment (TSS) (Devlin et al. 2013). TSS and light attenuation (measured as the diffuse attenuation coefficient for photosynthetically active radiation, K_dPAR) in the primary water type are typically ~36.8 ± 5.5 mg L^{-1} and 0.73 ± 0.54 m^{-1}, respectively (Devlin et al. 2013). The secondary water type is characterised by elevated Chl-a concentrations, with TSS concentrations reduced due to sedimentation. Figure 4.9 represents 22 weekly composite images of daily images from 1 December to 30 April per wet season (2006–2013) to minimise the amount of area without data due to masking of clouds (Álvarez-Romero et al. 2013), showing areas between approximately 2,000 and 4,000 km^2 within primary plume waters.

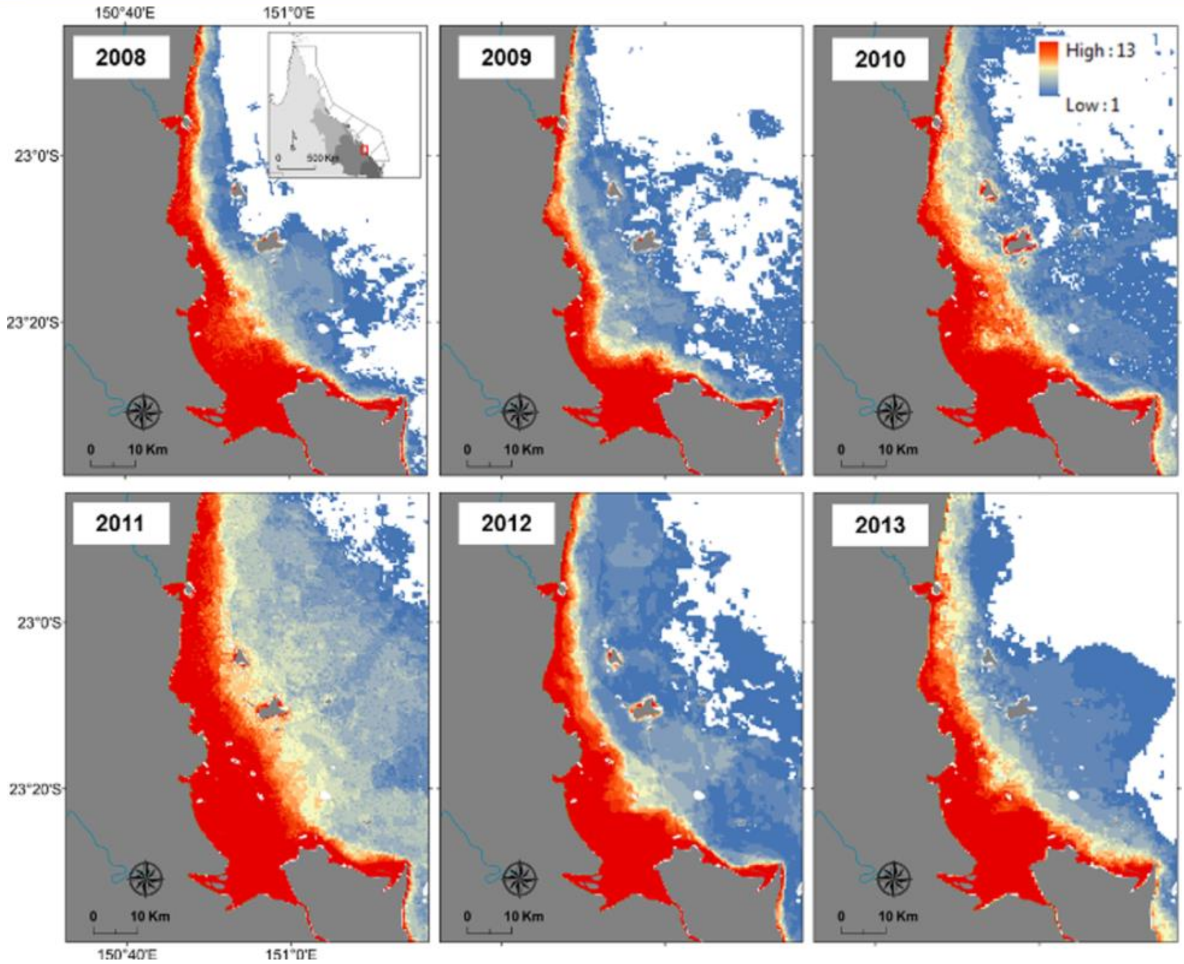

Figure 4.5. The 2008–2013 wet season frequency of primary plume waters in the Keppel Island group.
The extent of primary water exposure was 2,060, 2,397, 3,244, 4,078, 3,424, and 2,861 km^2 in 2008–2013, respectively.

Star et al. (2015) have also undertaken an assessment of priorities for management at a neighbourhood catchment scale which has been interpreted by FBA to derive spatial priorities for grazing and cropping management to reduce the sediment loads delivered to the GBR.

4.5 Future scenarios: Responses to higher frequency of extreme flood events

It is important to consider the future threats to the GBR ecosystems in the Fitzroy region, and how this might change the extent and severity of the relative risk from water quality factors in the future. Wenger et al. (2015) have recently examined how coral reefs in the Keppel Islands responded to moderate and major flooding events. The study: 1) assessed spatial and temporal variations in water quality and coral conditions; 2) identified key drivers of coral cover and macro-algae during a moderate and major flooding event; 3) examined the resilience of coral reefs against multiple flooding events; and 4) investigated the role of no-take zones in enhancing coral reef resilience to reduced water quality.

Moderate flood plumes from 2007–2009 did not result in coral cover declines on reefs in the Keppel Islands, suggesting intrinsic resistance against short-term exposure to reduced water quality. However, from 2009–2013, live coral cover declined sharply following extended periods of high exposure to turbid, low salinity water from major flood plume events in 2011 and subsequent moderate events in 2012 and 2013. Importantly, the ability of the reefs to cope with moderate disturbances following a major flooding event was lost. Although zone (no-take zones or fished) was identified as a significant driver of coral cover, we recorded consistently lower coral cover on reserve reefs than on fished reefs throughout the study period and significantly lower in 2011. **The findings suggest that even reefs with an inherent resistance to reduced water quality are not able to withstand repeated disturbance events.** The limitations of reserves in mitigating the effects of reduced water quality on near-shore coral reefs underscores the importance of integrated management approaches that combine effective land-based management with networks of no-take reserves.

These results have important implications for prioritising future management of the region, and the need to take into account the most appropriate management actions in the context of a changing climate. In this case, other considerations such as implementing management actions that protect relatively intact areas in the catchment and marine environments may become more important.

4.6 Conclusions and potential management priorities

Based on consideration of the Load Index and the areas of greatest relative risk identified in this assessment, we can draw the following conclusions regarding potential priorities for managing degraded water quality in the Fitzroy region.

1. The water quality influence in the region is generally constrained to the inshore areas, with hotspot areas in Shoalwater Bay and Keppel Bay for sediments, and Keppel Bay for nutrients. However, as noted above, the sediment influence in Shoalwater Bay is not believed to be linked to river discharge, and the area is naturally turbid due to shallow and large tidal variation. In addition, satellite imagery from flood events has shown the flood waters being steered 'past' the embayment of Shoalwater Bay. The influence of PSII herbicides does not appear to extend in the marine environment to any significant extent, supported by monitoring data where tebuthiuron was the only pesticide that exceeded the Water Quality Guidelines at a North Keppel Island routine monitoring site in 2012–13 (Gallen et al. 2013) and was below the guidelines in 2013–14 (Gallen et al. 2014).
2. The combined assessment of the relative risk of marine water quality variables (Section 3.3) highlights that the areas in the Very High relative risk class were located in Keppel Bay, extending out to the Keppel Island Group. Analysis of the zones of influence modelling indicates that the Fitzroy Basin has the greatest influence on this area, appearing to occur annually. This modelling also suggests that Water Park Creek and the Calliope River also influence the Keppel Island Group in larger flow events; however, these rivers only contribute 1–2% of the relative combined anthropogenic loads of the Fitzroy Basin. Nevertheless, when considering combined and cumulative impacts, it is still important to

ensure that the water quality from these basins does not decline to exert additional pressures on these receiving environments.

3. The areas around Port Curtis and extending up to Curtis Island are in the High and Moderate relative risk classes, and in this assessment, these areas were in the receiving areas of the zones of influence of the Calliope and Boyne rivers each year. While the influence of these rivers is small in comparison to the Fitzroy River in the context of the whole region, the Calliope and Boyne basins are important to consider in terms of localised impacts on these receiving environments and as above, need to managed to prevent increasing pressure from these basins in the future.
4. The proportion of surveyed seagrass area in the Very High and High assessment classes is greater than 66% and up to 100% for all sediment and nutrient variables. A large proportion of this seagrass is located in Shoalwater Bay and is not likely to be driven by anthropogenic influences; however, there are important and extensive seagrass areas in Port Curtis and Rodds Bay that are likely to be influenced by anthropogenic run-off from the Calliope and Boyne basins, and direct influences such as dredging. The proportion of deepwater modelled seagrass in the Very High and High assessment classes is less than 5% for all variables.
5. In the assessment of end-of-catchment pollutant loads (Section 3.4) the greatest relative contributions of combined end-of-catchment loads to the Fitzroy region is from the Fitzroy Basin. For the smaller coastal basins, the greatest loads are from the Styx Basin, but these are small in comparison to the Fitzroy Basin. Within the Fitzroy Basin, Dougall et al. (2014) identified that the Dawson catchment generates the largest proportion of total sediment to the GBR, followed by the Isaac and Lower Fitzroy. Grazing is the dominant land use across the region delivering sediment loads to the GBR.
6. This combined assessment of water quality variables can be used to guide overall management priorities for addressing the risks from degraded water quality to coral reefs and seagrass between Fitzroy region basins; however, given the dominance and large area of the Fitzroy Basin, further prioritisation between sub-catchments within the basin is required. This has been undertaken by Star et al. (2015) where several sub-catchments were identified as the highest priority sub-catchments for reducing TSS loads in the region.

5 Improvements and limitations to the risk assessment and future needs

The risk assessment described in this report provides the best available assessment of the relative risk of water quality pollutants to the GBR and the information outlined above can be used as the first step in prioritising management based on regional 'hotspots' for pollutant sources, contributing industries and resulting impacts in the marine environment.

We have applied a number of improvements from the 2013 risk assessment (Brodie et al. 2013a). These include:

1. Analysis of the reliability of Chl-a data obtained using remote sensing and replacement of this parameter with results from long-term Chl-a monitoring data.
2. Applied a revised method to assess the frequency of exceedance of TSS concentration data obtained using remote sensing data, factoring in the proportion of valid observations.
3. Definition of zones of influence for each basin in an attempt to attribute marine risk back to individual basins.
4. Incorporation of additional pollutants in the assessment of end-of-catchment loads; this assessment includes TSS, DIN, PSII herbicides as well as PN, DIP and PP.

However, there are several limitations to the assessment that are important to identify, and are summarised below.

Limitations to the input datasets in terms data collection, temporal and spatial resolution, influence the certainty of the outcomes. Several examples can be presented here:

The input for Chl-a is based on a long-term mean of in situ data collected between 1988 and 2006. TSS 2mg/L exceedance is based on daily remote sensing observations over a 10-year monitoring period (with a range of uncertainties described in Petus et al. 2015). While TSS, DIN and PN plume loading is based on a mean of 2003 to 2013 (which were, in fact, relatively wet years in the long-term record). PSII herbicide concentration modelling is based on single flood events in 2009 to 2011. In addition, the temporal resolution of the remote sensing data (which is used for daily observations, the plume loading and PSII herbicide modelling) is only one or two valid observations every five days. This presents difficulties in getting good temporal representation of the water quality parameter (e.g. TSS, Chl-a or DIN). For these reasons the final conclusions of the assessment are supported by additional evidence of known water quality conditions, spatial and temporal patterns and ecological impacts. Additional variables that were considered but not included due to the current lack of temporal and spatial data, and/or knowledge of ecological impacts include chronic exposure to PSII herbicides and non-PSII herbicides, particulate nutrients and phosphorus exposure, and the presence and distribution of micro-pollutants in the GBR.

The modelled estimates of anthropogenic end-of-catchment loads are long-term averages and do not capture the influence of large floods. Empirical datasets included in the assessment (e.g. TSS, DIN and PN plume loading) do factor in these events. In comparing the modelled results against empirical data, the relative contributions of individual basins are in general agreement with monitoring data (where available) except during extreme wet seasons.

The marine hydrological modelling is only available for the Fitzroy River to estimate the zones of influence, and the path-distance approach applied for the other rivers cannot be applied to the Styx and Shoalwater rivers as there is no gauging station in these basins. Therefore the full analysis cannot be extended to all rivers in the region.

The assessed risk posed by pesticides is most probably an underestimate. Only a few of the pesticides detected in the GBR lagoon are considered. The risk posed by multiple pesticides, in combination with other contaminants found in flood plumes (e.g. elevated TSS and nutrients) and other environmental stressors (temperature) have not been assessed. Cumulative impacts from the multiple plumes that occur each year are also not accounted for. Toxicity of PSII herbicides is time-dependent (Vallotton et al. 2008), i.e. the toxicity to phototrophs increases with exposure duration. For this risk assessment, only acute exposure was used to assess the potential impacts to seagrass and corals.

The risk classes for individual water quality variables are not equivalent in terms of ecological impact, and are therefore not directly comparable without recognition of these differences. Further studies should adequately address this limitation to provide a better representation of the severity of potential ecological impacts between assessment classes for each water quality variable. Community characteristics such as the sensitivity and resilience of particular seagrass or coral communities (e.g. associated with their natural levels of exposure to pollutants) are additional parameters that must be considered when defining the ecological consequences of the risk. Indeed, different species assemblages will respond differently to the same exposure (i.e. same likelihood magnitude of risk) to river plumes. The consequence of the exposure of species to a range of water quality conditions is complicated by the influence of multiple stressors and additional external influences including weather and climate conditions, and consequences are mostly unknown at a regional or species level.

The approach to classification used is also a potential weakness of multi-criteria analysis, which is an interval scale approach, while risk consequence is inherently oriented to a need for quantification of magnitudes. In addition, the assessment does not account for the potential synergistic or antagonistic effects that these multiple stressors when acting together may have on ecosystems.

Only a limited sensitivity analysis that tested weighting of variables has been conducted. More scenarios that scale or 'weight' individual factors or pollutants as being more or less important and the effect of only selecting the highest assessment classes in the final analysis should be tested. For example, a more detailed assessment of the patterns in the lower assessment classes should be considered in future work, particularly given the potential influence of chronic exposure to pollutants, or the effects of periodic exposure to high concentrations of pollutants.

Further validation of remote sensing-based results is required for locations where high turbidity that confounds existing algorithms may naturally occur. These areas include Shoalwater Bay and Broad Sound, which are naturally turbid. Uncertainties in products derived from remote sensing of these areas have not been resolved (see results and discussion in Petus et al. 2015). In addition, the number of valid observations for the remote sensing assessment varies between seasons and locations and over the year equates to an average of less than two valid observations every five days (refer to Maynard et al. 2015).

The scope of the assessment is limited in terms of the coverage of social and economic issues. It should be recognised and highlighted that the results presented in this study only represent the biophysical perspective of relative risk to guide management priorities to reduce pollutant impacts on the GBR. This aspect is being addressed in Star et al. (2015).

Assessment of anthropogenic nutrient loads. There is insufficient knowledge of the sources of anthropogenic nutrient sources in the region to make recommendations about management priorities for these variables. Further knowledge of the role of particulate nitrogen, which is largely derived from grazing lands, and the processing of this into dissolved inorganic nitrogen is important for making future management recommendations in the large grazing catchments of the Fitzroy region. Current research is showing differences in sediment and nutrient run-off from native brigalow scrub, newly planted legume-based ley pastures and cropping lands, which will be useful to guide management priorities in the future. Compared to the native brigalow scrub landscape, cropping exported more TSS and DIN, while grazing exported less total nitrogen and DIN, but more TSS (Thornton & Elledge 2013; 2014). The legume pastures do appear to pose a risk to water quality as they contribute higher nutrient loads than grass-only pasture systems, established grass-leucaena pastures, and the native brigalow scrub landscape representative of the environment in its pre-European condition (Thornton & Elledge 2013; 2014). The way that these variables are incorporated into the Source Catchments modelling also requires further attention to enable more detailed assessments of anthropogenic nutrient sources to be undertaken for the Fitzroy region.

These limitations have been translated into priority information needs for future risk assessments of water quality in the GBR:

1. Scoping the availability of, and acquisition of, more consistent temporal and spatial data for all water quality variables (including those not included in the most recent assessment such as phosphorus and particulate nutrients) and their ecological impacts to enable improved classification in terms of ecological risk and application of a formal risk assessment framework (which includes assessments of likelihood and consequence).
2. Refining the approach to estimate ‘zones of influence’ for each river.
3. Better understanding of the responses of key GBR ecosystem components to cumulative impacts of repeated exposure to poor water quality, and the cumulative impacts of multiple water quality pressures.
4. Validating the remote sensing data for turbidity, particularly in areas that are known to be naturally highly turbid or where existing validation data is limited, such as in Shoalwater Bay and Broad Sound.

5. Better understanding of the prevalence and associated effects of other pollutants (e.g. microplastics, endocrine disrupting substances, oil and polycyclic aromatic hydrocarbons, pharmaceuticals and heavy metals) on GBR ecosystems.
6. Improved measurement and understanding of the sources of anthropogenic nutrients in the region, and the delivery and fate of particulate nutrients and importance for coastal and marine ecosystems.
7. Extending the habitat assessments beyond coral reefs and seagrass to include coastal ecosystems such as freshwater and coastal wetlands, mangroves and estuarine environments, and non-reef bioregions.

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7 Closure

This document was prepared by FBA in collaboration with our Program partners. If you have any questions or require additional details, please contact the undersigned.

Our country, Our future.

Fitzroy Basin Association Incorporated
PO Box 139, Rockhampton QLD 4700
Phone: (07) 4999 2800
Fax: (07) 4921 2860

www.fba.org.au

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