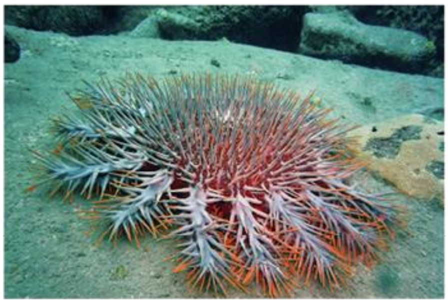
2017 Scientific Consensus Statement

CHAPTER THREE

The risks from anthropogenic pollutants to Great Barrier Reef coastal and marine ecosystems



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This document was prepared by a panel of scientists with expertise in Great Barrier Reef water quality. This document does not represent government policy.

Contents

Acronyms, units and definitions

Acronyms

CDF = cumulative distribution function
CDOM = colour dissolved organic matter
Chl-*a* = chlorophyll *a*
CoTs = Crown-of-Thorns starfish
DIN = dissolved inorganic nitrogen
DIN Anth. = anthropogenic dissolved inorganic nitrogen
DIP = dissolved inorganic phosphorus
ERA = ccological risk assessment
ETVs = ecotoxicity threshold values
Kd = light attenuation coefficient
LOR = limit of reporting
MoA = modes of action
ms-PAF = multisubstance-Potentially Affected Fraction
NetCDF = network common data form
NRM = natural resource management
PERA = probabilistic ecological risk assessment
PN = particulate nitrogen
PP = particulate phosphorus
PS wet season = primary and secondary wet season water types
PSII herbicides = photosystem II inhibiting herbicides
SSDs = species sensitivity distributions
TSS Anth. = anthropogenic total suspended sediments
TSS = total suspended sediment[1]

Units

km^3 = cubic kilometres
kt = kilotonnes
m = metre
mg/L = milligram per litre
$mol/m^2/d$ = moles of light per square metre per day
t = tonnes
µg/L = micrograms per litre
µm = micrometres (microns)

[1] TSS is also often referred to as total suspended solids.

Definitions

Basin: There are 35 basins that drain into the Great Barrier Reef. A basin can be made up of a single or multiple rivers (e.g. North and South Johnstone rivers belong to one basin, the Johnstone Basin). Basins are primarily used here when discussing the relative delivery of a pollutant to the marine system.

Catchment: The natural drainage area upstream of a point that is generally on the coast. It generally refers to the 'hydrological' boundary and is the term used when referring to modelling in this document. There may be multiple catchments in a basin.

Coastal ecosystems: Coastal freshwater wetlands and estuarine systems connect the land and sea and have the potential to influence the health and resilience of the Great Barrier Reef. This includes the Great Barrier Reef catchment and 10% of the Reef waters seawards of the coastline (GBRMPA, 2012). The risk assessment in Chapter 3 specifically includes floodplain wetlands (vegetated swamps and lakes) and floodplains, in line with the scope of the Reef Water Quality Protection Plan.

Ecological risk assessment: The process of determining the nature and likelihood of effect of anthropogenic actions on animals, plants and the environment (SETAC, 1997; US EPA, 1998). It is a systematic process for estimating the likelihood of occurrence (or probability) and the severity of the consequences (or magnitude) of the effects of human actions or natural events on ecosystems of ecological value and their sustainability (modified from Hart et al., 2005).

Ecosystem health: Ecosystem health is defined here as the state or condition of an ecosystem in which its dynamic attributes are expressed within normal ranges of activity relative to its ecological stage of development.

Hazard: A situation that poses a level of pressure or threat to ecosystem health.

Management unit: There are 47 management units in the Great Barrier Reef catchment, which incorporate the 35 basins that drain directly to the Great Barrier Reef including additional internal catchments or management units within the Burdekin and Fitzroy basins.

Other pollutants: Includes pollutants such as antifouling paints, coal particles, metals and metalloids, marine debris/microplastics, personal care products, petroleum hydrocarbons, and pharmaceuticals. In addition, contaminants such as nanomaterials, perfluorooctane sulfonate and perfluorooctanoic acid may be present, but no monitoring information is available for the Great Barrier Reef lagoon (Kroon et al., 2015a).

Pesticides: Herbicides, insecticides and fungicides.

Pollutants: Pollution means the introduction by humans, directly or indirectly, of substances or energy into the environment resulting in such deleterious effects as harm to living resources, hazards to human health, hindrance to aquatic activities including fishing, impairment of quality for use of water and reduction of amenities (GESAMP, 2001). This document refers to suspended (fine) sediments, nutrients (nitrogen, phosphorus) and pesticides as 'pollutants'. Within this chapter we explicitly mean enhanced concentrations of or exposures to these pollutants, which are derived from (directly or indirectly) human activities in the Great Barrier Reef ecosystem or adjoining systems (e.g. river catchments). Suspended sediments and nutrients naturally occur in the environment; all living things in ecosystems of the Great Barrier Reef require nutrients, and many have evolved to live in or on sediment.

Risk: The likelihood that an adverse effect will occur as a result of ecosystem exposure to a certain concentration of the stressor. Risk exists when there is the possibility of adverse or unintended consequences. It is often quantified as the product of the likelihood of an event occurring and the consequences (also measured as effects) of that event (US EPA, 1998).

Risk factors

- **Likelihood Score**: The likelihood of exposure of coral reefs and seagrass to total suspended sediments and dissolved inorganic nitrogen for each Marine Zone using the area of coral reefs and seagrass in the highest likelihood classes.
- **Load Index:** The proportional contribution of the anthropogenic pollutant load from each basin to the total anthropogenic pollutant load for each Marine Zone. The anthropogenic load is calculated as the difference between the long-term average annual load and the estimated pre-development annual load.
- **Likelihood Index:** Attributes the Likelihood Score for each basin using the Load Index: Likelihood Score x Load Index.
- **Consequence Score:** Areas of coral reef and seagrass that are exposed to nutrient and sediment effects; examples provided are Crown-of-Thorns starfish (link between dissolved inorganic nitrogen and coral reefs) and reduced light (link between total suspended solids and seagrass).
- **Risk Index:** The likelihood that an adverse effect will occur as a result of ecosystem exposure to dissolved inorganic nitrogen or total suspended solids: Likelihood Index x Consequence Score.

Region: There are six natural resource management (NRM) regions covering the Great Barrier Reef catchments. Each region groups and represents catchments with similar climate and bioregional setting, with boundaries extending into the adjacent marine area. The regions are Cape York, Wet Tropics, Burdekin, Mackay Whitsunday, Fitzroy and Burnett Mary.

Water types: The wet season water types are produced using MODIS true colour imagery reclassified to six distinct colour classes defined by their colour properties. The wet season water types are regrouped into three water types (primary, secondary and tertiary) characterised by different concentrations of optically active components (suspended sediment, colour dissolved organic matter and chlorophyll *a*), which control the colour of the water and influence the light attenuation, and different pollutant concentrations:

- Primary water type (colour classes 1–4): Corresponds to the brownish to brownish-green turbid water masses. These waters have high nutrient and phytoplankton concentrations but are also enriched in sediment and dissolved organic matter and have reduced light levels. They are typical for nearshore areas or inshore regions of flood river plumes.
- Secondary water type (colour class 5): Corresponds to the greenish to greenish-blue water masses and are typical of coastal waters dominated by algae, but also with some dissolved matter and some fine sediment present. Relatively high nutrient availability and increased light levels due to sedimentation favour an increased coastal productivity in this water type. This water type is typical for the coastal waters or the mid-region of river plumes.
- Tertiary water type (colour class 6): Transitional, greenish-blue water mass with slightly above ambient turbidity and nutrient concentrations. This water type is typical for areas towards the open sea or offshore regions of flood river plumes.

Time frames: The datasets used in this assessment are typically defined as ‘current’, which is 2011 to 2014 using the eReefs model, and ‘longer term’ which is 2003 to 2016 using the water type mapping. The modelled baseline is set at 2012-2013.

Note: Inshore coral reefs are equivalent in terminology here as inner shelf coral reefs, as distinct from mid-shelf reefs and outer shelf reefs.

Acknowledgements

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Executive summary

In this chapter, we applied an ecological risk assessment approach to assess the likelihood of exposure and potential risks from land-based pollutants to Great Barrier Reef coastal (floodplain wetlands and floodplains) and marine (coral reefs and seagrass meadows) ecosystems. Ecological risk is defined as the product of the *likelihood* of an effect occurring and the *consequences* if that effect was to occur.

The main water quality pollutants of concern to Great Barrier Reef coastal aquatic and marine ecosystems are enhanced levels of suspended sediments, excess nutrients and pesticides (predominantly photosystem II inhibiting [PSII] herbicides) discharged to the Great Barrier Reef lagoon from the adjacent catchments (refer to Chapter 2). The distinct wet and dry seasonal climate of the Great Barrier Reef results in most sediment, nutrients and pesticides being delivered to the Great Barrier Reef lagoon during the summer wet season (December–April) when high river discharge occurs, forming distinctive river plumes in the coastal zone that can move north along the coast but can occasionally move out towards the mid- and outer shelf area. In the dry season, sediments and nutrients can be remobilised by wind-driven resuspension, leading to conditions of elevated turbidity year-round, particularly in inshore areas. Coastal (floodplain wetlands and floodplains) ecosystems are similarly influenced by seasonal conditions. First-flush run-off during the early wet season can result in inputs of elevated pollutant loads. During the dry season, wetland water quality can be affected by irrigation and other localised run-off or cattle and other animal disturbance depending on location. The assessment of the likelihood of exposure of pollutants to marine ecosystems (coral reefs and seagrass) (Section 6) used several spatial layers to represent nutrients and sediments in wet season and annual average conditions. The factors were the distribution and frequency of anthropogenic dissolved inorganic nitrogen and fine sediment (referred to suspended sediment) loading in the wet season and assessment of the degree of difference between current (baseline) average annual concentration of chlorophyll *a* and light attenuation compared to pre-development load scenarios (derived from the eReefs coupled hydrodynamic-biogeochemical model).

The assessment included all 35 basins that discharge into the Great Barrier Reef, and the risk to marine ecosystems was assessed within eight Marine Zones: Cape York North, Cape York Central, Cape York South, Wet Tropics, Burdekin, Mackay Whitsunday, Fitzroy and Burnett Mary (see Appendix 1 for details). The boundaries for these Marine Zones differ from the marine natural resource management regions as they better reflect the collective influence of rivers which may extend across natural resource management boundaries. The Marine Zones typically incorporate the enclosed coastal and inner shelf water bodies and, in the northern areas, mid-shelf areas.

There were three primary steps in the marine assessment, each conducted separately for total suspended sediments and dissolved inorganic nitrogen:

1. ***Calculate the likelihood of pollutant exposure (A) (Section 6)***

- Likelihood Score (A1) = The likelihood of exposure of coral reefs and seagrass to total suspended sediments and dissolved inorganic nitrogen for each Marine Zone using the area of coral reefs and seagrass in the highest likelihood classes.
- Load Index (A2) = The proportional contribution of the anthropogenic pollutant load from each basin to the total anthropogenic pollutant load for each Marine Zone. The anthropogenic load is calculated as the difference between the long-term average annual load and the estimated pre-development annual load.
- Likelihood Index (A) = Attributes the Likelihood Score for each basin using the Load Index.

(A) Likelihood Index = (A1) Likelihood Score x (A2) Load Index

2. ***Calculate the consequence of pollutant exposure (B) (Section 7)***
   Consequence Score (B) = Areas of coral reef and seagrass that are exposed to Crown-of-Thorns starfish (dissolved inorganic nitrogen and coral reefs) and reduced light (total suspended solids and seagrass). The consequence, and therefore the risk, assessments were limited to two examples due to knowledge limitations: (i) the risk of dissolved inorganic nitrogen and the area of influence from Crown-of-Thorns starfish on coral reefs, and (ii) the risk of the benthic light thresholds for seagrass being exceeded due to excessive concentrations of fine sediment.
3. ***Calculate marine Risk Index (Section 8)***
   Risk Index = The likelihood that an adverse effect will occur as a result of ecosystem exposure to dissolved inorganic nitrogen or total suspended sediments.

   Risk Index = (A) Likelihood Index x (B) Consequence Score

Pesticides are also a pollutant of concern but were treated separately as it was not possible to conduct a full pesticide risk assessment for marine ecosystems at this stage. A case study is presented to demonstrate the capacity to model pesticide risk to seagrass and coral reefs in the future (Appendix 4). The risk assessment was performed using two methods that assess consequence and likelihood. Consequence was first determined using the multisubstance-Potentially Affected Fraction (ms-PAF) method. The analysis assessed whether concentrations of pesticides (as a mixture of five PSII herbicides) entering the Great Barrier Reef World Heritage Area would be protective of 99% of species. This approach assessed the compliance of monitoring data with the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000), and the Great Barrier Reef Water Quality Guidelines (GBRMPA, 2010). Likelihood could then be determined using methods of a probabilistic ecological risk assessment: the area under the curve of the ms-PAF cumulative frequency distribution. An ecotoxicity threshold assessment was completed for 28 individual pesticides (for which threshold values are available) collected over a three-year period (2013-2016), as many of these pesticides were not analysed prior to 2013.

A risk assessment of emerging pollutants (recently completed as part of the National Environmental Science Programme) on Great Barrier Reef ecosystems was also incorporated.

The key results are summarised below in conjunction with additional supporting evidence from published literature.

What is the likelihood of exposure of key pollutants to Great Barrier Reef aquatic coastal and marine systems, and when is the exposure from degraded water quality most likely to be highest? (Section 6)

- The greatest exposure of coral reef and seagrass to dissolved inorganic nitrogen is from the Herbert, Haughton, Johnstone, Russell-Mulgrave, Tully, Plane and Murray basins. The greatest exposure of coral reef and seagrass to fine sediment is from the Burdekin, Fitzroy, Mary, Herbert, Johnstone and Burnett basins.

- Anthropogenic particulate nitrogen is also likely to be of some importance in the same areas, as well as in the Fitzroy Basin; however, our knowledge on the bioavailabity of particulate nitrogen to the marine ecosystems relative to that of dissolved inorganic nitrogen is still limited.

- Given the small anthropogenic loads of dissolved organic nitrogen from most basins, and its limited bioavailablity, it is considered to be less important than dissolved inorganic nitrogen.
- Floodplain wetlands in six management units / basins (Dawson, Lower Burdekin, Herbert, Burnett, Burrum and Tully) have high likelihood of exposure to sediment, nutrient and pesticide pressures (Section 6.3). The areas of greatest likelihood of exposure of floodplain wetlands to nutrient pressures are in the Fitzroy and Dawson; for exposure to sediments, the Dawson and Lower Burdekin; and for exposure to pesticides, the Lower Burdekin and Herbert basins.
- Floodplains in seven management units / basins (Tully, Belyando, Plane, Dawson, Comet, Kolan and Burnett) have high likelihood of exposure to sediments, nutrients and pesticides (Section 6.3). The areas of greatest likelihood of exposure of floodplains to nutrient inputs are in the Belyando and Dawson; for exposure to sediments, the Dawson, Isaac and Mackenzie; and for exposure to pesticides, the Herbert, Lower Burdekin, Belyando, Pioneer and Plane basins.

What are the consequences of the water quality exposure? (Section 7)

- The greatest area of reefs in the High consequence class for Crown-of-Thorns starfish are in the Wet Tropics Marine Zone, followed by the Cape York South Marine Zone and, to a lesser extent, the Burdekin Marine Zone. None of the other Marine Zones contain reefs in the Crown-of-Thorns starfish influence area and are therefore not within the High consequence class for the consequence assessment.
- The greatest limitation in meeting benthic light thresholds was predicted in the Burnett Mary and Cape York South Marine Zones for surveyed seagrass and the Cape York South and Wet Tropics Marine Zones for modelled deepwater seagrass.

What is the risk from degraded water quality to Great Barrier Reef ecosystem health? (Section 8)

Nutrients and sediments

- The greatest area of risk to coral reefs from Crown-of-Thorns starfish influence is in the Wet Tropics Marine Zone followed by the Cape York South Marine Zone and, to a lesser extent, the Burdekin Marine Zone. The basin-scale assessment (estimated by linking the results to end-of-catchment dissolved inorganic nitroegen loads) indicates that the Herbert Basin has the greatest contribution to dissolved inorganic nitrogen risk to coral reefs. This is followed by the Johnstone, Russell-Mulgrave and Tully basins but to a lesser extent (approximately 50% lower than for the Herbert Basin).
- The greatest area of risk to surveyed seagrass from benthic light limitation is in the Burdekin Marine Zone followed by the Burnett Mary Marine Zone. For modelled deepwater seagrass, the greatest risk was predicted in the Burnett Mary Marine Zone followed by the Wet Tropics and Fitzroy Marine Zones. The basin-scale assessment (estimated by linking the results to end-of-catchment dissolved inorganic nitrogen loads) indicates that the Burdekin Basin has the greatest contribution to total suspended sediments risk to surveyed seagrass and total seagrass area. The Fitzroy Basin has the greatest contribution to total suspended sediments risk to modelled deepwater seagrass, and ranks second for surveyed and total seagrass area.

Pesticides

- Only a few basins present a Very High to Moderate risk to end-of-catchment ecosystems from PSII herbicides, with diuron presenting the highest risk. These basins are generally characterised as smaller coastal catchments with high proportions of sugarcane land use (i.e. basins within the Mackay Whitsunday region, Lower Burdekin and Wet Tropics).

- Management units that contribute the greatest potential pesticide exposure to floodplain wetland ecosystems are the Herbert and Lower Burdekin.
- Management units that contribute the greatest potential pesticide exposure to floodplain ecosystems are the Herbert, Lower Burdekin, Belyando, Pioneer and Plane.
- The ecotoxicity threshold assessment demonstrated that Great Barrier Reef ecosystems are exposed to a large number of other types of pesticides, some of which were a high risk on their own. Of the pesticides that indicated a risk to freshwater and estuarine ecosystems (i.e. <95% species protection) and to the Great Barrier Reef World Heritage Area (99% species protection), imidacloprid had a Very High to Moderate risk in a number of basins, and hexazinone, metolachlor and imazapic had a High to Moderate risk in some catchments.
- A case study presented here demonstrates the utility of the eReefs hydrodynamic model to model pesticide exposure and risk to seagrass and coral areas in the marine area.

Other pollutants

- In a qualitative risk assessment of emerging pollutants, marine plastic pollution poses the highest risk to the Great Barrier Reef marine ecosystems, particularly in the Cape York NRM region due to exposure to oceanic and local shipping sources. This is followed by chronic contamination of water and sediments with antifouling paints, and exposure to certain personal care products in natural resource management regions south of Cape York. The qualitative risks of all other emerging pollutants are relatively low with some minor differences between NRM regions.

Conclusions

This assessment has shown that the primary pollutants of concern to Great Barrier Reef coastal and marine ecosystems, that is, sediments, nutrients and pesticides, are all important at different scales and different locations. A summary table of the results in Section 9 highlights that several basins are identified as high exposure for two or more pollutants. These include the Russell-Mulgrave, Johnstone, Tully, Haughton, Burdekin, O'Connell, Pioneer, Plane, Fitzroy, Burnett and Mary.

This assessment and the supporting literature also show that:

- Exposure to dissolved inorganic nitrogen is most significant to all inner shelf areas and the mid-shelf area between Lizard Island and Townsville adjacent to basins with high anthropogenic dissolved inorganic nitrogen loads. The relative importance of dissolved inorganic nitrogen to seagrass ecosystems is still uncertain, but it may influence light availability for deepwater seagrass in areas deeper than 10–15 m due to increased phytoplankton growth.
- The greatest exposure of coral reef and seagrass to dissolved inorganic nitrogen is from the Herbert, Haughton, Johnstone, Russell-Mulgrave, Tully, Plane and Murray basins. The Herbert, Johnstone, Russell-Mulgrave and Tully basins also contribute the greatest dissolved inorganic nitrogen risk to coral reefs and primary Crown-of-Thorns starfish outbreaks.
- The Dawson and Lower Fitzroy management units contribute the greatest exposure of floodplain wetland ecosystem to nutrients. The Belyando and Dawson contribute the greatest exposure of floodplain ecosystems to nutrients.
- Exposure to fine sediment is most significant to areas of shallow seagrass and coral reefs on the inner shelf adjacent to basins with high anthropogenic fine sediment loads.
- The greatest exposure of coral reef and seagrass to fine sediment is from the Burdekin, Fitzroy, Mary, Herbert, Johnstone and Burnett basins. The Burdekin and Fitzroy basins also contribute the greatest fine sediment risk to seagrass ecosystems.

- The Dawson, Isaac and Mackenzie management units contribute the greatest exposure of floodplain wetland ecosystem to sediment. The Dawson and Lower Burdekin contribute the greatest exposure of floodplain ecosystem to sediment.
- Pesticides pose the greatest risk to ecosystems closest to the source of the pesticides; that is, freshwater wetlands, rivers and estuaries are exposed to the highest concentrations, followed by coastal ecosystems, seagrass and coral. Our understanding, at this stage, of the spatial exposure of pesticides in the marine area is very limited. However, the case study presented here with the use of the eReefs hydrodynamic modelling demonstrates the utility of this model for assessing the spatial exposure of pesticides in the marine area. It is anticipated that future risk assessments of pesticides will be conducted for the marine area using the eReefs hydrodynamic model and therefore lead to a better understanding of the risks that pesticides pose to coastal, seagrass and coral ecosystems.
- The Herbert and Lower Burdekin contribute to the greatest exposure of floodplain wetland ecosystems to pesticides. The Herbert, Lower Burdekin, Belyando, Pioneer and Plane contribute to the greatest exposure of floodplain ecosystems to pesticides.

Significant data limitations exist in the Cape York natural resource managment region; therefore, it is difficult to make conclusions about this region with confidence. Enough evidence is available to conclude that overall the eastern Cape York catchments currently present a relatively low risk to adjacent coastal and marine ecosystems. The basins in the Cape York Central Marine Zone – the Normanby, Hann and Stewart catchments – are likely to pose a risk to ecosystems in the Princess Charlotte Bay area from degraded water quality, particularly increased turbidity in wet season conditions. Until the 2016 bleaching event, the coral reef ecosystems in the Cape York region were typically in good condition. Due to the potential underestimation and lack of validation of models pertaining to risks in the Cape York South Marine Zone, this region also warrants further investigation and management of threats to water quality.

The limitations of the risk assessment have been translated into priority information needs for future risk assessments of water quality in the Great Barrier Reef:

- scoping of the availability 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)
- refinement of the approach to estimate ‘zones of influence’ for each basin
- limitations to nutrient measurements and chlorophyll *a* spatially and temporally. Direct measurement of chlorophyll *a* in the Great Barrier Reef lagoon is still limited in sample numbers and locations of sampling. Estimates of chlorophyll *a* concentrations can be made from water type analysis and by using the eReefs model in conjunction with direct measurements. However, a more intensive direct measurement program is still required to be able to answer questions regarding the influence of nutrient enrichment on populations of Crown-of-Thorns starfish
- 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 Great Barrier Reef ecosystems
- extending the habitat assessments beyond coral reefs and seagrass to other marine ecosystems and coastal aquatic ecosystems such as floodplain wetlands, floodplains, freshwater wetland and estuarine environments (mangrove and saltpan) and non-reef bioregions

- incorporation of the principles of conservation management and the increasing need to protect areas in the Great Barrier Reef and its catchments that are in good condition as many parts of the Great Barrier Reef ecosystem become more degraded.

Further discussion of the improvements to the 2013 assessment and the limitations to the current assessment is presented in Section 10.

The results of this new assessment provide an improved analysis of the likelihood of exposure of nutrients, sediments and pesticides to coastal aquatic and marine ecosystems. This information can be used to inform management priorities for improving water quality from the Great Barrier Reef catchments that is discharged into the Great Barrier Reef World Heritage Area.

1. Introduction

This Scientific Consensus Statement applies a risk management framework based on the ISO 31000 (AS/NZS, 2004) shown in Figure 1. Chapter 1 describes Great Barrier Reef coastal and marine ecosystem status and condition, identifies the primary hazards to these systems and the known effects of land-based pollutants and other contaminants based on understanding derived through monitoring and modelling (Schaffelke et al., 2017). Chapter 2 describes the sources of pollutants, otherwise considered as the hazard to Great Barrier Reef ecosystems (Bartley et al., 2017). This chapter applies the risk assessment components of the framework by evaluating the likelihood, consequences and quantified risk to the Great Barrier Reef coastal and marine ecosystems, specifically from different nutrient constituents, suspended sediment (including different size fractions) and pesticides. Chapter 4 considers management of the risks.

Knowledge of marine ecosystem exposure in Great Barrier Reef has improved, providing greater confidence in the ability to assess the risk of degraded water quality to Great Barrier Reef marine ecosystem health. However, there is still insufficient data and knowledge concerning the exposure, thresholds and effects for coastal aquatic ecosystems such as wetlands. This gap currently constrains full consideration of the risk of degraded water quality to these ecosystems and related impacts on ecological functions at the local or basin scale. In this assessment, we assess the likelihood of exposure of floodplain wetlands, floodplains, coral reefs and seagrass to pollutants, and we provide examples of consequence and therefore risk for coral reefs and seagrass. In addition, we draw on a recent review on contaminants other than sediment, nutrients and pesticides to examine their potential risk to Great Barrier Reef and Torres Strait marine ecosystems.

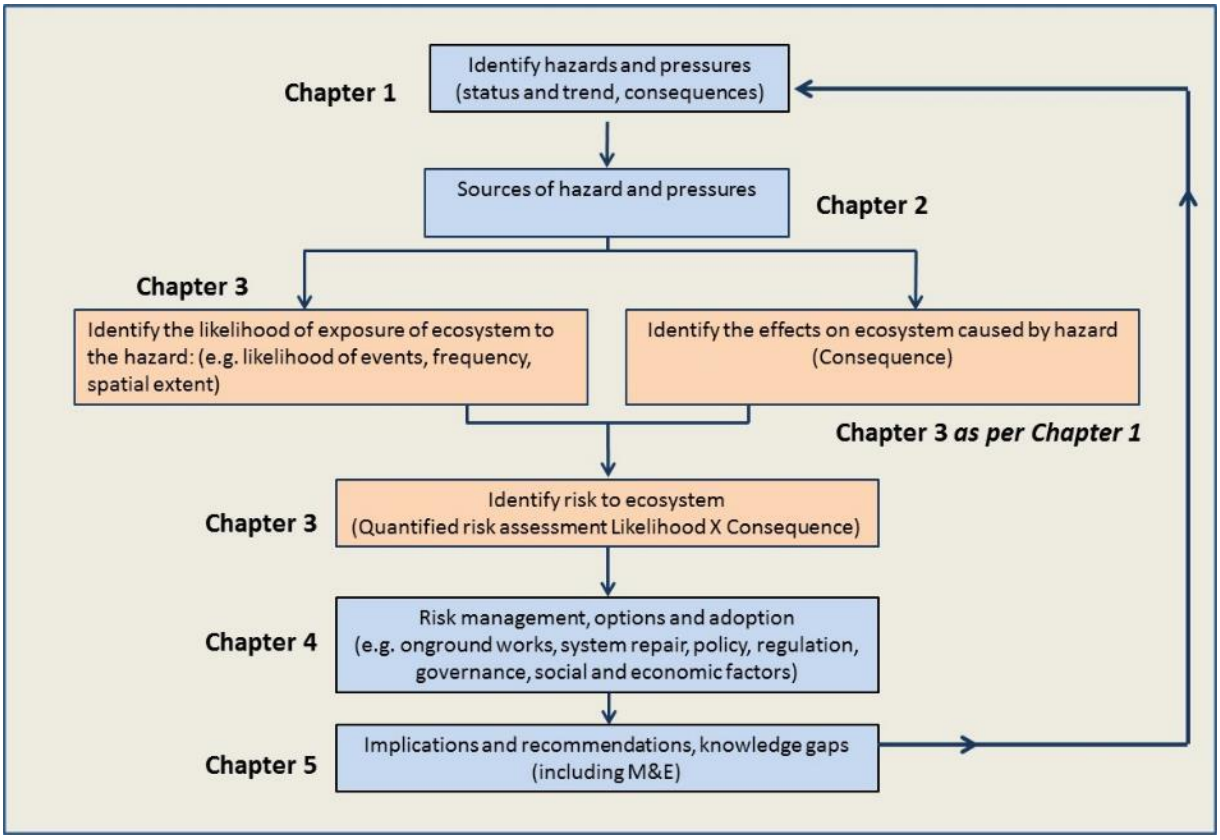


Figure 1. Risk management framework adopted for the 2017 Scientific Consensus Statement. Derived from AS/NZS (2004).

This chapter specifically addresses the overarching question: *What are the risks to ecosystem health in the Great Barrier Reef from degraded water quality arising from catchment land use?* with the following sub-questions:

- Drawing on chapters 1 and 2, what are the water quality hazards that pose the greatest potential risk to Great Barrier Reef aquatic coastal and marine ecosystems?
- What is the likelihood of exposure of key pollutants to Great Barrier Reef aquatic coastal and marine systems, and when is the exposure from degraded water quality most likely to be highest?
- What are the consequences of the water quality exposure?
- What is the risk from degraded water quality to Great Barrier Reef ecosystem health?

This chapter presents the risk assessment specifically completed to inform the 2017 Reef Water Quality Protection Plan update and draws on (i) regionally specific studies conducted between 2014 and 2016 to inform the regional Water Quality Improvement Plans, (ii) a landscape hazard assessment for wetlands by the Department of Science, Information Technology and Innovation (DSITI, 2015), and (iii) other peer-reviewed, published literature.

2. Previous findings

In the 2013 Scientific Consensus Statement, a combination of qualitative and semi-quantitative assessments were used to estimate the relative risk from water quality constituents to Great Barrier Reef ecosystems' health from major sources in the Great Barrier Reef catchments, focusing on agricultural land uses (Brodie et al., 2013a). Marine risk was defined as the area of coral reefs and seagrass within a range of assessment classes (Very Low to Very High relative risk) for several water quality variables in each natural resource management region. The variables included ecologically relevant thresholds for concentrations of total suspended solids and chlorophyll *a* from daily remote sensing observations and the distribution of key pollutants including total suspended sediment (TSS), dissolved inorganic nitrogen (DIN) 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 related to water quality influences on Crown-of-Thorns starfish outbreaks was included for coral reefs. The main finding was that increased loads of fine sediments, nutrients (nitrogen and phosphorus) and pesticides all pose a high risk to some parts of the Great Barrier Reef. It concluded that the risk to the marine ecosystem differs depending on the individual pollutants, between the source catchments and with distance of the ecosystem from the coast. The key findings from 2013, new information or insights and contentious, unresolved or unknown areas are summarised in Appendix 1. Coastal aquatic ecosystems were not included in the 2013 assessment and are included here for the first time, specifically, floodplain wetlands (i.e. vegetated swamps and lakes) and floodplains.

Prior to 2013, assessments of the relative risk of degraded water quality on Great Barrier Reef ecosystems were largely undertaken at a Great Barrier Reef–wide scale, with relative assessments between natural resource management regions (Brodie et al., 2013a; Waterhouse et al., 2012; Brodie and Waterhouse, 2009; Cotsell et al., 2009; Greiner et al., 2005) and, to a lesser extent, individual basins (Australian Government, 2014). The results of these assessments have been used to inform prioritisation across the natural resource management regions in terms of management effort (such as Reef Water Quality Protection Plan 2009 and 2013, the Queensland *Great Barrier Reef Protection Amendment Act, 2009*) or investment, including several Reef Water Quality Protection Plan initiatives. Since 2013, there has been more effort in regional-scale assessments to support the update and development of regional Water Quality Improvement Plans.

Several improvements in catchment modelling (see McCloskey et al., 2017a; McCloskey et al., 2017b for most recent published data), marine modelling (Brinkman et al., 2014; Schiller et al., 2014; Baird et al., 2016; Jones et al., 2016) and availability of longer time series of monitoring data to support this modelling effort have 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 (e.g. Waterhouse et al., 2016a; Waterhouse et al., 2016b). Better understanding of ecological thresholds for coral reefs and seagrass improves the ability to assess the impacts of water quality exposure. The Water Quality Improvement Plans assessments were based on revised methodology advanced from the relative risk assessment undertaken for the whole Great Barrier Reef for the 2013 Scientific Consensus Statement (see Brodie et al., 2013b) and modified for regional applications (Waterhouse et al., 2014a; Waterhouse et al., 2014b; Waterhouse et al., 2015a; Waterhouse et al., 2016a; Waterhouse et al., 2016b). Basin-scale priorities were identified in each region; however, the methods varied slightly between regions and are therefore not directly comparable to inform a Great Barrier Reef–wide assessment.

Part A: Hazard and systems at risk

3. Risk assessment framework

Ecological risk assessment 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* exists when there is the possibility of adverse or unintended consequences. 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 (Hart et al., 2005). A hazard is something that is likely to cause harm. In this context, the hazard is the source of the risk, largely described in Chapter 2 (Bartley et al., 2017).

Water quality within the Great Barrier Reef and its catchment is influenced by many factors (see chapters 1 and 2 for detailed descriptions). The primary influences are land-use contributions of pollutants, the volume and timing of seasonal rainfall and subsequent run-off events, which are determined by the monsoonal climate and extreme weather events (cyclones), tidal regimes and currents. These factors influence the relative risk of different pollutants at particular locations and to different habitats in the Great Barrier Reef and its catchment.

Different parts of the Great Barrier Reef 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 catchment (Brodie et al., 2012a) and subsequent events such as wind-driven resuspension leading to prolonged exposure (Fabricius et al., 2016). This differential spatial and temporal exposure to land-sourced pollutants results in varying levels of direct and indirect risks to coastal and marine ecosystems in the Great Barrier Reef including coral reefs and seagrass and wetland systems. Understanding these differences is important for prioritising investment between management areas. 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 and Pollino, 2008).

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 account for 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.

The *consequences* are the measured effects of the exposure. Current knowledge of the effects of degraded water quality on the health of coastal and marine ecosystems in the Great Barrier Reef are summarised in Chapter 1 (Schaffelke et al., 2017), but these ecological effects are still difficult to quantify for Great Barrier Reef coastal and marine ecosystems. Furthermore, 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 and their episodic nature (refer to Chapter 1).

The 2013 risk assessment (see Brodie et al., 2013a) incorporated factors that represent marine water quality in the context of water quality guidelines and thresholds, the influence of Crown-of-Thorns starfish and a factor representing end-of-catchment load contributions to assess the relative risk of degraded water quality among natural resource management regions. This method was further developed for the regional Water Quality Improvement Plans and conducted at a basin scale (Waterhouse et al., 2014a; Waterhouse et al., 2014b; Waterhouse et al., 2015a; Waterhouse et al., 2016a; Waterhouse et al., 2016b). These assessments were largely based on analysis of the likelihood of exposure of pollutants. While our knowledge of the consequence of degraded water quality has improved in the last three years, our ability to quantify the effects of the exposure to degraded water quality to coral reefs and seagrass is still limited. This assessment presents two examples of quantified consequence assessments for coral reefs and seagrass (to calculate risk); further analysis could be conducted with additional time and resource allocation.

Advances in understanding and new themes in this updated assessment include:

- a shift of focus from regions and towards basins
- incorporation of a hazard and likelihood of exposure assessment of wetland and floodplain ecosystems to expand the scope of ecosystems being considered in the assessment. Land-use driven pressures underpin this assessment
- inclusion of a pesticide risk assessment for freshwater and estuarine systems to recognise the importance of pesticide toxicity in these ecosystems; the marine assessment is still under development and is not quantified at this stage
- new knowledge on the timing, movement and transformation of pollutants within the Great Barrier Reef lagoon that will be used to assist in interpretation of the quantitative assessment
- consideration of the relative importance of all land use when linking marine risk to the basins
- recognition of the relative risk of emerging contaminants.

The scope of the assessment varies for different ecosystems due to data limitations:

- marine ecosystems: includes likelihood of exposure assessments for DIN and fine sediments, with an example of consequence and risk for each parameter. An assessment of pesticide risk directly in marine ecosystems has not been completed due to limitations in spatial and temporal pesticide data across the Great Barrier Reef
- coastal aquatic ecosystems: includes consideration of the likelihood of exposure for DIN, fine sediments and pesticides for floodplain wetlands and wetlands. The assessment of the consequences and risk for each parameter cannot be completed due to limitations in quantitative data across the Great Barrier Reef. A comprehensive risk analysis of

pesticides for freshwater and estuarine systems is presented, adopting multisubstance-Potentially Affected Fraction (ms-PAF) and probabilistic ecological risk assessment methods.

The main elements of the framework are shown in Figure 2. The approach is summarised in Box 1.

To provide justification for the methods and selection of input layers for the updated risk assessment, a summary of factors that influence the likelihood of pollutant exposure, the consequences and the risks from water quality in the Great Barrier Reef are presented below, structured using the questions being addressed in this chapter.

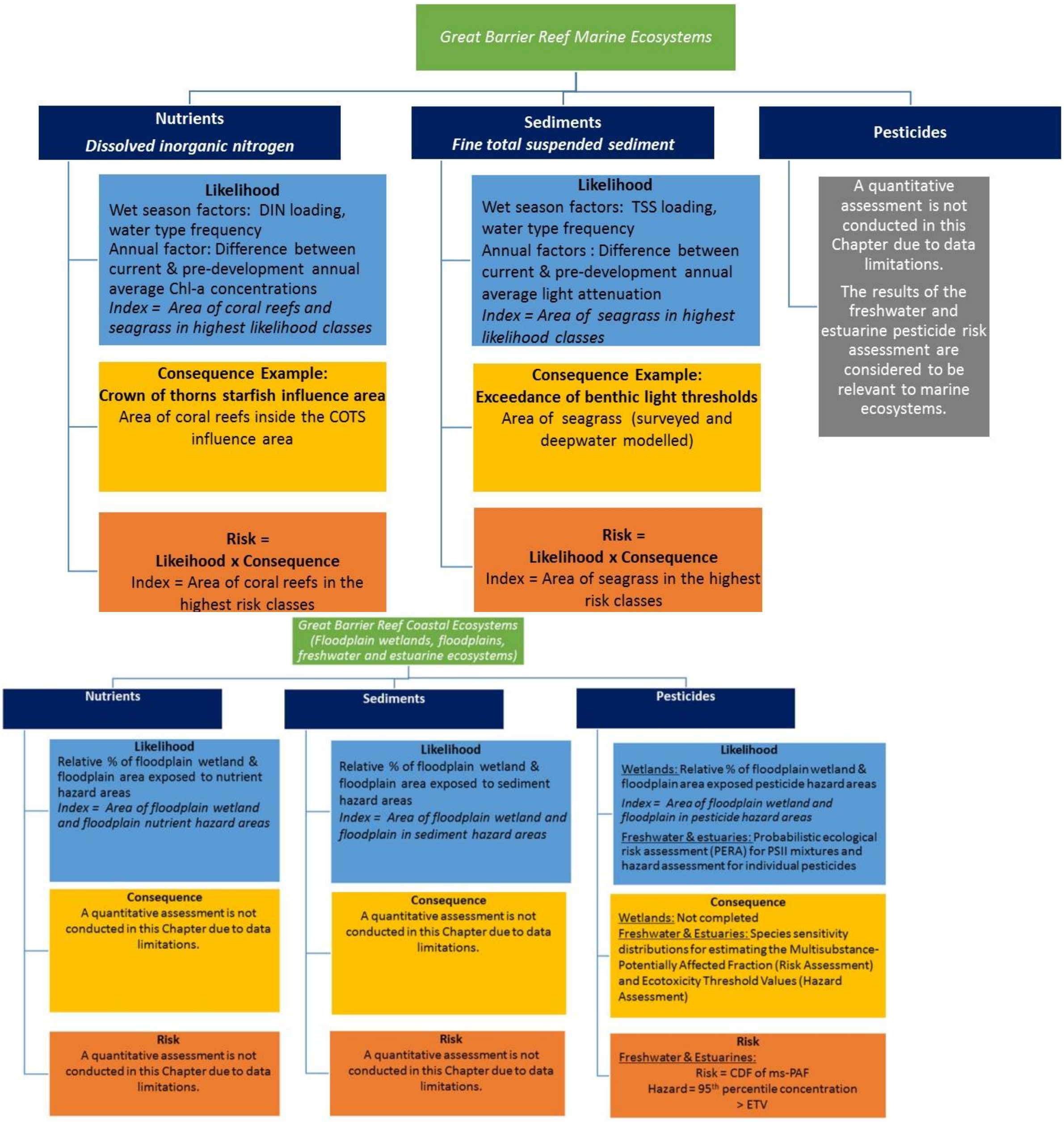


Figure 2. Framework for the assessment of the relative risk of degraded water quality to Great Barrier Reef coastal and marine ecosystems. Note that CDF refers to cumulative distribution function; ms-PAF refers to multisubstance-Potentially Affected Fraction; and ETV refers to Ecotoxicity Threshold Values.

BOX 1: Summary of the approach for assessing ecological risk in this chapter.

Step 1: Define assessment boundaries—spatial and temporal

- ***Marine zones:*** Regionally grouped areas of river influence in marine waters
- ***Marine habitats:*** Coral reefs and seagrass (surveyed composite and modelled deepwater) (data limitations exclude other ecosystems)
- ***Coastal aquatic ecosystems:*** Floodplain wetlands (lakes and vegetated swamps) and floodplains throughout the Great Barrier Reef catchment and above the tidal influence
- ***Catchments***: 35 basins for marine ecosystems; 47 management units (basins and catchments) for coastal aquatic ecosystems
- ***Time frame***: All eReefs modelled inputs are 2011-2014; other datasets are typically 2003-2016, presented as a multi-annual mean.

Step 2: Assess likelihood of exposure (Section 6)

Step 2a: Define Likelihood Score for each Marine Zone

Informed by identification of sources of risk (Chapter 2)

Marine ecosystems, for each Marine Zone:

- Assess frequency and area of influence of anthropogenic wet season and annual factors for nutrients (dissolved inorganic nitrogen - DIN) and fine sediments (TSS) for coral reefs and seagrass. Rate the areas High to Low.
- Calculate area of coral reefs and seagrass in highest likelihood categories to generate a Likelihood Score.
- Assess probability that the concentrations of pesticides (as a mixture) passing through the river mouth into the Great Barrier Reef lagoon exceed the concentrations that would be protective of 99% of species.

Coastal aquatic ecosystems:

- Calculate areas of floodplain wetlands and wetlands in High and Very High hazard areas for nutrients, sediments and pesticides. Apply a relative exposure classification.

Step 2b: Link likelihood of exposure to basins

Marine ecosystems, attribute Likelihood Scores for Marine Zones to each basin:

- Calculate DIN and TSS anthropogenic loads for Marine Zones and assess proportion of each basin's contribution to the total load for the Marine Zones to generate a Load Index.
- Multiply the basin Load Index by the Marine Zone Likelihood Score to generate a Likelihood Index for each basin.

Coastal aquatic ecosystems: The assessment is conducted within the 47 management units.

Step 3: Assess consequence of exposure (Section 7)

Informed by discussion of the impacts of pollutant exposure (Chapter 1)

Marine ecosystems, for each Marine Zone:

Example 1: DIN and coral reefs, Crown-of-Thorns starfish influence area. Calculate area of coral reefs in highest consequence category for Crown-of-Thorns starfish influence to generate a Consequence Score.

Example 2: TSS and seagrass, exceedance of benthic light thresholds. Calculate areas of seagrass in highest consequence categories to generate a Consequence Score.

Coastal aquatic ecosystems: Not completed due to data limitations.

Step 4: Assess ecological risk for Great Barrier Reef marine ecosystems (Section 8)

Step 4a: Assess examples of consequences on marine ecosystems from exposure to nutrients and sediments for each Marine Zone

Marine ecosystems, for each Marine Zone:

Example 1: DIN and coral reefs, Crown-of-Thorns starfish influence area.

- Multiply DIN Likelihood and DIN Consequence spatial layers to generate DIN Risk to coral reefs from Crown-of-Thorns starfish influence.
- Calculate area of coral reefs in highest risk categories to generate a Risk Score.

Example 2: TSS and seagrass, exceedance of benthic light thresholds.

- Combine TSS Likelihood and TSS Consequence spatial layers to generate TSS Risk to coral reefs and seagrass from reduced light.
- Calculate area of seagrass in highest risk categories to generate a Risk Score.

Coastal aquatic ecosystems: Not completed due to data limitations.

Step 4b: Assess examples of ecological risk from nutrients and sediments for each basin

- Multiply the basin Load Index (see Step 2b above) by the Marine Zone Risk Score to generate a Risk Index for each basin.

Step 4c: Assess pesticide risk for each basin

- Using monitored pesticide concentration data for each basin, assess risk using (i) probablistic ecological risk assessment (likelihood), and (ii) ms-PAF method (consequence).

4. Defining and mapping the ecosystems at risk of exposure to anthropogenic river-derived pollutants

4.1 Defining Marine Zones

The marine natural resource management (NRM) regions (as defined by the Great Barrier Reef Marine Park Authority; see Figure 4) extend seawards from the northern and southern boundaries of each of the six natural resource management regions to the outer edge of the Great Barrier Reef Marine Park. However, these are administrative boundaries and do not necessarily reflect the extent of influence of the catchments on the marine environment. Furthermore, rivers outside of a natural resource management region may influence the marine ecosystems within a region; for example, the northern Wet Tropics rivers influence the southern Cape York NRM region, the Burdekin River can influence the Wet Tropics NRM region, the Fitzroy River can influence the Mackay Whitsunday NRM region and the Burnett and Mary rivers can influence the Fitzroy NRM region in large-scale events (Lønborg et al., 2016). Accordingly, new Marine Zones have been defined for this assessment, which are intended to group waters in the Great Barrier Reef that regularly receive input from a group of rivers and are typically geographically constrained by coastal and marine features.

The Marine Zones used in this assessment (Figure 3) are based on a combination of (i) the long-term (2003-2014) primary and secondary wet season water type frequency map (see Devlin et al., 2015a) used to define the outer boundaries, (ii) the latest tracer modelling from eReefs (Baird et al., 2016) to define the northern and southern boundaries, qualified by, (iii) the existing natural resource management marine regions and Water Quality Improvement Plan assessment boundaries, and (iv) observations of plume extent from satellite imagery. The rivers that provide the primary influence in each Marine Zone are shown in Table 1.

Detailed methods, further justification and limitations of the extent of the Marine Zones are described in Appendix 1.

Table 1. Description of the Marine Zones assessed in this chapter and the primary basins of influence for each zone.

Marine zone	Primary basins of influence (refer also to Table 4)
Cape York North	Jacky Jacky, Olive, Pascoe, Lockhart
Cape York Central	Stewart, Hann, Normanby
Cape York South	Jeannie, Endeavour with limited influence from Daintree, Mossman, Russell-Mulgrave, Johnstone
Wet Tropics	Daintree, Mossman, Russell-Mulgrave, Johnstone, Tully, Murray, Herbert, Burdekin (limited)
Burdekin	Tully, Murray, Herbert, Black, Ross, Haughton, Burdekin, Don
Mackay Whitsunday	Proserpine, O’Connell, Pioneer, Plane
Fitzroy	Proserpine, O’Connell, Pioneer, Plane, Styx, Shoalwater Creek, Waterpark Creek, Fitzroy, Calliope, Boyne, Burnett (limited)
Burnett Mary	Waterpark Creek, Fitzroy, Calliope, Boyne, Baffle, Kolan, Burnett, Burrum, Mary

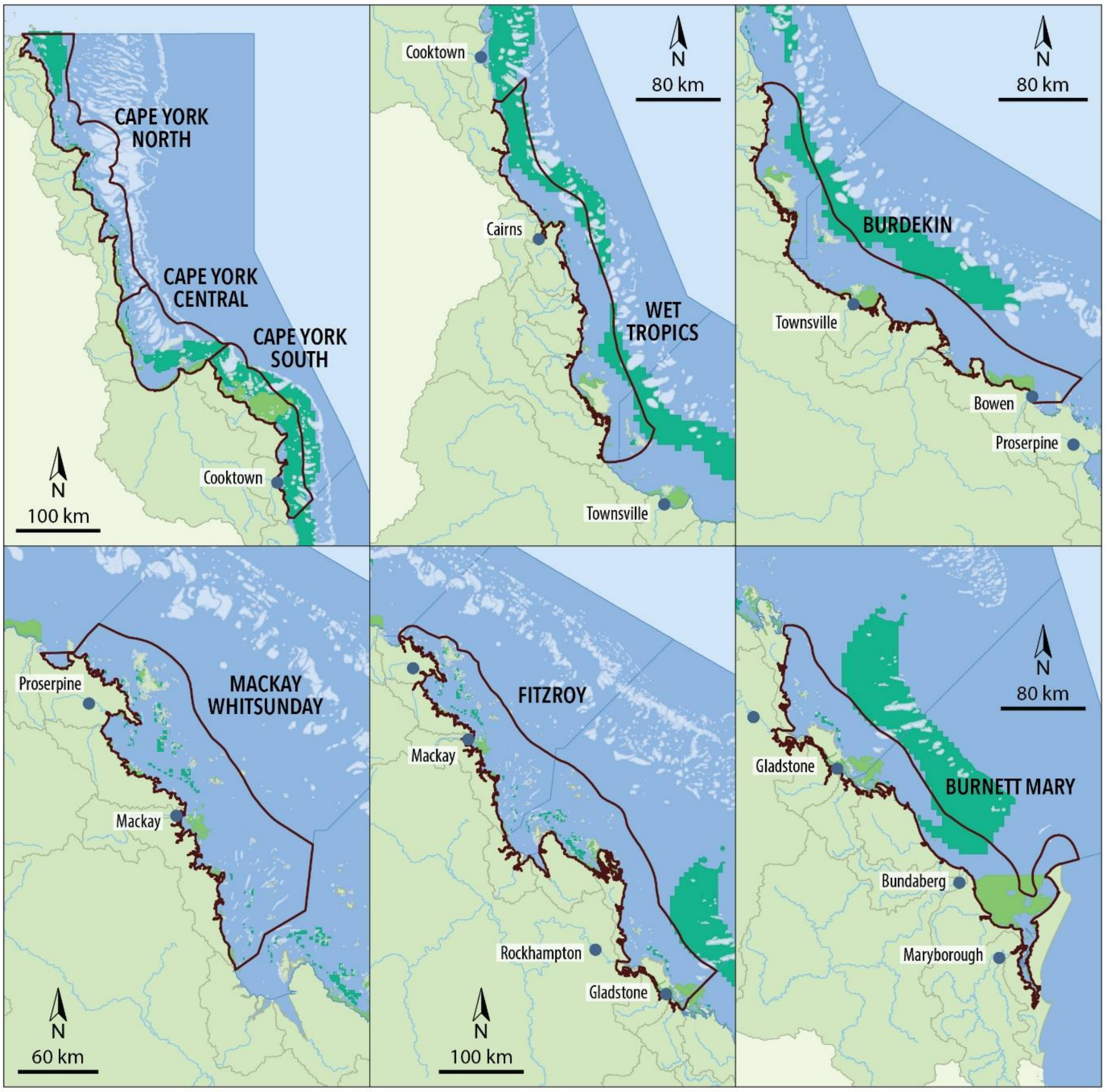


Figure 3. Panel showing the Marine Zones defined for the assessment, using the eReefs model tracer data (2011-2014), frequency of primary and secondary wet season water types (2003-2016), satellite imagery and expert knowledge of the influence of river plumes in the Great Barrier Reef.

4.2 Habitats

4.2.1 Marine ecosystems

The marine habitats considered in the marine ecosystems assessment are coral reefs and seagrass meadows, based on the best available information. There is insufficient data to inform the assessment of pollutant exposure to other ecosystems such as mangroves, soft bottom communities and fish; however, these systems are still recognised as important to the health of the Great Barrier Reef and should be included in future assessments as more information becomes available.

For coral reefs, the spatial layer used is the Great Barrier Reef Marine Park Authority Spatial Data Centre's coral reefs spatial data file (accessed September 2016).

The seagrass habitat map used is a combination of the following datasets:

1. collation of seagrass assessments undertaken in the Great Barrier Reef World Heritage Area from 1984 to 2014 (Carter et al., 2016)
2. seagrass mapping of 100 km of coastal seagrass meadows from Walsh Bay to Cape Flattery and four reef-top meadows, plus additional surveys near the Starcke River mouth (see Waterhouse et al., 2016a)
3. Hervey Bay seagrass mapping, which recognises the area of influence of the rivers in the Burnett Mary NRM region, particularly the Mary River (McKenzie et al., 2014).

Deepwater seagrass is also represented using a statistical model of seagrass present in Great Barrier Reef World Heritage Area waters >15 m depth. In this model, spatial distribution is a statistically modelled probability of seagrass presence (using generalised additive models based on binomial error and smoothed terms in relative distance across and along the Great Barrier Reef), based on field validation points (Coles et al., 2009). Locations with seagrass habitat probability >0.5 (50%) were included in the assessment.

Both datasets should only be presented as *potential* seagrass habitat.

4.2.2 Coastal aquatic ecosystems—floodplain wetlands and floodplains

The coastal aquatic ecosystems considered in this assessment are floodplain wetlands and floodplains. The spatial layers used are the Queensland Wetland Data Version 4.0—Wetland areas 2013 EHP and Queensland Floodplain Assessment Overlay 2013 NRM (see Figure 4).

4.2.3 Basins and catchments

The marine assessment links to the 35 main Great Barrier Reef river basins, and the coastal aquatic ecosystem assessment links to the 47 management units (described in the Introduction to theScientific Consensus Statement) within the Great Barrier Reef catchment (Figure 4). The management units were defined for the Water Quality Improvement Plans (NQ Dry Tropics, 2016; Fitzroy Basin Association, 2015) to recognise the relative contributions of catchments within the large Burdekin and Fitzroy Basins.

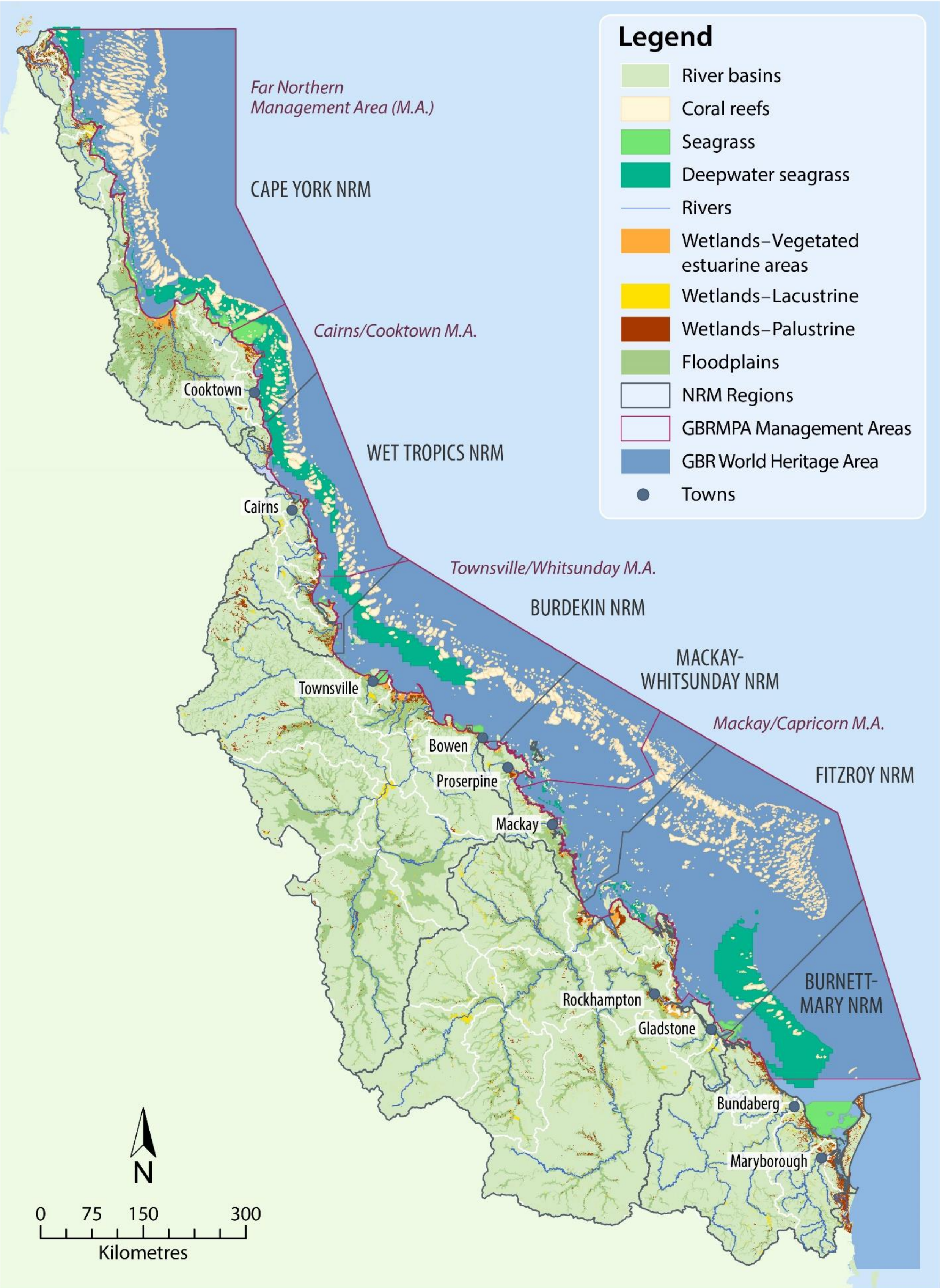


Figure 4. Map of the marine natural resource management boundaries, management units and coastal aquatic and marine habitats included in this chapter.

5. Drawing on Chapters 1 and 2: what are the water quality hazards that pose a potential risk to Great Barrier Reef aquatic coastal and marine ecosystems?

Chapter 1 identifies the pressures and threats to Great Barrier Reef coastal and marine ecosystems, highlighting degraded water quality from land-based pollutants as one of the primary threats. Chapter 2 describes the sources of land-based pollutants and discusses spatial and temporal differences in the Great Barrier Reef catchments. This section interprets that information in the context of the risk assessment to support the analysis of likelihood of exposure of pollutants to coastal aquatic ecosystems specifically (i) floodplains, and (ii) floodplain wetlands (i.e. lakes and vegetated swamps).

The drivers and sources of pressures on floodplain wetlands within the Great Barrier Reef catchments have been broadly described (Cape York NRM and South Cape York Catchments, 2016; DSITI, 2015; Folkers et al., 2014; NQ Dry Tropics, 2016; Terrain NRM, 2015). However, there are presently few data available on the generation, pathways and fate of potential pollutants in relation to floodplain wetlands whether from sub-catchment, basin or Great Barrier Reef–wide perspectives.

In one of the few Great Barrier Reef catchment-wide studies available, the Department of Science, Information Technology and Innovation (DSITI, 2015) mapped the level of hazard to both floodplain and non-floodplain wetlands from individual pressures associated with land-use drivers. Hazard from potential pressures to wetlands, including individual pollutants, was calculated by generating a land-use pressure profile, where each land use was assigned a numerical score based on the potential for that land use to drive a particular pressure on wetlands, including individual pollutants. The percentage area of each land use within a sub-catchment unit was then multiplied by the weighting derived from the land-use pressure score. Hazard was evaluated for each individual pressure, and the results for multiple pressures were summed to produce a single pressure hazard score for each sub-catchment unit. Mapping units were then attributed with separate hazard scores using a unique scoring range for each pressure (individual or combined) and displayed as hazard categories using an equally distributed five-point scale. The land uses most strongly associated with individual pressures and driving multiple pressures to wetlands were urban areas, irrigated cropping and horticulture, extensive grazing, intensively managed grazing and mining (DSITI, 2015). Land uses as drivers of other pressures such as groundwater abstraction and pesticide input were concentrated in specific locations.

In this risk assessment, we used the data from the Department of Science, Information Technology and Innovation (DSITI, 2015) for the High and/or Very High hazard categories to represent the sources of relative risk from elevated sediments, nutrients and pesticides to floodplain wetlands and floodplains. The extent of these highest hazard areas containing floodplain wetlands is discussed below. Potential pollutant exposure of these wetlands is discussed in Part B of this chapter.

5.1 Area of hazard from nutrient pressures

Nutrient pressures in the Great Barrier Reef catchment are primarily associated with intensive grazing and cropping including sugarcane and horticulture (see Chapter 2; Waters et al., 2014). Extensive grazing is also an important driver of particulate nutrient pressures (Waters et al., 2014; DSITI, 2015).

The area of the Great Barrier Reef catchment that is subject to High and Very High hazard from nutrient pressures is estimated to be around 5,561,000 ha. The areas of Very High hazard from nutrient pressures are typically concentrated within low-lying coastal areas including the Burdekin, Burrum and Plane Basins (Figure 5.). The areas of High hazard from nutrient pressures are widespread within the Fitzroy Basin and scattered across the Burnett and Burdekin basins, but occur in all basins.

Figure 6 shows the area of land lying within Very High or High nutrient hazard categories and the proportion of that area in which floodplain wetlands are found. Appendix 2 contains detailed results of the land use hazard analysis (DSITI, 2015).

There are 11 management units where 15% or more of the unit has Very High or High nutrient hazard areas containing floodplain wetlands. At least 30% of the Dawson, Lower Burdekin and Comet management units are subject to Very High or High hazard areas containing floodplain wetlands.

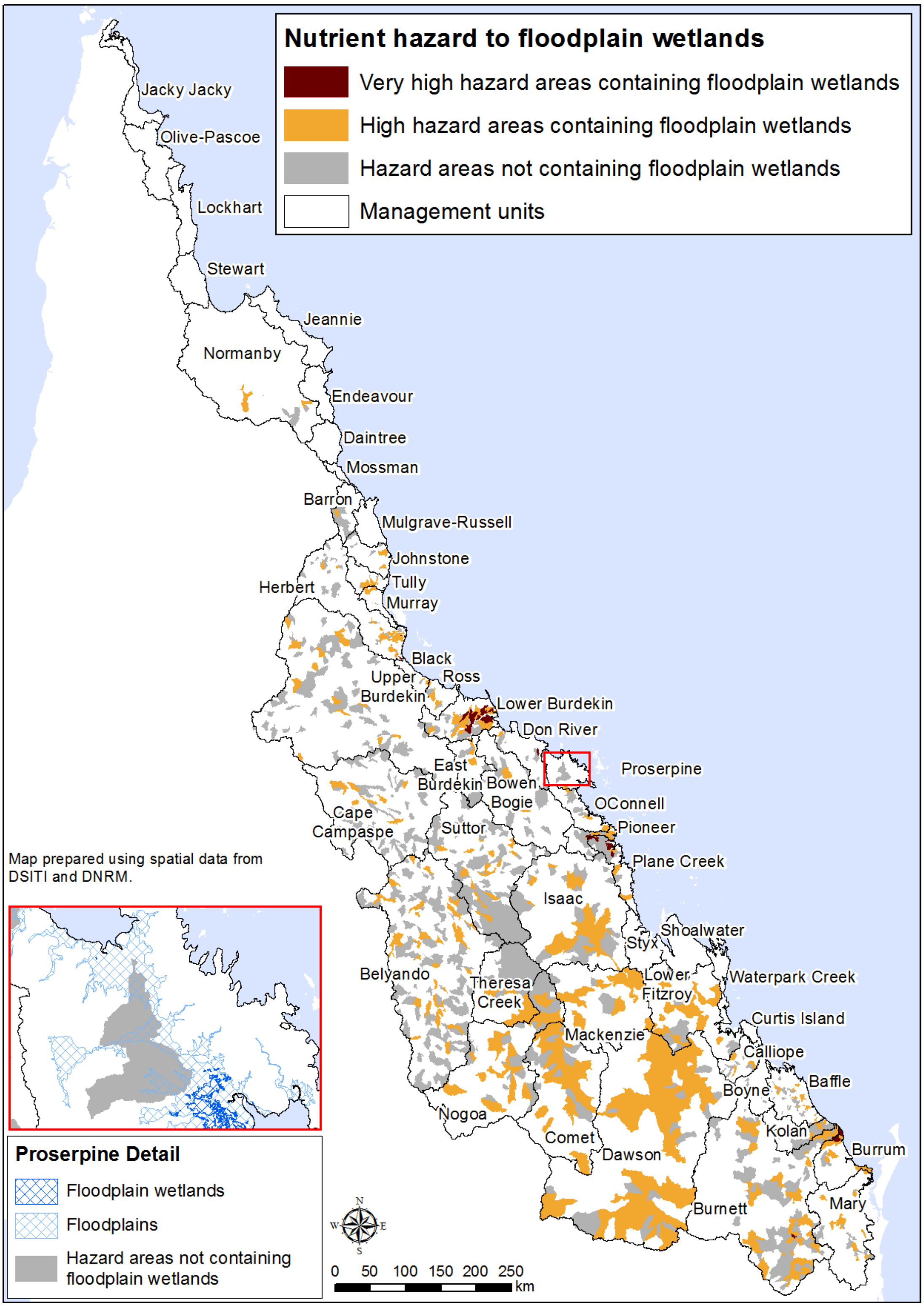


Figure 5. Areas categorised as Very High and High hazard from nutrient pressures (outputs derived from DSITI [2015] land-use hazard analysis). Inset shows example of detail at smaller scale showing the occurrence of High hazard area for nutrients that does not contain wetlands but where wetlands lie downstream. As this hazard area does not contain wetlands it is not included in this assessment.

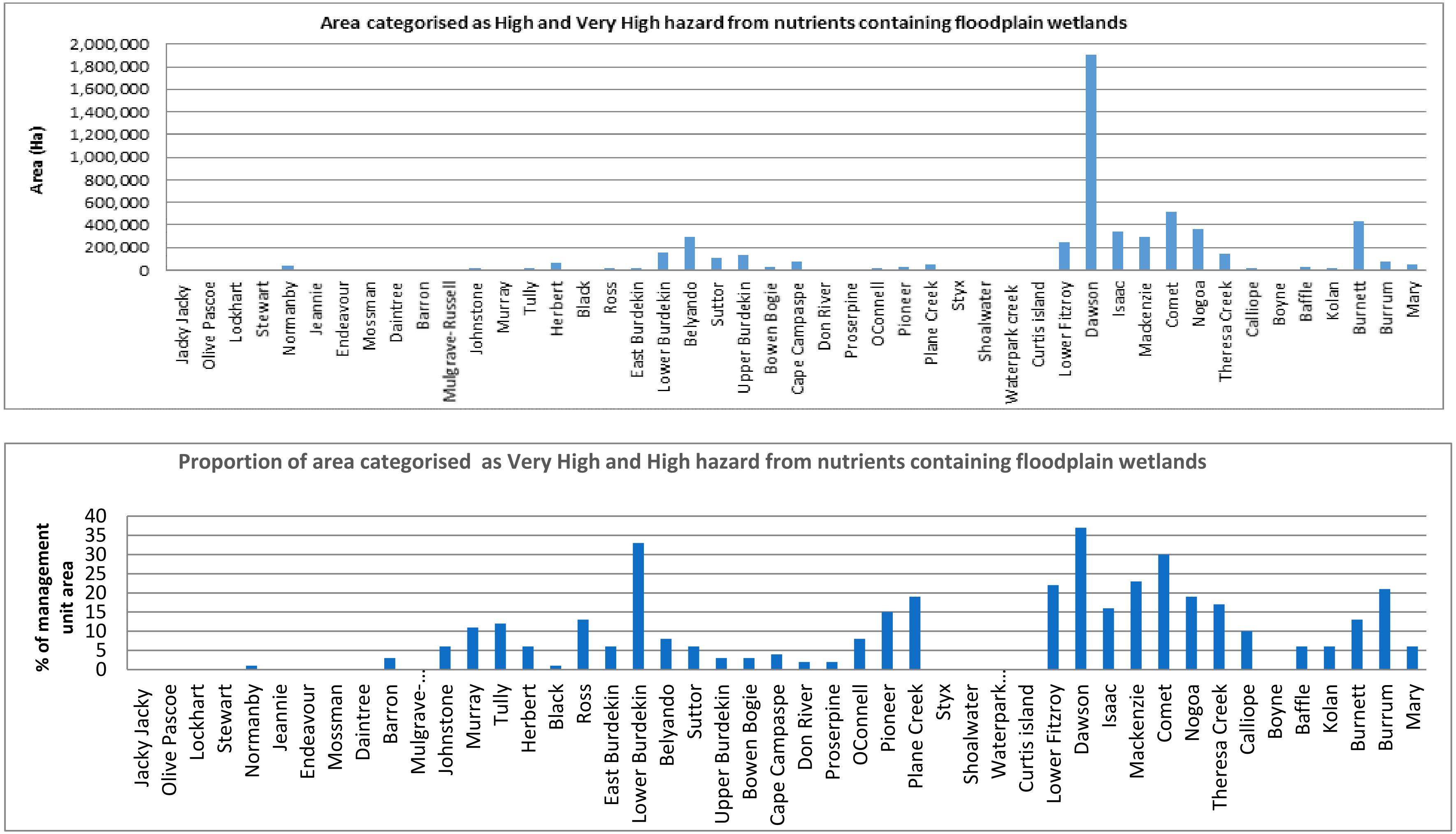


Figure 6. (Top) Area of each management unit categorised as Very High and High hazard from nutrients and also containing floodplain wetlands, and (bottom) the percentage of the Very High and High hazard area within the management unit that also contains floodplain wetlands (outputs derived from DSITI [2015] land-use hazard analysis).

5.2 Area of hazard from sediment pressures

Elevated levels of sediment influencing water quality are typically attributed to grazing land use and, to a lesser extent, cultivated cropping areas, mining and developing urban areas (DSITI, 2015; Fitzroy Basin Association, 2015; NQ Dry Tropics, 2016; Terrain NRM, 2015). Sediment hazard is pervasive within the Great Barrier Reef catchments; therefore, the focus of this section is on Very High hazard areas only, shown in Figure 7. The Great Barrier Reef catchment area categorised as Very High hazard from sediment pressures and which contains wetlands is estimated to be around 3,921,000 ha. This is around 59% of the total area categorised as Very High hazard from sediment pressures (see Appendix 2 for further detail).

The main areas of hazard from sediment pressures are in the upper parts of the Fitzroy and Burnett basins. Smaller areas of hazard from sediment pressures influencing floodplains and floodplain wetlands are also concentrated in the coastal plains of the Lower Burdekin, Herbert and Plane management units. These areas of hazard, which are based on the Department of Science, Information Technology and Innovation land-use pressure profiles, largely align with the highest areas of sediment delivery predicted by the Source Catchments modelling data (see Chapter 2).

As shown in Figure 8, a number of management units have at least 20% of the unit categorised as Very High hazard from sediment pressures in which floodplain wetlands area found. This includes the Dawson (35%), Comet (26%), Lower Burdekin (27%) and Mackenzie (21%) management units. Floodplain wetlands are also located in smaller areas categorised as Very High hazard for sediment pressures in the Isaac, Nogoa, Theresa Creek and Burnett management units.

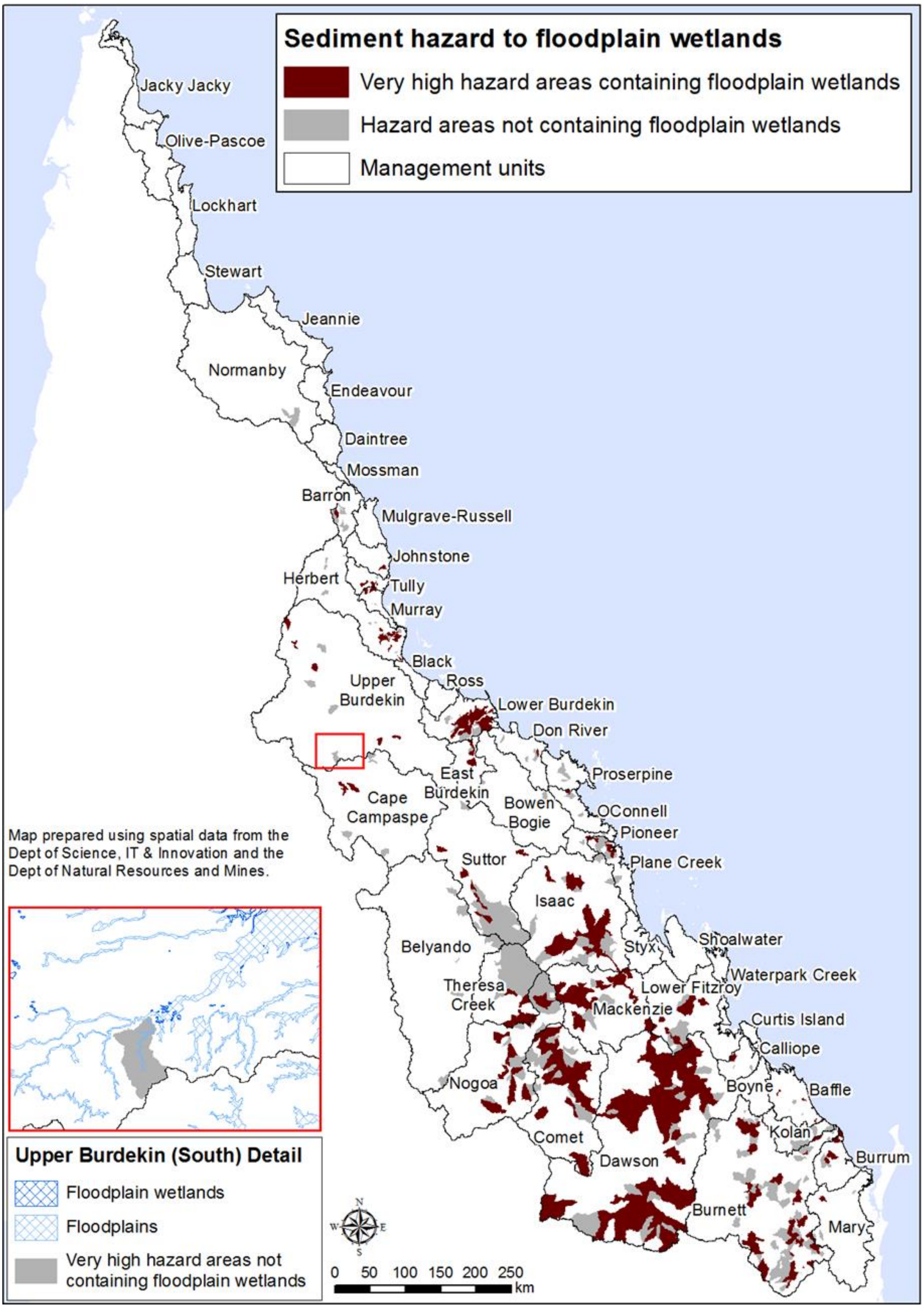
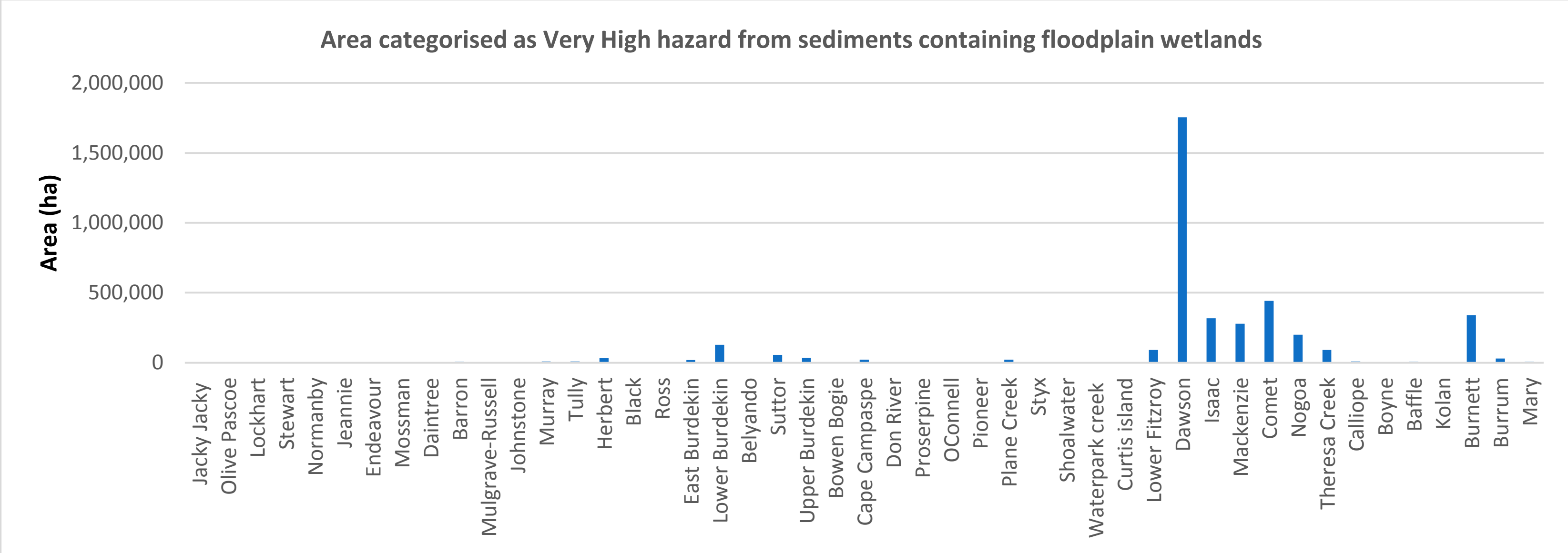


Figure 7. Areas categorised as Very High hazard from sediment pressures (outputs derived from DSITI [2015] land-use hazard analysis). Inset shows example of detail at smaller scale showing the occurrence of Very High hazard area that does not contain wetlands but where wetlands lie downstream. As this hazard area does not contain wetlands it is not included in this assessment.



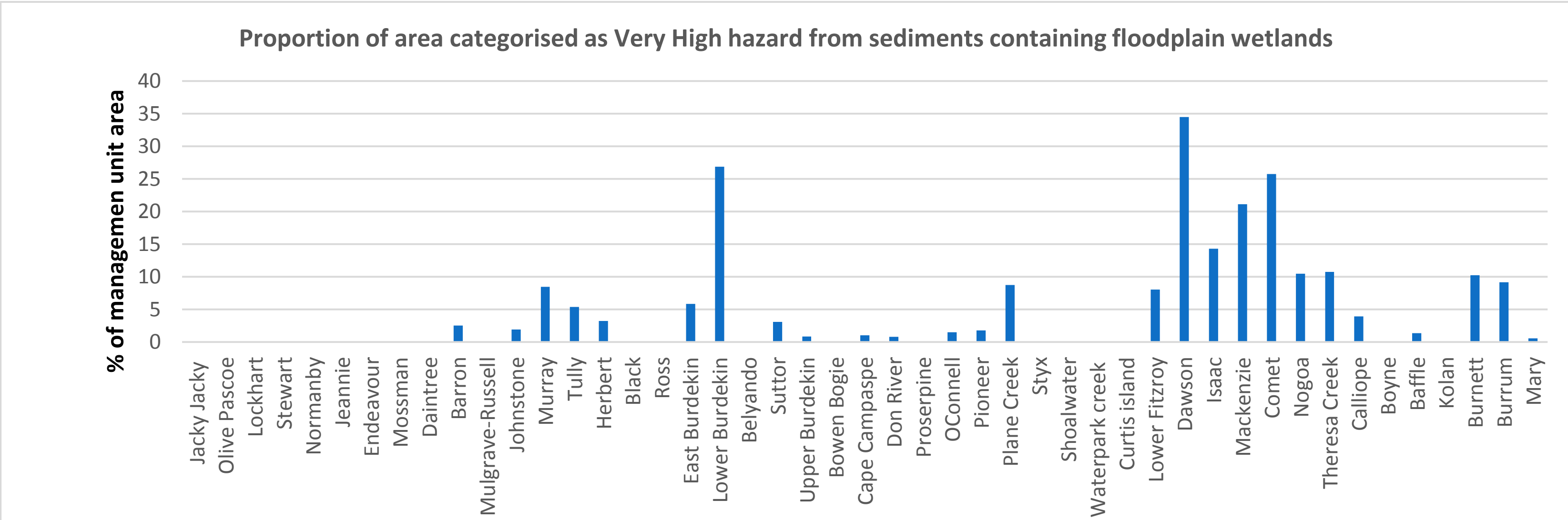


Figure 8. (Top) Area of each management unit categorised as Very High hazard from sediment pressures and which also contains floodplain wetlands, and (bottom) the percentage of the Very High sediment pressure area which contains floodplain wetlands (outputs derived from DSITI [2015] land-use hazard analysis).

5.3 Area of hazard from pesticide pressures

Regional Water Quality Improvement Plans attribute the input of regional pesticides (ametryn, atrazine, diuron, hexazinone) into wetlands primarily to sugarcane land use (NQ Dry Tropics, 2016; Folkers et al., 2014; Terrain NRM, 2015). A recent study of a limited set of wetlands in sugarcane areas in the Lower Burdekin, Pioneer and Burnett floodplain areas detected 19 different pesticides (or metabolites), with 14 pesticides detected at each of the two sites where samples were taken (Devlin et al., 2015b). Other land-use sources of pesticides include dryland cropping, horticulture and intensive animal production (e.g. dairy, poultry) (Terrain NRM, 2015; DSITI, 2015; Fitzroy Basin Association, 2015).

An estimated 287,000 ha is categorised as Very High or High hazard from pesticide pressures containing floodplain wetlands. These areas are mainly concentrated on the coastal floodplains where sugarcane cropping is the main land use (Figure 9) and, to a lesser extent, in dryland cropping areas in the Comet and Dawson management units. For example, the Lower Burdekin has around 78,000 ha categorised as being High and Very High hazard from pesticides and containing floodplain wetlands. Other significant areas categorised as subject to High and Very High hazard from pesticide pressures and which contain floodplain wetlands include Comet (~49,000 ha), Herbert (~38,000 ha), Plane (~22,000 ha), Pioneer (~21,000 ha), Dawson and Burnett (both ~17,000 ha). As shown in Figure 10, 16% of the Lower Burdekin and 13% of the Pioneer basins have pesticide hazard areas with floodplain wetlands in them; all other areas are less than 10% but, as highlighted in Chapter 1, some of these areas are relatively large given the toxicity of pesticides.

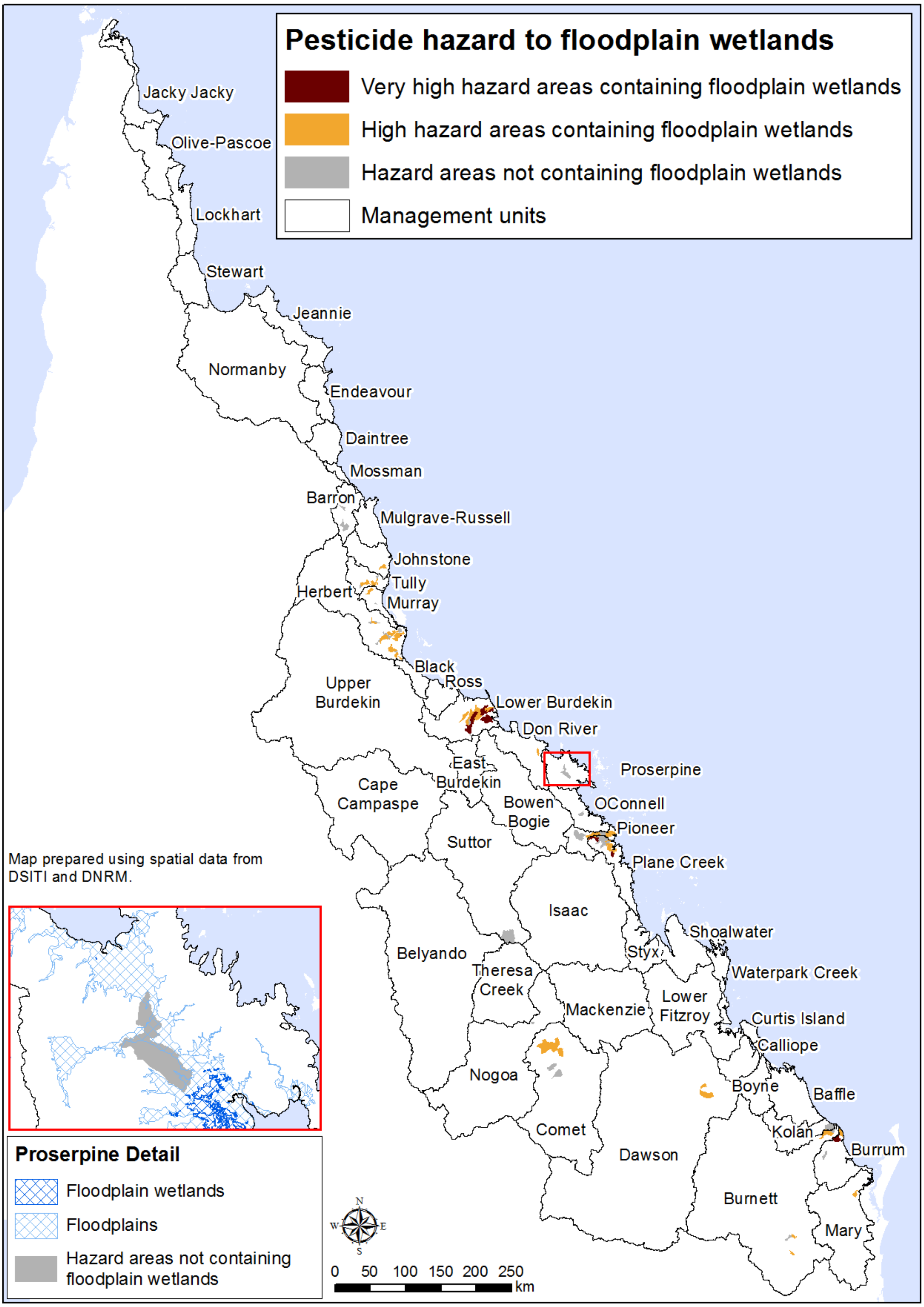


Figure 9. Areas categorised as High and Very High hazard from pesticide pressures (outputs derived from DSITI [2015] land-use hazard analysis). Inset shows example of detail at smaller scale showing the occurrence of High and Very High hazard for pesticide pressure that does not contain wetlands but where wetlands lie downstream. As this hazard area does not contain wetlands, it is not included in this assessment.

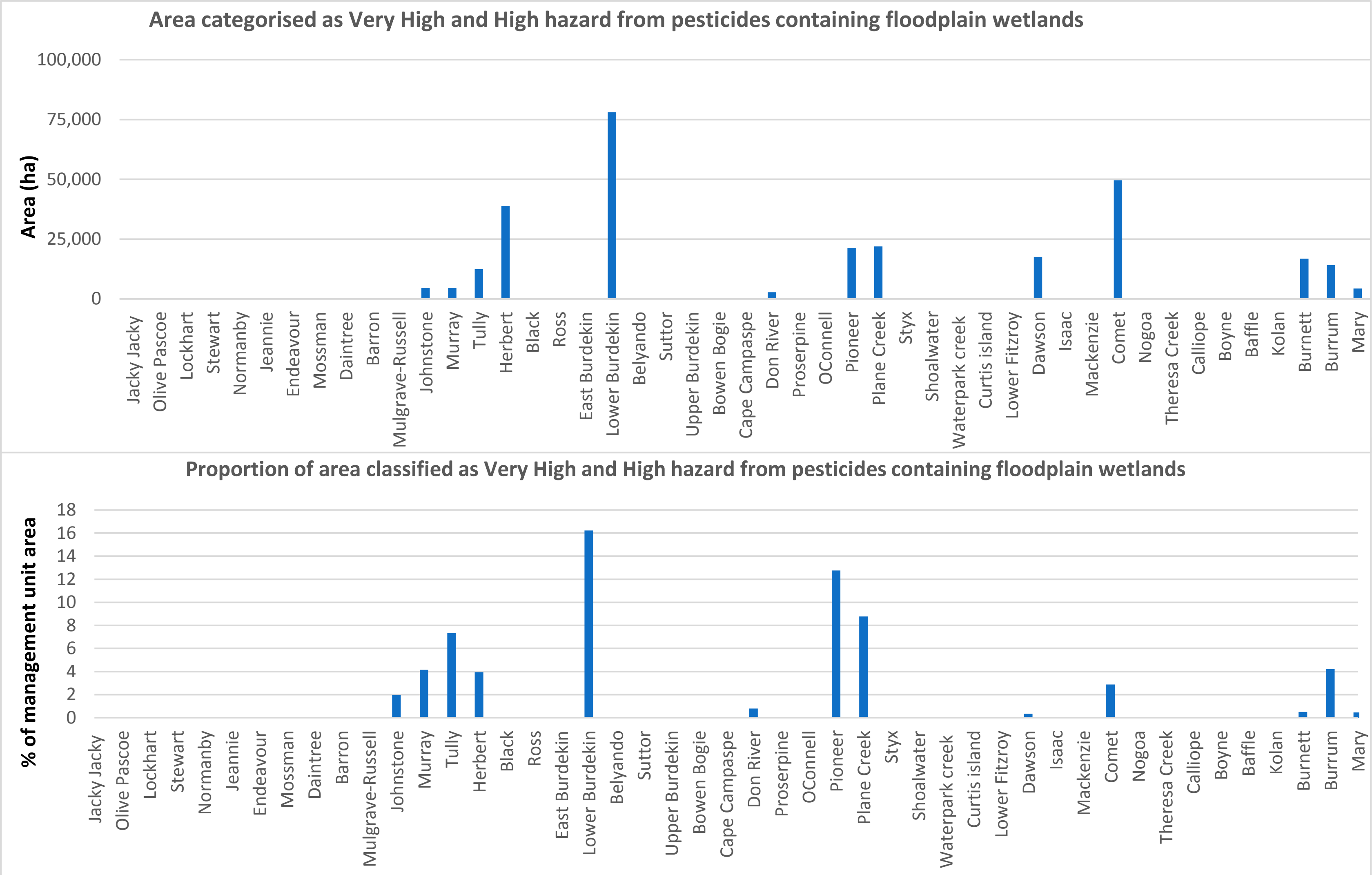


Figure 10. (Top) Area of each management unit categorised as Very High and High hazard from pesticide pressures and which also contains floodplain wetlands, and (bottom) the percentage of the Very High and High pesticide hazard areas which also contain floodplain wetlands (outputs derived from DSITI [2015] land-use hazard analysis).

5.4 Changes to hazards and pressures on floodplain wetlands and floodplains due to land-use intensification

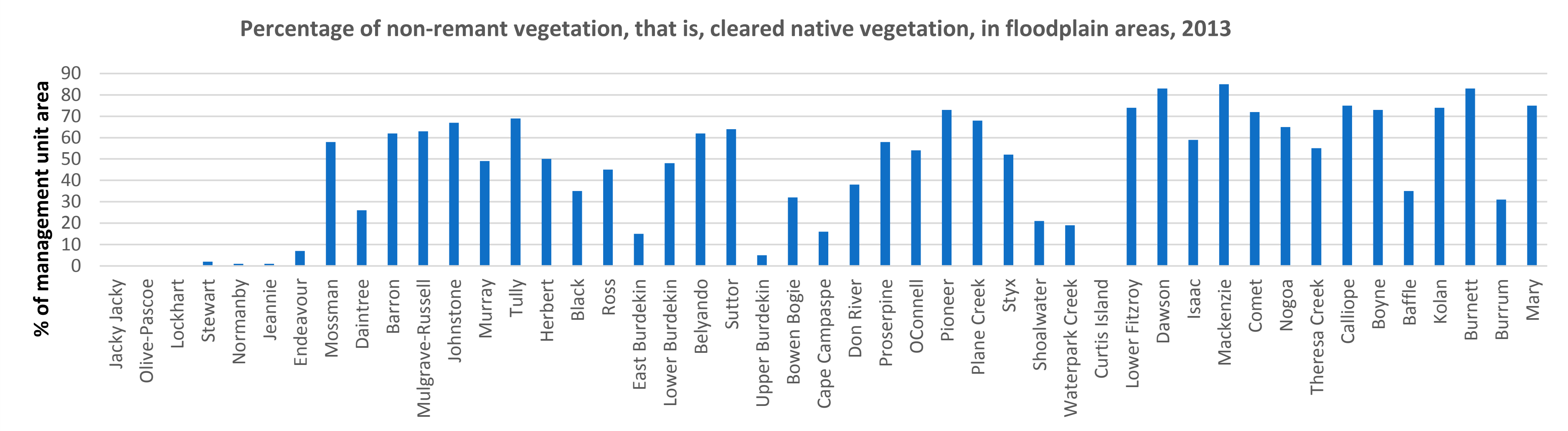
Land-use change and intensification are highly important in driving changes in land-use hazard and pressures on wetland ecosystems (Davis et al., 2017; DSITI, 2015; Boulton, 2014) and as described in chapters 1 (Schaffelke et al., 2017) and 2 (Bartley et al., 2017). Palaeolimnological records also suggest the degree of change and total impact on wetlands from resource development and land-use intensification in Australia has been underestimated (Gel et al., 2013).

In 2013, 77% of the total pre-European extent of freshwater wetlands (lakes and vegetated swamps) remained across the Great Barrier Reef catchments (Australian and Queensland governments, 2015). In Cape York and inland regions where there are extensive wetland areas, there has been little change in extent since European settlement. South of Cape York, wetland loss is largely concentrated in nearshore coastal lowland floodplain areas. For these areas this represents a significant loss in connected river, wetland and floodplain water quality improvement function. The wetland losses are mainly due to drainage, clearing and levelling associated with intensive agriculture or urban use (Australian and Queensland governments, 2015). Loss of wetlands compared to pre-development extent has been high in the Wet Tropics (48%), Mackay Whitsunday (42%) and Burnett Mary (37%) regions. Many smaller coastal catchments have undergone widespread loss of freshwater wetlands with, for example, more than 80% of wetland extent lost from the Kolan, Pioneer and Calliope basins. These losses are mainly of floodplain wetlands.

Remnant regional ecosystems in the Great Barrier Reef comprise woody (e.g. shrubland, woodland, rainforest) and non-woody (e.g. grassland, sedgeland, herbland) native vegetation. There is ongoing clearing of woody vegetation for the introduction of more intensive land uses including cropping, infrastructure, mining pasture, settlements and plantations (DSITI, 2016). The consequence of land-use change is the amplification of water quality and other pressures on aquatic ecosystems (DSITI, 2015). The Great Barrier Reef catchments had a total woody vegetation clearing rate of 108,000 ha/year during the 2014-2015 period. This was similar to the 2013-2014 period (105,000 ha/year) but 46% higher than the rate in 2011-2012 (DSITI, 2016). An analysis of the Statewide Landcover and Trees Study data (DSITI, 2016), which uses remote sensing techniques, shows floodplain wetlands are subject to ongoing clearing of remnant woody vegetation. Detailed figures are provided in Appendix 2. Conversion of non-woody remnant vegetation such as grassland, sedgelands and herblands cannot be reliably monitored using remote sensing techniques. Therefore, the contemporary rate of loss of these remnant floodplain regional ecosystems is uncertain.

There are few floodplain areas south of Cape York where there has not been significant past vegetation clearing, except in the Upper Burdekin, East Burdekin, Shoalwater and Waterpark Creek. In some management units, up to 85% of native woody vegetation in floodplain areas has been cleared (Figure 11, top). By 2013, only 15% of native woody vegetation extent remained in floodplains in the Mackenzie and Dawson catchments, and 17% remained in the Burnett Basin.

The relative rate of contemporary woody native vegetation clearing in floodplain areas is disproportionate to non-floodplain areas in some management units. For example, during the 2014-2015 reporting period, approximately 18,300 ha of remnant woody vegetation was cleared from floodplain areas. During this period, around 1% or more of remnant woody vegetation in floodplain areas was cleared in the Belyando (4526 ha), Suttor (1700 ha), Styx (1023 ha), Dawson (1023 ha), Isaac (2239 ha), Mackenzie (1722 ha), Baffle (863 ha), Kolan (771 ha), Burnett (447 ha) and Mary (368 ha, Figure 11). The Kolan stands out, where over 5.5% of remnant woody native floodplain vegetation was cleared during the 2014-2015 period. The detailed figures are provided in Appendix 2. This loss of woody native vegetation cover is indicative of land-use change and/or intensification which can, in turn, increase pressures on wetland ecosystems and water quality.



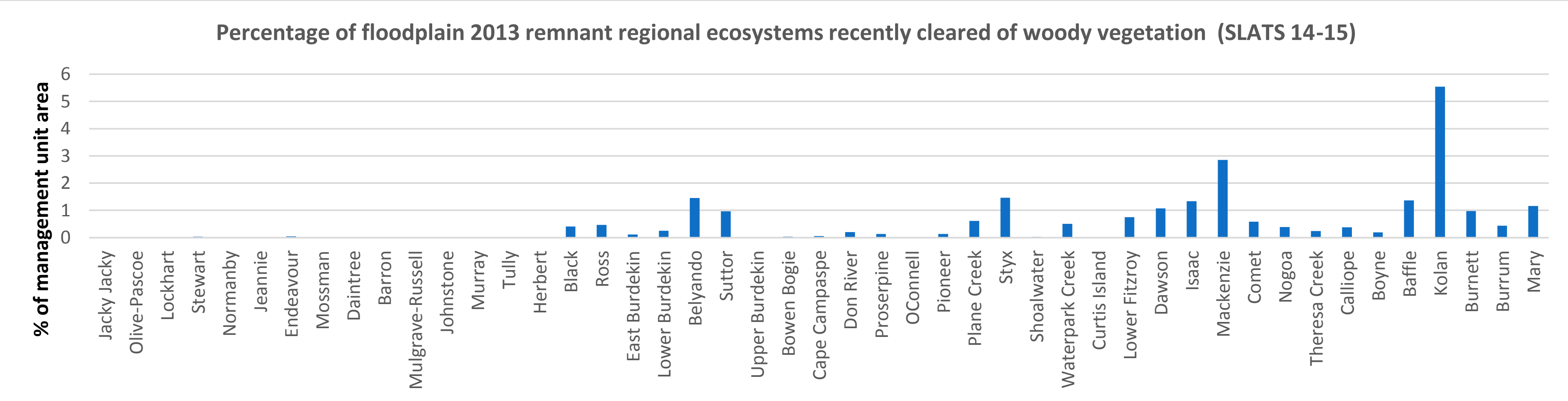


Figure 11. (Top) The percentage of remnant regional ecosystems cleared (i.e. the area cleared of native vegetation), and (bottom) percentage of remnant regional ecosystem in each management unit cleared of native woody vegetation in floodplain areas in the Great Barrier Reef catchment as at 2013. Source: DSITI (2016).

Part B: Likelihood and consequences of pollutant exposure

6. What is the likelihood and timing of exposure of key pollutants to Great Barrier Reef coastal and marine systems?

6.1 Supporting evidence

An assessment of the *likelihood of exposure* and *when the greatest exposure of wetlands, coral reefs, and seagrass to degraded water quality occurs* depends on the parameter of interest, seasonal climate and weather, geochemical and water residence times, marine physical processes, biogeochemical transformation processes, time-dependent ecological processes and interactions with other stressors.

The distinct wet and dry seasonal climate of the Great Barrier Reef results in most sediment, nutrients and pesticides being delivered to the Great Barrier Reef lagoon during the summer wet season (December–April) when high river discharge occurs. On average, 70 km^3 of fresh water is discharged each year by rivers and streams into the Great Barrier Reef lagoon (Lønborg et al., 2016). River flow is delivered in discrete flood events during the five-month summer wet season, forming distinctive river plumes in the coastal zone that can move north along the coast but can occasionally move out towards the mid- and outer shelf area. The episodic nature of the dry tropics rivers provides an opportunity for some recovery of marine ecosystems in the periods between flows, for example, seagrass recovery in Cleveland Bay given four years of limited sediment delivery from the Burdekin River (McKenzie et al., 2016). The quantity of material delivered is generally proportional to the run-off volume over any particular time interval, as this reflects the degree of erosion and transport of materials from catchments. However, the relationship is not a straightforward one as export in run-off depends upon a variety of factors, including quantity of material available in catchments, its geochemical mobility and dilution in the run-off. In general, early wet season flood events and the leading edge of most floods contain the highest concentrations of suspended sediment, nutrients, and pesticides (e.g. Furnas, 2003; Furnas et al., 2011; Devlin et al., 2012a; Devlin et al., 2015a).

It has also been shown that some floodplains can accumulate large amounts of material including sediment and nutrients that would otherwise be exported downstream (Noe and Hupp, 2009), and these can be remobilised in large events depending on the period of time between events (Croke et al., 2015). Episodic wetting and drying periods can also influence mineralisation and remobilisation of pollutants, especially sediments (e.g. Abera et al., 2012). However, as there has been little consideration of floodplains within the Great Barrier Reef catchments there is a significant gap in understanding of current and historic floodplain dynamics influencing exposure to pollutants, temporal and spatial accumulation of material and changes resulting from floodplain development. Many of the rivers, including the Fitzroy and Pioneer, have been 'river trained' through flood-mitigation structures (e.g. levees), and the floodplain is essentially isolated from the river (Webster and Mullins, 2003).

The likely exposure of specific wetlands to impacts is influenced by a range of factors such as natural drivers, position in the landscape, on-ground land-use management practices and the ecological resilience (DSITI, 2015). In a broadscale assessment, it is difficult to capture the location and proximity of individual pressures to individual wetlands and their likelihood of causing an impact on a wetland (DERM, 2011). Nevertheless, the Great Barrier Reef Wetland Monitoring pilot project showed that the state of the wetland environmental values co-varied with the surrounding land use in the local scale sub-catchment surrounding the wetland (Tilden et al., 2015). Wetlands in primary source areas have a greater exposure to pollutants. Thus, in this assessment, the presence of a wetland within a source area is equated to the likelihood of exposure. The greater the area of wetland within a source area

the greater the likelihood of exposure of an individual wetland and the greater the likelihood of detrimental impacts and adverse ecosystem change.

Further knowledge is available on the timing, movement and transformation of pollutants in the Great Barrier Reef lagoon and wetlands, described below and summarised in Table 2 that is relevant to the selection of the input data and interpretation of the results for assessing the *likelihood of exposure of ecosystems to pollutants*.

6.1.1 Nutrients

Almost all discharge of nutrients from rivers to the Great Barrier Reef occurs in the wet season (December–April inclusive) (Devlin and Brodie, 2005; Furnas, 2003; Furnas et al., 2011). Oceanic nutrient upwelling is also an important source of nutrients in the wet season in offshore areas and a potentially dominant source of new nutrients to some areas (Furnas et al., 2011), for example, the far northern and southern offshore Great Barrier Reef where few river-derived nutrients are found. As the risk assessment in this chapter is largely concerned with anthropogenic river-derived nutrients, these influences are not considered further here.

The negative impacts of river-derived nutrient inputs occur almost completely during the wet season due to high levels of exposure in these months. Hence, minor discharge of nutrients in the dry season (comprising approximately 10% of the annual inputs; Furnas et al., 2011), whether from river base flow or groundwater discharge, results in low exposure and is considered to be less important for Great Barrier Reef ecosystems.

Nutrients (both nitrogen and phosphorus) are discharged from rivers as a mixture of particulate and dissolved organic and inorganic forms. Dissolved nutrients are transported widely across and along the shelf, and those in dissolved inorganic form are taken up by phytoplankton (algae; Devlin et al. 2015a) and bacteria (Angly et al., 2016) as biological activity increases as the plume is transported further from the river mouth. Increased nutrient uptake is reflected in increased chlorophyll concentration as an indicator of phytoplankton biomass. Particulate nutrients are largely deposited from the water column close to the river mouth (Devlin et al., 2012b; Brodie et al., 2015; Lewis et al., 2014). Following deposition, the nutrients may be mineralised (through bacterial action) to dissolved inorganic nutrients and/or, for nitrogen, gaseous forms such as nitrogen (N_2) and nitrous oxide (N_2O) (Brodie et al., 2015). Following mineralisation, the DIN and phosphorus formed may be transported further into the Great Barrier Reef lagoon. For nitrogen, the proportion of denitrification versus mineralisation is not well quantified (Brodie et al., 2015). Further transport of the mineralised nutrients may also occur outside of the wet season; however, these are not considered to contribute to the most important periods of exposure for Great Barrier Reef ecosystems.

From a longer term perspective, nutrients can be mobilised from the sediments for years after they are deposited. Compared to the water column, large pools of nitrogen and phosphorus are stored in Great Barrier Reef sediments and benthic biota, principally as organic detritus but also as bioavailable soluble forms in pore waters and bound to sediment particles (Brodie et al., 2012b; Walker, 1981).

Nutrient influences can be episodic and dependent on the coincidence of several factors. For example, the influence of nutrient enrichment on coral bleaching susceptibility is episodic at three- to five-year intervals during high-temperature periods, for example, in El Niño years in summer (December–March) (Wooldridge, 2016; Wooldridge et al., 2017). During periods of prolonged elevated sea surface temperature, coral reefs are more susceptible to coral bleaching if these conditions coincide with degraded water quality (e.g. Wooldridge, 2009). Peak annual temperatures typically occur in the month of February coincident with the period of river discharge of nutrients. However, other factors can also affect the susceptibility of coral reefs exposed to increased temperatures.

Measuring nutrient exposure in the Great Barrier Reef marine waters is complicated by the large degree of processing and transformation that occurs between the river mouth and the mid-shelf and

outer Great Barrier Reef. Pollutant load dispersion models can help to develop risk maps by defining areas that may experience acute or chronic high exposure to pollutants or stressors (Álvarez-Romero et al., 2013; Lønborg et al., 2016). Details of the pollutant movement and frequency of exposure are key parameters in attributing water quality decline to ecosystem change. These contribute to the 'likelihood' component of the risk equation. Remote sensing techniques for measuring chlorophyll have low reliability in turbid waters, leading to low confidence in the ability of the algorithms to estimate chlorophyll in large areas of the Great Barrier Reef (Waterhouse et al., 2015b). Thus, manual measures of chlorophyll are considered the most reliable source; however, very little manual measurement of chlorophyll *a* has been undertaken in recent years across the Great Barrier Reef.

For coastal aquatic ecosystems, the timing and exposure of systems to excess nutrients varies according to land use and location in the Great Barrier Reef catchments. Fertilised land uses, including irrigated cropping, are a high source of nitrate (Waters et al., 2014), which can enter wetlands throughout the irrigation season or during unseasonal rain events; the net effect is a balance between the rate of fertiliser input and the rate of utilisation by vegetation. In grazing areas, nutrients influence the growth of weeds and declining water quality during the dry season which is exacerbated by more frequent cattle usage (Pettit et al., 2012). The effects of nutrients may be exhibited mainly at the local scale. For example, waterholes/wetlands can maintain reasonably good water quality if they are protected from direct disturbances. However, a single localised event such as a visit by a herd of cattle or introduction of a weed species can cause catastrophic effects to water quality and biota (Loong et al., 2005).

6.1.2 Suspended sediment

As with nutrients, almost all discharge of sediments from rivers to the Great Barrier Reef occurs in the wet season (December–April inclusive) (Devlin and Brodie, 2005; Furnas, 2003; Furnas et al., 2011). Sediment that is transported more than 1 km offshore is generally the clay to fine silt (<16 µm) fraction (Bainbridge et al., 2012). Fine sediments have potential residence times of decades on the shallow inner shelf, during which they remain available for resuspension and can affect coastal turbidity (Brodie et al., 2012b; Fabricius et al., 2013a, 2014, 2016; Lewis et al., 2015). Fine sediment supply is attenuated during the year by the processes of transport into mangroves, compaction and transport offshore into deeper waters where resuspension is unlikely to occur (Brodie et al., 2012b).

There is clear evidence that there is year-round, wind-driven resuspension of fine terrigenous sediments on the shallow inner shelf, reducing light penetration for periods of days to weeks. The degree of resuspension appears to attenuate towards the second half of the dry season (Fabricius et al., 2013a; Fabricius et al., 2014; Fabricius et al., 2016) as fine material deposited after wet season floods is moved to deeper water or coastal depositional areas (Larcombe et al., 1995). However, there is a pool of refractory fine sediments that varies depending on hydrodynamic and climate influences (Bainbridge et al., 2016).

Only a small proportion of the sediment load delivered by rivers ultimately makes it to coral reefs on the mid- and outer shelf. Most fresh sediment is initially deposited in the estuaries (e.g. Fitzroy River; see Bostock et al., 2007) or close to the river mouth (see Orpin et al., 2004; Bartley et al., 2014; Lewis et al., 2014).

In coastal aquatic ecosystems, the timing and exposure to sediments and reduced wetland water quality also varies according to land use and location. The largest load of sediment transported into wetlands occurs during the peak flow of the flood season, where large quantities of sediments from hillslope, gully and streambank erosion are transported into the rivers and the connected wetlands (Bartley et al., 2010; Davis et al., 2015; Davis et al., 2017). In grazing areas, increased turbidity and declining water quality during the dry season can be exacerbated by more frequent cattle usage of wetlands (Pettit et al., 2012).

6.1.3 Pesticides

Pesticide residues (including pesticides, herbicides and fungicides) have been detected in sediments and waters of rivers, creeks, wetlands, estuaries and the inshore and offshore parts of the Great Barrier Reef lagoon (Devlin et al., 2015b). The types and concentrations of pesticides in the fresh, estuarine and marine ecosystems vary between catchments and regions, reflecting the predominant land use in the catchment (Waters et al., 2014). Other factors that also contribute to the likelihood of an ecosystem being exposed to pesticides include the pesticide application regime (including existing regulations) within those land-use areas, the physico-chemical properties of the pesticides applied (e.g. soil half-life and sorption potential), the timing between application and rainfall or irrigation and the distance of the ecosystem from the source of application (O'Brien et al., 2016a). The likelihood of exposure varies spatially and temporally, as well as in magnitude (i.e. the concentration of the pesticide); thus, exposure can be measured using these three factors.

As a rule, freshwater ecosystems are exposed to higher concentrations and a greater variety of pesticides than estuarine and marine ecosystems (Table 2). Likewise, pesticide concentrations in the marine environment decrease with increased distance from the mouths of rivers and creeks. As pesticides move through successive ecosystems along the transport pathway, a number of variables are believed to influence the observed changes in the presence of pesticides (and therefore exposure to these ecosystems), for example dilution and mixing, land use with and without pesticide application, bacterial degradation and compartmentalisation (discussed in more detail below). Studies have demonstrated that soluble PSII herbicides in flood plumes display conservative mixing behaviour along the salinity gradient, becoming increasingly diluted as the river waters progressively mix with seawater (Lewis et al., 2009). Outside of the area directly affected by flood plumes, chronic (year-round) low-level concentrations (well below concentrations known to cause adverse effects) have been observed (Kennedy et al., 2012a; Kennedy et al., 2012b) due to the long half-lives of many pesticides in marine waters (Mercurio et al., 2015).

On a temporal scale, environmental (e.g. rainfall) and human (e.g. timing, frequency and rate of changes in management practices) factors determine the inter-annual and inter- and intra-seasonal variability in the likelihood of pesticide exposure (Kennedy et al., 2012b; Gallen et al., 2015).

Pesticide exposure of riverine, estuarine and marine ecosystems generally occurs in the wet season during run-off events, generating a 'pulsed' type of exposure that varies in frequency and intensity depending on the catchment conditions. For example, smaller coastal catchments (with intensive cropping land use) have short and frequent run-off events and, therefore, show distinct pulses of elevated pesticide concentrations (e.g. Sandy Creek; Wallace et al., 2017). Conversely, larger catchments with higher discharge volumes and longer run-off events exhibit more low-level and more constant pesticide concentrations for longer periods (e.g. Fitzroy River; Smith et al., 2012). Both scenarios present a different type of risk: short-term exposures to high concentrations compared to long-term exposures to low concentrations. Exceptions to this are catchments that are fed with irrigation tail water, such as Barratta Creek, which may present relatively high concentrations (above guideline values) over extended periods. In addition, freshwater wetland hydrology can be influenced by irrigation and other flow modifications, which can determine the timing and nature of pesticide exposure in wetlands (Devlin et al., 2015b).

Critical periods of pesticide input to the Great Barrier Reef are associated with the first flush of terrestrial discharge after pesticide applications. These events often coincide with or closely follow end-of-dry-season pesticide application in key agricultural industries (Lewis et al., 2009; Davis et al., 2017; O'Brien et al., 2016a). The likelihood and magnitude of exposure to any particular pesticide is largely dependent on timing between application and first rainfall. After the first flush, concentrations usually dissipate over time with successive run-off events (e.g. O'Brien et al., 2016a).

The likelihood of exposure is also driven by the physico-chemical characteristics of the individual pesticides. PSII herbicides have moderate to high persistence in freshwaters (aqueous hydrolysis) of

over 50 days (Lewis et al., 2016). Pesticide half-lives in the marine environment are longer than previously thought for a number of pesticides, including the priority PSII herbicides (Navarro et al., 2004; Meakins et al., 1995; Thomas et al., 2002; Mercurio et al., 2015), glyphosate, 2,4-D and metolachlor. These findings are consistent with pesticides being detected at relatively stable, albeit relatively low concentrations in the offshore Great Barrier Reef lagoon and throughout the dry season (Shaw et al., 2010; Kennedy et al., 2012b).

Table 2. Qualitative assessment of the likelihood of exposure of DIN, suspended sediment and pesticides run-off in the Great Barrier Reef basins to receiving aquatic environments and the most relevant temporal periods.

Receiving environment	Nutrients – DIN[1] Likelihood of exposure *Key times*	Sediment[2] Likelihood of exposure *Key times*	Pesticides[3] Likelihood of exposure *Key times*
Freshwater reaches of rivers and freshwater /coastal wetlands	Very high to low (depending on location within the Great Barrier Reef, particularly relating to the area of intensive agriculture upstream). Limited direct evidence of effects on aquatic plants; however, strongly implicated in freshwater eutrophication and aquatic weed infestation causing hypoxic and connectivity barriers to migrating fish species between freshwater and marine environments. *Dry season and first flush (e.g. hypoxic events)*	Uncertain but likely to vary according to natural clarity of freshwater ecosystems. Some regions' water clarity has greatly reduced (sometimes locally exacerbated by water resource management). Coarser sediments can be deposited in systems and result in sediment infill (shallower) and changed hydrology. *Wet season events*	High to very low (depending on location within the Great Barrier Reef, particularly relating to the area of intensive agriculture upstream). *Wet season pulsed exposure with highest concentrations in first-flush events. Irrigated areas can be exposed to high concentrations over long periods under low flow conditions prior to the wet season.*
Estuarine reaches of the rivers	Moderate to low (limited understanding of effects on biota and water quality data; trophic interactions). *Dry season and first flush*	Uncertain; however, there is likely to be coagulation and settling in the dry season. In the wet season, estuarine processes occur further offshore. *Potentially year-round depending on flow*	High to very low (depending on location within the Great Barrier Reef, particularly relating to the area of intensive agriculture upstream). *Wet season pulsed exposure with highest concentrations in first-flush events. Some high value wetlands downstream of irrigated areas can be exposed to high concentrations over long periods under low flow conditions.*
Coastal intertidal and subtidal seagrass	High to low. Interaction with sediment leads to flocs and increased turbidity; can promote growth of epiphytes and macroalgae. *Wet season and months following wet season*	High to low depending on the inputs of excess fine suspended sediment (link demonstrated between sediment input and photic depth). *Months following large river flows*	Moderate to very low. Increased risk under low-light, high-temperature conditions. *Wet season, especially first-flush events. Pulsed exposure is likely.*

Receiving environment	Nutrients – DIN[1] Likelihood of exposure *Key times*	Sediment[2] Likelihood of exposure *Key times*	Pesticides[3] Likelihood of exposure *Key times*
Coral reefs—inner shelf	Locally high (high flow conditions only). Interaction with sediment leads to flocs and increased turbidity, link to increased incidence of coral disease, increased bioeroders, macroalgae, increased bleaching susceptibility. *Wet season only*	High to low depending on the inputs of excess fine suspended sediment and location: turbid nearshore and shallow reefs likely unaffected, while reefs where large change in clarity has occurred are affected, particularly on the reef slope margins (link demonstrated between sediment input and photic depth and studies showing lower food quality for grazing herbivores on reefs). *Months following large river flows and resuspension events*	Low to negligible risk. *Possible low-level exposure in wet season only*
Seagrass—deepwater	Moderate to low *Wet season and months following wet season*	Moderate to low *Months following large river flows and resuspension events*	Low to negligible risk *Possible low-level exposure, especially in wet season*
Coral reefs—mid- and outer shelf	Moderate (Crown-of-Thorns starfish and bleaching; relative to Wet Tropics and Central Great Barrier Reef) to low *Large wet season events only*	Low to no risk (studies show some link between influence of excess fine suspended sediment and lowered photic depth on mid-shelf reefs) *Months following large river flows and resuspension events*	Very low to negligible risk *Large wet season events only*

Data sources:
[1] Brodie et al. (2016); Waterhouse et al. (2016c)
[2] Lewis et al. (2015); Davis et al. (2017)
[3] Lewis et al. (2013); Brodie et al. (2013a); Kennedy et al., (2012a, 2012b)

The data inputs selected to represent the likelihood of exposure to sediments, nutrients and pesticides to Great Barrier Reef marine and coastal ecosystems are described below.

6.2 Assessing the likelihood of exposure to pollutant pressures—marine ecosystems

6.2.1 Nutrients

Land-sourced run-off containing elevated nutrient concentrations results in flood plumes in the Great Barrier Reef lagoon which may result in a range of impacts on coral communities (Tomascik and Sander, 1985; Ward and Harrison, 1997). Dissolved inorganic and particulate forms of nutrients discharged into the Great Barrier Reef are both important in driving ecological effects. Most of the terrestrial nitrogen and phosphorus discharged into the Great Barrier Reef is in particulate form (Furnas, 2003), but this varies greatly between basins (see Chapter 2). However, dissolved inorganic forms of nitrogen and phosphorus are 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 time frames, and dissolved organic forms typically have limited and delayed bioavailability (Furnas et al., 2013), highlighting the relevance of wet season and annual nutrient conditions.

It is currently thought that increased nitrogen inputs are more important than phosphorus inputs, although this is still uncertain and is a point of ongoing global debate (e.g. Smith, 1984). Freshwater systems are most commonly regarded as phosphorus-limited (e.g. Schindler, 1977). Careful bioassay testing of nitrogen versus phosphorus limitation (e.g. Ptacnik et al., 2010) indicates a broad boundary between nitrogen and phosphorus limitation in natural systems which is very much influenced by conditions and affected communities at the time. Available nutrient data from the Great Barrier Reef system (reef waters, rainfall, Coral Sea, rivers) clearly show that phytoplankton biomass is constrained by the availability of readily bioavailable inorganic forms of nitrogen (Furnas et al., 2005; Furnas et al., 2011; Furnas et al., 2013). Further justification of this conclusion was provided in the 2013 Scientific Consensus Statement (drawn from Furnas et al., 2013) and there is limited new information for this report, therefore the evidence is not repeated here.

Based on this knowledge, the likelihood assessment for nutrients in this chapter focuses on DIN. The confidence and availability of data for DIN is also greater than for other nutrient forms, including phosphorus and particulate nutrients, due to complexities in transformation and processing.

While most nutrients are delivered to the Great Barrier Reef in wet season conditions, ongoing processing and availability of nutrients are also important; therefore, wet season and annual conditions are relevant in assessing the potential risk of nutrients to Great Barrier Reef ecosystems.

The likelihood of exposure to DIN in the Great Barrier Reef was assessed using three different spatial layers (Figure 12) and accounts for wet season and annual influences, summarised below and explained in further detail in Appendix 3.

DIN Likelihood of Exposure layer = $(\Delta DIN_{load} \times Freq_{P+S}) + \Delta Chla_{conc}$

Wet season influence:

i. Anthropogenic DIN loading (ΔDIN_{load}, $0-1$): Represents the dispersion of end-of-catchment DIN loads during the wet season, calculated as an anthropogenic influence by comparing the difference between long-term current (2003-2016) average DIN loading with a pre-development DIN load scenario. It highlights the areas of greatest change with current land-use characteristics. This layer was produced using an ocean colour (satellite) based model, and outputs were normalised to the maximum value across the Great Barrier Reef, between 0 (lowest) and 1 (highest).

ii. Wet season water type frequency map ($Freq_{P+S}$, 0- 1): Represents the long-term (2003-2016) likelihood of occurrence of DIN–enriched surface waters (also referred as the primary and secondary wet season water types). These waters are mapped through a supervised colour classification of MODIS satellite imagery (Álvarez-Romero et al., 2013). This layer highlights the areas with the greatest probability of being exposed to DIN–enriched (>23 µg/L) waters during the wet season. The long-term output was normalised to the maximum value, with a final value between 0 (lowest) and 1 (highest) allocated to each pixel (1 km^2 grid).

The anthropogenic DIN loading layer is combined with the wet season water type frequency layer by multiplying the two spatial layers. This combination provides an indication of where the greatest probability of being exposed to DIN–enriched waters is likely to be from river discharges, as opposed to other external drivers in the marine environment. The result is shown in Figure 13.

Annual influence:

iii. Anthropogenic Chla ($\Delta Chla_{conc}$, 0-1): Represents the difference between current (2011-2014) and pre-development annual average concentrations of nutrients (measured as Chl-*a*) in the water column as a measure of anthropogenic annual average Chl-*a* conditions. Produced using the eReefs coupled hydrodynamic-biogeochemical model

(Baird et al., 2016) current (GBR4_H1p85_B1p0_Cbas_Dhnd) and pre-development (GBR4_H1p85_B1p0_Cpre_Dhnd) simulations. The outputs were normalised to the maximum value, between 0 (lowest) and 1 (highest), and each pixel (size) was allocated a normalised value, as shown in Figure 13.

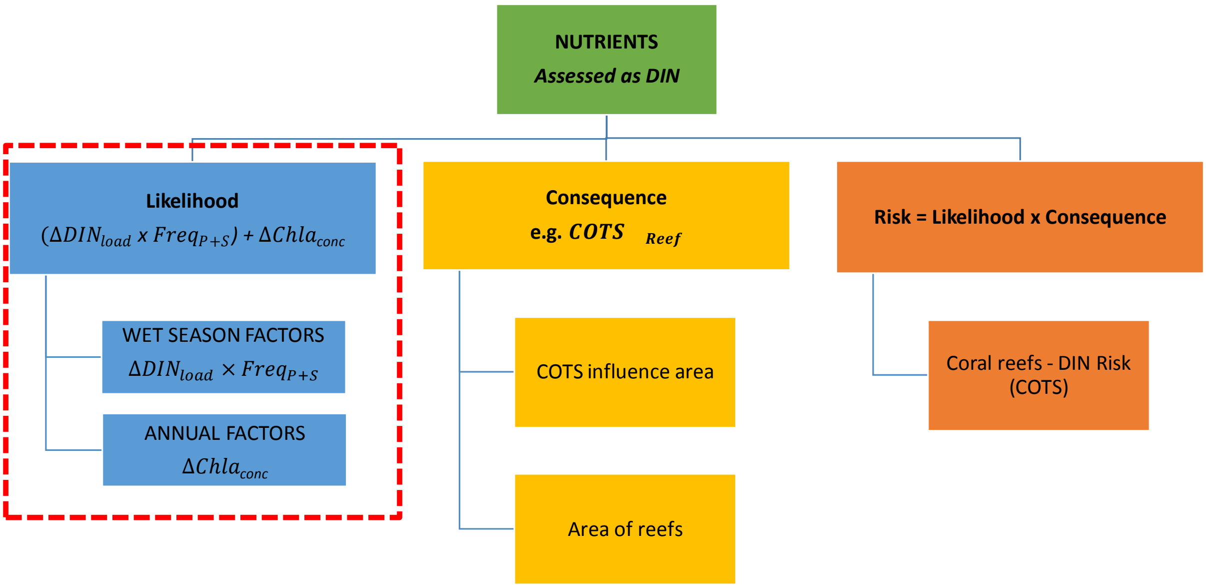


Figure 12. Framework for assessing DIN Likelihood, Crown-of-Thorns starfish consequence and Crown-of-Thorns starfish DIN Risk.

Combining the data

To combine the input data to develop a combined DIN Likelihood of Exposure map, all input data were converted to a common pixel size (~1 km^2 grid). The normalised input layers were added together and the resulting layer divided by its maximum value to normalise from 0 to 1. The final output was classified into six final categories from Negligible to Very High (see Figure 13).

The areas of coral reefs, surveyed seagrass, modelled deepwater seagrass and total areas in each of the final likelihood categories in each Marine Zone were calculated by overlaying the final DIN Likelihood of Exposure map with the spatial layers for each habitat type. A *DIN Likelihood Score* was generated for each Marine Zone by summing the area of coral reefs or seagrass in the Moderate, High and Very High classes and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum.

Example

If the maximum summed area of coral reefs in the Moderate, High and Very High areas of DIN exposure was 80 km^2 in the Wet Tropics Marine Zone, then the Wet Tropics Marine Zone is attributed a DIN Likelihood Score of 1.0, and all other results are presented relative to that. For instance, if the area of coral reefs in the Moderate, High and Very High area in the Fitzroy Marine Zone is 33 km^2, then the DIN Likelihood Score = 33/80 = 0.41, indicating that the likelihood of exposure of coral reefs to DIN in the Fitzroy Marine Zone is approximately 41% of that of the Wet Tropics Marine Zone.

Results

The final input layers and the final DIN Likelihood of Exposure map are shown in the panel in Figure 13. The total areas and the areas of coral reefs and seagrass (surveyed and modelled deepwater)

within each likelihood category in each of the Marine Zones and the DIN Likelihood Scores are presented in Table 3.

For *coral reefs, the greatest likelihood of exposure to anthropogenic DIN is in the Wet Tropics Marine Zone* because it has the largest area in the highest likelihood of exposure categories (80 km^2). The result for the Burdekin Marine Zone is similar (75 km^2), equating to 93% (equivalent to Index 0.93) of the area exposed in the Wet Tropics Marine Zone. The areas in the Fitzroy (33 km^2, Index 0.41), Mackay Whitsunday (21 km^2, Index 0.26) and Burnett Mary (13 km^2, Index 0.16) are relatively small in comparison (Table 3). The Cape York Marine Zones did not show any likelihood of exposure to DIN; while the results for this region are highly uncertain due to data limitations, this result is consistent with current understanding and previous assessments (Waterhouse et al., 2016a). The proportion of coral reefs in the highest likelihood of exposure categories (Moderate to Very High) ranged from 65% of the total area of coral reefs in the Burdekin Marine Zone, to 15% of the Wet Tropics and less than 10% in Mackay Whitsunday, Fitzroy and Burnett Mary. However, the total area of coral reefs in the Burdekin Marine Zone is relatively small compared to the Wet Tropics (115 km^2 compared to 531 km^2) but these proportional assessments may be important at a local scale.

The seagrass data are assessed separately as survey-monitored and modelled deepwater seagrass. For *surveyed seagrass, the greatest likelihood of exposure to anthropogenic DIN is in the Burdekin Marine Zone* because it has the largest area in the highest likelihood of exposure categories (427 km^2). The results for other Marine Zones are comparatively lower, with an Index of 0.41 in the Wet Tropics Marine Zone (175 km^2), Fitzroy (Index 0.37), Mackay Whitsunday (Index 0.27) and Burnett Mary (Index 0.1) (Table 3). Again, the Cape York Marine Zones did not show any likelihood of exposure to DIN. The proportion of surveyed seagrass in the highest likelihood of exposure categories (Moderate to Very High) ranged from 90% of the total area of surveyed seagrass in the Wet Tropics Marine Zone, 54% of the Burdekin, 41% of Mackay Whitsunday, 22% of Fitzroy and less than 2% in Burnett Mary. As with coral reefs, the large proportions within a Marine Zone may be important at a local scale.

For *modelled deepwater seagrass, the greatest likelihood of exposure to anthropogenic DIN is in the Wet Tropics Marine Zone* because it has the largest area in the highest likelihood of exposure categories (585 km^2). The results for other Marine Zones are comparatively lower, with an Index of 0.2 for the Fitzroy Marine Zone, Mackay Whitsunday (0.18), Burdekin (0.13), Burnett Mary (0.05) and Indexes less than 0.05 for all the Cape York Marine Zones (Table 3). The proportion of modelled deepwater seagrass in the highest likelihood of exposure categories (Moderate to Very High) ranged from 48% of the total area of modelled deepwater seagrass in the Mackay Whitsunday Marine Zone, 20% of the Burdekin and less than 10% of all other Marine Zones.

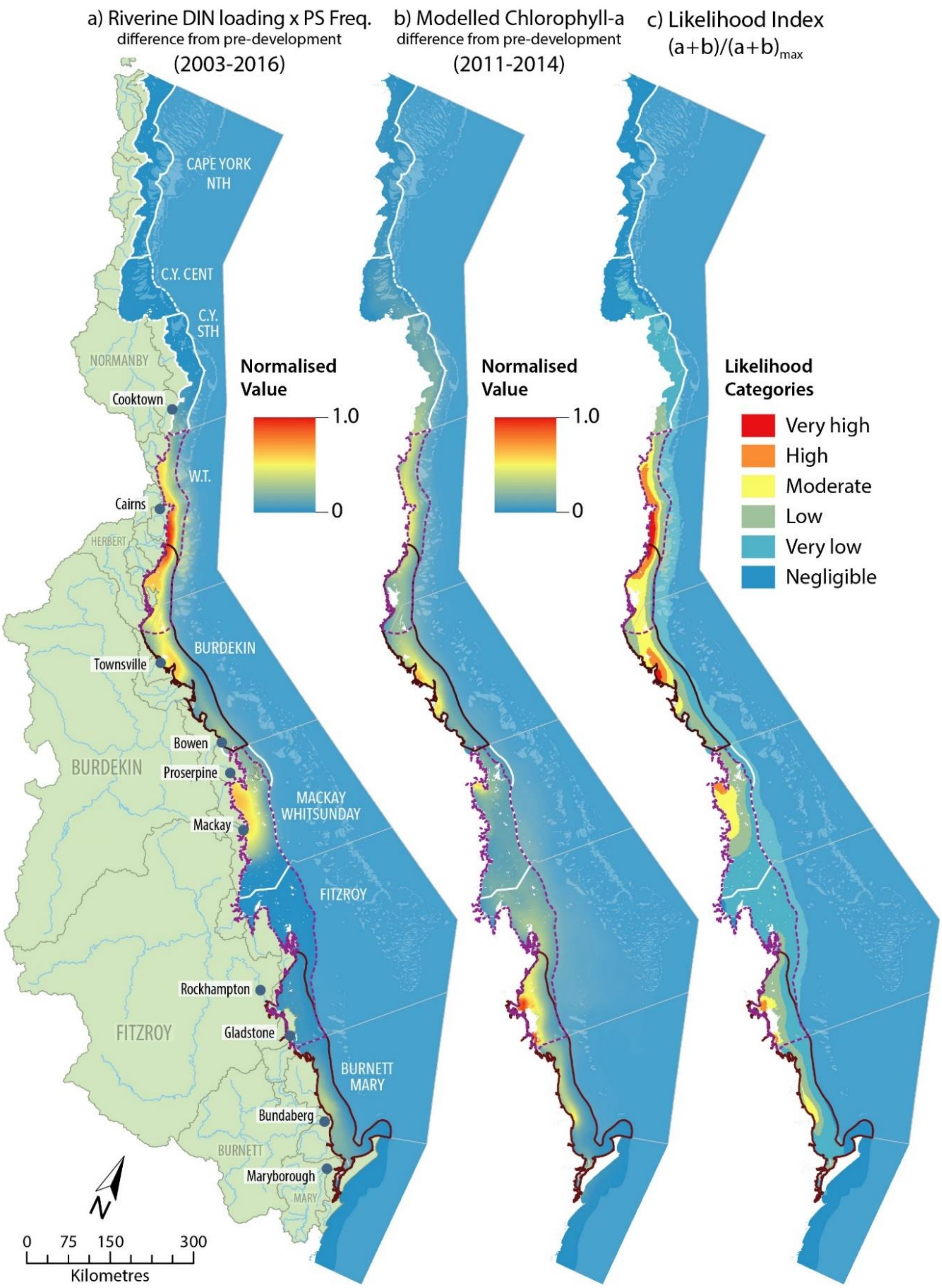


Figure 13. Inputs representing likelihood of exposure of anthropogenic river-derived DIN to coral reefs and seagrass. a) DIN loading, difference between multi-annual average (2003-2016) calculated from satellite-derived plume extents and pre-development load scenario; b) Difference between the current annual average chlorophyll *a* (2011-2014) and the pre-development scenario calculated from the eReefs biogeochemical model; and c) Modelled likelihood of exposure of Great Barrier Reef ecosystems to anthropogenic DIN. Purple lines show Marine Zones.

Table 3. Calculations of the total areas in each final DIN likelihood category within each Marine Zone and the areas of coral reefs and seagrass (surveyed and modelled deepwater). The DIN Likelihood Score is calculated for each Marine Zone by summing the area of coral reefs or seagrass in the Moderate, High and Very High Likelihood categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. The scores are ranked among the Marine Zones, from the highest (1) to lowest score.

CORAL REEFS	Area (km^2) per likelihood category							DIN LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	2,198	-	-	-	-	-	2,198	0.0	0.0%	0.00
Cape York Cent.	625	501	-	-	-	-	1,126	0.0	0.0%	0.00
Cape York Sth	63	775	134	-	-	-	971	0.0	0.0%	0.00
Wet Tropics	-	230	221	56	20	4	531	80	15%	1.00
Burdekin	1	18	21	60	13	2	115	75	65%	0.93
Mackay Whitsunday	3	163	83	20	1	-	270	21	7.7%	0.26
Fitzroy	42	318	117	31	2	-	511	33	6.5%	0.41
Burnett Mary	53	38	29	12	1	-	133	13	9.4%	0.16
							Max	*80*		

SEAGRASS (surveyed)	Area (km^2) per likelihood category							DIN LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	299	-	-	-	-	-	299	-	0.0%	0.00
Cape York Cent.	361	196	-	-	-	-	557	-	0.0%	0.00
Cape York Sth	-	1,66 4	41	-	-	-	1,705	-	0.0%	0.00
Wet Tropics	-	1	19	132	43	1	194	175	90%	0.41
Burdekin	0	44	313	307	118	2	784	427	54%	1.00
Mackay Whitsunday	1	57	111	110	7	-	287	117	41%	0.27
Fitzroy	2	262	306	139	21	-	730	159	22%	0.37
Burnett Mary	195	1,83 2	529	30	13	-	2,600	43	1.7%	0.10
							Max	*427*		

DEEPWATER SEAGRASS (modelled)	Area (km^2) per likelihood category							DIN LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	1,503	-	-	-	-	-	1,503	-	0.0%	0.00
Cape York Cent.	901	674	-	-	-	-	1,575	-	0.0%	0.00
Cape York Sth	58	3,37 3	754	9	-	-	4,194	9	0.2%	0.02
Wet Tropics	-	549	1,734	509	69	6	2,868	585	20%	1.00
Burdekin	-	396	570	74	-	-	1,040	74	7.1%	0.13
Mackay Whitsunday	-	18	96	99	5	-	218	104	48%	0.18
Fitzroy	10	319	126	114	5	-	574	119	21%	0.20
Burnett Mary	173	737	476	32	-	-	1,418	32	2.2%	0.05
							Max	*585*		

Linking DIN likelihood to individual basins

It is important to link the likelihood of exposure for each Marine Zone back to individual basins to inform priority areas for managing pollutant run-off. However, it is difficult to assess the contribution of pollutant discharge from each basin to the Great Barrier Reef. While the eReefs hydrodynamic model can assess the extent of river discharge into the Great Barrier Reef on a daily basis, this assessment is being conducted for longer term average conditions. In addition, a combination of methods was used to establish the Marine Zones (including the eReefs tracer data as one input), so it would be difficult to assign the basin contributions volumetrically with an adequate degree of confidence at this stage. Therefore, to attribute the likelihood of DIN exposure to individual basins, the anthropogenic end-of-catchment DIN load for each Marine Zone was calculated (using the modelled 2013 baseline estimates), and the proportion that each river contributes to that total load was calculated as a *DIN Load Index*.

To determine which rivers contribute to each Marine Zone, it was assumed that those rivers directly adjacent to the zone contributed 100% of their load to that zone. In some cases, where there was evidence that the discharge from rivers crosses between zones, a proportional contribution from that river into another zone was estimated. These estimations were based on knowledge of plume movement (e.g. Álvarez-Romero et al., 2013; Devlin et al., 2012a; Devlin et al., 2012b; Devlin et al., 2015a; Lønborg et al., 2016; Petus et al., 2014a; Petus et al., 2016), recent assessments to support the Water Quality Improvement Plans (Waterhouse et al., 2014a; Waterhouse et al., 2014b; Waterhouse et al., 2015a; Waterhouse et al., 2016a; Waterhouse et al., 2016b), assessment of satellite imagery and the eReefs tracer outputs. The proportional anthropogenic load contributions allocated for each parameter to the Marine Zones are shown in Table 4. It is recognised that the eReefs modelling period included several wet years for the Fitzroy region, so the extension to the Whitsunday regions is likely to be an overestimate in an average year. These limitations are discussed further in Appendix 1.

Table 4. Assumptions for allocating the proportional anthropogenic DIN and TSS load contributions from each basin to the total anthropogenic load of the Marine Zones.

Basin name	Which Marine Zones the basin is likely to contribute to for each parameter	
	DIN	TSS
Jacky Jacky Creek	Cape York North	Cape York North
Olive Pascoe River	Cape York North	Cape York North
Lockhart River	Cape York North	Cape York North
Stewart River	Cape York Central	Cape York Central
Normanby River	Cape York Central	Cape York Central
Jeannie River	Cape York South	Cape York South
Endeavour River	Cape York South	Cape York South
Daintree River	Wet Tropics + 5% Cape York South	Wet Tropics
Mossman River	Wet Tropics + 5% Cape York South	Wet Tropics
Barron River	Wet Tropics	Wet Tropics
Mulgrave-Russell River	Wet Tropics + 5% Cape York South	Wet Tropics
Johnstone River	Wet Tropics + 5% Cape York South	Wet Tropics
Tully River	Wet Tropics + Burdekin	Wet Tropics + Burdekin
Murray River	Wet Tropics + Burdekin	Wet Tropics + Burdekin
Herbert River	Wet Tropics + Burdekin	Wet Tropics + Burdekin
Black River	Burdekin	Burdekin
Ross River	Burdekin	Burdekin
Haughton River	Burdekin + 5% Wet Tropics	Burdekin
Burdekin River	Burdekin + 5% Wet Tropics	Burdekin
Don River	Burdekin	Burdekin
Proserpine River	Mackay Whitsunday + Fitzroy	Mackay Whitsunday + Fitzroy
O'Connell River	Mackay Whitsunday + Fitzroy	Mackay Whitsunday + Fitzroy
Pioneer River	Mackay Whitsunday + Fitzroy	Mackay Whitsunday + Fitzroy
Plane	Mackay Whitsunday + Fitzroy	Mackay Whitsunday + Fitzroy
Styx River	Fitzroy	Fitzroy
Shoalwater Creek	Fitzroy	Fitzroy
Waterpark Creek	Fitzroy + Burnett Mary	Fitzroy + Burnett Mary
Fitzroy River	Fitzroy + Burnett Mary	Fitzroy + Burnett Mary
Calliope River	Fitzroy + Burnett Mary	Fitzroy + Burnett Mary
Boyne River	Fitzroy + Burnett Mary	Fitzroy + Burnett Mary
Baffle Creek	Burnett Mary	Burnett Mary
Kolan River	Burnett Mary	Burnett Mary
Burnett River	Burnett Mary + 5% Fitzroy	Burnett Mary
Burrum River	Burnett Mary	Burnett Mary
Mary River	Burnett Mary	Burnett Mary

A *DIN Load Index* was calculated for each basin using the assumptions in Table 4. The DIN Load Index was then multiplied by the *DIN Likelihood Scores* in Table 3 for each Marine Zone, to generate a *DIN Likelihood Index*, presented in Table 5. Where a river had more than one Index result because it contributed to more than one Marine Zone, the highest Index was used in the overall ranking.

Example 1

The Herbert River is assessed as contributing DIN loads to the Wet Tropics and Burdekin Marine Zones. For the Wet Tropics Marine Zone calculation, it is assumed to contribute 100% of the annual average DIN load (886 tonnes), which is 32% of the total DIN load contributions to the Wet Tropics Marine Zone (2804 tonnes). This is the largest basin contributor to that zone, so the DIN Load Index for the Herbert Basin is 1.00 (the maximum). The DIN Likelihood Score for coral reefs in the Wet Tropics Marine Zone is 1.00 (the maximum), so the Herbert DIN Likelihood Index = DIN Load Index (1.00) x DIN Likelihood Score (1.00) = 1.00.

Following the same calculation for the Herbert contribution to the Burdekin Marine Zone [DIN Load Index (0.97) x DIN Likelihood Score (0.93) = 0.90], the DIN Likelihood Index is lower than in the Wet Tropics Marine Zone, so the highest score of 1.00 prevails in the final assessment.

Example 2

The Tully River is assessed as contributing DIN loads to the Wet Tropics Marine Zone. For the Wet Tropics Marine Zone calculation, it is assumed to contribute 100% of the annual average DIN load (384 tonnes), which is 14% of the total DIN load contributions (2804 tonnes) to the Wet Tropics Marine Zone. This is relative to the Herbert River which was the maximum, so the DIN Load Index for the Tully Basin is 0.43 (i.e. Tully 0.14/ max Herbert 0.32 = 0.43). The DIN Likelihood Score for coral reefs in the Wet Tropics Marine Zone is 1.00 (the maximum), so the Tully DIN Likelihood Index = DIN Load Index (0.43) x DIN Likelihood Score (1.00) = 0.43.

The Index is only presented for coral reefs, as although the exposure of seagrass is considered to be important, the direct effects of inorganic nutrient loading on seagrass are less well known. In part, this is because the effect of nutrient loads has so far been difficult to elucidate as light limitation, caused by sediment loads, has been the primary driver in recent years (Collier and Waycott, 2009; Petus et al., 2014b; Collier et al., 2011; Collier et al., 2012). From the limited number of studies in the Great Barrier Reef, and from research in other regions, we know that nutrient enrichment can stimulate seagrass growth, rather than impair growth (Udy and Dennison, 1997; Udy et al., 1999; Mellors, 2003; Collier et al., 2015) if other factors, such as light availability, are not limiting (Collier et al., 2015). Although a theoretical nutrient toxicity level does exist, nutrient overenrichment tends to impact at ecosystem scales and follow a path of eutrophication with excessive production of organic matter. Nutrients favour the growth of plankton, macroalgae and epiphytic algae, all of which attenuate light to seagrass leaves (Collier, 2013). In the Great Barrier Reef some very high epiphyte loads occur on seagrass meadows of the Great Barrier Reef (McKenzie et al., 2012a; McKenzie et al., 2012b) 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., 2012a; McKenzie et al., 2012b). Although nutrient enrichment has been linked to high algal cover (Campbell et al., 2002), seagrass loss has rarely been attributed to nutrient overenrichment. In summary, dissolved inorganic nutrients can have a direct benefit for seagrass growth, but impact seagrass indirectly through plankton, epiphyte and microalgae blooms. Our understanding of these pathways is insufficient for a quantitative risk assessment. The likelihood of exposure has, however, been assessed in acknowledgement that DIN exposure could be important for seagrass condition and resilience. Further discussion of the impact of flood plumes and degraded water quality on seagrass ecosystems in the Great Barrier Reef is included in Petus et al. (2014b), Chapter 1 (Schaffelke et al., 2017), and in the consideration of Consequences (Section 7) in this chapter.

Indexes among the rivers within a Marine Zone are driven by the anthropogenic loads (as all rivers in a zone are allocated the same DIN Likelihood Score), which is reflected in these results, but the assessment also provides a relative ranking of basins across the Great Barrier Reef.

The *assessment of the likelihood of exposure to anthropogenic DIN at a basin scale* indicates that the Herbert Basin has the greatest likelihood of coral reef exposure to anthropogenic DIN (Index 1.00), followed by the Haughton Basin (Index 0.93). Other Wet Tropics basins present approximately half the likelihood of these basins, including the Johnstone, Russell-Mulgrave and Tully basins (Indexes 0.56 to 0.43). The Plane Basin in the Mackay Whitsunday region also has a relatively high DIN Likelihood Index (0.41).

Table 5. Calculation of a DIN Likelihood Index for each basin (coral reefs) using a DIN Load Index based on the proportion of anthropogenic DIN load that each basin contributes to the total anthropogenic DIN load of the Marine Zone and the DIN Likelihood Scores for each Marine Zone (Table 3). The contribution weighting of each river to the Marine Zone is shown in Table 4, resulting in an adjusted anthropogenic load contribution to each Marine Zone for each basin (highlighted in red text). The DIN Likelihood Index is ranked; the top 5 rivers are highlighted in red shading, and the rivers ranked 5 to 10 are highlighted in orange shading.

Marine zone	Basin name	DIN Load Index						DIN Likelihood Index		
		DIN anth. baseline (2012-2013) (tonnes)	Contribution weighting to zone	DIN anth. load to apply	DIN anth. load as % Marine Zone load	Basin ranking within zone	DIN Load Index within Marine Zone	DIN Likelihood Score	DIN Likelihood Index (DIN Load Index x Likelihood Score)	RANK across GBR
Cape York North	Jacky Jacky Creek	0	1.00	0.1	0.08	2	0.09	0.00	0.00	43
	Olive Pascoe River	1	1.00	0.6	0.90	1	1.00	0.00	0.00	43
	Lockhart River	0	1.00	0.0	0.02	3	0.02	0.00	0.00	43
	REGIONAL TOTAL *and Max*			**0.7**	*0.90*					
Cape York Central	Stewart River	0	1.00	0.1	0.01	2	0.01	0.00	0.00	43
	Normanby River	9	1.00	9.0	0.99	1	0.01	0.00	0.00	43
	REGIONAL TOTAL			**9.1**	*0.99*					
Cape York South	Jeannie River	0	1.00	0.2	0.00	6	0.01	0.00	0.00	43
	Endeavour River	1	1.00	1.1	0.02	5	0.04	0.00	0.00	43
	Daintree River	135	0.05	6.7	0.11	3	0.27	0.00	0.00	43
	Mossman River	104	0.05	5.2	0.09	4	0.21	0.00	0.00	43
	Mulgrave-Russell River	423	0.05	21	0.36	2	0.85	0.00	0.00	43
	Johnstone River	499	0.05	25	0.42	1	1.00	0.00	0.00	43
	REGIONAL TOTAL			**59**	*0.42*					
Wet Tropics	Daintree River	135	1.00	135	0.05	6	0.15	1.00	0.15	17
	Mossman River	104	1.00	104	0.04	7	0.12	1.00	0.12	22
	Barron River	87	1.00	87	0.03	8	0.10	1.00	0.10	24
	Mulgrave-Russell River	423	1.00	423	0.15	3	0.48	1.00	0.48	5
	Johnstone River	499	1.00	499	0.18	2	0.56	1.00	0.56	4
	Tully River	384	1.00	384	0.14	4	0.43	1.00	0.43	6
	Murray River	232	1.00	232	0.08	5	0.26	1.00	0.26	9
	Herbert River	886	1.00	886	0.32	1	1.00	1.00	1.00	1

Marine zone	Basin name	DIN Load Index						DIN Likelihood Index		
		DIN anth. baseline (2012-2013) (tonnes)	Contribution weighting to zone	DIN anth. load to apply	DIN anth. load as % Marine Zone load	Basin ranking within zone	DIN Load Index within Marine Zone	DIN Likelihood Score	DIN Likelihood Index (DIN Load Index x Likelihood Score)	RANK across GBR
	Burdekin River	171	0.05	9	0.00	10	0.01	1.00	0.01	33
	Haughton River	914	0.05	46	0.02	9	0.05	1.00	0.05	29
	REGIONAL TOTAL			**2,804**	*0.32*					
Burdekin	Tully River	384	1.00	384	0.14	3	0.42	0.93	0.39	8
	Murray River	232	1.00	232	0.08	4	0.25	0.93	0.24	11
	Herbert River	886	1.00	886	0.32	2	0.97	0.93	0.90	3
	Black River	21	1.00	21	0.01	8	0.02	0.93	0.02	31
	Ross River	123	1.00	123	0.04	6	0.13	0.93	0.13	21
	Haughton River	914	1.00	914	0.33	1	1.00	0.93	0.93	2
	Burdekin River	171	1.00	171	0.06	5	0.19	0.93	0.17	15
	Don River	68	1.00	68	0.02	7	0.07	0.93	0.07	28
	REGIONAL TOTAL			**2,799**	*0.33*					
Mackay Whitsunday	Proserpine River	157	1.00	157	0.17	4	0.43	0.26	0.11	23
	O’Connell River	186	1.00	186	0.21	3	0.51	0.26	0.13	20
	Pioneer River	193	1.00	193	0.21	2	0.53	0.26	0.14	19
	Plane	366	1.00	366	0.41	1	1.00	0.26	0.26	10
	REGIONAL TOTAL			**902**	*0.41*					
Fitzroy	Proserpine River	157	1.00	157	0.17	4	0.43	0.41	0.18	14
	O'Connell River	186	1.00	186	0.21	3	0.51	0.41	0.21	13
	Pioneer River	193	1.00	193	0.21	2	0.53	0.41	0.22	12
	Plane	366	1.00	366	0.41	1	1.00	0.41	0.41	7
	Styx River	10	1.00	10	0.01	7	0.02	0.41	0.01	35
	Shoalwater Creek	5	1.00	4.9	0.00	9	0.01	0.41	0.00	37
	Waterpark Creek	4	1.00	3.7	0.00	10	0.01	0.41	0.00	38
	Fitzroy River	159	1.00	159	0.14	5	0.36	0.41	0.15	18
	Calliope River	6	1.00	5.6	0.01	8	0.01	0.41	0.01	36

Marine zone	Basin name	DIN Load Index						DIN Likelihood Index		
		DIN anth. baseline (2012-2013) (tonnes)	Contribution weighting to zone	DIN anth. load to apply	DIN anth. load as % Marine Zone load	Basin ranking within zone	DIN Load Index within Marine Zone	DIN Likelihood Score	DIN Likelihood Index (DIN Load Index x Likelihood Score)	RANK across GBR
	Boyne River	3	1.00	2.6	0.00	11	0.01	0.41	0.00	40
	Burnett River	207	0.05	10	0.01	6	0.02	0.41	0.01	34
	REGIONAL TOTAL			**1,097**	*0.41*					
Burnett Mary	Waterpark Creek	4	1.00	3.7	0.00	8	0.01	0.16	0.00	41
	Fitzroy River	159	1.00	159	0.15	4	0.44	0.16	0.07	27
	Calliope River	6	1.00	5.6	0.01	7	0.02	0.16	0.00	39
	Boyne River	3	1.00	2.6	0.00	9	0.01	0.16	0.00	42
	Baffle Creek	32	1.00	32	0.03	6	0.09	0.16	0.01	32
	Kolan River	68	1.00	68	0.07	5	0.19	0.16	0.03	30
	Burnett River	207	1.00	207	0.20	2	0.57	0.16	0.09	25
	Burrum River	186	1.00	186	0.18	3	0.51	0.16	0.08	26
	Mary River	361	1.00	361	0.35	1	1.00	0.16	0.16	16
	REGIONAL TOTAL			**1,024**	*0.35*					

6.2.2 Sediments

The likelihood assessment for sediment accounts for wet season and annual influences. The factors selected include model outputs that represent the predicted dispersion of end-of-catchment TSS loads in the wet season (multi-annual average) presented as an anthropogenic influence, suspended sediment exposure in the wet season (based on frequency of wet season water types relative to the Great Barrier Reef Water Quality Guidelines for TSS) and the difference between the current annual light attenuation and pre-development scenarios (Figure 14). Note that anthropogenic influences are assessed by modelling scenarios of pre-development TSS load estimates for each basin.

The likelihood of exposure to TSS in the Great Barrier Reef was assessed using four different spatial layers and accounts for wet season and annual influences, summarised below and explained in further detail in Appendix 3.

TSS Likelihood of Exposure layer = $(\Delta TSS_{load}\ x\ Freq_{P+S}) + \mathrm{Exp}_{\mathrm{TSS}} + \Delta Light$

Wet season influence:

i. Anthropogenic TSS loading $(\Delta TSS_{\mathrm{load}}, 0-1)$: Represents the dispersion of end-of-catchment TSS loads during the wet season, calculated as an anthropogenic influence by comparing the difference between long-term current (2003-2016) average TSS loading with a pre-development TSS load scenario. It highlights the areas of greatest change with current land use characteristics. This layer was produced using an ocean colour (satellite) based model, and outputs were normalised to the maximum value across the Great Barrier Reef, between 0 (lowest) and 1 (highest).

ii. Wet season water type frequency map $(\mathrm{Freq}_{\mathrm{P+S}}, 0\text{-} 1)$: Represents the long-term (2003-2016) likelihood of occurrence of TSS enriched surface waters (also referred as the primary and secondary wet season water types). These waters are mapped through a supervised colour classification of MODIS satellite imagery (Álvarez-Romero et al., 2013). This layer highlights the areas with the greatest probability of being exposed to the greatest TSS-enriched (>5.5 mg/L, i.e. 2 times the wet season guideline of 2.4 mg/L; GBRMPA, 2010) waters during the wet season. The long-term output is normalised to the maximum value, with a final value between 0 (lowest) and 1 (highest) allocated to each pixel.

The anthropogenic TSS loading layer (i) was combined with the wet season water type frequency layer (ii) by multiplying the two spatial layers. This combination provides an indication of whether the greatest probability of being exposed to TSS-enriched waters is likely to be from river discharges, as opposed to other external drivers in the marine environment. The result is shown in Figure 15.

iii. TSS exposure $(\mathrm{Exp}_{\mathrm{TSS}}: 0-1)$: Represents the long-term (2003-2016) frequency of TSS-enriched (> 2.4 mg/L) surface waters (also referred as the primary, secondary and tertiary wet season water types) assessed against the Great Barrier Reef Water Quality Guideline for TSS to represent the magnitude and duration of TSS exceedance in the wet season. The surface exposure for TSS is derived from the long-term (2003-2016) primary, secondary and tertiary frequency maps (satellite-derived) and the mean long-term current TSS value measured in-situ in each of the wet season water types relative to the wet season Great Barrier Reef Water Quality Guideline for TSS (2.4 mg/L; GBRMPA, 2010). The long-term output is normalised to the maximum value, with a final value between 0 (lowest) and 1 (highest) allocated to each pixel.

The combined layer from (i) and (ii) is used as a 'modifier' to the TSS exposure layer (iii), as TSS exposure does not specifically link to river derived TSS conditions and may also be driven by wind-driven resuspension. $(\Delta TSS_{load}\ x\ Freq_{P+S}) + \mathrm{Exp}_{\mathrm{TSS}}$

Annual influence:

iv. Anthropogenic Light attenuation ($\Delta Light$, 0-1): Represents the difference between current (2011-2014) average annual conditions of light attenuation relative to pre-development estimates as a measure of anthropogenic averaged annual turbidity conditions. Light attenuation and turbidity are typically dominated by river-derived sediment inputs and resuspension (e.g. Fabricius et al., 2016) but can also be affected by nutrient inputs and phytoplankton biomass (see note below). This layer is produced using the eReefs coupled hydrodynamic-biogeochemical model (Baird et al., 2016). The outputs were normalised to the maximum value, between 0 (lowest) and 1 (highest), and each pixel (size) was allocated a normalised value, as shown in Figure 15.

Note: Light attenuation can be caused by a number of factors in addition to TSS, including clear water attenuation, phytoplankton and colour dissolved organic matter. These factors also influence the water type mapping (used to assess TSS exposure and frequency) so are indirectly included in this assessment. However, when linking the likelihood back to individual basins, only TSS loads are considered as they dominate river influences on light availability in shallow (e.g. less than 12 m) areas through resuspension (Fabricius et al., 2016).

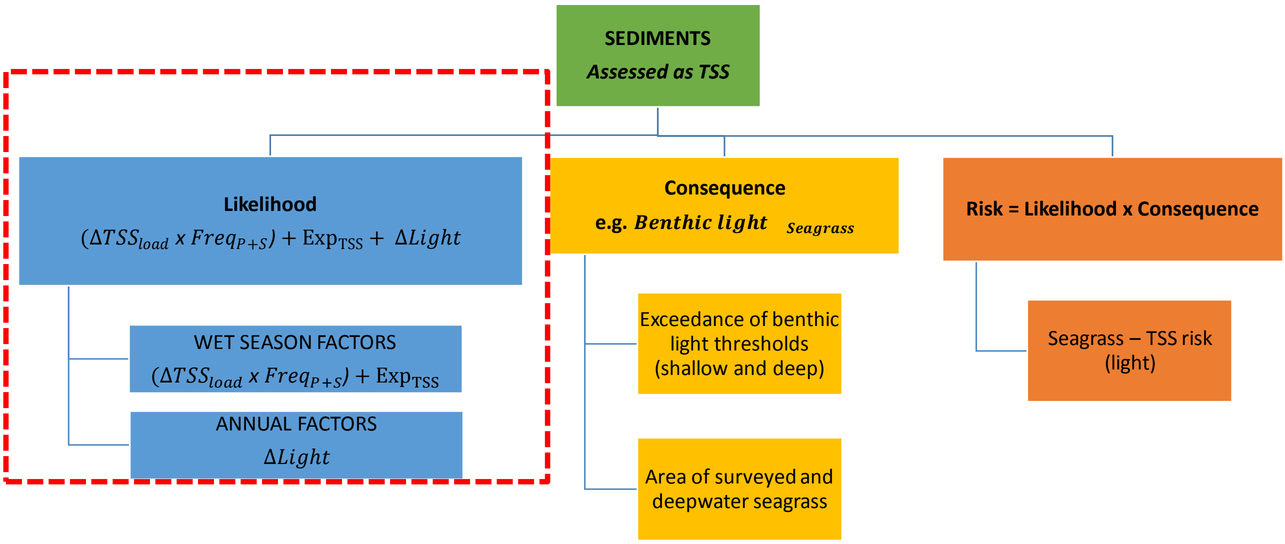


Figure 14. Framework for assessing the likelihood of exposure and consequence of TSS to seagrass and coral reefs. An example of consequence and risk is also provided for TSS, benthic light and seagrass.

Combining the data

To combine the input data to develop a combined TSS Likelihood of Exposure map, all input data were converted to a common pixel size (~1 km^2 grid). The normalised input layers were added together and the resulting layer divided by its maximum value to normalise from 0 to 1. The final output was classified into six final categories from Negligible to Very High (see Figure 15).

The areas of coral reefs, surveyed seagrass, modelled deepwater seagrass and total areas in each of the final likelihood categories in each Marine Zone were calculated by overlaying the final TSS Likelihood of Exposure map with the spatial layers for each habitat type. A *TSS Likelihood Score* was generated for each Marine Zone by summing the area of coral reefs or seagrass in the Moderate, High and Very High classes and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum.

Example

If the maximum summed area of seagrass in the Moderate, High and Very High areas to TSS exposure was 363 km^2 in the Burdekin Marine Zone, then that zone is attributed a TSS Likelihood Score of 1.0, and all other results are presented relative to that. For instance, if the area of seagrass in the Moderate, High and Very High area in the Burnett Mary Marine Zone is 82 km^2, then the TSS Likelihood Score = 82/363 = 0.23, indicating that the likelihood of exposure of seagrass to TSS in the Burnett Mary Marine Zone is approximately 23% of that of the Burdekin Marine Zone.

Results

The final input layers and the final TSS Likelihood of Exposure map are shown in the panel in Figure 15. The total areas and the areas of coral reefs and seagrass (surveyed and modelled deepwater) within each likelihood category in each of the Marine Zones and the TSS Likelihood Scores are presented in Table 6**.**

For *coral reefs, the greatest likelihood of exposure to anthropogenic fine sediments was in the Burdekin Marine Zone* because it has the largest area in the highest likelihood of exposure categories (13 km^2), but the areas of exposure were relatively small for all Marine Zones (Table 6). The results of the Fitzroy and Burnett Mary Marine Zones were similar (9 km^2), equating to 70% equivalent of the area exposed in the Burdekin Marine Zone. The area of highest exposure in the other Marine Zones was less than 5 km^2. The Cape York Marine Zones did not show any likelihood of exposure of coral reefs to TSS, which is likely to be an underestimate based on previous assessments (Waterhouse et al., 2016a); however, the results in this region are also highly uncertain. The proportion of coral reefs in the highest likelihood of exposure categories (Moderate to Very High) was less than 10% of the total area of coral reefs in each Marine Zone.

For *surveyed seagrass, the greatest likelihood of exposure to anthropogenic fine sediment was in the Burdekin Marine Zone*, because it has the largest area in the highest likelihood of exposure categories (363 km^2). The results for other Marine Zones are comparatively lower, with an Index of 0.23 for the Burnett Mary Marine Zone (82 km^2), Wet Tropics (Index 0.18, 66 km^2), Fitzroy (Index 0.17, 61 km^2) and Mackay Whitsunday (Index 0.07, 25 km^2) (Table 6). The Cape York Marine Zones did not show any likelihood of exposure to TSS which is contradictory to current understanding, particularly in the Cape York South Marine Zone which is only showing a small area of influence in this assessment (see Waterhouse et al., 2016a). The extent of influence is likely to be underestimated in the Cape York region due to limitations in the modelled light attenuation (improved understanding of load estimates is not reflected in the model) and the TSS loading maps associated with pre-development load estimates. The proportion of surveyed seagrass in the highest likelihood of exposure categories (Moderate to Very High) ranged from 46% of the total area of surveyed seagrass in the Burdekin Marine Zone, to 34% of the Wet Tropics, and less than 10% of all other Marine Zones.

For *modelled deepwater seagrass, there are no areas within the highest categories of TSS likelihood of exposure, indicating that the likelihood of exposure to the TSS factors included in the assessment is limited within the Marine Zones* (Table 6). This is likely to be associated with depth characteristics, as the model prediction is only for seagrass in more than 15 m depth, which is most likely to be located in the outer parts of the Marine Zones where it is too deep for resuspension in normal (non-cyclonic) wind and wave conditions.

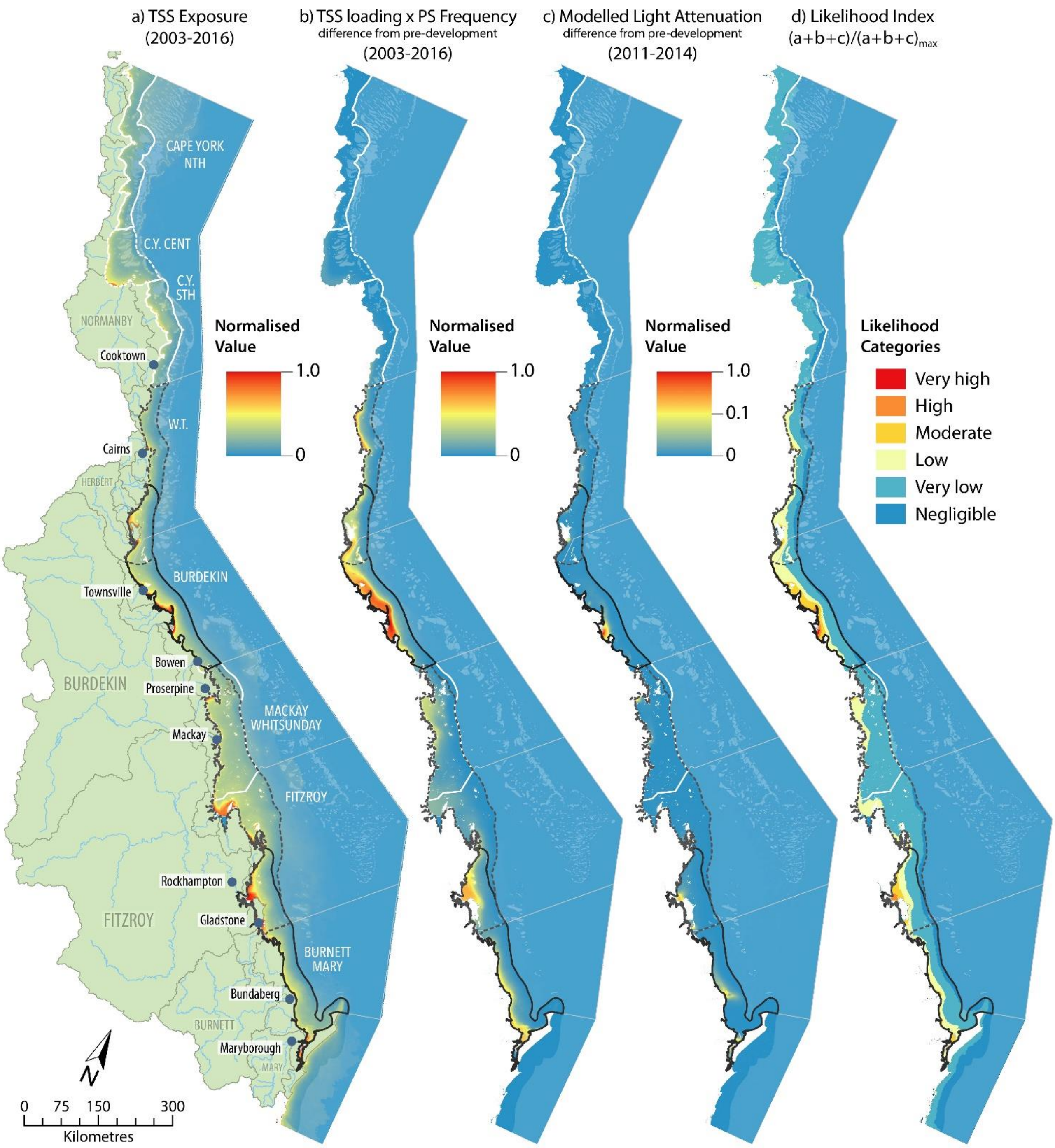


Figure 15. Inputs representing likelihood of exposure of anthropogenic TSS to coral reefs and seagrass. a) TSS exposure (2003-2016); b) TSS loading, difference between multiannual average (2003-2016) and pre-European load scenario x Frequency; c) Annual average light attenuation (2011-2014) difference from pre-development; d) Modelled likelihood of exposure of Great Barrier Reef ecosystems to anthropogenic TSS. Purple lines shown Marine Zones.

Table 6. Calculations of the total areas within each final TSS Likelihood category within each Marine Zone, and the areas of coral reefs and seagrass (surveyed and modelled deepwater). The TSS Likelihood Score is calculated for each Marine Zone by summing the area of coral reefs or seagrass in the Moderate, High and Very High Likelihood categories and normalising the value to the maximum result to provide a relative index between the Marine Zones; that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are then ranked among the Marine Zones, from the highest (1) to lowest score**.**

CORAL REEFS	Area (km^2) per likelihood category							TSS LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	1,838	358	2.0	0.0	0.0	0.0	2,198	-	0.0%	0.00
Cape York Cent.	172	949	5.2	0.0	0.0	0.0	1,126	-	0.0%	0.00
Cape York Sth	525	438	8.3	0.0	0.0	0.0	971	-	0.0%	0.00
Wet Tropics	332	127	67	3.9	0.0	0.0	531	3.9	0.7%	0.30
Burdekin	32	7.2	63	13	0.2	0.0	115	13	11.2%	1.00
Mackay Whitsunday	23	224	23	0.5	0.0	0.0	270	0.5	0.2%	0.04
Fitzroy	76	344	81	9.1	0.1	0.0	511	9.2	1.8%	0.71
Burnett Mary	66	30	29	8.6	0.1	0.0	133	8.7	6.6%	0.68
							Max	*13*		

SEAGRASS (surveyed)	Area (km^2) per likelihood category							TSS LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	12	255	32	-	-	-	299	-	0.0%	0.00
Cape York Cent.	7.3	429	121	-	-	-	557	-	0.0%	0.00
Cape York Sth	153	1,423	127	1.4	-	-	1,705	1.4	0.1%	0.00
Wet Tropics	4.1	10	114	66	-	-	194	66	34.1%	0.18
Burdekin	5.5	86	330	332	29	1.5	784	363	46.3%	1.00
Mackay Whitsunday	11	124	127	25	-	-	287	25	8.6%	0.07
Fitzroy	23	378	267	61	-	-	730	61	8.4%	0.17
Burnett Mary	8.4	1,940	569	82	0.1	-	2,600	82	3.2%	0.23
							Max	*363*		

DEEPWATER SEAGRASS (modelled)	Area (km^2) per likelihood category							TSS LIKELIHOOD SCORE		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	Likelihood Score
Cape York Nth	7.7	1,495	-	-	-	-	1,503	-	0.0%	-
Cape York Cent.	91	1,484	-	-	-	-	1,575	-	0.0%	-
Cape York Sth	1,270	2,922	2.9	-	-	-	4,195	-	0.0%	-
Wet Tropics	1,339	1,460	69	-	-	-	2,868	-	0.0%	-
Burdekin	596	431	13	-	-	-	1,040	-	0.0%	-
Mackay Whitsunday	-	203	16	-	-	-	218	-	0.0%	-
Fitzroy	76	434	63	-	-	-	574	-	0.0%	-
Burnett Mary	546	784	88	-	-	-	1,418	-	0.0%	-
							Max	-		-

Linking total suspended sediment likelihood to individual basins

A *TSS Load Index* was calculated for each basin using the assumptions in Table 4, as described for DIN above. The TSS Load Index was then multiplied by the *TSS Likelihood Scores* in Table 6 for each Marine Zone to generate a *TSS Likelihood Index*, presented in Table 7. As for DIN, where a river had more than one Index result because it contributed to more than one Marine Zone, the highest Index was considered in the overall ranking.

Example

The Herbert River is assessed as contributing TSS loads to the Wet Tropics and Burdekin Marine Zones. For the Wet Tropics Marine Zone calculation, it is assumed to contribute 100% of the annual average TSS load (331 kt), which is 35% of the TSS load contributions to the Wet Tropics Marine Zone (936 kt). This is the largest basin contributor to that zone, so the TSS Load Index for the Herbert Basin is 1.00 (the maximum).

Seagrass: The TSS Likelihood Score for seagrass in the Wet Tropics Marine Zone is 0.30, so the Herbert TSS Likelihood Index = TSS Load Index (1.00) x TSS Likelihood Score (0.30) = 0.30.

Coral reefs: The TSS Likelihood Score for coral reefs in the Wet Tropics Marine Zone is 0.18, so the Herbert TSS Likelihood Index = TSS Load Index (1.00) x TSS Likelihood Score (0.18) = 0.18.

Overall Herbert TSS Likelihood Index: Herbert TSS Likelihood Index for seagrass (0.30) + Herbert TSS Likelihood Index for seagrass (0.18) = 0.48. This result is then normalised to the maximum score across all the Great Barrier Reef basins, which is 2.00 for the Burdekin Basin (i.e. 0.48/2), resulting in a final TSS Likelihood Index of 0.24 for the Herbert Basin.

As with DIN, the Indexes between the basins within a Marine Zone are driven by the anthropogenic loads (as all rivers in a zone are allocated the same TSS Likelihood Score), which is reflected in these results. However, the assessment provides a relative ranking of basins across the Great Barrier Reef.

The assessment of the likelihood of exposure to TSS at a basin scale indicates that the *Burdekin Basin has the greatest likelihood of seagrass exposure to TSS (with dominant influence compared to other rivers), followed by the Fitzroy Basin (but only with around a quarter of the likelihood of exposure of the Burdekin Basin; maximum Index 0.23)*. The Herbert, Mary and Johnstone basins rank relatively high among the rivers, but the likelihood of exposure to TSS is relatively small compared to the Burdekin Basin (Index 0.12–0.18). The Mulgrave-Russell, Haughton, Don, O'Connell and Burnett basins have similar, results with around 6–8% of the influences of the Burdekin Basin (Indexes 0.06–0.08).

The *Burdekin Basin also has the greatest likelihood of coral reef exposure to TSS, followed by the Fitzroy Basin (around 75% of the likelihood of exposure of the Burdekin River; maximum Index 0.75)* (note that the Burdekin Marine Zone extends northwards into the Wet Tropics NRM region). The Mary and Herbert basins also rank relatively high among the rivers, and the likelihood of exposure to TSS is around a third of that of the Burdekin (Indexes 0.3 and 0.35 respectively). The Johnstone and Burnett basins have around a quarter of the influence of the Burdekin River (Indexes 0.24 and 0.22), and the Russell-Mulgrave, Herbert, O'Connell and Pioneer Basins have similar Indexes around 10–15% of the influence of the Burdekin Basin.

When the seagrass and coral reef Indexes are combined, the dominance of the Burdekin Basin is evident, with an Index almost double that of any of the other basins. The Fitzroy Basin is in turn almost double the influence of any of the other basins (maximum Index 0.46). The Herbert (0.24) and Mary (0.23) basins rank highly among the rest of the basins, followed by the Johnstone (0.19), Burnett (0.15), Russell-Mulgrave (0.11), O'Connell (0.08) and Don (0.07) basins. All other basins have Indexes less than 0.06 (or 6%) of the influence of the Burdekin Basin.

Table 7. Calculation of a TSS Likelihood Index for each basin (seagrass and coral reefs) using a TSS Load Index based on the proportion of anthropogenic TSS load that each basin contributes to the total anthropogenic TSS load of the Marine Zone and the TSS Likelihood Scores for each Marine Zone. As shown in Table 4, the contribution weighting of each river to the Marine Zone is 100% in all cases and is therefore not shown here. The results for seagrass and reefs are summed to give an overall TSS Likelihood Index for coral reefs and seagrass. The TSS Likelihood Index is ranked; the top 5 rivers are highlighted in red, and the rivers ranked 5–10 are highlighted in orange**.**

Marine zone	Basin name	TSS Load Index					TSS Likelihood Index: Seagrass (total area; surveyed + modelled)			TSS Likelihood Index: Coral Reefs			TSS Likelihood Index: Seagrass + Reefs		
		TSS anth. baseline (2012-2013)	TSS anth. load as % Marine Zone load	Basin ranking within Marine Zone	TSS Load Index within Marine Zone	Basin banking within Marine Zone	TSS Likelihood Score: Total seagrass	TSS Likelihood Index Seagrass (TSS Load Index x TSS Likelihood Score: Total seagrass)	RANK across GBR	TSS Likelihood Score: Reefs	TSS Likelihood Index Reefs (TSS Load Index x TSS Likelihood Score: Reefs)	RANK across GBR	TSS Likelihood Index: Seagrass + Reefs	TSS Likelihood Index: Total	RANK across GBR
Cape York North	Jacky Jacky Creek	43	0.29	3	0.80	3	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	Olive Pascoe River	54	0.36	2	1.00	2	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	Lockhart River	54	0.36	1	1.00	1	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	REGIONAL TOTAL and Max	**151**	*0.36*												
Cape York Central	Stewart River	41	0.21	2	0.27	2	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	Normanby River	151	0.79	1	0.27	1	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	REGIONAL TOTAL	**192**	*0.79*												
Cape York South	Jeannie River	31	0.53	1	1.00	1	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	Endeavour River	27	0.47	2	0.89	2	0.00	0.00	36	0.00	0.00	40	0.00	0.00	40
	REGIONAL TOTAL	**58**	*0.53*					0.00							
Wet Tropics	Daintree River	28	0.03	7	0.08	7	0.18	0.02	21	0.30	0.03	29	0.04	0.02	25
	Mossman River	6	0.01	8	0.02	8	0.18	0.00	33	0.30	0.01	39	0.01	0.00	39
	Barron River	32	0.03	6	0.10	6	0.18	0.02	20	0.30	0.03	25	0.05	0.02	22
	Mulgrave-Russell River	156	0.17	3	0.47	3	0.18	0.08	7	0.30	0.14	8	0.23	0.11	9
	Johnstone River	260	0.28	2	0.79	2	0.18	0.14	5	0.30	0.24	6	0.38	0.19	6
	Tully River	83	0.09	4	0.25	4	0.18	0.05	12	0.30	0.08	12	0.12	0.06	12

Marine zone	Basin name	TSS Load Index					TSS Likelihood Index: Seagrass (total area; surveyed + modelled)			TSS Likelihood Index: Coral Reefs			TSS Likelihood Index: Seagrass + Reefs		
		TSS anth. baseline (2012-2013)	TSS anth. load as % Marine Zone load	Basin ranking within Marine Zone	TSS Load Index within Marine Zone	Basin banking within Marine Zone	TSS Likelihood Score: Total seagrass	TSS Likelihood Index Seagrass (TSS Load Index x TSS Likelihood Score: Total seagrass)	RANK across GBR	TSS Likelihood Score: Reefs	TSS Likelihood Index Reefs (TSS Load Index x TSS Likelihood Score: Reefs)	RANK across GBR	TSS Likelihood Index: Seagrass + Reefs	TSS Likelihood Index: Total	RANK across GBR
	Murray River	39	0.04	5	0.12	5	0.18	0.02	18	0.30	0.04	19	0.06	0.03	20
	Herbert River	331	0.35	1	1.00	1	0.18	0.18	3	0.30	0.30	5	0.48	0.24	4
	REGIONAL TOTAL	**936**	*0.35*												
Burdekin	Tully River	83	0.02	5	0.03	5	1.00	0.03	14	1.00	0.03	23	0.06	0.03	19
	Murray River	39	0.01	7	0.01	7	1.00	0.01	22	1.00	0.01	33	0.03	0.01	33
	Herbert River	331	0.09	2	0.12	2	1.00	0.12	6	1.00	0.12	10	0.24	0.12	8
	Black River	34	0.01	8	0.01	8	1.00	0.01	25	1.00	0.01	35	0.02	0.01	34
	Ross River	49	0.01	6	0.02	6	1.00	0.02	19	1.00	0.02	30	0.04	0.02	29
	Haughton River	157	0.04	4	0.06	4	1.00	0.06	10	1.00	0.06	14	0.11	0.06	14
	Burdekin River	2,786	0.76	1	1.00	1	1.00	1.00	1	1.00	1.00	1	2.00	1.00	1
	Don River	183	0.05	3	0.07	3	1.00	0.07	9	1.00	0.07	13	0.13	0.07	11
	REGIONAL TOTAL	**3,663**	*0.76*												
Mackay Whitsunday	Proserpine River	75	0.13	4	0.31	4	0.07	0.02	17	0.04	0.01	34	0.03	0.02	31
	O'Connell River	241	0.41	1	1.00	1	0.07	0.07	8	0.04	0.04	18	0.11	0.06	15
	Pioneer River	173	0.29	2	0.72	2	0.07	0.05	11	0.04	0.03	24	0.08	0.04	16
	Plane	99	0.17	3	0.41	3	0.07	0.03	15	0.04	0.02	31	0.05	0.02	23
	REGIONAL TOTAL	**589**	*0.41*												
Fitzroy	Proserpine River	75	0.03	6	0.06	6	0.17	0.01	27	0.71	0.04	17	0.05	0.03	21
	O'Connell River	241	0.11	2	0.19	2	0.17	0.03	13	0.71	0.13	9	0.16	0.08	10
	Pioneer River	173	0.08	3	0.13	3	0.17	0.02	16	0.71	0.10	11	0.12	0.06	13
	Plane	99	0.05	4	0.08	4	0.17	0.01	23	0.71	0.05	15	0.07	0.03	17

Marine zone	Basin name	TSS Load Index					TSS Likelihood Index: Seagrass (total area; surveyed + modelled)			TSS Likelihood Index: Coral Reefs			TSS Likelihood Index: Seagrass + Reefs		
		TSS anth. baseline (2012-2013)	TSS anth. load as % Marine Zone load	Basin ranking within Marine Zone	TSS Load Index within Marine Zone	Basin banking within Marine Zone	TSS Likelihood Score: Total seagrass	TSS Likelihood Index Seagrass (TSS Load Index x TSS Likelihood Score: Total seagrass)	RANK across GBR	TSS Likelihood Score: Reefs	TSS Likelihood Index Reefs (TSS Load Index x TSS Likelihood Score: Reefs)	RANK across GBR	TSS Likelihood Index: Seagrass + Reefs	TSS Likelihood Index: Total	RANK across GBR
	Styx River	94	0.04	5	0.07	5	0.17	0.01	24	0.71	0.05	16	0.06	0.03	18
	Shoalwater Creek	59	0.03	7	0.05	7	0.17	0.01	30	0.71	0.03	20	0.04	0.02	24
	Waterpark Creek	57	0.03	8	0.04	8	0.17	0.01	31	0.71	0.03	21	0.04	0.02	27
	Fitzroy River	1292	0.60	1	1.00	1	0.17	0.17	**4**	0.71	0.71	**2**	0.88	0.44	**3**
	Calliope River	50	0.02	9	0.04	9	0.17	0.01	32	0.71	0.03	27	0.03	0.02	32
	Boyne River	16	0.01	10	0.01	10	0.17	0.00	35	0.71	0.01	37	0.01	0.01	38
	REGIONAL TOTAL	**2,157**	*0.60*												
Burnett Mary	Waterpark Creek	57	0.02	4	0.04	4	0.23	0.01	28	0.68	0.03	22	0.04	0.02	26
	Fitzroy River	1,292	0.50	1	1.00	1	0.23	0.23	**2**	0.68	0.68	**3**	0.91	0.46	**2**
	Calliope River	50	0.02	6	0.04	6	0.23	0.01	31	0.68	0.03	28	0.03	0.02	30
	Boyne River	16	0.01	9	0.01	9	0.23	0.00	38	0.68	0.01	38	0.01	0.01	37
	Baffle Creek	53	0.02	5	0.04	5	0.23	0.01	30	0.68	0.03	26	0.04	0.02	28
	Kolan River	30	0.01	7	0.02	7	0.23	0.01	35	0.68	0.02	32	0.02	0.01	35
	Burnett River	426	0.16	3	0.33	3	0.23	0.08	9	0.68	0.22	7	0.30	0.15	7
	Burrum River	17	0.01	8	0.01	8	0.23	0.00	37	0.68	0.01	36	0.01	0.01	36
	Mary River	666	0.26	2	0.52	2	0.23	0.12	7	0.68	0.35	4	0.47	0.23	5
	REGIONAL TOTAL	**2,607**	*0.50*									*Max.*	*2.00*		

6.2.3 Pesticides

The likelihood of pesticide exposure in the Great Barrier Reef World Heritage Area is determined based on the probability that the concentrations of pesticides (as a mixture) passing through the river mouth into the Great Barrier Reef lagoon exceed the concentrations that would be protective of 99% of species. Unlike the methods used for nutrient and sediment to determine the likelihood of exposure to coral and seagrass, this method takes a simpler approach to ensure that all ecosystems in the Great Barrier Reef World Heritage Area have the same level of protection. It is envisioned that in the future, eReefs hydrodynamic models will also be used to map the risk of pesticides to coral and seagrass. At the time of publishing this report, the methods for modelling pesticide exposure were still being developed. A case study is provided in Appendix 4 to present a first look at how the eReefs hydrodynamic model can be used to map pesticide exposure in the Marine Zone.

The probability that the concentration of pesticides (in a mixture) exceeds the concentrations protective of 99% of species for a selected day is determined for the wet season period only. The wet season period defined for these calculations is a 182-day (6 month) period, independent of the period of time between the first and last run-off events of the season. The 182-day period is measured from 1 November to 30 April, unless the first run-off event with elevated pesticide concentrations occurs earlier, in which case the 182-day period starts from the start of that event. The reason for this is to maintain a consistent time period to assess the likelihood of exposure between years and catchments. As previously mentioned, the time period from the first to the last event in a wet season varies (i.e. when pesticides are transported down catchments and exposure is most likely) between catchments and year to year. Therefore, the period of time of exposure to downstream ecosystems varies. When considering the risk of pesticides to organisms, the level of impact is dependent on both the magnitude and period of exposure (US EPA, 1998). A long exposure period often has a greater impact on an organism as it reduces the capacity for the organism to recover (Reinert et al., 2002). Hence, the proportion of time in which organisms are exposed to elevated pesticide concentrations relative to the proportion of time in which recovery can occur is important to include in the risk calculations. The method used to assess the effect based on the magnitude of exposure (i.e. concentration level) is discussed later.

6.3 Assessing the likelihood of exposure to pollutant pressures—floodplain wetlands and floodplain ecosystems

Wetlands are highly susceptible to the delivery of pollutants as they receive water directly flushed through agricultural lands. The likelihood of degraded water flushing into floodplain wetlands is closely linked to their hydrology and connectivity and, thus, to the climate and any factors that affect the water cycle.

In Australia's wet-dry tropics, wetland health is mainly driven by the cycle of wetting and drying (Douglas et al., 2011). The rhythm of the flooding—predictability of timing and magnitude of flood—predicts species richness and productivity of these wetlands in Australian tropical systems (Jardine et al., 2015). Similarly, within the Great Barrier Reef, exposure of floodplain wetlands within the Great Barrier Reef catchment to degraded water quality is closely linked to the timing and magnitude of flooding events (Davis et al., 2017). For example, floodplain wetlands from the Tully River catchment are well flushed, with relatively frequent floods of high magnitude. Thus, pollutants in these wetlands are diluted, maintaining better water quality compared to catchments like the Herbert and Burdekin, which have arrhythmic and infrequent floods (Kennard et al., 2010; Pearson et al., 2013). Temporally, greatest exposure to degraded water quality is generally during the first flooding events when pollutants are highly concentrated, during peak flow when the largest loads of pollutants are transported and during long dry seasons when the input of pollutants is continuous and prolonged (Davis et al., 2017).

Future climate-related changes in the frequency, timing and intensity of rainfall will have consequences for flooding rhythm, and thus for the impacts of degraded water quality on the floodplain wetlands and floodplains of the Great Barrier Reef catchment (GBRMPA, 2016; Davies et al., 2016).

In areas with intensive agriculture or urban development, natural flow, flooding and inundation patterns can be significantly altered influencing landscape and water quality processes. For example, within the Lower Burdekin, many floodplain wetlands that once dried out during the dry season are now permanently filled due to irrigation tail-water drainage (NQ Dry Tropics, 2016; Perna et al., 2012). In some agricultural areas of the Wet Tropics, drainage can be directed into wetlands influencing the water regime and wetland water quality (Terrain NRM, 2014).

It is a two-step process to determine the likelihood of exposure to hazard from pollutant pressures for floodplains and floodplain wetlands within a management area relative to exposure across the whole Great Barrier Reef catchment:

1) Determine the extent of hazard for each management unit:

    a. Calculate the area of High and/or Very High hazard for individual pollutant pressures.

    b. Calculate the area categorised as High and/or Very High hazard *that also contains wetlands*.

    c. Provide a spatial analysis of hazard and non-hazard areas (hectares and percentage area) for each management unit.

    The results of these calculations are provided in Section 5.

2) Calculate likelihood of exposure of wetlands to hazard from pollutant pressures:

    The likelihood of exposure of floodplain wetlands and floodplains to elevated nutrients, sediment and pesticides is assessed using the area of floodplain wetland (or floodplain) within a management unit, that is, exposed to a defined pollutant hazard (elevated nutrients, sediment and pesticides). The method for calculating that area as a percentage of the total Great Barrier Reef floodplain wetland area exposed to the hazard is outlined in Table 8.

Table 8. Calculation of the likelihood of exposure of floodplain wetlands to pollutants based on land-use hazard and area of floodplain wetlands.

A	Hectares of floodplain wetland within management unit exposed to hazard from the defined pollutant pressure (Very High and/or High hazard)
B	Hectares of floodplain wetland within the Great Barrier Reef catchment exposed to hazard from the defined pollutant
A/B*100	Per cent of wetland area within management unit exposed to High and Very High hazard from the defined pollutant relative to the exposed Great Barrier Reef wetland area

6.3.1 Floodplain wetlands

Nutrients

Approximately 17% of the total floodplain wetland area within the Great Barrier Reef catchment falls within areas categorised as Very High and High hazard for nutrients.

As identified in Section 5.1, hazard areas for nutrient pressures to floodplain wetlands are relatively widespread across the Great Barrier Reef catchment. Floodplain wetlands in 33 of the 47 management units (Figure 6) have a likelihood of exposure to nutrient pressures. Approximately 29,000 ha of floodplain wetlands lie within the High and Very High hazard areas for nutrient pressures (see detailed figures in Appendix 2).

The most extensive areas of floodplain wetlands in the nutrient hazard areas are in the Dawson and Lower Fitzroy catchments of the Fitzroy Basin (Figure 16). Together they represent an estimated 33% of the

floodplain wetland area in the Great Barrier Reef catchment lying within areas categorised as Very High and High hazard for nutrient pressures. Areas of intensive land use most strongly associated with driving nutrient pressures are restricted to the coastal lowland areas of the Lower Burdekin and Burrum basins. Concentrated areas of wetlands in the Lower Burdekin and Herbert also lie within hazard areas for nutrient pressures. See Appendix 2 for details of extent of exposure and management unit percentages.

As shown in the inset of Figure 5, floodplain wetlands such as Goorganga Plains wetlands, which lie downstream of nutrient hazard areas in Proserpine Basin, are likely to be exposed to nutrient inputs exported from source areas further up the system. These downstream wetlands are not included in this exposure assessment because they are not within categorised hazard areas.

Floodplain wetlands receive large quantities of nutrients—continuously through groundwater transport and in pulses during floods. In the Great Barrier Reef catchment, nutrients can affect metabolism and growth of organisms, shifts in species composition and changes in ecosystem function (Brodie and Mitchell, 2005; Pearson et al., 2015). While the highest nutrient loads are delivered during the wet season, probably the largest detrimental impacts of degraded water quality will typically be seen in the dry season. During the dry season, water quality deteriorates because of increased algal blooms, hypoxia, and ammonia concentrations, all of which can cause fish kills (Townsend and Edwards, 2003). For example, in the Burdekin River, algal blooms and poor water quality are common during the dry season in periods of low flows (Congdon and Lukacs, 1996). In shallow wetlands where cattle or feral animals frequently visit, the degradation of the water quality during the dry season is more pronounced compared with deeper ponds or those without cattle (Pettit et al., 2012). Overall, the effects of nutrients are likely to be the most detrimental during long dry spells in freshwater wetlands that are shallow, disconnected and are visited by cattle. Instant effects will also be seen during pre-flush pulses, where floodwater with high concentration of pollutants and high oxygen demand can cause fish kills.

Sediments

Almost 8% of the total floodplain wetland area within the Great Barrier Reef catchment is within areas of Very High hazard for sediment pressures.

Approximately 12,900 ha of floodplain wetlands are located within approximately 6,664,000 ha of the Very High hazard for sediment pressures. Wetlands in 26 of the 47 catchment management units have a likelihood of exposure to sediment pressures. This exposure to sediment hazard is mainly from grazing land use and some intensive cropping areas. The most extensive areas of wetlands lying within areas of Very High sediment hazard for sediment pressures are located in Fitzroy Basin, particularly the Dawson, Isaac and Mackenzie, representing 54% of the floodplain wetland area exposed to Very High sediment hazard from sediment pressures. The Lower Burdekin, Herbert and Burnett also have relatively large areas of wetland where there is a likelihood of exposure to elevated sediment inputs. The relative percentage area of Great Barrier Reef floodplain wetland area exposed to Very High hazard areas for sediment by management unit are summarised in Figure 17.

Although in some management units the extent of floodplain wetlands exposed to sediment pressures may be small, for those basins the proportion of wetland area exposed can be considerable. For example, 58% (55 ha) of the 95 ha of freshwater floodplain wetland area in the Barron Basin is exposed to sediment hazard. The details of the extent of exposure and management unit percentages are summarised in Appendix 2.

Areas of hazard for sediment pressures that do not have wetlands can expose floodplain wetlands lower in the system to sediment risk. For example, the Figure 7 inset shows a part of the floodplain and wetland system in the Cape Campaspe management unit lying immediately downstream of a significant sediment hazard area which itself does not contain wetlands. The complexity of river, floodplain and wetland systems currently precludes any attempt to quantify the exposure of these wetlands or impacts on them.

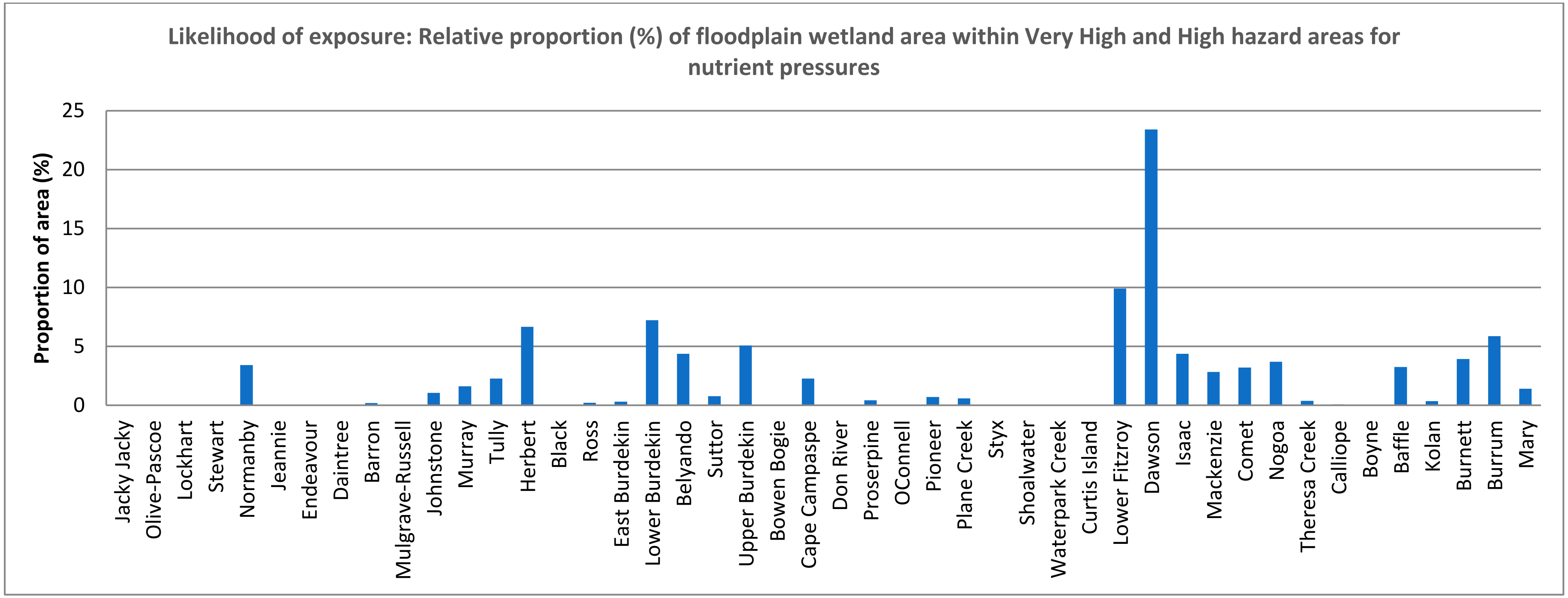


Figure 16. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain wetland area (per management unit) within areas categorised as Very High and High hazard for nutrient pressures. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain wetland area exposed to Very High and/or High hazard areas for nutrient pressures are shown.

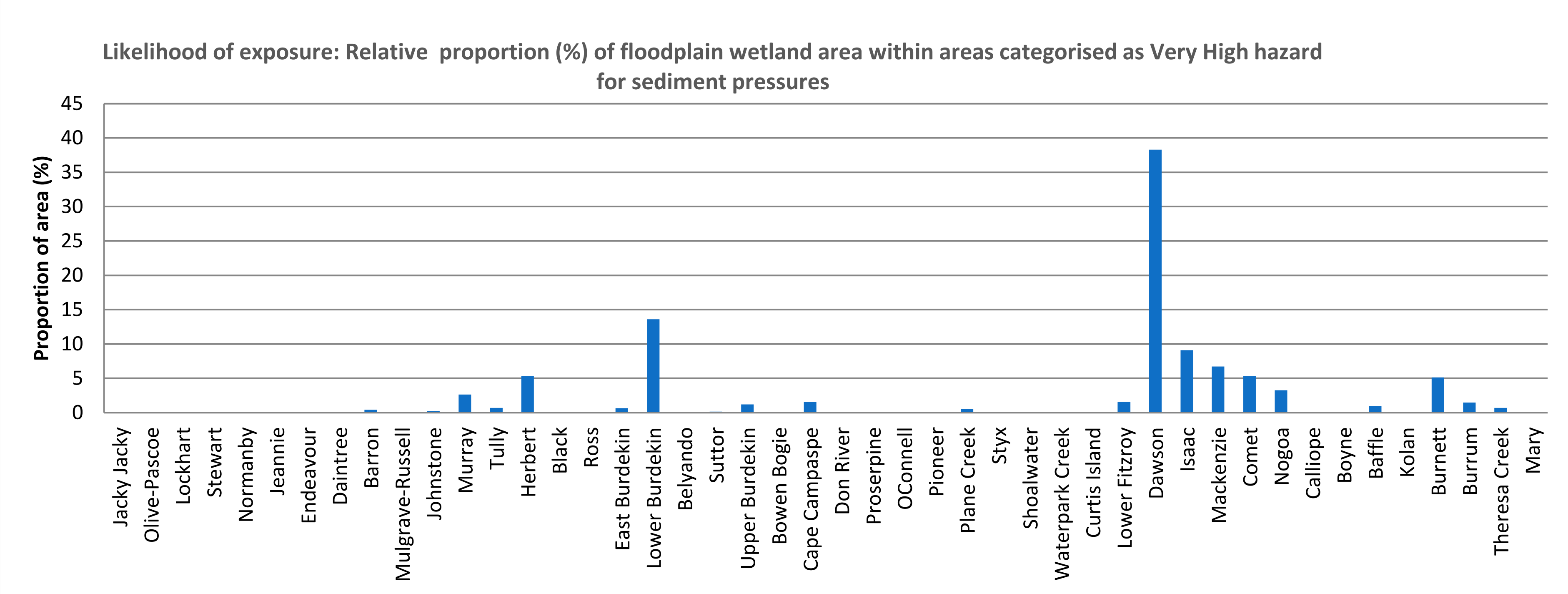


Figure 17. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain wetland area (per management unit) within areas categorised as Very High hazard for sediment pressure. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain wetland area within Very High sediment hazard areas are shown**.**

Floodplain wetlands are subject to increased sediment deposition after peak floods, where most sediments are mobilised from the land and transported through wetlands (Bartley et al., 2010; Davis et al., 2017). As with the Great Barrier Reef marine ecosystems, the most detrimental sediments to wetlands and floodplains are those with a smaller particle size (<10 μm to <63 μm), which are enriched in potentially bioavailable nutrients (Lewis et al., 2015; Garzon-Garcia et al., 2016). Floodplain wetlands and floodplains are areas of the landscape that are prone to retaining these finer sediment fractions (Prosser et al., 2001; Zierholz et al., 2001).

Pesticides

Almost 2% of the total floodplain wetlands area within the Great Barrier Reef catchment is within areas of High and Very High hazard from pesticide pressures. As identified in Section 5.3, approximately 2,800 ha of floodplain wetlands lie within approximately of 439,600 ha of High and Very High hazard for pesticides within the Great Barrier Reef catchments. See Appendix 2 for details.

Wetlands in 13 of the 47 catchment management units recorded a likelihood of exposure to pesticide pressures. These areas are typically situated on nearshore coastal plains where there has been a significant and continuing loss of natural wetland extent and a decline in condition compromising capacity for water quality threat abatement (NQ Dry Tropics, 2016; Folkers et al., 2014; Terrain NRM, 2015; Australian and Queensland governments, 2016). Pesticides have been shown to occur in waterways and in wetland soils and waters in these areas (Devlin et al., 2015b). Exposure of floodplain wetlands to pesticide inputs is primarily associated with intensive land use including cropping, horticulture and intensive animal production.

The likelihood of floodplain wetlands receiving water of degraded quality is especially high during the first flush at the end of the dry season (especially after a long dry spell), when concentrated pesticides are locally transported to nearby wetlands through run-off and groundwater (Davis et al., 2017; Devlin et al., 2015b). In irrigation areas exposure can be throughout the dry season as irrigation tailwater is drained into local wetlands (NQ Dry Tropics, 2016).

The Lower Burdekin has the greatest area (~1,200 ha) of floodplain wetlands within Very High or High pesticide hazard areas comprising 43% of the floodplain wetlands in the area, followed by the Herbert (640 ha or 23% of the basin's floodplain wetlands) (see Figure 18). The Burnett, Tully and Burrum basins also have a relatively high proportion of floodplain wetlands in these hazard areas, with a combined area of 618 ha of floodplain wetlands but all with less than 10% of the floodplain wetlands in each basin.

Floodplain wetlands downstream of hazard areas may also be receiving areas for pesticides exported from further up in the system. For example, the Proserpine Basin has extensive areas of floodplain wetland that lie downstream of intensive cropping, primarily sugarcane land use, as shown in the inset of Figure 9. Thus, the extent of wetlands influenced by the pesticide hazard area is likely to be greater than just those wetlands lying within the hazard areas.

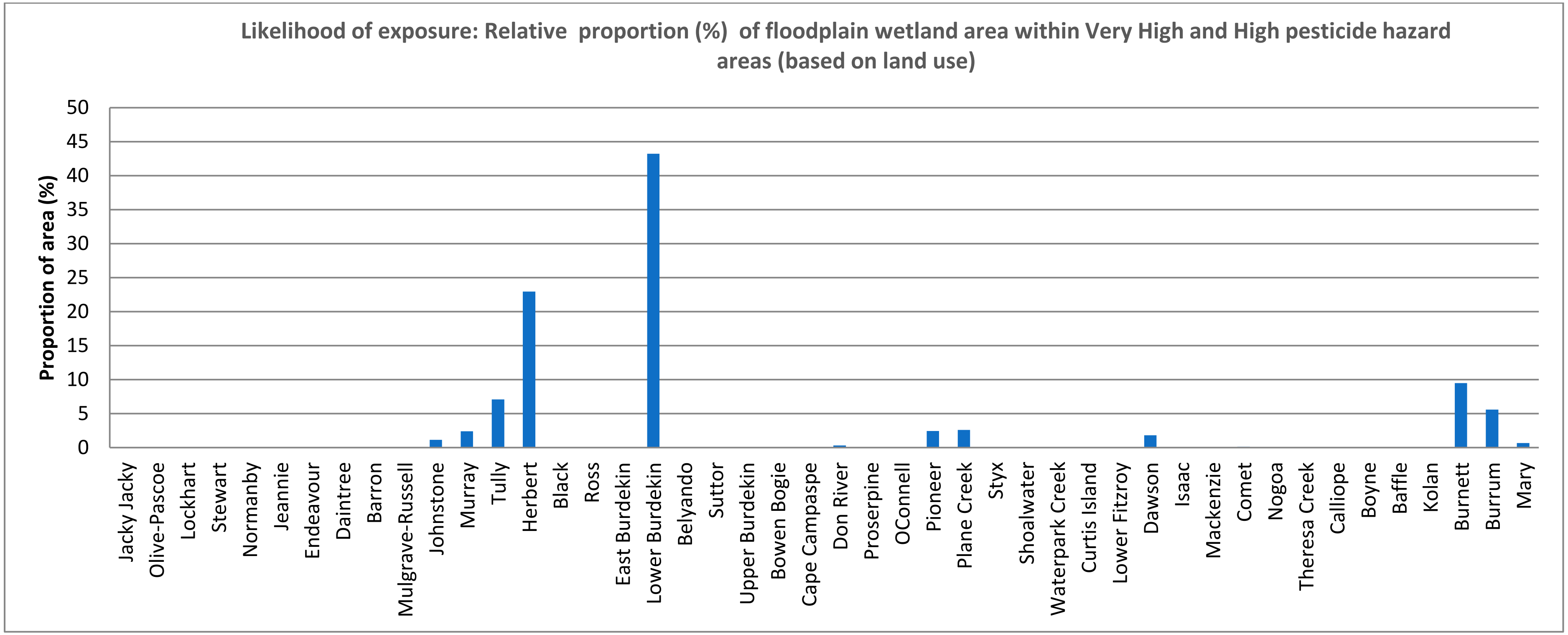


Figure 18. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain wetland area (as per management unit) within Very High and High pesticide hazard areas. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain wetland area within Very High and High pesticide hazard areas are shown.

6.3.2 Floodplain ecosystems

The Department of Science, Information Technology and Innovation (DSITI, 2015) provided a characterisation of pressures that are driven by land uses identified as potential sources of risk (hazards) for near-natural floodplain and non-floodplain wetlands. While this conceptual characterisation was conducted to be specific to wetland environments it was noted that those pressures are also broadly applicable to other aquatic ecosystems including floodplains. Floodplains are areas that are intermittently inundated by lateral overflow of riverine systems and the floodout areas of palustrine and lacustrine systems (Aquatic Ecosystems Task Group, 2012; Tockner et al., 2008). They can occur along rivers throughout a catchment.

Floodplains can act to ameliorate water quality pressures by accumulating materials such as sediment and nutrients (Noe and Hupp, 2009). On the other hand, depending on land uses present, they can accrue and combine multiple pressures including changes in water regimes and water quality processes at the local, regional and catchment scale (Noe and Hupp, 2009), thus also being a potential source of pollutants.

Almost 30% of the total floodplain area within the Great Barrier Reef catchment is exposed to areas of Very High and High hazard from nutrient pressures. Of the total floodplain area within the Great Barrier Reef catchment, 17% is exposed to areas of Very High hazard for sediment pressures. Nearly 3% of the total floodplain area within the Great Barrier Reef catchment is exposed to areas of Very High and High pesticide hazard.

Thirty-four of the 47 catchment management units recorded a likelihood of floodplain exposure to nutrient hazard. Thirty-two of the 47 catchment management units recorded a likelihood of floodplain exposure to sediment hazard. Nineteen of the 47 catchment management units recorded a likelihood of floodplain exposure to pesticide hazard.

Figure 19, Figure 20 and Figure 21 summarise the relative extent of floodplain lying within hazard areas for nutrient, sediment and pesticides.

The Dawson has the greatest relative area of floodplain (16%) lying within areas of hazard for nutrient pressure; this is followed by the Belyando (12%). These are followed by additional management units in the Burdekin and Fitzroy basins and the Burnett Basin. A similar distribution applies to exposure of floodplains to sediment hazard, with the Dawson having 25% of the Great Barrier Reef floodplain area that is exposed to Very High hazard for sediment pressures. Floodplains within pesticide hazard areas are more concentrated; 33% of the total Great Barrier Reef floodplain area that is exposed to pesticide hazard is within the Lower Burdekin management unit. A further 40% of the floodplain area exposed to pesticide hazard is within the Plane, Herbert and Pioneer management units. Detailed figures are provided in Appendix 2.

Exposure to pesticide pressures is influenced by the distinctive character of the flooding regimes of most Great Barrier Reef rivers, typically marked by hydrological seasonality with major flows occurring over only a few months of the year during the wet season. There is high variability between years and in the degree of flow cessation, or intermittency, over the dry season (Warfe et al., 2011). Floodplain inundation and flooding patterns are also influenced by land uses and regional and local hydrological modification (Waterhouse et al., 2015c). Since floodplains are pulsed systems with distinct and diverse flow, sediment, resource and thermal pulses, human modifications that truncate or amplify these pulses will have cascading effects on river–floodplain interactions (Junk et al., 1989; Tockner et al., 2000) including exposure to, and transport of, materials.

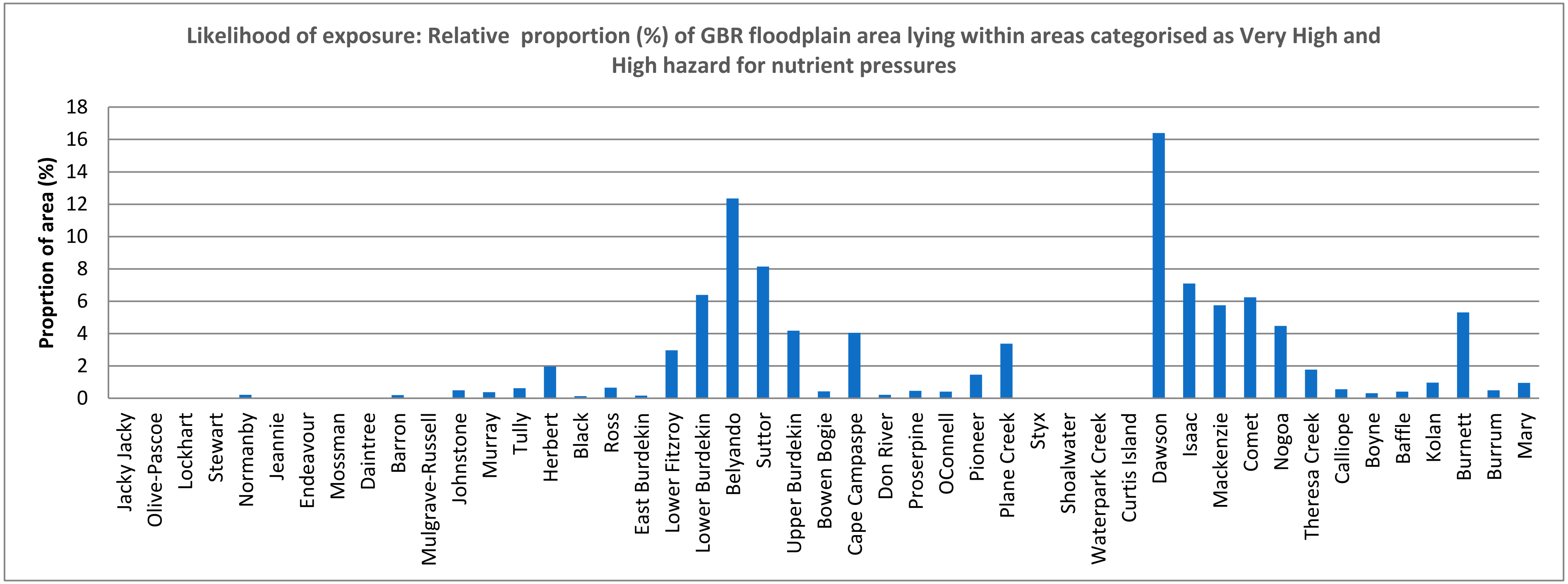


Figure 19. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain area lying within areas categorised as Very High and High hazard for nutrient pressure. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain area within Very High and High nutrient hazard areas are shown.

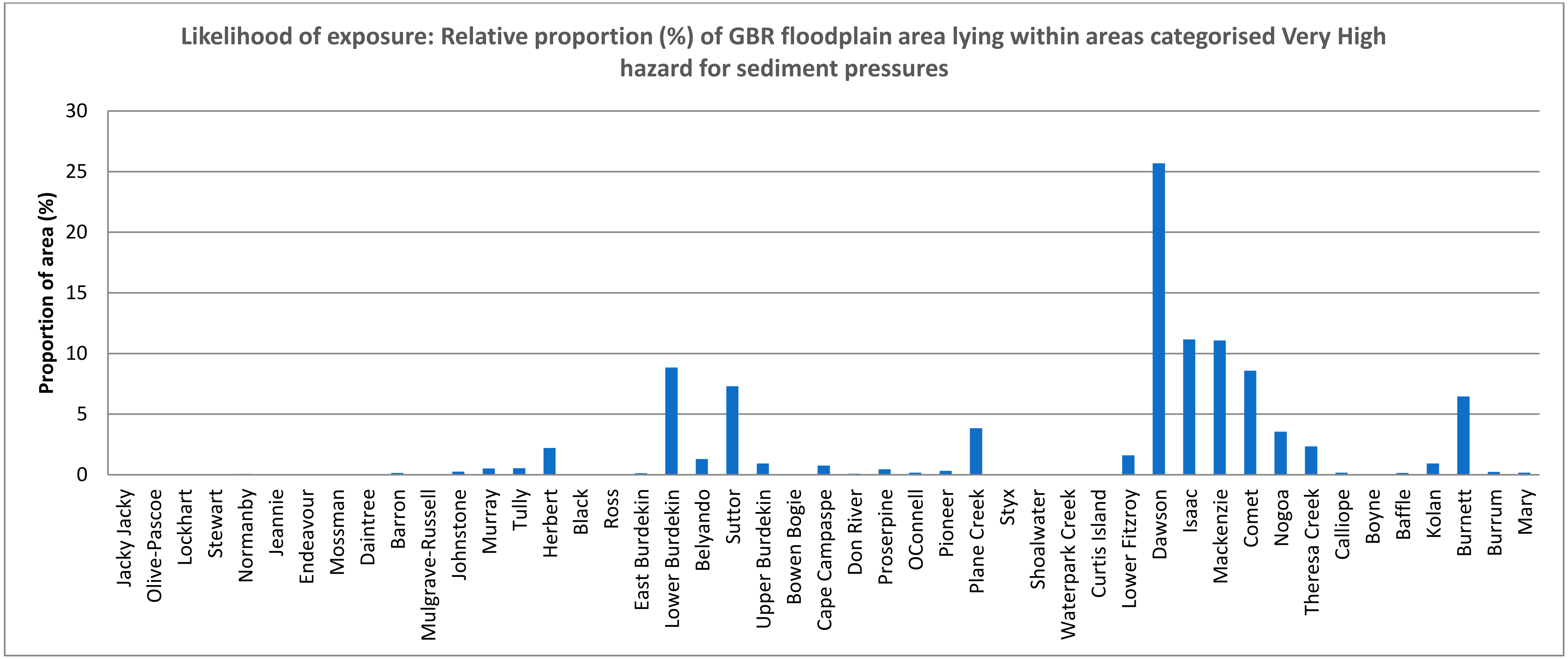


Figure 20. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain area lying within areas categorised as Very High hazard for sediment. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain area exposed to Very High and High sediment hazard areas are shown.

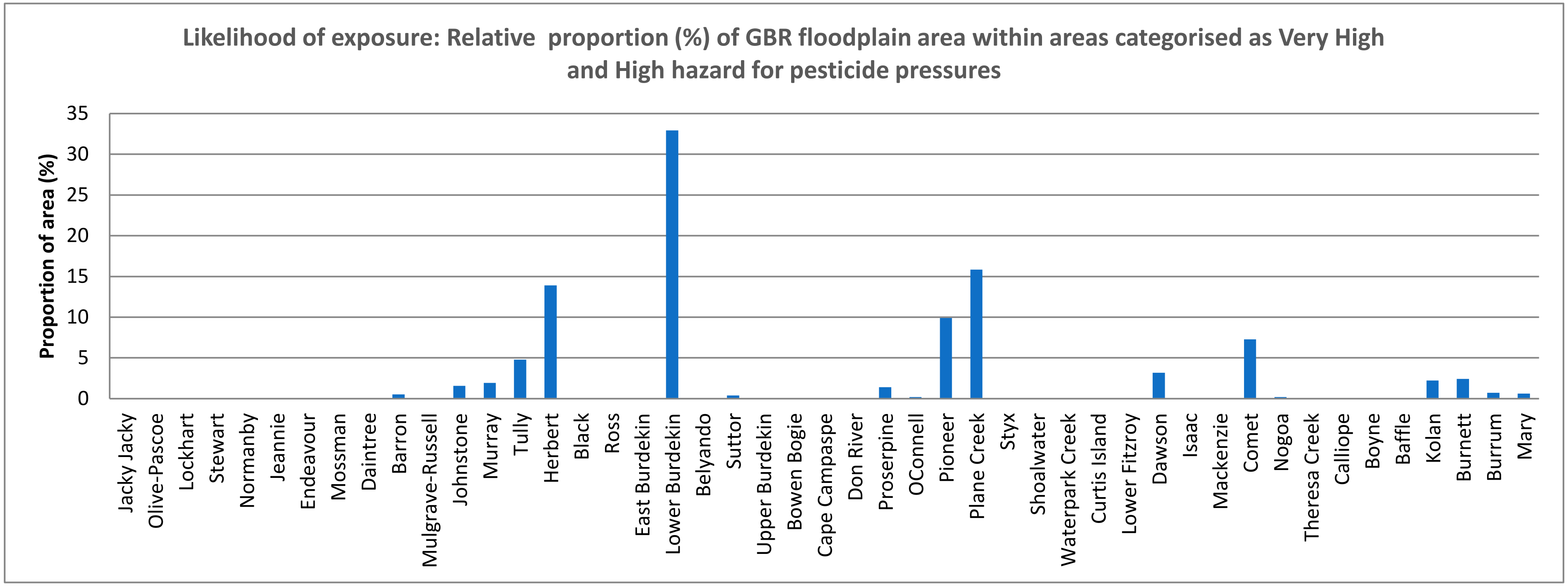


Figure 21. Likelihood of exposure: Relative proportion (%) of Great Barrier Reef floodplain area lying within areas categorised as Very High and High hazard for pesticide pressures. Only those management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain area within Very High and High pesticide hazard areas are shown.

7. What are the consequences of the water quality exposure?

The evidence of the consequences of degraded water quality on Great Barrier Reef coastal and marine ecosystems is reviewed in Chapter 1, and is therefore not covered in detail here. Rather, this section summarises the direct and secondary consequences of degraded water quality on coastal and marine ecosystems to inform selection of the most relevant parameters for the risk assessment.

7.1 Supporting evidence

7.1.1 Nutrients

Reefs in the Great Barrier Reef are naturally exposed to episodic river run-off carrying elevated nutrient and suspended sediment loads. There is little evidence that elevated nutrient concentrations *per se* increase coral mortality; however, enhanced nutrient availability within the ecosystem can have deleterious indirect effects on reef communities (e.g. De'ath and Fabricius, 2010; also covered in 2013 Scientific Consensus Statement, see Schaffelke et al., 2013).

There is strong evidence for several effects of nutrients on Great Barrier Reef ecosystems, including increased outbreaks of Crown-of-Thorns starfish, macroalgae abundance correlated with lower coral diversity, increased coral bleaching susceptibility, increased bioerosion (coral), increases in some coral diseases, reduced benthic light due to algal blooms and increased macroalgae and epiphytes on seagrass. While most effects occur in the wet season during river discharge (plumes) conditions, some effects have consequences beyond the wet season and continue for many years, for example Crown-of-Thorns starfish outbreaks (e.g. Fabricius et al., 2010). There is some evidence of the relative severity of these effects on Great Barrier Reef ecosystems, particularly in relation to the extensive effects of Crown-of-Thorns starfish (see Chapter 1, De'ath et al., 2012), but most influences are spatially specific and difficult to quantify. For example, macroalgal effects are dominant in inner shelf areas, and bleaching susceptibility may vary depending on the extent of influence of nutrients to cross shelf areas (driven by distance to the coast). Quantification of the severity of these effects on different ecosystems is a limitation to the current assessment.

The effects of increased nutrient loads to Great Barrier Reef coastal aquatic and marine ecosystems are discussed in more detail in Chapter 1 and summarised below.

1. Crown-of-Thorns starfish (derived from Chapter 1)

Crown-of-Thorns starfish are one of the major causes of coral mortality for the Great Barrier Reef (De'ath et al., 2012). During the wet season, river nutrients can influence Crown-of-Thorns starfish outbreak dynamics (Chapter 1) when large discharges (approximately more than 10 km^3) occur during the early wet season (November–February) in the region between Hinchinbrook and Lizard Islands from the Wet Tropics and the Burdekin Rivers, while phytoplankton-feeding Crown-of-Thorns starfish larvae are present in the water column (Brodie et al., 2005; Brodie et al., 2017; Fabricius et al., 2010). The Crown-of-Thorns starfish spawning period is mainly from November to February (Babcock et al., 2016). Further development of a Crown-of-Thorns starfish outbreak, however, depends on there being sufficient live coral cover to sustain adult populations and, for the initiation of a wave of outbreaks, reinforcing hydrodynamic conditions in the area between Cairns and Lizard Island (Hock et al., 2014; Wooldridge and Brodie, 2015). After waves are initiated in the Cairns – Lizard Island area (as they were in 1962, 1978, 1993 and 2009) outbreaks progress down the Great Barrier Reef (mainly on mid-shelf reefs) over a period of about 12 years to the areas approximately offshore from Mackay. It is generally assumed that numbers of outbreaks have increased in the period since pre-development in Queensland and that the frequency of Crown-of-Thorns starfish waves (about every 15 years over the last 60 years) has increased greatly (possibly from frequencies of 1 in 50–80 years) (Fabricius et al., 2010).

Hydrodynamic modelling has been applied to rank the influence of individual rivers with discharges that affect the Cairns – Lizard Island region (Brinkman et al., 2014). The analysis indicates those rivers between (and including) the Burdekin and the Daintree have some degree of influence. Riverine inputs were ranked using both the magnitude and duration of exposure and show that discharges from the Daintree, Russell-Mulgrave, Tully and Barron Rivers dominate the northern Cairns – Lizard Island region, with the Burdekin River also contributing significantly to the southern part of this region. When the contribution of DIN was considered, the Russell-Mulgrave, Tully, Johnstone, Burdekin and Haughton basins together contributed more than 80% to the total region.

Following the initiation of a primary outbreak, massive larval production (Uthicke et al., 2015) leads to secondary outbreaks to the north and south of the initiation area. A wave of secondary outbreaks occurs to the south of the initiation area from Cairns towards Townsville over approximately 8–9 years after the primary outbreak and then offshore from Mackay (12 years), after which the outbreaks appear to diminish (Pratchett et al., 2017). There is evidence that nutrient enrichment from rivers can be enhancing secondary outbreaks, especially in the offshore Wet Tropics region, and further south to the mid-shelf areas off Townsville (Brodie et al., 2017). Crown-of-Thorns starfish larvae are in the plankton stages during November to February and this is also when the Wet Tropics rivers are in a high discharge period every year and regularly produce phytoplankton blooms on the Great Barrier Reef shelf (Devlin et al., 2012a; Devlin et al., 2012b; Álvarez-Romero et al., 2013; Blondeau-Patissier et al., 2014; Furnas et al., 2005). Measurements of high chlorophyll concentrations during this period represent an extended period of time in which conditions of high phytoplankton biomass are optimal, thereby providing an adequate food source for Crown-of-Thorns starfish larvae (see also Devlin et al., 2015a; Wolfe et al., 2015). Therefore, wet season nutrient inputs in the extended Crown-of-Thorns starfish influence area are considered to be important in assessing the risk of nutrients to the Great Barrier Reef.

2. Macroalgae versus coral diversity (from Chapter 1)

As a direct effect, excess nutrient availability, especially of nitrogen, can promote the growth of fleshy macroalgae at locations with sufficient light (Schaffelke et al., 2005). Macroalgae are more abundant on reefs with high concentrations of water column chlorophyll, which is responsive to nutrient availability (De'ath and Fabricius, 2010). High macroalgal biomass on reefs has a number of adverse effects on corals: space competition (e.g. McCook et al., 2001), affecting coral metabolism by altering the corals' microenvironment (Hauri et al., 2010), reducing coral settlement (Birrell et al., 2008) and increasing the susceptibility to coral disease (Morrow et al., 2012).

3. Increased coral bleaching susceptibility (see also Chapter 1)

The availability of excess DIN alone has few direct adverse effects on corals (Fabricius, 2011); however, indirect effects through the coral–zooxanthellae relationship have been shown to be important. A number of environmental factors influence thermal bleaching resistance of corals. Elevated nutrient concentrations have been identified as one of the most important (aside from sea surface temperature) (Carilli et al., 2009; Carilli et al., 2010; Wagner et al., 2010; Wooldridge, 2009; Wooldridge and Done, 2009). DIN availability is important in the functioning of the coral–algae symbiosis, and elevated DIN concentrations can cause changes that disrupt the ability of the coral host to maintain an optimal population of algal symbionts (Wooldridge, 2016). Together with increased temperature, elevated DIN concentrations and changes in nitrogen:phosphorus ratios can increase the susceptibility of corals to coral bleaching (Wooldridge, 2016; Wooldridge et al., 2017; Vega Thurber et al., 2013; Wiedenmann et al., 2013; D'Angelo and Wiedenmann, 2014).

Fabricius et al. (2013b) propose a conceptual framework that synthesises the apparently inconsistent result of recent studies that suggest either greater or reduced thermal tolerance in response to changes in nutrient status. The framework illustrates two important points: (i) nutrients and light can be either a stress or a

beneficial factor, with optimum responses at species-specific tolerance levels and detrimental effects if rates are much higher or lower, (ii) shifts in the trophic status of the environment (from oligotrophic to eutrophic) do not easily translate into shifts in the trophic status of reef corals (from starved to well fed), because the food preferences and trophic plasticity vary greatly between species. The review concludes that in more eutrophic environments, as found on parts of the inner shelf Great Barrier Reef south of Cooktown, exposure to additional nutrients is predominantly a stress factor for most coral species, and that improvement of water quality would improve the tolerance of inner shelf corals to thermal stress. The responses of early life history stages of the common inshore coral *Acropora tenuis* to a combination of excess inorganic and organic nutrients and elevated temperatures indicate that recruitment and recovery potential of this species may be limited at Great Barrier Reef inshore reefs (Humanes et al., 2016).

4. Increased bioerosion (coral)

Bioerosion of corals (both live and dead) occurs via a large range of organisms but two of the main types are microborers, often algae and sponges, and macroborers, often worms and bivalves (Hutchings et al., 2005). The growth of both of these types of borers can be increased by nutrient enrichment, for example algal borers via increased growth due to increased dissolved inorganic nutrient availability, and filter feeding worms and bivalves through increased phytoplankton biomass. Increased bioerosion by these organisms can interact with reduced calcification due to ocean acidification to additively reduce reef net calcification (DeCarlo et al., 2015).

5. Increases in some coral diseases (from Chapter 1)

Coral disease is a significant cause of coral cover declines on the Great Barrier Reef (Osborne et al., 2011) and is predicted to worsen with global pressures of increasing temperature and ocean acidification (Maynard et al., 2015; O'Brien et al., 2016b). While coral disease is considered a general stress response of corals, it has been positively correlated to sedimentation and elevated concentrations of nutrients and organic matter (Harvell et al., 2007; Haapkylä et al., 2011; Vega Thurber et al., 2013; Thompson et al., 2014; Pollock et al., 2016).

6. Reduced benthic light due to algal blooms

Production of phytoplankton due to inputs of river-derived nutrients reduces water clarity and, hence, light availability for benthic plant communities, for example seagrass and coral (Collier et al., 2016a; Petus et al., 2014b). In inner shelf waters, the light reduction due to resuspended sediment is usually a more dominant effect but in deeper waters (>15 m) where resuspension does not normally occur, the light reduction due to phytoplankton may be an important factor, for example for deepwater seagrass communities (see below).

7. Macroalgae and epiphytes on seagrass

Epiphytes growing on the leaves of seagrasses and macroalgae growing in seagrass meadows increase their productivity and biomass turnover with increasing nutrient loads (Cebrian et al., 2013). When they increase to bloom proportions, epiphytes and macroalgae have been known to cause seagrass die-off at various locations throughout the world (e.g. Cambridge et al., 1986; Cabaço et al., 2013). Epiphytes are highly variable in the Great Barrier Reef (see Chapter 1) and temporarily reach bloom proportions, but there is not yet any documented evidence of lasting effects to seagrass condition, which may be due to several reasons, including complex biological interactions (e.g. grazing) and rapid seagrass growth (Cebrian et al., 2013; Unsworth et al., 2015). However, there is potential for epiphyte and macroalgal blooms from nutrient enrichment that warrants further investigation.

8. Nutrient inputs to coastal aquatic ecosystems (summarised from Chapter 1)

Wetlands receive large quantities of nutrients continuously through groundwater transport and in pulses during floods. In the Great Barrier Reef catchment, nutrients can affect metabolism and growth of organisms, shifts in species composition and changes in ecosystem function (Brodie and Mitchell, 2005; Pearson et al., 2015).

In the short term, excessive nutrients can cause increase in plant and algal productivity if suspended sediments are low. Excessive algal growth can affect invertebrate fauna by reducing light and suitable substrate (Connolly and Pearson, 2007) and by causing toxicity of the water if cyanobacteria blooms develop (Brodie and Mitchell, 2005). Increase in nutrients can also favour the growth of exotic weeds (Brodie and Mitchell, 2005) and floating weed mats (MacKinnon and Herbert, 1996). For animals, degraded water quality can cause rapid and fatal consequences. For example, in tropical floodplains, the first flood of the wet season can be rich in nutrients and carbon and have a high oxygen biological demand, causing strong hypoxia which can result in the sudden death of hundreds of fish (Townsend and Edwards, 2003). While the highest nutrient loads are delivered during the wet season, the largest detrimental impacts of degraded water quality will be seen in the dry season. During the dry season, water quality deteriorates because of increased algal bloom, hypoxia, and ammonia concentrations, all of which can cause fish kills (Townsend and Edwards, 2003). In shallow wetlands where cattle or feral animals frequently visit, the degradation of the water quality during the dry season is more pronounced compared to deeper ponds or those without cattle (Pettit et al., 2012).

7.1.2 Suspended sediment and turbidity

There is also strong evidence for several effects of sediments on Great Barrier Reef ecosystems, including light reduction for seagrass and coral, sedimentation on coral, sedimentation in turf algae and herbivore feeding and fine suspended sediment effects on coral reef fish. As for nutrients, the relative severity of these effects on Great Barrier Reef ecosystems is difficult to quantify and is a limitation for this assessment. However, the dominant influence of turbidity on reduced light for seagrass is well documented, as described below.

The effects of increased sediment loads to the Great Barrier Reef are discussed in more detail in Chapter 1 and summarised below.

1. Light reduction for seagrass

Light limitation is presently regarded as the primary driver of seagrass production (Collier and Waycott, 2009) in the Great Barrier Reef, and reductions in light availability have been directly linked to seagrass loss (Collier et al., 2012). Light penetration into coastal waters is strongly regulated by the resuspension of fine sediments, which may occur year-round (Fabricius et al., 2013a; Fabricius et al., 2014; Fabricius et al., 2016).

Seagrass meadows of the Great Barrier Reef undergo seasonal changes in growth, with the peak growing season typically during spring when benthic light availability is highest and there is a low risk of extreme water temperatures, resulting in a maximum abundance (i.e. distribution and density) in late spring and early summer (Chartrand et al., 2016; McKenzie, 1994; Rasheed et al., 2014). This is also a time for carbohydrates reserve formation (Collier et al., 2016b), and many species are flowering (McKenzie et al., 2016). Conversely, abundance declines in the wet season from early summer and reaches the lowest in winter. They also undergo decadal-scale changes in abundance caused by climatic variations in riverine discharge and water temperature and by periods of elevated disturbance from cyclones (e.g. Birch and Birch, 1984; Coles et al., 2011; McKenzie et al., 2016; Petus et al., 2014b; Rasheed and Unsworth, 2011).

The consequences of degraded water quality for seagrass meadow abundance will be affected by the timing relative to these cycles. The peak growing season is sensitive to degraded water quality because the increase in abundance, formation of reserves and sexual reproduction producing seed banks is critical to survival in

the following years. For example, the 2010-2011 wet season began earlier than average in October, and this appeared to hamper growing season recovery from previous wet season losses leading to the lowest abundances ever observed throughout the Great Barrier Reef (McKenzie et al., 2016; Rasheed et al., 2014). However, severely degraded water quality during the wet season can drive rapid seagrass loss, and place them in poor state for the following growing season. For example, substantial seagrass loss occurred in the Burdekin region between 2008 and 2009 when there were very large riverine flows in January and February and high frequency of exposure to turbid primary water (Petus et al., 2014b; McKenzie et al., 2016; Collier et al., 2012). The relative sensitivity and resilience has not yet been quantitatively assessed in relation to these cycles.

2. Sedimentation on corals (summarised from Chapter 1)

High concentrations of suspended sediment can cause direct biological effects (e.g. interfering with filter feeding), alter the light quantity and quality and smother the corals' surface with a fine layer of sediment (Jones et al., 2015). Most importantly, light reduction, elevated suspended sediments and sediment deposition negatively affect the reproductive cycle and early life histories of corals (Jones et al., 2015). Two of the newly recognised mechanisms for the negative effects of suspended sediments were the entanglement and entrapment of coral sperm by sediment particles (Ricardo et al., 2015) and ballasting of the buoyant egg-sperm bundles (Ricardo et al., 2016a), both reducing fertilisation success of corals. Conversely, developing embryos and larvae tolerated exposure to suspended sediments by having mechanisms to remove particles (Ricardo et al., 2016b). The following coral life history stage, successful larval settlement, is again reduced by a thin layer of fine, terrigenous settlement (Perez et al., 2014), supporting earlier research (Jones et al., 2015).

3. Sedimentation in turf algae and herbivore feeding

Increased benthic sediment loads, within the epilithic algal matrix found on coral reefs, can suppress fish herbivory and detritivory on coral reefs (Bellwood and Fulton, 2008; Goatley and Bellwood, 2013; Gordon et al., 2016). In doing so, sediments may drive a change in the state of the epilithic algal matrix from palatable, short, productive algal turfs to unpalatable, long, sediment-laden algal turfs. These changes may reduce the resilience of coral reefs to other stressors (Goatley et al., 2016). Finer sediments have greater effects on the suppression of herbivory (Tebbett et al., 2017a; Tebbett et al., 2017b).

4. Fine suspended sediment adverse effects on coral reef fish (summarised from Chapter 1)

Coral reef–associated damselfish respond to suspended sediment, and their larval development, foraging success and habitat use are adversely affected at concentrations that have been observed at Great Barrier Reef inshore reefs (Johansen and Jones, 2013; Wenger et al., 2012; Wenger and McCormick, 2013; Wenger et al., 2014).

5. Sediment inputs to coastal aquatic ecosystems (summarised from Chapter 1)

Excess sediment can put pressure on freshwater wetlands directly through increases in turbidity, but also indirectly, as sediments carry pollutants and nutrients attached to them (Neil et al., 2002; Bainbridge et al., 2014). In freshwater coastal wetlands, suspended sediments can reduce temperature, decrease light availability, lead to physical impacts on fish such as gill clogging (Ryan, 1991), impede fish foraging, reduce habitat spaces for some species (Ebner et al., 2016) and change the behaviour of macroinvertebrates (Wetzel, 2001; Connolly and Pearson, 2007; Wallace et al., 2015) and turtles (Schaffer et al., 2015).

In estuarine wetlands, excessive sedimentation can cause immediate mortality in mangroves and marshes by burying plants and seedlings and covering pneumatophores (Ellison, 1998; Saintilan and Williams, 1999; Lovelock and Ellison, 2007). Excessive sedimentation can also cause drastic changes in hydrology, which can significantly affect wetlands. For example, during tropical storms, sedimentation can cause the death of large areas of mangrove forests due to changes in soil elevation which can drastically change the hydrology within the forest (Ellison, 1998).

Increased sedimentation can impact plant communities through changes in germination rates and reduced seedling emergence, seedling establishment and species richness (Wang et al., 2013; Maheny et al., 2004). These changes can influence ecosystem function: its productivity, stability and sustainability (Maheny et al., 2004). A significant impact of excess sediments is the filling up of some hydrologically isolated wetlands and the subsequent loss of function.

The impacts of increased sediments in naturally clear systems might be greater compared to the impacts on naturally turbid catchments (e.g. Connolly and Pearson, 2007; Davis et al., 2017). In the long term, increased sedimentation can choke lagoons and waterways, which can result in the loss of connectivity and consequent deterioration of the wetland.

7.1.3 Pesticides

As is true of most stressors in an ecosystem, measuring the direct consequences of pesticide exposure *in situ* of Great Barrier Reef ecosystems is difficult because of the range of variables and stressors present that can alter community structure and ecosystem function (Schäfer et al., 2007). It is, however, possible to ascertain what consequences may have occurred in Great Barrier Reef ecosystems based on the mode of action of the pesticides, which organisms are likely to be exposed to the pesticides, and laboratory studies that identify the sensitivity of species relative to the environmental concentrations observed.

A range of pesticides with different modes of action have been detected in Great Barrier Reef ecosystems (freshwater, estuarine and marine) of the Great Barrier Reef (see Devlin et al., 2015b for a review). The variety of modes of action means that different organisms are impacted depending on which type of pesticide they are exposed to. Table 9 presents a list of the main modes of actions of the pesticides detected and the taxa at risk, based on species sensitivity distributions (King et al., 2017a; King et al., b). The majority of herbicides have a specific mode of action that directly impacts biochemical processes in phototrophic species (e.g. PSII inhibitors, acetolactate enzyme inhibitors). For these chemicals, non-phototrophic species are largely insensitive to their exposure at environmental concentrations but may be affected indirectly through loss of primary production services. In contrast, some of the insecticides that have been detected in Great Barrier Reef ecosystems (e.g. imidacloprid, fipronil) have a specific mode of action on biochemical pathways in arthropods, making species within this taxa the most sensitive to these chemicals. In addition, a few herbicides have a mode of action that can affect both plant and animal species (e.g. haloxyfop, metolachlor, chlorothalonil, pendimethalin).

Table 9. The main modes of action of pesticides detected in freshwater, estuarine and marine ecosystems of the Great Barrier Reef and the taxa at risk, based on species sensitivity distributions in King et al. (2017a; 2017b).

Pesticide type	Mode of action	Examples detected	Taxa most at risk
Herbicides	Photosystem II inhibitors	Ametryn, atrazine, diuron, hexazinone, tebuthiuron, metribuzin, simazine, prometryn, bromacil	Phototrophic species
	Synthetic auxins	2,4-D, triclopyr, fluroxypyr, MCPA,	Phototrophic species, particularly dicot species, e.g. macrophytes
	Acetolactate enzyme inhibitors (inhibits plant amino acid synthesis)	Imazapic, imazethapyr, metsulfuron-methyl	Phototrophic species
	Acetyl Coenzyme A (CoA) carboxylase (ACCase) inhibitor (inhibitor of fatty acid synthesis)	Haloxyfop	Phototrophic species

Pesticide type	Mode of action	Examples detected	Taxa most at risk
	Inhibition of cell division	Metolachlor	Phototrophic species
	Inhibition of carotenoid biosynthesis	Isoxaflutole	Phototrophic species
Insecticides	Acetylcholine receptor (nAChR) agonist	Imidacloprid	Arthropods (insects and crustaceans)
	γ-aminobutyric acid GABA-gated chloride channel antagonist	Fipronil	Arthropods (insects and crustaceans)
Fungicides	Inhibition of spore germination and zoospore motility	Chlorothalonil	Fungi
	Growth inhibitor through inhibition of ergosterol biosynthesis	Propiconazole	Fungi

The consequences of exposure to PSII herbicides on Great Barrier Reef marine species are understood the most and have been recently reviewed in Devlin et al. (2015b). Based on laboratory studies, PSII herbicides have been reported to affect corals (Jones and Kerswell, 2003; Negri et al., 2005), microalgae (Bengtson Nash et al., 2005), crustose coralline algae (Negri et al., 2011), foraminifera (van Dam et al., 2012) and seagrass (Haynes et al., 2000; Gao et al., 2011). Effects of chronic herbicide exposures in inshore Great Barrier Reef environments are unknown but are likely to affect coral reproduction (Cantin et al., 2009).

Less is known of the adverse effects of pesticides to the Great Barrier Reef freshwater, wetland and estuarine ecosystems, although the proximity of these ecosystems to pesticide sources would suggest that exposure to high levels is likely. Pesticide run-off was suggested as the cause of sublethal effects observed in barramundi (*Lates calcarifer*) from Great Barrier Reef freshwater ecosystems (Kroon et al., 2015b; Kroon et al., 2015c; Hook et al., 2017). In laboratory studies, atrazine affected cell health of benthic diatoms collected *in situ* of Great Barrier Reef catchments (Wood et al., 2014); PSII herbicides and imazapic affected population growth of three tropical freshwater microalgal species (Stone et al., 2016); chlorpyrifos impacted the eastern rainbowfish (*Melanotaenia splendida splendida)* (Humphrey and Klumpp, 2003); and environmentally relevant concentrations of atrazine produced sublethal effects in fish and amphibians (Rohr and McCoy, 2010). Impacts to estuarine organisms have also been observed. For example, PSII herbicides reduced photosynthesis and growth (Magnusson et al., 2008) and influenced species composition (Magnusson et al., 2012).

The results for individual pesticide impacts may underestimate real-world effects in Great Barrier Reef ecosystems as pesticides are most often detected in mixtures with other pesticides and/or other stressors (e.g. high turbidity and nutrients) (Smith et al., 2012; Davis et al., 2012; Davis et al., 2013; Davis et al., 2014). Pesticide mixtures produce a cumulative effect in organisms (Faust et al., 1994), and when chemical pollutants (including pesticides) are combined with other natural stressors the combined effects are often found to be synergistic (Holmstrup et al., 2010); such synergistic effects are known for pesticides (Beketov and Leiss, 2012).

7.2 Assessing the consequence of nutrient and sediment exposure to coral reefs and seagrass

The data inputs selected to represent the consequence of exposure to sediments, nutrients and pesticides to coastal and marine Great Barrier Reef ecosystems are described below. However, the ability to conduct quantitative analyses is limited; therefore, only two examples are presented here.

7.2.1 Nutrients

It is outside the scope of this assessment to analyse the consequence of nutrient exposure in the context of all of the factors outlined above, and for many factors there are limited quantified data to complete the assessment. However, due to the dominance of the impact of Crown-of-Thorns starfish on coral reefs explained in Section 7.1.1, an example of Crown-of-Thorns starfish influence is presented to illustrate a consequence factor to calculate nutrient (DIN) risk for coral reefs (see Figure 22). This factor is only relevant to areas north of Mackay Whitsunday region.

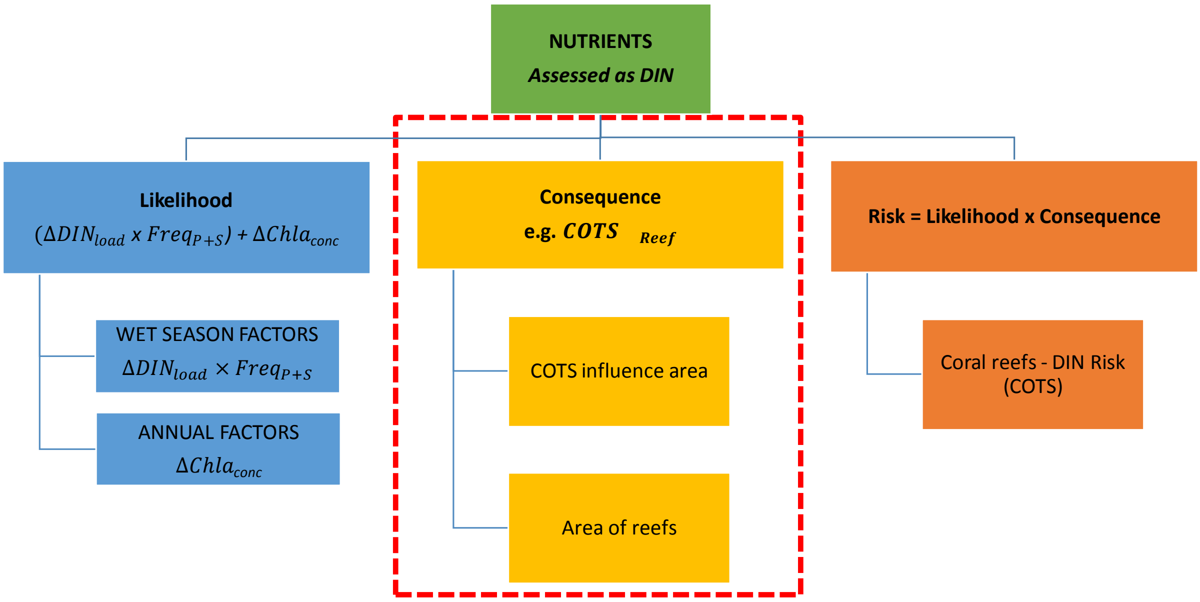


Figure 22. Framework for assessing the likelihood of exposure and consequence of DIN to coral reefs. The consequence example presented here for coral reefs highlights the importance of the Crown-of-Thorns starfish influence area for assessing risk to coral reefs.

Following the initiation of a primary Crown-of-Thorns starfish outbreak, massive larval production leads to secondary outbreaks to the north and south of the Crown-of-Thorns starfish initiation area (approximately Lizard Island to Cairns; Fabricius et al., 2010; Brodie et al., 2017). Wet Tropics river discharges influence this area every year, and the Burdekin River influences this area periodically in larger flow years. The Crown-of-Thorns starfish influence area represents the region in the Great Barrier Reef that provides continued nutrient inputs to sustain the survival of Crown-of-Thorns starfish larvae following the initiation of an outbreak (Brodie et al., 2017).

The Crown-of-Thorns starfish influence area is shown spatially in Figure 23. To apply a weighting of greater consequence to these areas, the pixels in the Crown-of-Thorns starfish influence area were allocated a score of 1.0, and those outside of the Crown-of-Thorns starfish influence area were allocated a score of 0.5. Note that these weightings are not justified by quantified evidence and were applied as a demonstration, derived by expert elicitation. Since the consequence layer is essentially applied as a binary weighting to the DIN Likelihood of Exposure (the product of the two layers), the influence is relative regardless of the actual weighting values selected. However, a more sophisticated approach would be required if management conclusions were to be drawn from this analysis.

The final output was classified into two final categories of High (inside the Crown-of-Thorns starfish influence area) and Low (outside the Crown-of-Thorns starfish influence area). The total areas and area of coral reefs, surveyed seagrass and modelled deepwater seagrass in each consequence category in each Marine Zone was calculated by overlaying the final DIN Consequence (CoTs) layer with the spatial layers for each habitat type.

The *DIN Consequence (CoTs) Score* was calculated for each Marine Zone by summing the area of coral reefs in the High consequence category and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score.

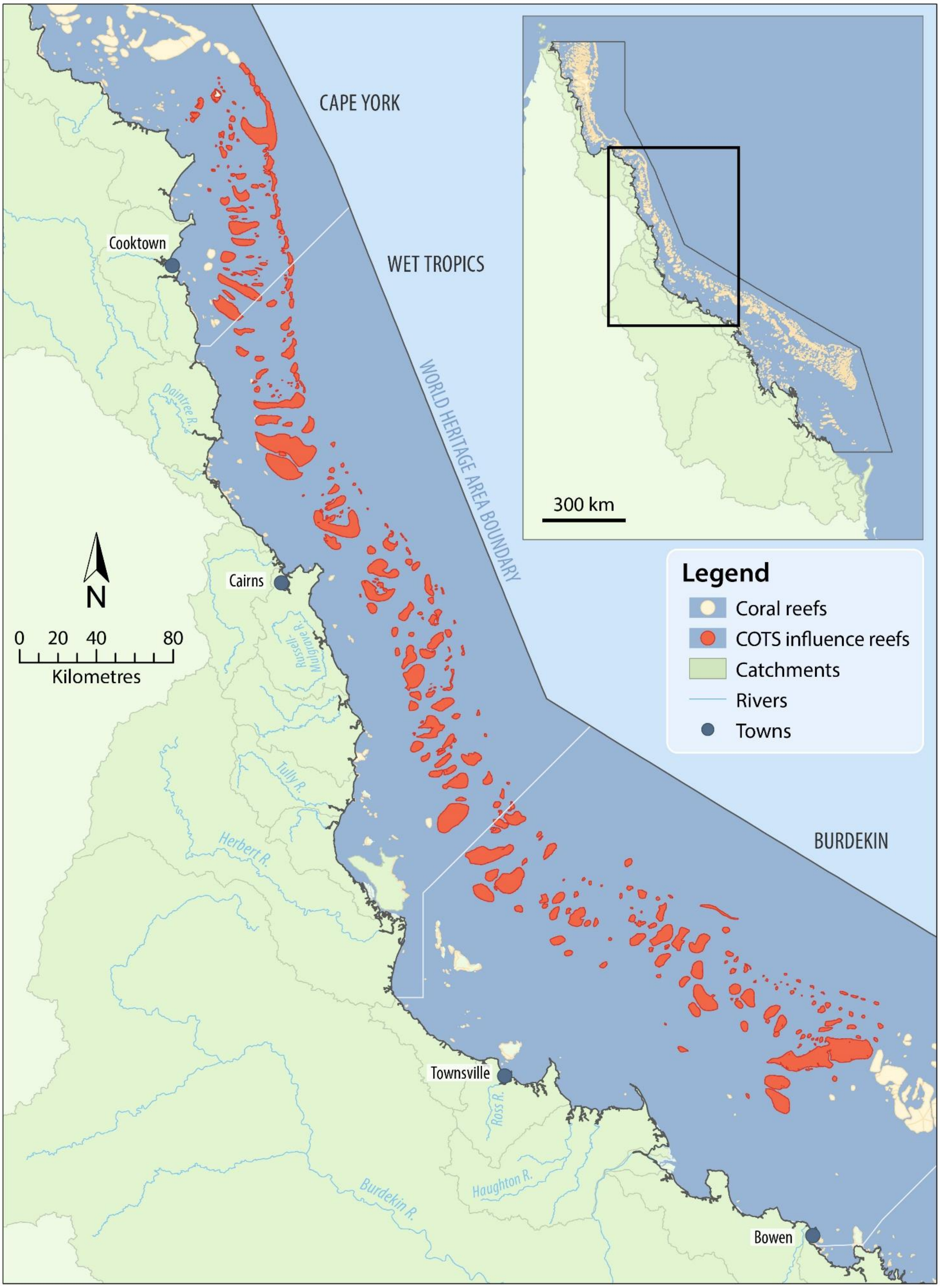


Figure 23. The Crown-of-Thorns starfish influence area, defined as one of example of DIN consequence for this assessment.

Results

The area calculations for the DIN Consequence (CoTs) example are shown in Table 10. The results show that the *greatest area of reefs in the High consequence class is in the Wet Tropics Marine Zone*, followed by the Cape York South Marine Zone and, to a lesser extent, the Burdekin Marine Zone. None of the other Marine Zones contain reefs in the Crown-of-Thorns starfish influence area and are therefore not within the High class for the consequence assessment.

The DIN Consequence (CoTs) Scores have not been ascribed to individual basins as this is considered in the final assessment.

Table 10. Area of coral reefs in the two DIN Consequence (CoTs) classes, High (inside) and Low (outside). The DIN Consequence (CoTs) Score is calculated for each Marine Zone by summing the area of coral reefs or seagrass in the High consequence category and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score**.**

CORAL REEFS	Area (km^2) per consequence category			DIN CONSEQUENCE (CoTs) SCORE		
	Low	High	Total	Area in H (km^2)	% of area in H	DIN Consequence (CoTs) Score
Cape York Nth	2,198	-	2,198	-	0.00	0.00
Cape York Cent.	1,126	-	1,126	-	0.00	0.00
Cape York Sth	703	268	971	268	0.28	0.66
Wet Tropics	123	408	531	408	0.77	1.00
Burdekin	112	3	115	3	0.03	0.01
Mackay Whitsunday	270	-	270	-	0.00	0.00
Fitzroy	511	-	511	-	0.00	0.00
Burnett Mary	133	-	133	-	0.00	0.00

7.2.2 Sediments

As for nutrients, it is outside the scope of this assessment to analyse the consequence of TSS exposure in the context of all the factors outlined above, and for many factors there are limited quantified data to complete the assessment. However, due to the dominance of the impact of light on seagrass and coral reefs, an example applying benthic light thresholds is presented to illustrate a consequence factor to calculate sediment risk for seagrass (see Figure 24).

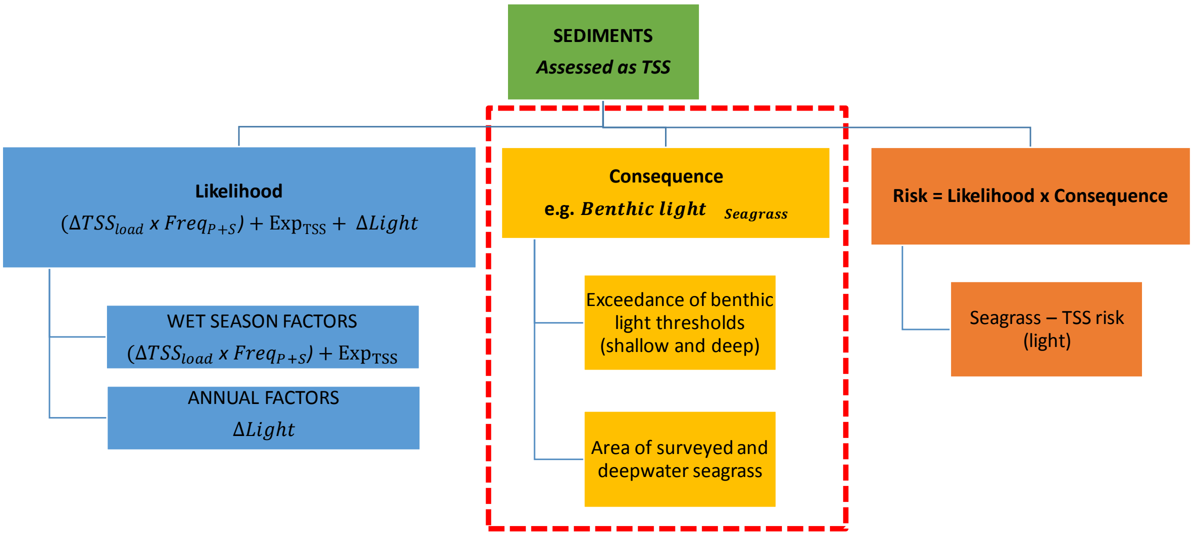


Figure 24. Framework for assessing the likelihood of exposure and consequence of TSS to seagrass and coral reefs. The consequence example presented here for seagrass focuses on the percentage of time that benthic light thresholds are not met for shallow and deepwater seagrass.

Despite the many potential consequences of increased suspended sediments as described above, it is necessary to assess risk on the basis of quantifiable consequences. There are well-documented effects of increased suspended sediment loads reducing benthic light and driving seagrass decline. Furthermore, light thresholds leading to this loss have been defined for several Great Barrier Reef seagrass species. Therefore, the risk of increased sediment loads has been assessed against potential consequences for seagrass meadows.

Suspended sediments attenuate light as it passes through the water column, and the greater the concentration of sediments, the greater the attenuation of light (the light attenuation coefficient, Kd, rises). This in turn reduces benthic light levels. Seagrasses have relatively high light requirements compared to other marine plants (Dennison et al., 1993), and yet they are widely distributed in inshore coastal areas where the likelihood of exposure to suspended sediments is high and where increases in anthropogenic loads have the greatest effect. This assessment was conducted on mapped seagrass distribution (composite distribution from all surveys), largely in the inshore region of the Great Barrier Reef (Carter et al., 2016), and on modelled deepwater seagrass distribution (>15 m), which occurs predominantly in the mid-outer Great Barrier Reef (Coles et al., 2009).

Light thresholds for seagrass have been derived using experimental approaches (Collier et al., 2016a; Chartrand et al., 2016) and have also been determined from changes in seagrass density in relation to natural variations in light (Chartrand et al., 2016; Collier et al., 2012), which revealed a relatively high degree of consistency among approaches. These studies were focused on *Zostera muelleri*, *Halophila uninervis*, *Cymodocea serrulata* and *Halophila ovalis* because of their predominance in habitat at risk from declining water quality from terrestrially derived pollutants and from dredging in port regions. These studies identified light thresholds based on short-term reductions in light. These thresholds were summarised in Collier et al. (2016b); in that report, thresholds were assigned to all species in the Great Barrier Reef (13 species) using the published results and unpublished results and, for some species, based on experiments that did not define thresholds but provided insight into light requirements (e.g. shading studies). In summary, if light declines below 6 mol/m^2/d, then long-bladed species of seagrass (i.e. not *Halophila* spp.), which typically grow in shallower water, are at risk of light limitation. However, the period of time below 6 mol/m^2/d that leads to loss varies among species. An impact is predicted after 28 days of continuous shading for the faster

growing (opportunistic species) and after more than 50 days in the persistent slower growing species, but in some cases it may take more than 100 days of continuous shading for an impact in these species. *Halophila* species growing in deepwater can be impacted if light falls below 2–2.5 mol/m^2/d continuously for 14–28 days (Collier et al., 2016b).

Elevated TSS loads will have lower consequence (i.e. cause less or no seagrass loss) in habitat that is naturally high in light and rarely reaching thresholds than in habitat that is already low in light and more frequently exceeds light thresholds. Therefore, the consequence of elevated TSS loads in this assessment was calculated from the number of days that did not meet light thresholds such that those already frequently not meeting thresholds were assigned the highest consequence rating. Specifically, locations where light was <6 mol/m^2/d for 42–100% days in the whole year (equivalent to wet season duration or longer) will suffer the greatest consequence of elevated TSS (i.e. seagrass loss is almost certainly expected if TSS loads increase in these habitats). The studies informing light thresholds on Great Barrier Reef species typically apply continuous shading, but intermittent or pulsed shading has lower impact (Chartrand et al., 2016); specifically, pulsed exposure to high light for the same duration as low light exposure (nine days shade, nine days light) prevents long-term decline (Biber et al., 2009). Therefore, the frequencies used to assess vulnerability in the consequence layer at annual timescales are longer than those derived from short-term studies. An explanation for categories is given in Table 11. A single annual threshold of 3 mol/m^2/d was used for modelled deepwater meadows, which is a conservative threshold (York et al., 2013; Collier et al., 2016b). Light was <3 mol/m^2/d at almost all modelled deepwater seagrasses (and therefore was included as the highest consequence category), but most modelled deepwater seagrass meadows fell outside of the Marine Zones and were therefore excluded from the analysis. The percentage of days that did not meet the benthic light threshold is shown spatially in Figure 25.

Table 11. Vulnerability of shallow water long-bladed seagrass meadows to low light. These were used to inform the consequence categories for the mapped seagrass (composite).

% days below threshold (days)	Consequence of continuous exposure (published)	Consequence of annual exposure (inferred)
0	Healthy seagrass meadows, assuming no other impacts	Healthy seagrass meadows, assuming no other impacts
10% (37 d)	Would cause reduction in biomass in some species if under continuous shading (Chartrand et al., 2016; Collier et al., 2016a)	Intermittent exceedance is unlikely to have an impact (Chartrand et al., 2016)
28% (102 d)	Will cause complete mortality in some species (Collier et al., 2012)	Likely to already affect density of seagrass and make them vulnerable to further impacts. This is commonly exceeded in monitoring meadows (shallow subtidal and coastal meadows; McKenzie et al., 2016).
42% (153 d)	Will cause complete mortality in most species (Collier et al., 2012)	Will have a large impact on some species, probably affecting species composition. Highly vulnerable to further impacts. This level of exceedance occurs in some monitored meadows in most years, even low flow years (McKenzie et al., 2016).

The area of coral reefs, surveyed seagrass, modelled deepwater seagrass and total areas in each consequence category in each Marine Zone was calculated by intersecting the final TSS Consequence spatial layer with the spatial layers for surveyed and modelled deepwater seagrass.

The *TSS Consequence Score* was calculated for each Marine Zone by summing the area of seagrass in the Moderate and High consequence categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score.

Results

The area calculations for the *TSS Consequence (Light)* example for seagrass are shown in Table 12.

For *surveyed seagrass, the greatest area not meeting benthic light thresholds was in the Burnett Mary Marine Zone (2423 km^2).* This is followed by the Cape York South Marine Zone (Index 0.62, 1494 km^2), Burdekin (Index 0.32, 771 km^2), Fitzroy (Index 0.27, 657 km^2) and Cape York Central (Index 0.2, 496 km^2) (Table 12). The Cape York North and Mackay Whitsunday Marine Zones had similar areas (Index around 0.1).

For *modelled deepwater seagrass, the greatest area not meeting benthic light thresholds was in the Cape York South Marine Zone* (4134 km^2). This is followed by the Wet Tropics (Index 0.69, 2857 km^2), Cape York North, Cape York Central and Burnett Mary Marine Zones (all around Index 0.35, 1500 km^2), Burdekin (Index 0.25, 1040 km^2) and a relatively small area in the Fitzroy and Mackay Whitsunday Marine Zones (Table 12).

The greatest area not meeting benthic light thresholds for total seagrass area was also in the Cape York South Marine Zone (5628 km^2). This is followed by the Burnett Mary Marine Zone (Index 0.68, 3840 km^2), Wet Tropics (Index 0.54, 3047 km^2), Cape York North, Cape York Central and Burdekin (Indexes all around 0.35), Fitzroy (Index 0.2, 1152 km^2) and Mackay Whitsunday (Index 0.08, 427 km^2) (Table 12).

The *TSS Consequence (Light) Scores* have not been ascribed to individual basins as this is considered in the final assessment of an example of TSS risk assessment for seagrass.

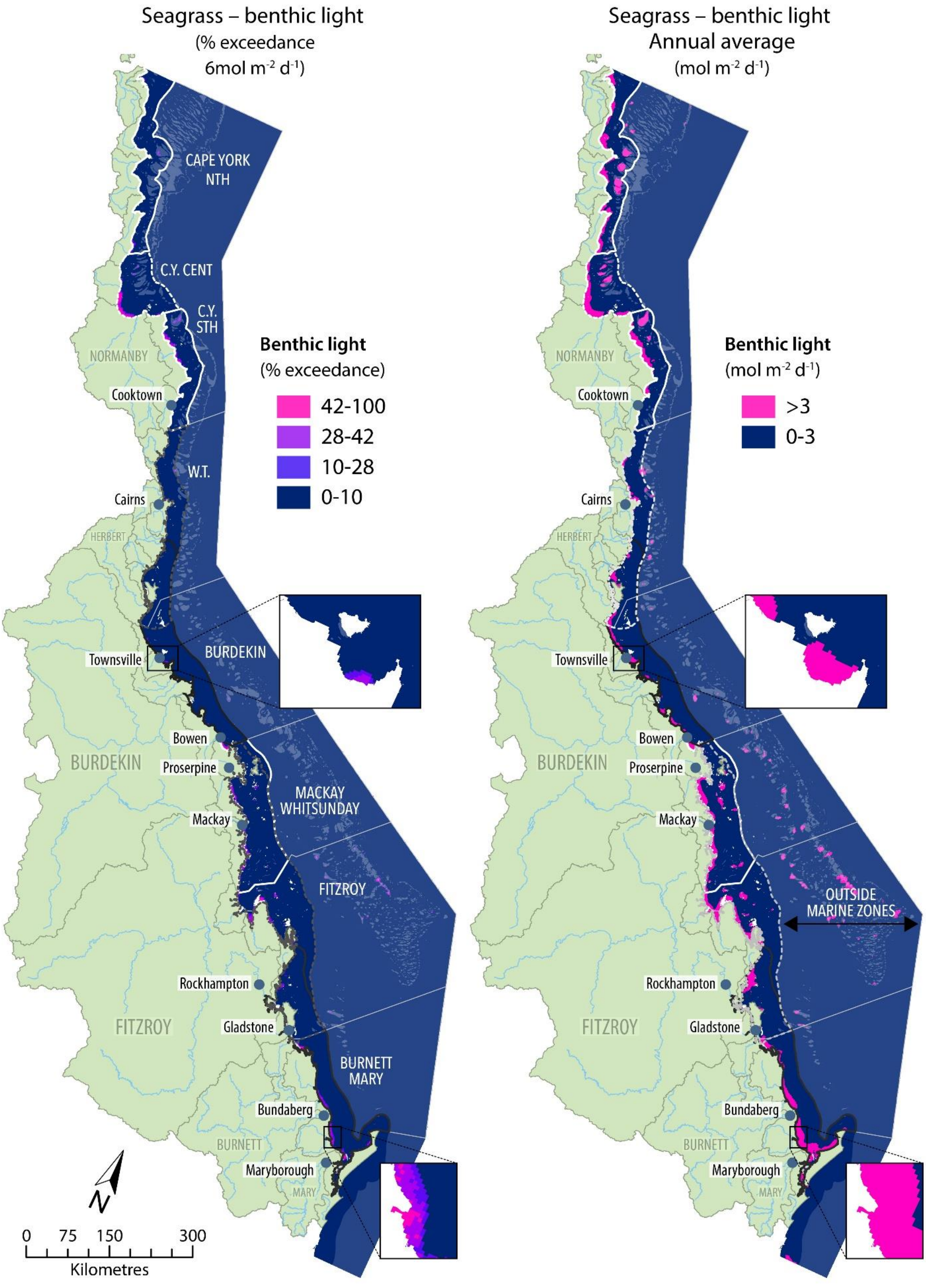


Figure 25. Per cent of days in each year that the benthic light thresholds were not met for surveyed seagrass (<6 mol/m^2/d) and modelled deepwater seagrass (<3 mol/m^2/d), 2011-2014.

Table 12. Area of seagrass (surveyed, modelled deepwater and total area) in the TSS Consequence (Light) classes assessed by the area not meeting benthic light thresholds. The TSS Consequence (Light) Index is calculated for each Marine Zone by summing the area of seagrass in the Moderate and High consequence categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score**.**

SEAGRASS (surveyed)	Area (km^2) per category						TSS CONSEQUENCE (LIGHT) INDEX		
	Neg.	V. Low	Low	Mod.	High	Total	Area (km^2) (M+H)	% of area in M+H	TSS Consequence Index
Cape York Nth	-	0	11	28	262	301	290	0.96	0.12
Cape York Cent.	-	31	32	35	461	558	496	0.89	0.20
Cape York Sth	-	65	148	82	1,412	1,708	1,494	0.87	0.62
Wet Tropics	-	1.2	5.2	6.2	184	196	190	0.97	0.08
Burdekin	-	0.8	16	25	746	788	771	0.98	0.32
Mackay Whitsunday	-	11	46	17	217	291	234	0.81	0.10
Fitzroy	-	9.5	71	35	622	738	657	0.89	0.27
Burnett Mary	-	49	132	84	2,338	2,604	2,423	0.93	1.00
						Max	*2,423*		

DEEPWATER SEAGRASS (modelled)	Area (km^2) per category						TSS CONSEQUENCE (LIGHT) INDEX		
	Neg.	V. Low	Low	Mod.	High	Total	Area (km^2) (M+H)	% of area in M+H	TSS Consequence Index
Cape York Nth	-	15	-	-	1,487	1,503	1,487	0.99	0.36
Cape York Cent.	-	0.0	-	-	1,575	1,575	1,575	1.00	0.38
Cape York Sth	-	61	-	-	4,134	4,194	4,134	0.99	1.00
Wet Tropics	-	11	-	-	2,857	2,868	2,857	1.00	0.69
Burdekin	-	-	-	-	1,040	1,040	1,040	1.00	0.25
Mackay Whitsunday	-	25	-	-	193	218	193	0.88	0.05
Fitzroy	-	78	-	-	495	574	495	0.86	0.12
Burnett Mary	-	0.1	-	-	1,417	1,418	1,417	1.00	0.34
				Max			*4,134*		

TOTAL SEAGRASS	Area (km^2) per risk category						TSS CONSEQUENCE (LIGHT) INDEX		
	Neg.	V. Low	Low	Mod.	High	Total	Area (km^2) (M+H)	% of area in M+H	TSS Consequence Index
Cape York Nth	-	15	11	28	1,750	1,804	1,777	0.99	0.32
Cape York Cent.	-	31	32	35	2,036	2,133	2,071	0.97	0.37
Cape York Sth	-	126	148	82	5,545	5,902	5,628	0.95	1.00
Wet Tropics	-	12	5	6	3,041	3,064	3,047	0.99	0.54
Burdekin	-	1	16	25	1,786	1,828	1,811	0.99	0.32
Mackay Whitsunday	-	36	46	17	410	509	427	0.84	0.08
Fitzroy	-	88	71	35	1,117	1,311	1,152	0.88	0.20
Burnett Mary	-	49	132	84	3,756	4,022	3,840	0.95	0.68
						Max	*5,628*		

7.2.3 Pesticides

Consequence of pesticide exposure to Great Barrier Reef ecosystems was measured using two methods: (i) an ecological risk assessment based on five PSII herbicides: ametryn, atrazine, diuron, hexazinone and tebuthiuron; and (ii) a hazard assessment of the individual pesticides detected in monitored catchments. The inputs for assessing the potential ecological consequences for an ecosystem from pesticide exposure were ecotoxicity data of species sensitivity distributions. The species sensitivity distribution (SSD) concept is a well-acknowledged and -utilised approach for the derivation of environmental quality criteria (e.g. water quality guideline values) and ecological risk assessments (Posthuma et al., 2002). Species sensitivity distributions are a compilation of the available ecotoxicology data (that pass quality criteria) sourced from the literature and various ecotoxicology databases. Consequence of exposure of pesticides to an ecosystem can therefore be assessed in terms of the proportion of species in the ecosystem likely to be adversely affected by the exposure. Applying this technique has a number of advantages: it is the same method that is used to determine the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000) for freshwater and marine protection and it provides an assessment of the potential impact to multiple species and taxa rather than to a single species. Species sensitivity distributions for 28 pesticides relevant to the Great Barrier Reef were developed for the purpose of generating water quality guideline values for the revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000). At the time of publishing this report, that review process was ongoing; thus, the generated values were published as ecotoxicity threshold values for interim use (King et al., 2017a; King et al., 2017b; Warne et al., in press).

The ecotoxicity data for the five PSII herbicides embedded in their individual species sensitivity distributions were extracted to generate the consequence input data for the risk assessment (the results are presented in the following section, on risk). The inputs required for the risk assessment vary slightly from the input data used to generate the ecotoxicity threshold values. Ecotoxicity threshold values are exclusive to each pesticide and are specific for either freshwater or marine ecosystems, whereas the risk assessment aims to determine the risk of pesticides from mixtures of the five PSII herbicides across multiple ecosystem types (freshwater, estuarine and marine). Therefore, the ecotoxicity data for marine and freshwater species were combined for each of the PSII herbicides and a new species sensitivity distribution was developed using the combined species. Combining freshwater and marine species in the species sensitivity distributions has two purposes: (i) the risk assessment is consistent and covers freshwater, estuarine and marine ecosystems without having to be spatially exclusive; and (ii) confidence in the estimations of the species sensitivity distributions is improved with larger datasets. A few studies have looked at the effects of salinity on the toxicity of pesticides. DeLorenzo et al. (2011) found that atrazine and diuron had higher toxicities to a halophilic microalga (*Dunaliella tertiolecta*) under higher salinities, suggesting that at higher salinities the rate of chemical diffusion across the cell membrane increases, resulting in greater herbicide toxicity. However, the same pattern was not observed for ametryn (DeLorenzo et al., 2011). Wheeler et al. (2002) found that, in general, differences between freshwater and marine species sensitivity distributions were not great. In particular, for pesticides, marine species were found to be more sensitive than freshwater species, although this assessment was restricted to organochlorine and organophosphate insecticides. Overall, the differences in sensitivities are likely to be minor compared to the higher levels of uncertainty that are generated from species sensitivity distributions with small datasets.

For the hazard assessment, the ecotoxicity threshold values published by King et al. (2017a; 2017b) were used to assess the potential consequences for ecosystems exposed to individual pesticides. In doing so, we could assess the existing level of protection—that is, 99%, 95%, 90% or 80% of species—present in each catchment based on the concentrations of the individual pesticides detected in those catchments. These levels of protection are synonymous with the levels of protection recommended for use in water quality management by the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC

and ARMCANZ, 2000) for the protection of ecosystems of high ecological value (99% species protection), slightly to moderately disturbed systems (95% of species protection) and highly disturbed systems (90–80% species protection).

8. What is the risk of degraded water quality to Great Barrier Reef ecosystems?

As described in the Introduction, several assessments of the relative risk of degraded water quality to Great Barrier Reef coral reef and seagrass ecosystems have been conducted in recent years. The current assessment was conducted separately for coral reefs and seagrass and also for each water quality parameter (TSS, DIN and pesticides) at a basin scale (the 35 major basins in the Great Barrier Reef catchment). This section presents the examples of DIN and Crown-of-Thorns starfish, and reduced light and seagrass. A full assessment of the risk of pesticides on freshwater and estuarine ecosystems is included. At this stage there is insufficient information on floodplain wetlands and floodplains to extend that assessment beyond consideration of the likelihood of exposure.

8.1 Assessing the risk of pollutant exposure to Great Barrier Reef marine ecosystems

8.1.1 Risk from DIN to coral reefs (Crown-of-Thorns starfish example)

The risk assessment uses the spatial outputs for the DIN Likelihood and Consequence assessments presented in previous sections to calculate risk (Figure 26). The final input layers are shown in the panel in Figure 27.

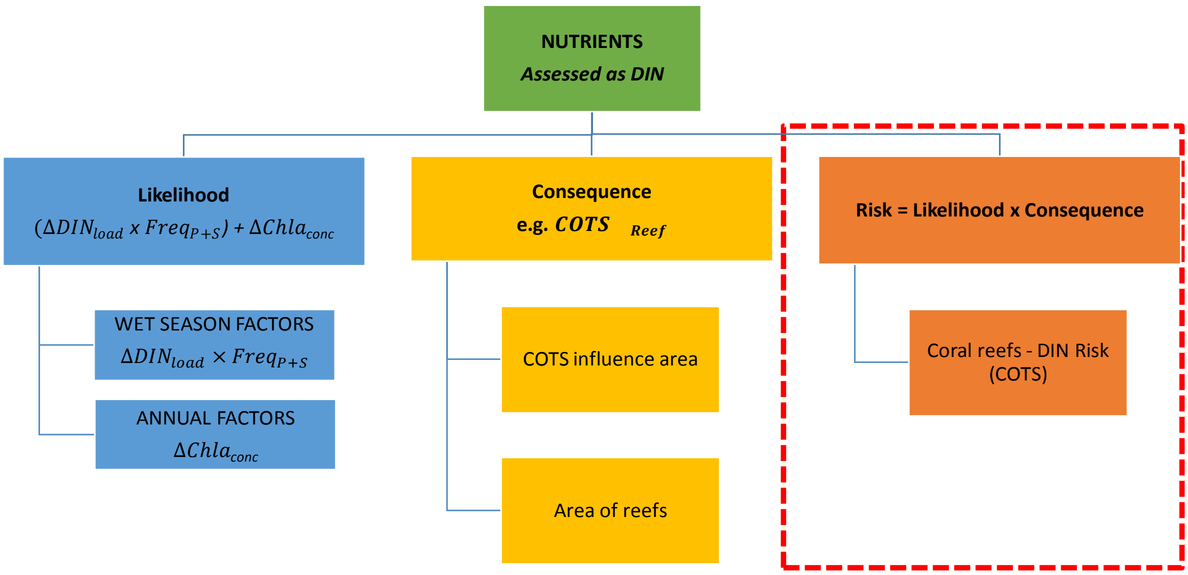


Figure 26. Framework for assessing the likelihood of exposure and consequence of DIN to coral reefs. The consequence example presented here for coral reefs highlights the importance of the Crown-of-Thorns starfish influence area for assessing risk to coral reefs.

A risk matrix for the product of the spatial layers of DIN Likelihood of Exposure and DIN Consequence (CoTs Influence Area) is shown in Table 13. The final categories were derived by expert opinion and are not fully quantified, so should be used as an example only.

Table 13. Risk matrix for the DIN risk to coral reefs for the example of the Crown-of-Thorns influence area.

		DIN Consequence (CoTs Influence Area)	
DIN Likelihood		Low (0.5)	High (1.0)
Very High	**0.8–1.0**	H	H
High	**0.6–0.8**	M	H
Moderate	**0.4–0.6**	M	H
Low	**0.2–0.4**	L	M
Very Low	**0.05–0.2**	L	L
Negligible	**0–0.05**	N	L

To generate the DIN Risk (CoTs) layer, the two normalised input layers for the DIN Likelihood of Exposure and DIN Consequence (CoTs) were multiplied together and the resulting layer divided by its maximum value to again normalise from 0 to 1.

The total area and areas of coral reefs in each risk category in each Marine Zone were calculated by overlaying the final DIN Risk (CoTs) spatial layer with coral reefs.

The *DIN Risk (CoTs) Score* was calculated for each Marine Zone by summing the area of coral reefs in the Moderate, High and Very High Risk categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score.

Results

The final DIN risk map for Crown-of-Thorns starfish influence is shown in the panel in Figure 27, and the results are shown in Table 14. The results show that *greatest area of risk to coral reefs from Crown-of-Thorns starfish influence is in the Wet Tropics Marine Zone (484 km^2)*, followed by the Cape York South Marine Zone (Index 0.31, 151 km^2) and, to a lesser extent, the Burdekin Marine Zone (0.16, 78 km^2). The risk in the other Marine Zones is associated with the classification shown in Table 13, where a Very High likelihood of DIN exposure can result in High risk. The assessment should therefore be constrained to the geographic area where the Crown-of-Thorns influence area exists, that is, the Cape York South, Wet Tropics and Burdekin Marine Zones.

Table 14. Calculations of the total areas within each DIN Risk (CoTs) category within each Marine Zone and the areas of coral reefs. The DIN Risk (CoTs) Score is calculated for each Marine Zone by summing the area of seagrass in the Moderate and High consequence categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score**.**

CORAL REEFS	Area (km^2) per risk category							DIN RISK (CoTs) SCORE –		
	Neg.	V. Low	Low	Mod.	High	V. High	Total	M+H+VH (km^2)	% of area (M+H+VH)	DIN Risk Score
Cape York Nth	2,198	-	-	-	-	-	2,198	0	0.0%	0.00
Cape York Cent.	625	501	-	-	-	-	1,126	0	0.0%	0.00
Cape York Sth	61	650	109	65	87	-	971	151	16%	0.31
Wet Tropics	-	14	33	145	218	121	531	484	91%	1.00
Burdekin	1.4	18	18	64	13	1.6	115	78	67%	0.16
Mackay Whitsunday	3.2	163	83	20	1.0	-	270	21	7.7%	0.04
Fitzroy	42	318	117	31	2.0	-	511	33	6.5%	0.07
Burnett Mary	53	38	29	12	0.9	-	133	13	9.4%	0.03
							Max	*484*		

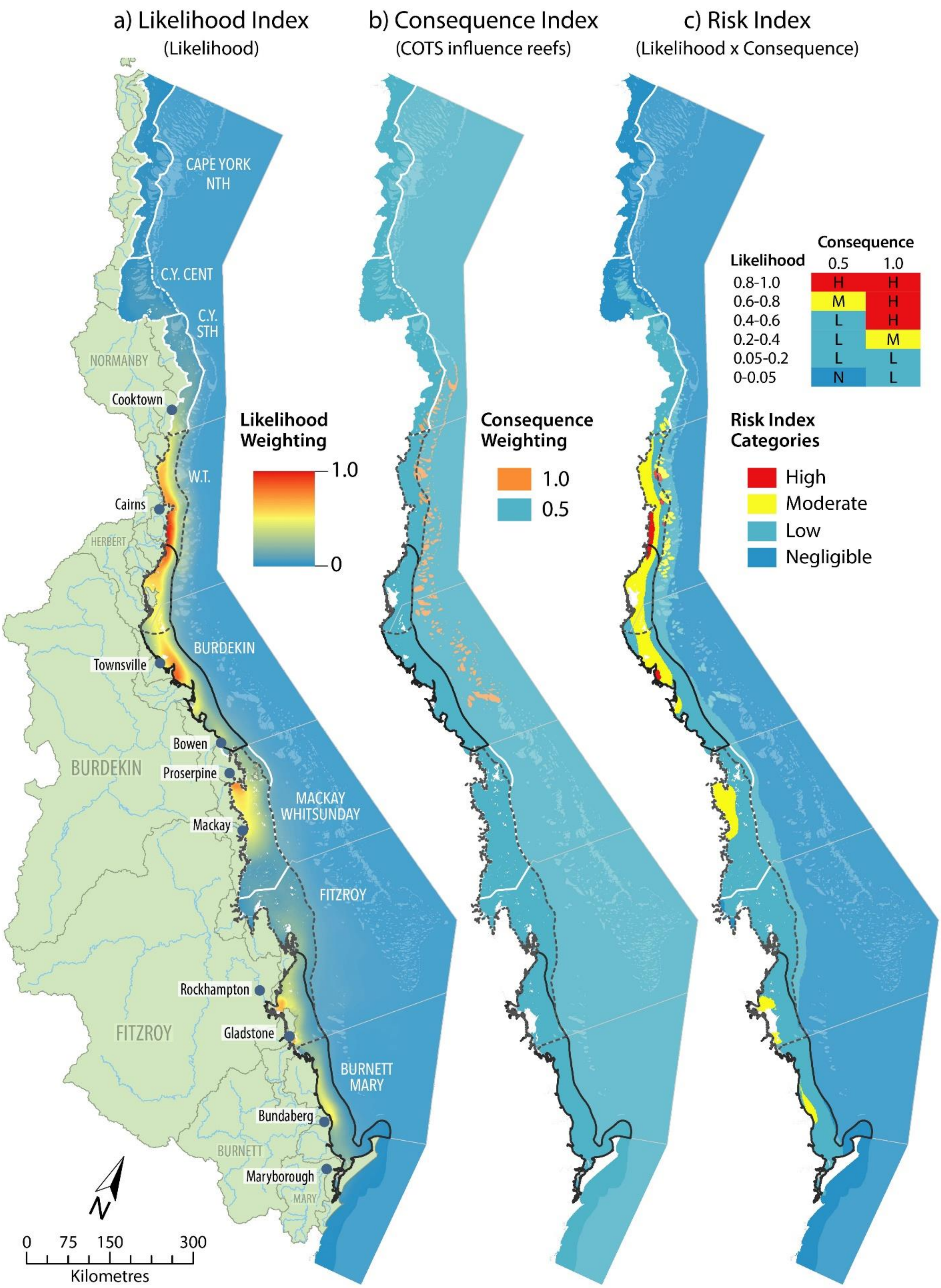


Figure 27. Inputs representing a) likelihood of exposure of anthropogenic river-derived DIN, b) an example of the consequence of exposure using the Crown-of-Thorns starfish influence area, and c) the estimated risk from river-derived DIN to coral reefs from Crown-of-Thorns starfish.

Linking DIN Risk (CoTs example) to individual basins

To attribute the risk of DIN to coral reefs from Crown-of-Thorns starfish to individual basins, the DIN Load Index described in Section 6.2.1 was multiplied by the DIN Risk (CoTs) Scores in Table 14 for each Marine Zone to generate a *DIN Risk Index (CoTs)* for each basin (Table 15. Where a river had more than one Index result because it contributed to more than one Marine Zone, the highest Index was considered in the overall ranking.

Table 15. Calculation of DIN Risk (CoTs example) to coral reefs for each basin using a DIN Load Index based on the proportion of anthropogenic DIN load that each basin contributes to the total anthropogenic DIN load of the Marine Zone and the DIN Risk Scores for each marine Zone (Table 14). The DIN Risk Index for each basin is the product of the DIN Load Index and the DIN Risk for each Marine Zone and is ranked across the 35 Great Barrier Reef basins; the top 5 rivers are highlighted in red, and the rivers ranked 5 to 10 are highlighted in orange. Only the rivers in the Cape York South, Wet Tropics and Burdekin Marine Zones are shown in this example as the assessment is not relevant outside of the Crown-of-Thorns starfish influence area.

Marine zone	Basin name	DIN Load Index within Marine Zone	DIN Risk Score (CoTs)	DIN Risk Index (CoTs) (DIN Load Index x DIN Risk Score)	RANK across the Great Barrier Reef
Cape York South	Jeannie River	0.01	0.31	0.00	
	Endeavour River	0.04	0.31	0.01	
	Daintree River	0.27	0.31	0.08	
	Mossman River	0.21	0.31	0.06	
	Mulgrave-Russell River	0.85	0.31	0.26	6
	Johnstone River	1.00	0.31	0.31	5
Wet Tropics	Daintree River	0.15	1.00	0.15	8
	Mossman River	0.12	1.00	0.12	9
	Barron River	0.10	1.00	0.10	10
	Mulgrave-Russell River	0.48	1.00	0.48	3
	Johnstone River	0.56	1.00	0.56	2
	Tully River	0.43	1.00	0.43	4
	Murray River	0.26	1.00	0.26	6
	Herbert River	1.00	1.00	1.00	1
	Burdekin River	0.01	1.00	0.01	
	Haughton River	0.05	1.00	0.05	
Burdekin	Tully River	0.42	0.16	0.07	
	Murray River	0.25	0.16	0.04	
	Herbert River	0.97	0.16	0.15	8
	Black River	0.02	0.16	0.00	
	Ross River	0.13	0.16	0.02	
	Haughton River	1.00	0.16	0.16	7
	Burdekin River	0.19	0.16	0.03	
	Don River	0.07	0.16	0.01	

The assessment of DIN risk from the Crown-of-Thorns starfish influence area at a basin scale indicates that the *Herbert Basin has the greatest contribution to DIN risk to coral reefs through Crown-of-Thorns starfish influence (Index 1.00). This is followed by the Johnstone (Index 0.56), Russell-Mulgrave (Index 0.46) and Tully (0.43) basins.* The Murray (0.26) and Daintree (Index 0.15) also rank relatively high in the contribution to the

DIN risk in the Crown-of-Thorns influence area. The Haughton Basin also contributes to DIN risk from Crown-of-Thorns starfish but to a lesser extent than the Wet Tropics basins (Index 0.16).

8.1.2 Risk from fine sediment to seagrass (light)

The risk assessment uses the spatial outputs for the TSS Likelihood and Consequence assessments presented in previous sections to calculate risk (represented in Figure 28). The final input layers are shown in the panel in Figure 29.

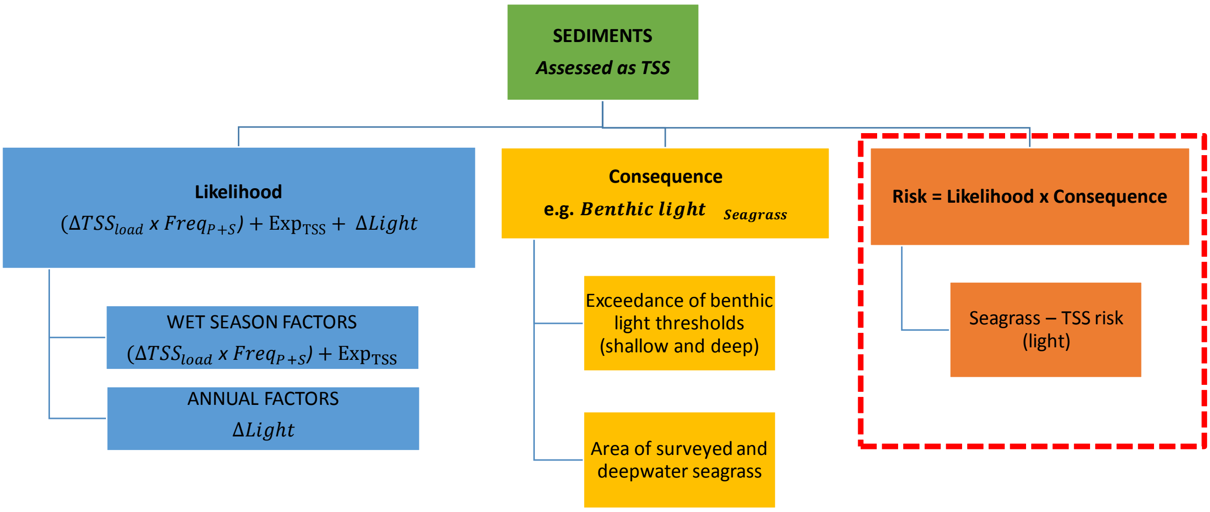


Figure 28. Framework for assessing the likelihood of exposure and consequence of TSS to seagrass and coral reefs. An example of consequence and risk is also provided for TSS, benthic light and seagrass.

A risk matrix for the product of the spatial layers of TSS Likelihood of Exposure and TSS Consequence (benthic light) is shown in Table 16. As for DIN, *the final categories* were derived by expert opinion and are not fully quantified, so *should be used as an example or demonstration of the concept only*. For example, if seagrass were already under frequent exposure to low light (42–100% of days), then a likelihood of this increasing (of >0.4) was considered to be significant (High risk).

Table 16. Likelihood x Consequence rating used in the assessment. The risk categories were derived through expert elicitation and were categorised as High (red), Medium (orange), Low (Yellow) and Very Low (Green).

		Consequence, % days below threshold			
		0–10%	**10–28%**	**28–42%**	**42–100%**
Likelihood of TSS exposure		**0.5**	**0.67**	**0.83**	**1**
negligible	**0**	0	0	0	0
0–0.2	**0.2**	0.1	0.134	0.166	0.2
0.2–0.4	**0.4**	0.2	0.268	0.332	0.4
0.4–0.6	**0.6**	0.3	0.402	0.498	**0.6**
0.6–0.8	**0.8**	0.4	0.536	**0.664**	**0.8**
0.8–1.0	**1**	0.5	**0.67**	**0.83**	**1**

To generate the Risk layer, the two normalised input layers for TSS Likelihood of Exposure and TSS Consequence (benthic light) were multiplied together and the resulting layer divided by its maximum value to normalise from 0 to 1.

The total area and area of surveyed seagrass and modelled deepwater seagrass in each risk category in each Marine Zone were calculated by overlaying the final TSS Risk (benthic light) spatial layer with the layers for seagrass.

The *TSS Risk (benthic light) Score* was calculated for each Marine Zone by summing the area of surveyed, modelled deepwater and total seagrass in the Moderate, High and Very High Risk categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score.

Results

The final TSS risk map for TSS, benthic light and seagrass is shown in the panel in Figure 29, and total areas within each final risk category within each Marine Zone and the areas of seagrass (surveyed, modelled deepwater and total) are presented in Table 17.

Most of the modelled deepwater seagrass falls outside of the Marine Zone (Figure 31) and therefore is not included in the risk index (Table 17).

The results show that *greatest area of risk to surveyed seagrass from not meeting benthic light thresholds is in the Burdekin Marine Zone (687 km^2)*. This is followed by the Burnett Mary Marine Zone (Index 0.63, 432 km^2), the Fitzroy (Index 0.38, 263 km^2), Wet Tropics (Index 0.26, 179 km^2) and Mackay Whitsunday (Index 0.16, 110 km^2) (Table 17). The Cape York Marine Zones were ranked relatively low, but the Cape York Central Marine Zone had the highest relative risk (Index 0.11) compared to the Cape York North (Index 0.03) and Cape York South (Index 0.06) Marine Zones.

The *greatest area of risk to modelled deepwater seagrass from not meeting benthic light thresholds is in the Burnett Mary Marine Zone (87 km^2)*. This is followed by the Wet Tropics (Index 0.79, 69 km^2), the Fitzroy (Index 0.71, 62 km^2), Mackay Whitsunday (Index 0.17, 15 km^2) and Burdekin (Index 0.15, 13 km^2) (Table 17). The Cape York Marine Zones were relatively low, with Indexes all less than 0.05.

The *greatest area of risk to the total seagrass area from not meeting benthic light thresholds is in the Burdekin Marine Zone (700 km^2)*. This is followed by the Burnett Mary Marine Zone (Index 0.74, 520 km^2), the Fitzroy (Index 0.46, 325 km^2), Wet Tropics (Index 0.35, 248 km^2) and Mackay Whitsunday (Index 0.18, 125 km^2) (Table 17). The Cape York Marine Zones were ranked relatively low, but the Cape York Central Marine Zone had the highest relative risk (Index 0.11) compared to the Cape York North (Index 0.03) and Cape York South (Index 0.06) Marine Zones.

Linking TSS Risk (seagrass benthic light example) to individual basins

To attribute the risk of TSS to seagrass from reduced benthic light to individual basins, the TSS Load Index described in Section 6.1.2 was multiplied by the TSS Risk (benthic light) Score in Table 17 for each Marine Zone to generate a *TSS Risk Index* (benthic light) for each basin (Table 18). Where a river had more than one Index result because it contributed to more than one Marine Zone, the highest Index was considered in the overall ranking.

The assessment of TSS Risk from failure to meet the benthic light thresholds at a basin scale indicates that the *Burdekin Basin has the greatest contribution to TSS risk to surveyed seagrass and total seagrass area. The Fitzroy Basin has the greatest contribution to TSS risk to deepwater modelled seagrass and ranks second for surveyed and total seagrass area. The Mary, Herbert, Johnstone, Burnett and Russell-Mulgrave basins also rank relatively highly for all seagrass assessments, although these areas are relatively small contributors compared to the Burdekin and Fitzroy.*

Table 17. Calculations of the total areas within each TSS Risk (benthic light) category within each Marine Zone and the areas of seagrass. The TSS Risk (Benthic Light) Score is calculated for each Marine Zone by summing the area of seagrass in the Moderate and High consequence categories and normalising the value to the maximum result to provide a relative index between the Marine Zones, that is, the Marine Zone with the highest area is assigned a value of 1.0, and all other areas are expressed as a value between 0.0 and 1.0, relative to the maximum. These scores are ranked among the Marine Zones, from the highest (1) to lowest score**.**

SEAGRASS (surveyed)	Area (km^2) per category						TSS RISK (BENTHIC LIGHT) SCORE		
	Neg.	V. Low	Low	Mod.	High	Total	M+H (km^2)	% of area (M+H)	TSS Risk Score
Cape York Nth	1.5	42	234	24	0.0	301	24	7.9%	0.03
Cape York Cent.	1.3	57	425	75	0.0	558	75	13.4%	0.11
Cape York Sth	2.8	352	1,312	39	1.4	1,708	41	2.4%	0.06
Wet Tropics	1.8	4.1	11	125	54	196	179	91.2%	0.26
Burdekin	4.2	7.9	88	356	331	788	687	87.2%	1.00
Mackay Whitsunday	4.1	20	157	102	7.4	291	110	37.8%	0.16
Fitzroy	8.2	38	429	229	33	738	263	35.6%	0.38
Burnett Mary	4.6	13	2,154	402	30	2,604	432	16.6%	0.63
				Max	*331*		*687*		

DEEPWATER SEAGRASS (modelled)	Area (km^2) per category						TSS RISK (BENTHIC LIGHT) SCORE		
	Neg.	V. Low	Low	Mod.	High	Total	M+H (km^2)	% of area (M+H)	TSS Risk Score
Cape York Nth	-	23	1,480	-	-	1,503	0.0	0.0%	0.00
Cape York Cent.	-	91	1,484	-	-	1,575	0.0	0.0%	0.00
Cape York Sth	-	1,299	2,892	2.9	-	4,195	2.9	0.1%	0.03
Wet Tropics	0.0	1,347	1,452	69	-	2,868	69	2.4%	0.79
Burdekin	-	596	431	13	-	1,040	13	1.2%	0.15
Mackay Whitsunday	-	24	179	15	-	218	15	6.8%	0.17
Fitzroy	-	154	357	62	-	574	62	10.9%	0.71
Burnett Mary	-	546	784	87	-	1,418	87	6.2%	1.00
				Max	-		*87*		

TOTAL SEAGRASS	Area (km^2) per category						TSS RISK (BENTHIC LIGHT) SCORE		
	Neg.	V Low	Low	Mod.	High	Total	M+H (km^2)	% of area (M+H)	TSS Risk Score
Cape York Nth	1.5	65	1,713	24	-	1,804	24	1.3%	0.03
Cape York Cent.	1.3	148	1,908	75	-	2,133	75	3.5%	0.11
Cape York Sth	2.8	1,651	4,205	42	1.4	5,902	44	0.7%	0.06
Wet Tropics	1.9	1,351	1,463	194	54	3,06	248	8.1%	0.35
Burdekin	4.2	604	519	369	331	1,828	700	38.3%	1.00
Mackay Whitsunday	4.1	44	336	117	7.4	509	125	24.5%	0.18
Fitzroy	8.2	191.8	786	292	33	1,311	325	24.8%	0.46
Burnett Mary	4.6	559.6	2,938	489	30	4,022	520	12.9%	0.74
				Max	331		*700*		

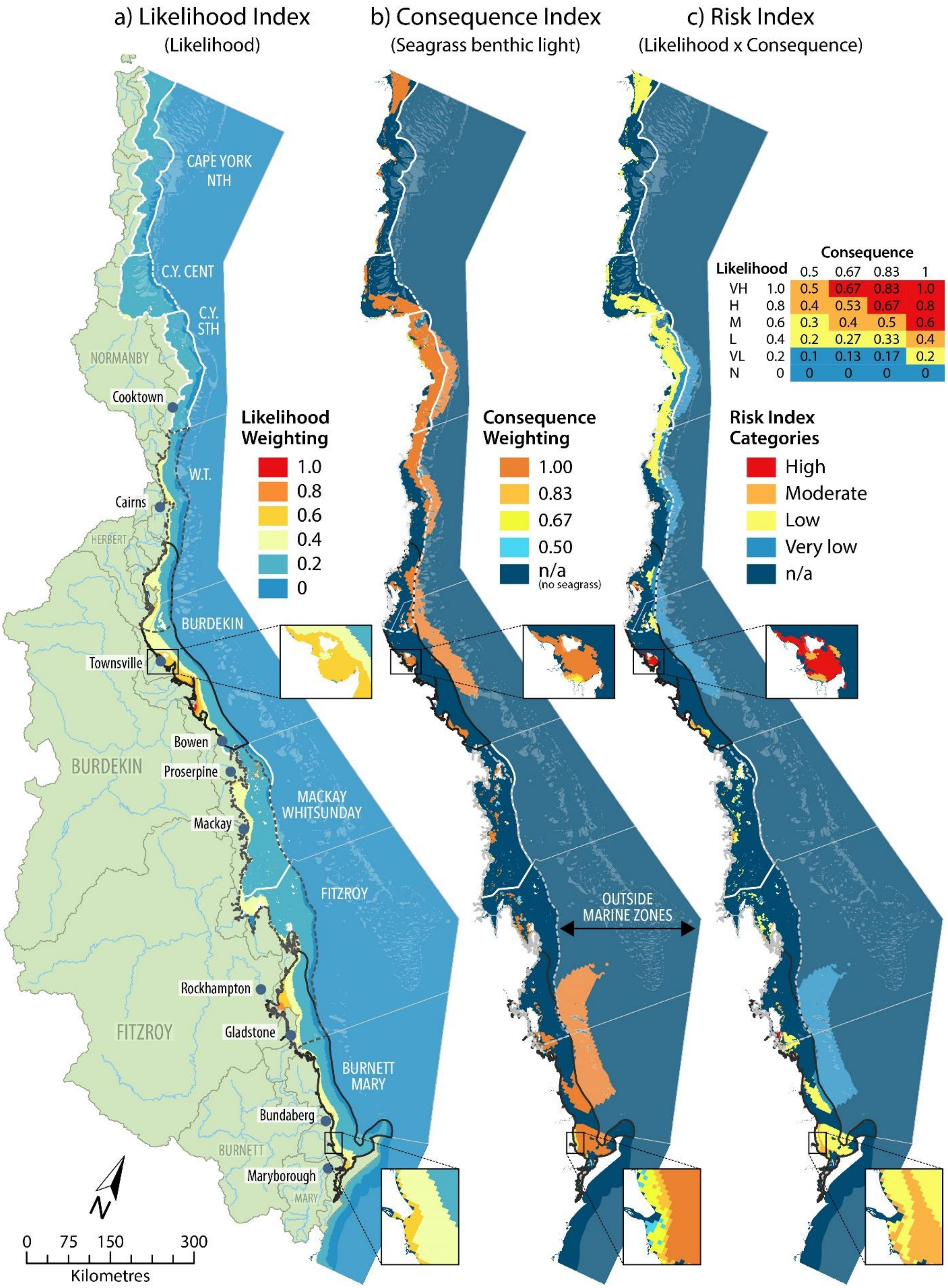


Figure 29. Inputs representing a) likelihood of exposure of anthropogenic TSS to seagrass, b) an example of the consequence of exposure using an analysis of the failure to meet benthic light thresholds for seagrass, and c) the estimated risk from anthropogenic TSS to seagrass.

Table 18. Calculation of TSS Risk (seagrass and benthic light example) for each basin using a TSS Load Index based on the proportion of anthropogenic TSS load that each basin contributes to the total anthropogenic TSS load of the Marine Zone. The TSS Risk Index for each basin is the product of the TSS Load Index and the TSS Risk for each Marine Zone (Table 17) and is ranked across the 35 Great Barrier Reef basins; the top 5 rivers are highlighted in red shading, and the rivers ranked 5 to 10 are highlighted in orange shading**.**

Marine zone	Basin name	TSS Load Index within Marine Zone	SURVEYED SEAGRASS			MODELLED DEEPWATER SEAGRASS			TOTAL AREA SEAGRASS		
			TSS Risk Score	TSS Risk for each basin	Rank	TSS Risk Score	TSS Risk for each basin	Rank	TSS Risk Score	TSS Risk for each basin	Rank
Cape York North	Jacky Jacky Creek	0.80	0.03	0.02	33	0.00	0.00	42	0.03	0.02	35
	Olive Pascoe River	1.00	0.03	0.03	23	0.00	0.00	42	0.03	0.03	28
	Lockhart River	1.00	0.03	0.03	22	0.00	0.00	42	0.03	0.03	27
Cape York Central	Stewart River	0.27	0.11	0.03	24	0.00	0.00	42	0.11	0.03	29
	Normanby River	0.27	0.11	0.03	24	0.00	0.00	42	0.11	0.03	29
Cape York South	Jeannie River	1.00	0.06	0.06	16	0.03	0.03	27	0.06	0.06	17
	Endeavour River	0.89	0.06	0.05	18	0.03	0.03	29	0.06	0.05	20
Wet Tropics	Daintree River	0.08	0.26	0.02	35	0.79	0.07	17	0.35	0.03	32
	Mossman River	0.02	0.26	0.01	45	0.79	0.02	32	0.35	0.01	45
	Barron River	0.10	0.26	0.02	31	0.79	0.08	15	0.35	0.03	24
	Mulgrave-Russell River	0.47	0.26	0.12	9	0.79	0.37	6	0.35	0.17	9
	Johnstone River	0.79	0.26	0.20	7	0.79	0.62	4	0.35	0.27	6
	Tully River	0.25	0.26	0.07	15	0.79	0.20	8	0.35	0.09	12
	Murray River	0.12	0.26	0.03	21	0.79	0.09	14	0.35	0.04	21
	Herbert River	1.00	0.26	0.26	5	0.79	0.79	2	0.35	0.35	5
Burdekin	Tully River	0.03	1.00	0.03	26	0.15	0.00	38	1.00	0.03	31
	Murray River	0.01	1.00	0.01	41	0.15	0.00	40	1.00	0.01	41
	Herbert River	0.12	1.00	0.12	10	0.15	0.02	31	1.00	0.12	11
	Black River	0.01	1.00	0.01	42	0.15	0.00	41	1.00	0.01	42
	Ross River	0.02	1.00	0.02	36	0.15	0.00	39	1.00	0.02	39
	Haughton River	0.06	1.00	0.06	17	0.15	0.01	37	1.00	0.06	18

Marine zone	Basin name	TSS Load Index within Marine Zone	SURVEYED SEAGRASS			MODELLED DEEPWATER SEAGRASS			TOTAL AREA SEAGRASS		
			TSS Risk Score	TSS Risk for each basin	Rank	TSS Risk Score	TSS Risk for each basin	Rank	TSS Risk Score	TSS Risk for each basin	Rank
	Burdekin River	1.00	1.00	1.00	1	0.15	0.15	10	1.00	1.00	1
	Don River	0.07	1.00	0.07	14	0.15	0.01	35	1.00	0.07	15
Mackay Whitsunday	Proserpine River	0.31	0.16	0.05	20	0.17	0.05	19	0.18	0.06	19
	O'Connell River	1.00	0.16	0.16	8	0.17	0.17	9	0.18	0.18	8
	Pioneer River	0.72	0.16	0.12	11	0.17	0.12	12	0.18	0.13	10
	Plane	0.41	0.16	0.07	13	0.17	0.07	16	0.18	0.07	14
Fitzroy	Proserpine River	0.06	0.38	0.02	34	0.71	0.04	22	0.46	0.03	34
	O'Connell River	0.19	0.38	0.07	12	0.71	0.13	11	0.46	0.09	13
	Pioneer River	0.13	0.38	0.05	19	0.71	0.10	13	0.46	0.06	16
	Plane	0.08	0.38	0.03	27	0.71	0.05	18	0.46	0.04	22
	Styx River	0.07	0.38	0.03	29	0.71	0.05	20	0.46	0.03	23
	Shoalwater Creek	0.05	0.38	0.02	37	0.71	0.03	25	0.46	0.02	36
	Waterpark Creek	0.04	0.38	0.02	38	0.71	0.03	26	0.46	0.02	37
	Fitzroy River	1.00	0.38	0.38	3	0.71	0.71	3	0.46	0.46	3
	Calliope River	0.04	0.38	0.01	40	0.71	0.03	28	0.46	0.02	38
	Boyne River	0.01	0.38	0.00	46	0.71	0.01	36	0.46	0.01	46
Burnett Mary	Waterpark Creek	0.04	0.63	0.03	28	1.00	0.04	21	0.74	0.03	25
	Fitzroy River	1.00	0.63	0.63	2	1.00	1.00	1	0.74	0.74	2
	Calliope River	0.04	0.63	0.02	32	1.00	0.04	24	0.74	0.03	33
	Boyne River	0.01	0.63	0.01	44	1.00	0.01	34	0.74	0.01	44
	Baffle Creek	0.04	0.63	0.03	30	1.00	0.04	23	0.74	0.03	26
	Kolan River	0.02	0.63	0.01	39	1.00	0.02	30	0.74	0.02	40
	Burnett River	0.33	0.63	0.21	6	1.00	0.33	7	0.74	0.24	7
	Burrum River	0.01	0.63	0.01	43	1.00	0.01	33	0.74	0.01	43
	Mary River	0.52	0.63	0.32	4	1.00	0.52	5	0.74	0.38	4

8.1.3 Pesticides

A risk assessment for mixtures of five PSII herbicides—ametryn, atrazine, diuron, hexazinone and tebuthiuron—detected at end-of-system sites in Great Barrier Reef catchments (freshwater and estuarine locations) was conducted to assess the likelihood that concentrations of pesticide mixtures were protective of 99% of species in the Great Barrier Reef World Heritage Area. In addition, to further assess other pesticides, monitored concentrations were compared against ecotoxicity threshold values to assess if the concentrations of individual pesticides detected at the end of catchments were protective of 99% of species in the Great Barrier Reef World Heritage Area. The framework is shown in Figure 30.

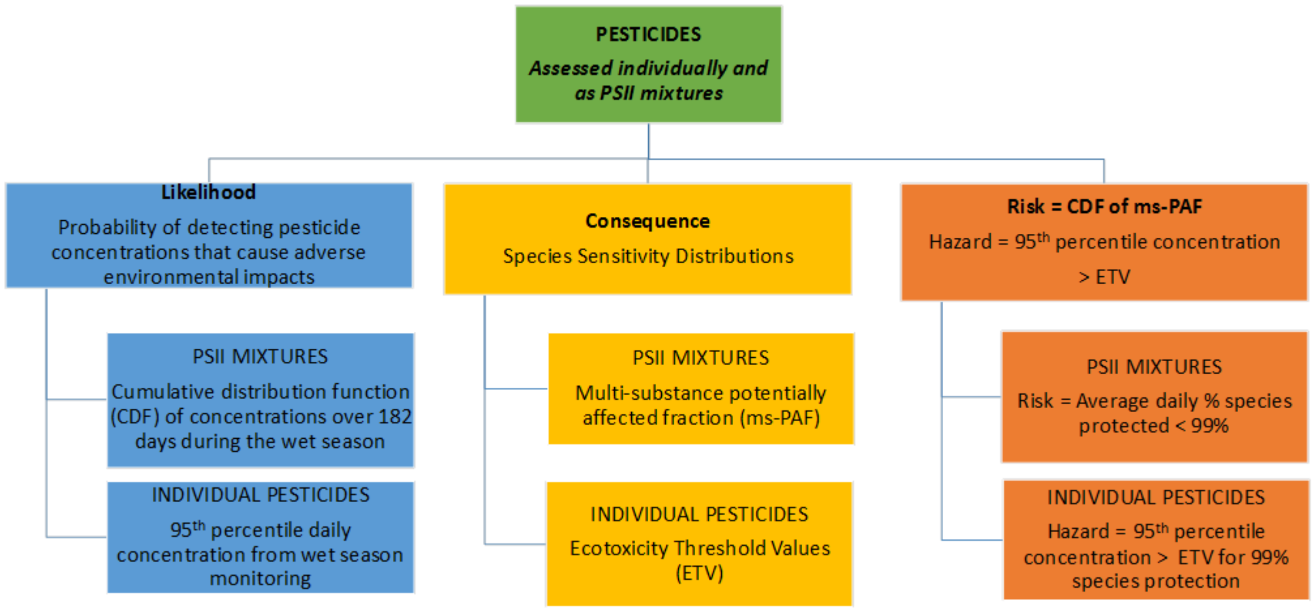


Figure 30. Framework for the ecological risk assessment for pesticides. Two different assessment methods were applied to evaluate: (i) PSII Mixtures—the ecological risk of mixtures of five photosystem II herbicides, including ametryn, atrazine, diuron, hexazinone and tebuthiuron, and (ii) Individual pesticides—the exceedance of ecotoxicity threshold values for up to 28 individual pesticides.

The risk and hazard assessments were conducted using monitored pesticide concentration data collected as part of the Great Barrier Reef Catchment Loads Monitoring Program. A description of the methods used for sample collection and quality acceptance / quality control is reported in Wallace et al. (2016). The risk assessment was conducted using monitoring data of five photosystem II herbicides (ametryn, atrazine, diuron, hexazinone and tebuthiuron) collected over six years (2010-2016). Exceedances of the ecotoxicity threshold values were examined using monitoring data of up to 28 pesticides (for which an ecotoxicity threshold value was available) collected over a three-year period (2013-2016), as many of these pesticides were not analysed prior to 2013.

The risk assessment was performed using two methods that assess consequence and likelihood. Consequence was first determined using the multisubstance-Potentially Affected Fraction (ms-PAF) method. Likelihood could then be determined using methods of a probabilistic ecological risk assessment: the area under the curve of the ms-PAF cumulative frequency distribution. Both approaches are discussed in more detail below.

(1) Multisubstance-Potentially Affected Fraction method

The ms-PAF method, originally described by Traas et al. (2002), allows for the estimation of the effect of multiple pollutants on an ecosystem. Species sensitivity distributions form the basis of the method, similar to what is used to generate the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000) for ecosystem protection. Traas et al. (2002) describe two approaches: concentration addition for pollutants with the same mode of action and response addition for pollutants with different modes of action. The ms-PAF method used here had four modifications from the concentration addition method: (i) a logistic cumulative distribution function, rather than a log-logistic cumulative distribution function, was fitted to the ecotoxicity data of individual pollutants in order to generate the individual species sensitivity distributions; (ii) the midpoint of the curve (α) when y = 50 was used rather than the median to calculate the hazard units; (iii) the curve parameters, α and β, of the cumulative distribution function used to calculate the potentially affected fraction of the mixture were calculated using a global[2] cumulative distribution function using SigmaPlot 13.0 (Systat) rather than assuming α = 1, and β was calculated from the average of the βs from the individual pollutant cumulative distribution functions; and (iv) the global α and β were calculated from 13 different PSII herbicides (ametryn, atrazine, diuron, hexazinone, tebuthiuron, simazine, bromacil, fluometuron, metribuzin, terbuthylazine, terbutryn, prometryn, propazine) rather than just the five PSII herbicides in question. Tests (not presented here) demonstrated that these four modifications provided greater accuracy in the ms-PAF estimations.

Ecotoxicity data were obtained from both marine and freshwater species sensitivity distributions generated for deriving pesticide water quality guidelines (King et al., 2017a; King et al., 2017b; Warne et al., in press). It should be noted that the marine and freshwater species were combined to generate single chemical species sensitivity distributions for the ms-PAF calculations. The toxicity data for 13 PSII herbicides were each fitted with a logistic cumulative distribution function and α was calculated to convert the toxicity data to hazard units, such that the toxicity data of all herbicides were on a relative scale with the midpoint of the curve sitting at x = 1. The species sensitivity distributions of the 13 PSII herbicides were then combined and one global logistic cumulative distribution function was fitted to the whole dataset. From the global cumulative distribution function, α_G and β_G were calculated. The ms-PAF value, based on the concentrations of ametryn, atrazine, diuron, hexazinone and tebuthiuron in each sample, could then be calculated according to Traas et al. (2002):

$$HU_i = \frac{C_i}{\alpha_i} \qquad \textbf{Equation 1}$$

where HU_i is the hazard unit for chemical *i*, C_i is the concentration of chemical *i* in a sample and α_i is the concentration of chemical *i* at which 50% of species are affected (calculated from the species sensitivity distribution of chemical *i*). The *HU* values of the mixture constituents are then summed, resulting in a hazard unit for the mixture (HU_{mix}) (Equation 6).

$$HU_{mix} = \sum_i HU_i \qquad \textbf{Equation 2}$$

The ms-PAF was then calculated for each sample:

$$msPAF = \frac{1}{1+\frac{HU_{mix}}{\alpha_G}^{-\beta_G}} \times 100 \qquad \textbf{Equation 3}$$

[2] The global regression function in SigmaPlot 13.0 (Systat) allows one regression function to be fitted to data from multiple variables. This allowed us to fit one logistic cumulative distribution function to the ecotoxicity data from all pollutants combined and therefore generate the required α and β parameters that represented the combined data. This is different from Traas et al. (2002), in which it was assumed that α = 1, and β was determined for each herbicide in the mixture and then the average β was calculated from these.

Where α_G is the midpoint of the global cumulative distribution function curve and β_G is the slope of the global fit curve.

(2) *Probabilistic ecological risk assessment*

Probabilistic ecological risk assessments are species sensitivity distributions combined with contaminant concentration distributions to describe the likelihood of exceedences of effect thresholds and thus the risk of adverse effects (Solomon and Takacs, 2001; Aldenberg et al., 2002). For each site and year (2010-2016), the concentrations of ametryn, atrazine, diuron, hexazinone and tebuthiuron in each sample were used to calculate an ms-PAF value. The daily average was calculated when multiple samples were collected on the same day. The cumulative distribution of ms-PAF values for samples collected within a 182-day period (i.e. over the wet season) between 1 November and 30 April was generated and fitted with an inverse log-normal distribution. The area under the inverse log-normal curve was calculated and divided by 182 days.

Procedures 1 and 2 were used to calculate an ms-PAF risk metric, which estimates the average number of species potentially affected over the annual wet season. This risk metric alone was used in the risk assessment for the freshwater and estuarine ecosystems and was categorised according to the protection levels of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000), as presented in Table 19.

Table 19. Risk categories for freshwater and estuarine ecosystems based on the ms-PAF risk metric (average per cent of species affected during the wet season) and the equivalent ANZECC and ARMCANZ (2000) protection levels.

Risk category	ms-PAF risk metric (average % species affected)	ANZECC & ARMCANZ (2000)	
		Protection level	Ecosystem value
Very High	≥20	≤80% species protection	Highly disturbed
High	10–19	80% species protection	Highly disturbed
Moderate	5–9	90% species protection	Highly disturbed
Low	1–4	95% species protection	Slightly – moderately disturbed
Very Low	<1	>99% species protection	High ecological value

(3) *Exceedance of ecotoxicity threshold values*

To assess whether pesticides other than the five PSII herbicides used in the risk assessment may pose a hazard to the freshwater, estuarine and marine ecosystems, exceedances of their ecotoxicity threshold values were evaluated. The 95th percentile concentrations, calculated for samples collected during each monitoring year (1 July – 30 June) from 2013 to 2016, of pesticides were compared with their respective ecotoxicity threshold values (Table 20). For the majority of pesticides reported in Table 20, both freshwater and marine ecotoxicity threshold values are provided. For the purpose of conservation, and because the objective is to protect freshwater, estuarine and marine ecosystems, the most conservative ecotoxicity threshold value (freshwater or marine) was used to compare against the 95th percentile concentration. The 95th percentile concentrations were then categorised according to the risk categories presented in Table 20, based on the ANZECC and ARMCANZ (2000) protection levels. For the catchments and estuarine ecosystems, a protective concentration that will protect 95% of species (PC95—Low risk category) is applied. For the Great Barrier Reef World Heritage Area and other ecosystems of high ecological value (e.g. Bowling Green Bay Ramsar wetland in the Lower Burdekin), a protective concentration that will protect at least 99% of species (PC99—Very Low risk category) was applied.

Table 20. Proposed ecotoxicity threshold values for pesticides in freshwater and marine ecosystems. (Source: King et al. 2017a; King et al., 2017b; Warne et al., in press.) The values presented are the estimated protective concentrations that will protect at least 99% (PC99), 95% (PC95), 90% (PC90) or 80% (PC80) of species. The reliability of the ecotoxicity threshold values is also provided as Very High (VH), High (H), Moderate (M) or Low (L).

Pesticide	Freshwater (μg/L)					Marine (μg/L)				
	Reliability	PC99	PC95	PC90	PC80	Reliability	PC99	PC95	PC90	PC80
Herbicide										
2,4-D	Data not available					L	1,040	2,516	3,751	5,788
Ametryn	M	0.074	0.33	0.66	1.4	M	0.1	0.61	1.3	2.8
Bromacil	L	1.6	3.6	5.2	7.7	M	0.23	1.1	2.2	4.8
Diuron	VH	0.08	0.23	0.42	0.9	VH	0.43	0.67	0.86	1.2
Fluometuron*	M	20	40	55	77	M	20	40	55	77
Fluroxypyr*	M	87	200	291	437	M	87	200	291	437
Glyphosate	M	144	246	341	532	Data not available				
Haloxyfop*	L	589	1,969	3,399	6,147	L	589	1,969	3,399	6,147
Hexazinone	L	0.31	1.1	1.9	3.4	L	1.8	2.5	3.1	4
Imazapic*	VL	0.036	0.41	1.2	4	VL	0.049	0.44	1.2	3.6
Isoxaflutole	L	0.068	0.46	1.1	2.8	M	0.33	0.69	1.1	2
MCPA						L	1	17	60	241
Metolachlor	VH	0.016	0.71	3.7	19					
Metribuzin*	VH	2	2.6	3.1	3.9	M	2	2.7	3.1	3.9
Metsulfuron-methyl	M	0.0047	0.025	0.069	0.28	Data not available				
Pendimethalin	M	1.3	2.1	2.9	4.5	M	0.24	0.97	1.9	4.1
Prometryn	L	0.094	0.49	1	2.3	M	0.11	0.52	1.1	2.2
Propazine	L	1.3	3.1	4.5	6.8	L	2.2	4.6	6.4	9.2
Simazine	H	3.2	10	17	29	L	28	63	89	132
Tebuthiuron	M	4.8	13	19	31	M	4.7	11	17	26
Terbuthylazine	VH	0.43	1.2	2	3.8	M	0.4	0.97	1.6	2.8
Terbutryn	M	0.079	0.26	0.51	1.2	M	0.079	0.26	0.51	1.2
Triclopyr	L	1.6	6.4	12	24	L	0.36	4	11	32
Fungicide										
Chlorothalonil	H	0.24	0.48	0.74	1.3	M	0.34	1	1.7	2.9
Propiconazole	M	3.7	10	18	35	M	2.1	8.2	15	30
Insecticide										
Fipronil	Data not available					M	0.0034	0.0089	0.016	0.033
Imidacloprid	M	0.025	0.074	0.14	0.34	M	0.034	0.099	0.19	0.45

*pesticides with only one set of guideline values for both fresh and marine ecosystems.

Risk assessment

Results for the risk assessment of the five PSII herbicides at the end-of-system monitored sites are presented in Table 21. Two regions, Mackay Whitsunday and the Lower Burdekin, had the highest ms-PAF risk scores, which fell within the Very High risk category. These occurred at Sandy Creek in three consecutive years (2014- 2016) and Barratta Creek (2013-2014). All other years in Sandy Creek were considered High risk, whereas in Barratta Creek, all other years were High or Moderate. Pioneer River in the Mackay Whitsunday

had the next highest ms-PAF risk scores, ranging from High to Moderate. The only other catchment to score a High risk was O'Connell, also in the Mackay Whitsunday region, with one year scored as High and the other two years scored as Good. In the Wet Tropics, Tully was scored as Moderate to Good with all other catchments scored as Very Good or Good. The Fitzroy and Burnett Mary catchments scored Very Good or Good.

For the freshwater and estuarine ecosystems, a score of Good is required for the protection of these ecosystems as they are considered to be slightly to moderately disturbed ecosystems (Table 19). However, for the Great Barrier Reef World Heritage Area a score of Very Good is required as it is a high ecological value system (Table 19). Thus, these results indicate a risk from pesticides in five catchments for the marine ecosystems (Sandy Creek, Barratta Creek, Pioneer River, O'Connell River and Tully River) and the marine ecosystems in the majority of catchments are potentially at risk from pesticides, based on the point in the catchment that was monitored. It should be acknowledged that only a proportion of all catchments have been assessed here due to the limitations of the monitoring data. A number of other catchments, particularly the smaller coastal catchments with high proportions of intensive cropping, are likely to also have a Very High to Moderate ecological risk from pesticides. For instance, in the Mackay Whitsunday region, a number of other small catchments in the Plane Basin have similar or higher concentrations of PSII herbicides as Sandy Creek (Folkers et al., 2014). These small catchments in the Plane Basin may seem insignificant on their own in terms of their impact to the marine area, but when combined they make up approximately one-third of the total discharge of the Mackay Whitsunday region. This is further substantiated by results from sub-catchment monitoring in Sandy Creek (Wallace et al., 2017) and the Herbert River (O'Brien et al., 2013).

The monitoring sites in the catchments are generally located either at, or downstream of, the upper extent of the tidal zone (Wallace et al., 2016); therefore, the ms-PAF risk score is a good reflection of the ecological risk from pesticides in these ecosystems. Ecological risk is likely to be higher than the reported ms-PAF risk score upstream or downstream of the monitoring sites where the proportion of intensive cropping land use increases, particularly sugarcane. Wallace et al. (2017) demonstrated that the concentrations of PSII herbicides are much higher in sub-catchments upstream of the Sandy Creek at Homebush monitoring site, but also demonstrated that concentrations in the main channel remained steady for at least 22 km downstream of the Homebush site and into the tidal zone. O'Brien et al. (2016a) also demonstrated increased concentrations of PSII herbicides in sub-catchments upstream of the Herbert River at Ingham monitoring site, that is, where the upstream land use area in the sub-catchments has a higher proportion of sugarcane than the proportion of sugarcane in the Herbert River catchment upstream of Ingham. The Johnstone River sub-catchment monitoring site (North Johnstone River at Tung Oil) is located upstream of the majority of the sugarcane land use in the Johnstone catchment; as such, the ms-PAF risk score was higher at the end-of-catchment monitoring site (Johnstone River at Coquette), which captures a much larger area of the sugarcane land use in the Johnstone catchment (Table 21). In most cases, it is expected that the ecological risk from pesticides would decrease towards the mouth of the catchment due to dilution from run-off from other land sources and tidal flushing. This was demonstrated at Barratta Creek (O'Brien et al., 2016a) where PSII herbicide concentrations decreased towards the end of catchment.

Table 21. Risk assessment of five PSII herbicide mixtures to monitored rivers and estuaries based on the ms-PAF risk metric, which accounts for the magnitude of exposure on a temporal scale. Values represent the average per cent of species affected during the wet season**.**

Wet Season	Wet Tropics						Burdekin			Mackay Whitsunday			Fitzroy*	Burnett Mary	
			Johnstone				Lower Burdekin								
	Russell	Mulgrave	North	Coquette	Tully	Herbert	Haughton	Barratta	Burdekin	O'Connell	Pioneer	Sandy	Fitzroy	Burnett	Mary
2010-2011	ND	ND	0.3	ND	2	3	ND	7	0.8	ND	8	18	1	0.4	ND
2011-2012	ND	ND	0.2	ND	3	0.7	ND	12	0.3	ND	11	16	0.4	1	ND
2012-2013	ND	ND	0.2	ND	5	3	ND	17	0.5	ND	13	19	1	1	ND
2013-2014	3	0.9	0.5	ND	3	1	0.4	25	0.9	4	16	24	0.4	0.3	0.2
2014-2015	7	4	0.7	ND	5	2	1	18	0.7	10	14	37	1	0.7	0.8
2015-2016	4	3	0.5	2	6	2	4	17	0.3	2	13	27	1	0.6	0.5

ND = No monitoring data available.

Exceedances of ecotoxicity threshold values

Of the 39 pesticides (and metabolites) that were detected above the limits of reporting in monitored catchments between 2013 and 2016, 21 were compared against their respective ecotoxicity threshold values, including the five PSII herbicides included in the ms-PAF risk metric. Similar patterns persisted across the three years reported here (Table 22–Table 24):

1. Diuron and imidacloprid were the only two pesticides with scores in the Very High risk category (the 95th percentile concentrations were higher than the PC80 ecotoxicity threshold values).
2. Similar to the risk assessment, Very High scores principally occurred at Pioneer River and Sandy Creek in the Mackay Whitsunday region and Barratta Creek in the Lower Burdekin (Table 23); the exception was a Very High score of imidacloprid in the Fitzroy in 2014-2015 (Table 24).
3. The 95th percentile concentration exceeded the PC95 ecotoxicity threshold values (Moderate risk) for ametryn, hexazinone, imazapic and metolachlor..
4. Sandy Creek had the highest number of pesticides (six) recorded as Moderate risk or higher, followed by Barratta Creek (five) and Pioneer River (three) (Table 23).

Diuron was also recorded as High risk (the 95th percentile concentrations were ≥PC90 in the Tully (Table 22) and Haughton Rivers in 2015-2016 (Table 23) and the Russell and O'Connell Rivers in 2014-2015 (Table 23). Imidacloprid was recorded as High (≥PC90) for the North Johnstone and Johnstone at Coquette, Tully, Herbert, O'Connell and Pioneer Rivers in 2015-2016, Tully and O'Connell Rivers in 2014-2015 and Tully, O'Connell and Pioneer rivers in 2013-2014 (Table 22 and Table 23).

Monitored concentrations of the majority of pesticides did not exceed the PC99 and/or PC95 ecotoxicity threshold values; however, what should be noted is the total number of pesticides detected. The reason for this is that the small effect from each pesticide is cumulative, resulting in a higher overall risk from the mixture as a whole. Other pesticides and pesticide metabolites also detected but not reported in Table 22–Table 24 (i.e. because they do not have assigned ecotoxicity threshold values) included 3,4-dichloroaniline, acetamiprid, acifluorfen, AMPA, atrazine, clomazone, clothianidin, desethyl atrazine, desisopropyl atrazine, imazapyr, imazethapyr, imidacloprid metabolites, MCPB, methoxyfenozide, N-demethyl acetamiprid, propazin-2-hydroxy, thiacloprid and terbuthylazine desethyl.

Table 22. Exceedances of ecotoxicity threshold values for individual pesticides in catchments of the Wet Tropics region between 2013 and 2016. Exceedances were determined using the annual 95th percentile concentration from samples collected during 2013-2014 (2013), 2014-2015 (2014) and 2015-2016 (2015) monitoring years. Risk categories: dark green = >99% species protection, light green = 95% species protection, yellow = 90% species protection, orange = 80% species protection, red = <80% species protection, grey = no analysis, white = < limit of reporting (LOR). Where the 95th percentile concentration was <LOR, but the pesticide was detected (above the LOR) in ≤ 5% of samples, the cell was shaded the risk category colour dependent on the LOR concentration relative to the ETV.

Pesticides	Mulgrave River			Russell River			Johnstone River* North Johnstone			Johnstone River* Coquette			Tully River			Herbert River		
	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015
2,4-D																		
Ametryn																		
Bromacil																		
Diuron																		
Fluometuron													<LOR					
Fluroxypyr																		
Glyphosate																		
Haloxyfop													<LOR					
Hexazinone																		
Imazapic																		
Isoxaflutole																		
MCPA																		
Metolachlor							<LOR											
Metribuzin																		
Metsulfuron-methyl							<LOR						<LOR	<LOR				
Prometryn																		
Simazine																		
Tebuthiuron																		
Terbuthylazine																		
Triclopyr																		
Imidacloprid																		

Table 23. Exceedances of ecotoxicity threshold values for individual pesticides in catchments of the Burdekin and Mackay Whitsunday regions between 2013 and 2016. Exceedances were determined using the annual 95th percentile concentration from samples collected during 2013-2014 (2013), 2014-2015 (2014) and 2015-2016 (2015) monitoring years. Risk categories: dark green = >99% species protection, light green = 95% species protection, yellow = 90% species protection, orange = 80% species protection, red = <80% species protection, grey = no analysis, white = < limit of reporting (LOR). Where the 95[th] percentile concentration was <LOR, but the pesticide was detected (above the LOR) in ≤ 5% of samples, the cell was shaded the risk category colour dependent on the LOR concentration relative to the ETV.

Pesticides	Burdekin region									Mackay Whitsunday region								
	Haughton			Barratta			Burdekin			O'Connell			Pioneer			Sandy		
	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015	2013	2014	2015
2,4-D																		
Ametryn										<LOR								
Bromacil																		
Diuron																		
Fluometuron																		
Fluroxypyr																		
Glyphosate																		
Haloxyfop																		
Hexazinone							<LOR											
Imazapic																		
Isoxaflutole																		
MCPA																		
Metolachlor																		
Metribuzin																		
Metsulfuron-methyl	<LOR				<LOR													
Prometryn																		
Simazine																<LOR		
Tebuthiuron																		
Terbuthylazine																		
Triclopyr																		
Imidacloprid	<LOR																	

Table 24. Exceedances of ecotoxicity threshold values for individual pesticides in catchments of the Fitzroy and Burnett Mary regions between 2013 and 2016. Exceedances were determined using the annual 95th percentile concentration from samples collected during 2013-2014 (2013), 2014-2015 (2014) and 2015-2016 (2015) monitoring years. Risk categories: dark green = >99% species protection, light green = 95% species protection, yellow = 90% species protection, orange = 80% species protection, red = <80% species protection, grey = no analysis, white = < limit of reporting (LOR).

Pesticides	Fitzroy region			Burnett Mary region					
	Fitzroy			Burnett			Mary		
	2013	2014	2015	2013	2014	2015	2013	2014	2015
2,4-D									
Ametryn									
Bromacil									
Diuron									
Fluometuron									
Fluroxypyr									
Glyphosate									
Haloxyfop									
Hexazinone									
Imazapic									
Isoxaflutole									
MCPA									
Metolachlor							<LOR		
Metribuzin									
Metsulfuron-methyl									
Prometryn									
Simazine							<LOR		
Tebuthiuron									
Terbuthylazine									
Triclopyr									
Imidacloprid									

8.1.4 Other pollutants

Pollutants other than sediment, nutrients and pesticides are known to be present in the coastal and marine waters of the Great Barrier Reef (Chapter 1). This includes pollutants such as antifouling paints, coal particles, metals and metalloids, marine debris/microplastics, personal care products, petroleum hydrocarbons, and pharmaceuticals. In addition, pollutants such as nanomaterials, perfluorooctane sulfonate and perfluorooctanoic acid may be present, but no monitoring information is available for the Great Barrier Reef lagoon (Kroon et al., 2015a).

Kroon et al. (2015a) assessed the qualitative risk for nine classes of emerging pollutants for each of the natural resource management NRM regions separately, using an established risk assessment framework considering 'Likelihood' and 'Consequence' (refer to Great Barrier Reef Outlook 2014 report; GBRMPA, 2014). 'Likelihood' gives an indication of the expected frequency of a given threat, ranging on a scale from 1 ('not expected to occur within the next 100 years') to 5 ('expected to occur more or less continuously throughout a year'). 'Consequence' gives an indication of the impact to the ecosystem at local and broad scale, as well as to the heritage value, based on current management, ranging on a scale from 1 ('insignificant') to 5 ('catastrophic'). Preliminary risk assessments revealed that only using 'Likelihood' and 'Consequence' would not separate out the nine emerging pollutants for prioritisation. Hence, it was agreed to include an additional assessment category, namely 'Area' to include the number of water bodies the risk may occur in and allow better separation for prioritisation purposes. 'Area' ranges from 1 ('any one water body) to 4 ('all four water bodies').

The qualitative risk assessments were conducted by the Team Lead and associated project team members for each emerging pollutant. The risk assessment scores for all nine emerging contaminants were subsequently compared by the Project Leader and adjusted in case of inconsistencies. For example, emerging contaminants discharged from sewage treatment plants (e.g. pharmaceuticals, personal care products) can be expected to have similar categories for 'Area'. Similarly, the 'Likelihood' of emerging contaminants released during ship collision and/or grounding (e.g. petroleum hydrocarbons, coal particles, antifouling paints) would be similar.

The final qualitative risk assessment scores for each of the nine emerging contaminants were compared for each of the NRM regions separately and used to develop a list of priority emerging contaminants based on the risk to the Great Barrier Reef and Torres Strait marine ecosystems.

The qualitative risk assessment conducted by Kroon et al. (2015a) determined that of the nine classes of emerging contaminants, marine plastic pollution poses the highest risk to the Great Barrier Reef marine ecosystems, particularly in the Cape York NRM region due to exposure to oceanic and local shipping sources. This is followed by chronic contamination of water and sediments with antifouling paints and exposure to certain personal care products in natural resource management regions south of Cape York. The qualitative risks of all other emerging pollutants are relatively low with some minor differences between NRM regions.

To inform management of and policy for emerging pollutants in the Great Barrier Reef marine environments, Kroon et al. (2015a) recommended the following key areas of research:

- ensuring availability of valuable existing environmental data in the public domain for building marine baselines on emerging pollutants in the study region;
- conducting local, targeted monitoring campaigns for priority emerging contaminants with little or no recent monitoring data for the Great Barrier Reef and Torres Strait regions; and
- examining the ecological impacts of marine plastic pollution, chronic contamination of antifouling paints and certain personal care products on Great Barrier Reef marine organisms and ecosystems.

9. Synthesis of key findings

A summary of each of the assessments for each parameter for each habitat is presented below.

9.1 Relative priorities between basins and pollutants

Table 25 summarises all of the results of the likelihood of exposure assessments to assist in the identification of patterns among basins and between pollutants. While the Index scores vary between basins and pollutants, the rankings are relatively consistent, with several basins identified as high exposure for two or more pollutants. These include the *Russell-Mulgrave, Johnstone, Tully, Herbert, Haughton, Burdekin, Pioneer, Plane, Fitzroy and Mary basins*.

The assessment of the *likelihood of exposure of coral reefs to DIN* indicates that the *Herbert, Haughton, Johnstone, Russell-Mulgrave and Tully basins* are the highest contributors across the Great Barrier Reef. The example of the assessment of *DIN Risk (CoTs)* also indicates that coral reefs in the Wet Tropics Marine Zone are at greatest risk from DIN exposure, followed by the Cape York South Marine Zone and, to a lesser extent, the Burdekin Marine Zone. At a basin scale the assessment emphasises the contribution of the *Herbert Basin to the DIN risk to coral reefs through Crown-of-Thorns starfish influence, followed by the Johnstone, Russell-Mulgrave and Tully basins*. This should be taken into account when defining management priorities across the Great Barrier Reef.

The assessment of the *likelihood of exposure of coral reefs and seagrass to TSS* shows that the *Burdekin, Fitzroy and Herbert basins* are the highest contributors to coral reef and seagrass exposure across the Great Barrier Reef. The example of the assessment of *TSS Risk (seagrass benthic light)* indicates that the greatest risk to seagrass from TSS exposure is in the Burdekin Marine Zone, followed by the Burnett Mary Marine Zone. At a basin scale the assessment emphasises the contribution of the *Burdekin and Fitzroy basins* to the TSS risk to seagrass.

The assessment of the *average ms-PAF risk* shows that the *Plane, Haughton and Pioneer basins* rank the highest.

Floodplain wetlands in six management units / basins—the *Dawson, Lower Burdekin, Herbert, Burnett, Burrum and Tully*—have *high likelihood of exposure to sediment, nutrient and pesticide pressures* (Section 6.3). The areas of greatest likelihood of exposure of floodplain wetlands to nutrient pressures are in the Fitzroy and Dawson; for sediments it is the Dawson and Lower Burdekin; and for pesticides the Lower Burdekin and Herbert basins.

Floodplains in seven management units / basins—*Tully, Belyando, Plane, Dawson, Comet, Kolan and Burnett*—have *high likelihood of exposure to sediments, nutrients and pesticides*. The areas of greatest likelihood of exposure of floodplains to nutrient inputs are in the Belyando and Dawson; for sediments it is the Dawson, Isaac and Mackenzie; and for pesticides the Herbert, Lower Burdekin, Belyando, Pioneer and Plane basins.

From these results, the following conclusions can be drawn regarding the likelihood of exposure of anthropogenic pollutant loads to coastal aquatic and marine ecosystems and the relative importance of pollutants to these ecosystems.

Fine sediment

- Exposure to fine sediment is most significant to areas of shallow seagrass and coral reefs on the inner shelf adjacent to basins with high anthropogenic fine sediment loads.
- The greatest exposure of coral reef and seagrass to fine sediment is from the Burdekin, Fitzroy, Mary, Herbert, Johnstone and Burnett Basins. The Burdekin and Fitzroy basins also contribute the greatest fine sediment risk to seagrass ecosystems.

- The Dawson, Isaac and Mackenzie basins contribute the greatest exposure of floodplain wetland ecosystem to sediment pressures. The Dawson and Lower Burdekin contribute the greatest exposure of floodplain ecosystems to sediment pressures.

Nitrogen

- Exposure to DIN is significant to all inner shelf areas and the mid-shelf area between Lizard Island and Townsville adjacent to basins with high anthropogenic DIN loads. The relative importance of DIN to seagrass ecosystems is still uncertain, but it may influence light availability for deepwater seagrass in areas deeper than 10–12 m due to increased phytoplankton growth.
- The greatest exposure of coral reefs and seagrass to DIN is from the Herbert, Haughton, Johnstone, Russell-Mulgrave, Tully, Plane and Murray basins. The Herbert, Johnstone, Russell-Mulgrave and Tully basins also contribute the greatest DIN risk to coral reefs and primary Crown-of-Thorns starfish outbreaks.
- Anthropogenic particulate nitrogen is also likely to be of some importance in the same areas, as well as in the Fitzroy Basin; however, our knowledge on the bioavailability of particulate nitrogen to the marine ecosystems relative to that of DIN is still limited.
- Given the small anthropogenic loads of dissolved organic nitrogen from most basins, and its limited bioavailablity, it is considered to be less important than DIN.
- The Dawson and Lower Fitzroy management units contribute the greatest exposure of floodplain wetland ecosystem to nutrient pressures. The Belyando and Dawson contribute the greatest exposure of floodplain ecosystems to nutrient pressures.

Phosphorus

- Anthropogenic phosphorus loads are considerable from many basins, and although our knowledge of the relative importance of nitrogen and phosphorus is still limited, nitrogen is considered to be the limiting nutrient and hence more important than phosphorus. Hence, phosphorus is not considered to be as important as nitrogen in any form.

Pesticides

- Only a few basins present a Very High to Moderate risk to end-of-catchment ecosystems from PSII herbicides, with diuron presenting the highest risk. These basins are generally characterised as smaller coastal catchments with high proportions of sugarcane land use (i.e. basins within the Mackay Whitsunday region, Lower Burdekin and Wet Tropics).
- While the risk assessment only assessed the concentration and temporal exposure of five PSII herbicides, developments in our understanding of the ecotoxicity of other pesticides detected in Great Barrier Reef catchments has allowed us to examine other pesticides individually.
- The ecotoxicity threshold assessment demonstrated that Great Barrier Reef ecosystems are exposed to a large number of other types of pesticides, some of which were a high risk on their own. Of the pesticides that indicated a risk to ecosystems (i.e. <95% species protection), imidacloprid had a Very High to Moderate risk in a number of basins, and hexazinone, metolachlor and imazapic had a High to Moderate risk in some basins. Including all pesticides in the ms-PAF risk metric in future risk assessments will provide a more accurate assessment of the potential risk pesticides have to these ecosystems.
- Pesticides pose the greatest risk to ecosystems closest to the source of the pesticides; that is, freshwater wetlands, rivers and estuaries are exposed to the highest concentrations, followed by coastal ecosystems, seagrass and coral reefs. Our understanding, at this stage, of the spatial exposure of pesticides in the marine environment is very limited.

- The Herbert and Lower Burdekin contribute the greatest exposure of floodplain wetland ecosystems to pesticide pressures. The Herbert, Lower Burdekin, Belyando, Pioneer and Plane contribute the greatest exposure of floodplain ecosystems to pesticide pressures.

Other pollutants

- In a qualitative risk assessment of pollutants other than sediment, nutrients and pesticides, marine plastic pollution poses the highest relative risk to the Great Barrier Reef marine ecosystems, particularly in the Cape York NRM region due to exposure to oceanic and local shipping sources. This is followed by chronic contamination of water and sediments with antifouling paint components and exposure to certain personal care products in natural resource managment regions south of Cape York. The relative risks of other pollutants are likely to be relatively low with some minor differences between natural resource management regions.

Significant data limitations exist in the Cape York region; therefore, it is difficult to make conclusions about this region with confidence. There is enough evidence to conclude that, overall, the eastern Cape York catchments currently present a relatively low risk to coral reef and seagrass ecosystems in the Great Barrier Reef and that the ecosystems in the region are typically in good condition (see Waterhouse et al., 2015c). The basins in the Cape York Central Marine Zone—the Normanby, Hann and Stewart catchments—are likely to pose a risk to ecosystems in the Princess Charlotte Bay area from degraded water quality, particularly increased turbidity in wet season conditions.

Due to the potential underestimation and lack of validation of models pertaining to risks in the Cape York South Marine Zone, this region also warrants further investigation and management of threats to water quality. In particular, high levels of impacts have been documented from current and historic land uses in the Annan and Endeavour catchments, and increased levels of disturbance from urban and semi-urban development are expected in this area (Howley et al., 2012; Shellberg et al., 2016). Reefs and seagrass meadows in this region are regularly inundated by flood plumes that may contain high levels of suspended sediments (Davies and Hughes, 1983; Davies and Eyre, 2005).

Other threats to water quality in the eastern Cape York region should also be considered and include shipping traffic, particularly on the inner route, which may pose an increasing risk to the region with predicted increases in traffic.

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 and pollutant loads for some variables in the model. However, the results do correlate with current status reported in Coppo et al. (2015) and Chapter 1 (Schaffelke et al., 2017).

It is important to note that many mid- and outer shelf parts of the Great Barrier Reef are not impacted to any extent by terrestrial run-off impacts. This is particularly true for the Torres Strait, Cape York, the Pompeys and The Swains which form a large part of the area of the Great Barrier Reef. The main mid- and outer shelf reef locations directly impacted by terrestrial run-off are those regions located between Lizard Island and Townsville.

Table 25. Summary of the likelihood of exposure and risk assessments for coastal aquatic and marine ecosystems for sediments, nutrients and pesticides. The shading represents relative classes of high to very low exposure. For marine ecosystems: red classes = very high, orange = high, yellow = moderate, light green = low, dark green = very low. For coastal ecosystems: scores are based on the relative proportion (%) of Great Barrier Reef floodplain wetland area exposed to High and Very High nutrient, sediment or pesticide hazard; red: >10%, orange: 5–9%, yellow: 1–4%. Grey shading indicates areas that are not included in the assessment.

Management unit	Marine ecosystems: Likelihood of exposure						Freshwater and estuarine		Coastal ecosystems: Likelihood of exposure					
							Pesticide risk		Floodplain wetlands			Floodplains		
	DIN Index within Marine Zone	DIN Likelihood Index (reefs)	Rank	TSS Index within Marine Zone	TSS Likelihood Index (seagrass + reefs)	Rank	Average ms-PAF Risk: per cent species affected (2013-2016)	Rank	Nutrient	Sediment	Pesticide	Nutrient	Sediment	Pesticide
Jacky Jacky Creek	0.09	0.00	29	0.80	0.00	29			0	0	0	0	0	0
Olive Pascoe River	1.00	0.00	29	1.00	0.00	29			0	0	0	0	0	0
Lockhart River	0.02	0.00	29	1.00	0.00	29			0	0	0	0	0	0
Stewart River	0.01	0.00	29	0.27	0.00	29			0	0	0	0	0	0
Normanby River	0.01	0.00	29	0.27	0.00	29			3	0	0	<1	<1	0
Jeannie River	0.01	0.00	29	1.00	0.00	29			0	0	0	0	0	0
Endeavour River	0.04	0.00	29	0.89	0.00	29			0	0	0	0	0	0
Daintree River	0.15	0.15	13	0.08	0.02	19			0	0	0	0	0	0
Mossman River	0.12	0.12	16	0.02	0.00	28						0	0	0
Barron River	0.10	0.10	17	0.10	0.02	17			<1	<1	0	<1	<1	1
Mulgrave-Russell River	0.48	0.48	4	0.47	0.11	7	3.7	6	0	0	0	0	0	0
Johnstone River	0.56	0.56	3	0.79	0.19	5	1.3	8	1	<1	1	<1	<1	2
Tully River	0.43	0.43	5	0.25	0.06	10	5	5	2	1	7	1	1	5
Murray River	0.26	0.26	7	0.12	0.03	15			2	3	2	<1	1	2
Herbert River	1.00	1.00	1	1.00	0.24	3	2	7	7	5	23	2	2	14
Black River	0.02	0.02	22	0.01	0.01	24			<1	0	0	<1	0	0
Ross River	0.13	0.13	15	0.02	0.02	22			<1	0	0	1	0	0
Haughton River	1.00	0.93	2	0.06	0.06	12	10.9	3						
Burdekin River	0.19	0.17	11	1.00	1.00	1	0.6	10						

Management unit	Marine ecosystems: Likelihood of exposure						Freshwater and estuarine		Coastal ecosystems: Likelihood of exposure					
							Pesticide risk		Floodplain wetlands			Floodplains		
	DIN Index within Marine Zone	DIN Likelihood Index (reefs)	Rank	TSS Index within Marine Zone	TSS Likelihood Index (seagrass + reefs)	Rank	Average ms-PAF Risk: per cent species affected (2013-2016)	Rank	Nutrient	Sediment	Pesticide	Nutrient	Sediment	Pesticide
Burdekin - East									<1	1	0	<1	<1	0
Burdekin - Lower Burdekin									7	14	43	6	9	33
Burdekin - Belyando									4	0	0	12	1	0
Burdekin - Suttor									1	<1	0	8	7	<1
Burdekin - Upper									5	1	0	4	1	0
Burdekin - Bowen Bogie									<1	0	0	<1	<1	0
Burdekin - Cape Campaspe									2	2	0	4	1	0
Don River	0.07	0.07	20	0.07	0.07	9			<1	<1	<1	<1	<1	<1
Proserpine River	0.43	0.18	10	0.06	0.03	16			<1	0	0	<1	<1	1
O'Connell River	0.51	0.21	9	0.19	0.08	8	5.3	4	<1	<1	0	<1	<1	<1
Pioneer River	0.53	0.22	8	0.13	0.06	11	13.5	2	1	<1	2	1	<1	10
Plane Creek	1.00	0.41	6	0.08	0.03	13	29.3	1	1	1	3	3	4	16
Styx River	0.02	0.01	24	0.07	0.03	14			0	0	0	0	0	0
Shoalwater Creek	0.01	0.00	26	0.05	0.02	18			0	0	0	0	0	0
Waterpark Creek	0.01	0.00	27	0.04	0.02	20			0	0	0	0	0	0
Fitzroy River	0.36	0.15	14	1.00	0.46	2	0.8	9						
Fitzroy - Lower Fitzroy									10	2	0	3	2	0
Fitzroy - Dawson									23	38	2	16	26	3
Fitzroy - Isaac									4	9	0	7	11	0
Fitzroy - Mackenzie									3	7	0	6	11	0
Fitzroy - Comet									3	5	<1	6	9	7
Fiztroy - Nogoa									4	3	0	4	4	<1

Management unit	Marine ecosystems: Likelihood of exposure						Freshwater and estuarine		Coastal ecosystems: Likelihood of exposure					
							Pesticide risk		Floodplain wetlands			Floodplains		
	DIN Index within Marine Zone	DIN Likelihood Index (reefs)	Rank	TSS Index within Marine Zone	TSS Likelihood Index (seagrass + reefs)	Rank	Average ms-PAF Risk: per cent species affected (2013-2016)	Rank	Nutrient	Sediment	Pesticide	Nutrient	Sediment	Pesticide
Fitzroy - Theresa Creek									<1	<1	0	2	2	0
Calliope River	0.01	0.01	25	0.04	0.02	23			<1	0	0	1	<1	0
Boyne River	0.01	0.00	28	0.01	0.01	27			0	1	0	<1	<1	0
Baffle Creek	0.09	0.01	23	0.04	0.02	21			3	0	0	<1	<1	0
Kolan River	0.19	0.03	21	0.02	0.01	25			<1	5	0	1	1	2
Burnett River	0.57	0.09	18	0.33	0.15	6	0.5	11	4	1	9	5	6	2
Burrum River	0.51	0.08	19	0.01	0.01	26			6	1	6	<1	<1	1
Mary River	1.00	0.16	12	0.52	0.23	4	0.5	12	1	<1	1	1	<1	1
[a] calculated from the average of Haughton River and Barrata Creek ms-PAF risk values														
[b] calculated from the average of Russell River and Mulgrave River ms-PAF risk values														
[c] calculated from the Johnstone River at Coquette site 2015-2016 ms-PAF risk values														

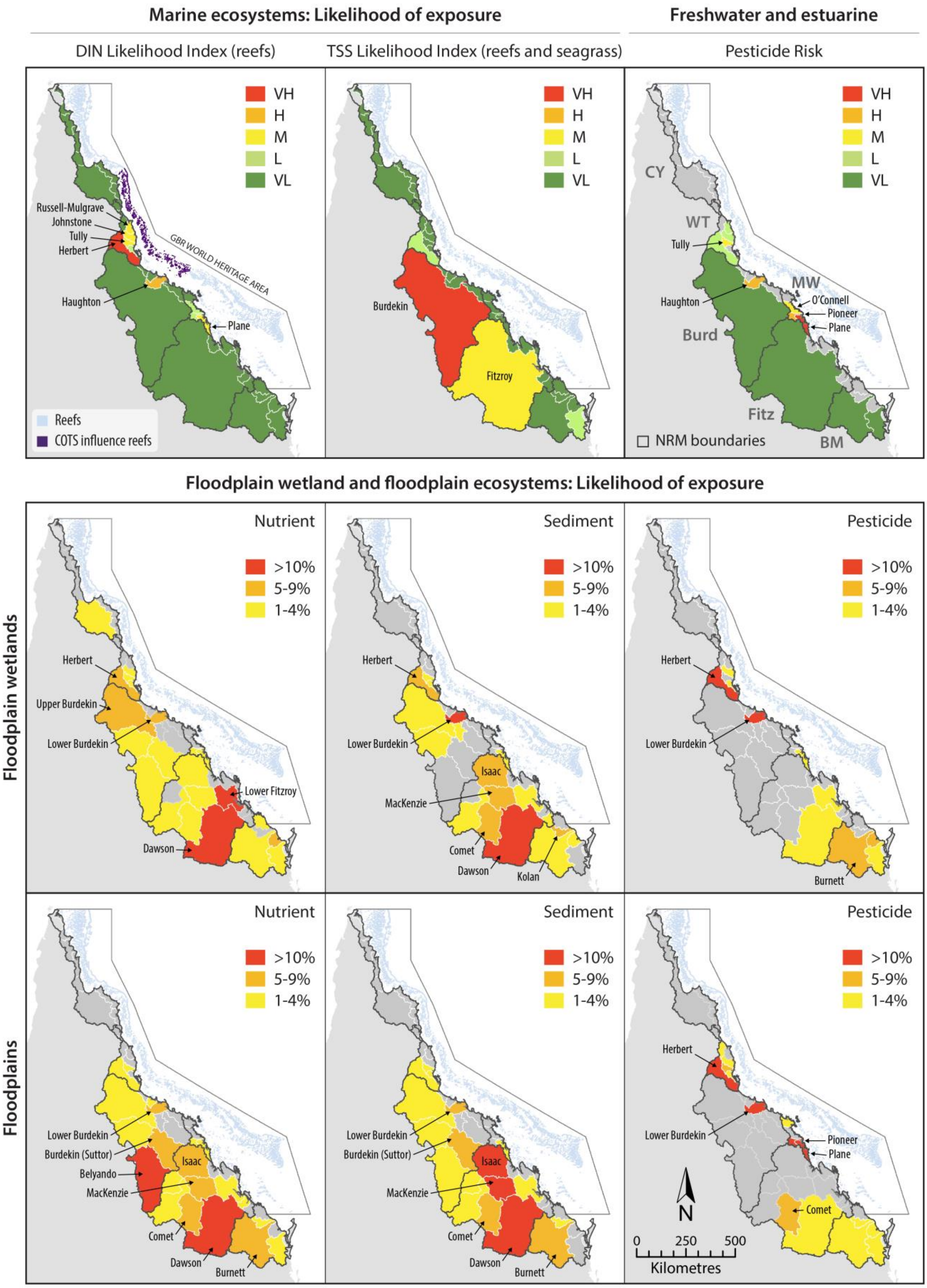


Figure 31. Summary of the likelihood of exposure and risk assessments for coastal aquatic and marine ecosystems for sediments, nutrients and pesticides.

9.2 Changes from the 2013 assessment

In the 2013 relative risk assessment, the relative risk of degraded water quality to coral reefs and seagrass were considered (Brodie et al., 2013a). Coastal aquatic ecosystems were not included.

Fine sediment: In the 2013 Scientific Consensus Statement, the Burdekin NRM and Fitzroy NRM regions were identified as the highest priorities for fine sediment management. The current assessment highlights the importance of individual basins in the assessment of the likelihood of exposure of coral reefs and seagrass to fine sediment, including the Burdekin, Fitzroy, Herbert, Mary and Johnstone basins. It also recognises that other basins within the Burdekin NRM and Fitzroy NRM regions (apart from the Burdekin and Fitzroy basins) have lower contributions to the likelihood of fine sediment exposure to coral reefs and seagrass. The Russell-Mulgrave Basin in the Wet Tropics NRM region, O'Connell Basin in the Mackay Whitsunday NRM region and the Don Basin in the Burdekin NRM region also rank relatively highly (within the top 10 basins) but the results are relatively low compared to the Burdekin and Fitzroy basins (Indexes around 10% of the likelihood of exposure).

The assessment for the Cape York basins is not considered to be adequate to draw conclusions regarding the likelihood of exposure. This is essential for future assessments.

Nutrients: In the 2013 Scientific Consensus Statement, the Wet Tropics NRM region was identified as the priority area for management of nitrogen. The current assessment of the likelihood of exposure of coral reefs to DIN maintains this result, but refines the assessment to highlight the relative importance of individual basins in the Wet Tropics NRM region, with the Herbert Basin ranked the highest, followed by the Johnstone, Russell-Mulgrave, Tully and Murray basins; the Daintree, Mossman and Barron basins are ranked comparatively lower. The assessment identifies the importance of the Haughton Basin in the Burdekin NRM region and, to a lesser extent, the Plane in the Mackay Whitsunday NRM region. The other basins in the Mackay Whitsunday NRM region (Pioneer, O'Connell, Proserpine) also rank relatively high (within the top 10 basins).

Pesticides: In the 2013 Scientific Consensus Statement, Mackay Whitsunday NRM and Burdekin (Lower Burdekin) NRM regions were identified as the priority areas for management of PSII herbicides. The current assessment supports those conclusions, even after inclusion of a wider range of pesticides including some insecticides.

Other pollutants: Emerging pollutants were identified in the 2013 Scientific Consensus Statement, but there were limited data available to conduct an assessment so it was highlighted as a knowledge gap.

10. Improvements to the risk assessment and limitations

The assessments described in this report provide the best available assessment of the likelihood exposure of TSS and DIN anthropogenic pollutants to Great Barrier Reef marine ecosystems (coral reefs and seagrass) and of nutrients, sediments and pesticides to floodplain wetlands and floodplains. The risk of pesticides to freshwater and estuarine ecosystems for basins where monitoring data are available was included, and examples of consequence and risk for TSS and DIN were also provided.

Several improvements from the 2013 risk assessment of DIN, TSS and pesticides on Great Barrier Reef marine ecosystems (Brodie et al., 2013a) were incorporated. These include:

1. incorporation of a hazard and likelihood of exposure assessment of floodplain wetlands and floodplains to expand the scope of ecosystems being considered in the assessment. Pollutant contributions associated with different land uses underpinned this assessment
2. a pesticide risk assessment for freshwater and estuarine systems, recognising the importance of pesticide toxicity in these ecosystems; the marine assessment is still under development and not quantified across the Great Barrier Reef at this stage

3. expanding the assessment of pesticides from five PSII herbicides to up to 28 pesticides, including herbicides, insecticides and fungicides
4. inclusion of a synthesis of qualitative risk (or prioritisation) of emerging contaminants
5. a shift of focus from assessment at a regional scale towards basins, and for floodplain wetlands and wetlands, management units (sub-basins for the Fitzroy and Burdekin catchments)
6. incorporation of new knowledge on the timing, movement and transformation of pollutants within the Great Barrier Reef lagoon to assist in interpretation of the quantitative assessment
7. establishment of Marine Zones that represent the extent of influence of groups of rivers in the Great Barrier Reef that are likely to overlap
8. inclusion of anthropogenic influences in the modelling data to attempt to account for areas where there is naturally high exposure of sediments and nutrients
9. application of the high resolution eReefs modelling outputs to define areas of river influence and spatial and temporal variation in chlorophyll *a* concentrations and light attenuation.

However, there are several limitations to the assessment that are important to identify:

- The nutrient and sediment assessment is conducted using average conditions, which does not fully account for the impact of large-scale events (e.g. those seen in 2010-2011) and most likely overestimates the likelihood of exposure in the drier years (e.g. 2014-2015). However, similarly to the Water Quality Improvement Plans assessments, it was concluded that most management decisions in the catchment are made on average conditions, and there is limited information to distinguish different management choices depending on extreme conditions at this stage. Future assessments could present scenarios of periods representative of dry, wet and average conditions and compare the results.
- Ideally, the results would be attributed back to individual basins, but recognition of the overlap between rivers and the potential impact of that influence is important. The Marine Zones attempt to represent the extent of influences of groups of rivers that are not necessarily in the same NRM regions but their areas of influence overlap, but it has been modelled using average conditions over the period of the eReefs tracer data, 2011-2014. This period is not representative of average conditions in some areas, for example the modelled period had above average flow conditions in Burdekin, Fitzroy and Burnett Mary regions, average flow conditions in the Wet Tropics and below average flow conditions in the Mackay Whitsunday region. Cape York is not well represented in the model. This has been accounted for to some extent, but may result in overestimates in the average likelihood of exposure in the Burdekin, Fitzroy and Burnett Mary regions (see Appendix 1 for further detail).
- Utilisation of the eReefs modelling outputs for nutrients and sediments has enabled higher resolution datasets to be incorporated, but it only includes 17 rivers in the Great Barrier Reef catchments. In particular, Cape York basins are poorly represented in the model, leading to reduced confidence in the results for this region. Major rivers in other regions are adequately represented; however, smaller coastal creeks and rivers, which have potential important influences in the inshore areas, are missing. The flow estimates included in the model are also underestimated for some rivers because of the position of the gauging station in the basin, leading to an underestimate in river flow and, thus, pollutant dispersion (see Appendix 1 for further detail).
- Application of the eReefs 4 km output for some input layers results in large interpolation of data across the Great Barrier Reef. This coarse grid size is particularly limiting along the coastline, where shallow waters and resuspension events can dominate conditions, and where intertidal seagrass beds are often located. This limitation is likely to result in underestimates in the calculation of

potential exposure of seagrass to TSS and DIN. This is also relevant to coral reefs, although there are comparably smaller areas of reefs in these nearshore coastal waters.

- Representation of anthropogenic conditions in the assessment provides a positive advancement in assessing likelihood of exposure and accounting for the influence of river-derived anthropogenic pollutants loads on marine water quality conditions and distinguishing those from areas which may be naturally high in nutrients or turbidity. However, these models are all dependent on the Source Catchments pollutant load modelling scenarios that have limitations in the definition of pre-development loads, especially for sediment (and hydrological modifications). Improvements in these estimates and incorporation of more accurate river discharge data (see above) would result in greater confidence in the results.
- It is still difficult to fully represent nutrients in the assessment as there is uncertainty in the current understanding of the relationship between end-of-catchment loads of nutrients (all forms) and measurements of chlorophyll *a* in the marine environment due to complex nutrient processing, which is affected by many factors. This applies to dissolved and particulate nutrients. Importantly, as described in Chapter 1, remote sensing techniques have limitations in shallow and turbid waters, resulting in low confidence in the outputs of these methods in the Great Barrier Reef. Further effort is required to progress this understanding and generate high resolution datasets of nutrient condition and links to riverine load inputs in the Great Barrier Reef. There is greatest confidence in the relationship between DIN loads and chlorophyll *a*, used in this assessment, but quantitative correlations between phosphorus and particulate nutrients were too limited to be included here. These will improve as the eReefs model is further developed.
- An assessment of the spatial exposure of pesticides in the marine area is currently unfeasible. The eReefs hydrodynamic model presents the best way forward to achieve a spatial exposure assessment of mixtures of pesticides (see Appendix 4), but at this stage the methods to carry out the assessment are still being explored.
- The current spatial extent of the pesticide risk assessment was limited to just those catchments and basins where monitoring data were available. In many cases, the end-of-system sample collection site for the monitored data was situated above the tidal zone, which often occurs several kilometres inland from the coast. Thus, the monitoring data do not capture the whole catchment and do not provide a true representation of the pesticide concentrations at the mouth of the estuary, where the pesticides enter the World Heritage Area and Great Barrier Reef marine park. Expansion of the monitoring program to include more catchments and sites that are closer to the mouth of the estuary will improve the representativeness of the pesticide risk assessment to the World Heritage Area and Great Barrier Reef marine park.
- The use of the pesticide Source Catchment modelled data for calculating pesticide risk was not possible. To date the Paddock to Reef modelling has been designed to report on relative changes in annual average load between a baseline and a management change scenario. To move towards reporting on a pesticide target that uses daily concentration data, a number of upgrades to the modelling would be required, including transfer of pesticides from paddock-scale models to the (Source) catchment-scale model would need to occur on a daily timescale; run-off generated from the paddock models would need to be transferred to Source Catchments; paddock modelling relative to pesticide application in cane lands needs to be improved; and multiple crop planting dates need to be represented in the paddock models, particularly for sugarcane.
- The ms-PAF risk metric only included five PSII herbicides in the risk assessment. It is anticipated that 28 pesticides, for which species sensitivity distributions have been generated, will be included in the future ms-PAF risk metric calculations.
- The ability to translate consequence factors into quantified spatial layers, and therefore calculate risk, is still relatively crude and requires considerably more effort to improve confidence in

calculating ecological risk from degraded water quality to the Great Barrier Reef. This requires better understanding and quantification of the severity and effect of pressures on Great Barrier Reef coastal and marine ecosystems, which could be progressed to some extent through further interrogation of existing monitoring results (e.g. from the AIMS Long-term Monitoring Program, Reef Plan Marine Monitoring Program and Seagrass Watch). Consideration of cumulative impacts of multiple pressures is a major challenge and has not been considered here at all.

- The current method does not include a specific measure of uncertainty, which would add value to interpretation of the results at smaller spatial scales.
- Incorporation of the current condition of ecosystems and trends over time (correlated with water quality condition) would enable management objectives for managing Great Barrier Reef water quality to be better defined, in terms of priorities for protection, restoration or efforts to maintain current values.

The assessment of the likelihood of exposure of floodplain wetlands and floodplains to sediments, nutrients and pesticides has been included for the first time. This assessment provides valuable information for managers to account in management planning and decisions for the areas that have the greatest potential to expose pollutants to coastal aquatic ecosystems. However, there are some limitations to the assessment, highlighted below.

- There is little data on the condition of floodplain wetlands or targeted research to enable quantification of the relationships between pollutant pressures, wetland condition and water quality impacts to inform a comprehensive risk assessment at the Great Barrier Reef or regional scale.
- The complexity of river, floodplain and wetland systems currently precludes any attempt to quantify the exposure of these wetlands or impacts on them. The ability to undertake a risk assessment would benefit from an improved understanding and empirical data on wetland-floodplain-river processes and functioning within the Great Barrier Reef catchments.
- Local characteristics for and vulnerabilities of wetlands and contributing catchments are needed before overlaying risk can be confidently attempted. The challenges and complexities associated with determining risk at the wetland scale are multiplied at the management unit and basin scales.
- There is insufficient knowledge of the consequences of pollutants to coastal aquatic ecosystems at the local wetland or wide scales to quantitatively assess pollutant risks at this stage.

11. Knowledge gaps and areas of further research

The limitations identified above have been translated into priority information needs for future risk assessments of water quality in the Great Barrier Reef:

1. scoping of the availability 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. refinement of the approach to estimate zones of influence for each basin
3. while the assessment of DIN risk to Crown-of-Thorns starfish has been attributed back to individual basins using end-of-catchment DIN loads, this would be improved by an updated assessment of DIN contributions from basins using the new eReefs model. This would assist in refining relative priorities between the Wet Tropics basins

4. better understanding of the responses of key Great Barrier Reef ecosystem components to cumulative impacts of repeated exposure to poor water quality, the cumulative impacts of multiple water quality pressures and responses to extreme weather events
5. limitations to nutrient measurements and chlorophyll *a* spatially and temporally. Direct measurement of chlorophyll *a* in the Great Barrier Reef lagoon is still limited in sample numbers and locations of sampling. Estimates of chlorophyll *a* concentrations can be made from water type analysis and by using the eReefs model in conjunction with direct measurements. However, a more intensive direct measurement program is still required to be able to answer questions regarding the influence of nutrient enrichment on populations of Crown-of-Thorns starfish
6. better understanding of the prevalence and associated effects of other pollutants (e.g. marine debris including microplastics, antifouling components, pharmaceuticals and others) on Great Barrier Reef species and ecosystems to assess their ecological risk, including relative to the current contaminants of concern
7. extension of the habitat assessments beyond coral reefs and seagrass to other marine ecosystems and coastal aquatic ecosystems such as floodplain wetlands, floodplains, freshwater wetland and estuarine environments (mangrove and saltpan), fish and predator fish and non-reef bioregions
8. incorporation of the principles of conservation management and the increasing need to protect areas in the Great Barrier Reef and its catchments that are in good condition as many parts of the Great Barrier Reef ecosystem become more degraded.

Additional knowledge is required to progress the underpinning science for understanding the likelihood of exposure and risk of land-based pollutants to Great Barrier Reef ecosystems. The key knowledge gaps and areas for further research are listed below.

Marine ecosystems

- Implications of the timing of seagrass exposure to sediments and nutrients: The key drivers of changes in light in the dry season are unknown as the higher resolution water quality or water type data from the dry season when seagrasses are doing most of their growing are not assessed.
- The role of nutrient-enriched water in seagrass habitat, especially in the context of secondary or tertiary water types which are typically found further offshore: While there is demonstrated evidence that primary water types are correlated to seagrass loss (e.g. Petus et al., 2014b), the secondary and tertiary water types might be having a separate, chronic effect that has not been assessed.
- There is some evidence of the relative severity of pollutant effects on Great Barrier Reef ecosystems, particularly in relation to the extensive effects of Crown-of-Thorns starfish (see Chapter 1, De'ath et al., 2012), but most influences are spatially specific and difficult to quantify. For example, macroalgal effects are dominant in inner shelf areas, and bleaching susceptibility may vary depending on the extent of influence of nutrients to cross shelf areas (driven by distance to the coast). Quantification of the severity of these effects on different ecosystems is a limitation to the current assessment.
- Temporal and spatial exposure of pesticides in marine and wetland ecosystems is largely unknown and has not been formally quantified.
- Species sensitivity distributions have not been generated for a number of pesticides detected in Great Barrier Reef ecosystems. In addition, ecotoxicity data are not available for many of these pesticides and, furthermore, the sensitivity of Great Barrier Reef freshwater and marine species is also largely unknown.

- Impacts of (i) multiple pesticides other than PSII herbicides, (ii) additional stressors related to flood plumes (e.g. low salinity, nutrients and light) and climate change (e.g. cyclones, floods, high temperature, drought, increased ocean acidification), and (iii) repeated pulses (flood plumes) and chronic exposures on organisms/communities of high conservation value. Future research would incorporatethese influences in species sensitivity distributions and review of water quality guidelines.
- The effects of contaminants other than sediment, nutrients and pesticides on Great Barrier Reef marine organisms and ecosystems are largely unknown and need to be examined for those contaminants prioritised as potentially high risk (marine plastic pollution, antifouling paints and certain personal care products). This information would contribute to a more comprehensive risk assessment of all pollutants and contaminants known to be present in the Great Barrier Reef and Torres Strait coastal and marine waters.
- Establish techniques to quantify the connectivity of the coastal freshwater and estuarine ecosystems to seagrass and coral communities and the implications of modification or loss in terms of ecological risk.

Coastal aquatic ecosystems

- Knowledge of the condition of floodplain wetlands and floodplains and quantified information on the consequences or impacts of degraded water quality on wetlands to enable a more comprehensive quantitative risk assessment for coastal aquatic ecosystems.
- Improved understanding of floodplain processes and functioning within the Great Barrier Reef catchments.
- Improved understanding of the pressures on wetlands from land uses at a smaller scale, which could include data on landholder perceptions and the uptake of best practices for wetland management. This understanding would aid in the assessment of the likelihood of exposure of pollutants. This should include a condition assessment of current vulnerable wetlands to see if exposure and ultimately risk are as high as expected.
- Exposure of wetlands to pesticide pollutants via drainage systems in nearshore areas (e.g. irrigation drainage, spoon drains), including mapping of drainage infrastructure carrying pesticide pollutants to nearshore coastal wetlands to determine basin-scale risk to nearshore coastal ecosystems.
- Historic and current rates of sediment accumulation in floodplain wetlands to improve understanding of landscape change, sediment exposure and risk to wetlands in prioritised management units including the Dawson, Lower Burdekin, Isaac, Mackenzie and Herbert.
- Understanding of the impacts of broadscale land use on changes in catchment hydrology and run-off and potential effects on coastal ecosystems and reef health.
- Improved understanding of the sediment-retention and nutrient-filtering capacity of floodplains under different land uses to better understand the role of floodplains in water quality improvement and therefore water quality risk from changed land use.
- Identification of wetland characteristics in the landscape that maximise contaminant removal, supported by data on pollutant assimilation rates, capacity and load/concentration thresholds.
- Thresholds for vegetation clearing on floodplains and riparian areas to indicate the extent that poses significant risk to floodplain function; wetland, river and marine connectivity; and the quality of water entering coastal aquatic and marine waters.

A summary table of the key findings in the 2013 Scientific Consensus Statement, a synthesis of established knowledge in this assessment, new information and areas of further research is provided in Table 26.

Table 26. Update of the key findings in the 2013 Scientific Consensus Statement, a synthesis of established knowledge, new information and areas of further research.

Established knowledge and understanding (based on previous Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
• Overall, nitrogen poses the greatest risk of pollution to coral reefs from catchments between the Daintree and Burdekin Rivers. Run-off from these rivers during extreme and early wet seasons is associated with outbreak cycles of the coral-eating Crown-of-Thorns starfish on the northern Great Barrier Reef shelf (15 to 17 degrees south) that subsequently generate secondary outbreaks throughout the central Great Barrier Reef. Great Barrier Reef–wide loss of coral cover due to Crown-of-Thorns starfish is estimated to be 1.4% per year over the past 25 years, and a new outbreak is underway. It is estimated that Crown-of-Thorns starfish have affected more than 1000 of the approximately 3000 reefs within the Great Barrier Reef over the past 60 years.	• Assessment is confirmed by new research on the influence of elevated nutrients in the Great Barrier Reef on Crown-of-Thorns starfish outbreaks. Further evidence is available about the relative importance of secondary outbreaks in overall coral condition. • The analysis of the likelihood of exposure of nutrients to the Great Barrier Reef and the risk to coral reefs from Crown-of-Thorns starfish influence by DIN exposure emphasises the dominant contribution of the Wet Tropics rivers to nutrient exposure in the Great Barrier Reef, particularly the Herbert, Johnstone, Russell-Mulgrave, Tully and Murray basins. The basin-specific assessment also identifies areas of high DIN exposure from the Haughton and Plane basins (although the Plane is not linked to Crown-of-Thorns starfish).	• Quantification of the influence of river run-off in sustaining secondary outbreaks in different years requires further work. • Difficulty in fully representing nutrients as there is uncertainty in the current understanding of the relationship between end-of-catchment loads of nutrients (all forms) and measurements of chlorophyll *a* in the marine environment due to complex nutrient processing which is affected by many factors. This applies to dissolved and particulate nutrients.
• Of equal importance is the risk to seagrass from suspended sediments discharged from rivers in excess of natural erosion rates, especially the fine fractions (clays). Whether carried in flood plumes or resuspended by waves, suspended solids create a turbid water column that reduces the light available to seagrass and corals. Increased sedimentation of fine particles interferes with many functions of benthic animal and plant communities. Run-off-associated risk decreases with increasing distance from rivers. High turbidity affects approximately 200 inshore reefs and most seagrass areas. On a regional basis, the Burdekin and Fitzroy regions present the greatest risk to the Great Barrier Reef in terms of sediment loads. Loss of seagrass habitat as a result of cyclones, floods and degraded water quality appears	• Assessment is confirmed by new research and the incorporation of an analysis of the extent of reduced benthic light to seagrass. • Regional assessments of the relative risk of degraded water quality show that fine sediment has a significant influence on photic depth and seagrass health. • Analysis of photic depth confirms that the ecosystems in shallower waters (<15 m) are at greatest risk. • The revised risk assessment will assess relative differences between basins across the Great Barrier Reef. • Importance of the frequency of large-scale events has been assessed in some locations as has the ability for coral reefs to recover from repeated disturbances.	• Effects of wet season river plumes versus annual turbidity conditions driven by resuspension. • Long-term light requirements for seagrass and coral reefs. • Effects of sedimentation on seagrasses, in particular changes to the biogeochemistry of sediments.

Established knowledge and understanding (based on previous Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
to be associated with higher mortality of dugong and turtles.	• Acute light thresholds have been identified for seagrasses. • Since the last assessment, a number of seagrass meadows have had a high frequency of exposure to water classified as secondary water, which is high in phytoplankton and also attenuates light and reduces light availability for photosynthesis. • Areas of greatest fine sediment exposure to marine ecosystems are the Burdekin, Fitzroy, Mary, Herbert, Johnstone and Burnett. The Burdekin and Fitzroy basins also have the greatest total suspended sediment risk to seagrass.	
• At smaller scales, particularly in coastal seagrass habitats and freshwater and estuarine wetlands, pesticides can pose a high risk. Concentrations of a range of pesticides exceed water quality guidelines in many fresh and estuarine waterbodies downstream of cropping lands. Based on a risk assessment of the six commonly used PSII herbicides, the Mackay Whitsunday and Burdekin regions are considered to be at highest risk, followed by the Wet Tropics, Fitzroy and Burnett Mary regions. However, the risk of only a fraction of pesticides has been assessed, with only six of the 34 pesticides currently detected included in the assessment; therefore, the effect of pesticides is most likely to have been underestimated.	• The evidence to support this statement is strengthened. Monitoring data continue to show exceedances in freshwater and estuarine systems, particularly in the Lower Burdekin and Mackay Whitsunday regions. • Only a few basins showed a Very High to Moderate risk from PSII herbicides, with diuron presenting the highest risk. These basins are generally characterised as smaller coastal catchments with high proportions of sugarcane land use (i.e. basins within the Mackay Whitsunday region and Lower Burdekin). • New data are available on a greater suite of pesticides. • The ms-PAF assessment technique is further developed and was used as part of the risk assessment for freshwater and estuarine ecosystems. A case study of marine risk is presented.	• Toxicity of emerging pesticides. • Thresholds for assimilation and breakdown and impacts of pesticides on freshwater wetlands connected to Great Barrier Reef waters and therefore the risk to these ecosystems. • Cumulative impacts from multiple stressors, particularly chronic effects.
• The relative risk of degraded water quality ranked between the regions in the Great Barrier Reef in the 2013 assessment (from highest to lowest) was: – Wet Tropics – Fitzroy – Burdekin	• The updated assessment was completed at a basin scale. Several basins were identified as high exposure for two or more pollutants, including the Russell-Mulgrave, Johnstone, Tully, Haughton, Burdekin, O’Connell, Pioneer, Plane, Fitzroy, Burnett and Mary. The Cape York region is difficult to assess with data	• Comprehensive assessment of the consequence or effects of degraded water quality on Great Barrier Reef ecosystems is challenging, as well as consideration of water quality influences in the context of other stressors such as climate change.

Established knowledge and understanding (based on previous Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
– Mackay Whitsunday – Burnett Mary – Cape York. Priority areas for management of degraded water quality in the Great Barrier Reef are Wet Tropics for nitrogen management, Mackay Whitsunday and the Lower Burdekin for PSII herbicide management and Burdekin and Fitzroy for suspended sediment management. • From a combined assessment of water quality variables in the Great Barrier Reef (using the total area of habitat affected in the areas identified to be of highest relative risk) and end-of-catchment anthropogenic loads of nutrients, sediments and PSII herbicides, the regional ranking of water quality risk to coral reefs in the 2013 assessment (from highest risk to lowest) was: – Wet Tropics – Fitzroy – Mackay Whitsunday – Burdekin – Cape York – Burnett Mary. The regional ranking of water quality risk to seagrass (from highest risk to lowest) was: – Burdekin – Wet Tropics – Fitzroy – Mackay Whitsunday – Burnett Mary – Cape York. • Importantly in the Mackay Whitsunday region, 40% of the seagrass area is in the highest relative risk class compared to less than 10% for all other regions.	limitations but is considered to remain as relatively low risk from pollutant exposure; the southern parts of the Cape York region are now known to be exposed to pollutants. • Areas of greatest fine sediment exposure to marine ecosystems are the Burdekin, Fitzroy, Mary, Herbert, Johnstone and Burnett. The Burdekin and Fitzroy basins also have the greatest TSS risk to seagrass. • Areas of greatest floodplain wetland ecosystem exposure to hazard from sediment pressure are Dawson, Isaac and Mackenzie. • Areas of greatest floodplain ecosystem exposure to hazard from sediment pressures are Dawson and Lower Burdekin. • The areas of greatest exposure of DIN to marine ecosystems are Herbert, Haughton, Johnstone, Russell-Mulgrave, Tully, Plane and Murray. The Herbert, Johnstone, Russell-Mulgrave and Tully basins have the greatest contributions to DIN risk and Crown-of-Thorns starfish in this assessment. • Areas of greatest floodplain wetland ecosystem exposure to hazard from nutrient pressure are Dawson and Lower Fitzroy. • Areas of greatest floodplain ecosystem exposure to hazard from nutrient pressures are Belyando and Dawson. • As per the 2013 assessment, only a few basins showed a Very High to Moderate risk from PSII herbicides, with diuron presenting the highest risk. These basins are generally characterised as smaller coastal catchments with high proportions of sugarcane land use (i.e. basins within the Mackay Whitsunday region and Lower Burdekin). • Pesticides pose the greatest risk to ecosystems closest to the source of the pesticides; that is,	• Analysis of the risk of pesticides to marine ecosystems is still in development. • Pollutant exposure and consequence assessments for coastal aquatic ecosystems are limited. • Use of modelling as main input for pollutant loads and comparability between regional modelling outputs. However, this will be backed up by additional interpretation of the results with monitoring results (multiple lines of evidence).

Established knowledge and understanding (based on previous Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
	freshwater wetlands, rivers and estuaries are exposed to the highest concentrations, followed by coastal ecosystems, seagrass and coral. • Areas of greatest floodplain wetland ecosystem exposure to hazard from pesticide pressure are Herbert and Lower Burdekin. • Areas of greatest floodplain ecosystem exposure to hazard from pesticide pressures are Herbert, Lower Burdekin, Belyando, Pioneer and Plane.	
• Both dissolved (inorganic and organic) and particulate forms of nutrients discharged into the Great Barrier Reef are important in driving ecological effects. Overall, increased nitrogen inputs are more important than phosphorus inputs. Dissolved inorganic forms of nitrogen and phosphorus are considered to be of greatest concern compared to dissolved organic and particulate forms as they are immediately bioavailable for supporting algal growth. Particulate forms of nitrogen and phosphorus mostly become bioavailable, but over longer time frames. Most dissolved organic nitrogen typically has limited and delayed bioavailability.	• There remains considerable uncertainty in the relative importance of nitrogen versus phosphorus, and particulate versus dissolved nutrients. • Anthropogenic particulate nitrogen is also likely to be of some importance in the same areas, but is considered less important than DIN due to the immediate bioavailability of DIN versus potential delayed bioavailability of particulate nitrogen.	• The role of particulate nutrients in driving marine ecosystem response requires further consideration. • The effects of enriching seagrass sediments with organic material. • Refinement of the magnitude of nutrient interaction with bleaching and other causal factors.
• Little is known about the types and concentrations of contaminants bound to sediment discharged by rivers into the Great Barrier Reef and the risk that these pose to marine ecosystems.	• Further evidence is available on the relative importance of emerging pollutants in the Great Barrier Reef. • Pollutants other than sediment, nutrients and pesticides are known to be present in the coastal and marine waters of the Great Barrier Reef. This includes antifouling paints, coal particles, heavy and trace metals/metalloids, marine debris/microplastics, personal care products, petroleum hydrocarbons and pharmaceuticals. • Based on current knowledge, it is likely that the following contaminants are of most concern to the Great Barrier Reef marine ecosystems:	• Pollutants such as nanomaterials, perfluorooctanoic acid and perfluorooctane sulfonate may be present, but no monitoring information is available for the Great Barrier Reef lagoon. • Little is known about the types and concentrations. The risk of these pollutants to Great Barrier Reef organisms and ecosystems is largely unknown.

Established knowledge and understanding (based on previous Scientific Consensus Statement findings—highlights)	New information in 2017	Unresolved or unknown areas (e.g. for further research)
	1. marine plastic pollution, particularly in the Cape York NRM and Torres Strait NRM regions due to exposure to oceanic and local shipping sources 2. chronic contamination of water and sediments with antifouling paints, in natural resource management regions south of Cape York 3. exposure to certain personal care products, in natural resource management regions south of Cape York.	

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Appendix 1: Method for defining Marine Zones

The Marine Zones (Table 27) used in this assessment are based on a combination of (i) the long-term (2003-2014) primary and secondary wet season water type frequency map (see Devlin et al., 2015), (ii) the latest tracer modelling from eReefs (Baird et al., 2016), (iii) the existing natural resource management marine regions and Water Quality Improvement Plans assessment boundaries, and (iv) observations of plume extent from satellite imagery.

The *outer boundaries* of the Marine Zones were derived from the 2003 to 2016 primary and secondary wet season water type (brown/green water) frequency maps by applying an estimated 5% threshold, that is, when the water types are present for at least 5% of the wet season (Figure 32). This represents, on average, approximately one week per wet season where primary or secondary water types are present, noting that these waters only need to be observed for one day for the week to be classified as a certain water type, so it is likely to be a conservative estimate of water type extent. Because the wet season water type frequency map has an increasingly irregular edge below a value of ~10%, the 5% threshold was estimated by (i) reclassifying to a raster mask with pixel values greater than 0.1, (ii) converting from raster to polygon, (iii) smoothing this 10% contour using the polynomial approximation with exponential kernel algorithm in ArcGIS and adding a 10 km buffer to the outer edge.

Tracer values from the eReefs hydrodynamic model between 1% and 2% were used to define the *northern and southern boundaries* of the Marine Zones, between 2011 and 2014 (Figure 33 and Figure 34). However, only 17 rivers in the Great Barrier Reef are modelled: the Normanby, Daintree, Barron, Russell-Mulgrave, Johnstone, Tully, Herbert, Haughton, Burdekin, Don, O'Connell, Pioneer, Fitzroy, Boyne, Calliope, Burnett and Mary rivers. Furthermore, the representativeness of river discharge in this period and, therefore, in the Marine Zones varies between regions. During the four-year modelled period, river discharge was above average in some regions of the Geat Barrier Reef, including the Burdekin, Fitzroy and Burnett Mary regions, and below average in the Mackay Whitsunday region.

To account for this, satellite imagery (Landsat and MODIS) of several river plumes were examined and used to qualify and, where necessary, adjust the outer boundaries of the Marine Zones to better represent those captured in observed events, particularly outside of the eReefs modelling period (2011-2014). Some examples are shown in Figure 35. The Mackay Whitsunday region was adjusted to match imagery from 2005 where there was greater discharge as there was relatively low discharge during the modelling period.

The confidence in the estimated Marine Zones for Cape York region using the modelling data is severely constrained by the representation of the rivers in the eReefs model: only the Normanby River is currently included. Therefore, the northern and southern assessment boundaries used in the Water Quality Improvement Plans (Cape York NRM and South Cape York Catchments, 2016) were adopted, with three zones: Cape York North, Cape York Central and Cape York South.

The eReefs model is limited in some locations by the input river flows, as the main gauge for some rivers is located upstream and, therefore, a proportion of the river flow below the gauge is not accounted for. The impact of this varies between rivers which, in many cases, means that the predicted discharge areas for some rivers will be underestimated (e.g. <50% in the Daintree, Russell-Mulgrave, Haughton, Don and O'Connell), while others where the gauge is located close to the end of the catchment (e.g. >85% in the Barron, Tully, Herbert, Burdekin, Pioneer, Burnett and Fitzroy rivers) are likely to provide a better representation of river flow (Waterhouse et al., 2017 - 2015-16 MMP report). Accordingly, the wet season water type mapping was used to define outer boundaries of the Marine Zones, and the north and south boundaries were cross-checked with satellite imagery.

Table 27. Description of the Marine Zones assessed in this chapter and the primary rivers of influence for each zone.

Marine zones	Description *Note that all outer boundaries are informed by the multi-annual wet season water type frequency maps and satellite imagery.*	Area of the Marine Zone (km^2) and proportion of the GBR combined Marine Zone	Primary rivers of influence
Cape York North	Extends from the northern boundary of the GBRWHA south to the Nesbit River mouth.	11,482 km^2 13%	Jacky Jacky, Olive, Pascoe and Lockhart
Cape York Central	Extends from the Nesbit River mouth south to Cape Melville.	8,153 km^2 9%	Stewart, Hann and Normanby
Cape York South	Extends from Cape Melville to the southern boundary of the marine NRM region.	7,259 km^2 8%	Jeannie, Endeavour with limited influence from Daintree, Mossman, Russell-Mulgrave and Johnstone
Wet Tropics	The northern boundary of the 1−1.5% tracer, to just beyond the southern boundary of the Wet Tropics NRM region to include the Palm Islands.	12,176 km^2 13%	Daintree, Mossman, Russell-Mulgrave, Johnstone, Tully, Murray, Herbert, Burdekin (limited)
Burdekin	The northern boundary of the 1.5−2% tracer, to the bottom of Edgecumbe Bay in the south.	15,677 km^2 17%	Tully, Murray, Herbert, Black, Ross, Haughton, Burdekin, Don
Mackay Whitsunday	Gloucester Island (the existing Burdekin marine NRM region boundary) to the southern Mackay Whitsunday marine NRM boundary.	14,637 km^2 16%	Proserpine, O'Connell, Pioneer, Plane
Fitzroy	Cape Conway as the northern boundary (approximately where the northern 1.5−2% tracer boundary comes in), to the southern Fitzroy marine NRM boundary.	32,436 km^2 36%	Proserpine, O'Connell, Pioneer, Plane, Styx, Shoalwater Creek, Waterpark Creek, Fitzroy, Calliope, Boyne, Burnett (limited)
Burnett Mary	Port Clinton in the north, to southern boundary of the Burnett Mary Water Quality Improvement Plan area in the south (Hervey Bay).	17,575 km^2 19%	Waterpark Creek, Fitzroy, Calliope, Boyne, Baffle, Kolan, Burnett, Burrum and Mary
Combined Marine Zone	Northern boundary of the GBRWHA to the southern boundary of the Burnett Mary Water Quality Improvement Plan area in the south (includes Hervey Bay).	91,329 km^2	

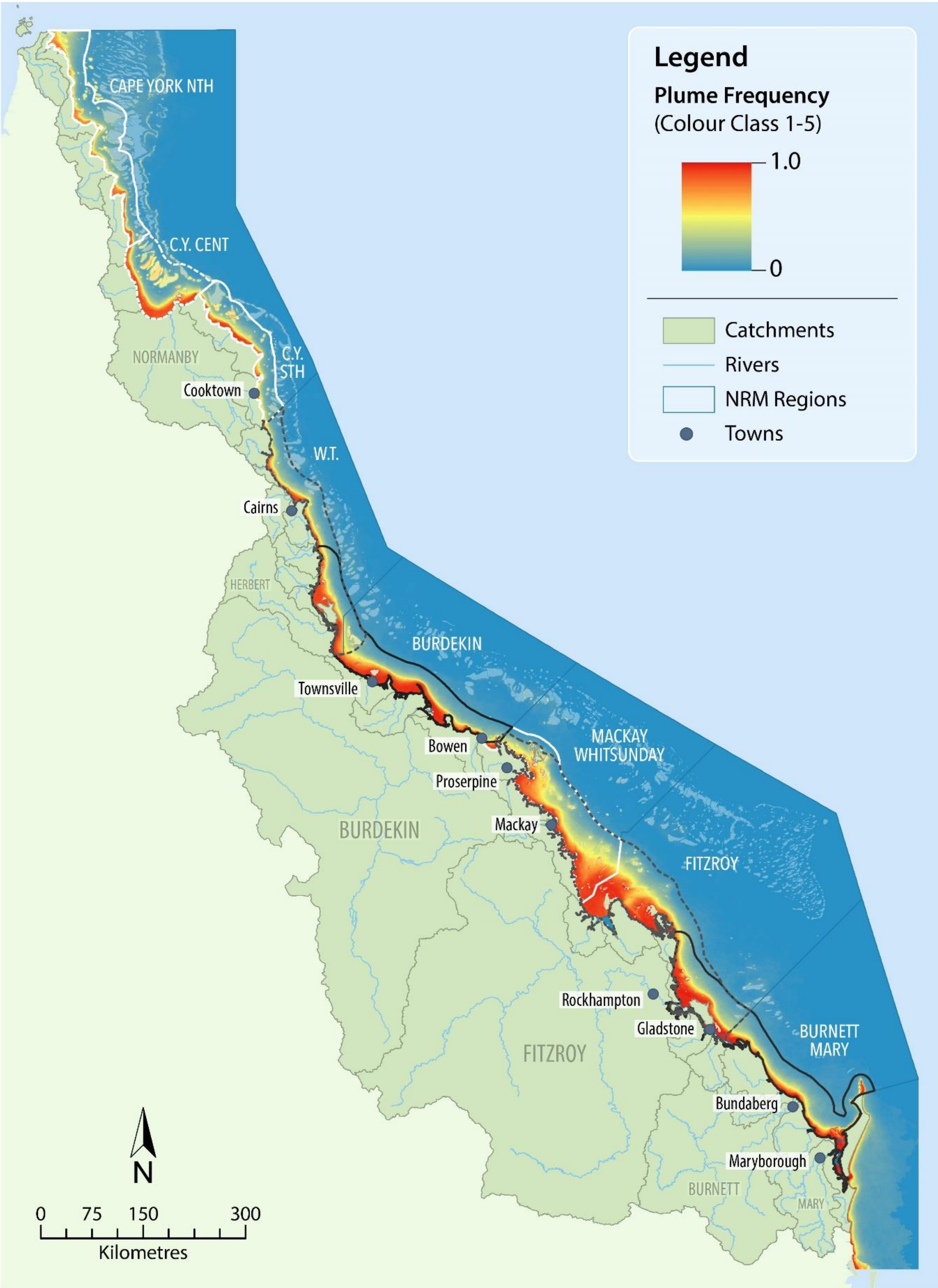

Figure 32. Wet season water type frequency (primary + secondary) map (2003-2016). Weekly primary and secondary (and colour classes) water type composites of the Great Barrier Reef coastal waters for each wet season (c.a., December to April, 22 weeks) from 2003 to 2016 were averaged to generate a multi-annual map. Map provided by JCU under the Marine Monitoring Program.

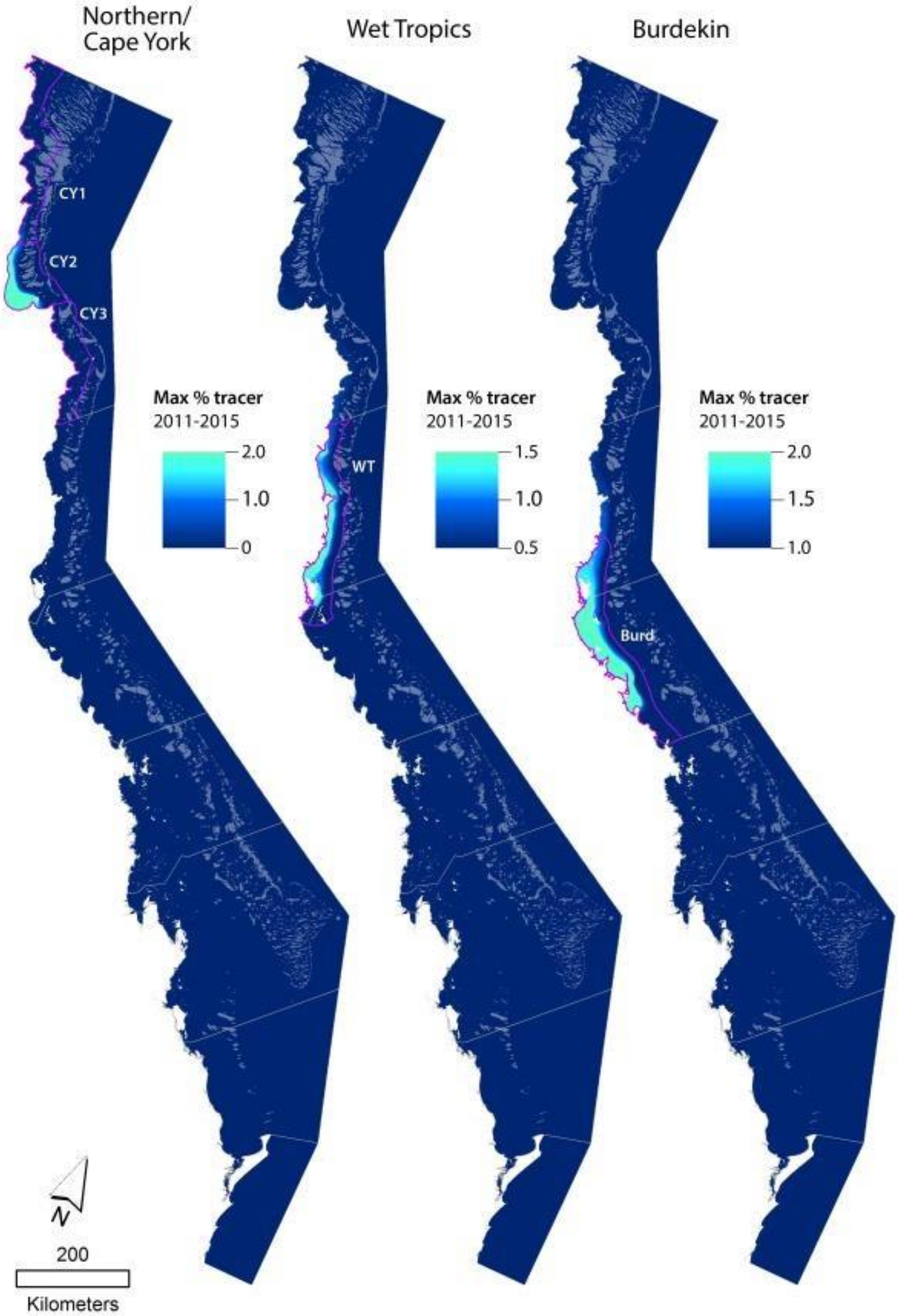

Figure 33. Panel showing the Cape York, Wet Tropics and Burdekin Marine Zones defined for the assessment, showing the eReefs tracer data (2011-2014).

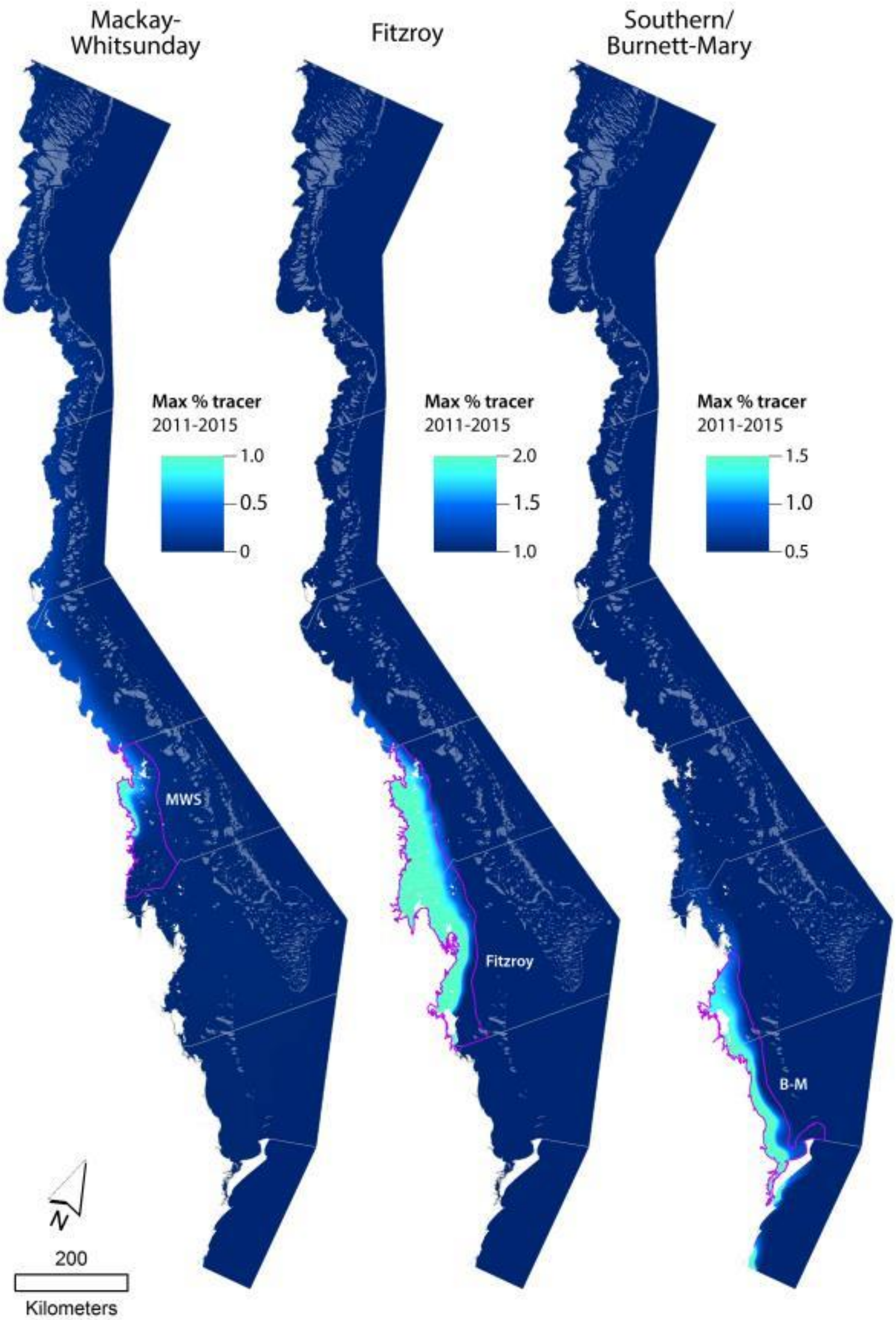

Figure 34. Panel showing the Mackay Whitsunday, Fitzroy and Burnett Mary Marine Zones defined for the assessment, showing the eReefs tracer data (2011-2014).

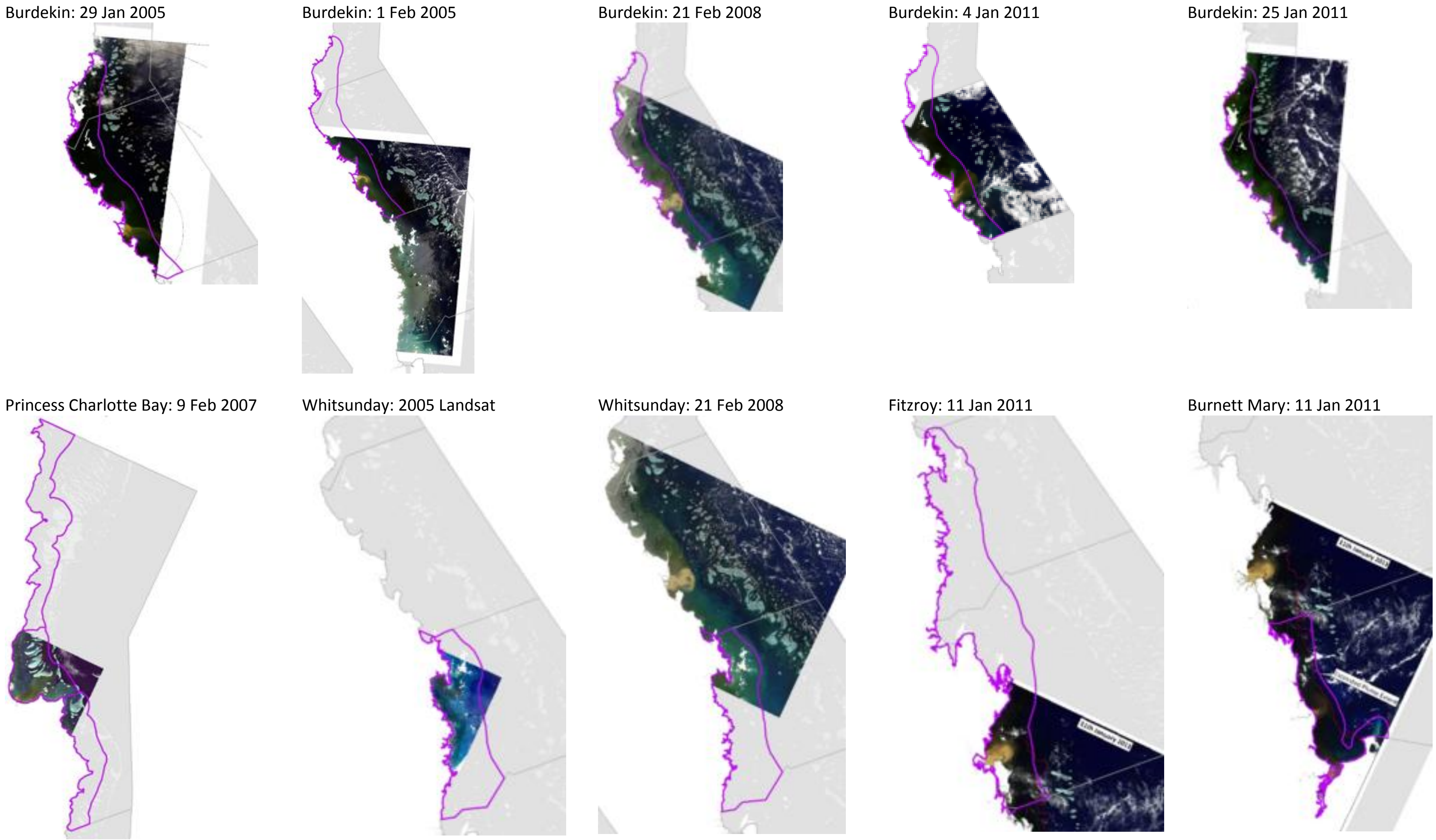

Figure 35. Examples of the comparison of satellite imagery to assess outer plume extent with the defined Marine Zones.

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Appendix 2: Floodplain wetland and floodplain hazard and likelihood of exposure assessments

Hazard to floodplain wetlands from nutrient, sediment and pesticides (based on DSITI, 2015 land-use hazard assessment)

Nutrient hazard

Table 28. Nutrient hazard data: area of High and Very High hazard within management unit, area of High and Very High hazard containing wetlands and management unit area percentages (outputs derived from DSITI, 2015 land-use hazard analysis)

Management units	Area (ha) of nutrient hazard	Area of nutrient hazard as per cent area of the management unit area	Area (ha) of nutrient hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Jacky Jacky	0	0	0	0
Olive Pascoe	0	0	0	0
Lockhart	0	0	0	0
Stewart	0	0	0	0
Normanby	75,134	3	35,448	1
Jeannie	0	0	0	0
Endeavour	0	0	0	0
Daintree	0	0	0	0
Mossman	0	0	0	0
Barron	63,228	29	6,873	3
Mulgrave-Russell	0	0	0	0
Johnstone	34,891	15	14,679	6
Tully	22,485	13	19,734	12
Murray	12,919	12	11,887	11
Herbert	137,469	14	59,361	6
Black	3,464	3	1,256	1
Ross	34,677	21	22,198	13
East Burdekin	79,083	24	19,295	6
Lower Burdekin	235,915	49	160,131	33
Belyando	1,272,804	36	293,840	8
Suttor	854,953	46	108,148	6
Upper Burdekin	737,066	18	133,279	3
Bowen Bogie	187,840	16	32,245	3
Cape Campaspe	486,901	24	71,540	4
Don River	45,095	13	7,048	2
Proserpine	39,227	16	4,360	2
O’Connell	53,405	23	18,249	8
Pioneer	73,963	44	25,230	15
Plane Creek	121,428	49	47,130	19
Styx	0	0	0	0
Shoalwater	0	0	0	0
Waterpark creek	0	0	0	0
Curtis Island	0	0	0	0
Lower Fitzroy	394,926	35	251,161	22
Dawson	2,244,080	44	1,908,855	37
Isaac	724,450	33	347,042	16
Mackenzie	368,941	28	296,709	23
Comet	732,868	43	519,451	30
Nogoa	683,878	36	370,125	19
Theresa Creek	476,321	56	148,808	17
Calliope	45,060	21	20,672	10

Management units	Area (ha) of nutrient hazard	Area of nutrient hazard as per cent area of the management unit area	Area (ha) of nutrient hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Boyne	14,481	6	0	0
Baffle	62,034	15	25,476	6
Kolan	82,416	29	17,165	6
Burnett	1,111,134	33	439,456	13
Burrum	86,062	26	70,259	21
Mary	107,097	11	53,839	6
Total area	11,705,696		5,560,949	

Sediment hazard

Table 29. Sediment hazard data: area of Very High hazard within management unit, area of Very High hazard containing wetlands and management unit area percentages (outputs derived from DSITI, 2015 land-use hazard analysis)

Management units	Area (ha) of sediment hazard	Area of sediment hazard as per cent area of the management unit area	Area (ha) of sediment hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Jacky Jacky	0	0	0	0
Olive Pascoe	0	0	0	0
Lockhart	0	0	0	0
Stewart	0	0	0	0
Normanby	34,516	1	0	0
Jeannie	0	0	0	0
Endeavour	0	0	0	0
Daintree	0	0	0	0
Mossman	0	0	0	0
Barron	35,940	16	5,520	3
Mulgrave-Russell	0	0	0	0
Johnstone	4,540	2	4,540	2
Tully	10,285	6	9,115	5
Murray	10,191	9	9,344	8
Herbert	52,101	5	32,075	3
Black	0	0	0	0
Ross	0	0	0	0
East Burdekin	23,385	7	19,295	6
Lower Burdekin	172,498	36	129,257	27
Belyando	97,419	3	0	0
Suttor	376,575	20	57,488	3
Upper Burdekin	78,136	2	34,872	1
Bowen Bogie	19,319	2	0	0
Cape Campaspe	55,961	3	21,237	1
Don River	16,232	5	2,809	1
Proserpine	19,660	8	0	0
O'Connell	14,659	6	3,552	2
Pioneer	8,743	5	2,953	2
Plane Creek	72,050	29	21,911	9
Styx	0	0	0	0
Shoalwater	0	0	0	0
Waterpark creek	0	0	0	0
Curtis Island	0	0	0	0

Management units	Area (ha) of sediment hazard	Area of sediment hazard as per cent area of the management unit area	Area (ha) of sediment hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Lower Fitzroy	183,637	16	91,215	8
Dawson	2,076,773	41	1,754,956	34
Isaac	512,319	23	317,578	14
Mackenzie	366,852	28	278,113	21
Comet	631,689	37	441,588	26
Nogoa	445,957	23	200,879	10
Theresa Creek	408,103	48	92,142	11
Calliope	12,058	6	8,529	4
Boyne	644	0	0	0
Baffle	10,096	2	5,621	1
Kolan	48,769	17	0	0
Burnett	803,051	24	339,980	10
Burrum	44,961	13	30,728	9
Mary	16,863	2	5,387	1
Total area	6,663,982		3,920,684	

Pesticide hazard

Table 30. Pesticide hazard data: area of High and Very High hazard within management unit, area of High and Very High hazard containing wetlands and management unit area percentages (outputs derived from DSITI, 2015 land-use hazard analysis).

Management units	Area (ha) of pesticide hazard	Area of pesticide hazard as per cent area of the management unit area	Area (ha) of pesticide hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Jacky Jacky	0	0	0	0
Olive Pascoe	0	0	0	0
Lockhart	0	0	0	0
Stewart	0	0	0	0
Normanby	0	0	0	0
Jeannie	0	0	0	0
Endeavour	0	0	0	0
Daintree	0	0	0	0
Mossman	0	0	0	0
Barron	13,317	6	0	0
Mulgrave-Russell	0	0	0	0
Johnstone	4,540	2	4,540	2
Tully	14,677	9	12,378	7
Murray	5,225	5	4,592	4
Herbert	50,892	5	38,826	4
Black	0	0	0	0
Ross	0	0	0	0
East Burdekin	0	0	0	0
Lower Burdekin	80,991	17	78,073	16
Belyando	0	0	0	0
Suttor	30,226	2	0	0
Upper Burdekin	0	0	0	0
Bowen Bogie	0	0	0	0

Management units	Area (ha) of pesticide hazard	Area of pesticide hazard as per cent area of the management unit area	Area (ha) of pesticide hazard area containing wetlands	Area (ha) of hazard containing wetlands as per cent area of the management unit area
Cape Campaspe	0	0	0	0
Don River	2,809	1	2,809	1
Proserpine	9,836	4	0	0
O'Connell	3,324	1	0	0
Pioneer	41,890	25	21,277	13
Plane Creek	40,010	16	21,911	9
Styx	0	0	0	0
Shoalwater	0	0	0	0
Waterpark creek	0	0	0	0
Curtis Island	0	0	0	0
Lower Fitzroy	0	0	0	0
Dawson	17,579	0	17,579	0
Isaac	0	0	0	0
Mackenzie	0	0	0	0
Comet	67,398	4	49,589	3
Nogoa	1,271	0	0	0
Theresa Creek	0	0	0	0
Calliope	0	0	0	0
Boyne	0	0	0	0
Baffle	0	0	0	0
Kolan	10,670	4	0	0
Burnett	22,339	1	16,780	1
Burrum	18,310	5	14,155	4
Mary	4,289	0	4,289	0
Total area	439,592		286,799	

Remnant regional ecosystem clearing

Table 31. Area historically cleared of native vegetation (referred to as non-remnant) within management unit, recent clearing of remnant woody regional ecosystems outside of and inside of floodplains (derived from the Statewide Landcover and Trees Study (SLATS) data [DSITI, 2016].

Management units	Per cent of floodplain area in management unit that is non-remnant in 2013 (i.e. cleared of native vegetation)	Area (ha) recently cleared (SLATS 14-15) outside floodplains	Area (ha) remnant regional ecosystems recently cleared (SLATS 14-15) in floodplains	Per cent of floodplain 2013 remnant regional ecosystems in management unit recently cleared (SLATS 14-15)
Jacky Jacky	0	19	1	0.00
Olive Pascoe	0	19	0	0.00
Lockhart	0	1	0	0.00
Stewart	2	10	55	0.04
Normanby	1	1,730	83	0.01
Jeannie	1	4	0	0.00
Endeavour	7	45	10	0.05
Daintree	26	0	0	0.00
Mossman	58	0	0	0.00
Barron	62	0	0	0.00
Mulgrave-Russell	63	0	0	0.00

Management units	Per cent of floodplain area in management unit that is non-remnant in 2013 (i.e. cleared of native vegetation)	Area (ha) recently cleared (SLATS 14-15) outside floodplains	Area (ha) remnant regional ecosystems recently cleared (SLATS 14-15) in floodplains	Per cent of floodplain 2013 remnant regional ecosystems in management unit recently cleared (SLATS 14-15)
Johnstone	67	0	0	0.00
Tully	69	0	0	0.00
Murray	49	0	0	0.00
Herbert	50	0	0	0.00
Black	35	59	66	0.41
Ross	45	89	197	0.47
East Burdekin	15	214	23	0.11
Lower Burdekin	48	97	391	0.25
Belyando	62	17,695	4,526	1.45
Suttor	64	4,694	1,700	0.97
Upper Burdekin	5	1,406	114	0.02
Bowen Bogie	32	274	22	0.04
Cape Campaspe	16	1,235	207	0.06
Don River	38	417	82	0.20
Proserpine	58	407	35	0.13
O’Connell	54	78	2	0.01
Pioneer	73	80	18	0.13
Plane Creek	68	107	328	0.61
Styx	52	75	1,023	1.46
Shoalwater	21	137	20	0.03
Waterpark Creek	19	453	303	0.51
Curtis Island	0	49	0	0.00
Lower Fitzroy	74	3,860	565	0.75
Dawson	83	13,187	1,023	1.07
Isaac	59	6,079	2,239	1.34
Mackenzie	85	4,760	1,722	2.85
Comet	72	2,548	502	0.58
Nogoa	65	2,853	276	0.39
Theresa Creek	55	2,585	72	0.24
Calliope	75	571	44	0.38
Boyne	73	189	34	0.19
Baffle	35	3,532	863	1.37
Kolan	74	1,637	771	5.54
Burnett	83	11,384	447	0.98
Burrum	31	1,432	186	0.44
Mary	75	4,327	368	1.16
Total area		88,338	18,318	

Floodplain wetland exposure to pollutant hazard

Floodplain wetland nutrient exposure

Table 32. Likelihood of exposure: The relative proportion (%) of GBR floodplain wetland area exposed to High and Very High nutrient hazard. Showing management units with a relative GBR catchment-wide proportion of 1% or more of floodplain wetland area exposed to High and Very High nutrient hazard: red = >10%, orange = 5–9%, yellow = 1–4%.

Management units	Relative per cent of GBR floodplain wetland area exposed to High and Very High hazard from nutrient pressures	Floodplain wetland area (ha) exposed to High and Very High hazard from nutrient pressures	Per cent floodplain wetland area exposed to High and Very High hazard from nutrient pressures in the management unit
Jacky Jacky	0	0	0
Olive Pascoe	0	0	0
Lockhart	0	0	0
Stewart	0	0	0
Normanby	3	979	5
Jeannie	0	0	0
Endeavour	0	0	0
Daintree	0	0	0
Mossman	0	0	0
Barron	<1	55	58
Mulgrave-Russell	0	0	0
Johnstone	1	305	30
Tully	2	653	18
Murray	2	462	11
Herbert	7	1,909	17
Black	<1	13	15
Ross	<1	60	5
East Burdekin	<1	88	16
Lower Burdekin	7	2,073	21
Belyando	4	1,252	21
Burdekin - Suttor	1	221	18
Upper Burdekin	5	1,455	8
Bowen Bogie	<1	3	3
Cape Campaspe	2	654	15
Don River	<1	18	1
Proserpine	<1	123	2
O'Connell	<1	9	14
Pioneer	1	204	75
Plane Creek	1	169	24
Styx	0	0	0
Shoalwater	0	0	0
Waterpark Creek	0	0	0
Curtis Island	0	0	0
Lower Fitzroy	10	2,845	23
Dawson	23	6,713	64
Isaac	4	1,257	33
Mackenzie	3	811	26
Comet	3	919	21
Nogoa	4	1,057	51
Theresa Creek	<1	113	45
Calliope	<1	21	18

Management units	Relative per cent of GBR floodplain wetland area exposed to High and Very High hazard from nutrient pressures	Floodplain wetland area (ha) exposed to High and Very High hazard from nutrient pressures	Per cent floodplain wetland area exposed to High and Very High hazard from nutrient pressures in the management unit
Boyne	0	0	0
Baffle	3	931	10
Kolan	<1	101	57
Burnett	4	1,129	66
Burrum	6	1,686	22
Mary	1	405	6
Total area		28,693	

Floodplain wetland sediment exposure

Table 33. Likelihood of exposure: The relative proportion (%) of GBR floodplain wetland area exposed to Very High sediment hazard. Showing management units with a relative GBR catchment-wide proportion of 1% or more of floodplain wetland area exposed to High and Very High sediment hazard: red = >10%, orange = 5–9%, yellow = 1–4%.

Management units	Relative per cent of GBR floodplain wetland area exposed to Very High hazard from sediment pressure	Floodplain wetland area (ha) exposed to Very High hazard from sediment pressure	Per cent floodplain wetland area exposed to hazard from sediment pressure in the management unit
Jacky Jacky	0	0	0
Olive Pascoe	0	0	0
Lockhart	0	0	0
Stewart	0	0	0
Normanby	0	0	0
Jeannie	0	0	0
Endeavour	0	0	0
Daintree	0	0	0
Mossman	0	0	0
Barron	<1	55	58
Mulgrave-Russell	0	0	0
Johnstone	<1	32	3
Tully	1	92	3
Murray	3	339	8
Herbert	5	688	6
Black	0	0	0
Ross	0	0	0
East Burdekin	1	88	16
Lower Burdekin	14	1,756	18
Belyando	0	0	0
Suttor	<1	21	2
Upper Burdekin	1	156	1
Bowen Bogie	0	0	0
Cape Campaspe	2	199	4
Don River	<1	9	<1
Proserpine	0	0	0
O’Connell	<1	3	5
Pioneer	<1	12	5
Plane Creek	1	71	10
Styx	0	0	0

Management units	Relative per cent of GBR floodplain wetland area exposed to Very High hazard from sediment pressure	Floodplain wetland area (ha) exposed to Very High hazard from sediment pressure	Per cent floodplain wetland area exposed to hazard from sediment pressure in the management unit
Shoalwater	0	0	0
Waterpark Creek	0	0	0
Curtis Island	0	0	0
Lower Fitzroy	2	208	2
Dawson	38	4,944	47
Isaac	9	1,175	31
Mackenzie	7	870	28
Comet	5	686	16
Nogoa	3	421	20
Calliope	<1	7	6
Boyne	0	0	0
Baffle	1	125	1
Kolan	0	0	0
Burnett	5	663	39
Burrum	1	192	2
Theresa Creek	1	92	37
Mary	<1	4	<1
Total area		12,906	

Floodplain wetland pesticide exposure

Table 34. Likelihood of exposure: The relative proportion (%) of GBR floodplain wetland area exposed to High and Very High pesticide hazard. Showing management units with a relative GBR catchment-wide proportion of 1% or more of floodplain wetland area exposed to High and Very High pesticide hazard: red = >10%, orange = 5–9%, yellow = 1–4%.

Management unit	Relative per cent of GBR floodplain wetland area exposed to High and Very High hazard from pesticide pressure	Floodplain wetland area (ha) exposed to High and Very High hazard from pesticide pressures	Per cent floodplain wetland area exposed to hazard from pesticide pressure in the management unit
Jacky Jacky	0	0	0
Olive Pascoe	0	0	0
Lockhart	0	0	0
Stewart	0	0	0
Normanby	0	0	0
Jeannie	0	0	0
Endeavour	0	0	0
Daintree	0	0	0
Mossman	0	0	0
Barron	0	0	0
Mulgrave-Russell	0	0	0
Johnstone	1	32	3
Tully	7	198	5
Murray	2	67	2
Herbert	23	639	6
Black	0	0	0
Ross	0	0	0
East Burdekin	0	0	0
Lower Burdekin	43	1,203	12

Management unit	Relative per cent of GBR floodplain wetland area exposed to High and Very High hazard from pesticide pressure	Floodplain wetland area (ha) exposed to High and Very High hazard from pesticide pressures	Per cent floodplain wetland area exposed to hazard from pesticide pressure in the management unit
Belyando	0	0	0
Suttor	0	0	0
Upper Burdekin	0	0	0
Bowen Bogie	0	0	0
Cape Campaspe	0	0	0
Don River	<1	9	<1
Proserpine	0	0	0
O'Connell	0	0	0
Pioneer	2	68	25
Plane Creek	3	73	10
Styx	0	0	0
Shoalwater	0	0	0
Waterpark Creek	0	0	0
Curtis Island	0	0	0
Lower Fitzroy	0	0	0
Dawson	2	51	0
Isaac	0	0	0
Mackenzie	0	0	0
Comet	<1	4	<1
Nogoa	0	0	0
Theresa Creek	0	0	0
Calliope	0	0	0
Boyne	0	0	0
Baffle	0	0	0
Kolan	0	0	0
Burnett	9	264	15
Burrum	6	156	2
Mary	1	19	1
Total area		2,783	

Floodplain exposure

Nutrient

Table 35. Likelihood of exposure: The relative proportion (%) of GBR floodplain area exposed to High and Very High hazard from nutrient pressures. Showing management units with a relative GBR catchment-wide proportion of 1% or more of floodplain wetland area exposed to High and Very High nutrient hazard: red = >10%, orange = 5–9%, yellow = 1–4%.

Management units	Relative per cent of GBR catchment floodplain area exposed to High and Very High hazard from nutrient pressures	Floodplain area (ha) exposed to High and Very High total nutrient hazard pressures	Per cent floodplain area exposed to High and Very High hazard from nutrient pressures in the management unit	Total area of floodplain within management unit
Jacky Jacky	0	0	0	103,883
Olive Pascoe	0	0	0	45,027
Lockhart	0	0	0	59,816
Stewart	0	0	0	130,443
Normanby	<1	5,296	<1	1,082,753
Jeannie	0	0	0	52,980
Endeavour	0	0	0	21,067
Daintree	0	0	0	32,401
Mossman	0	0	0	10,482
Barron	<1	5,051	29	17,623
Mulgrave-Russell	0	0	0	46,105
Johnstone	<1	12,312	26	47,783
Tully	1	15,582	33	46,838
Murray	<1	9,376	19	48, 367
Herbert	2	48,779	35	140,335
Black	<1	3,288	13	24,717
Ross	1	16,207	21	76,183
East Burdekin	<1	4,236	17	24,245
Lower Burdekin	6	158,881	52	307,132
Belyando	12	307,053	37	822,153
Suttor	8	202,386	41	490,390
Upper Burdekin	4	103,910	17	608,008
Bowen Bogie	<1	10,617	14	76,221
Cape Campaspe	4	100,654	22	447,917
Don River	<1	5,193	8	67,220
Proserpine	<1	11,309	18	61,638
O'Connell	<1	10,139	28	36,139
Pioneer	1	36,374	74	49,330
Plane Creek	3	84,119	51	166,525
Styx	0	0	0	146,605
Shoalwater	0	0	0	90,507
Waterpark Creek	0	0	0	72,085
Curtis Island	0	0	0	0
Lower Fitzroy	3	73,710	25	293,830
Dawson	16	407,775	72	565,748
Isaac	7	176,432	43	406,569
Mackenzie	6	142,805	35	410,055
Comet	6	155,207	50	309,127

Management units	Relative per cent of GBR catchment floodplain area exposed to High and Very High hazard from nutrient pressures	Floodplain area (ha) exposed to High and Very High total nutrient hazard pressures	Per cent floodplain area exposed to High and Very High hazard from nutrient pressures in the management unit	Total area of floodplain within management unit
Nogoa	4	111,221	54	206,339
Theresa Creek	2	43,945	66	66,668
Calliope	1	14,010	30	46,549
Boyne	<1	7,888	12	68,004
Baffle	<1	10,166	10	97,231
Kolan	1	24,196	46	52,585
Burnett	5	131,924	50	266,190
Burrum	<1	12,285	20	61,804
Mary	1	23,745	19	124,531
Total area		2,486,071		8,428,149

Sediment

Table 36. Likelihood of exposure: The relative proportion (%) of GBR floodplain area exposed to High and Very High hazard from sediment pressures. Showing management units with a relative Great Barrier Reef catchment-wide proportion of 1% or more of floodplain wetland area exposed to High and Very High sediment hazard: red = >10% red, orange = 5–9%, yellow = 1–4%.

Management units	Relative per cent of GBR catchment floodplain area exposed to Very High hazard from sediment pressures	Floodplain area (ha) exposed to Very High total sediment hazard pressures	Per cent floodplain area exposed to Very High sediment hazard in the management unit	Total area of floodplain within management unit
Jacky Jacky	0	0	0	103,883
Olive Pascoe	0	0	0	45,027
Lockhart	0	0	0	59,816
Stewart	0	0	0	130,443
Normanby	<1	886	<1	1,082,753
Jeannie	0	0	0	52,980
Endeavour	0	0	0	21,067
Daintree	0	0	0	32,401
Mossman	0	0	0	10,482
Barron	<1	1,992	11	17,623
Mulgrave-Russell	0	0	0	46,105
Johnstone	<1	3,673	8	47,783
Tully	1	7,526	16	46,838
Murray	1	7,167	15	48,367
Herbert	2	31,169	22	140,335
Black	0	0	0	24,717
Ross	0	0	0	76,183
East Burdekin	<1	1,868	8	24,245
Lower Burdekin	9	124,685	41	307,132
Belyando	1	18,438	2	822,153
Suttor	7	102,920	21	490,390
Upper Burdekin	1	13,181	2	608,008
Bowen Bogie	<1	42	<1	76,221
Cape Campaspe	1	10,707	2	447,917

Management units	Relative per cent of GBR catchment floodplain area exposed to Very High hazard from sediment pressures	Floodplain area (ha) exposed to Very High total sediment hazard pressures	Per cent floodplain area exposed to Very High sediment hazard in the management unit	Total area of floodplain within management unit
Don River	<1	1,325	2	67,220
Proserpine	<1	6,359	10	61,638
O'Connell	<1	2,408	7	36,139
Pioneer	<1	4,424	9	49,330
Plane Creek	4	53,973	32	166,525
Styx	0	0	0	146,605
Shoalwater	0	0	0	90,507
Waterpark Creek	0	0	0	72,085
Curtis Island	0	0	0	0
Lower Fitzroy	2	22,740	8	293,830
Dawson	26	362,398	64	565,748
Isaac	11	157,411	39	406,569
Mackenzie	11	156,401	38	410,055
Comet	9	121,036	39	309,127
Nogoa	4	50,223	24	206,339
Theresa Creek	2	33,053	50	66,668
Calliope	<1	2,406	5	46,549
Boyne	<1	573	1	68,004
Baffle	<1	1,965	2	97,231
Kolan	1	12,980	25	52,585
Burnett	6	91,033	34	266,190
Burrum	<1	3,300	5	61,804
Mary	<1	2,434	2	124,531
Total area		1,410,697		8,428,149

Pesticides

Table 37. Likelihood of exposure: The relative proportion (%) of GBR floodplain area exposed to High and Very High hazard from pesticide pressures. Showing management units with a relative GBR catchment-wide proportion of 1% or more of floodplain area exposed to High and Very High pesticide hazard: red = >10%, orange = 5–9%, yellow = 1–4%.

Management units	Relative per cent of GBR catchment floodplain area exposed to High and Very High pesticide hazard	Floodplain area (ha) exposed to High and Very High pesticide hazard	Per cent floodplain area exposed to High and Very High pesticide hazard in the management unit	Total area of floodplain within management unit
Jacky Jacky	0	0	0	103,883
Olive Pascoe	0	0	0	45,027
Lockhart	0	0	0	59,816
Stewart	0	0	0	130,443
Normanby	0	0	0	1,082,753
Jeannie	0	0	0	52,980
Endeavour	0	0	0	21,067
Daintree	0	0	0	32,401
Mossman	0	0	0	10,482
Barron	1	1,245	7	17,623
Mulgrave-Russell	0	0	0	46,105
Johnstone	2	3,673	8	47,783
Tully	5	11,211	24	46,838
Murray	2	4,562	9	48,367
Herbert	14	32,650	23	140,335
Black	0	0	0	24,717
Ross	0	0	0	76,183
East Burdekin	0	0	0	24,245
Lower Burdekin	33	77,294	25	307,132
Belyando	0	0	0	822,153
Suttor	<1	879	<1	490,390
Upper Burdekin	0	0	0	608,008
Bowen Bogie	0	0	0	76,221
Cape Campaspe	0	0	0	447,917
Don River	<1	40	<1	67,220
Proserpine	1	3,294	5	61,638
O’Connell	<1	475	1	36,139
Pioneer	10	23,228	47	49,330
Plane Creek	16	37,171	22	166,525
Styx	0	0	0	146,605
Shoalwater	0	0	0	90,507
Waterpark Creek	0	0	0	72,085
Curtis Island	0	0	0	0
Lower Fitzroy	0	0	0	293,830
Dawson	3	7,488	1	565,748
Isaac	0	0	0	406,569
Mackenzie	0	0	0	410,055
Comet	7	17,092	6	309,127
Nogoa	<1	462	<1	206,339
Theresa Creek	0	0	0	66,668
Calliope	0	0	0	46,549
Boyne	0	0	0	68,004

Management units	Relative per cent of GBR catchment floodplain area exposed to High and Very High pesticide hazard	Floodplain area (ha) exposed to High and Very High pesticide hazard	Per cent floodplain area exposed to High and Very High pesticide hazard in the management unit	Total area of floodplain within management unit
Baffle	0	0	0	97,231
Kolan	2	5,198	10	52,585
Burnett	2	5,650	2	266,190
Burrum	1	1,652	3	61,804
Mary	1	1,412	1	124,531
Total area		234,678		8,428,149

References

DSITI 2015. A landscape hazard assessment for wetlands in the GBR catchment, Department of Science, Information technology and Innovation, Queensland Government.

DSITI 2016. Land cover change in Queensland 2014–15: a Statewide Landcover and Trees Study (SLATS) report. Queensland Department of Science, Information Technology and Innovation. 2016. Brisbane.

Appendix 3: Technical details for spatial inputs to the marine likelihood of exposure assessments

The following spatial layers have been applied in the assessment of the likelihood of exposure of coral reefs and seagrass to dissolved inorganic nitrogen (DIN) and total suspended sediments (TSS).

1. Frequency of wet season water types

The frequency of wet season water type (also referred as the primary, secondary and tertiary wet season water types) composite represents the long-term (2003-2016) mean frequency of exposure of coastal water types that are typically enriched in pollutants during the wet season (surface waters only). These water types are mapped through a supervised classification (colour classification) of MODIS true colour satellite imagery (Álvarez-Romero et al., 2013). This colour classification has been successfully used in the Great Barrier Reef (e.g. Devlin et al., 2013; Devlin et al., 2015; Petus et al., 2014; Petus et al., 2016).

The wet season water type layers are produced using MODIS true colour imagery reclassified to six distinct colour classes defined by their colour properties (see detailed techniques below) and typical of colour gradients existing across the coastal waters and flood river plumes of the Great Barrier Reef during the wet season (Figure 36). The wet season water types are further regrouped into three water types (primary, secondary and tertiary wet season water types) with colour classes 1–4 corresponding to the primary water type, the colour class 5 to the secondary water type and the colour class 6 to the tertiary water type (defined originally by Devlin and Schaffelke (2009) and Devlin et al. (2012). The colour classification allows a finer-scale characterisation of the water constituents inside of the primary water type.

Each of the three water types is characterised by different concentrations of optically active components (total suspended sediment, colour dissolved organic matter and chlorophyll *a*) which control the colour of the water and influence the light attenuation, as well as different pollutant concentrations which can vary the impact on the underlying ecological systems. Typical water quality concentrations measured across the wet season water types during high flow events in the Great Barrier Reef are presented in Figure 37.

- The *primary water type (colour classes 1-4)* corresponds to the brownish to brownish-green turbid water masses. These waters are with high nutrient and phytoplankton concentrations, but are also enriched in sediment and dissolved organic matter and have reduced light levels. They are typical for nearshore areas or inshore regions of flood river plumes.
- The *secondary water type (colour class 5)* corresponds to the greenish to greenish-blue water masses and is typical of coastal waters dominated by algae, but also with some dissolved matter and some fine sediment present. Relatively high nutrient availability and increased light levels due to sedimentation favour an increased coastal productivity in this water type. This water type is typical for the coastal waters or the mid-region of river plumes.
- The *tertiary water type (colour class 6)* is the transitional, greenish-blue water mass with slightly above ambient turbidity and nutrient concentrations. This water type is typical for areas towards the open sea or offshore regions of flood river plumes.

This method produces weekly primary, secondary and tertiary (and six colour classes) water type composites of the Great Barrier Reef coastal waters for each wet season (defined as December to April, 22 weeks) from 2003 to 2016. The weekly outputs are overlaid to calculate the annual or long-term frequency of occurrence of each water type: the water type frequency is calculated as the number of weeks within a wet season that a pixel was exposed to each water type. Finally, the annual outputs are averaged to generate a multi-annual map (2003-2016) and normalised to the maximum value between 0 (lowest) and 1 (highest).

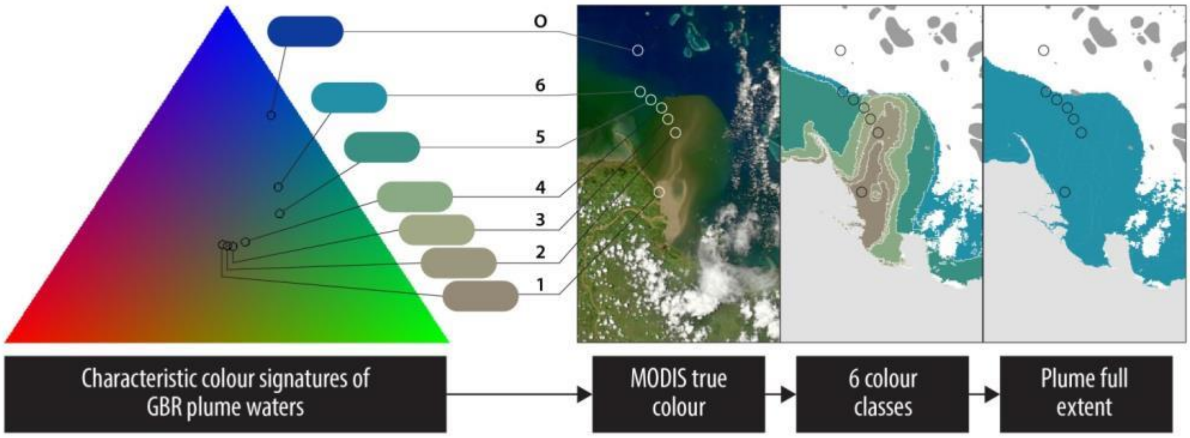

Figure 36. Triangular colour plot showing the characteristic colour signatures of the GBR wet season waters in the Red-Green-Blue (or true colour) space. Álvarez-Romero et al. (2013) developed a method to map these characteristic coastal water masses in the GBR using a supervised classification of MODIS true colour data (modified from Devlin et al., 2015).

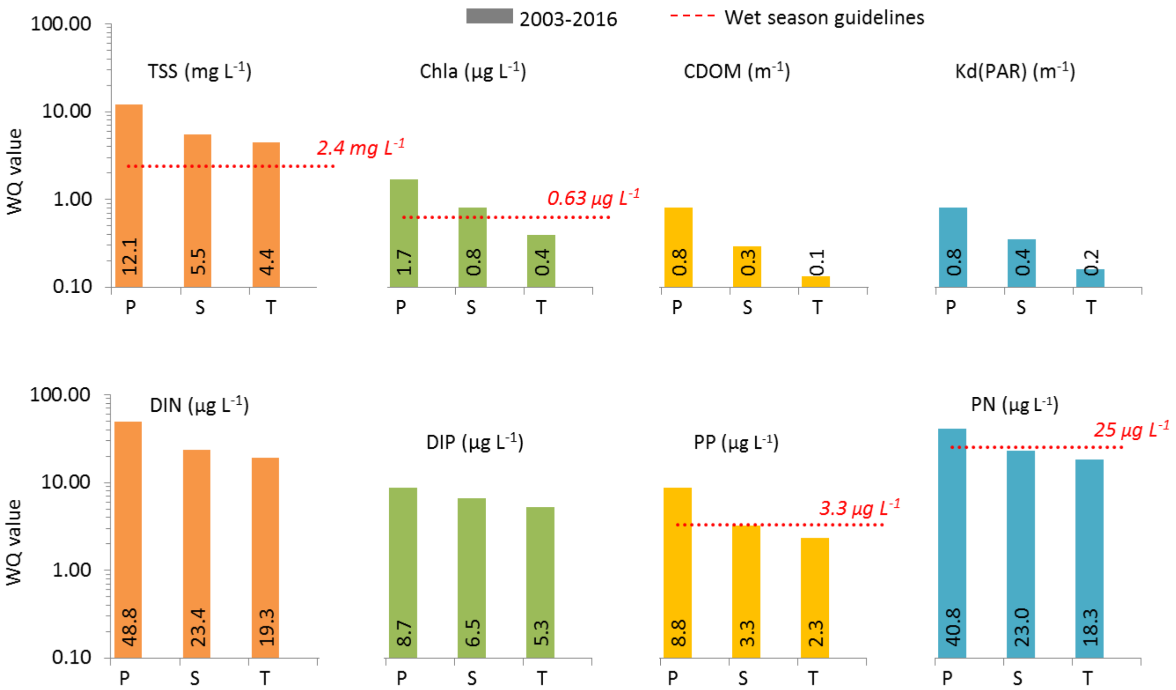

Figure 37. Mean long-term (2003-to 2016) water quality concentrations measured across the three wet season water types: primary (P), secondary (S) and tertiary (T) water types. *In situ* water quality data were collected through the flood plume monitoring program of the Marine Monitoring Program (Waterhouse et al., 2017): Total Suspended Sediment (TSS in mg/L), chlorophyll *a* (Chl-*a* in µg/L), DIN and phosphorus (DIN and DIP in µg/L), particulate nitrogen and phosphorus (PN and PP in µg/L). The wet season water quality guideline values for the GBR open coastal and mid-shelf waters are indicated with dotted red lines (GBRMPA, 2010).

2. Detailed techniques used to classify the MODIS true colour into six colour classes and primary, secondary and tertiary wet season water types composites

The method used to classify the MODIS true colour images into six colour classes is detailed in Álvarez-Romero et al. (2013, Section 2.1.1) and summarised in Figure 38. This methods involve: (i) converting MODIS true colour images from Red-Green-Blue (RGB) to Intensity-Hue-Saturation (IHS) colour schemes, (ii) the definition of six colour classes corresponding to coastal waters and plume areas and that describe a gradient in the river-borne pollutants, as well as two classes corresponding to non-plume areas (cloud and sun glint signatures), (iii) the creation of spectral signatures for these respective areas, and (iv) the utilisation of the created spectral signature to map the distinct wet season colour classes (six colour classes or 'CC') and water types (primary, secondary, tertiary water types).

Figure 38a: Step 1 and Figure 38b: Steps A and B

Wet season colour classes corresponding to coastal waters, rivers plumes, dense clouds and sun glint were created using a MODIS true colour image with large plumes occurring along the whole Great Barrier Reef coast to ensure that colour variations within coastal waters and river plumes along the latitudinal gradient were incorporated into the spectral signature. The selected image included large areas with no plumes, varied atmospheric conditions (light to dense clouds, haze and sun glint) and sections with no data (not covered by satellite swath). The ArcMap Spatial Analyst isodata clustering tool was used to perform an unsupervised classification of the selected image and create spectral signatures for coastal waters and plume areas as well as the non-plume areas. The resulting structure allowed characterisation of the natural groupings of cells (i.e. pixels within an image) in multidimensional attribute space, that is, IHS and RGB spaces for plumes and clouds/sun glint, respectively.

A supervised classification was used to map the full extent of plumes and to create a mask representing dense clouds and intense sun glint. Supervised classification uses labelled training data (i.e. the colour classes defined in the previous step) to create a spectral signature for each class, which is then used to classify all of the daily input MODIS imagery into wet season colour classes. The classification was selected based on six colour classes as the most appropriate for wet season mapping. These classes represent a gradient in exposure to pollutants typical for the wet season, from highest in class 1 to lowest in class 6 (Devlin et al., 2012). The ArcMap Spatial Analyst maximum-likelihood classification tool (ESRI, 2013) was used to produce: (i) daily six-class plume maps representing variations in satellite-derived water quality parameters (also used to identify the plume boundary (Table 38), and (ii) masks representing dense clouds and intense sun glint, used to eliminate areas with insufficient information to map plumes. A number of images covering different years, regions and months were selected to confirm the plume extent and overall congruence of our classified plume maps against plume maps produced and to visually validate the clouds/sun glint masks.

Figure 38a: Step 2 and Figure 38b: Step C

Weekly six-class composites were created to minimise the amount of area without data per image due to masking by dense cloud cover, common during the wet season, and intense sun glint (Álvarez-Romero et al., 2013). The minimum colour-class value of each cell/week is used to map the colour class with the highest level of exposure to pollutants for each week (i.e. assuming the colour classes represented a gradient in exposure to pollutants).

Figure 38a: Step 3 and Figure 38b: steps D and E

The six colour classes were further reclassified into three plume water types corresponding to the three wet season water types (primary, secondary and tertiary water types) defined by, for example, Devlin and Schaffelke (2009) and Devlin et al. (2012). The turbid sediment-dominated waters or primary water type is defined as corresponding to colour classes 1–4; the productive waters or secondary water type is defined as corresponding to colour class 5; and the tertiary water type is defined as corresponding to colour class 6. Land is removed using a shapefile of the Great Barrier Reef marine natural resource management boundaries (source: GBRMPA, Great Barrier Reef: Features) and by assigning 'No Data' values to any pixels

outside of the shapefile boundaries (including land and offshore areas outside of the Great Barrier Reef Marine Park).

Weekly composites were thus overlaid (i.e. presence/absence of primary, secondary or tertiary water type) and normalised, to compute annual normalised frequency maps of occurrence of primary, secondary or tertiary water type (hereafter annual primary, secondary or tertiary water frequency maps). Mean long-term frequency composites of occurrence of primary, secondary or tertiary water types are created by overlaying several years of frequency maps in ArcGIS and calculating the average frequency values of each cell/year.

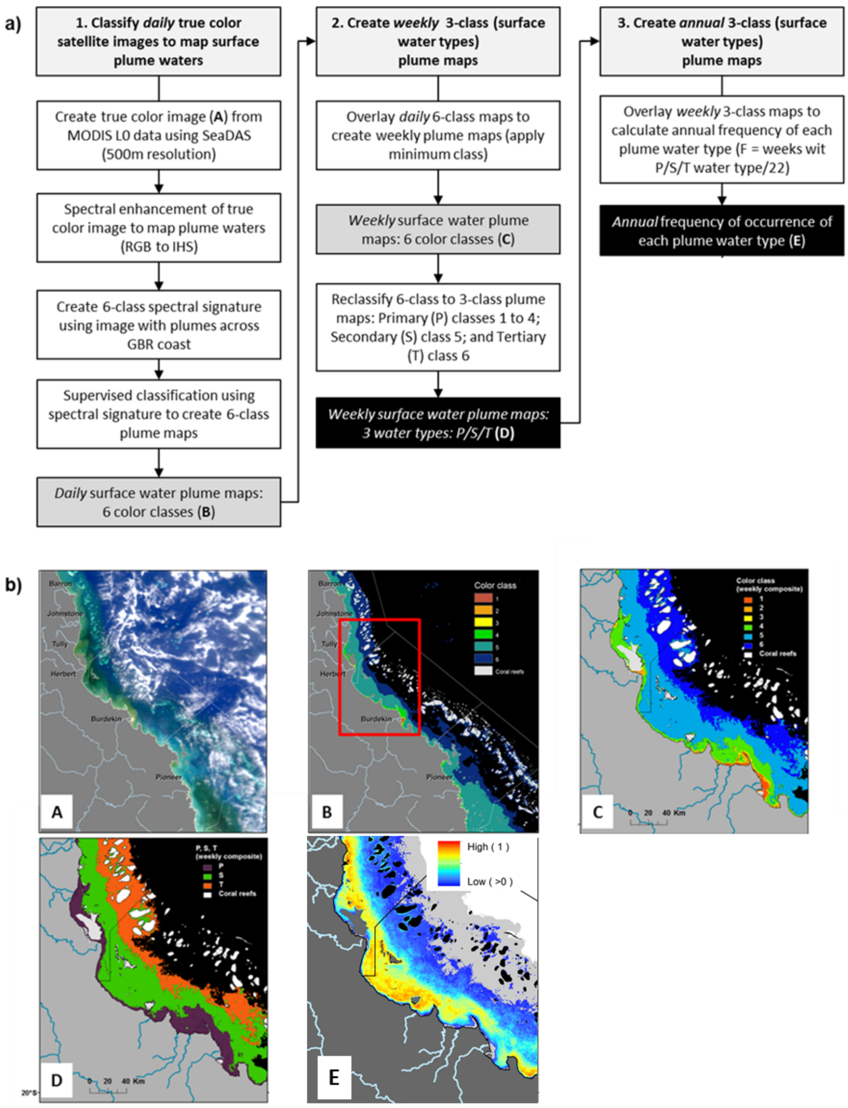

Figure 38. Summary of the process followed to build wet season water type maps with examples of inputs and outputs. (a) Plume mapping process: different shadings represent steps (light grey), analyses within steps (white), intermediate outputs (dark grey) and final outputs (black); (b) A: MODIS-Aqua true colour image used to create the spectral signature defining six colour classes for GBR plumes (25/01/2011), B and C: daily six-colour class map (25/01/2011) and weekly composite (19–25/01/2011) of six-class map; D: reclassified map into weekly primary (P), secondary (S) and tertiary (T) composite (19–25/01/2011); E: Frequency of occurrence of the secondary water type in 2011; Figures C–E are zoomed in the Tully-Burdekin area (see red box on panel B). Modified from Álvarez-Romero et al. (2013) and Devin et al. (2012; 2013).

Table 38. Selected satellite-derived (Level 2 or L2) parameters mapped from MODIS satellite imagery. The gradient in L2 parameters is indicative of variations in water quality within plumes as described in Devlin et al. (2012) and thus was used to identify the plume boundary and to represent the potential dispersal of pollutants (see Álvarez-Romero et al. [2013] for the references in the Table).

L2 parameter	Description	Application in plume mapping	Algorithms
nLw_{667}	Normalized water-leaving radiance at 667 nm	Normalized water-leaving radiance at higher wavelengths (555 nm or 667 nm) is sensitive to suspended sediment in the water column and is correlated with plume location (Salisbury et al., 2004; Thomas and Weatherbee, 2006). We use this parameter as a proxy for TSS and to approximate the dispersal of sediment within the plume.	Defined as upwelling radiance just above the sea surface, in the absence of an atmosphere, and with the sun directly overhead (Gordon and Wang, 1994).
adg_{443}	Colored dissolved organic and detrital matter (CDOM + D) absorption coefficient at 443 nm	CDOM has been proposed as a proxy for salinity and thus been used to estimate the maximum extent of river plume influence (Schroeder et al., 2012). We use it to estimate the dispersal of dissolved inorganic nitrogen assuming conservative mixing.	Derived using the quasi-analytical algorithm (QAA) model (Lee et al., 2002), which estimates absorption due to gelbstof and detrital material at sensor wavelength of 443 nm.

3. Derived spatial layers used in the current risk assessment

Two spatial layers were derived from the *frequency of wet season water type* composites and were used for the current risk assessment:

i. *The wet season water type (primary + secondary) frequency map ($Freq_{P+S}$):* Represents the long-term frequency of exposure to DIN and TSS–enriched surface waters during the wet season. It was used in the assessment of the likelihood of exposure of marine ecosystems to nutrients and sediments, respectively.

This spatial layer represents the long-term (2003-2016) frequency of exposure to the combined primary and secondary wet season water types. This layer highlights the areas of the Great Barrier Reef with the greatest probably of being exposed to DIN–enriched (DIN >23µg L^{-1}) and TSS-enriched (TSS >5.5 mg/L i.e. two times the wet season guideline of 2.4 mg/L, GBRMPA, 2010) waters during the wet season (Figure 37). The long-term composite is normalised to the maximum value, with a final value between 0 (lowest) and 1 (highest) allocated to each pixel.

ii. *TSS exposure (Exp_{TSS}):* Represents the long-term (2003-2016) frequency of exposure TSS-enriched surface waters assessed against the TSS Water Quality Guideline to represent the magnitude and duration of TSS exceedance in the wet season. It was used in the assessment of the likelihood of exposure of marine ecosystems to sediments.

This layer is derived from the long-term frequency of exposure to the primary, secondary and tertiary water types, which represent the likelihood of exposure, as well as the long-term TSS value measured *in situ* in each of the wet season water types (Figure 37) relative to the wet season Great Barrier Reef Water Quality Guidelines for TSS (2.4 mg/L, GBRMPA, 2010), which represent the magnitude of the exposure. For each cell (pixel) of the GBR assessment area:

$$Exp_{TSS} = Exp_{TSS\_P} + Exp_{TSS\_S} + Exp_{TSS\_T}$$

With: $Exp_{TSS_P} = ((TSS_P - TSS_{guideline}) / TSS_{guideline}) \times Freq_P$; $Exp_{TSS\_S} = (TSS_S - TSS_{guideline}) / TSS_{guideline}) \times Freq_S$ and $Exp_{TSS\_T} = ((TSS_T - TSS_{guideline}) / TSS_{guideline}) \times Freq_T$;

And TSS_P, TSS_S and TSS_T are the long-term mean TSS values measured in the primary ($TSS_P = 12.1$mg/L), secondary ($TSS_S = 5.5$mg/L) and tertiary ($TSS_T = 4.4$mg/L) water types during the wet season, $TSS_{guideline}$ is the wet season GBR Water Quality Guidelines for TSS ($TSS_{guideline} = 2.4$ mg/L) and $Freq_P$, $Freq_S$ and $Freq_T$ are the

long-term frequency of exposure (cell-specific) to the primary, secondary and tertiary water types, respectively.

Note: A DIN exposure (Exp_{DIN}) layer was not produced similarly to the Exp_{TSS} layer as there is no wet season GBR Water Quality Guidelines for DIN.

4. Anthropogenic pollutant loading (DIN and TSS)

The anthropogenic pollutant loading (DIN and TSS) composite represents the dispersion of end-of-catchment DIN loads during the wet season, calculated as an anthropogenic influence by comparing the difference between long-term current (2003-2016) average DIN loading with a pre-development DIN load scenario. It highlights the areas of greatest change with current land-use characteristics. This layer was produced using an ocean colour (satellite) based model and outputs were normalised to the maximum value across the Great Barrier Reef, between 0 (lowest) and 1 (highest).

An ocean colour based model was used to estimate the dispersion of DIN ($\mathrm{DIN} = NH_4^+ + NO_2^- + NO_3^-$) and TSS delivered by river plumes to Great Barrier Reef waters (Lønborg et al., 2016). The following summary describes the DIN modelling, and similar concepts were applied for TSS. *A full description of the method is included in Appendix 1.10 of Lønborg et al. (2016).*

This model, built on Álvarez-Romero et al. (2013), combines *in situ* data, moderate resolution imaging spectroradiometer (MODIS satellite) imagery and modelled annual end-of-catchment DIN loads from the Great Barrier Reef catchments. Monitored end-of-catchment DIN loads provide the amount of DIN delivered to the Great Barrier Reef, *in situ* data provides the DIN mass in river plumes, and satellite imagery provides the direction and intensity of DIN mass dispersed over the Great Barrier Reef lagoon. The eReefs hydrodynamic model also provides an estimate of the boundary of plume extent in the wet season. This model produces annual maps of average DIN concentration in the Great Barrier Reef waters. Maps are in a raster format, which is a spatial data model that defines space as an array of equally sized cells arranged in rows and columns (ESRI, 2013).

The model has four main components: (i) modelling of individual river plumes, (ii) DIN dispersion function, (iii) DIN decay function, and (iv) mapping of DIN concentration over the Great Barrier Reef lagoon. The conceptual model in Figure 39 shows how each model component is set up and how they are combined to produce the DIN dispersion maps. The basic idea of the DIN dispersion maps is to produce river plume maps for each individual river in the model. The end-of-catchment load of each river can then be dispersed over its individual river plume. To control this dispersion, a relationship based on the mass proportion of DIN in each plume colour class determined at the Great Barrier Reef scale is used. To account for potential DIN uptake, the ratio between an *in situ* DIN x salinity relationship and the theoretical DIN decay due to dilution (i.e. freshwater – marine water mixing) is used. This ratio defines a DIN decay coefficient, which is multiplied by the dispersed DIN load. After the load has been dispersed over each individual river plume, and corrected for DIN uptake, the resultant dispersed DIN from each river is summed together to represent the total annual DIN dispersion over the Great Barrier Reef lagoon discharged by the rivers.

The method developed for the dispersion of land-based DIN was also applied for TSS. Details of the methods used for this study are presented in Lønborg et al. (2016, Appendix A1.10).

For this assessment, the difference between the estimated wet season DIN concentration and TSS concentrations in the Great Barrier Reef lagoon for the 2016 water year (1 October to 30 September) and compared to the pre-development loads were calculated. This can be interpreted as anthropogenic DIN or TSS concentrations, highlighting the areas of greatest change with current land-use characteristics.

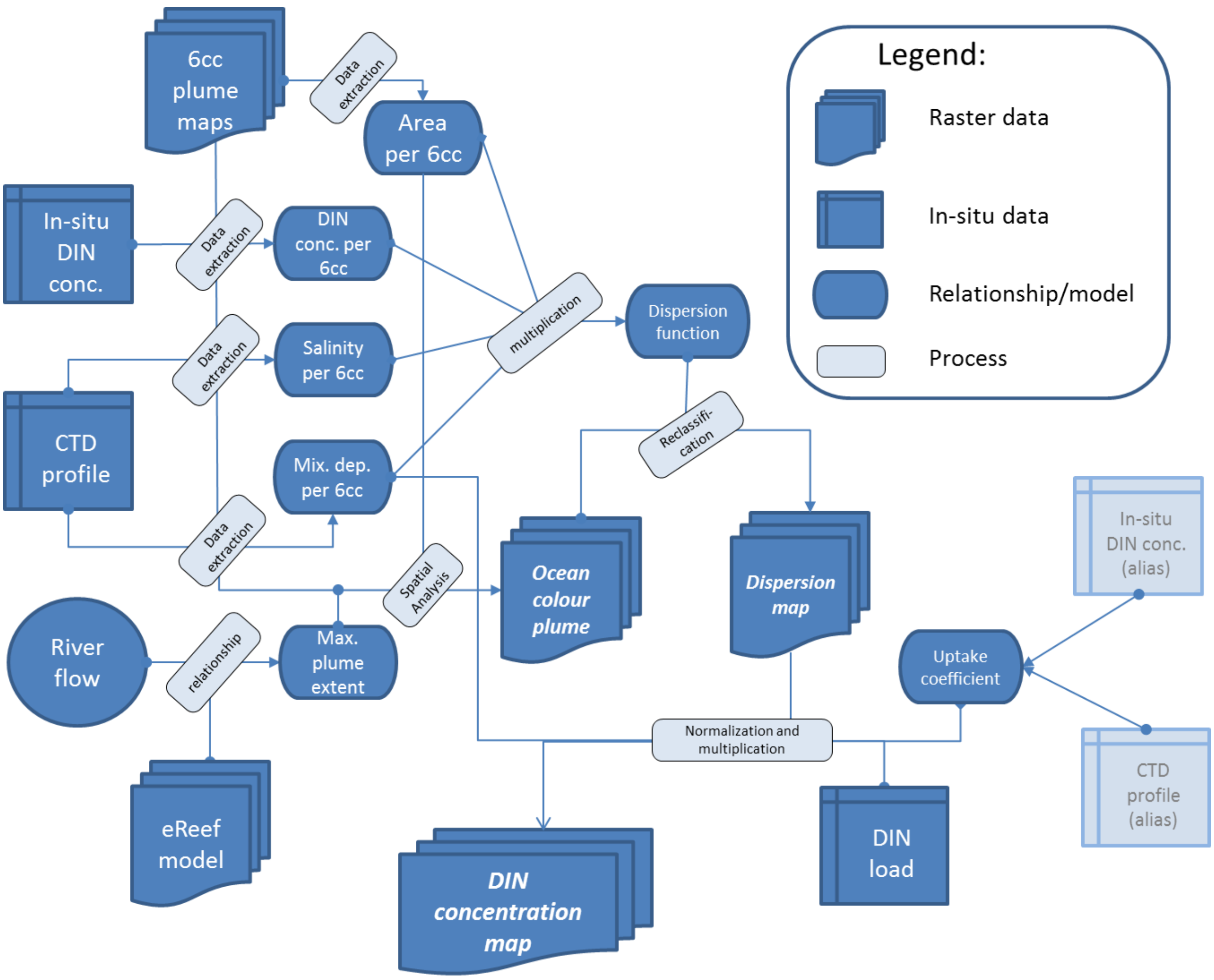

Figure 39. Conceptual model for DIN concentration load mapping. Note: 6cc = six colour class classification. See text for explanation.

5. eReefs model outputs: Predicted difference between annual average chlorophyll *a* concentrations and light attenuation conditions in the baseline and pre-development scenarios

These spatial layers represent the difference between current (2011-2014) annual average and pre-development load scenarios of (i) concentrations of nutrients (measured as Chl-*a*) in the water column, and (ii) turbidity measured as light attenuation, as a measure of annual water quality characteristics.

The eReefs coupled hydrodynamic-biogeochemical model (Baird et al., 2016; Figure 40) was forced with the Source Catchments modelled estimates of current sediment and nutrient load estimates (using the baseline model output which is the 2012-2013 management practice applied in the 2014-2015 Source Catchments model run) and pre-development sediment and nutrient loads (based on pre-development land use). In this assessment, the resulting conditions of chlorophyll *a* and light attenuation were used to predict the areas of influence of river-derived pollutant loads in the Great Barrier Reef. The difference between the two simulations was used to represent anthropogenic DIN and TSS end of catchment load influences by subtracting the pre-development model output from the maximum annual average chlorophyll *a* concentrations (or light attenuation) in the four-year modelling period (2011-2014).

The output from the eReefs biogeochemical model (chlorophyll *a* and light attenuation) was generated in the multi-dimensional NetCDF format at 4 km resolution on a curvilinear grid. The NetCDF files were converted and resampled to the common 0.01 decimal degree (dd) grid via the following steps: (i) the data was imported to a point feature layer and saved in an ArcGIS geodatabase, (ii) converted from points to a triangular irregular network (TIN), and (iii) from TIN to 0.01 dd raster, (iv) masked to the area bounded by

the eReefs data (i.e. pixels at this stage are non-null only to the area enclosed by the outermost grid cell centres of the model data), (v) interpolated outward at the edges by twice replacing null pixels with the 5x5 px focal mean of the neighbouring pixels, and (vi) masked to the rasterised area of the Great Barrier Reef Marine Park plus additional Burnett Mary Water Quality Improvement Plans Region. The outputs were normalised to the maximum value, between 0 (lowest) and 1 (highest) and each pixel (size) was allocated a normalised value.

It should be noted that the application of the 4 km modelled output results in large interpolation of data across the Great Barrier Reef. This coarse grid size is particularly important along the coastline, where shallow waters and resuspension events can dominate conditions and also where intertidal seagrass beds are often located. This limitation is likely to result in underestimates in the calculation of potential exposure of seagrass to TSS and DIN. This is also relevant to coral reefs, although there are comparably smaller areas of reefs in these coastal waters.

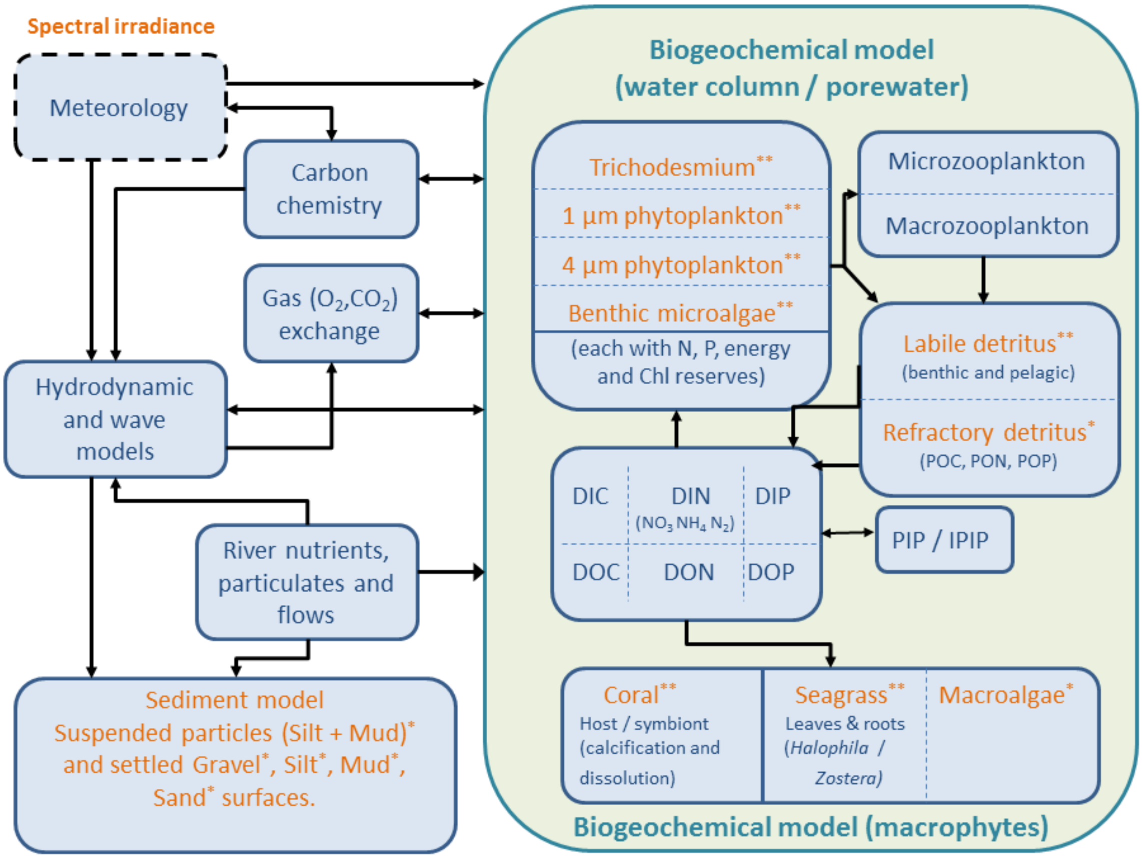

Figure 40. Conceptual framework of the eReefs coupled hydrodynamic-biogeochemical model. The model contains a hydrodynamic, sediment and biogeochemical model. The orange variables are optically active (i.e. either scatter or absorb light), influencing the vertical attenuation of light and the bottom light field. The model is forced by 21 rivers with nutrient and sediment loads (Baird et al., 2016) using the Source Catchments model.

6. Combining the data

To combine the input data to develop combined likelihood of exposure maps, all input data was converted to a common raster grid with a resolution of 0.01 x 0.01 decimal degrees (dd) in the area bounded by longitude 142.5°E to 154°E and latitude 10.68°S to 26°S. Data derived from MODIS RS imagery (i.e. TSS exposure, PS frequency, and DIN and TSS loading maps) was resampled from 0.0053 dd to 0.01 dd

resolution. eReefs model data was resampled from 4 km resolution to 0.01 dd (approximately 1 km grid) as described above.

Since the various input layers are assessed using different units and scales (i.e. they have different maximum values), the values for all indicators are divided by the maximum value for the indicator, resulting in a normalised scale from 0 to 1. Calculations on input layers were performed in ArcGIS 10.2 using the Raster Calculator in ArcGIS Spatial Analyst. For the Likelihood layer, the normalised input layers were added together and the resulting layer divided by its maximum value to again normalise from 0 to 1. The final output was classified into six final categories.

The areas of coral reefs, surveyed seagrass, modelled deepwater seagrass and total areas in each likelihood category in each Marine Zone were calculated by intersecting the final likelihood spatial layers with the spatial layers for each habitat type. All area calculations were performed on ArcGIS geodatabase feature classes (i.e. in vector format—similar to shapefiles). Combined layers were converted from raster to polygon without simplification (i.e. the output polygons conform to the pixel edges). Area calculations were made using habitat shapefiles to clip the risk features (i.e. a 'cookie cutter' approach), then habitat, risk and habitat x risk layers were clipped using the Marine Zone areas. The feature classes were projected using the Australia Albers Equal Area Conic projection (spatialreference.org/ref/sr-org/6644/) and area calculation was performed using 'calculate geometry' in ArcGIS 10.2 (ESRI, 2013). All data were exported in CSV format and compiled into summary tables using the Pandas module for Python (PythonWin 2.7.3).

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Appendix 4: Case study of the assessment of pesticide risk in the Great Barrier Reef

Case study: Modelling pesticide exposure of five PSII herbicides in the Marine Zone using the eReefs hydrodynamic model

The following approach highlights a possible way forward to determine herbicide exposure and risk in the Great Barrier Reef using daily monitored concentration data (supplied by the Great Barrier Reef Catchment Loads Monitoring Program) with the eReefs hydrodynamic model. The aim of the case study was to model mixtures of five PSII herbicides (ametryn, atrazine, diuron, hexazinone and tebuthiuron) that posed a potential risk to marine ecosystems during a run-off event (i.e. a 4–5-day period) for the 2014 water year (1 October 2013 to 30 September 2014). Pesticide monitoring data collected from end-of-system sites in 15 catchments were first converted to an additive PSII herbicide concentration (diuron-equivalent concentration), using the toxic equivalency factors in King et al. (2016a) (Figure 41). Diuron-equivalent concentrations were then assessed for their potential risk to the marine ecosystem. Catchments in which the diuron-equivalent concentrations exceeded the proposed diuron protective concentration for 99% of species (PC99) (0.08 µg/L) (King et al., 2017a; King et al., 2017b) were considered a potential risk to the marine area and were modelled; all other catchments were discarded from further analysis. The marine area of the Great Barrier Reef World Heritage Area is considered as a high ecological value ecosystem and therefore has the highest level of protection, that is, the PC99 (ANZECC and ARMCANZ, 2000). We note that the diuron PC99 for the Great Barrier Reef is currently the subject of further examination where it is yet to be decided what is the most appropriate value to adopt given the differences between the diuron freshwater high reliability (0.08 µg/L) and the diuron marine low reliability (0.39 µg/L) proposed guideline values (King et al., 2017b). Here we adopt the more conservative value.

The Haughton, Fitzroy, Burnett and Mary rivers were the only rivers in which diuron-equivalent concentrations did not exceed the diuron PC99 (Figure 41). Of the catchments that did exceed the diuron PC99, the Mulgrave, Tully, Herbert and Pioneer rivers were selected for this case study to assess the spatial exposure of PSII herbicides in the marine area using the hydrodynamic model. Most commonly, the diuron-equivalent concentrations exceeded the diuron PC99 over a 4–5-day period (with some exceptions). In that regard, we chose a 4–5-day period for each of these four rivers where the diuron-equivalent concentrations exceeded the diuron guideline value and where sufficient discharge occurred which would result in a sizeable flood plume offshore.

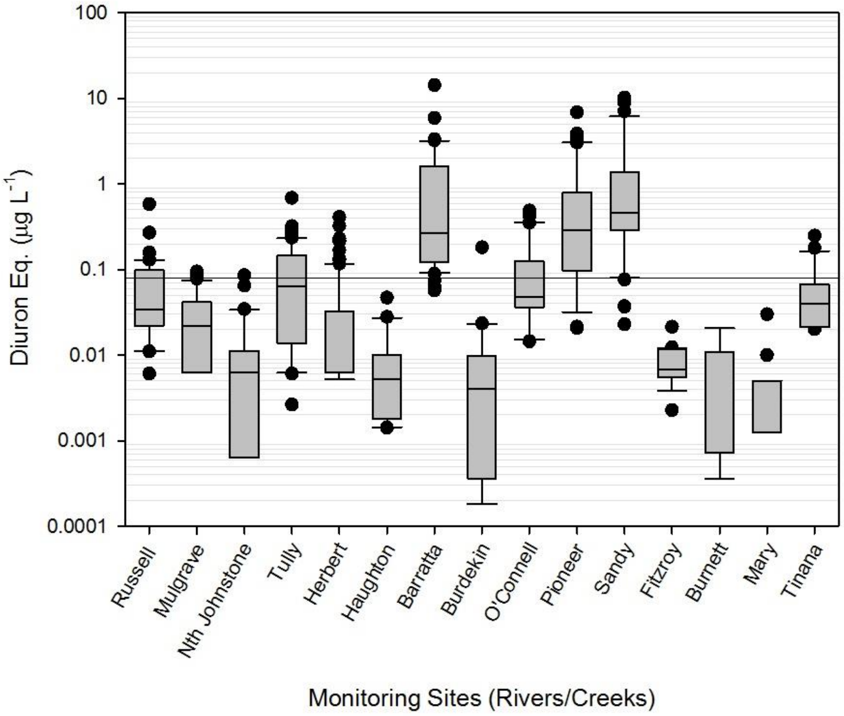

Figure 41. Box plots of diuron-equivalent (Eq.) concentrations (µg/L) calculated from pesticide monitoring data of five PSII herbicides (ametryn, atrazine, diuron, hexazinone and tebuthiuron) across 15 catchments in 2013-2014. The horizontal line across the graph represents the proposed diuron PC99 (0.08 µg/L). Boxes represent the distribution of diuron-equivalent concentrations with the top and bottom boundaries of the boxes representing the 75th and 25th percentiles of the distribution, the lines within the boxes representing the median, the whiskers above and below the boxes representing the 90th and 10th percentiles and the dots representing outliers.

The eReefs hydrodynamic model was run for the selected data using the dilution tracer; that is, the tracer is 'released' into the end-of-river with a value of 1 and modelled offshore until the value becomes 0.01, or 1% of the original value. Using the dilution tracer to map changes in diuron-equivalent concentrations in flood plumes assumes a conservative mixing behaviour of the PSII herbicides once they reach the Great Barrier Reef lagoon. This assumes conservative mixing behaviour is supported by marine monitoring data (e.g. Lewis et al., 2009; C. Gallen, unpublished data) and the fact that PSII herbicides are predominantly transported in the dissolved phase (Davis et al., 2012; Packett, 2013). To take into account the antecedent herbicide concentrations in the lagoon, we use an iterative approach to estimate the cumulative diuron-equivalent concentration from inputs over multiple days (see Figure 42). First, the day 1 diuron-equivalent concentrations at the river mouth were modelled mixing into the lagoon with a seawater end member concentration of '0'. The concentration of this tracer was then modelled over the following four days with no new tracer (i.e. 0 value) being released from the river into the Great Barrier Reef lagoon to calculate the influence of the contribution of this day over this period. Day 2 concentrations were then modelled to take into account the residual concentrations in the Great Barrier Reef lagoon from the day 1 input as well as the 'new' concentration released into the Great Barrier Reef on that day. This was repeated for days 3, 4 and 5 (Figure 42). Using this approach, the diuron-equivalent concentrations can be modelled on a 3D scale (lateral and vertical) of the water column, to estimate the space and depth of exposure (note that the vertical mixing outputs are not shown in the analysis). The pixel size for the eReefs model of the Great Barrier Reef lagoon is represented by a 500 × 500 m grid area, thus the diuron-equivalent concentrations represent an average concentration for each pixel. Hence, it is likely that areas within a pixel have much higher (or lower) concentrations than the reported average value.

The mapping outputs (Figure 43; Figure 44; Figure 45; Figure 46) display the maximum and 3-day mean diuron-equivalent concentrations discharged by each river during the selected 4–5-day period. The maps indicate that the proposed diuron PC99 was not exceeded by the modelled diuron-equivalent concentrations offshore from the Russell-Mulgrave, Tully, Herbert and Pioneer rivers. However, given that the diuron-equivalent concentrations did exceed the proposed diuron PC99 at the end-of-system catchment monitoring sites for all of these rivers, exceedances of the proposed diuron PC99 may have occurred in part of the pixel (500 m x 500 m grid) at the mouth and river estuary. Note that the sampling sites on the Tully, Mulgrave and Herbert rivers do not capture the full agricultural area where herbicides are applied, and so the actual end-of-river concentration may be underestimated. The modelled 4–5-day periods were first-flush events, where the diuron-equivalent concentrations were highest but the daily discharge was small relative to other events of the season, and therefore dilution with marine waters would occur relatively quickly. The monitored diuron-equivalent concentrations in the larger events were below the proposed diuron PC99, indicating a low probability of ecological risk from PSII herbicides to the marine area. In that regard, it would be useful to examine periods where the first-flush event of the season also coincided with larger river flows. Previous monitoring data show periods (i.e. 2004-2005) where a much larger area of the Great Barrier Reef lagoon would have exceeded guidelines (e.g. see Lewis et al., 2009; Lewis et al., 2012). Unfortunately, the eReefs hydrodynamic model does not go back to this period and so a validation between model predictions and monitoring data cannot be performed at this stage. Furthermore, smaller streams discharging into the lagoon that are not modelled by eReefs (e.g. Sandy Creek, Barratta Creek) would also increase the area of PSII herbicide exposure in the Great Barrier Reef coastal zone. This is a clear limitation in the analysis, although there are currently no options to resolve this issue.

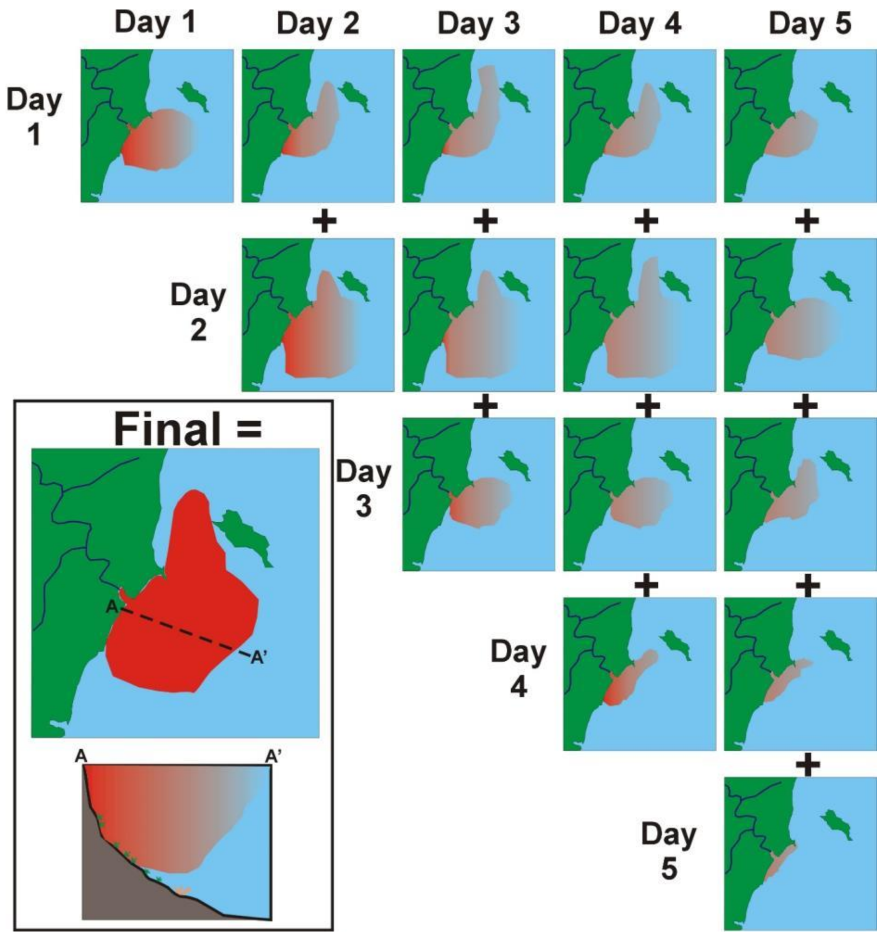

Figure 42. Schematic diagram to summarise the iterative methods used to generate the spatial risk maps of additive PSII herbicide concentrations in flood plumes of the GBR lagoon over a five-day period. The red shading in each map represents dilution of PSII herbicide concentrations (herbicide-equivalent relative to diuron). The final product (map) outlines the total area where the diuron-equivalent concentrations exceeded the proposed diuron PC99 value (0.08µg/L), laterally and vertically through the water column.

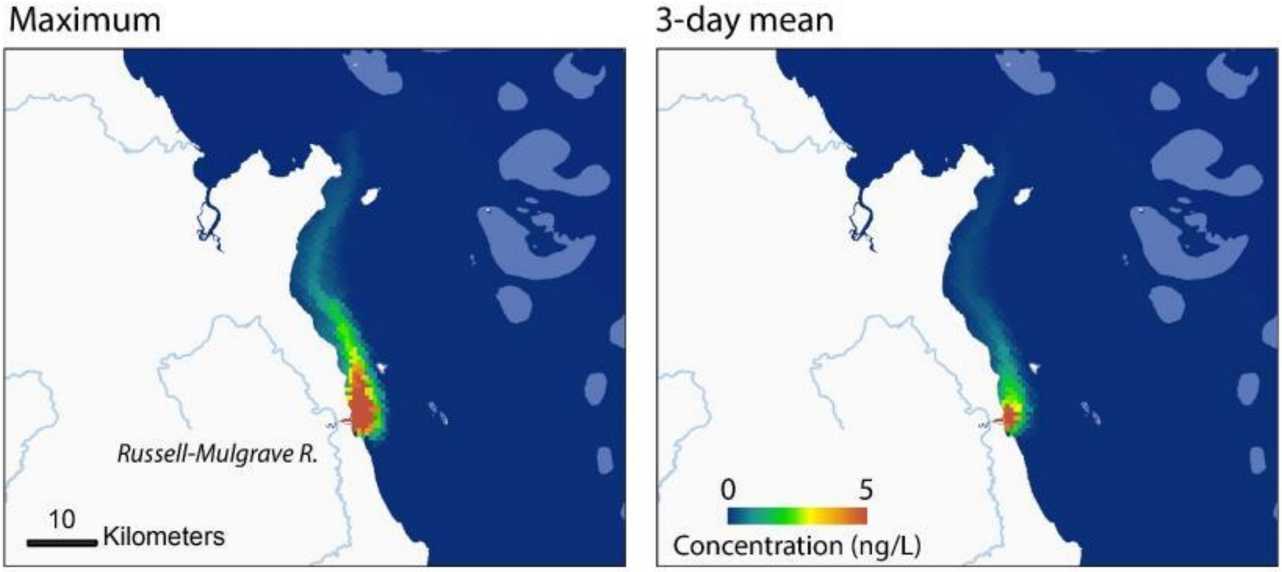

Figure 43. Modelled herbicide-equivalent (relative to diuron) concentrations offshore from the Russell-Mulgrave River including maximum and 3-day mean concentration for the period in the 2014 water year coinciding where the highest concentrations were measured.

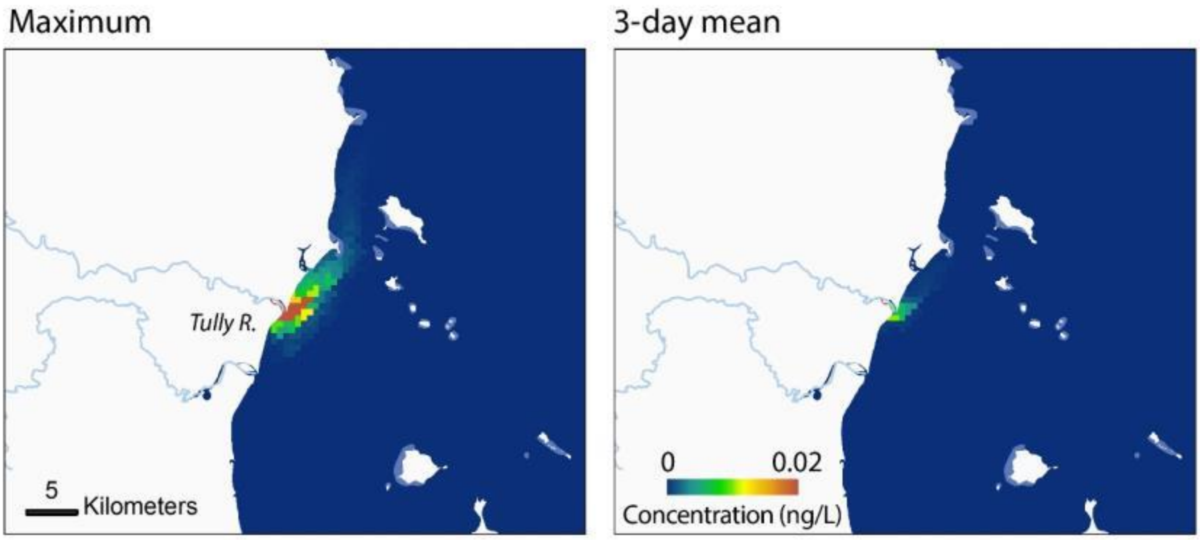

Figure 44. Modelled herbicide-equivalent (relative to diuron) concentrations offshore from the Tully River including maximum and 3-day mean concentration for the period in the 2014 water year coinciding where the highest concentrations were measured.

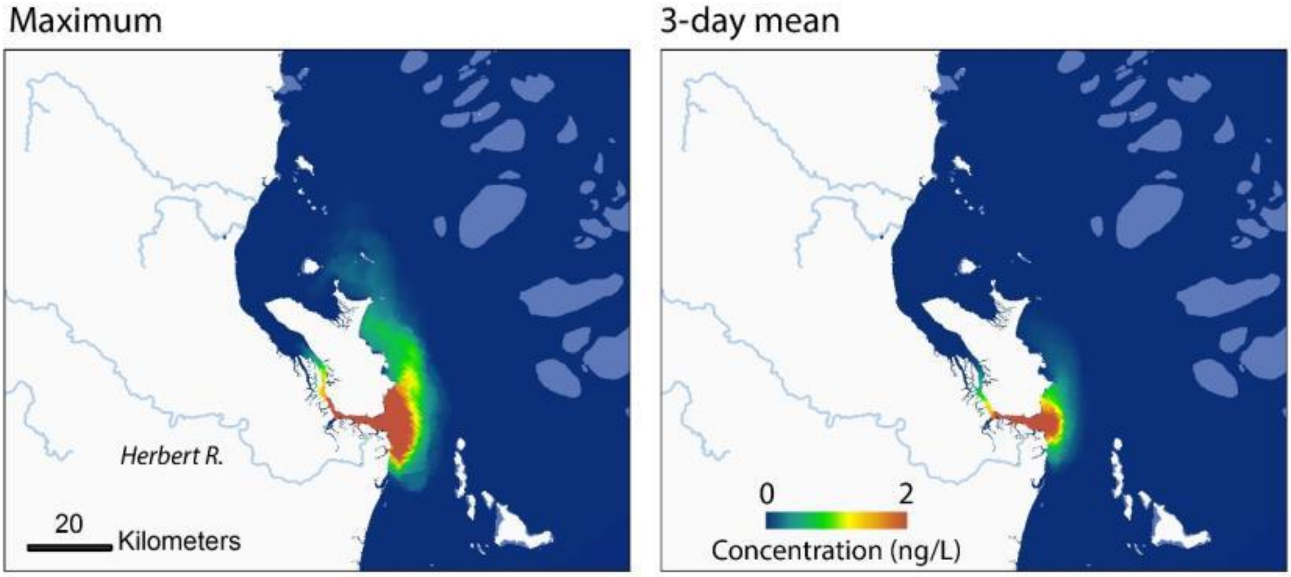

Figure 45. Modelled herbicide-equivalent (relative to diuron) concentrations offshore from the Herbert River including maximum and 3-day mean concentration for the period in the 2014 water year coinciding where the highest concentrations were measured.

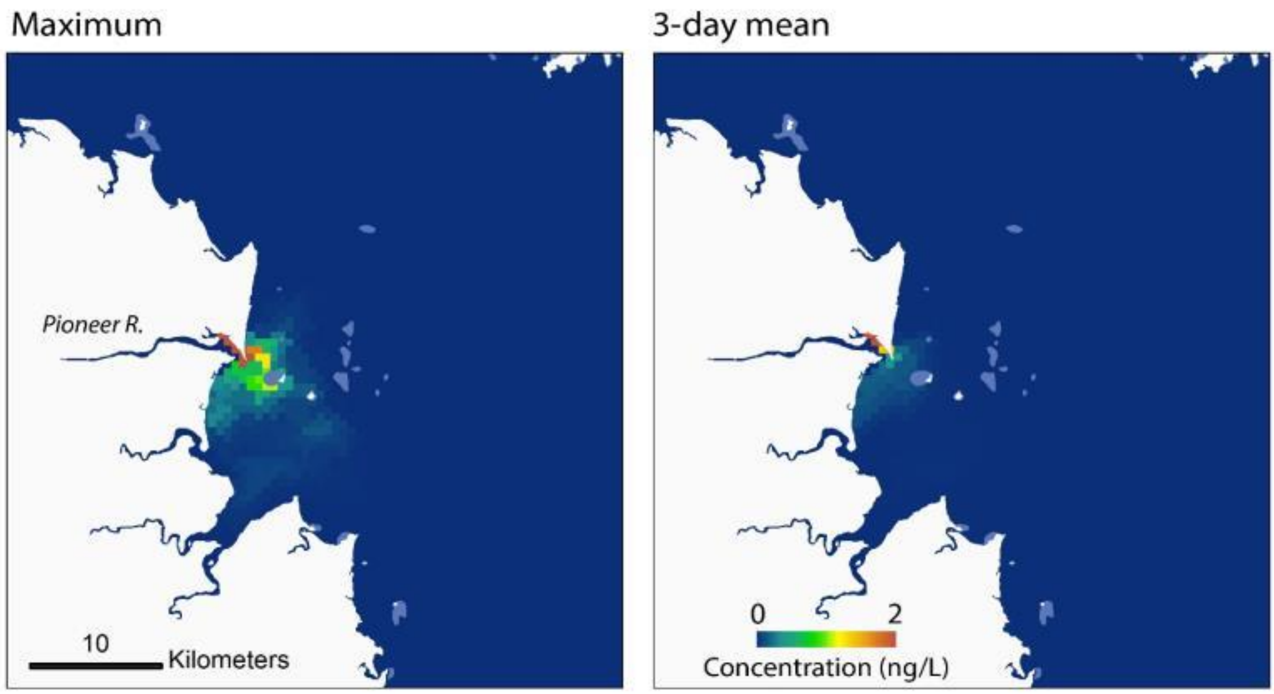

Figure 46. Modelled herbicide-equivalent (relative to diuron) concentrations offshore from the Pioneer River including maximum and 3-day mean concentration for the period in the 2014 water year coinciding where the highest concentrations were measured. We note that this model only examines the Pioneer River and does not include the streams to the south of the river such as Bakers or Sandy creeks. Monitoring at these sites suggest that additive PSII herbicide concentrations would also exceed guideline values at the mouth of these streams and hence concentrations would extend further south if all streams could be modelled.

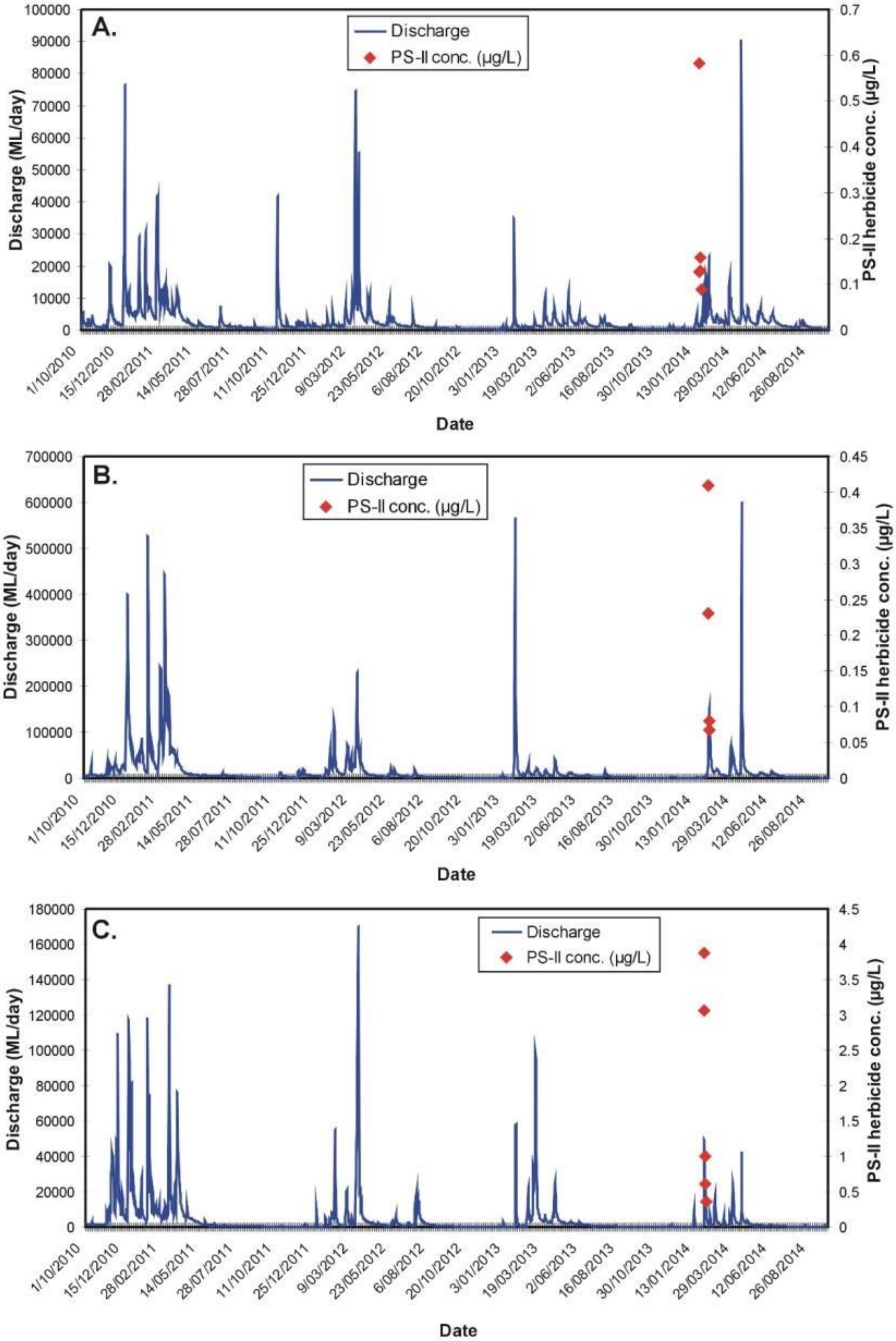

Figure 47. Daily discharge in ML over four water years (2010-2011 to 2013-2014) highlighting the diuron-equivalent concentrations (PSII herbicide concentrations) over 4-5 day periods where the model was run for the (A) Russell-Mulgrave River (using Mulgrave River at Peets Bridge gauge), **(B)** Herbert River and **(C)** Pioneer River**.** No daily discharge data were available for the Tully River for the 2013-2014 water year at the time of the analysis**.**

The assessment of the 5-day additive PSII herbicide concentration (plotted as maximum values and 3-day mean concentrations) using the hydrodynamic model was a tedious exercise where we could not alter the daily concentrations exported at the mouth of the river and hence had to produce a number of maps which then needed to be combined and further analysed. The CSIRO eReefs team have the capacity to do this analysis by changing the daily concentration and so this option should be explored in the future as longer time periods and additional rivers can then be covered. In that regard, the model could be validated using passive samplers which commonly provide mean herbicide concentrations over 1–2-month deployment periods as well as grab samples taken during flood plumes. We note that a combination of end-of-catchment modelling and monitoring (where available) data would be required to examine herbicide exposure and risk for all 35 basins of the Great Barrier Reef catchment area. Furthermore, the model only covers ~15 of the major rivers of the Great Barrier Reef, and so some of the smaller streams which have very high herbicide concentrations but lower discharge (e.g. Sandy Creek, Barratta Creek) would not be considered in the model.

This approach highlights a possible way forward to determine herbicide exposure and risk in the Great Barrier Reef using a combination of daily monitoring/modelling data with a hydrodynamic model. However, further discussions would be needed through an expert panel to decide how the metric would be constructed including what constitutes an A, B, C, D and E rating. There are two likely scenarios to consider which include (i) applying similar metric rules for exceedance of TSS and chlorophyll *a* guidelines, or (ii) a multiple species-Potentially Affected Fraction (ms-PAF) type framework to examine areas where >20%, 10–20%, 5–10%, 1–5% and <1% of species would be affected. Both scenarios could transfer directly into an A–E rating, and there are pros and cons for both methods. The guideline exceedance metric is the most simplistic approach and quite straightforward to communicate and would form a consistent approach with the other metrics for TSS and chlorophyll *a*; however, it would not account for non-PSII herbicides. Furthermore, concentrations measured in the Great Barrier Reef lagoon are commonly at the lower range where effects to biological productivity may be expected but not in the range which would result in high order to catastrophic effects such as immediate coral bleaching or organism mortality. However, this approach does not account for the complexity of the influences of multiple pulses of herbicide exposure over a wet season where the productivity of the ecosystem may not recover in between exposure periods. In that regard, the ms-PAF approach can account for this influence of multiple exposures and provide an overall rating for each pixel based on not only the percentage of species affected but also the compromised functioning of the ecosystem.

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