

Exploring trading in water quality credits as a cost-effective approach for managing water quality in the Great Barrier Reef

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Cover photographs: (front) Wetland on cane farm. Image: Joshua Dyke; (back) Gullying in the Burdekin region. Image: Graeme Curwen.

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ACRONYMS

APVC	Annualised Present Value Cost
CE	Cost Effectiveness
CVaR	Conditional Value at Risk
DAWE	Department of Agriculture, Water and the Environment
DIN	Dissolved Inorganic Nitrogen
DoEE	Department of the Environment and Energy
EA	Environmental Authority
EIAT	Economic Impact Analysis Tool
EoC	End of Catchment
EP Act	Environmental Protection Act
EPBC	Environmental Protection and Biodiversity Conservation (Act)
EPP	Environmental Protection Policy
ERA	Environmentally Relevant Activity
FTE	Full-Time Equivalent
GBR	Great Barrier Reef
GBRWhA	Great Barrier Reef World Heritage Area
GBRMPA	Great Barrier Reef Marine Park Authority
GRP	Gross Regional Product
I-O	Input Output Analysis
MAC	Marginal Abatement Cost
MIP	Major Integrated Project
MNES	Matter Of National Environmental Significance
MU	Management Unit
NESP	National Environmental Science Programme
NRM	Natural Resource Management (Group)
OUV	Outstanding Universal Value
P2R	Paddock to Reef
PN	Particulate Nitrogen
PP	Particulate Phosphorus
PSWQ	Point Source Water Quality (Offset Policy)
QLUMP	Queensland Land Use Mapping Program
RRRC	Reef and Rainforest Research Centre Limited
SPP	State Planning Policy
STP	Sewage Treatment Plant
SWIM	Statewide Water Information Management
TA	Tourism Australia
TAC	Total Abatement Cost
TN	Total Nitrogen
TPVC	Total Present Value Cost
TSS	Total Suspended Sediment
TWQ	Tropical Water Quality
UNESCO	United Nations Educational, Scientific and Cultural Organisation
WQIP	Water Quality Improvement Plan
WQO	Water Quality Objective
WSAA	Water Services Association Australia

WTA..... Willingness To Accept
WTP..... Willingness To Pay

ABBREVIATIONS

\$/kgDIN..... dollar per kilogram of dissolved inorganic nitrogen
capex..... capital expenditure
kg..... kilogram
O&M..... operating and maintenance
opex..... operating expenditure
M&E..... monitoring and evaluation
mgN/L..... milligrams of nitrogen per litre
tonne⁻¹..... per tonne

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EXECUTIVE SUMMARY

This National Environmental Science Program (NESP) Tropical Water Quality (TWQ) Hub Project 3.1.6 (hereafter Project 3.1.6) investigates whether trading in water quality credits can provide a cost-effective pathway for *future* economic expansion in the Great Barrier Reef (GBR) catchment *once the Reef 2050 basin-specific water quality objectives have been achieved* (State of Queensland, 2019a). The Project scopes opportunities for water quality credit trading and develops a framework for trading markets in dissolved inorganic nitrogen (DIN) and sediment.

The Australian and Queensland governments have made very significant investments to improve GBR water quality with the aim of achieving defined Reef 2050 Water Quality Improvement Plan (WQIP) basin-specific water quality objectives (State of Queensland, 2018). Once the Reef 2050 WQIP water quality objectives have been achieved future economic development along the Reef coastline could potentially jeopardise these hard-won gains unless cost-effective approaches can be found for facilitating economic expansion with no net decline in GBR water quality. Project 3.1.6 considers the potential for water quality credit trading to address this challenge by estimating the potential supply of and demand for DIN and sediment credits in representative catchment settings along the GBR coastline. Key findings are:

Supply of DIN credits

Using catchments in the Wet Tropics as examples, we consider potential options for the supply of DIN credits. We assume full adoption of Best Practice ('moderate-low risk') nutrient management in sugarcane production (The Australian and Queensland Governments, n.d.) as a prerequisite for achieving the Reef 2050 water quality targets. Cost modelling from this starting point suggests that post-delivery of the Reef 2050 water quality targets, it will typically cost 60 – 250 \$/kgDIN to supply DIN credits from further practice change in cane¹. Initial cost data from nitrogen treatment systems constructed under the Wet Tropics Major Integrated Project (MIP), and representative average DIN removal rates from the literature, suggest that nitrogen treatment wetlands in the Wet Tropics could supply DIN credits at ~40 – 60 \$/kgDIN, with 1ha of wetland potentially supplying 550 - 750 kg of DIN credits annually². Consequently, in sugarcane catchments, it is likely that constructed treatment wetlands will provide the lowest cost source of DIN credits post-delivery of the Reef 2050 water quality targets.

Demand for DIN credits

As licenced point source emitters, sewage treatment plants (STPs) and aquaculture facilities are potential buyers of DIN credits. We estimated the potential demand for DIN credits from these sources. Results indicate that 22 tonnes, 11 tonnes and 10 tonnes of annual demand for DIN credits could arise in the Herbert, Mulgrave-Russell, and Mossman catchments, respectively, if older-specification STPs sought to purchase DIN credits to avoid the need for

¹ The current Reef Regulations align with Minimum Standard ('Moderate risk') nutrient management practice in cane production (The Australian and Queensland Governments, n.d.). The Queensland government have made a commitment to "not substantially change the minimum practice agricultural standards for five years (from commencement of [the] Amendment Regulations)" (Queensland Government, 2019; p.4, paragraph 4).

² Peer-reviewed DIN removal rates from wetlands constructed under the Wet Tropics MIP should be available during 2021.

upgrading to a de facto discharge standard of 5 mgN/L. Accounting for avoided annualised expenditure on upgrading capital expenditure (capex) and avoided incremental annual operating expenditure (opex), STP operators should be willing to pay c. 45 – 100 \$/kg DIN for the quantities of credits required³. Further south, significant annual demands for DIN offsets from STPs currently discharging at concentrations above 5mgN/L could arise in Townsville, Mackay, Rockhampton, Bundaberg and Gympie.

STPs that already deliver 5 mgN/L discharge performance but are approaching their capacity limits due to population growth may seek to buy nitrogen credits as a temporary measure to defer the cost of capacity augmentation. If large STPs in Townsville or Cairns sought to use this approach, they would generate c. 0.5 tonne of annual demand for diffuse-source DIN credits per 1,000 additional population. In this context, based on cost savings due to deferral of capex, STP operators would potentially be willing to pay up to ~ 70 \$/kg DIN credit⁴.

Supply of sediment credits

We considered potential supply of sediment credits from targeted remediation of alluvial gullies in grazing catchments, using the Bowen-Bogie subcatchments in the Burdekin as an example. Recent results from modelling undertaken for the Landholders Driving Change (LDC) MIP indicate that c. 120 thousand tonnes of sediment credits could be supplied from targeted gully remediation at prices below c. 45 \$ tonne⁻¹ year⁻¹.

Demand for sediment credits

There appears to be only limited demand for sediment credits along the GBR coastline. Defined activities within the Great Barrier Reef Marine Park (GBRMP) can offset unmitigated impacts by in-lieu payments to the Reef Trust's Special Account. The Reef Trust Offsets Plan and Calculator specifies a one-off payment of 464 \$ tonne⁻¹ of unmitigated sediment. This equates to an annualised equivalent of c. \$40 tonne⁻¹ year⁻¹⁵. At this pricing c.85 thousand tonnes of sediment credits could potentially be sourced from gully remediation in the Bowen-Bogie, if sufficient demand arises from coastal development.

Under State Planning Policy, post-construction emissions from urban development sites can be managed by developers making in-lieu payments to local government. We revised the Alluvium Report's (Alluvium, 2016) predictions of sediment credit demand from this source in line with recent estimates of population growth. The resulting estimates of annual demand for sediment credits at major population centres along the GBR coast are very modest: Cairns 55 tonnes, Townsville 28 tonnes, Mackay 16 tonnes, Rockhampton 13 tonnes.

Employment opportunities from water quality credit markets

Applying Flinders University's Economic Impact Analysis Model, we estimated that the in-catchment investment actions specified under the Reef 2050 Water Quality Improvement Plan could potentially create 1083 full-time equivalent positions annually. This equates to approximately 1 full-time position per \$114,000 of annual investment in restoration activities.

³ This allows for a 1.5:1 delivery ratio for offsetting point source emissions by diffuse source credits in Queensland's Point Source Water Quality Offsets Policy (State of Queensland, 2019b).

⁴ These price and quantity estimates are approximate. An STP-specific calculation and discussions with STP operators would be required to generate more precise estimates.

⁵ Annualised equivalent value calculated over 25 years using a discount rate of 7% per annum.

Similar employment opportunities could reasonably be anticipated from in-catchment works to supply nitrogen and sediment offsets to water quality credit markets. This equates to c. 1 equivalent full-time position per 2 tonnes of DIN credits, or per 2,500 tonnes of sediment credits, supplied to the market.

Our findings suggest that – once the Reef 2050 WQIP water quality objectives have been achieved – water quality credit trading has the potential to be a facilitator of cost-effective economic expansion along Queensland’s GBR coast, with no net decline in hard-won water quality. However, there remain few examples worldwide of water quality credit trading markets that have delivered on their potential. Literature identifies several key challenges that must be addressed:

- Credit buyers’ anxiety regarding the longer-term viability and availability of credits.
- Credit suppliers’ concerns regarding high transaction costs and inadequate compensation for exposure to up-front risk – particularly around credit generation from land use change and land remediation.
- The Environmental Regulator’s risk aversion regarding the effectiveness of credit generation from on-ground actions and continuity of the requisite maintenance.

Realising the potential: suggested market configuration

Recognising these challenges, the market configuration in Figure ES1, supported by relevant technological innovations, is recommended for implementing water quality credit trading along the Reef coastline. The elements in Figure ES1 are described in the following paragraphs.

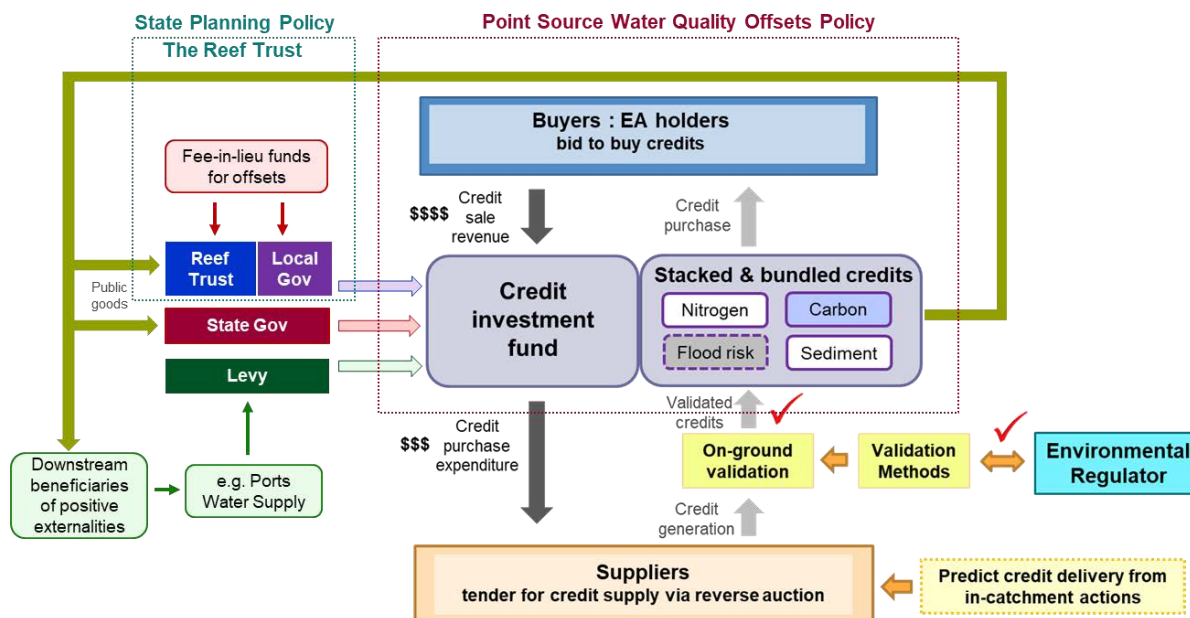


Figure ES1: Suggested configuration for a market in water quality credits to facilitate economic expansion along the Reef coast, whilst also maintaining Reef 2050 water quality outcomes.

In Figure ES1, monies from the Reef Trust Offset Fund and State Planning Policy urban stormwater offsets, combined with funding from state and federal government, provide capital for a water quality credit investment fund. This fund can cover a substantial proportion of the

up-front costs arising from credit generation via land use change and land remediation. Landholders and project proponents tender for offset supply, choosing offset actions from a permitted list appropriate to their catchment and location⁶. With a substantial proportion of major upfront costs already covered by the investment fund, the financial risk to suppliers is considerably reduced; supplier participation should increase as a result. On-ground actions are validated through methodologies approved by the Environmental Regulator. Validated credits from those actions accrue in credit ledgers. The proportion of credits retained by the investment fund relative to the proportion assigned to the landholder/project proponent will reflect the distribution of financial risk. A proportion of credits will generally be assigned to the landholder/proponent as an incentive to continue undertaking necessary maintenance. Periodic spot checks will be carried out for continuing validation of credits in subsequent years; credit validation will be withdrawn if the necessary maintenance has not been completed. Validated nitrogen and sediment credits in the credit ledgers will be eligible for purchase as Point Source Water Quality Policy offsets by buyers who bid for credits in standard fashion⁷. The revenue from credit sales accrues to the credit investment fund and landholders/proponents in proportion to their credit holdings. If investments are focused on relevant catchments, findings from this study suggest that credit buyers' willingness to pay should at least be sufficient to cover the pricing required to elicit credit supply, including an adequate return on the capital injected by the credit investment fund.

If the list of eligible credit generating actions and locations is appropriately informed by state-of-the-art catchment modelling, actions and locations that produce nitrogen and/or sediment credits for the offset markets will also produce additional public good outcomes for downstream beneficiaries (the thick green arrow in Figure ES1). Some credit supply actions will deliver downstream benefits to, for example, water supply companies (via reduced drinking water treatment costs) and ports (through reduced dredging costs). This could potentially be used to support a levy on these beneficiaries as an additional funding stream for the catchment investment fund. Other positive externalities – such as a reduction in the rate of rise and extent of flooding and in-catchment job creation – will benefit multiple downstream beneficiaries. Provision of these public goods strengthens the case for mobilising public funds to initiate the credit investment fund.

A major advantage of adopting the market configuration proposed in Figure ES1 is that the credit investment fund would provide the capital investment injection required to initiate water quality trading at sufficient scale to ensure that adequate quantities of credits would be supplied to the market. This would alleviate credit buyers' concerns regarding inadequate credit supply – which might otherwise severely restrict credit demand. The fee-in-lieu offset funds would generate a 'base load' demand for sediment and nitrogen credits in catchments where urban populations are expanding. The credit investment fund could then engage more heavily to facilitate water quality credit trading in catchments where the need for STP upgrades or expansions may otherwise act as a handbrake on economic development.

The following key technological innovations would further facilitate effective operation of water quality trading markets: a distributed network of water quality sensors, state-of-the-art

⁶ Tenders to supply credits will be submitted via a credit suppliers smart phone 'app'.

⁷ Offers to buy credits will be submitted via a credit buyers smart phone 'app'.

modelling of the predicted effectiveness of in-catchment actions to reduce pollutant loads, and smart phone 'apps' to assist credit suppliers' and credit buyers' selling and buying.

Recommendations for future research

- Investigation of DIN transport via drainage/groundwater pathways.
- Investigation of bioavailable nitrogen reductions through sediment abatement.
- Catchment water quality modelling to incorporate the latest advances in DIN transport and reductions in bioavailable nitrogen from sediment reductions.
- Validation of the cost-effectiveness of DIN removal by constructed treatment systems.
- Validation of the cost-effectiveness of sediment reductions from gully remediation.
- Continuing technological developments:
 - Low-cost distributed real-time water quality sensing – particularly for bioavailable nitrogen.
 - Smart phone apps for credit suppliers and credit buyers.

1.0 INTRODUCTION

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The Great Barrier Reef is a unique ecological asset of global importance and an economic asset that has been estimated to generate more than \$6 billion of revenue per annum for the Australian economy (Deloitte Access Economics 2017). The future of the Reef is threatened by increasing sea temperatures that cause coral bleaching, ocean acidification that inhibits coral formation, increasing frequency and severity of cyclones that physically damage Reef structure, and major outbreaks of coral-eating Crown of Thorns starfish (*Acanthaster planci*) (Waterhouse et al. 2017a). Previous research has linked nutrient (particularly DIN) export from catchments draining to the Great Barrier Reef lagoon with reductions in coral biodiversity, heightened vulnerability of Reef corals to thermally-induced bleaching, increased presence of macroalgae, and Reef damage due to coral-eating crown of thorns starfish (see Brodie et al., 2017 p.8). Fine sediment has been found to impair growth of ecologically-important seagrass beds by inhibiting light transmission, to negatively affect the reproductive cycle of corals, and to impair larval development in some species of coral reef fish (see Brodie et al., 2017 p.14-15).

Recognising the importance of addressing these threats, the Reef 2050 Long-Term Sustainability Plan (Commonwealth of Australia 2015) sets out objectives to secure the sustainability of the Reef and thus maintain its status as an iconic and valuable global, national and regional asset. The Reef 2050 Water Quality Improvement Plan 2017-2022 (Reef 2050 WQIP) (State of Queensland 2018b) sets out the catchment-specific water quality improvements required to achieve the objectives of the Reef 2050 Long-Term Sustainability Plan (Commonwealth of Australia 2015). The Reef 2050 Long-Term Sustainability Plan and the Reef 2050 WQIP are underpinned by scientific research summarised in the 2017 Scientific Consensus Statement (Waterhouse et al. 2017a).

Intending to deliver the Reef 2050 WQIP, the federal and Queensland governments have made, and are continuing to make, very significant investments in improving Reef water quality. Economic development along the Reef coastline could potentially jeopardise these hard-won gains unless cost-effective approaches can be found for facilitating economic expansion with no net decline in Reef water quality.

Project 3.1.6 aimed to investigate how trading in water quality credits could help facilitate *future* economic expansion in Great Barrier Reef (GBR) catchments *once the Reef 2050 water quality objectives have been met*. The project scopes opportunities for water quality credit trading and develops a framework for DIN and sediment trading along the Reef coast. The overarching research question addressed is whether trading in water quality credits can provide a cost-effective pathway for *future* economic expansion without jeopardising Reef water quality – *once the Reef 2050 basin-specific water quality objectives have already been achieved* (State of Queensland 2019a). The project also estimates how revenues from credit trading could potentially stimulate rural economies and create jobs.

This Report presents the findings of research undertaken during the project. The layout of the Report is as shown in Figure 1.1 and described in the following paragraphs.

1	Introduction	
2	Regulatory & governance framework	
3	Water quality credit trading: fundamentals	
4	Nitrogen losses from sugarcane	
5	DIN credits from practice change in sugarcane	9 Sediment credits from gully remediation
6	DIN credits from land use change	10 Demand for sediment credits
7	Demand for DIN credits	11 DIN credits from sediment reduction
8	Trading ratios for DIN credits	
12	Market configurations for multiple types of credit	
13	Economic impacts from credit generation	
14	Achieving the potential: realising opportunities	
Addendum	Q-Method study: Revealing innovation discourses in the sugarcane sector	

Figure 1.1: Topic coverage in the Report; showing chapter numbers and abbreviated chapter titles.

Chapter 2 outlines the regulatory and governance framework surrounding water quality of the GBR. Chapter 3 describes the water quality targets for catchments draining to the GBR and sets out the fundamentals of water quality credit trading. In Chapter 4, the focus turns to nitrogen losses from sugarcane production because the reductions in nitrogen losses that follow from improvements in fertiliser management practice in cane production are potentially a major source of dissolved inorganic nitrogen (DIN) credits. Chapter 4 briefly describes the cane production cycle, the physical and chemical processes through which DIN losses arise, and the Paddock to Reef (P2R) modelling, monitoring and reporting framework that is used to account for pollutant loads entering the Reef lagoon. Chapter 4 also presents results from agricultural production simulation modelling in P2R (APSIM P2R) for predicted annual average, and year-specific, DIN losses and cane yields at fine spatial resolution. Drawing on the background and APSIM P2R results from Chapter 4, Chapter 5 describes the methodologies and data sources used to estimate the levels of compensation that cane farmers would potentially require to adjust their fertiliser management practices to reduce DIN losses to the Reef – and, in so doing, generate DIN credits for a DIN credit trading market. These compensation estimates, in combination with estimates of the resulting reductions in DIN load at End-of-Catchment (EoC), are used to produce DIN credit supply curves for the Mulgrave-Russell, Johnstone, Tully, Murray and Herbert catchments in the Wet Tropics. The supply curves order DIN reductions from successive steps of fertiliser practice change on individual cane management units from the most cost-effective (i.e. lowest cost) to least cost-effective (highest cost) for the catchment concerned. This allows the total quantity of DIN credits available within a given price limit to be calculated for each catchment. Driving the supply curve in the opposite ‘direction’, the maximum credit price required to achieve a desired total quantity (kg) of EoC DIN reduction can be calculated.

Chapter 6 estimates the costs of supplying DIN credits via land use change (e.g. setting aside areas of less profitable cane land) and from constructed nitrogen treatment systems (e.g. landscape wetlands, embellished wetlands and field-scale bioreactors). This chapter uses the latest findings from nitrogen treatment systems installed using funding from the Wet Tropics Major Integrated Program (MIP). Chapter 7 switches emphasis to consider the demand side of a water quality trading market in DIN. Data from the Queensland Government’s Environmental Authorities Register is used to catalogue the total licenced nitrogen load emitted by sewage treatment plants (STPs) and aquaculture facilities in Wet Tropics catchments. Additional data on the nitrogen concentrations and loads discharged by STPs in GBR catchments, together with projections for population growth in the region’s towns and cities, is used to estimate the potential scale of demand for nitrogen credits from STPs that are looking to offset emissions in excess of a de facto 5mgN/L concentration standard, or from STPs that might seek to buy nitrogen credits to defer the costs of augmenting treatment capacity as population increases. Chapter 8 addresses the important issue of the ‘trading ratio’ that is applied in state policy to safeguard environmental outcomes when diffuse source credits are used to offset emission exceedances on point source licences. The predicted year-to-year variation in the EoC DIN reduction from different levels of practice change in cane is used to exemplify the challenge inherent in setting a trading ratio that is environmentally robust whilst also still supportive of point source to diffuse source credit trading.

In Chapter 9 the focus moves to sediment credits. Chapter 9 describes the background to sediment erosion from gullies, and the potential of gully remediation as a mechanism for supplying sediment credits to a water quality trading market. This chapter uses the latest

predictions from research undertaken for the Landholders Driving Change MIP that estimate the cost-effectiveness of supplying sediment credits from gully remediation in the Bowen – Bogie subcatchments of the Burdekin Region. Chapter 10 considers potential demand for sediment credits in water quality trading markets along the Reef coastline, combining potential demand from multiple sources under different policies. Chapter 11 addresses the possibility of generating nitrogen credits from interventions that reduce fine sediment loads to GBR catchments. This chapter draws on the latest research from the Department of Environment and Science on quantifying the bioactive nitrogen content of eroded fine sediment, and shows how these innovations can be incorporated alongside P2R-derived data to quantify the potential of sediment reductions as a source of DIN credits. Continuing on a similar theme, Chapter 12 considers market configurations for trading credits in multiple pollutants, using co-production of DIN, sediment and carbon credits as an example. Chapter 13 uses results from input-output analysis to estimate the potential economic impact of investments in credit-generation in GBR catchments on gross regional product and employment. Chapter 14 summarises key findings and suggests a market configuration and supporting technological innovations to facilitate water quality trading as a mechanism for enabling economic expansion along the Reef coastline without adverse impacts on Reef water quality.

An separate technical report presents findings from a Q-method study to identify key discourses around water quality credit trading amongst stakeholders in the sugarcane industry in the Wet Tropics.

2.0 REGULATORY AND GOVERNANCE FRAMEWORK OF THE GREAT BARRIER REEF

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2. Summary

Concern at state, national and international levels regarding the ecological condition of the Great Barrier Reef has led to a suite of policies that seek to improve Reef health. The primary focus of these policies is to reduce the threat posed by high loads of DIN and sediment discharged from catchments along the Reef coastline. End-of-Basin targets for anthropogenic DIN and fine sediment loads have been set for 35 catchments in the 6 Reef regions along the Queensland coastline (State of Queensland 2019a). Reef Regulations have been amended to mandate adoption of defined best management practices by the different agricultural sectors (State of Queensland 2019d, a, c). Policies have been developed to support purchase of DIN and sediment credits through different forms of water quality offsetting mechanisms as a potentially cost-effective approach for achieving and then retaining water quality objectives (Reef Trust 2017; State of Queensland 2018a, 2019f). This provides the legal and governance background within which water quality credit trading mechanisms can be implemented.

2.1 Introduction

The Great Barrier Reef (GBR) is the largest coral reef system on Earth and at 348 000 square kilometres in area, one of the world's richest and most diverse natural ecosystems. The unique qualities of this property were recognised in 1981 when it was inscribed on the World Heritage List. The GBR was one of 15 Australian World Heritage locations included in the National Heritage List on 21 May 2007.

Successive reports since the late 1990s have identified that the Great Barrier Reef is showing declining trends in condition. The recent *2017 Scientific Consensus Statement: Land use impacts on Great Barrier Reef water quality and ecosystem condition* identifies that poor water quality remains one of the leading causes of the poor state of the GBR's coastal and reef ecosystems (Waterhouse et al. 2017a). The 2017 Scientific Consensus Statement identifies discharges of nutrients, fine sediments, and pesticides as the greatest water quality-related risks to the Reef.

The Reef receives run-off from 35 major catchments within six Reef regions that drain 424,000 square kilometres of coastal Queensland. The main source of excess nutrients and fine sediments from GBR catchments is diffuse source pollution from agriculture (Waterhouse et al. 2017a). Other uses, including urban areas contribute relatively small and concentrated

pollutant loads, but these can be important at local scales. The loads from different catchments are quite varied but result in a significant cumulative impact on the Reef, which is also facing other stressors such as climate change and impacts from direct uses like shipping and fishing. The Great Barrier Reef Water Science Taskforce recommended that water quality trading approaches may be viable in some settings in the future, but will require a staged pathway of regulation and detailed farm-level information to support implementation (Great Barrier Reef Water Science Taskforce and Queensland Government 2016).

This chapter outlines the current regulatory and governance mechanisms which would be relevant to the establishment of a terrestrial water quality credit trading scheme for nutrient and sediment pollution in GBR catchments.

2.2 The players

The varying jurisdictional responsibilities of International, Australian, Queensland and local governments, layered with the sheer scale and diversity of landscapes comprising the Great Barrier Reef World Heritage Area (GBRWHA) can make the system of legal protection, management planning and institutional arrangements appear complex.

The *Commonwealth of Australia Constitution Act* (the Constitution) came into effect in 1901, establishing a federal Parliament and uniting into a federation what had until then been separate states. Under the Constitution, regulation of natural resource management and environmental protection on land are primarily the responsibility of State Governments.

Environment is not one of the heads of power granted to the Australian Government under section 51 of the Constitution, however since around 1970 the Australian Government has increased its role in environmental legislation and has amended s.51 to provide a head of power in relation to:

- external affairs⁸
- international and interstate trade and commerce
- fisheries in Australian waters beyond territorial limits.

Jurisdictional responsibility for the management, use, access and protection of the GBRWHA is therefore shared between the Australian and Queensland governments. There are 26 different Australian and Queensland government Acts and regulations directly relevant to the management (use and protection) of the GBRWHA (Tingay et al. 2014) and without a radical overhaul it would be impossible for a single entity to have full and complete jurisdictional authority over all threats to, impacts on, uses of, and research into the GBRWHA.

2.3 Intergovernmental agreement⁹

In order to deal with the existence of multiple legislative instruments and multiple levels of government, a cooperative approach to managing the GBRWHA, was formalised by the Emerald Agreement, between the Australian and Queensland governments, in 1979. This

⁸ Including World Heritage Listings

⁹ <http://www.environment.gov.au/marine/gbr/protecting-the-reef/intergovernmental-agreement>

agreement was updated in July 2009 as the Great Barrier Reef Intergovernmental Agreement, which provided a more contemporary framework for cooperation as well as recognising challenges such as climate change and catchment water quality that had not been foreseen in the 1979 Agreement.

The GBR Intergovernmental Agreement 2015 (an updated version of the 2009 Agreement) provides a framework for the Australian and Queensland governments to work together to protect the Great Barrier Reef and reflects the shared vision for the future outlined in the Reef 2050 Plan (Commonwealth of Australia 2015). The Great Barrier Reef Ministerial Forum oversees the implementation of the Intergovernmental Agreement. The Ministerial Forum is comprised of two ministers each from the Australian and Queensland governments with responsibility for matters relating to the environment and marine parks, science, tourism and/or natural resource management. Ministers responsible for mining may not be members.

2.4 Commonwealth Government

The *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act)¹⁰ provides a legal framework to protect and manage nationally and internationally important flora, fauna, ecological communities, including the GBRWHA, and heritage places defined in the Act as matters of national environmental significance (MNES). The EPBC Act is administered by the Federal Department of Agriculture, Water and the Environment (DAWE) (formerly the Department of Environment and Energy (DoEE)).

The Great Barrier Reef Marine Park Authority (GBRMPA) are a portfolio agency of the DAWE and have responsibility for the management of the Great Barrier Reef Marine Park under the *Great Barrier Reef Marine Park Act 1975* (GBRMP Act). The following activities within the Great Barrier Reef Marine Park and Queensland's Great Barrier Reef Coast Marine Park require a GBRMPA permit:

- aquaculture facilities
- dredging and dumping of dredge material
- waste discharge from a fixed structure

The EPBC Act allows for the use of an offset for any remaining impacts on biodiversity-related MNES after all on-site avoidance and mitigation has been implemented. The offset policy requires that the proponent of a defined activity within the Great Barrier Reef Marine Park, Queensland's Great Barrier Reef Coast Marine Park makes sufficient payment into an offset fund to enable any residual impacts to be mitigated through relevant on-ground actions. The relevant on-ground actions are financed from the offset fund which is administered by the Reef Trust (see below). An offsets methodology has been developed for the Reef Trust through research undertaken via the National Environmental Science Programme Tropical Water Quality Hub (NESP TWQ Hub) Project 3.12 (Maron et al. 2016).

¹⁰ <https://www.environment.gov.au/epbc>

Under the Reef Trust offsets methodology, reduction of adverse impacts on water quality by offsetting nitrogen and sediment pollution can be regarded as surrogates¹¹ for biodiversity-related MNES when calculating a monetised offset. The offset payments required per unit residual impact (i.e. per tonne of residual sediment impact or per kilogram of residual nitrogen impact) are specified in the Reef Trust Offsets Plan and Calculator (Reef Trust 2017). Payment rates for dissolved inorganic nitrogen (DIN) and fine sediment offsets are based on the actual costs on-ground restoration actions in Reef catchments (Rolfe and Windle 2016), with adjustments for estimated success rate, uncertainty regarding restoration cost, the suitability of the restoration site as a surrogate for the impacted site (for which the offsets are sought), and the time taken for the offset to become fully effective (at a 5% per annum annual discount rate). It is important to note that the Reef Trust Offsets implementation guidance document (Commonwealth of Australia 2017a) states “A proponent’s liability in relation to Reef Trust-delivered offsets will be discharged on payment of offset funds into the Reef Trust Special Account.” (Commonwealth of Australia, 2017, p.13).

As an example of the Reef Trust offsets policy in action, conditions attached to the Commonwealth’s approval of the Rookwood Weir¹² Environmental Impact Statement included a requirement for water quality offsets using the Reef Trust Offsets Plan and Calculator methodology, if monitoring determines that impacts to MNES are occurring as a result of the development.

The Reef Trust¹³ is one of the key mechanisms focusing on known critical areas for investment - improving water quality and coastal habitat along the Great Barrier Reef, controlling outbreaks of crown-of-thorns starfish and protecting threatened and migratory species, particularly dugong and turtles. The Reef Trust has a strong focus on evaluation and adaptive management, to ensure it effectively contributes to the long-term sustainable management of the Great Barrier Reef.

The Reef Trust is designed to allow for the consolidation of investment from a wide range of sources to deliver the most beneficial outcome for the Reef from every dollar spent. It seeks to complement existing Reef investment by providing prospective investors with new opportunities to support the delivery of conservation and protection projects that align with the desired outcomes of the Reef Trust.

The Reef Trust is being delivered by the Australian Government, in collaboration with the Queensland Government, and the Great Barrier Reef Marine Park Authority. Reef Trust governance and administrative arrangements are in place to ensure all funds are effectively managed and directed towards priority on-ground actions.

¹¹ A biodiversity surrogate is a relatively easily-measured metric that acts as a proxy for other components of biodiversity that are harder to measure.

¹² Rookwood Weir is an investment project on the lower Fitzroy River to provide new water storage infrastructure to meet future water demands and improve water security for residential, industrial and agricultural users in Rockhampton, Gladstone and the Capricorn Coast. <https://infrastructurepipeline.org/project/lower-fitzroy-infrastructure-project---rookwood-weir/>

¹³ <https://www.environment.gov.au/marine/gbr/reef-trust>

2.5 Queensland Government

State governments manage laws related to any matter not listed in Section 51 of the Constitution. The Queensland Government has responsibility for regulating trade and commerce and therefore creates and enforces laws relating to urban, residential and industrial activities, transport, infrastructure, and the use of state land for agriculture, mining and other purposes. The Queensland *Marine Parks Act 2004* establishes Queensland as responsible for the management of the Great Barrier Reef Coast Marine Park, which covers the portion of the GBR that is within the boundaries of the State of Queensland.

The *Environmental Protection Act 1994* (EP Act¹⁴) is the primary legislation dealing with regulation of environmental harm in Queensland and is administered by Queensland Government's Department of Environment and Science (DES). The Office of the Great Barrier Reef (OGBR) has been established within DES to ensure a coordinated response to GBR-related matters across the Queensland Government. Specific GBR protection measures exist under Chapter 4A of the EP Act. The objective of these measures is to support the outstanding universal value of the GBR, protect and enhance the integrity and biodiversity of the aquatic ecosystems of the GBR, and improve the GBR's health and resilience. GBR protection measures under Chapter 4A are consistent with The Reef 2050 Water Quality Improvement Plan 2017-2022 (Reef 2050 WQIP) (State of Queensland 2018b). The Reef 2050 WQIP sets out the water quality improvements required to achieve the objectives of the Reef 2050 Long-Term Sustainability Plan (Commonwealth of Australia 2015). The Reef 2050 Long-Term Sustainability Plan and the Reef 2050 WQIP are underpinned by scientific research summarised in the 2017 Scientific Consensus Statement (Waterhouse et al. 2017a).

To support delivery of the Reef 2050 WQIP, Chapter 4A of the EP Act was revised in 2019, together with subordinate legislation in the form of The Environmental Protection (Water & Wetlands) Policy 2019 (EPP Water and Wetland Biodiversity¹⁵), and the Environmental Protection Regulation 2019 (EP Regulation¹⁶). End-of-Basin Load Water Quality Objectives (End-of-Basin WQOs) were also set in 2019 for river basins draining to the GBR (State of Queensland 2019b). The End-of-Basin WQOs operate under EPP (Water and Wetland Biodiversity) 2019 and set limits for End-of-Basin annual loads of anthropogenic dissolved inorganic nitrogen and fine sediments for river basins draining to the GBR (Table 2.1). End-of-Basin WQOs are set to achieve desired load reductions from 2013 anthropogenic load baselines by 2025 in order to meet the water quality improvements specified in the Reef 2050 WQIP. End-of-Basin WQOs are represented as annual anthropogenic pollutant load limits (e.g. no more than 232,000 tonnes of anthropogenic fine sediment leaving the Herbert River basin by 2025) to help guide licence conditions for environmentally relevant activities (ERAs) that release nutrients, sediment or other pollutants of concern to land or water within catchments that drain to the GBR. ERAs can release pollutant emissions to land and water from point-sources, such as sewage treatment plants and aquaculture facilities, or diffuse sources, such as agriculture operations and urban stormwater runoff.

¹⁴ <https://environment.des.qld.gov.au/management/policy-regulation>

¹⁵ <https://www.legislation.qld.gov.au/view/html/inforce/current/sl-2019-0156>

¹⁶ <https://www.legislation.qld.gov.au/view/whole/html/asmade/sl-2019-0155>

Collectively, Chapter 4A of the EP Act, EPP Water and Wetland Biodiversity, EP Regulation, and the End-of-Basin WQOs support revised Reef Protection Regulations¹⁷ that were also introduced in 2019. Protection measures under Chapter 4A of the EP Act form the main tool for regulating pollution from agricultural ERAs in GBR catchments. Under Chapter 4A of the 2019 EP Act an agricultural ERA is any cattle grazing, horticulture (including banana production) or cultivation of other crops (including sugarcane and grains) conducted commercially on land in a GBR catchment. The Reef Protection Regulations seek to assist in delivering the End-of-Basin WQOs, and thus the Reef 2050 WQIP, for nutrient and sediments for all 35 river basins that drain into the GBR Lagoon.

¹⁷ <https://www.qld.gov.au/environment/agriculture/sustainable-farming/reef/reef-regulations>

Table 2.1: End-of-basin anthropogenic load WQOs for dissolved inorganic nitrogen and fine sediments in the Great Barrier Reef catchment to be achieved by 2025 (State of Queensland, 2019a; Table 1, p.6)

<i>RM Region</i>	<i>River Basin Name</i>	<i>Anthropogenic Dissolved Inorganic Nitrogen (DIN) (tonnes per year)</i>	Anthropogenic Fine Sediment (tonnes per year)
Cape York	Jacky Jacky	0	43,000
	Olive Pascoe	1	54,000
	Lockhart	0	53,000
	Stewart	0	39,000
	Normanby	9	136,000
	Jeannie	0	29,000
	Endeavour	1	24,000
Wet Tropics	Daintree	135	28,000
	Mossman	52	6,000
	Barron	35	32,000
	Mulgrave-Russell	123	140,000
	Johnstone	149	160,000
	Tully	194	66,000
	Murray	112	31,000
	Herbert	266	232,000
Burdekin	Black	21	34,000
	Ross	49	49,000
	Haughton	274	157,000
	Burdekin	71	1,946,000
	Don	68	128,000
Mackay Whitsunday	Proserpine	47	75,000
	O'Connell	56	145,000
	Pioneer	53	138,000
	Plane	106	99,000
Fitzroy	Styx	10	94,000
	Shoalwater	5	59,000
	Water Park	4	57,000
	Fitzroy	159	902,000
	Calliope	6	35,000
	Boyne	3	10,000
Burnett Mary	Baffle	16	42,000
	Kolan	34	24,000
	Burnett	57	341,000
	Burrum	93	14,000
	Mary	181	536,000

The Reef Protection Regulations adopt a mandatory best practice approach, with best practice defined separately for different agricultural commodities¹⁸ (sugarcane, bananas, beef grazing, grains etc.). Mandatory best practice for the grazing industry focuses on reducing sediment erosion. Mandatory best practices for sugarcane and bananas address nutrient, sediment, herbicide and pesticide losses. Key elements of mandatory best practice under the 2019 Reef Regulations are:

For sugarcane (State of Queensland 2019c, a)

- Development of a whole-of-farm budget for nitrogen and phosphorus loads applied annually (Figure 2.1), using fertiliser application rates determined from District Yield Potential (DYP) and block-specific soil testing, with appropriate adjustments for crop class (plant cane, first ratoon, second ratoon etc.), legume fallow, application of nutrients from mill by-products, and historical block- and management zone-specific cane yields (State of Queensland 2019a; SC10-19 p.6-8, 2019e, 2019c; Part B). This is designated as Minimum Standard ('moderate risk') nutrient management practice in the Sugarcane Water Quality Risk Framework (2017-2022) (The Australian and Queensland Governments, n.d.)
- Record keeping regarding soil testing (for organic carbon and extractable phosphorus content (State of Queensland, 2019b Part A)), and applications of fertilisers, soil conditioners, pesticides and fungicides
- Adhere to best practice standards to minimise sediment runoff (e.g. maintain adequate surface cover on paddocks after cane harvest, construct drainage structures such as spoon drains to reduce water runoff velocity etc.) and to avoid ground-based broadcast surface application of fertilisers

For grazing (State of Queensland 2019e)

- Adhere to best management practice by maintaining appropriate record keeping
- If there is less than 50% groundcover across an area at end of September, implement an appropriate measure to improve groundcover and land condition to reduce the risk of sediment erosion

The whole-of-farm nitrogen budget allows for nitrogen applications to be re-allocated between blocks within the same farm, provided that the overall load of nitrogen applied stays within the budget. This feature could assist in encouraging farmers to investigate supplying nitrogen credits to a water quality credit market by reducing nitrogen application rate on some blocks within the farm. Within the whole-of-farm budget, nitrogen application rates could then be increased on other blocks and the (potential) income from supply of water quality credits could be viewed as a form of insurance against reduced yield on blocks with the reduced fertiliser rate. Readily available advice on predicted yields and DIN reductions for individual blocks under reduced nitrogen applications would assist the farmer to consider these types of flexible nitrogen management options.

¹⁸ <https://www.qld.gov.au/environment/agriculture/sustainable-farming/reef/reef-regulations/resources>

Zones	Block/s	OC%	BSES P	PBI	N	P	Area (ha)	N x area	P x area	Notes
1	11, 12	1.4	80	150	130	0	7.5	975		No P for crop cycle – mill mud applied at 150 t/ha in 2017
2	13, 17	1.5	45	155	110	40	8	880	320	Crop cycle P requirement applied at plant (2018)
3	15, 16	1.1	55	150	140	0	4	560		No P for crop cycle – mill mud applied at 150 t/ha in 2015
Fallow	14	-	-	-	0	0	5	0	0	Grassy fallow
Whole of Farm amount							19.5 ha	2415	No whole of farm P amount	

Farm map for year 1 (2018)

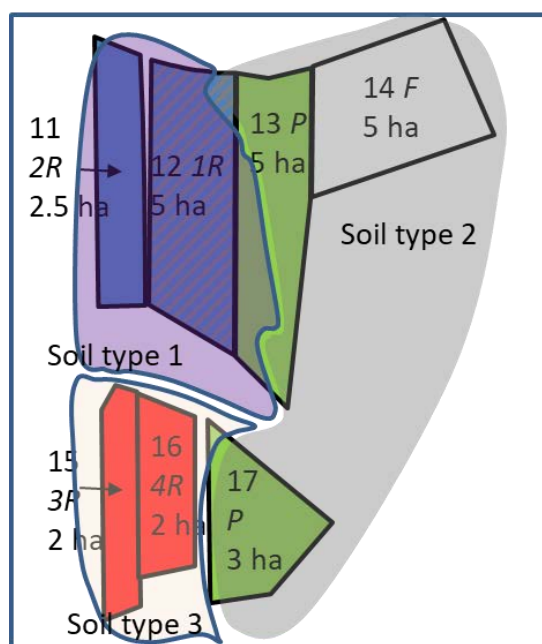


Figure 2.1: Example whole-of-farm nitrogen budget specifying nitrogen applications for land blocks on the farm, allowing for district yield potential for sugarcane, soil type, soil organic carbon content (derived from soil testing), application of soil conditioners such as mill mud, nitrogen inputs from legume fallow, and historical block- and management zone-specific cane yields. Whole-of-farm budget calculated using the methodology described in the agricultural ERA standard for sugarcane cultivation in GBR catchments (State of Queensland 2019a; SC10-19, p.6-8). Diagrams copied from (State of Queensland, 2019e p.18).

Emissions from point source ERAs also contribute to End-of-Basin nutrient and/or sediment loads. To protect environmental quality, the State government requires that point source ERAs operate under an Environmental Authority (EA) that stipulates conditions and limits for discharges to the environment¹⁹. Two categories of point source ERAs are defined: *prescribed ERAs* and *resource activities*. Prescribed ERAs are industrial or intensive agricultural activities (as defined under Schedule 2 in EP Regulation 2019) that carry associated environmental risks e.g. sewage treatment plants, aquaculture operations, intensive poultry units. Resource activities are primarily mining, coal seam gas and petroleum-related activities as defined under relevant resource-related Acts in State law. Prescribed ERAs and resource activities are relevant to Project 3.1.6 because their EAs set conditions and limits on acceptable discharges of nutrients and/or sediment from these point sources to river catchments and receiving water bodies.

The Queensland government's Point Source Water Quality Offsets Policy 2019 (PS WQ Offsets Policy) (State of Queensland 2019f) describes how existing or potential new EA holders can use water quality offsets as a voluntary mechanism for managing their ERAs within the terms of their EA. The PS WQ Offsets Policy operates in accordance with the management hierarchy specified under EPP Water and Wetland Biodiversity 2019: avoid, mitigate, offset such that no environmental harm is caused to the environmental values of receiving waters.

Under the PS WQ Offsets Policy a point source entity²⁰ may offset exceedance of the annual discharge mass load limit stipulated in its EA for particular contaminants (e.g. total nitrogen (TN), dissolved inorganic nitrogen (DIN), or total suspended solids (TSS)) by obtaining sufficient water quality offsets to cover those exceedances. A point source entity can obtain water quality offsets for relevant contaminants by (State of Queensland, 2019e p.5-6):

- Undertaking actions that reduce relevant contaminants from rural, urban or other diffuse sources²¹. Actions that may provide water quality offsets from diffuse sources include:
 - Restoration of riparian areas
 - Restoration of eroding streambanks and gullies
 - Construction or remediation of wetlands
 - Riparian fencing for stock exclusion
 - Bioremediation technology
 - Reducing on-farm nutrient runoff through improved management of fertiliser applications above required minimum standards
 - Reducing sediment runoff through improving grazing management practices above required minimum standards

¹⁹ <https://www.business.qld.gov.au/running-business/environment/licences-permits/applying/activities>

²⁰ The Point Source Water Quality Offsets Policy 2019 defines a point source entity as the holder of an EA that allows the discharge or release of wastewater into waterways at the release point stated in the EA (State of Queensland, 2019e p.16).

²¹ The Point Source Water Quality Offsets Policy 2019 defines a diffuse source as [a source of] non-point source pollutants (i.e. without a single point of origin or not introduced into a receiving stream from a specific outlet). Common diffuse sources are agriculture, forestry, urban areas and historical mining sites (State of Queensland, 2019e p.15).

- Purchasing water quality credits through a recognised market-based mechanism. The water quality credits may have been obtained from diffuse sources via improvements in agricultural land management practice and changes in agricultural land use as listed immediately above, or through verified reductions in contaminant emissions from other point sources. The market-based mechanism is intended to incentivise cost-effective supply water quality credits from diffuse and point sources by allowing point source entities to purchase these credits to offset emission exceedances under their EAs where they would find it unreasonably expensive to avoid or mitigate those exceedances on-site.
- Where an EA-holder holds multiple EAs in reasonable geographical proximity, the EA-holder can apply to combine those EAs into a single amalgamated EA. Under appropriate environmental conditions, an approved amalgamated EA can provide an opportunity for the EA-holder to exceed prior location-specific discharge limits at one of their operating locations provided that discharges from their other locations are reduced sufficiently to satisfy the discharge limits in the amalgamated EA.

The intention of the PS WQ Offsets Policy is to allow ERAs to employ greater flexibility in meeting the regulatory requirements inherent in their EA, whilst delivering improvements in water quality in river catchments and receiving water bodies. To ensure that water quality is not impaired, water quality offsets and water quality credits purchased as offsets in a water quality credit market must therefore deliver adequate reductions in relevant contaminants in the receiving environment to which the purchasing point source's EA conditions apply. The PW WQ Offsets Policy 2019 includes two mechanisms that aim to account for uncertainty regarding the offsetting capability of diffuse source offsets in the specified receiving environment. These are a *delivery ratio* and an *environmental equivalency ratio* which operate as illustrated in Figure 2.2²².

The delivery ratio (Figure 2.2) aims to account for uncertainty regarding the ability of a water quality offset to deliver an equivalent reduction in the mass load of the relevant pollutant in the receiving environment specified in the purchasing point source's EA. The delivery ratio thus intends to account for

- Uncertainty in offsetting capacity due to the distance between the offset location and the receiving environment to which the EA conditions apply
- Flow, flow duration and the attenuation of pollutants as they are carried down a catchment (i.e. the rate at which pollutant loads are reduced through natural processes as they move down a catchment under different flow conditions). Emissions from the purchasing point source and the offset location may be attenuated to different extents before reaching the receiving environment.
- Whether the offset location is located upstream or downstream, or in a connected catchment, relative to the receiving environment to which the EA conditions apply
- Uncertainty regarding the success (or otherwise) of offset implementation at the offset location

²² The delivery ratio and environmental equivalency ratio perform similar roles to the success rate multiplier and surrogate condition multiplier under the Reef Trust Offset Plan and Calculator. The Point Source Water Quality Offset Policy does not, however, include a time delay factor like that used in the Reef Trust Offset Plan.

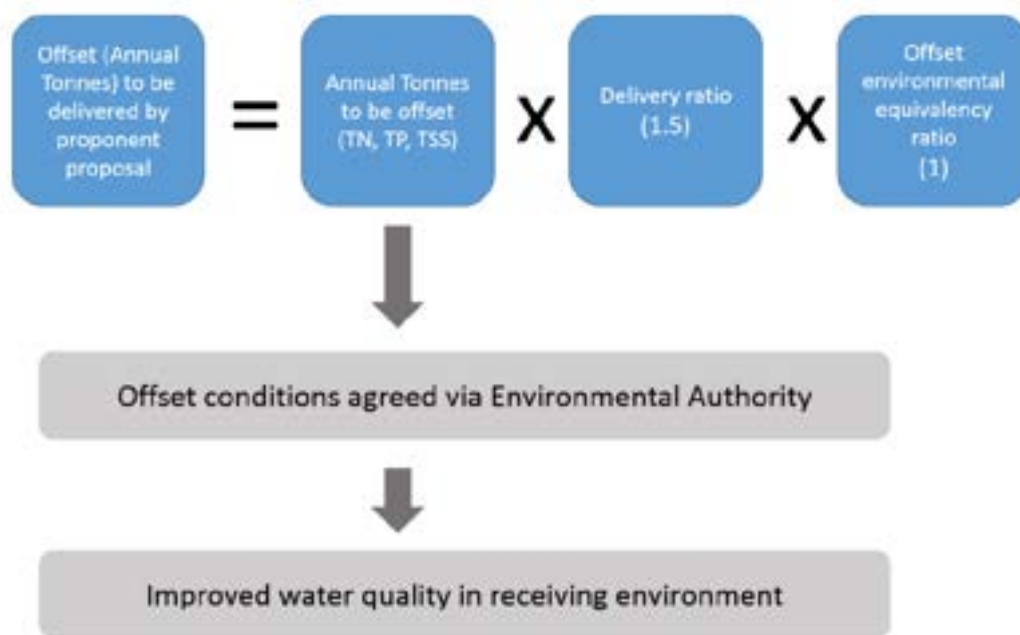


Figure 2.2: Conceptual diagram illustrating how delivery ratio and offset environmental equivalency ratio address uncertainty regarding the offsetting capacity of water quality offsets in the receiving environment to which the purchasing point source's EA conditions apply. Copied from Point Source Water Quality Offsets Policy 2019, Figure 1, p.3 (State of Queensland 2019f).

To account for these uncertainties, the delivery ratio will be greater than one. The PS WQ Offsets Policy 2019 currently stipulates that a delivery ratio of 1.5 should be applied to both point source and diffuse source offsets. Using DIN load as an example, a delivery ratio of 1.5 requires that 1.5 kg of DIN offset would have to be purchased to constitute an adequate offset for 1 kg of DIN exceedance on a point source's EA.

The offset environmental equivalency ratio (Figure 2.2) aims to account for uncertainty associated with the chemical form or species of pollutant reduction provided by a pollution offset relative to the chemical form or species of the point source's emission exceedance that the offset is intended to negate. The PS WQ Offsets Policy 2019 currently stipulates that an environmental equivalency ratio of 1 should be applied to both point source and diffuse source offsets.

The PS WQ Offsets Policy specifies that the Administering Authority can adjust the delivery ratio and the environmental equivalency ratio as additional information becomes available. Notwithstanding inclusion of a delivery ratio and an environmental equivalency ratio, PS WQ Offsets Policy 2019 stipulates that, under the conditions of their EA, the proponent (i.e. the purchaser of the offset) remains accountable for impacts on the receiving environment. The proponent also remains accountable if the offset was acquired through a market-based mechanism. This contrasts starkly with a proponent's liability once fee in-lieu payments have been made to the Reef Trust Offsets plan.

The *Planning Act 2016* (The Planning Act²³) and its associated Planning Framework came into effect on 3 July 2017. These contain the principal legislation for Queensland's planning and development system for urban land uses. The Planning Act is administered by the Department for State Development, Manufacturing, Infrastructure and Planning. The Planning Act and the State Planning Policy set the requirements for achieving best management practice for erosion and sediment control and stormwater management from urban development. State Planning Policy allows for the management of unmitigated post-construction pollutant emissions from urban stormwater by off-site implementations of appropriate catchment-based green infrastructure solutions (e.g. riverbank rehabilitation, gully remediation, wetland construction). The 2018 Draft Implementation Guidance (State of Queensland 2018a) states that post-construction emissions from an urban development site can be managed by developers making appropriate in-lieu payments to local government. Local government then have the responsibility to deliver off-site treatment solutions that provide water quality outcomes equivalent to those that would have been achieved by on-site treatment.

An approval is required under the Planning Act wherever activities materially change the use of land, realign a property boundary or undertake significant operational works. Some activities that impact on water quality may be approved independent of the Planning Framework under the *Water Act 2000*, the *Vegetation Management Act 1999*, the *Coastal Protection and Management Act 1995* and the *State Development and Public Works Organisation Act 1971*.

2.6 Natural Resource Management Groups

Natural Resource Management Groups (NRM groups) are community-based not for profit organisations largely funded by the Australian and Queensland governments. NRM groups build local and regional partnerships to secure the health of natural resources (including water, soil and biodiversity) and provide planning direction, information, advice and practical support. The six NRM regions that cover the GBR are:

- Burnett Mary
- Cape York
- Fitzroy Basin
- Mackay Whitsunday
- Burdekin
- Wet Tropics

2.7 The United Nations Educational, Scientific and Cultural Organisation (UNESCO)

The UNESCO seeks to encourage the identification, protection and preservation of cultural and natural heritage around the world that is considered to be of outstanding value to humanity. This is embodied in an international treaty called the Convention Concerning the Protection of the World Cultural and Natural Heritage (the Convention²⁴), adopted by UNESCO in 1972.

²³ <https://www.legislation.qld.gov.au/view/pdf/inforce/current/act-2016-025>

²⁴ <https://whc.unesco.org/en/conventiontext/>

The World Heritage Committee has been established with responsibility for the implementation of the Convention and defines the use of the World Heritage Fund and allocates financial assistance upon request. The Committee determines whether a property is inscribed on the World Heritage List, assesses the state of conservation of inscribed properties and requests action when properties are not being adequately managed. It also decides on the inscription or deletion of properties on the List of World Heritage in Danger.

At its 36th meeting in June 2012, the World Heritage Committee considered the state of conservation of the GBR World Heritage Area (GBRWHA) and noted a report from a reactive monitoring mission that visited Australia earlier that year. Subsequently, in February 2013, January 2014 and January 2015 the Australian Government provided reports on the state of conservation of the GBR, and also on the implementation of the Committee's decisions. The Committee considered this material and in July 2015 determined that the GBR did not face ascertained or potential danger to its outstanding universal value (OUV).

In July 2017, the World Heritage Committee recognised Australia's significant effort in the implementation of the Reef 2050 Long-term Sustainability Plan (Commonwealth of Australia 2015) and the development of the Investment Framework for the Reef (Commonwealth of Australia 2016). Given the satisfactory inception of the Reef 2050 Plan, the World Heritage Committee will next consider the overall state of the Reef in 2020

2.8 Summary

Concern at state, national and international levels regarding the ecological condition of the Great Barrier Reef has led to a suite of policies at state and national levels that seek to improve Reef health. The primary focus of these policies is to reduce the threat posed by high loads of DIN and sediment discharged from catchments along the Reef coastline. End-of-Basin targets for anthropogenic DIN and fine sediment loads have been set for 35 catchments in the 6 Reef regions along the Queensland coastline (State of Queensland 2019b). Reef Regulations have been amended to mandate adoption of defined best management practices by the different agricultural sectors (State of Queensland 2019d, a, c). Policies have been developed to support purchase of DIN and sediment credits through different forms of water quality offsetting mechanisms as a potentially cost-effective approach for achieving and then retaining water quality objectives (Reef Trust 2017; State of Queensland 2018a, 2019f). This provides the legal and governance background within which water quality credit trading mechanisms can be implemented.

3.0 FUNDAMENTALS OF WATER QUALITY CREDIT TRADING

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3. Summary

This Chapter describes the fundamentals of water quality credit trading. The target reductions in End-of-Catchment anthropogenic DIN and sediment loads required to achieve the 2025 Water Quality Objectives (State of Queensland 2019a) are reported for GBR catchments. The commodities that would be traded in water quality credit trading markets for nutrient and sediment are defined, and the process through which supply and demand curves are constructed for the relevant credits is described. The role of the environmental regulator, a key actor in water quality credit trading, is defined with regard to setting the discharge licence conditions specified in Environmental Authorities (EAs) for Environmentally Relevant Activities (ERAs), and setting the delivery ratio and environmental equivalency ratio specified in Queensland's Point Source Water Quality Offsets Policy.

Point source emissions offsetting via diffuse source credits is explained, for situations in which the point source discharges to End-of-Catchment or to a receiving water further upstream. Basic configurations for water quality trading are described and the roles of a credit bank, credit aggregator and clearing house in these configurations are explained. The material covered in this Chapter sets the background for quantification of supply and demand for DIN and sediment credits in subsequent chapters and for more detailed consideration of complexities such as trading in stacked credits in multiple pollutants later in the Report.

Project 3.1.6 considers whether trading in water quality credits could provide a cost-effective pathway to facilitate *future* economic expansion²⁵ in Great Barrier Reef catchments without jeopardising Reef water quality – once the Reef 2050 water quality objectives have been met. Specifically, Project 3.1.6 explores trading in water quality credits primarily as a potential approach for *retaining* the 2025 End-of-Catchment anthropogenic Water Quality Objectives at least cost to society in the face of economic expansion in the region. This Chapter provides background on conventional water quality credit markets that trade credits in a single type of pollutant (i.e. dissolved inorganic nitrogen (DIN) or fine sediment) between regulated point source emitters and agricultural diffuse source credit suppliers. This basic structure of *point*

²⁵ Economic expansion here could involve any sector of the economy, e.g. agriculture, tourism, mining, aquaculture, retail etc. New agricultural developments are subject to strict provisions under the ERA standards for agriculture in the Reef catchment (State of Queensland 2019c, d). These standards operate in tandem with legislation relating to land clearing.

source to diffuse source credit trading is the focus of this Chapter. Inclusion of *non-emitters as buyers* in the credit markets is also described. Participation of non-emitting buyers *would* enable water quality credit trading to contribute towards achieving End-of-Catchment Water Quality Objectives; this is discussed briefly in this Chapter. Descriptions of market configurations for trading in multiple pollutants as *stacked credits* are postponed until Chapter 12. The following topics are addressed in this Chapter:

- 2025 End-of-Catchment anthropogenic Water Quality Objectives for nitrogen and sediment
- nitrogen or sediment credits as a traded commodity
- demand for and supply of credits
- delivery ratio and offset environmental equivalency ratio as they relate to the role of the Environmental Regulator

3.1 Target reductions for nutrients and sediments

The Reef 2050 Water Quality Improvement Plan (2017-2022) (Reef 2050 WQIP) (State of Queensland 2018b) sets basin-specific targets for reductions in anthropogenic pollutants of concern to be achieved by 2025. These basin-specific reductions are expressed relative to modelled 2012-2013 End-of-Basin anthropogenic pollutant loads and are consistent with the End-of-Basin anthropogenic load targets for 2025 set as Water Quality Objectives (End-of-Basin WQOs) (State of Queensland 2019b) under the Queensland government's Environmental Protection (Water & Wetlands) Policy 2019 (EPP Water and Wetland Biodiversity²⁶).

The End-of-Basin reduction targets and WQOs reflect findings from a report by Brodie et al. (2017) regarding the reductions in anthropogenic catchment-sourced pollution loads required to maintain the ecological health of the GBR (Brodie et al. 2017). Drawing on multiple previous scientific studies, Brodie et al's report identifies fine sediment (< 16µm), particulate nitrogen (PN), particulate phosphorus (PP), dissolved inorganic nitrogen (DIN), and pesticides as primary pollutants of concern for the ecological health of the Reef. End-of-Basin WQOs and pollutant reduction targets are specified for DIN and fine sediment. Separate WQOs and reduction targets are not set for PN and PP as these pollutants are emitted jointly with fine sediment and it is assumed that the necessary load reductions in PN and PP will be delivered if the fine sediment WQOs are fulfilled. Brodie et al's report collates research linking nutrient (particularly DIN) export from catchments draining to the GBR Lagoon with reductions in coral biodiversity, heightened vulnerability of Reef corals to thermally-induced bleaching, increased presence of macroalgae, and Reef damage due to coral-eating crown of thorns starfish (*Acanthaster planci*) (Brodie et al., 2017 p.8). Fine sediment has been found to impair growth of ecologically-important seagrass beds by inhibiting light transmission, to negatively affect the reproductive cycle of corals, and to impair larval development in some species of coral reef fish (Brodie et al., 2017 p.14-15).

The percentage reductions in DIN and fine sediment required to achieve the End-of-Basin WQOs vary between regions along the GBR coastline, and between catchments within regions

²⁶ <https://www.legislation.qld.gov.au/view/html/inforce/current/sl-2019-0156>

(Table 3.1). The Reef 2050 WQIP regards achieving the WQOs for DIN from most catchments in the Wet Tropics as outcomes of high or very high priority (Table 3.1). DIN WQOs for most catchments in the Wet Tropics are very challenging; 70% reductions from 2012-2013 anthropogenic DIN loads by 2025 are specified for the Mulgrave-Russell, Johnstone, Tully and Herbert catchments (State of Queensland 2018b). Similarly, achieving the WQO for fine sediment from the Burdekin River basin is regarded as a very high priority and is again very challenging; a 30% load reduction (840,000 tonnes) is required from the 2012-2013 anthropogenic load level (State of Queensland 2018b). The Johnstone and Herbert catchments in the Wet Tropics are unusual in having attainment of their DIN and sediment WQOs assigned moderate, high or very high priorities. Achieving these targeted reductions in DIN and fine sediment loads by 2025 will be challenging; retaining them in the face of economic expansion thereafter is likely to be equally challenging. Project 3.1.6 considers the feasibility of water quality credit trading as an approach for addressing the second of these challenges.

Given the priorities and challenges identified in the Reef 2050 WQIP, and the body of prior research conducted under the TWQ Hub and available via published literature, Project 3.1.6 explores potential *future* water quality credit trading markets for:

- DIN in the Russell-Mulgrave, Johnstone, Tully, Murray and Herbert catchments in the Wet Tropics
- Fine sediment in the Bowen and Bogie sub-catchments within the Burdekin River catchment
- DIN and fine sediment in combination in the Johnstone catchment in the Wet Tropics and the Bowen sub-catchment in the Burdekin region

This Chapter describes key elements that form the basic structure of a market for trading water quality credits between *point source credit buyers* and *diffuse source credit suppliers*. The remainder of this Chapter proceeds as follows. Section 3.2 describes the traded commodity in a water quality credit market. Section 3.3 provides background on the demand for, and supply of, nitrogen or sediment credits. Section 3.4 discusses the role of the environmental regulator in a water quality credit market and expands on the potential application of the *delivery ratio* and *offset environmental equivalency ratio* (that were described in Chapter 2) as policy levers to encourage market participation whilst also maintaining environmental quality in receiving waters. Section 3.5 explains operation of a credit market, considers participation of non-emitters as credit buyers, and shows how markets can potentially be used to retain pollution load reduction targets at least cost to society. Section 3.6 concludes.

Table 3.1: Load reduction targets for DIN and fine sediment by 2025 for catchments in the Wet Tropics and the Burdekin regions. Specified in the Reef 2050 WQIP (Australian Government and Queensland Government, 2018 Table 2, p18). Management priority colour coded. MCL = maintain current load. ND = not determined.

Management priority

	very high	moderate	minimal
	high	low	

Region	Catchment / Basin	2025 Reduction Targets (relative to 2012-13 loads)			
		Dissolved inorganic nitrogen		Fine sediment (<16µm)	
		(tonnes)	% reduction	(kilo-tonnes)	% reduction
Wet Tropics	Daintree	MCL	MCL	MCL	MCL
	Mossman	52	50	MCL	MCL
	Barron	52	60	MCL	MCL
	Mulgrave-Russell	300	70	16	10
	Johnstone	350	70	100	40
	Tully	190	50	17	20
	Murray	120	50	8	20
	Herbert	620	70	99	30
Burdekin	Black	ND	ND	ND	ND
	Ross	74	60	ND	ND
	Haughton	640	70	MCL	MCL
	Burdekin River	100	60	840	30
	Don	MCL	MCL	55	30

3.2 Traded commodity

A *water quality credit market* is a general term to describe a market involving trading of reductions in a specific pollutant between buyers and sellers within the geographic boundary of a catchment. If the pollutant reductions traded are reductions in DIN, the market is called *nitrogen credit market*. Similarly, if buyers and sellers trade reductions in fine sediment, this exchange takes place in a *sediment credit market*. In either market, the traded commodity is defined as reductions in the relevant pollutants realised at a defined receiving water. For nitrogen or sediment credit trading in GBR catchments the traded commodity would be defined as the reduction in the quantity of DIN or fine sediment load delivered to End-of-Catchment.

If DIN is the pollutant of interest, an abatement activity (e.g. land use change to constructed or remediated wetlands, installation of on-farm bioreactors, improved fertiliser management above regulated minimum standards) generates a specified quantity of nitrogen load *reduction* at the abatement site compared to the baseline nitrogen load emitted prior to abatement. This on-farm reduction in DIN load is converted into its End-of-Catchment equivalent by multiplying by a *transport coefficient*. The transport coefficient for DIN is a value between zero and one indicating the proportion of the DIN reduction at the abatement site that reaches End-of-

Catchment. The DIN load reduction at End-of-Catchment corresponds to the amount nitrogen credits that can be offered for purchase in the nitrogen credit market by interested parties²⁷.

In principle, a market in sediment credits works in a similar way. The traded commodity is the reduction in fine sediment load achieved at End-of-Catchment by implementing an abatement action. An activity to reduce soil erosion (e.g. physical remediation of gullies or streambanks, riparian revegetation, or improvement in grazing land management) yields a specific reduction in sediment load emitted at the abatement site. A *transport coefficient* is applied to this sediment saving to convert the reduction in sediment load at the abatement site into the equivalent End-of-Catchment load reduction²⁸. The transport coefficient for fine sediment is a value between zero and one indicating the proportion of fine sediments from a given location that are transported to End-of-Catchment during rainfall events. The fine sediment load reduction at End-of-Catchment generated by an abatement action corresponds to the amount of sediment credits that can be offered for purchase in a sediment credit trading market. Remediation of gullies and streambanks, if appropriately implemented and maintained, would be expected to reduce soil erosion for a lengthy period, e.g. 25 years. In this instance, sediment credits would be generated for each year in which End-of-Catchment reductions in fine sediment load were produced, relative to the 'baseline' emissions that would have been expected had the site not been remediated. As a traded commodity, nitrogen or sediment credits are scarce products, and therefore command positive prices in their respective markets.

As noted earlier, Project 3.1.6 considers the potential of water quality credit trading for facilitating economic expansion once basin-specific the Reef 2020 water quality objectives have been met. For catchments with relatively modest anthropogenic pollutant loads (e.g. the Daintree catchment in Table 3.1) an appropriate management strategy might be to prevent any 'economic expansion' with adverse water quality impacts. Were such regulatory restrictions to be implemented, there would not be a role for water quality credit trading in those catchments.

3.3 Supply and demand for credits

Supply of credits

Landholders can choose to participate in a water quality credit trading market as *credit supplier* (i.e. as providers of credits that can be purchased by point source emitters seeking to offset liabilities under their Environmental Authorities (EAs)). A credit supplier incurs private costs in undertaking abatement actions that produce quantifiable reductions in DIN and/or sediment loads at End-of-Catchment; these 'quantifiable reductions' can then be offered for sale as credits in a credit trading market.

A landholder's willingness to participate in a water quality credit trading market as a diffuse source supplier of credits is likely to be dependent, at least to some degree, on the total

²⁷ assuming that the receiving water is the Reef lagoon

²⁸ The transport coefficient for fine sediment is frequently referred to as the 'sediment delivery ratio' in the sediment literature (e.g. Brooks et al., 2016). We use 'transport coefficient' instead 'sediment delivery ratio' in Project 3.1.6 to avoid confusion with the delivery ratio that is specified in the Point Source Water Quality Offsets Policy to account for uncertainty surrounding the capability of diffuse source credits to offset point source emissions.

(private) costs they incur and their perception of whether expected revenues from credit sales will be sufficient to cover those costs. Landholders' total costs of abatement should at least include opportunity costs and transaction costs, together with other relevant costs associated with implementing the abatement action. For example, if DIN abatement involves construction of a treatment wetland, the total cost of abatement may include opportunity costs (i.e. foregone revenue from no longer growing crops on the land), up-front construction cost, periodic maintenance costs, and transaction cost²⁹. If the abatement action involves reducing stocking density and physical repair of streambanks to reduce sediment erosion, the landholder's total abatement costs would likely include opportunity costs (i.e. foregone profit from reduced stocking density), remediation costs (e.g. fencing, earthworks, grade control structures, revegetation etc.), cost of erecting off-stream watering points for livestock, and transaction cost.

For some agricultural businesses, changing land management practice to reduce DIN (e.g. sugarcane farming) or sediment runoff (e.g. livestock grazing) may also introduce costs associated with increased risks and uncertainty about the future impacts of abatement actions on profitability. For example, in sugarcane, risks and uncertainty associated with reduced fertiliser application may be reflected in concerns that increased variability in yield will affect profitability (Beverly et al. 2016; Kandulu et al. 2018; Farr et al. 2019). In these circumstances, the costs of exposure to risk and uncertainty should be quantified and added to the landholder's opportunity costs and transaction costs to fully reflect the total abatement cost associated with practice change.

The sum of these different components of the private total cost of abatement represents the *minimum* level of compensation payment required by the credit supplier for supplying the relevant number of credits into a water quality credit trading market. Hence, when a landholder offers to supply credits to the credit market, the revenue they receive from sale of those credits needs to at least cover this minimum compensation requirement. This enables the landholder's minimum *willingness to accept* (WTA) compensation to be expressed as a minimum required price per credit (i.e. per unit load reduction) at End-of-Catchment. Landholder j 's minimum required credit price (in \$/credit) for sourcing credits via abatement action k is calculated by dividing their total abatement cost (TAC_k) by the number of End-of-Catchment water quality credits (Q_k) generated:

$$WTA_{jk} = \frac{TAC_{jk}}{Q_{jk}} \text{ for landholder } j \text{ and abatement action } k \quad (3.1)$$

WTA_{jk} can be regarded as the minimum *offer price* P_{jk} that landholder j will require for supply of Q_k End-of-Catchment credits (*offer quantity*) into the market from abatement action k . If landholder j undertakes multiple abatement actions on their property (e.g. riparian revegetation and reducing stocking density below the regulated maximum level to decrease soil erosion) abatement costs and credit production can be summed to produce the landholder's minimum offer price for the total quantity of credits generated as shown in equation (3.2):

²⁹ Transaction costs generally refer to the cost of financial and time resources required to learn about and engage with the credit market (Coggan et al. 2014).

$$P_j = WTA_j = \frac{\sum_{k=1}^K TAC_{jk}}{\sum_{k=1}^K Q_{jk}} \text{ for landholder } j \text{ and abatement actions } k = 1, 2, \dots, K \quad (3.2)$$

The offer quantities $Q_j = \sum_{k=1}^K Q_{jk}$ and offer prices $P_j = WTA_j$ are expressed in End-of-Catchment credits and \$/End-of-Catchment credits, respectively. On-farm credits are converted to End-of-Catchment credits knowing tc_{j-EoC} the transport coefficient for the pollutant concerned between farm j and End-of-Catchment:

$$Q_{j@EoC} = tc_{j-EoC} \cdot Q_{j@farm} \quad (3.3)$$

Hereafter, unless noted otherwise, quantities of credits supplied will be expressed at End-of-Catchment. Generalising across all participating landholders ($j = 1, 2, \dots, J$) in a given catchment, and noting heterogeneity in abatement costs and spatial variability in transport coefficients to End-of-Catchment, the aggregate supply of credits (for either DIN or sediment) to the market can be constructed as shown in Figure 3.1 by organising the quantity of credits generated by each offer to supply in order of increasing offer price.

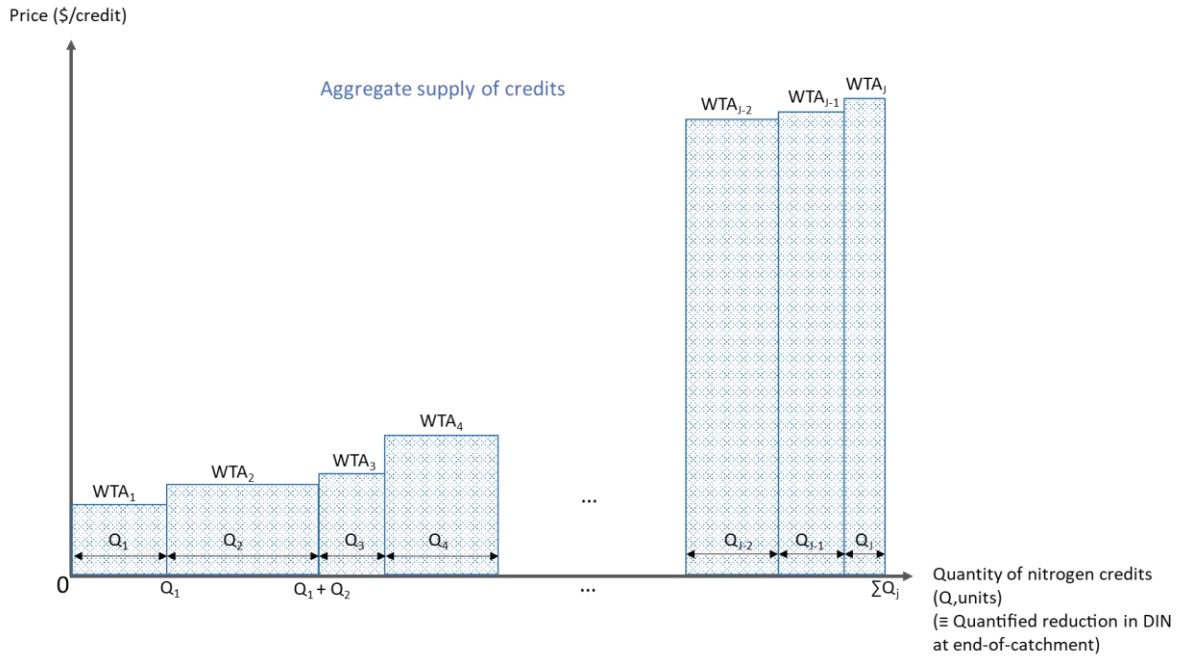


Figure 3.1: Aggregate supply of nitrogen credits from J participating landholders ($j = 1, 2, \dots, J$). (For simplicity, Figure 1.1 assumes that each landholder undertakes only a single abatement action which generates Q_j credits).

Demand for credits

The Queensland Government's Point Source Water Quality Offsets Policy (hereafter, PS WQ Offsets Policy) allows point source emitters and proponents of new development projects (i.e. existing and potential new EA holders) to trade bilaterally with landholders as offset providers, or to purchase water quality credits from a market (State of Queensland 2019f). Under the PS WQ Offsets Policy, point source emitters such as sewage treatment plants or aquaculture operators are allowed to exceed the end-of-pipe nitrogen and/or sediment discharge limits in their EAs provided they have acquired sufficient quantities of relevant water quality credits to *offset* their discharge limit exceedances. Proponents of new development projects have the

option to purchase water quality credits to offset residual water pollution emissions (after conducting on-site abatement) as a condition of obtaining EA approval. When existing and new EA holders are looking to offset point source emission exceedances via credits supplied from diffuse sources, the *delivery ratio* and *offset environmental equivalency ratio* specified in the PS WQ Offsets Policy will generally require that more than the nominal quantity of credits are purchased to offset their actual load exceedance (see Figure 2.2 in Chapter 2 Regulatory and Governance Framework). The role of the delivery ratio and offset environmental equivalency ratio will be discussed further in Section 3.5 below, and again in Chapter 8.

The value of offsets, and thus credits, to a point source buyer is reflected in their maximum *willingness to pay* (WTP) to obtain those credits. A point source emitter would choose to purchase credits to offset exceedance liabilities under their EA if the total cost of credit purchase (i.e. the market price of a credit multiplied by the quantity of credits required) is *lower* than the total cost of abatement that the point source would otherwise have to incur to reduce emissions to within their licenced limit. A buyer's maximum WTP for the final credit purchased equates to the cost that buyer would incur in abating the last unit of the pollutant necessary to reduce emissions to within their licenced limit. This is termed the marginal abatement cost (MAC) for the final unit of pollution reduction. For example, for point source n , if 1 DIN credit is sufficient to offset 1 kg of DIN licence exceedance at End-of-Catchment³⁰, in Figure 3.2, point Q_n indicates the total DIN load exceedance that point source n would seek to offset by purchasing credits.

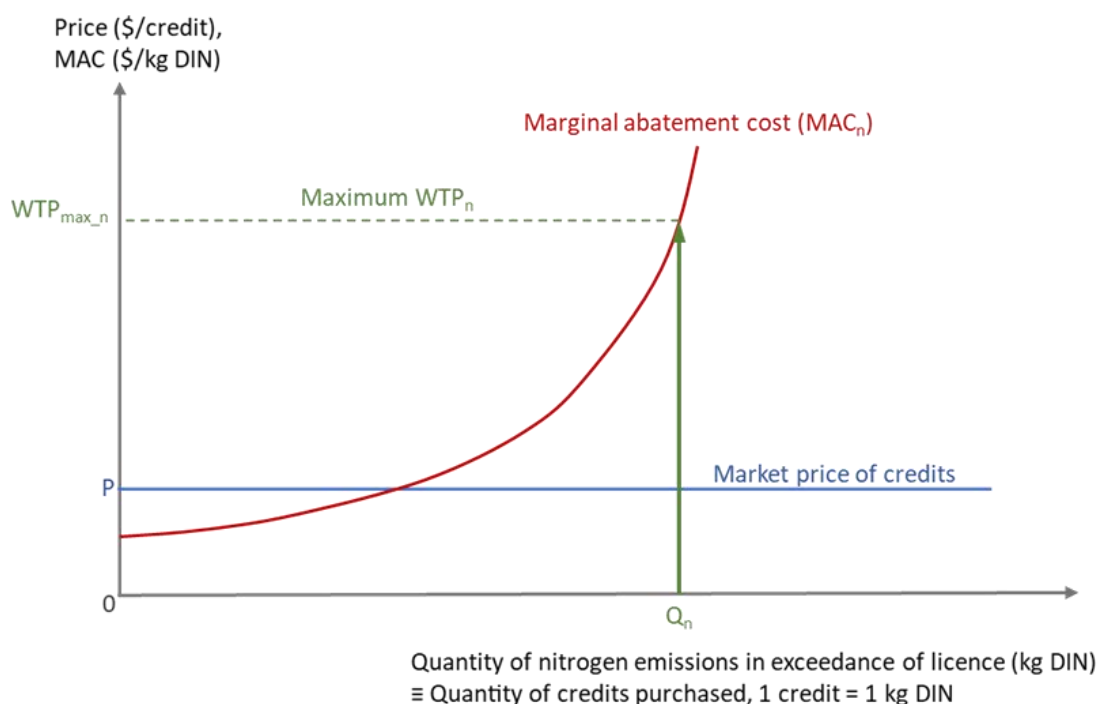


Figure 3.2: Determination of a point source DIN emitter's maximum willingness to pay (WTP) for a DIN credit.

³⁰ In this example, we ignore any additional offsetting requirements imposed by the delivery ratio and environmental equivalency ratio in the PS WQ Offsets Policy.

The total abatement cost (TAC_n) required to reduce emissions by Q_n kg by abatement at the point source is given by the area under the MAC curve from 0 up to Q_n . At Q_n , it is cheaper for this emitter to purchase Q_n credits at price P than to incur cost TAC_n to abate Q_n kg of End-of-Catchment DIN load. This enables the maximum price that point source emitter n would be prepared to pay for an End-of-Catchment credit (P_n) to be calculated as shown in Equation 3.4:

$$P_n = WTP_n = \frac{TAC_n}{Q_n} \text{ for point source emitter } n \quad (3.4)$$

The maximum price that the buyer would be willing to pay to obtain End-of-Catchment credits is termed their *bid price* P_n . Analogously, the quantity of End-of-Catchment credits demanded by a buyer at that price is termed their *bid quantity* Q_n .

Many point source emitters in GBR catchments are located at, or very close to, End-of-Catchment (e.g. sewage treatment plants for cities along the coastline, or aquaculture facilities). The ‘receiving water’ specified in these emitters’ EAs will therefore be End-of-Catchment. A further complication arises, however, for point source emitters whose ‘receiving water’ is appreciably upstream from End-of-Catchment, as illustrated in Figure 3.3.

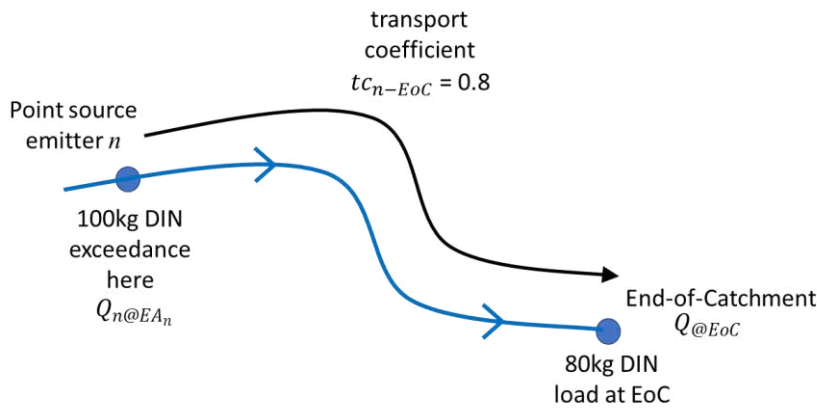


Figure 3.3: Converting point source loads to End-of-Catchment loads for points sources appreciably upstream from End-of-Catchment.

For point source emitter n located appreciably upstream from End-of-Catchment, the quantity of End-of-Catchment credits ($Q@EoC$) that equate to a given level of load to be offset at point source n 's receiving water ($Q_{n@EA_n}$) is calculated as follows, knowing the transport coefficient (tC_{n-EoC}) for the relevant pollutant between the point source's receiving water and End-of-Catchment:

$$Q@EoC = Q_{n@EA_n} \cdot tC_{n-EoC} \text{ or equivalently } Q_{n@EA_n} = \frac{Q@EoC}{tC_{n-EoC}} \quad (3.5)$$

Consequently, since tC_{n-EoC} is less than one for the situation illustrated in Figure 3.3, point source emitter n will be prepared to pay a higher price per credit expressed in End-of-

Catchment load than they would have been if the credits had been expressed in terms of pollutant load at the local receiving water for point source n .³¹

Given that emissions under EAs should already be included in End-of-Catchment loads, credits purchased to offset emissions in excess of EA limits allow point source emitters to fulfil their EA obligations at reduced cost, but *do not* contribute towards meeting water quality targets because credit purchase generates no net reduction in emissions below the level permitted in the EA³².

Non-emitting buyers such as natural resource management groups, NGOs, and government agencies could also be allowed to purchase credits from the market. Commercial businesses who do not operate under EAs (e.g. cosmetics companies, tourism operators) but seek voluntarily to offset adverse environmental impacts of their businesses to enhance their environmental credentials may also choose to purchase water quality credits in the market. Credits purchased by these non-emitting buyers would contribute directly to meeting the water quality targets (Table 3.1).

Generalising across all potential emitting buyers of credits ($n = 1, 2, \dots, N$) (i.e. EA holders such as sewage treatment plants, quarries, abattoirs and aquaculture businesses) and potential non-emitting buyers (e.g. NGOs and tourism operators) in a given catchment, an aggregate demand for credits (in either DIN or sediment) can be constructed as shown in Figure 3.4. Each buyer in Figure 3.4 has their own maximum WTP (WTP_n) (which equates to their bid price P_n) and bid quantity of credits demanded (Q_n). The sum of the individual bid quantities, ordered by decreasing bid price, produces the aggregate demand curve for credits.

³¹ The impacts of the relevant transport coefficients have already been taken into account in the pricing shown in Equations 2.2 and 2.4.

³² The delivery ratio and environmental equivalency ratio will typically require that the quantity of credits purchased is higher than the nominal licence exceedance that is being offset. In situations (e.g. favourable weather conditions) when the load reduction generated by the purchased credits is higher than the licence exceedance this 'surplus' offsetting will contribute towards achieving Water Quality Targets. This issue is explored further in Chapter 8.



Figure 3.4: Aggregate demand for DIN credits from potential DIN credit buyers ($n = 1, 2, \dots, N$) where buyers could be existing and potential new environmental authority holders who purchase credits to offset emissions in exceedance of their licences, and non-emitters who purchase credits to enhance their environmental credentials as a way of making their products more appealing to environmentally-conscious consumers.

Credit market

A water quality credit trading market brings credit suppliers and credit buyers together into a single platform where the buyers place bids to buy by submitting their bid prices and bid quantities, and sellers offer to supply specified quantities of credits by submitting their offer prices and offer quantities. Provided that bids to buy and offers to supply eventuate, and that the highest bid price exceeds the lowest offer price, the aggregate supply of credits (Figure 3.1) and aggregate demand for credits (Figure 3.4) will enable a market price for credits, P^* to emerge. Water quality credit trading markets can be implemented in various configurations. Coggan et al. (2013) provide a helpful summary of commonly-encountered arrangements. These will be described briefly in Section 1.6.

3.4 Environmental regulator

In water quality credit trading a market manager oversees market operation, safeguards the scarcity of the traded commodity and maintains a market environment conducive to trading. The environmental regulator is another key actor in water quality credit trading. The environmental regulator sets pollutant discharge limits in EAs to safeguard environmental quality in the receiving water(s). The environmental regulator must be satisfied of the validity of credits traded in the market before they will allow credit purchasers to offset EA exceedances through credits purchases. In some situations, the environmental regulator also performs the market manager role, but in many cases the market manager will be an independent entity.

3.4.1 Delivery ratio and environmental equivalency ratio

Under the PS WQ Offsets Policy, the environmental regulator sets the *delivery ratio* and the *offset environmental equivalency ratio* to be applied when diffuse source credits are used to offset point source exceedances of EA licensed pollutant loads. The *delivery ratio* aims to account for uncertainty regarding the ability of a diffuse source offset to deliver an equivalent reduction in the mass load of the relevant pollutant in the receiving environment specified in the purchasing point source's EA (State of Queensland 2019f). The *offset environmental equivalency ratio* aims to account for uncertainty associated with the chemical form or species of pollutant reduction provided by a pollution offset relative to the chemical form or species of the point source emission exceedance that the offset is intended to negate (State of Queensland 2019f).

As explained in Chapter 2, Section 2.5, the *delivery ratio* (D_r) and *environmental equivalency ratio* (E_{er}) are multiplying factors that are applied as shown in Equation 3.6 to determine the quantity of diffuse source credits that a point source has to purchase in order to offset a given level of EA exceedance.

$$\text{Offset required} = \text{Load to be offset} \times D_r \times E_{er} \quad (3.6)$$

In Equation 3.6 the offset required is expressed in units of credits (e.g. kg DIN credit or tonnes sediment credit) and the load to be offset is expressed in pollutant load at the receiving water specified in the point source emitter's EA. The delivery ratio and environmental equivalency ratio would each be expected to be at least one. Currently, under the PS WQ Offsets Policy, the delivery ratio is 1.5 and the environmental equivalency ratio is 1.0 (State of Queensland 2019f).

3.5 Point source offsetting via diffuse source credits

In Project 3.1.6, unless noted otherwise, water quality credits are expressed in terms of their load offsetting capability at End-of-Catchment. This provides a 'common currency' for pollutant load reductions in terms of the reduction in pollutant load delivered to the GBR lagoon. Where, as will often be the case, the point sources that are purchasing offsets are located at End-of-Catchment (e.g. sewage treatment plants for major cities along the coastline, or aquaculture facilities) the quantity of End-of-Catchment credits that have to be purchased to offset a particular load exceedance can be calculated directly via Equation 3.5. For example, if an aquaculture facility discharging to End-of-Catchment wants to offset 90kg DIN exceedance, and if (for illustration only) $D_r = 1.5$ and $E_{er} = 1.2$ then the diffuse source credit purchase required to offset this exceedance will be $90 \times 1.5 \times 1.2 = 162$ kg DIN (expressed in End-of-Catchment diffuse source DIN credits).

If, however, returning to the example in Figure 3.3, point source n wants to use diffuse source credits to offset 100kg DIN exceedance at receiving water location n (where n is upstream from End-of-Catchment) then the required diffuse source credit purchase at location n will be $100 \times 1.5 \times 1.2 = 180$ kg DIN. Referring back to Figure 3.3, allowing for the transport coefficient of 0.8 between location n and End-of-Catchment, this offset requirement can be satisfied by purchasing $180 \times 0.8 = 144$ End-of-Catchment DIN credits. However, for these credits to be effective in offsetting point source n 's EA exceedance at receiving water n , the credits must be

supplied by diffuse sources upstream of location n . To protect water quality at location n , the environmental regulator would have to ensure that this constraint was applied. This could be implemented easily by attaching a 'source location' (e.g. quoted in terms of distance and connection pathway from End-of-Catchment) to End-of-Catchment diffuse source credits offered in the market.

3.6 Market configurations

A range of different market configurations have been applied in environmental offset markets. Coggan et al. (2013) provide a helpful summary and suggest standardised nomenclature. Market configurations with various forms of intermediary between buyers and sellers generally aim to increase market participation by reducing transaction costs and providing mechanisms for devolving risks. The following descriptions draw heavily on Coggan et al's research.

3.6.1 *Bi-lateral credit supply agreement*

The PS WQ Offset Policy allows an individual point source emitter to negotiate with an individual diffuse source credit supplier to implement changes in land use or land management that are sufficient to generate an adequate quantity of diffuse source credits to offset the point source emitter's EA exceedance (taking due account of delivery ratio and environmental equivalency ratio). In practice, these one-off credit supply agreements are effectively tri-lateral because the adequacy of the proposed offset will have to be demonstrated to the satisfaction of the environmental regulator. The costs incurred in providing the necessary level of reassurance to the environmental regulator (e.g. by commissioning a catchment modelling study) will be borne by the credit purchaser. The lengthy and detailed negotiations that inevitably arise between all three parties increase the transaction cost of bi-lateral credit supply agreements considerably. This type of bespoke offsetting arrangement is not a market in the sense of Project 3.1.6, so it is not considered further here.

3.6.2 *Fee in-lieu arrangement*

In a fee in-lieu arrangement emitters can offset their residual environmental impact after undertaking appropriate on-site mitigation by making a stipulated level of payment to a designated intermediary³³. Once payment is made, the emitter's liability has been discharged and it is the responsibility of the designated intermediary to secure an appropriate offset with the monies provided. The financial risk (i.e. in this context, the risk of having insufficient funds to secure an appropriate offset) rests with the intermediary. The Reef Trust Offsets Policy and off-site implementations of urban stormwater management under State Planning Policy operate as fee in-lieu arrangements, but a fee in-lieu arrangement is not available under the PS WQ Offsets Policy. The intermediary will typically operate a reverse tender mechanism to elicit offers to supply credits, but, as this constitutes only the supply side of a full credit market, fee in-lieu arrangements are not considered further here.

³³ The intermediary in fee in-lieu arrangements is typically, but not necessarily, a public entity (Coggan et al. 2013).

3.6.3 Credit bank

Coggan et al. describe the operation of a credit bank as shown in Figure 3.5.

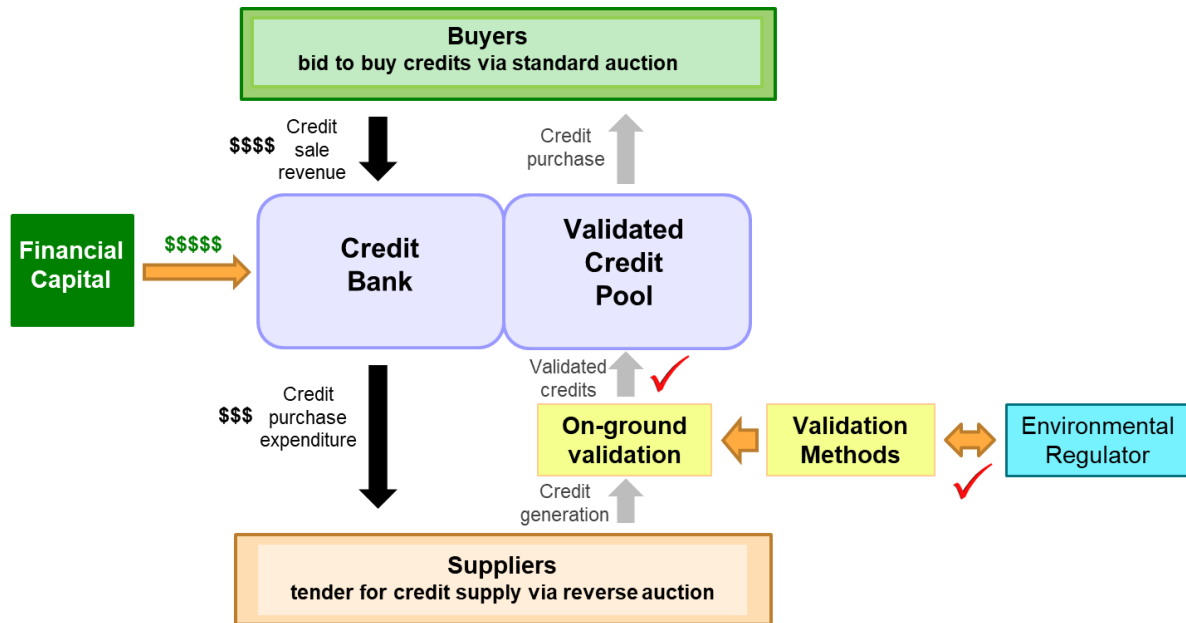


Figure 3.5: A credit trading market with a credit bank as an intermediary between credit suppliers and credit buyers. The bank funds generation of a pool of validated credits. Credits are validated through application of validation methods that have been pre-approved by the environmental regulator (Adapted from Fig. 4 p.148 in Coggan et al. (2013)).

The credit bank enters into up-front financial arrangements with credit suppliers to fund on-ground actions that generate credits. Credits from those actions accrue to the bank who will then seek to sell the credits to credit purchasers. In order for credits to be marketable as offsets against EA exceedances, they must be deemed valid by the environmental regulator. This would typically be achieved by the bank developing a set of validation 'methods' that are approved by the environmental regulator. Validation using the appropriate 'method' would then have to be checked by an independent verifier before the credits generated from a particular on-ground action could be offered for sale by the bank to prospective credit buyers. In the arrangement described above, the bank carries the financial risk and the production risk associated with credit supply (Coggan et al. 2013). Financial risk surrounds the bank's outlay on on-ground works prior to securing buyers for the credits those works will generate. Production risk surrounds assessment of both the level and validity of the credits generated. These risks could potentially be shared between the bank and the credit supplier by, for example, the bank offering the credit supplier a portion of the revenues from the sale of the credits generated by the project in return for the supplier committing some of their own costs and resources to implement the on-ground actions.

Although in the arrangement described above the bank (or the bank and the credit supplier in combination) carries the financial and production risks, under the PS WQ Offsets Policy liability for the adequacy of the purchased offset to satisfy the conditions of the purchaser's EA remains with the credit purchaser (State of Queensland 2019f).

3.6.4 Credit aggregator

Coggan et al. describe the operation of a credit aggregator as shown in Figure 3.6.

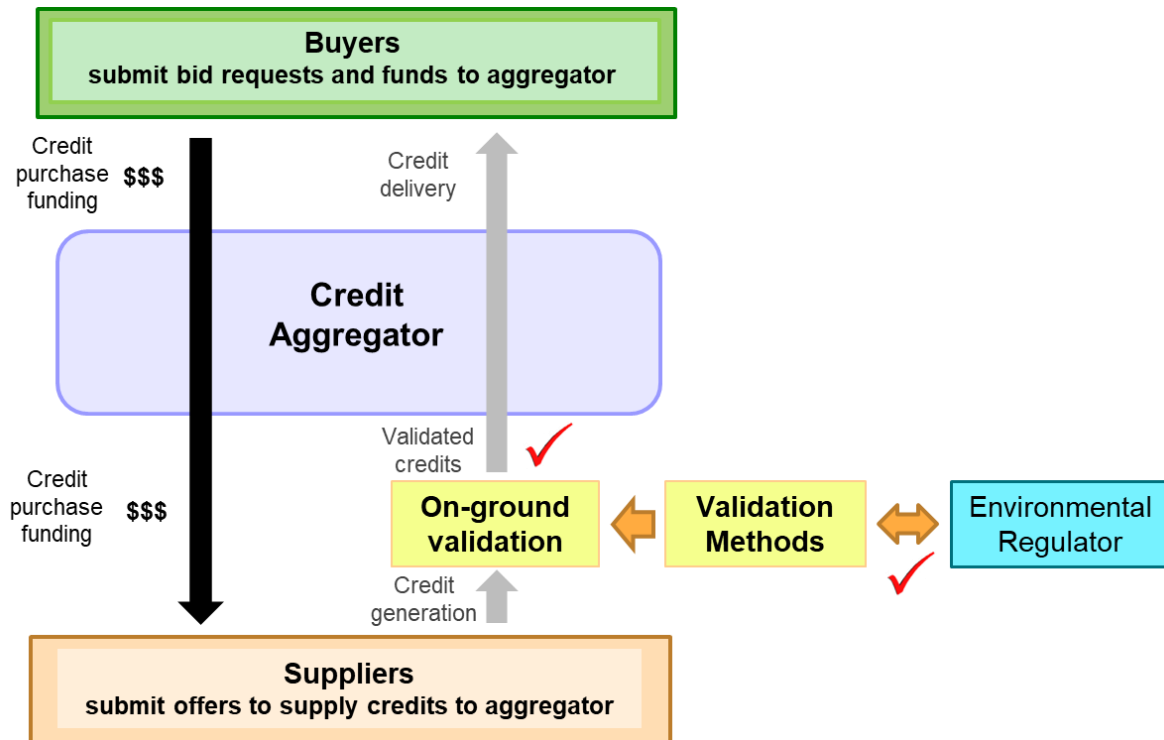


Figure 3.6: A credit trading market with a credit aggregator as an intermediary between credit suppliers and credit buyers. The aggregator maintains a database of credit buyers and credit suppliers. Buyers submit bid requests to the aggregator. A buyer provides credit purchase funding when the aggregator identifies a willing supplier. (Adapted from Fig. 3 p.148 in Coggan et al. (2013)).

The credit aggregator conducts negotiations to identify potential credit suppliers and potential credit buyers and compiles this information into a database. When an offset buyer contacts the aggregator to request purchase of a given quantity of credits, the aggregator accepts payment for the credits from the purchaser and undertakes to source the credits from the suppliers in their database. The aggregator differs from a credit bank because the aggregator does not provide up-front payments for credit supply and also because the offset suppliers in the aggregator's database have not yet implemented the on-ground actions required to generate verified credits. This considerably reduces the aggregator's exposure to financial and production risk. There will, however, be a delay between the time at which the purchaser transfers credit purchase funding to the aggregator and the time at which verified credits are obtained. This delay could be relatively short if credits are sourced via changes in land management practice (e.g. reductions in fertiliser applications to cane or reductions in cattle stocking density) but would be considerably longer for credits sourced from land use change (e.g. construction of treatment wetlands or remediation of large gullies).

3.6.5 Clearing house

Coggan et al. describe the operation of a clearing house as shown in Figure 3.7.

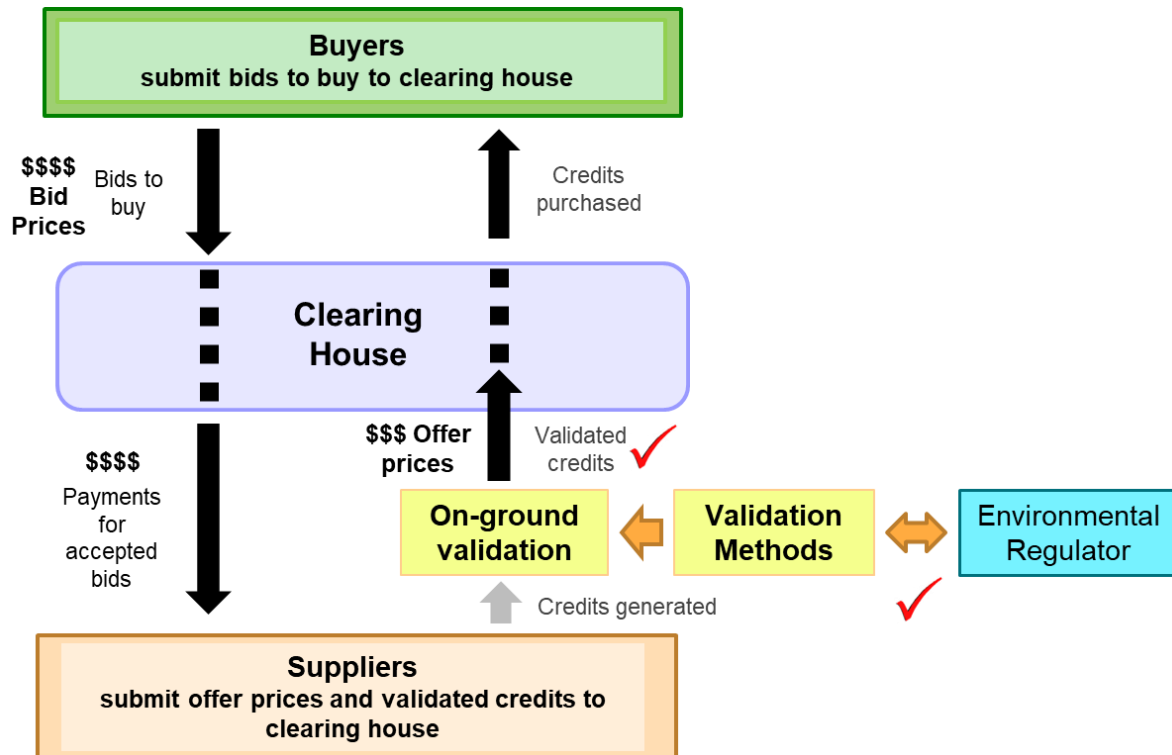


Figure 3.7: A credit trading market with a clearing house as an intermediary between credit suppliers and credit buyers. (Adapted from Fig. 6 p.149 in Coggan et al. (2013)).

The clearing house provides a marketplace in which multiple credit buyers can purchase validated credits from multiple credit suppliers, within constraints imposed by the environmental regulator (e.g. credit trades must ensure that the discharge limits specified in EAs are respected, and relevant delivery ratios and environmental equivalency ratios are enforced). Credit suppliers lodge quantities of validated credits and their minimum offer prices with the clearing house. Credit buyers lodge their demand for purchases of validated credits and their maximum bid prices. The clearing house identifies the best match between buyers' and sellers' requirements, within relevant constraints, and provides the suggested matches to both parties for their approval. In the form described above, financial risk sits with the credit suppliers, and there is no production risk because offers lodged with the clearing house are for pre-validated credits. This level of exposure to financial risk may be unacceptable to credit suppliers, particularly in the early stages of market development when there is considerable uncertainty regarding the level of demand. To accommodate these concerns, a variant of the clearing house arrangement can operate on a 'look ahead' basis. A 'look ahead' clearing house would accept 'next round' offers to supply credits provided that minimum offer prices are met and 'next round' bids to buy credits at prices not exceeding maximum bid prices. Best matches between 'next round' offers to supply and bids to buy would be identified and the relevant parties would enter into supply – purchase contracts that included a specified timescale for credit delivery (within the same operating year for credits sourced from land management change, and within an agreed time period for credits sourced from land use change). 'Next round' operation reduces credit suppliers' exposure to financial risk, but they would now be

exposed to production risk because they have contracted to supply credits to the purchaser before their on-ground actions have been validated.

3.6.6 Smart market

A smart market (Prabodanie et al. 2010; Raffensperger et al. 2017) is an automated clearing house operating under a cap-and-trade system in which the total gains from trading 'next round' bids to buy and offers to supply are maximised via a linear program. A smart market could be configured to accept bids and offers generated via a smart phone app that appended details of suppliers' and buyers' spatial locations, suppliers' proposed practice and land use changes, and buyers' EA licence conditions to offers to supply and bids to buy. This should further reduce transaction costs for all parties. This type of cap-and-trade systems are not considered further in Project 3.1.6.

3.7 Summary

This Chapter has described the fundamentals of water quality trading. The target reductions in End-of-Catchment anthropogenic DIN and sediment loads required to achieve 2025 Water Quality Objectives (State of Queensland 2019b) were reported for GBR catchments. The commodities traded in water quality trading markets for nutrient and sediment were defined, and the construction of supply and demand curves for the relevant credits was described. The role of the environmental regulator, a key actor in water quality credit trading, was defined with regards to setting licence conditions in EAs and setting the delivery ratio and environmental equivalency ratio as specified in the PS WQ Offsets Policy. Point source offsetting via diffuse source credits was explained, for situations in which the point source discharges to End-of-Catchment or to a receiving water further upstream. Basic configurations for water quality trading were then described and the roles of a credit bank, credit aggregator and clearing house were explained. The material covered in this Chapter sets the background for quantification of supply and demand for DIN and sediment credits in subsequent chapters and for more detailed consideration of complexities such as trading in stacked credits in multiple pollutants.

4.0 NITROGEN LOSSES FROM SUGARCANE PRODUCTION

Jim Smart^{1,2}, Syezlin Hasan¹, Joe McMahon¹, Chantal Saint Ange¹: using P2R APSIM modelling data from Melanie Shaw Qld DNRM

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4. Summary

This Chapter summarises the background to nitrogen losses from sugarcane production prior to considering, in Chapter 5, how nitrogen credits could be generated from improvements in cane management practice. Topics addressed in this Chapter are:

- the Paddock to Reef (P2R) modelling and monitoring framework
- modelling of nitrogen losses from sugarcane production
- fertiliser management practices for sugarcane
- P2R predictions of nitrogen losses and cane yields from sugarcane production in Wet Tropics catchments under different levels of nitrogen management practice.

P2R modelling predictions indicate that for sugarcane production in 563 distinct management units across eight catchments in the Wet Tropics, on average:

- nitrogen losses vary with soil type and soil permeability
- within-year average (across the cane production cycle) nitrogen losses and cane yields for a given soil type are considerably more variable than across-year averages
- nitrogen losses reduce as fertiliser management practice improves
- differences in nitrogen loss and cane yield between catchments under given levels of management practice broadly track catchment soil types.

Also:

- substantial reductions in average across-year nitrogen losses can be achieved by improving fertiliser management practice to adjust fertiliser application rates to:
 - (i) modify District Yield Potential-based rates for soil organic carbon, nitrogen inputs from legume fallow crops, and mill by-products, and
 - (ii) further account for finer scale variation in 'block yield potential' (determined from records of past yields)
- within-year average nitrogen losses typically also reduce as fertiliser management practice improves; particularly when fertiliser applications account for variation in 'block yield potential'.
- Across-year and within-year average cane yields are much less sensitive than nitrogen losses to reductions in fertiliser application.

4.1 Modelling, monitoring and reporting

The land area draining into the GBR Lagoon covers more than 420,000 km², comprises 35 main river basins and stretches for 2,300km along the Queensland coastline (Waters et al. 2014). The Paddock to Reef (P2R) modelling, monitoring and reporting framework has been developed as a tractable, scientifically robust mechanism for tracking progress towards the Reef Water Quality Objectives across this entire drainage area (Ellis and Searle 2014; Waters et al. 2014; McCloskey et al. 2017).

Relevant to Project 3.1.6, P2R modelling uses a combination of sub-catchment- and paddock-scale models operating on a daily timestep to predict End-of-Catchment loads of DIN and fine sediment to the GBR Lagoon (Figure 4.1). Key inputs to P2R modelling are the areas of different land uses, levels of land management practice adoption on agricultural land, and daily weather. Land use data detailing areas of grazing, native forest, sugarcane, bananas etc. are obtained from the Queensland Government's land use mapping (QLUMP) and collated to sub-catchment scale. Management practice adoption data are collated regularly and published as part of the Reef Report Card reporting cycle. Agricultural land management practices are classified on a D, C, B, A scale using the P2R Water Quality Risk Framework. D-level practices present the highest risk to water quality and A-level practice the lowest. Land management practice adoption is collated to sub-catchment scale for P2R modelling. Daily weather data (rainfall, sunshine hours, temperature and potential evapotranspiration) at approximately sub-catchment resolution are obtained from the SILO weather database (Queensland Government 2020).

Hydrological modelling using the eWater CRC's Source Catchments (eWater Ltd 2020) is the backbone of the P2R modelling framework. Calibrated Source Catchments hydrological models have been developed for each of the 35 Reef basins (Waters et al. 2014; McCloskey et al. 2017). Knowing land uses, land management practices and weather, basin-specific Source models predict the transport and fate of DIN and fine sediment to derive predictions of End-of-Catchment DIN and fine sediment loads to monitor progress towards the End-of-Catchment WQOs.

Separate daily time step, paddock-scale models are used for sugarcane production (APSIM (Keating et al. 2003)), bananas and horticulture (HowLeaky (McClymont et al. 2011)), and grazing (GRASP (McKeon et al. 1990)) to predict DIN and/or fine sediment losses from the different agricultural sectors under given management practices and weather. DIN and sediment loss predictions from APSIM and HowLeaky feed into the Source Catchments hydrological models at sub-catchment scale. For grazing, the GRASP model is used to predict ground cover on grazing land as an input into the Universal Soil Loss Equation (Renard and Ferreira 1993) to predict erosion of fine sediment from hillslopes, gullies and riverbanks on grazing land, again at sub-catchment scale. DIN transport to End-of-Catchment is modelled by apportioning paddock-scale DIN losses from APSIM or HowLeaky into runoff to surface waters and infiltration to groundwater drainage, and then applying appropriate transport coefficients for the relevant pathway from the sub-catchment in which the paddock loads are generated through to End-of-Catchment. Fine sediment transport to End-of-Catchment is modelled using SedNet functionality within Source, allowing for downstream transport and deposition (Ellis and Searle 2014). Calibration of the P2R modelling framework was undertaken using daily weather

data for 1987 – 2013 and observed river flows and pollutant concentrations from 10 End-of-System gauging stations (Waters et al. 2014).

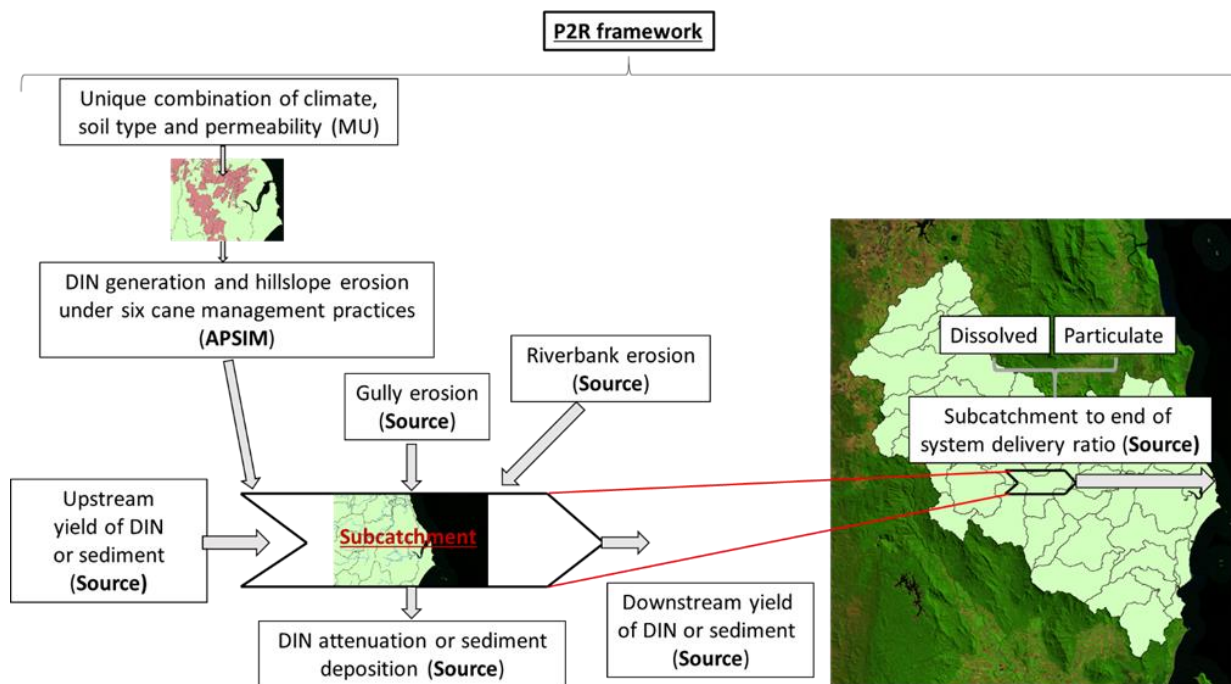


Figure 4.1: Model components within the Paddock to Reef (P2R) modelling framework, operating at management unit (MU) resolution. The model component used at each stage is indicated in brackets (APSIM or Source).

4.2 Modelling nitrogen losses in sugarcane production

This Section provides a brief description of nitrogen losses from cane production. It then presents P2R modelling results for average (1983-2013) and year-specific cane yields and DIN losses for near-paddock-scale management units in Wet Tropics catchments under different levels of nitrogen management practice. These data enable the changes in End-of-Catchment DIN loads for specific changes in management practice on individual management units to be calculated, as will be described in Chapter 5.

4.2.1 Nitrogen losses from cane production

The nitrogen cycle in cane production systems is illustrated in Figure 4.2.

Nitrogen cycle in cropping systems

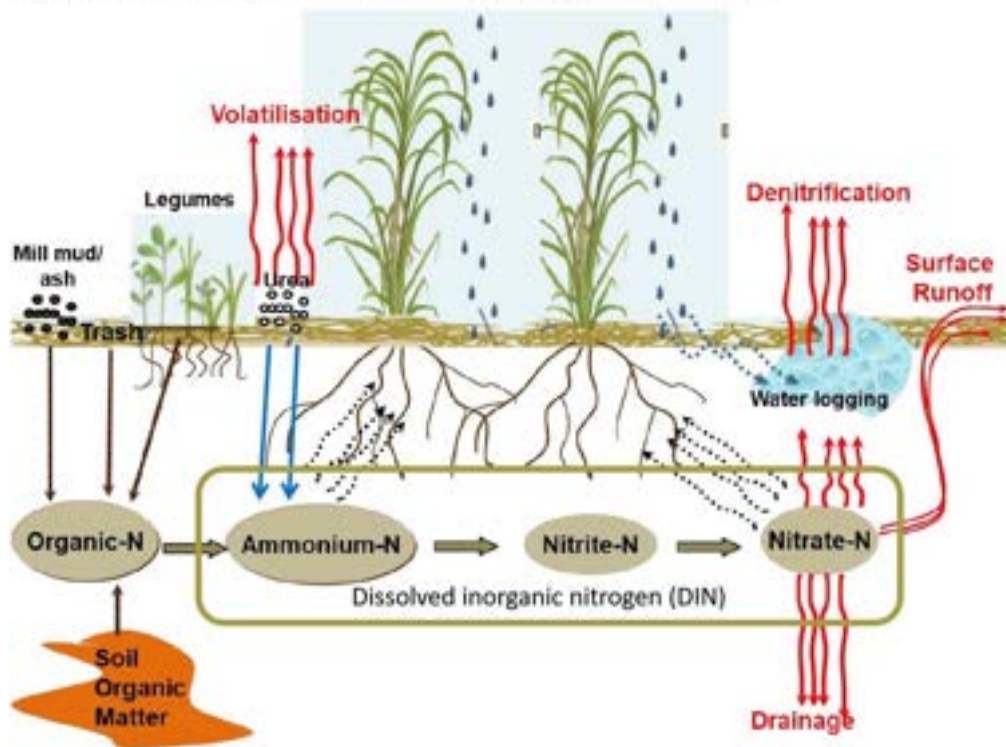


Figure 4.2: The nitrogen cycle in sugarcane cropping systems (from Moody 2016: Nitrogen Treatment Forum (2016))

Soil organic matter, fixation of atmospheric nitrogen by legumes, cane trash and applications of mill mud on the soil surface contribute organic nitrogen to the root zone in the soil below the cane crop. Some of this organic nitrogen transforms to dissolved inorganic nitrogen (DIN) species (ammonium-N, nitrite-N and nitrate-N) in pore water in the root zone. Artificial fertilisers such as urea applied to the crop contribute to this DIN pool directly. In agricultural settings, provided conditions are not anoxic, the majority of DIN will be present as nitrate-N. DIN is highly bio-available and its uptake via the crop's root system assists plant growth and boosts crop yield. DIN that is not taken up by the crop is highly susceptible to being transported from the crop root zone to waterways via one of two main pathways. The first pathway is via surface runoff. During rain events in the Wet Tropics surface water pathways are highly connected and flow rates are also high. In these situations, DIN is transported rapidly to End-of-Catchment via the surface water pathway. There is little opportunity for DIN to be removed from the aquatic system via plant uptake, soil accretion or denitrification, so – in Wet Tropics catchments - a very high proportion of the DIN that leaves the root zone via the surface water pathway will reach End-of-Catchment. This is confirmed by catchment monitoring data (Turner et al. 2012). The second transport pathway is via infiltration to ground water below the rootzone. DIN transport through the groundwater pathway (referred to as 'drainage' in P2R) to End-of-Catchment is slower and less direct, making the DIN more susceptible to attenuation due to adsorption, accumulation or retention. Studies by Rasiah et al. (2005, 2003b, 2003a), however, suggest that much of the DIN that is flushed into shallow groundwater below the root zone during rain events in the wet season re-emerges later via lateral flow into surface water systems during base flow conditions. Drawing on Rasiah et al.'s research and Webster et al. (2012), a recent study of the economic and environmental consequences of farmers' adoption

of improved fertiliser management practices in the Tully catchment by van Grieken et al. (2019) considered that at least 60% of the DIN lost from sugarcane land to drainage ultimately reaches End-of-Catchment via lateral flow connections to surface water. Following van Grieken et al. (2019), Project 3.1.6 assumes that 60% of the DIN that leaves the paddock by the drainage pathway ultimately reaches End-of-Catchment.

4.2.2 Spatial management units in P2R predictions

Project 3.1.6 uses P2R predictions of DIN losses and cane yield in Wet Tropics catchments under different levels of fertiliser management practice as input data for estimating supply of nitrogen credits from improvements in fertiliser management practice in cane production. Dr Melanie Shaw in the Department of Natural Resources and Mines provided APSIM-derived P2R simulations of average and individual year-specific DIN losses and cane yields from cane production under six levels of fertiliser management practice in the Wet Tropics. These results were produced by running APSIM P2R on daily weather data for each year from 1987 – 2013. DIN losses and cane yields were provided for cane land in unique combinations of six soil types (Table 4.2), three soil permeabilities (low medium and high permeability) and 101 SILO climate zones (Queensland Government 2020). These unique climate zone, soil type, soil permeability combinations of cane land, within a single P2R hydrological sub-catchment, are the basic **management units** that Project 3.1.6 uses for nitrogen credit generation from fertiliser management practice improvement in cane. Figure 4.3, Figure 4.4, and Figure 4.5 show management units in the Johnstone catchment. Maps showing management units in the Russell-Mulgrave, Johnstone, Tully, Murray and Herbert catchments are provided in Appendix 4A.

Table 4.1: Soil types used in APSIM P2R modelling in the Wet Tropics

<i>Name</i>	<i>Area of cane land modelled (ha)</i>	<i>Description</i>	<i>Australian Soils Classification</i>	Closest hydrological functional group
Coom	28,800	Structured clay loam	Brown dermosol	Dermosols (structured clay/clay loam surface)
Eton	12,400	Sandy clay loam (alluvial)	Sodosol	Sodosols
Mari	118,950	Heavy clay loam	Brown chromosol	Dermosols (sealing loamy surface), includes non-sodic: chromosols, kurosols, kandosols and calcarosols
Stew	4,850	Seasonally wet soil	Hydrosol	Hydrosols (sandy surfaced)
Thor	6,300	Well drained red clay	Ferrosol	Ferrosols
Todd	8,600	Yellow sandy loam	Yellow kandosol	Dermosols (sandy surface), includes non-sodic chromosols, kurosols, kandosols and calcarosols

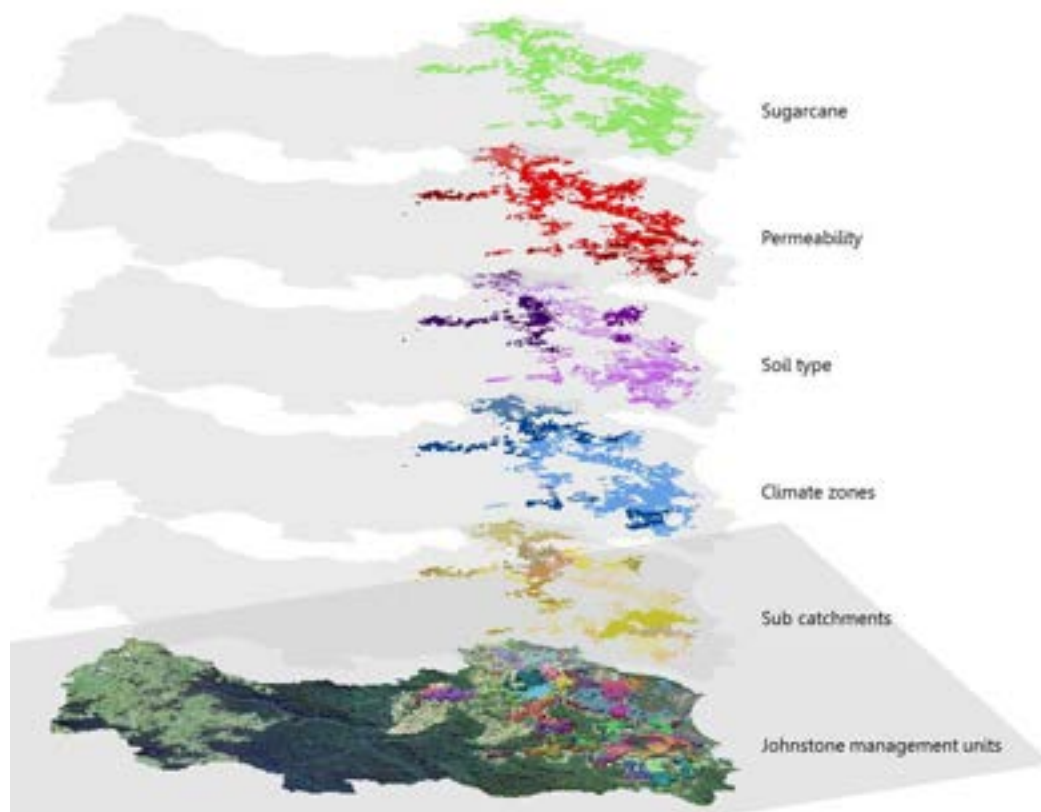


Figure 4.3: Individual ‘management units’ for nitrogen credit generation in Project 3.1.6, comprising the sugarcane production area in unique combinations of climate zone, soil type and soil permeability within a single P2R hydrological sub-catchment.

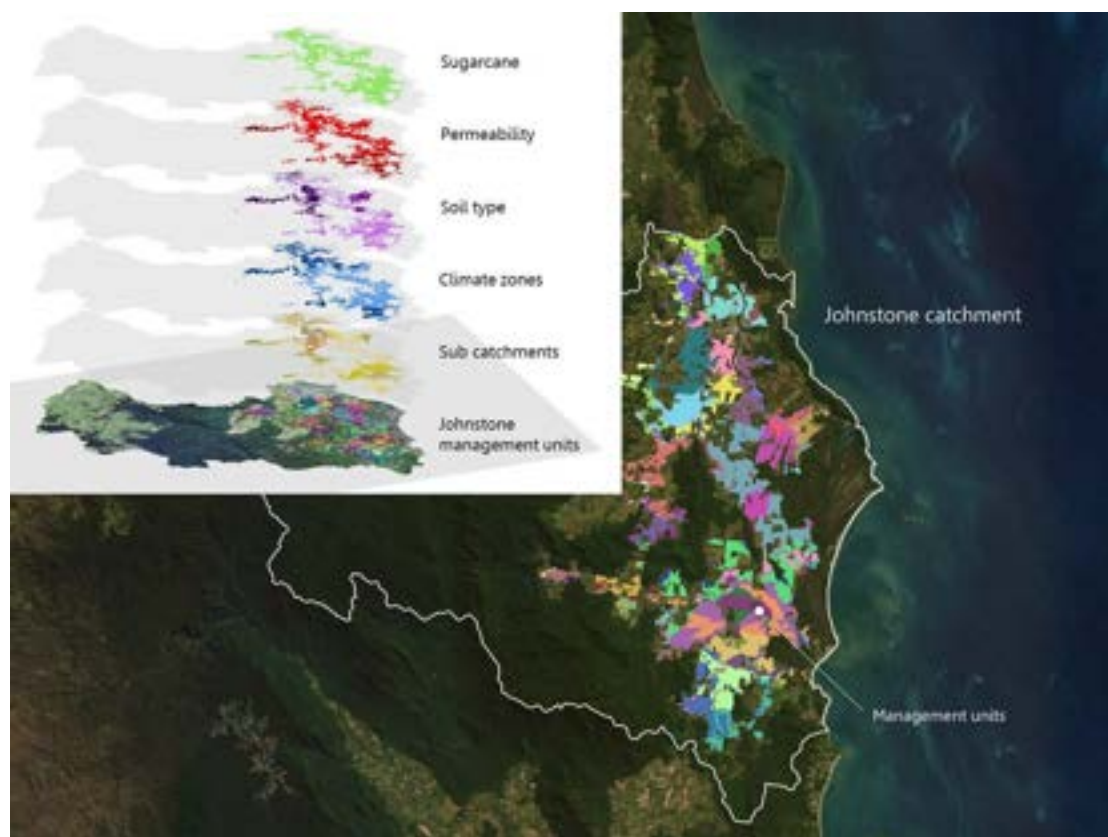


Figure 4.4: Individual management units in the Johnstone catchment.

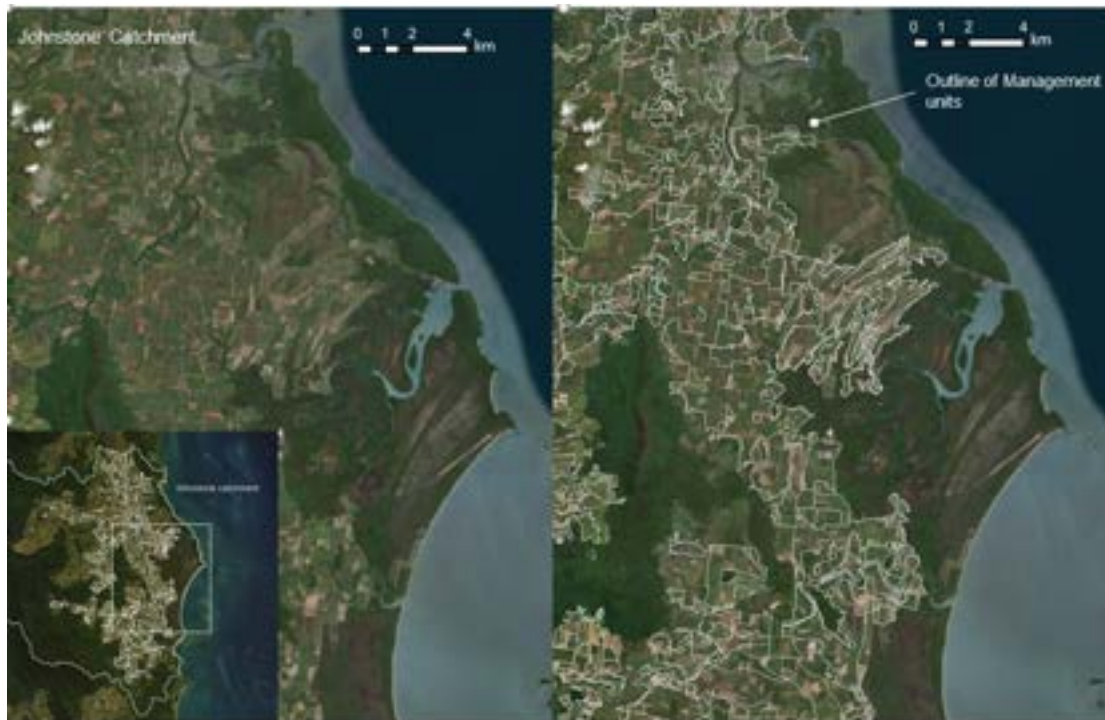


Figure 4.5: Expanded view of individual management units in the Johnstone catchment, illustrating that management units can be of approximately paddock-scale (depending on the spatial variation in soil class and soil permeability).

4.3 Fertiliser management practices for sugarcane in P2R predictions

(This subsection draws heavily on documentation provided by Dr Melanie Shaw at Qld DNRM regarding implementation of APSIM sugarcane modelling in the P2R framework for the 2016 Reef Report Card)

The DIN loss and cane yield modelling provided by P2R used the terminology of the Reef Report Card 2016 for categorising management practices. Simulations were produced under management scenarios for soil, fertiliser, and pesticide that aligned with the 4-step Water Quality Risk Framework in which each type of practice was classified as high (D), moderate (C), moderate-low (B) or lowest (A) risk. Emission reductions would be expected as practices improved in the sequence $D > C > B > A$. A second level of categorisation defined whether a given level of practice was implemented fully (f) or partially (p). Hence Bf denotes full implementation of practice level B, whereas Bp denotes only partial implementation.

With regard to cane production, nitrogen credits are the main focus of Project 3.1.6, so interest centres on the predicted reductions in DIN losses and changes in cane yield that follow from improvements in fertiliser management practice along a 6-step scale (Df to Af). Soil

management practice was fixed at practice level Bp throughout³⁴. Bp was the most common level of soil management practice adopted in the Wet Tropics at the time the P2R simulations were produced (*Pers. comm.* Melanie Shaw). Pesticide practices were changed in step with fertiliser practice. The combined suite of management practices applied in a P2R APSIM simulation scenario is described by a code comprising three sets of two letters e.g. AfBfCf where:

- Soil management is denoted by the first two letters (Af in the example above)
- Fertiliser management is denoted by the middle two letters (Bf)
- Pesticide management is denoted by the last two letters (Cf)

The management practices included in the P2R simulations for Project 3.1.6 are BpDfDf, BpCpCp, BpCfCf, BpBpBp, BpBfBf, BpAfAf, i.e. soil management practice is held fixed at Bp whilst fertiliser and pesticide management practice step together through practice levels Df to Af.

Six fertiliser management practice levels were modelled in the P2R APSIM simulations (Table 4.2). Fertiliser application rate and timing of application are the only factors that differ between practices. Fertiliser applications are applied post-harvest, so the timing of applications within the year follows the harvesting schedule for the different crop stages (Table 4.3). Application timing is important because DIN losses will be higher if a rain event (in that particular year's weather) occurs shortly after fertiliser application.

The fertiliser application rates modelled (Table 4.2) are consistent with traditional practice (for practices Df and Cp), recommended application rates under the Six Easy Steps fertiliser management plan (Schroeder et al. 2010) derived from District Yield Potential (DYP), with allowances for soil organic carbon, legume fallow and application of mill by-products (for practices Cf and Bp), application rates similarly derived from historical 'block yield potential', (for practice Bf), or rates that account for 'within-block' variation in 'historical yield potential' (for practice Af). Application rates in Table 4.2 map to the terminology of the Sugarcane Water Quality Risk Framework 2017-2022 (The Australian and Queensland Governments, n.d.) as follows: Df and Cp are regarded as Superseded ('high risk'), Cf and Bp are regarded as Minimum Standard ('moderate risk'), Bf is regarded as Best Practice ('moderate-low risk'), and Af is regarded as Innovative ('lowest risk'). Minimum Standard ('moderate risk') nutrient management practice is understood to comply with the 2019 Revisions to the Reef Regulations³⁵ (State of Queensland 2019a). However, ***for the future-focused purposes of Project 3.1.6, fertiliser application rates at practice level Bf (Best Practice 'moderate-low risk') are assumed to be required to achieve the Reef 2050 WQIP's basin-specific water quality objectives (State of Queensland, 2018).***

³⁴ Soil management practice Bp in the 2016 Water Quality Risk Framework comprised: 9 tillage operations including planting the cane and cowpea legume crop in the fallow; no tillage through the ratoons; green cane trash blanketing; cowpea legume in fallow.

³⁵ The Queensland government have made a commitment to "not substantially change the minimum practice agricultural standards for five years (from commencement of [the] Amendment Regulations)" (Queensland Government, 2019; p.4, paragraph 4).

Each management unit in Project 3.1.6 comprises a single combination of soil type and soil permeability, within a particular SILO climate zone and P2R sub-catchment. Under practice level Bf, specific fertiliser application rates for plant cane and ratoon cane would thus apply across the entire management unit under the Prescribed Methodology (assuming consistent organic carbon content and mill mud application history across the management unit). The only practice improvement from this position would be to adjust fertiliser application rate *within* the management unit, based on differences in recorded crop yield³⁶ (i.e. adopt practice Af). According to the APSIM P2R documentation supplied, adoption of practice Af would be expected to reduce fertiliser application rate to 5% below that of practice Bf.

Table 4.2: Nitrogen fertiliser application rates modelled for the Wet Tropics in APSIM P2R simulations for Project 3.1.6

<i>Practice</i>	<i>Description</i>	<i>Plant cane application rate (kgN/ha)</i>	Ratoon application rate (kgN/ha)
Df	Rule of thumb	160	180
Cp	General recommendation	140	160
Cf	Six Easy Steps <i>derived from District Yield Potential</i> with adjustment for soil organic carbon	110 - 140	130 - 150
Bp	As for Cf, but with further adjustment to plant cane rate to account for application of mill mud and/or legume fallow	Follow Six Easy Steps: Fertiliser application to ratoon crops derived using a nitrogen utilisation index after Keating et al.(1997) (1.4 kgN/ha per tonne/ha maximum annual cane yield up to 100 tonnes/ha, plus 1 kgN/ha per tonne/ha maximum cane yield thereafter), adjusting for soil organic carbon, mill mud and legume fallow. Minimum of 30 kgN/ha applied to plant cane. Maximum annual cane yield determined from <i>District Yield Potential</i> .	
Bf	Six Easy Steps <i>derived from block-specific yield potential</i> , with adjustments for soil organic carbon, mill mud and legume fallow ³⁷	Application to ratoon crops derived using a nitrogen utilisation index after Keating et al.(1997) (1.4 kgN/ha per tonne/ha maximum annual cane yield up to 100 tonnes/ha, plus 1 kgN/ha per tonne/ha maximum cane yield thereafter), adjusting for soil organic carbon, mill mud and legume fallow. Maximum annual cane	

³⁶ Tracking differences in crop yield within cropping blocks would require all harvesters and fertiliser spreaders to be fitted with GPS loggers.

³⁷ Management unit-specific fertiliser application rates for practice Bf were calculated from the nitrogen utilisation index of Keating et al.(1997) as set out in the Prescribed Method for Sugar Cane Production (Qld Govt 2019), using the maximum modelled cane fresh weight yield reported by APSIMP2R for each management unit over the period 1987-2013 and assuming a medium-low level for soil organic carbon (drawing from Schroeder et al. (2007) and Wood et al. (2003)). For management units in the Herbert catchment, for which year-specific yield data were not provided, mean fertiliser application rates for practice Bf were assigned based on soil type, drawing on results from other catchments. For the eton soil type (which only occurs in the Herbert catchment in this dataset) practice Bf application rate was assumed to be the average of the practice Bf application rates for soil types coom and mari, based on Wood et al. (2003).

	yield determined from <i>block-specific APSIM P2R simulations</i> across years 1987-2013.	
Af	As for Bf, but accounting for variation in cane yield <i>within</i> blocks	Assumed to be 5% lower than Bf

P2R APSIM modelling of DIN losses and cane yield proceeds on a daily timestep, using daily weather for each of the 101 SILO climate zones across Wet Tropics cane land for the years 1987 – 2013. The cane cropping cycle on which the modelling was based is shown in Table 4.3. This cropping cycle embodies common features of cane production in the Wet Tropics. Modelling uses characteristics of cane cultivar q117, grown on a plant cane plus four ratoons production cycle, followed by a cowpea legume fallow.

Table 4.3 Cane cropping cycle for the Wet Tropics used in APSIM P2R modelling for Project 3.1.6

	<i>Plant</i>	<i>1st ratoon</i>	<i>2nd ratoon</i>	<i>3rd ratoon</i>	<i>4th ratoon</i>	Fallow
Crop start	15 th July	15 th Aug	15 th July	15 th Aug	15 th Sept	15 th Nov
Harvest	14 th Aug	14 th July	14 th Aug	14 th Sept	14 th Nov	14 th Jul

In each management unit all six crop stages were simulated individually for each year 1987-2013. Results were then combined across crop stages (fallow, plant, 1st to 4th ratoon) to produce overall average results across all years and all crop stages for DIN loss and cane yield. P2R APSIM predictions of **overall average DIN losses and cane yields (1987-2013)** were provided for all Wet Tropics catchments. Separate predictions detailing DIN loss and crop yields **averaged across the six crop stages for each individual year 1987-2013** were provided for all Wet Tropics catchments except the Herbert.

4.4 Nitrogen losses from sugarcane production in the Wet Tropics

DIN losses from sugarcane production predicted by P2R APSIM will vary with fertiliser management practice (Table 4.2), soil type (Table 4.1), soil permeability (low, medium, high), crop stage and thus fertiliser application rates and timings (Table 4.3), and site-specific weather. The impacts of crop stage, fertiliser application timing and weather are averaged across APSIM P2R simulation years 1987-2013 to produce **average (1987-2013) DIN losses** for individual management units. APSIM P2R **year-specific average DIN losses** (averaged across crop stages) for individual management units for each year 1987-2013 allow the effects of year to year weather variations to be observed. Year-specific DIN losses from P2R APSIM were not provided for the Herbert catchment, so figures illustrating the year-specific outcomes do not include management units in the Herbert, or the eton soil type that (in this dataset) is present only in the Herbert catchment.

4.4.1 Average DIN losses across 1987-2013

As explained in Section 4.2.1, DIN losses from cane production comprise losses to surface runoff and losses to drainage. Figure 4.6 shows P2R APSIM predicted average annual DIN losses (1987-2013) to runoff and drainage for 563 cane land management units in the Wet Tropics, grouped by fertiliser management practices, soil type and soil permeability. Figure 4.6 shows that DIN losses from the paddock to drainage are typically at least an order of magnitude

higher than losses to runoff. Average DIN transport to End-of-Catchment via the surface water runoff pathway is 99.6% for the P2R sub-catchments in the Wet Tropics dataset³⁸. As explained in Section 4.2.1, in line with previous published literature, Project 3.1.6 assumes that 60% of the DIN that leaves the paddock via the groundwater drainage pathway reaches End-of-Catchment.

Figure 4.6 shows that average 1987-2013 DIN losses through both pathways reduce as fertiliser practice improves (Df to Af). The step reductions in DIN loss to drainage following practice improvement are considerably larger than those in DIN loss to runoff. Sizeable reductions are delivered by improving fertiliser practice from Df to Cp or Cf, from Cp or Cf to Bp, and from Bp to Bf. This illustrates that after having switched from high water quality risk practice, further sizeable reductions in average 1987-2013 DIN loss to drainage are obtained by adjusting DYP-derived fertiliser applications to plant cane to account for nitrogen inputs from mill mud and/or legume fallow (i.e. switching from practice Cf to Bp), and by then adjusting fertiliser rates to reflect 'block-specific yield potential' (switching from practice Bp to Bf). However, based on the P2R APSIM 2016 modelling results provided to Project 3.1.6, only a small further reduction in average 1987-2013 DIN loss would be achieved by improving fertiliser practice from Bf to Af, i.e. by further adjusting fertiliser rates to reflect within-block variation in yield potential.

Across management units, the spread of average 1987-2013 DIN loss about its median differs considerably for different soil types. Particularly wide variation in DIN losses to drainage is evident for coom and thor soils³⁹. As expected, DIN losses to drainage are lower in low permeability soils, and higher in high permeability soils, with the opposite progression evident for DIN losses to runoff⁴⁰.

³⁸ Calculated from P2R sub-catchment-specific surface water DIN transport ratios to End-of-Catchment supplied by Queensland DNRM.

³⁹ In Figure 4.6 these variations are confounded by changes in fertiliser practice and soil permeability.

⁴⁰ The variations in DIN loss with permeability in Figure 4.6 are confounded by changes in fertiliser practice and soil type.

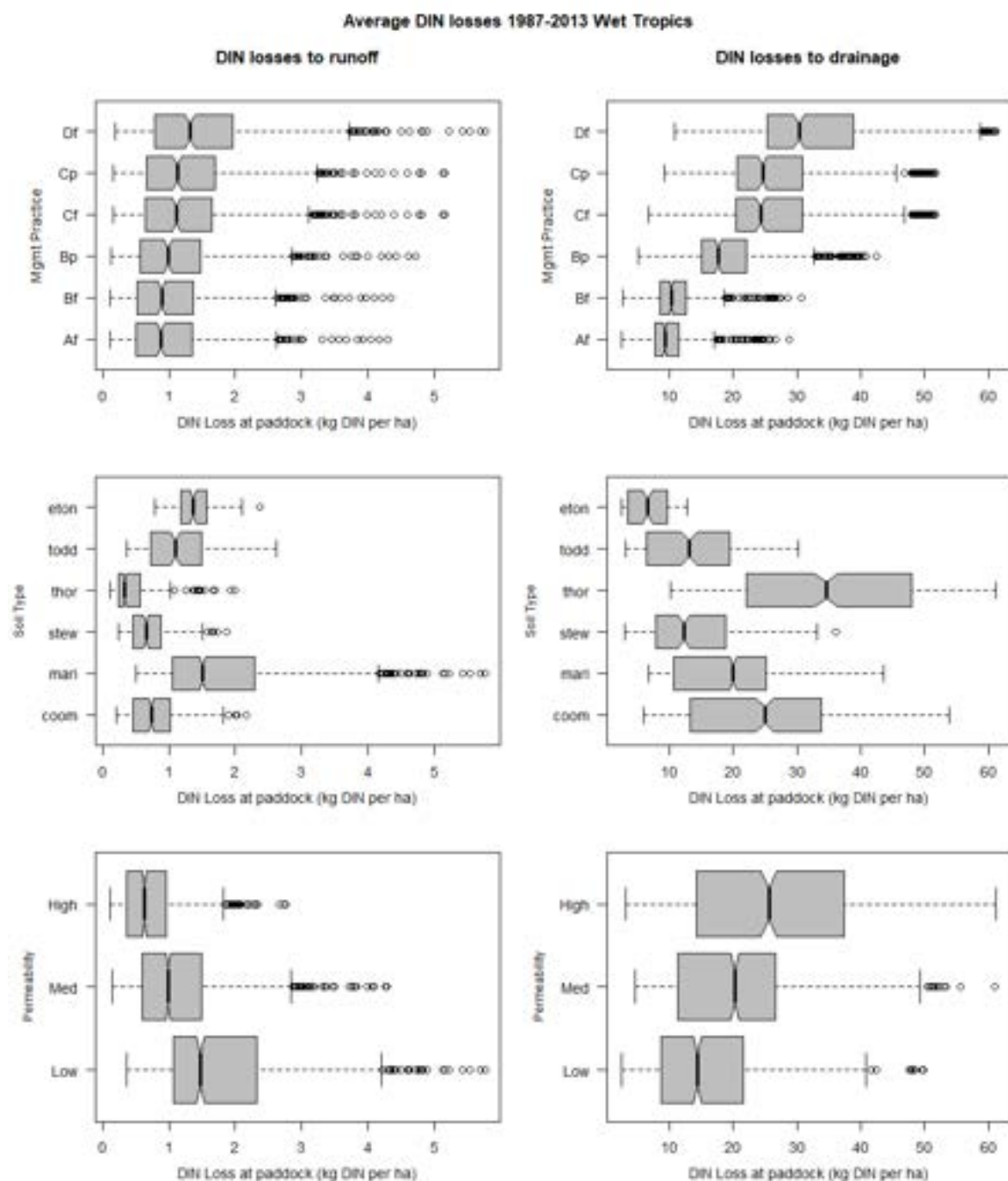


Figure 4.6: Average annual DIN losses (1987-2013) to runoff and drainage predicted by P2R APSIM for 563 cane land management units in the Wet Tropics, grouped by fertiliser management practices, soil type and soil permeability.

Figure 4.7 shows average total annual DIN loss⁴¹(1987-2013) for fertiliser management practices Df, Cf, Bp and Bf for cane land management units in the Wet Tropics, grouped by soil type. Average 1987-2013 DIN losses reduce as practice improves from Df to Bf. Considerable differences between soil types are still evident in the spread of average DIN losses around their mean under a given level of management practice (Figure 4.7). Spread is

⁴¹ Total DIN loss from the paddock is the sum of DIN losses to runoff and drainage at the paddock.

high for soil types coom, stew and thor, lower for mari and todd, and extremely low for eton. Variation in average DIN loss at a given practice level will arise because geographically distant management units on a particular soil type will have experienced different weather over the 1987-2013 modelling period, and also because soil permeability may differ between sites. All eton soils in the dataset are of low permeability and are located within approximately a 25km radius in the Herbert catchment (Figure 4.10 and Table 4.4). Todd and coom soils only occur in the dataset at medium and high permeability; all other soils occur at all three permeability levels. Considerable areas of mari soil occur in all catchments, generally at low and medium permeability. It is therefore notable that mari soils show a relatively modest variation in average DIN loss at all levels of fertiliser practice.

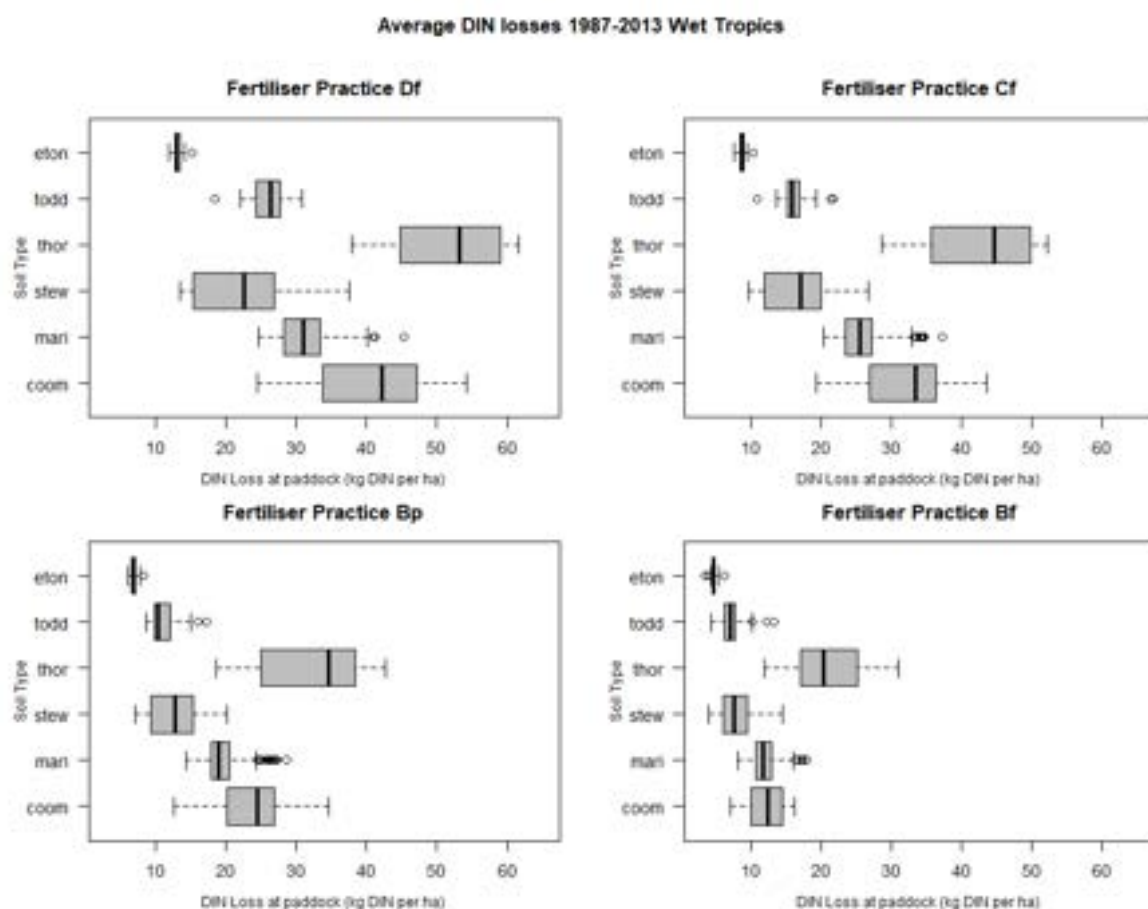


Figure 4.7: Average total annual DIN loss (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by soil type.

Average total annual DIN loss (1987-2013) for fertiliser management practices Df, Cf, Bp and Bf, with management units grouped by catchment, is shown in Figure 4.8. DIN loss characteristics of catchments are as expected, given the soil types present (Table 4.5). Catchments that contain a wide variety of soil types – and particularly catchments such as the Herbert that contain a mix of high (e.g. thor) and low (e.g. eton and stew) DIN loss soil types – show very wide variation in average DIN loss for a given management practice across management units. Figure 4.8 suggests that the effectiveness of practice improvement as a mechanism for delivering reductions in DIN losses will likely vary considerably within and between catchments.

Table 4.4: Main cane land soil types (by area) in Wet Tropics catchments

<i>Catchment</i>	<i>Main soil types</i>	Soil map
Daintree	Mari low & med permeability Thor med permeability	Figure 4.1
Mossman	Mari low & med permeability Thor med permeability	Figure 4.2
Barron (inland)	Thor med & high permeability Todd med & high permeability	Figure 4.3
Barron (coastal)	Coom high permeability Mari med & high permeability	Figure 4.4
Mulgrave-Russell	Coom high permeability Mari low & med permeability	Figure 4.5 Figure 4.6
Johnstone	Coom med & high permeability Mari low & med permeability Thor med permeability Todd med & high permeability	Figure 4.7
Tully	Coom med permeability Mari low & med permeability	Figure 4.8
Murray	Mari low & med permeability	Figure 4.9
Herbert	Coom med permeability Eton low permeability Mari low, med & high permeability Stew low permeability Thor high permeability	Figure 4.10

Average DIN losses 1987-2013 Wet Tropics

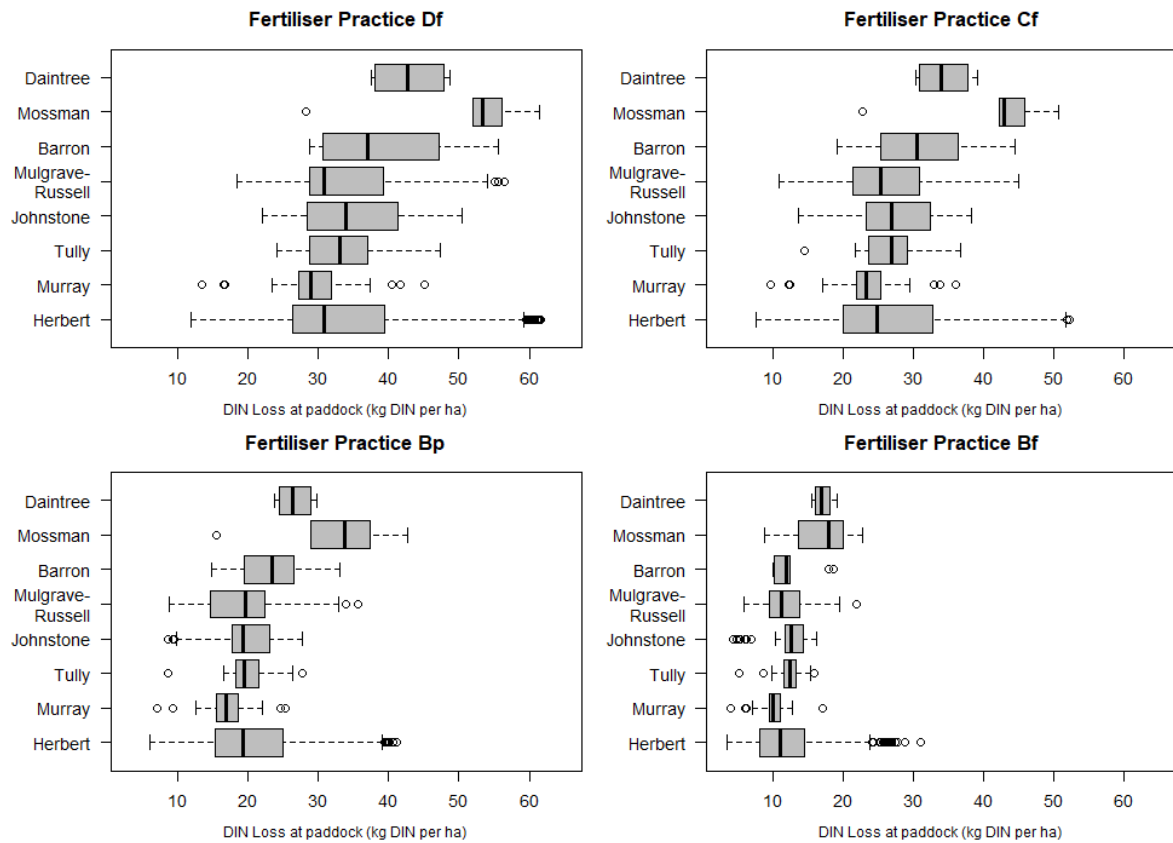


Figure 4.8: Average total annual DIN loss (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by catchment.

4.4.2 Year-specific DIN losses 1987-2013

APSIM P2R **year-specific** total DIN losses for individual cane land management units in the Wet Tropics, grouped by soil type are shown in Figure 4.9. The rank ordering of median year-specific DIN loss across soil types under different fertiliser practices matches that for average 1987-2013 DIN loss (Figure 4.7).

Figure 4.9 shows high DIN loss outlier ‘tails’ (the points plotted as individual circles extending above the upper whisker of the box plots) for all soil types, under all plotted management practices, Df through to Bf. This indicates that, under weather conditions that occurred between 1987-2013, extremely high DIN losses could result, even under ‘moderate-low water quality risk’ management practices (e.g. Bf). Management units on coom and mari soils – the soil types that comprise the largest areas of cane land in the Wet Tropics catchments (Table 4.2) – can produce at-paddock DIN losses of 90kg DIN per hectare at practice Bf under unfavourable weather conditions. Figure 4.9 also shows that practice improvement from Df through to Bp is less successful in limiting extreme losses than it is in reducing median losses; however, practice improvement from Bp to Bf is more successful in this regard. Given the links between high DIN concentrations in the Reef Lagoon, heightened susceptibility to coral bleaching, and increased risk of outbreaks of Crown-of-Thorns starfish infestations (Brodie et al. 2017), Figure 4.9 suggests that whole-of-farm nitrogen budgets that are consistent with fertiliser management practice Bf could be important for protecting Reef health.

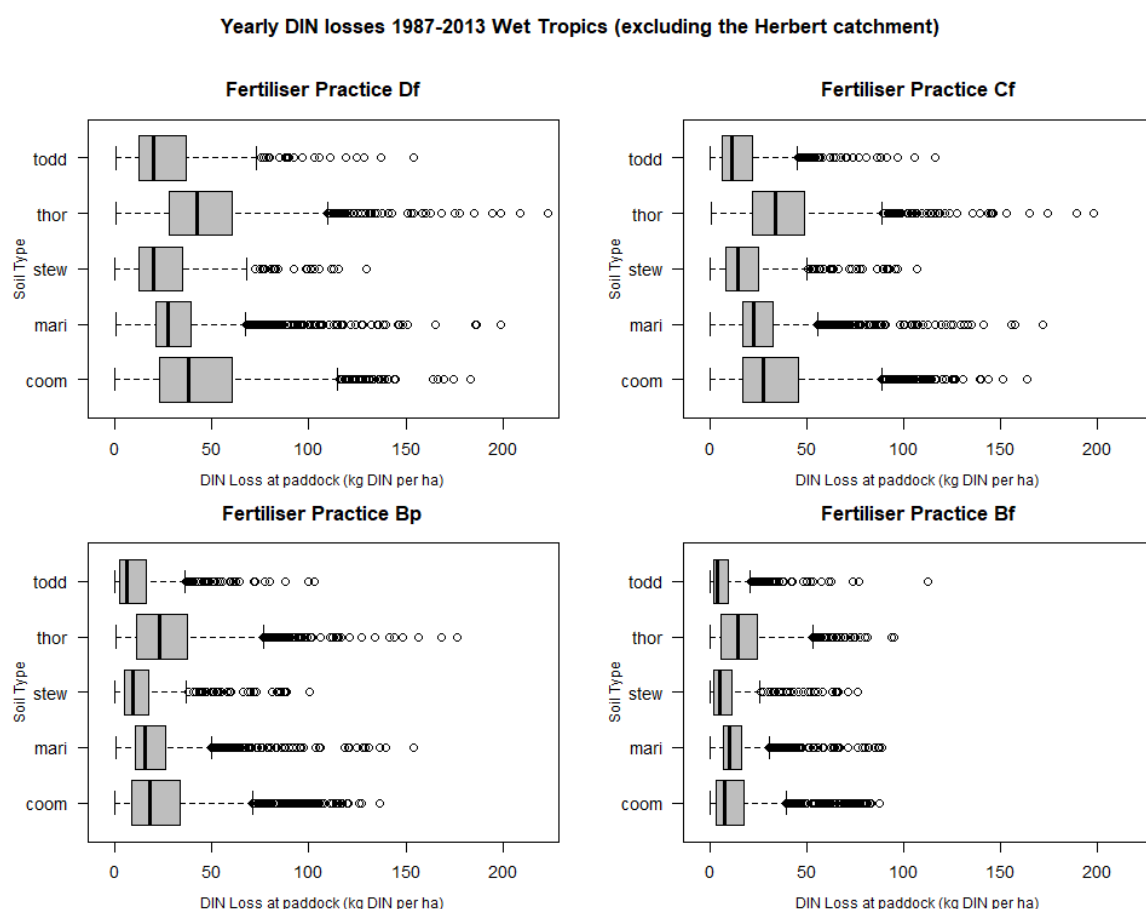


Figure 4.9: Yearly total annual DIN loss (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by soil type.

4.5 Cane yields from sugarcane production in the Wet Tropics

APSIM P2R simulations for the years 1987-2013 produced **average (1987-2013) cane yields** and **year-specific cane yields (1987-2013)** for individual cane land management units across the Wet Tropics under different fertiliser management practices. Average cane yields were provided for all Wet Tropics catchments; year-specific cane yields were provided for all catchments except the Herbert. Changes in cane yield under changing management practice could result in opportunity costs to the farmer if reductions in fertiliser expenditure are not sufficient to offset any reduction in cane revenues.

4.5.1 Average cane yields across management units 1987-2013

Figure 4.10 shows P2R APSIM predicted average annual cane yields (1987-2013) from individual cane land management units across the Wet Tropics, grouped by soil type. Average cane yields and spreads around median cane yield under a given fertiliser practice differ between soil types. Predicted average 1987-2013 cane yields and spreads around median cane yield change only slightly as fertiliser practice improves from Df through to Bp. This suggests that, under P2R APSIM yield modelling, average 1987-2013 opportunity costs from practice improvement to practice Bp are likely to be very low, and often negative, given the

substantial associated reductions in fertiliser expenditure. This matches findings from van Grieken et al. (2010, Section 2.2, Table 8 p.6 & Table 11 p.7).

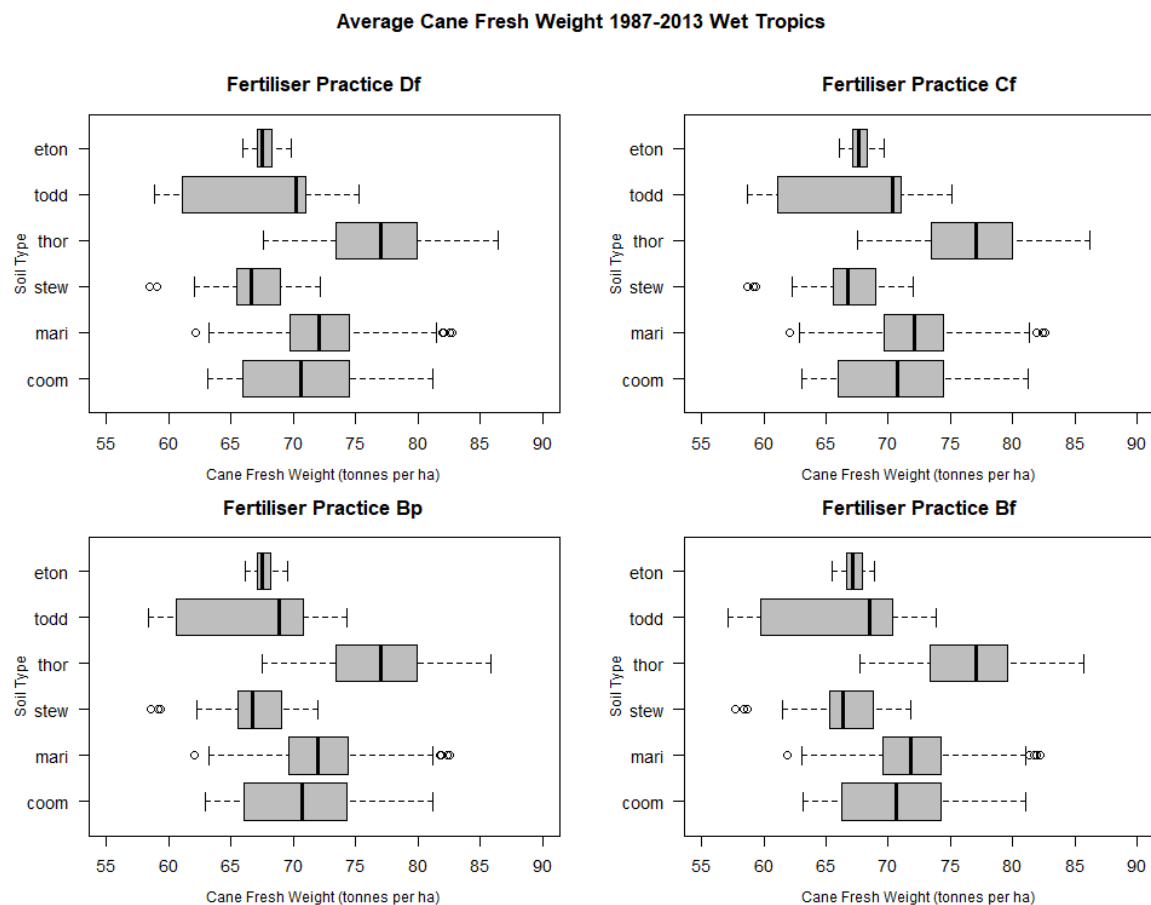


Figure 4.10: Average annual cane yields (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by soil type.

4.5.2 Year-specific cane yields across management units 1987-2013

Figure 4.11 shows P2R APSIM-predicted year-specific cane yields (1987-2013) from individual cane land management units across the Wet Tropics, grouped by soil type. The rank ordering of median year-specific cane yield is the same as that of average cane yields (Figure 4.10), although, as was the case for the DIN loss data, the spread of year-specific cane yields is much higher than that of average cane yields.

Figure 4.11 shows low cane yield outlier ‘tails’ for all soil types, under all plotted management practices, Df through to Bf. This indicates that, under weather conditions that occurred between 1987-2013, extremely low cane yields could result, even under high fertiliser application rates. Management units on mari soils – the soil type that comprises the largest area of cane land in the Wet Tropics catchments (Table 4.2) – are predicted to produce cane yields below 30 tonnes cane fresh weight per hectare under adverse weather conditions for all plotted fertiliser management practices. The results in Figure 4.11, are consistent with explanations provided by Thorburn et al. (2018, 2011) regarding the ability of year-to-year variations in weather to disrupt cane yield. Thorburn et al. (2018; Figure 2, p.213) show that large variations in nitrogen

loss to the environment⁴² under adverse weather conditions can leave insufficient nitrogen available for the crop – even under high nitrogen application rates – and low cane yields can thus result. Adopting Thorburn et al’s explanation regarding the influence of variation in nitrogen losses to the environment under year-to-year weather fluctuations in crop yield, the lengthy ‘lower tails’ in cane yields in Figure 4.11 and the lengthy upper tails in DIN losses in Figure 4.9 may be consequences of the same underlying phenomenon⁴³.

Figure 4.11 shows that higher water quality risk fertiliser management practices do not provide a substantive buffer against low cane yields, should adverse weather conditions arise, nor do they appear to provide an opportunity to realise substantially higher cane yields when weather conditions are more opportune. Differences in yield characteristics between soils are evident. For example, mari soil appears more prone to very low yield under adverse weather conditions than does coom soil. This suggests that the return from nitrogen credits sales that would be required to persuade farmers to reduce fertiliser application rates could differ between soil types. Farmers on soils that are prone to produce low yields under adverse conditions might be expected to be more risk averse. Such farmers may therefore require evidence of higher returns from credit sales before committing to practice improvements.

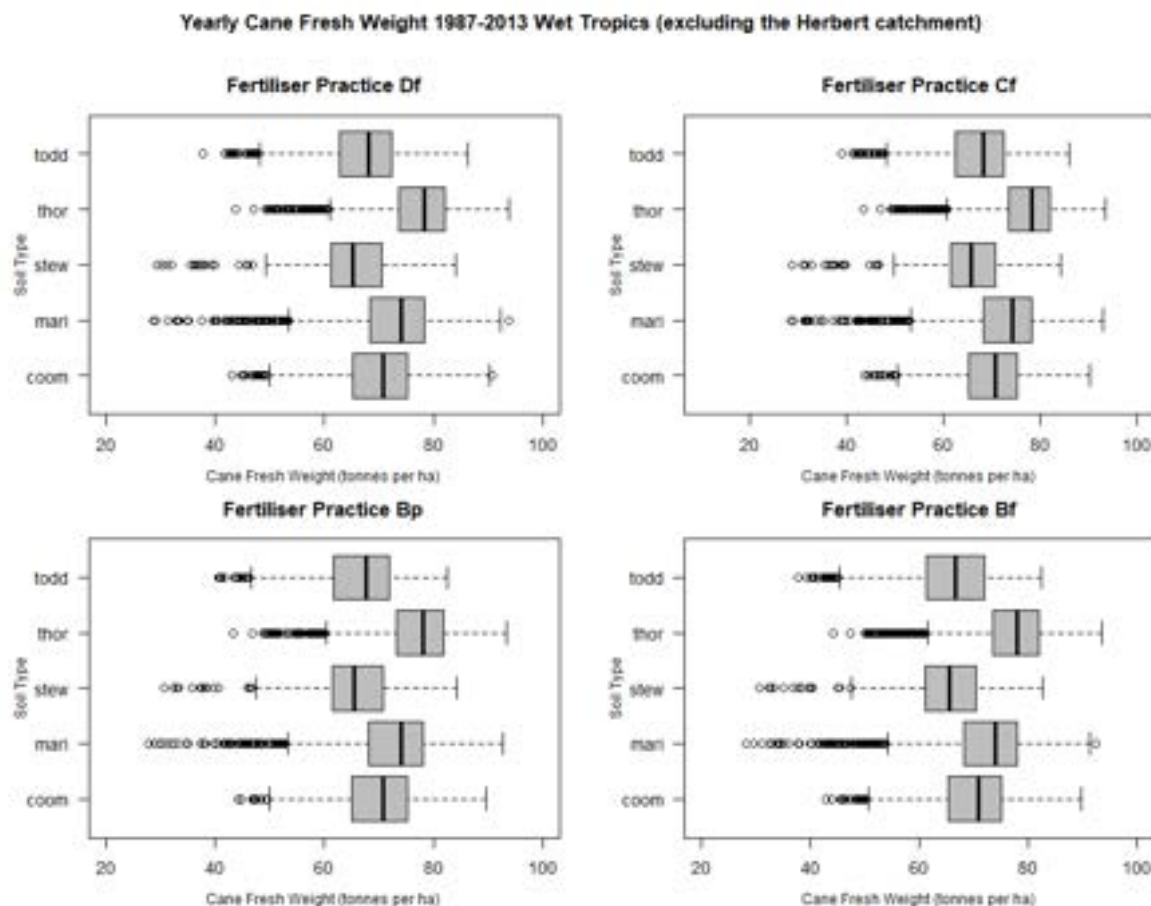


Figure 4.11: Yearly annual cane yields (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by soil type.

⁴² via denitrification, losses to drainage and losses via runoff (Thorburn et al., 2018; p215-216)

⁴³ Additional nitrogen losses via denitrification in water-logged soils (Figure 4.2) would be expected on low permeability soils in wet years.

4.6 Summary

To maintain the ecological health of the GBR, the Queensland government's Environmental Protection (Water & Wetlands) Policy 2019 sets End-of-Basin reduction targets⁴⁴ to be achieved by 2025 for anthropogenic DIN and fine sediment loads from catchments draining into the GBR Lagoon. These reduction targets are challenging, particularly for DIN load reduction from Wet Tropics cane catchments and fine sediment load reductions from the Burdekin and Fitzroy catchments. Challenging DIN and fine sediment load reductions have been set for the Herbert and Johnstone catchments in the Wet Tropics. Project 3.1.6 explores the potential of water quality credit trading as a mechanism for facilitating future economic expansion once the End-of-Catchment load reductions have been achieved. The Project specifically explores potential water quality markets for:

- DIN in the Russell-Mulgrave, Johnstone, Tully, Murray and Herbert catchments in the Wet Tropics
- Fine sediment in the Bowen and Bogie sub-catchments within the Burdekin River catchment
- DIN and fine sediment in combination in the Johnstone catchment in the Wet Tropics and the Bowen sub-catchment in the Burdekin region

To quantify the potential for supply of nitrogen credits from changes in fertiliser management practice, Project 3.1.6 uses Paddock to Reef (P2R) modelling results to predict the changes in DIN losses and cane yield that would result from improvements in fertiliser management practice in sugarcane production. Drawing on P2R APSIM simulations, reductions in DIN losses and changes in cane yield are produced for six levels of fertiliser management practice from high to low water quality risk across 563 individual cane land management units in Wet Tropics catchments. Each management unit comprises a unique combination of one of six soil types, one of three permeability levels, and one of 101 climate zones within a specific P2R sub-catchment. For individual management units, P2R modelling produces average DIN loss and cane yield results over the period 1987-2013, and year-specific DIN loss and cane yield results for each year in the period 1987-2013. Under any given level of fertiliser management practice, considerable variations in average DIN loss and average cane yield are evident between management units over the period 1987-2013, and very substantial variations arise in year-specific results. Consistent differences are evident between soil types in median levels of DIN loss and cane yield under given levels of fertiliser management practice. Differences in DIN loss and cane yield between catchments under given levels of management practice follow soil composition within catchments.

Management unit-specific P2R APSIM results underpin multiple aspects of the catchment-specific supply curves for nitrogen credits from fertiliser practice change in cane that will be produced in Chapter 5, and the exploration of the importance of 'delivery ratio' within the Queensland Government's Point Source Water Quality Offsets Policy (State of Queensland 2019f) in Chapter 8.

⁴⁴ Hereafter, in line with standard usage, these targets will be referred to as End-of-Catchment targets. The usage of 'basins' in the Water Quality Objectives document State of Queensland (2019a) matches the definitions of GeoScience Australia; however, practitioners will typically refer to e.g. 'the Tully catchment' rather than 'the Tully basin'. We adopt standard practitioner terminology in the remainder of this Report.

Appendix 4A: Management Units in the Mulgrave-Russell, Johnstone, Tully, Murray and Herbert catchments

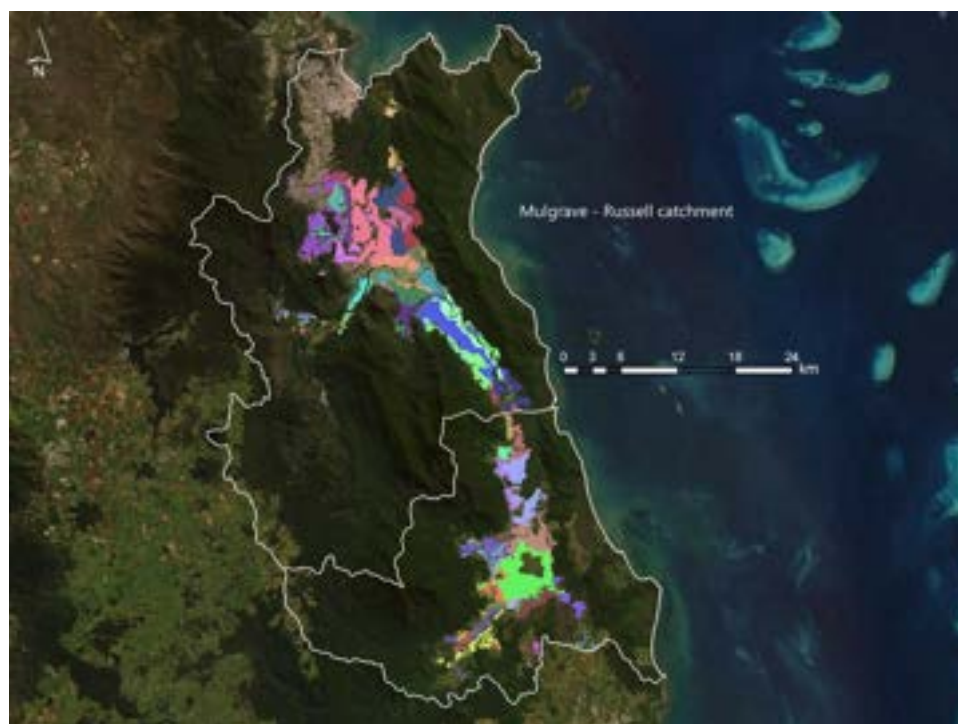


Figure 4.12: Individual 'management units' in the Mulgrave-Russell catchment

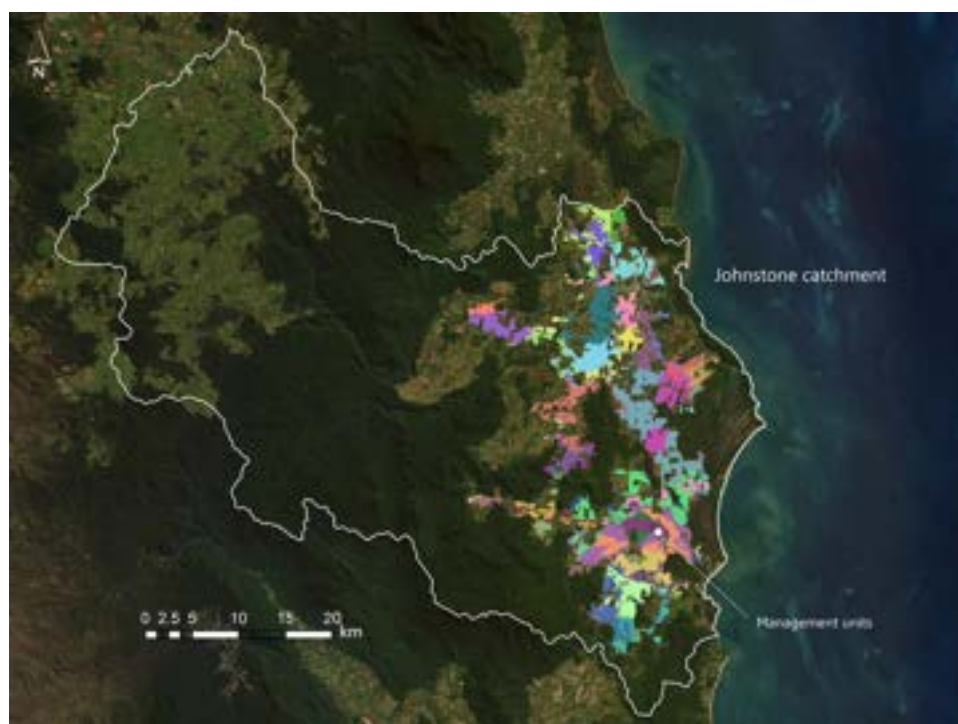


Figure 4.13: Individual 'management units' in the Johnstone catchment

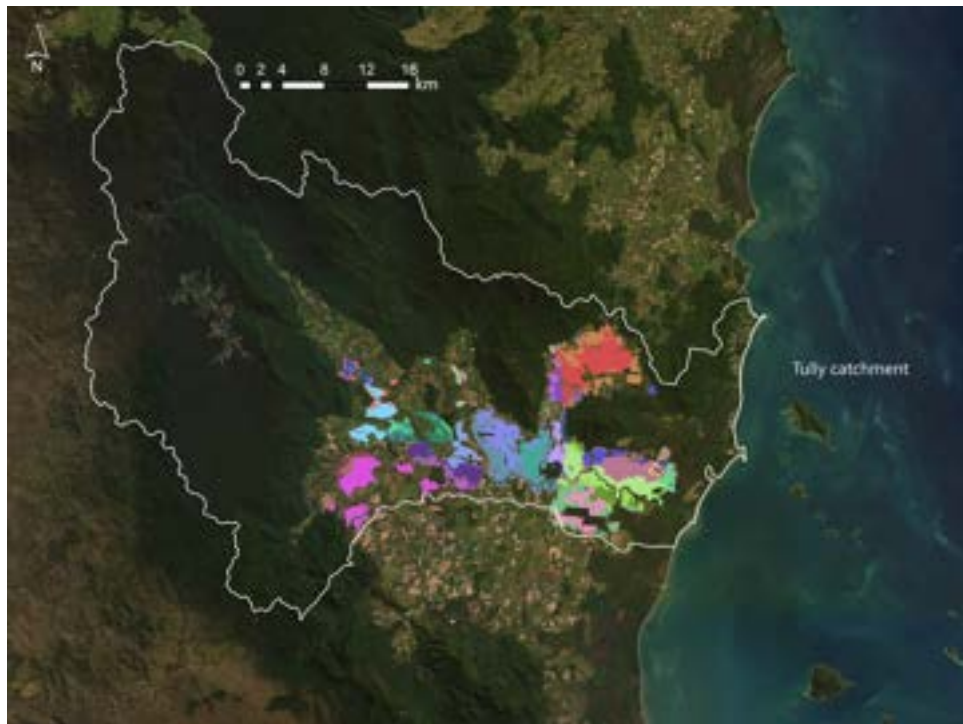


Figure 4.14: Individual 'management units' in the Tully catchment

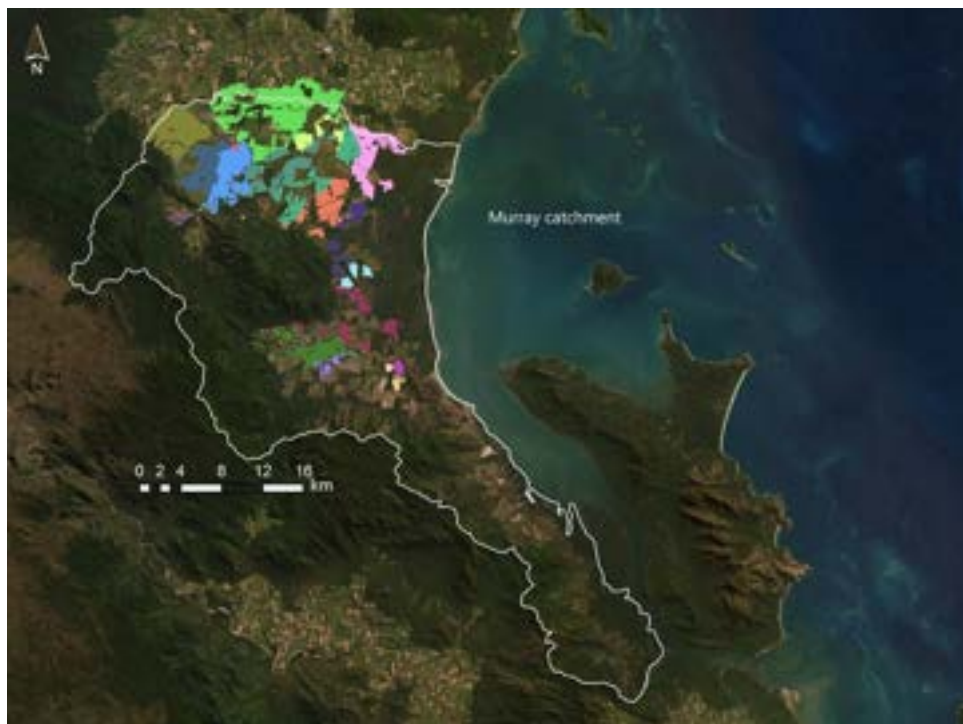


Figure 4.15: Individual 'management units' in the Murray catchment

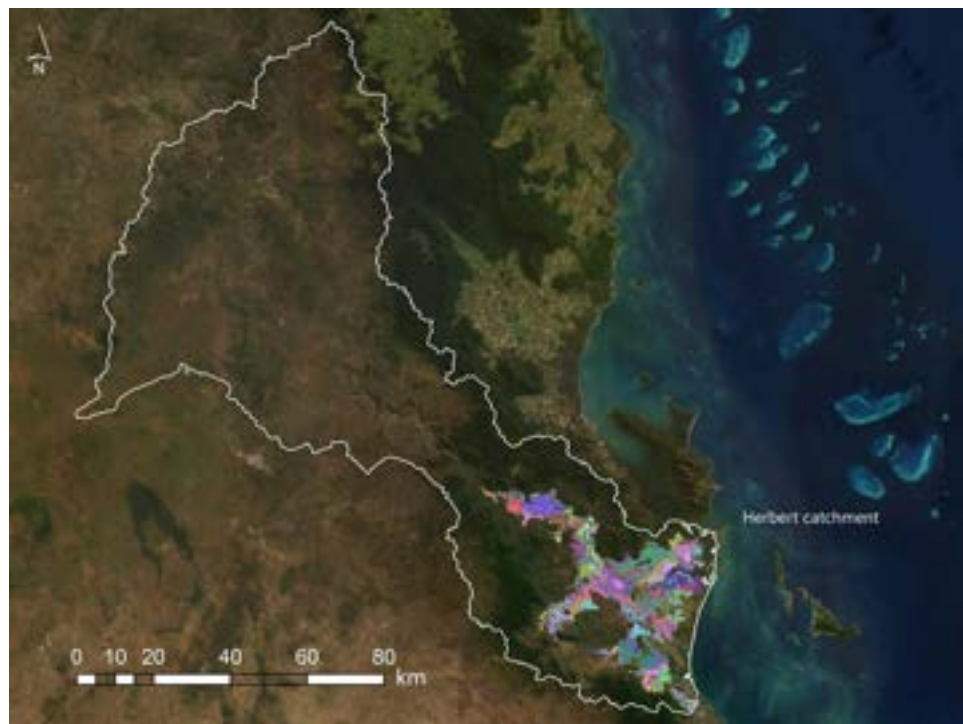


Figure 4.16: Individual 'management units' in the Herbert catchment

Appendix 4B: Soil types in the Daintree, Mossman, Barron, Russell-Mulgrave, Johnstone, Tully, Murray and Herbert catchments

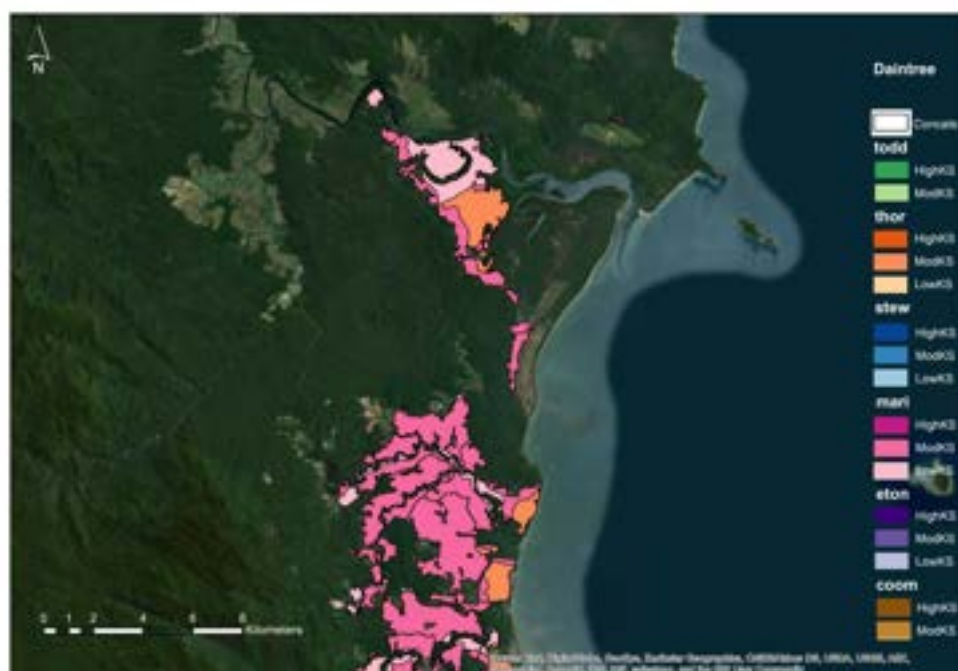
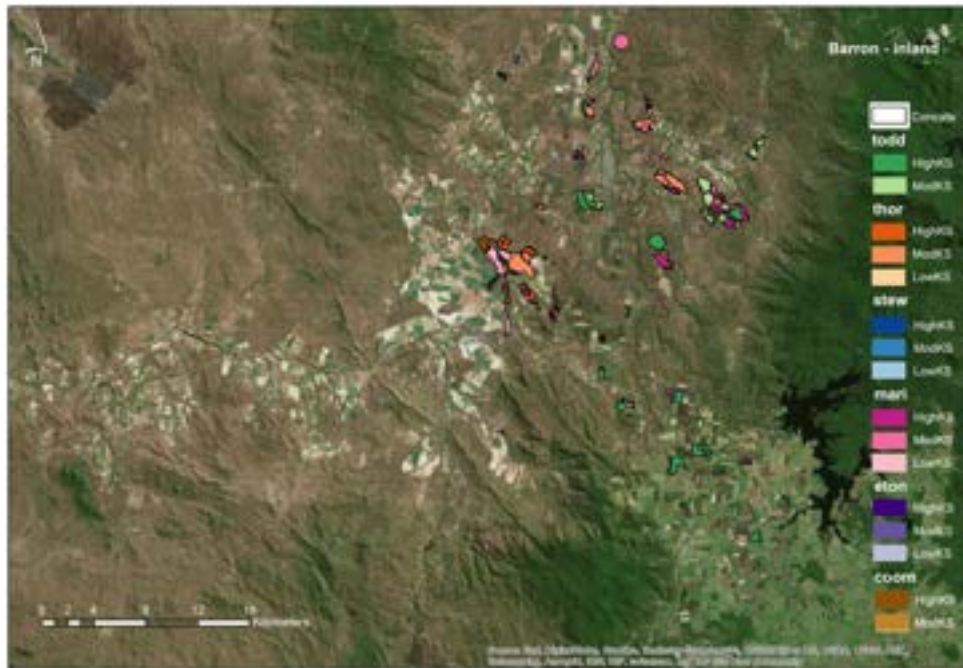


Figure 4.17: Soil types of cane land in the Daintree and northern section of the Mossman catchments. Mapped from soil layers provided by Paddock to Reef.



Figure 4.18: Soil types of cane land in the southern section of the Mossman catchment. Mapped from soil layers provided by Paddock to Reef.



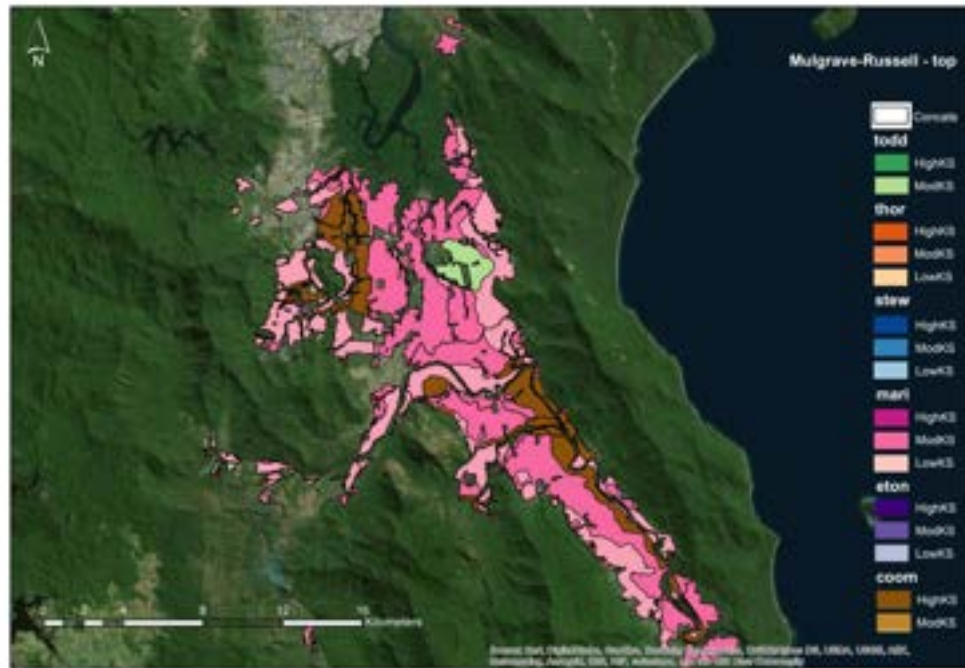


Figure 4.21: Soil types of cane land in the northern section of the Mulgrave-Russell catchment. Mapped from soil layers provided by Paddock to Reef.

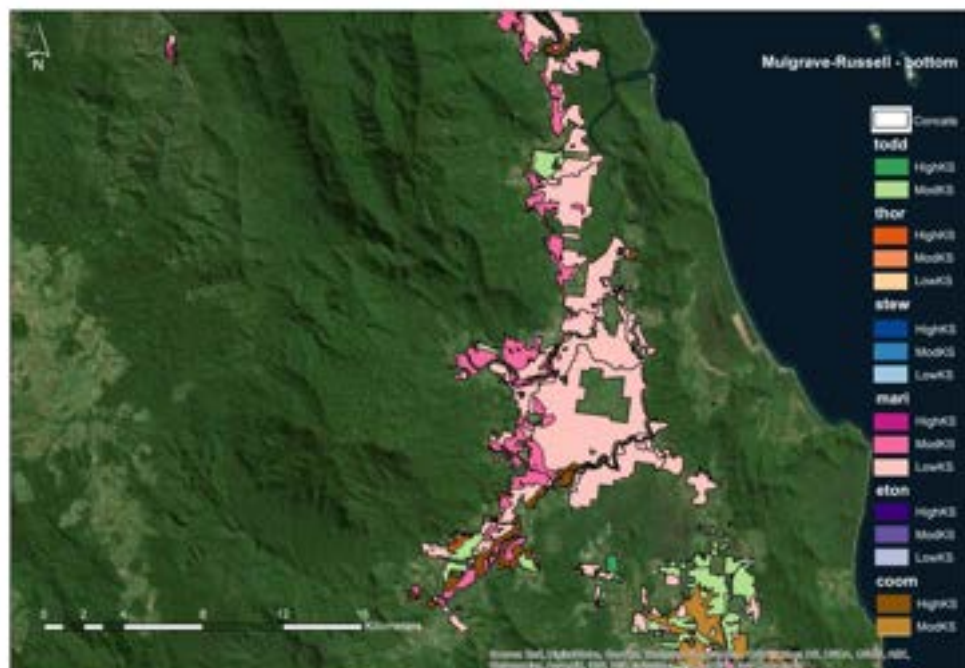


Figure 4.22: Soil types of cane land in the southern section of the Mulgrave-Russell catchment. Mapped from soil layers provided by Paddock to Reef.

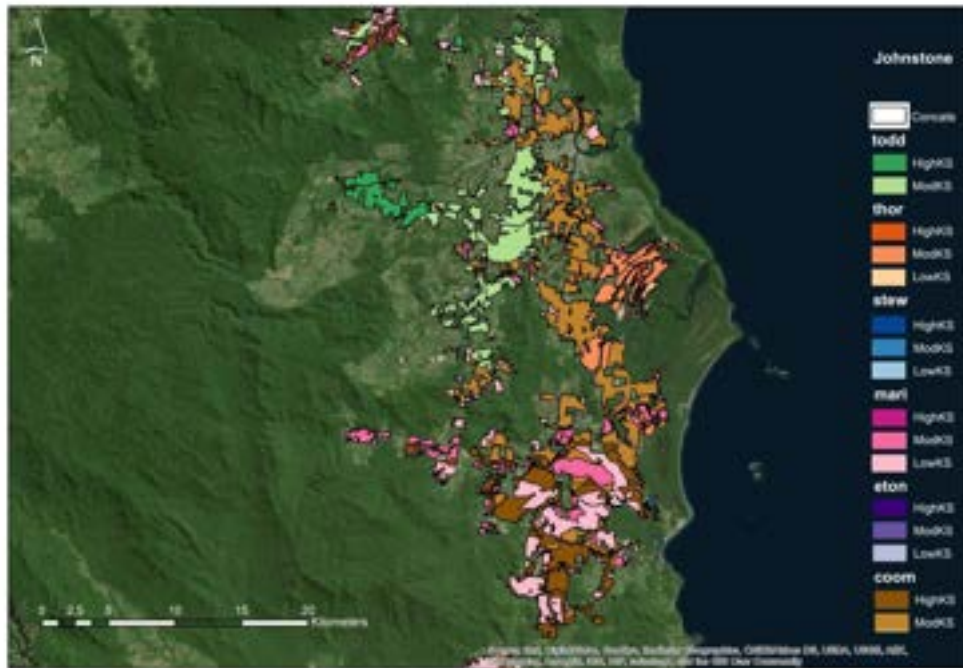


Figure 4.23: Soil types of cane land in the Johnstone catchment. Mapped from soil layers provided by Paddock to Reef.

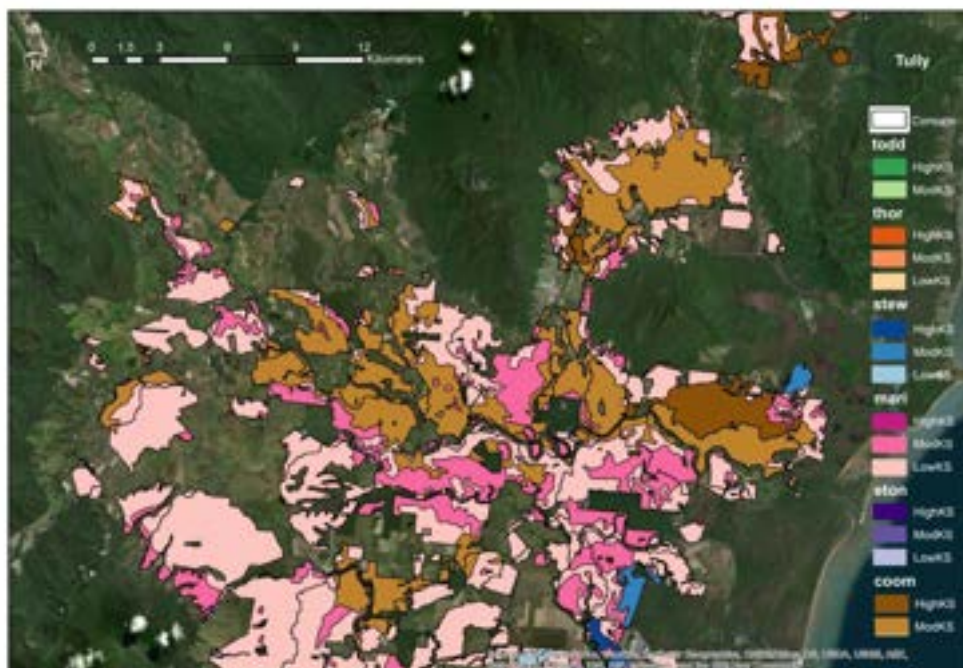


Figure 4.24: Soil types of cane land in the Tully catchment. Mapped from soil layers provided by Paddock to Reef.



Figure 4.25: Soil types of cane land in the Murray catchment. Mapped from soil layers provided by Paddock to Reef.

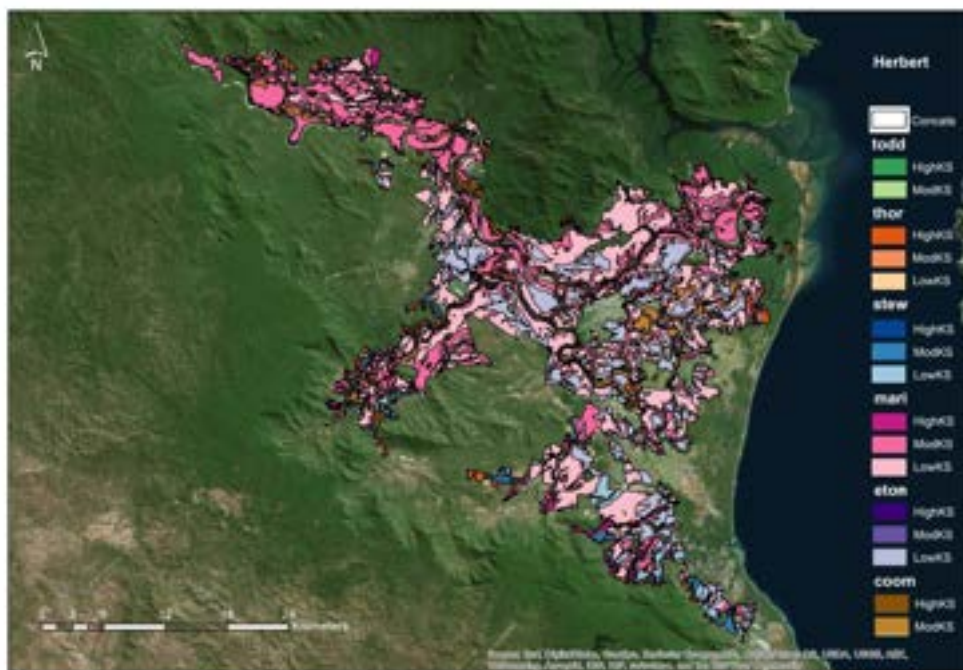


Figure 4.26: Soil types of cane land in the Herbert catchment. Mapped from soil layers provided by Paddock to Reef.

5.0 NITROGEN CREDITS FROM PRACTICE CHANGE IN SUGARCANE

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5. Summary

This Chapter used a bottom-up approach to estimate sugarcane growers' minimum compensation requirements for improving fertiliser management practice as a mechanism for supplying nitrogen credits to a nitrogen trading market. Compensation requirements were calculated as the sum of four separate cost elements: opportunity cost, the compensation required for exposure to production risk, transition cost, and transaction cost. Catchment-specific supply curves for nitrogen credits were produced for the Mulgrave-Russell, Johnstone, Tully, Murray and Herbert catchments by estimating minimum compensation requirements for successive levels of fertiliser practice change for individual cane land management units. Stochastic simulation was used to randomise allocations of management units to farm sizes and to randomise allocations of baseline nutrient practice to management units in accordance with recently reported data for the catchments as a whole.

Modelling results suggest that all catchments show a similar progression in the per kilogram cost of delivering DIN reductions to End-of-Catchment (EoC), or, equivalently, the cost of supplying EoC DIN credits for nitrogen credit trading. Practice changes from Df to Cp/Cf and Bp typically deliver large reductions in average DIN loss at zero or modest cost (generally below \$10 - 12/kg DIN @ EoC). Further practice improvement to Bf generally delivers a considerable additional reduction in DIN loss at costs in the range \$15 – 60/kg DIN @ EoC). Practice improvement to Af, however, incurs large additional transition costs and delivers only modest further reductions in DIN, resulting in DIN reduction costs that increase very rapidly from around \$50/kg DIN @ EoC to more than \$250/kg DIN @ EoC. This suggests that other approaches for DIN reduction should be considered for catchments such as the Johnstone and the Herbert that appear unlikely to be able to meet their End-of-Catchment Water Quality Target through full adoption of level Bf fertiliser practice.

The supply curves produced assume that full adoption of each successive level of practice improvement can be achieved. Lower rates of adoption will reduce the total DIN load reduction that can be achieved through fertiliser practice change and will thus likely necessitate introduction of other DIN reduction mechanisms to deliver the End-of-Catchment Water Quality Objectives in most catchments.

Chapter 5 proceeds as follows. Section 5.1 provides an overview of the separate components of the private cost of changing management practice as incurred by cane farmers. Sections 5.1.1 through to 5.1.4 explain each cost component in more detail, showing how these costs are calculated for individual management units using data from P2R APSIM predictions. Using the estimated costs from Sections 5.1.1 – 5.1.4 and DIN loss predictions under different levels of management practice as predicted by P2R APSIM, Section 5.2 predicts management unit-specific supply of nitrogen credits and presents these results as catchment-specific supply curves for nitrogen credits from practice change in cane production. Section 5.3 discusses the results; Section 5.4 concludes.

5.1 Costs incurred in improving nitrogen management practice

Landholders will typically incur private costs when changing from their standard management practice to reduce impacts on water quality. Landholders therefore require some minimum level of compensation payment in order to participate in water quality improvement programs that require some form of practice change, such as a nitrogen credit trading scheme. In nitrogen credit trading, improvement in fertiliser management practice by landholders can generate a supply of nitrogen credits. A *supply curve for nitrogen abatement*, or equivalently, a *supply curve for nitrogen credits from practice change*, can be produced by estimating landholders' minimum *willingness to accept (WTA) compensation* for specific practice changes, and the DIN reduction at End-of-Catchment that would result from those changes.

Enabling landholders to participate in nitrogen abatement mechanisms is a major challenge for water quality management globally and a considerable literature has developed around this topic. Drawing on this literature, Project 3.1.6 assumes that a landholder's minimum WTA comprises four main components of private cost: *opportunity costs*, *transition costs*, *transaction costs*, and costs associated with actual and perceived increases in *production risk* (Coggan et al. 2014, 2017; Motallebi et al. 2016, 2017; Rolfe and Windle 2016; O'Connell et al. 2017).

Opportunity cost associated with improving management practice is the anticipated reduction in farm profits that result from changes in farming practice (Rolfe and Windle 2016; Farr et al. 2019).

Transition costs associated with improving management practice are the one-off costs of purchasing the equipment and machinery required to implement a practice change (van Grieken et al. 2019).

Transaction costs generally refer to the costs of resources, including time, that a landholder has to commit to learn about and engage with a practice improvement program (Coggan et al. 2010).

Costs associated with increased production risk following changes in land management practice typically arise from perceived and/or actual concerns about impacts on profits due to low yields under adverse weather conditions, or the inability to obtain a bumper yield when weather conditions are particularly favourable (Beverly et al. 2016; Kandulu et al. 2018; Farr et al. 2019).

Given that, in combination, these different components of private cost are assumed to comprise landholders' minimum WTA compensation to change fertiliser management practice to reduce nitrogen emissions, obtaining reliable estimates of these costs for individual cane land management units in each Wet Tropics catchment is a critical first step in developing catchment-specific supply curves for sourcing nitrogen credits. As explained in Chapter 4, Project 3.1.6 uses P2R APSIM predictions of changes in cane yield following a change in management practice as the basis for deriving management unit-specific opportunity costs. Similarly, P2R APSIM predictions of the difference between average cane yield in 'good years' and 'bad years' before and after a change in fertiliser management underpin management unit-specific estimates of compensation requirements for perceived or actual exposure to production risk. Opportunity costs and the cost of exposure to production risk are calculated separately for each level of practice change for all 563 management units across the Wet Tropics (see Sections 5.1.1 to 5.1.4 for further details). Transition costs for individual management units are calculated for relevant changes in practice, categorised by farm size, using estimates of farm size-specific transition costs from the literature (van Grieken et al. 2010a, b, 2014, 2019) (see Section 5.1.3). Transaction costs for individual management units are also obtained from the literature, although in this case farm size is not considered to be a significant driver of variation (Coggan et al. 2014) (see Section 5.1.2). The sum of these four cost components is assumed to provide a reasonable reflection of the minimum compensation landholders would be WTA to undertake practice change, and thus also the minimum return that landholders would need to achieve from sale of nitrogen credits in order to choose to supply credits to a nitrogen credit market.

Farmer surveys and stated preference valuation methods such as choice experiments provide alternative approaches for estimating landholders' WTA requirements for changing land management practice or changing land use. Many such examples exist in the international literature (e.g. Beharry-Borg et al., 2013; Franzén et al., 2016; Haile et al., 2019; Hasler et al., 2019; Wu and Ge, 2019) and some in Australian settings (e.g. Brown et al., in press; Greiner, 2016). Whilst these survey-based approaches are helpful for identifying features such as socio-economic drivers of WTA (Haile et al. 2019), landholder groupings with similar WTA requirements (Brown et al.; Beharry-Borg et al. 2013), and variation around mean WTA for particular landholder groupings (e.g. Villanueva et al., 2017), the data required to use findings from such studies to construct WTA estimates at fine spatial resolution are not generally available for GBR catchments. For example, it is likely that landholder age and existing level of management practice will both affect WTA for implementing a given change in practice; however, such data are not publicly available at fine spatial resolution for sugarcane production in the Wet Tropics. This prevents location-specific predictions of landholders' WTA from being constructed from survey-based approaches. Post-hoc analysis of farmers' responses to the Federal Government's Reef Trust Tenders (Commonwealth of Australia 2014) could, in theory, also be used to identify key drivers of differences in WTA for reducing fertiliser application rates. Here again, however, individual-specific data on tender offers are not publicly available. Since P2R APSIM predictions indicate that the effectiveness of practice improvements for generating reductions in DIN load at End-of-Catchment differ significantly with local variations in soil type, soil permeability and climate, it is important to estimate landholders' WTA requirements for implementing practice changes at equivalent spatial resolution. For these reasons a 'bottom up' approach of constructing an estimate of WTA from its separate

components was used in Project 3.1.6 rather than a ‘top down’ approach of estimating WTA based on the characteristics of farmers and farm businesses⁴⁵.

The following subsections describe how each cost component of landholders’ WTA was constructed using data from P2R APSIM and the literature.

5.1.1 Opportunity Cost

P2R APSIM modelling provides estimated average cane yields over the period 1987-2013 for each level of fertiliser management practice (Df, Cp, Cf, Bp, Bf, Af – see Table 4.3) for each of 563 cane land management units in the Wet Tropics. This enables the difference in average cane yield to be calculated for each step change in practice (Df to Cf, Cf to Bp, Bp to Bf etc.). Figure 5.1 shows how the difference in average cane yield for step changes in fertiliser management practice varies across management units, grouped by soil type. For most practice changes on most soil types, average reductions in cane yield are less than 1 tonne per hectare. For comparison, median average cane fresh weight yields vary between 66 and 76 tonnes per hectare. Thus, the predicted reductions in average cane yield under fertiliser practice change are relatively minor, especially for practice changes Df to Cf, Cf to Bp, and Bf to Af. The largest reductions in cane yield are typically predicted for practice change Bp to Bf (the shift from nitrogen applications based on DYP to applications based on ‘block yield potential’). The variation in the reduction in average cane yield about its median under practice change differs by soil type, with stew and todd soils displaying wider variation.

⁴⁵ A further reason for not using a choice experiment in Project 3.1.6 to determine landholders compensation requirements for changing land management practices or changing land use is that the Wet Tropics Major Integrated Project (MIP) and the commercial Reef Credit aggregator GreenCollar were actively seeking practice change and land use change commitments from landholders in the Wet Tropics whilst NESP TWQ Hub Project 3.1.6 was being undertaken. To quantify landholders’ responses to compensation payments, choice experiments need to include some compensation payments at levels that almost all landholders will accept – even if those payment levels would not be viable for a commercial scheme. Considerable confusion and potential disruption could therefore have ensued if a NESP TWQ Hub Project 3.1.6-administered choice experiment included payment levels well in excess of those on offer from GreenCollar or the Wet Tropics MIP.

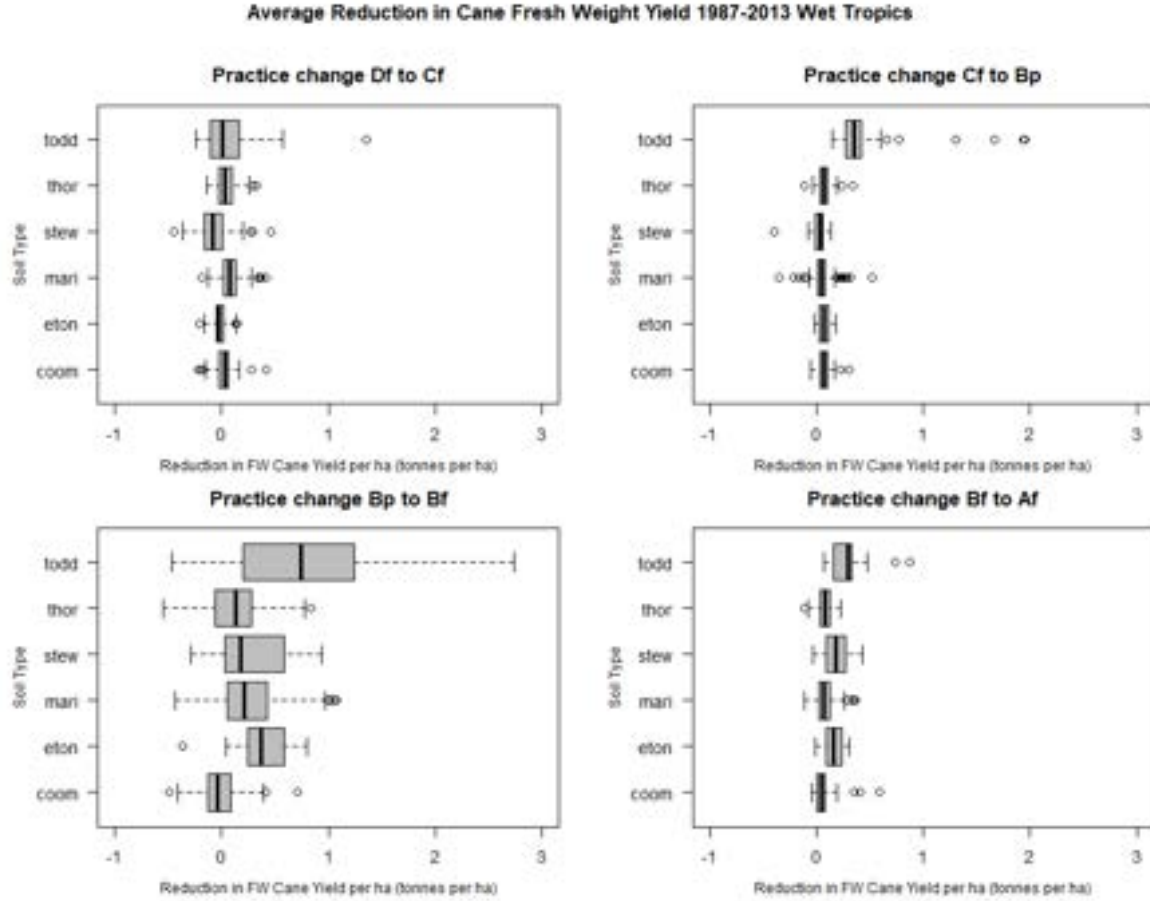


Figure 5.1: Average (1987-2013) annual reduction in fresh weight cane yield for practice changes Df to Cf, Cf to Bp, Bp to Bf and Bf to Af for 563 cane land management units in the Wet Tropics, grouped by soil type. (For comparison, median average cane fresh weight yields range from 66 – 76 tonnes depending on soil type (Figure 4.10)).

In the P2R APSIM modelled data supplied to Project 3.1.6, fertiliser management practices differ only in terms of the fertiliser rate applied (see Table 4.3). All other cane cultivation operations remain unchanged between the steps in management practice. The opportunity cost incurred by the landholder for a step change in fertiliser practice can therefore be calculated from differences in cane revenue, harvesting cost and fertiliser expenditure between two practice levels using Equation 5.1.

$$OppCost_{i,(m1,m2)} = (FW_{i,m1} - FW_{i,m2}) * P_c - (FW_{i,m2} - FW_{i,m1}) * P_h - (N_{i,m2} - N_{i,m1}) * P_n \quad (5.1)$$

where:

$OppCost_{i,(m1,m2)}$ = opportunity cost for management unit i of changing fertiliser practice from practice $m1$ to practice $m2$

$FW_{i,m1}$ = cane fresh weight (tonnes) produced by management unit i under management practice $m1$

$FW_{i,m2}$ = cane fresh weight (tonnes) produced by management unit i under management practice $m2$

$N_{i,m1}$ = nitrogen application (kg) by management unit i under management practice $m1$

$N_{i,m2}$ = nitrogen application (kg) by management unit i under management practice $m2$

P_c = farm price for fresh weight cane (AUD\$/tonne cane fresh weight)⁴⁶
 P_h = harvesting cost (AUD\$/tonne cane fresh weight harvesting)⁴⁷
 P_n = nitrogen cost (AUD\$/kg N)⁴⁸

Figure 5.2 shows how opportunity costs for step changes in fertiliser management practice vary across management units, grouped by soil type. For the majority of practice changes on most soil types, average opportunity costs are predicted to be negative, i.e. cost savings are realised through practice change. This is because the relatively small predicted reductions in cane yield for most practice change steps (Figure 5.1) are more than offset by reductions in fertiliser expenditures.

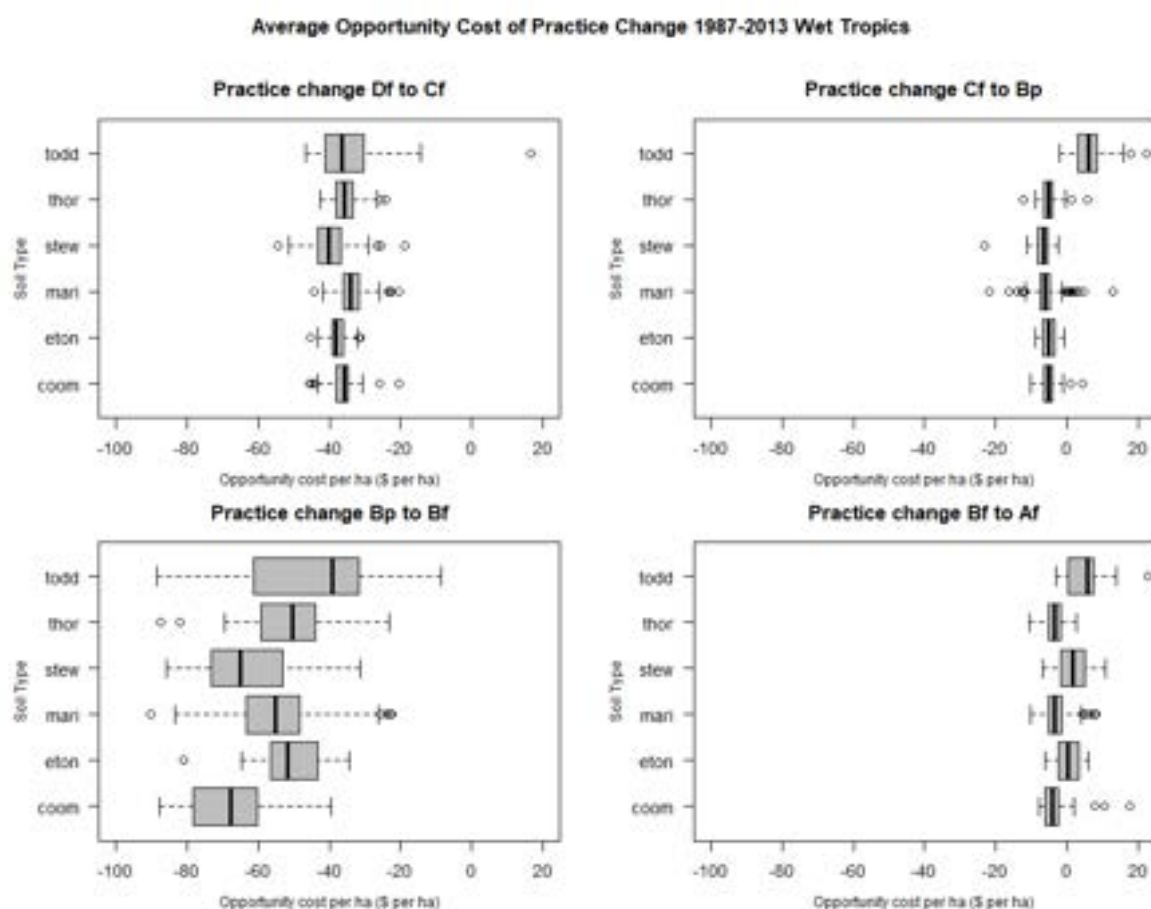


Figure 5.2: Average (1987-2013) annual opportunity cost of practice changes Df to Cf, Cf to Bp, Bp to Bf and Bf to Af for 563 cane land management units in the Wet Tropics, grouped by soil type.

⁴⁶ Average cane fresh weight price over the years 1987-2013 (<https://www.abs.gov.au/AUSSTATS>) (\$47/tonne) is calculated using the cane pricing formula (MSF Sugar) as follows:

$$P_c = (\text{Average QSL Sugar Pool Price} \times 0.009 \times (\text{Average CCS} - 4)) + 0.662$$

with pricing expressed in 2018 AUD\$/tonne.

⁴⁷ P_h = harvesting cost per tonne of fresh weight cane from Kandulu et al. (2018)= AUD\$ 7.50/tonne

⁴⁸ P_n = average cost of nitrogen over the years 1987-2013, derived from the average cost of urea 1987-2013 (\$708 2018 AUD\$/tonne)(<https://www.abs.gov.au/AUSSTATS>), assuming the nitrogen content in urea is 46% (<https://impactfertilisers.com.au/products/straights/urea/>).

5.1.2 Cost of exposure to risk

Project 3.1.6 uses management unit-specific, year-specific P2R APSIM predictions of cane yield as the basis for estimating the changes in farm-specific gross margin from low yields under adverse weather conditions, or from high yields under particularly favourable weather conditions, that would follow from a change in fertiliser management practice. We follow Kandulu et al (2018) in using a *conditional value at risk* (CVaR) approach to develop a quantitative metric of farm-specific average net returns in ‘good years’ and ‘bad years’, when weather conditions are particularly favourable or unfavourable, respectively. The difference between average net return from ‘good years’⁴⁹ under a given level of fertiliser application and average net return from ‘good years’ under reduced fertiliser application provides an indication of the production value farmers might consider to be at risk by changing practice. Kandulu et al’s approach can be extended to calculate the reduction in average net return in ‘bad years’⁵⁰ under stepwise reductions in fertiliser application.

In Project 3.1.6 we take the sum of the reduction in P2R APSIM predicted average net return from 1-in-10 good years (CVaR^{0.90}) and from 1-in-10 bad years (CVaR^{0.10}) over the period 1987-2013 for a specific change in fertiliser practice (e.g. Df to Cf) to be a management unit-specific estimate of the payment that the farmer would require as compensation to cover their exposure to production risk when changing fertiliser practice. Figure 5.4 and Figure 5.5 show, respectively, reductions in management unit-specific, year-specific gross margins from 1-in-10 good years (CVaR^{0.90}) and 1-in-10 bad years (CVaR^{0.10}) under different levels of practice change.

These results show that 1-in-10 ‘good year’ and ‘bad year’ gross margins typically increase, rather than reduce, for all soils as fertiliser practice improves from Df to Cf. Further practice improvements generally continue to increase ‘good year’ and ‘bad year’ gross margins on coom, mari and thor soils. The yield response of stew and – particularly – todd soils is more varied, with practice improvement from Bp to Bf under some circumstances liable to produce substantial reductions in gross margins per hectare in ‘good’ and ‘bad’ years. These soils comprise only approximately 3% (stew) and 5% (todd) of Wet Tropics cane land but are locally important in parts of the Barron (inland), and Johnstone catchments.

Management unit-specific compensation requirements for exposure to production risks in ‘good years’ and ‘bad years’ are added to the management unit-specific annual opportunity costs described in Section 5.1.1.

⁴⁹ *Conditional value at risk* (CVaR) can be configured to report the average net return from, for example, the top 10% or top 20% of a distribution of historical or modelled net returns. The difference between average top 10% or top 20% of net returns and overall average net return provides a representation of the value that a farmer might consider to be at risk if, through changing practice, they miss out on the full extent of net returns that could be achieved in a 1-in-10 (CVaR^{0.90}) or 1-in-20 (CVaR^{0.95}) ‘good year’.

⁵⁰ Similarly, the average net return from, for example, the mean net return obtained during a bottom 1-in-10 or bottom 1-in-20 ‘bad year’ could be estimated from CVaR^{0.10} or CVaR^{0.05}.

Average Reduction in Gross Margin per hectare for 90%-ile 'good years' 1987-2013 Wet Tropics (excl. Herbert)

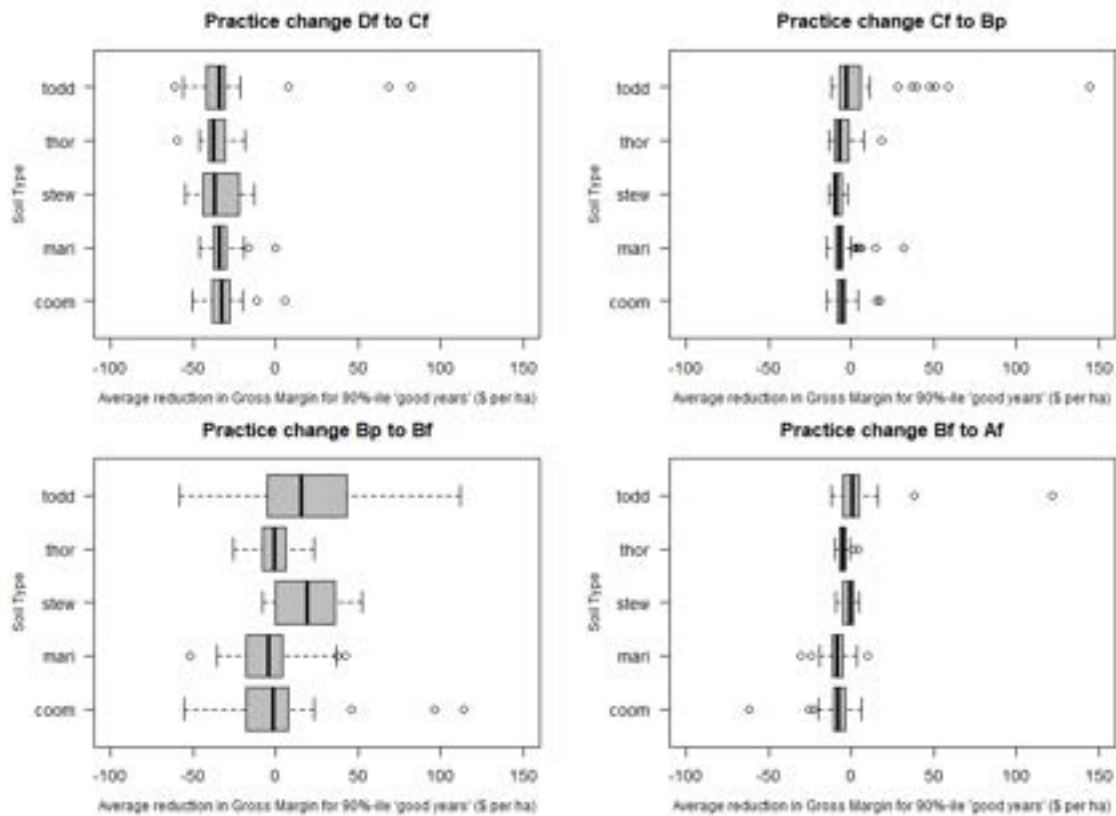


Figure 5.3: Average (1987-2013) reduction in gross margin per hectare for 90th percentile 'good years' for practice changes Df to Cf, Cf to Bp, Bp to Bf and Bf to Af for 360 cane land management units in the Wet Tropics (excluding the Herbert catchment), grouped by soil type.

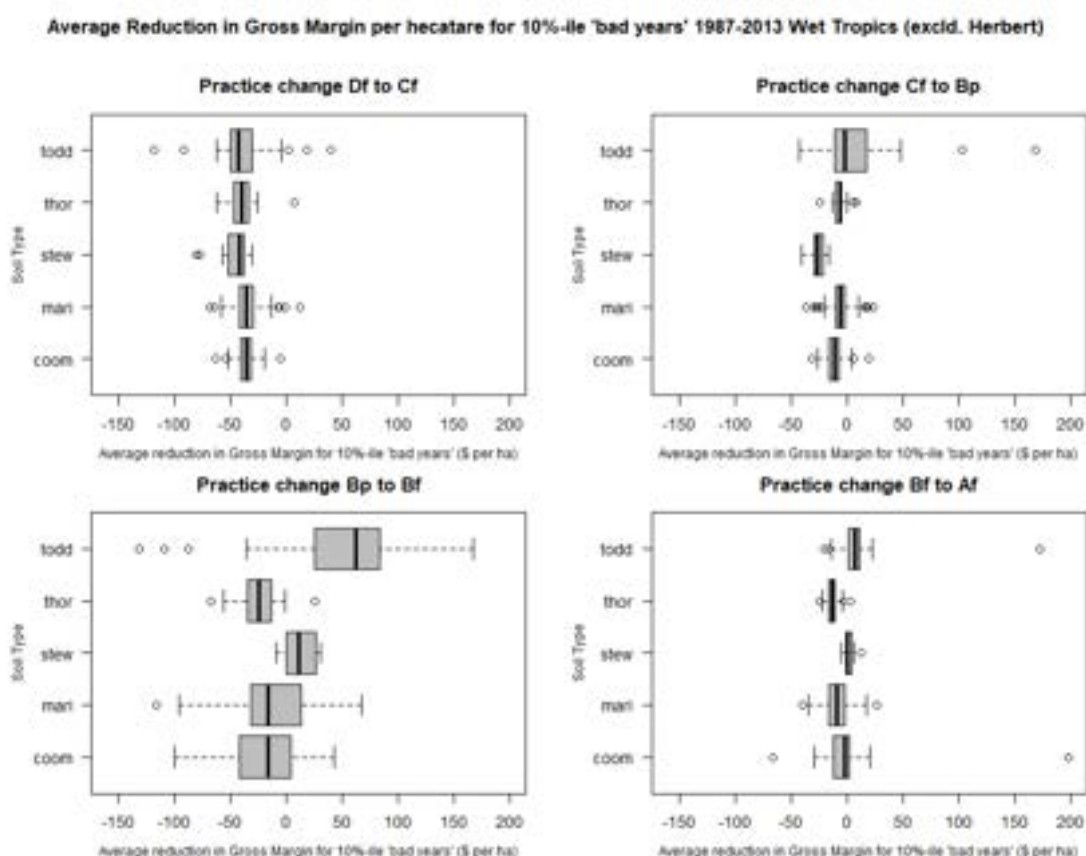


Figure 5.4: Average (1987-2013) reduction in gross margin per hectare for 10th percentile 'bad years' for practice changes Df to Cf, Cf to Bp, Bp to Bf and Bf to Af for 360 cane land management units in the Wet Tropics (excluding the Herbert catchment), grouped by soil type.

5.1.3 Transition cost

The transition costs incurred in improving management practice in sugarcane production have been studied in detail for more than a decade. Van Grieken et al. (2010c) specify the on-ground farming activities required to achieve each step in the D, C, B, A practice improvement sequence for management of sediment, fertiliser and pesticide runoff from cane production. Van Grieken et al. detail the capital costs required for stepwise transitions between the different practice levels for representative farms in all Reef regions including the Wet Tropics (M.E. van Grieken et al., 2010a; Section 4.3 p.11-13). The transition costs reported by van Grieken et al. assume that the farm improves soil, fertiliser and pesticide management simultaneously. Only some of the capital expenditures listed by van Grieken et al. are thus relevant in the context of nitrogen-based water quality credit trading. Drawing from van Grieken et al. (2010a; 2019), assumed transition costs for the stepwise improvements in fertiliser management practice considered in Project 3.1.6, as set out in Table 4.3, are listed in Table 5.1.

Table 5.1: Transition costs for improvements in nitrogen practice (2018 AUD\$) on a 120ha cane farm in the Wet Tropics, from van Grieken et al. (2010a; Section 4.3 p.11-13), inflated to 2018 AUD\$. Annualised transition cost is calculated over a 6-year cane cycle at a 7% discount rate.

Management class change	Capital items	Capital Cost (2018 AUD\$)	Annualised transition cost (2018 AUD\$)
D to C	No capital investment required	0	0
D or C to Bp	Soil tests (annual cost) 6 x E42 or E43 tests ⁵¹	\$625	\$625
Bp to Bf	Stool splitter fertiliser box	\$47,200	
	Harvester modifications	\$14,750	\$13,350
	Farm tractor modifications	\$1,770	
Bf to Af	GPS on farm tractor for variable within-block fertiliser application rate	\$47,200	\$14,850
	Ripper/rotary hoe modifications	\$23,600	

For inclusion in compensation requirements alongside annual opportunity costs and production risks, transition costs are annualised over a 6-year cane production cycle using Equation 5.2.

$$A = C \cdot \left(\frac{r}{1 - (1+r)^{-T}} \right) \quad (5.2)$$

where:

A = annualised cost

C = cost incurred at the start of the annualization period (i.e. at time $t=0$)

r = discount rate (assumed to be 7% per annum)

T = number of years over which the incurred cost is to annualised (assumed to be the 6-year cane production cycle)

Van Grieken et al (2019) further report that transition costs per hectare of cane land in the Tully-Murray catchment vary significantly with farm size; transition cost per hectare being considerably higher for small- and medium-sized farms (van Grieken et al., 2019; Table 2, p.4). Taking the annualised transition costs in Table 5.2 as representative for van Grieken et al's medium-sized farm, transition costs for small and large farms are scaled in the ratio of the results provided by van Grieken et al. (2019; Table 2, p.4). Table 5.2 shows estimated annualised transition costs for small (20 – 80 ha), medium (80 – 180ha) and large (180 – 1230ha) cane farms in the Wet Tropics.

⁵¹ Pricing for soil testing compliant with 2019 Reef Regulations kindly provided by Maria Ribbeck (Canegrowers Organisation and Terrain NRM, Tully).

Table 5.2: Annualised transition costs (2018 AUD\$) for practice changes on small, medium and large cane farms in the Wet Tropics, drawing from van Grieken et al. (2019; Table 2, p.4).

Farm category	Small	Medium	Large
<i>Farm size range</i>	20 – 80 ha	80 -180 ha	more than 180ha
<i>Mean size</i>	40ha	115ha	428ha
Practice change	Annualised transition cost (2018 AUD \$)		
<i>D to C</i>	\$0	\$0	\$0
<i>D or C to Bp</i>	\$250	\$625	\$2,300
<i>Bp to Bf</i>	\$10,000	\$13,350	\$18,150
<i>Bf to Af</i>	\$11,000	\$14,850	\$20,200
Practice change	Annualised transition cost per hectare (2018 AUD \$)[#]		
<i>D to C</i>	\$0	\$0	\$0
<i>D or C to Bp</i>	\$6.25	\$5.45	\$5.40
<i>Bp to Bf</i>	\$250	\$115	\$40
<i>Bf to Af</i>	\$275	\$130	\$45

[#]Per hectare costs derived using mean farm size.

The annualised transition costs in Table 5.2 are included in the management unit-specific costs of supplying nitrogen credits.

Transition costs accrue to the farm and vary significantly with farm size. Unfortunately, no data were available to allocate the 563 individual cane land management units into separate farm businesses. It was not therefore possible to assign transition costs to each of the 563 management units deterministically. However, Sing and Barron (2014) report the distribution of farm sizes as a proportion of the total cane land area in each of the Wet Tropics catchments (Sing and Barron, 2014; Table 14, p.42), shown here in Table 5.3. These data allow management units to be allocated stochastically to small, medium or large farms (using van Grieken et al's (2019) size designations), thus enabling transition costs for each management unit to be scaled as a proportion of its stochastically-allocated farm size. For example, a 46 ha management unit that had been allocated to a medium-sized farm (mean medium-sized farm area = 115 ha) would be assigned 40% of a full farm's transition cost for a particular change in nutrient practice. Stochastic catchment-specific cost curves for supply of nitrogen credits are created by producing 50 random allocations of management units to farm sizes within each catchment. For each random allocation, the proportion of the catchment's total cane land area in each farm size class matches Sing and Barron's reported data to within 1%.

Table 5.3: Percentage of cane land held by small, medium and large farms within Wet Tropics catchments. From data reported by Sing & Barron (2014; Table 14, p.42).

Farm category	Small	Medium	Large
<i>Farm size range</i>	20 – 80 ha	80 -180 ha	more than 180ha
<i>Mean size</i>	40ha	115ha	428ha
Catchment	Percentage of farms by size class		
<i>Daintree</i>	18%	32%	50%
<i>Mossman</i>	18%	32%	50%
<i>Barron</i>	23%	33%	44%
<i>Mulgrave-Russell</i>	18%	33%	49%
<i>Johnstone</i>	26%	25%	49%
<i>Tully</i>	8%	15%	77%
<i>Murray</i>	8%	15%	77%
<i>Herbert</i>	13%	28%	59%

5.1.4 Transaction Cost

Using survey data collected in 2012 from 110 landholders who successfully secured funding in the 2010/11 and/or 2011/12 Australian Government-funded Reef Rescue rounds, Coggan et al. (2014; Table 5, p.512) report that, on average, transaction costs per farm for engaging with changes in farming activities that improve nutrient management practice⁵² are \$9,937 (in 2018 AUD\$)⁵³. The annualised equivalent transaction cost over a 6-year cane cycle at 7% discount rate is \$2,084 (2018 AUD\$). This annualised transaction cost is included in the management unit-specific costs of supplying nitrogen credits whenever a land management unit changes nutrient practice from its current baseline level.

Unfortunately, no data were available to allocate baseline levels of nutrient management practice to the 563 individual cane land management units. It was not therefore possible to assign transaction costs for shifts away from baseline practice to each of the 563 management units deterministically. However, the proportions of cane land area at each level of nutrient management practice is provided for each Wet Tropics catchment in the Management Practice Adoption report that accompanies the 2017 and 2018 Reef Report Card (Commonwealth of Australia and Queensland Government 2018), reported here in Table 5.4. These data allow an initial level of nutrient practice to be allocated to individual management units stochastically, respecting the reported percentages of land area under different levels of nutrient practice within each catchment. As explained in the preceding section, individual land management units have been stochastically allocated into small, medium or large farms. Knowing the size of the farm to which a particular management unit has been allocated enables the transaction cost for that management unit to be scaled in proportion to the total land area of its

⁵² Transaction cost for engaging to improve nutrient management from class C to class B are reported to be \$9,887 (AUD 2018\$), only slightly different to the overall average. We therefore use the overall average transaction as the transaction cost of engaging in nutrient management practice improvements to supply nitrogen credits to a nitrogen credit market.

⁵³ The sample of farmers on which Coggan et al's transaction costs are based is somewhat atypical as it is likely to contain a high proportion of early adopters. This could bias the estimated transaction costs in either direction. Lacking further data to quantify this potential bias, we use Coggan's estimate.

stochastically-allocated farm size. For example, a 46 ha management unit stochastically-allocated to a medium-sized farm (mean medium-sized farm area = 115 ha) would be assigned 40% of a full farm's transaction cost (\$834/year) when nutrient practice changed from its *baseline* level. Whereas, if the same management unit had been allocated to a large-sized farm (mean large-sized farm area = 428 ha), it would be assigned 11% of the full farm's transaction cost (\$229/year).

Baseline nutrient practice level is also allocated stochastically across management units in accordance with the proportion of each catchment's total cane land area under each level of management practice as reported in the 2017 and 2018 Reef Report Card. Stochastic catchment-specific cost curves for supply of nitrogen credits are created by producing 50 random allocations of baseline nutrient management practice to individual management units within each catchment.

Table 5.4: Percentage of cane land area at different baseline fertiliser practices within Wet Tropics catchments. From data reported in the Management Practice Adoption from the 2017 and 2018 Reef Report Card (Commonwealth of Australia and Queensland Government 2018).

Baseline Fertiliser Practice	D	C	B	A
<i>Daintree</i>	48%	32%	13%	7%
<i>Mossman</i>	29%	42%	19%	10%
<i>Barron</i>	42%	28%	15%	15%
<i>Mulgrave-Russell</i>	65%	21%	9%	5%
<i>Johnstone</i>	45%	37%	10%	7%
<i>Tully</i>	46%	28%	15%	11%
<i>Murray</i>	50%	27%	14%	9%
<i>Herbert</i>	52%	13%	20%	15%

5.2 Catchment-specific supply curves for nitrogen credits from practice change

The cost components described in the preceding section are applied to the 563 individual management units in the Wet Tropics to produce catchment-specific supply curves for nitrogen credits from practice change.

5.2.1 Supply of DIN credits from improvement in nutrient practice

Table 5.5 shows an example of the estimated minimum compensation required to persuade a landholder to supply End-of-Catchment DIN reductions (i.e. 'End-of-Catchment DIN credits') from a 120 ha management unit on medium permeability coom soil assigned to a medium-sized farm in the Johnstone catchment, assuming that the management unit is initially at nutrient practice Df⁵⁴.

⁵⁴ This example management unit is in P2R APSIM sub-catchment SC_390 and in climate zone 1750_14605.

Table 5.5: Cost of End-of-Catchment DIN credits sourced via practice change from 120ha of medium permeability coom soil on a medium-sized farm in the Johnstone catchment. (DIN transport coefficient to End-of-Catchment (surface water & ground water pathways in combination) = 0.618).

Practice change	DIN reduction End-of- Catchment (kg DIN)	Landholder's minimum compensation requirement (\$)	End-of-Catchment DIN credit price required to meet min WTA (\$/kgDIN EoC)
<i>Df to Cf ‡</i>	724	0	0
<i>Cf to Bp</i>	553	23	0.04
<i>Bp to Bf</i>	768	6,478	8.43
<i>Bf to Af</i>	103	14,970	145.43

‡ DIN losses are the same for practices Cp and Cf for this management unit.

For this management unit, P2R APSIM-predicted average fresh weight cane yield remains almost constant as fertiliser practice improves from Df through to Af, but fertiliser expenditures reduce considerably as application rate decreases. This increases net revenues for practice change to Cf, and almost maintains net revenues in the face of increased soil sampling costs for the change to Bp, even when transaction and transition costs (to Bp) are accounted for. Subsequent practice change to Bf incurs higher transition cost. The accompanying large reduction in End-of-Catchment DIN, however, means that a modest \$8.43/kg DIN credit price would be sufficient to cover the landholder's minimum compensation requirement for this step of practice change. The final step of practice change to Af incurs high costs and only delivers a modest further reduction in End-of-Catchment DIN. Consequently, the DIN credit price required to meet the farmer's minimum WTA corresponding to this final step of practice change is much higher at \$145.43/kg DIN. For most management units, including the example in Table 5.5, P2R APSIM does not predict that production risk will arise from fertiliser practice change in 1-in-10 'good' or 'bad' years. For those management units where production risk does arise⁵⁵, this increases the DIN credit price required to cover minimum compensation requirements for switching to practices Bf and Af.

5.2.2 Stochastic simulation

Stochastic simulation is used to randomise allocations of management units to farm sizes (that affects transition cost and transaction cost), and to randomise allocations of baseline nutrient practice to management units (that affects which steps of fertiliser practice improvement are still available for that management unit). The random allocation of baseline practice across management units aims to match the levels of catchment-wide practice adoption reported in the 2017 and 2018 Reef Report Card to within 1%. Even though the overall percentages of catchment cane land at different levels of nutrient management practice remain essentially constant across stochastic simulations, random allocation of baseline nutrient practices and farm sizes across management units affects the maximum End-of-Catchment nutrient load reduction that can be achieved and the total cost of achieving that load reduction.

Maximum achievable End-of-Catchment load reduction varies with the stochastic allocation because the End-of-Catchment load reduction delivered by a specific practice change varies

⁵⁵ Almost all management units with exposure to 1-in-10 'good year' or 'bad year' production risk are on todd soil.

across management units due to differences in soil type, soil permeability, climate zone, and DIN transport to End-of-Catchment via the surface water pathway. Similarly, the cost of delivering the maximum achievable End-of-Catchment load reduction varies with the random allocation of management units to farm sizes because of the link between farm size and transition cost. If, for example, many of the management units that are allocated baseline nutrient practice Df are also allocated to small farms then the cost of delivering (large) End-of-Catchment DIN load reductions by improving practice on those management units to level Af will be high – because the transition costs per hectare of improving practice to levels Bf and Af are considerably higher for small farms. Conversely, the costs of achieving the same level of End-of-Catchment DIN reduction would have been much lower had these same management units been allocated to large farms (refer to Table 5.2).

Stochastic estimations of catchment-specific supply curves for End-of-Catchment DIN credits, constructed as described above, are produced for the Mulgrave-Russell, Johnstone, Tully, Murray and Herbert⁵⁶ catchments.⁵⁷ These catchment-specific supply curves are shown in Figure 5.5 to Figure 5.9.

⁵⁶ Minimum WTA compensation calculated for management units in the Herbert catchment does not include the cost of exposure to production risk. However, as noted earlier, this is primarily a feature of todd soil. Since none of the management units in the Herbert are on todd soil, this omission is unlikely to have a noticeable effect on the DIN credit supply curve produced for Herbert catchment.

⁵⁷ There are insufficient management units in the Daintree, Mossman and Barron catchments to support meaningful stochastic simulation.

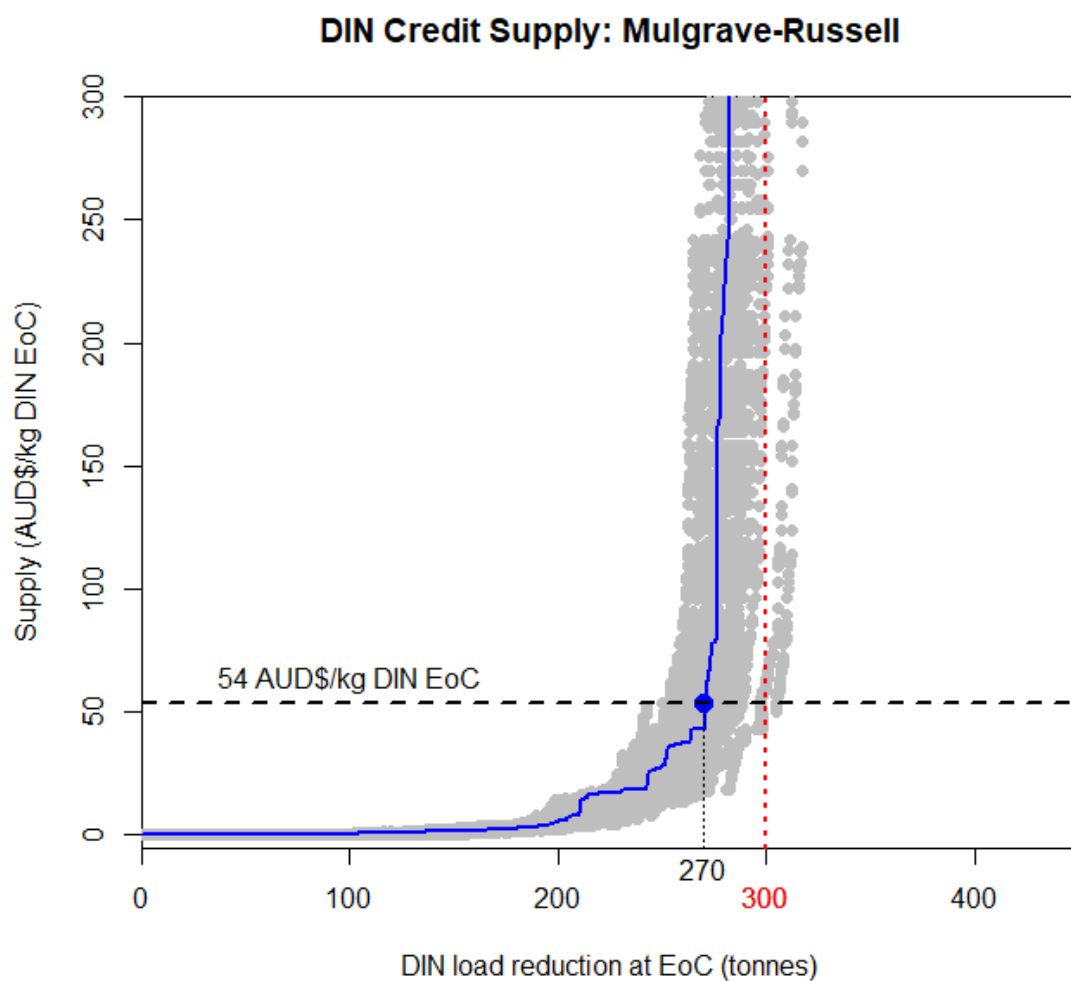


Figure 5.5: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Mulgrave-Russell catchment. Grey dots show 50 randomised replicate supply curves. Blue line shows a representative mid-band supply curve. The End-of-Catchment DIN load reduction target to meet the catchment's Water Quality Objective is shown in red. Credit supply price and quantity required to achieve more than 95% of the available cumulative DIN load reduction from adoption of practice level Bf are shown in black for the mid-band supply curve.

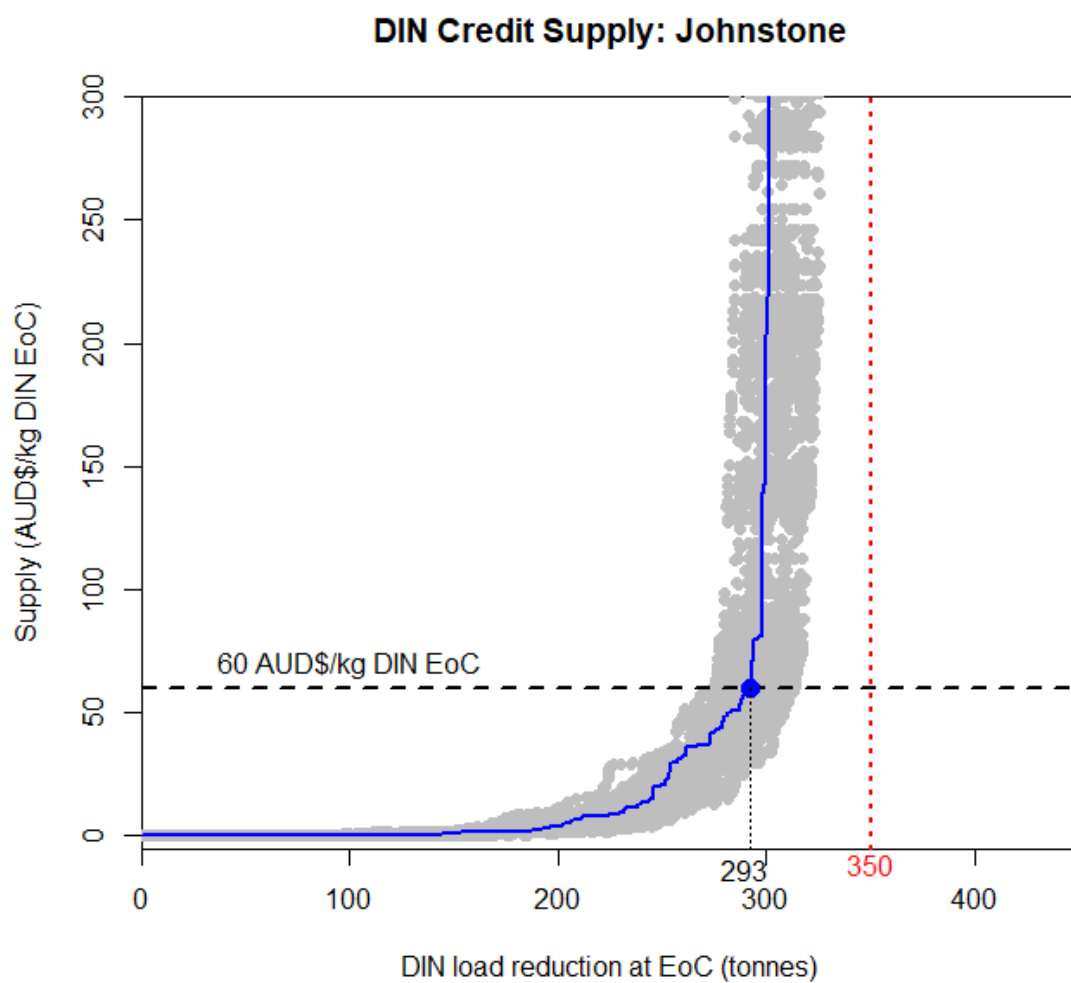


Figure 5.6: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Johnstone catchment. Other features as detailed for Figure 5.5.

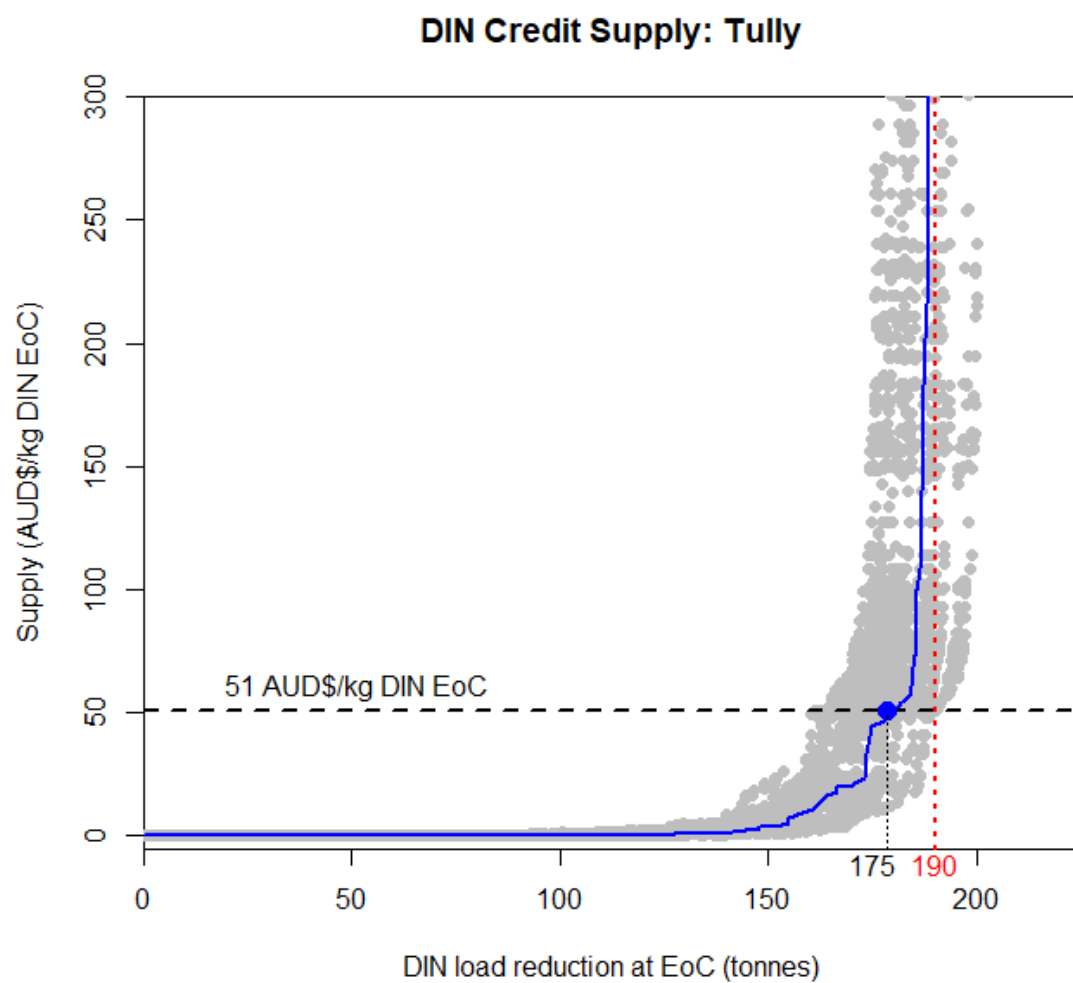


Figure 5.7: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Tully catchment. Other features as detailed for Figure 5.5.

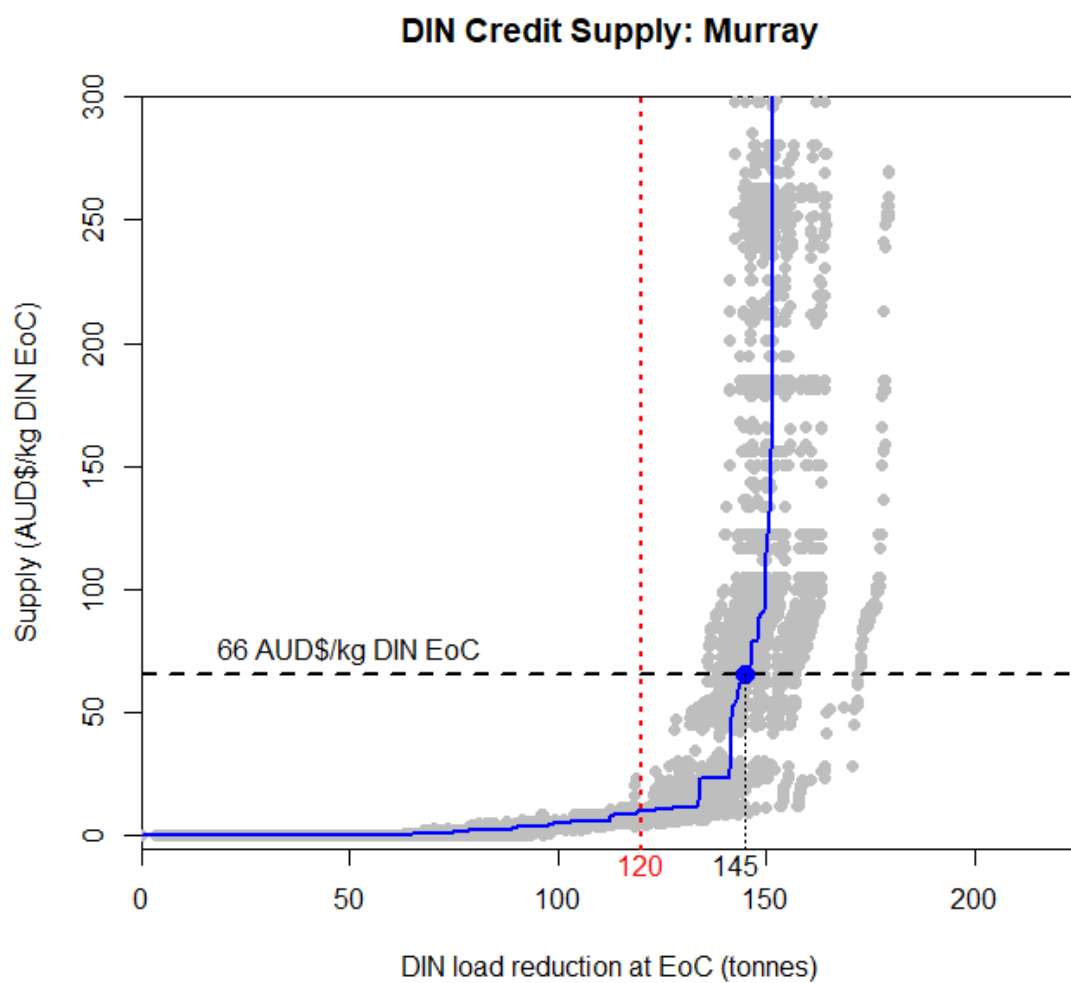


Figure 5.8: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Murray catchment. Other features as detailed for Figure 5.5.

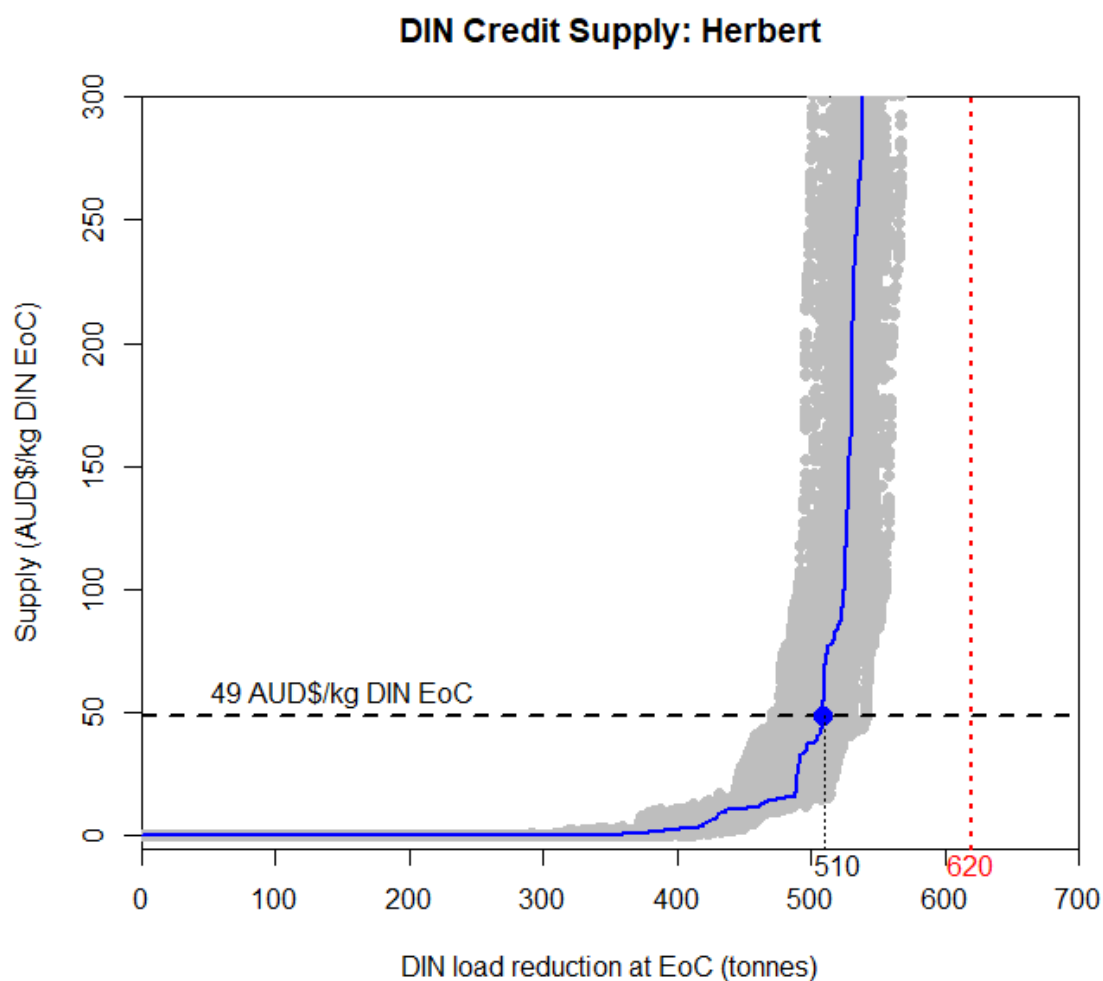


Figure 5.9: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Herbert catchment. Other features as detailed for Figure 5.5.

5.3 Discussion

Figure 5.5 to Figure 5.9 suggest that, under the assumptions of the DIN loss modelling in P2R APSIM and Project 3.1.6, the only major Wet Tropics catchment that appears likely to achieve its End-of-Catchment Water Quality Target for DIN through full implementation of fertiliser practice Bf is the Murray (Figure 5.8). End-of-Catchment DIN targets could potentially be achieved through full implementation of fertiliser practice Bf in the Mulgrave-Russell (Figure 5.5) and the Tully (Figure 5.7), but appear unlikely to be met through full implementation of fertiliser practice Bf in the Johnstone (Figure 5.6) and the Herbert (Figure 5.9).

All catchments show a similar progression in the per kilogram cost of delivering DIN reductions to End-of-Catchment, or, equivalently, the cost of supplying End-of-Catchment DIN credits for nitrogen credit trading. This is because, according to the P2R APSIM predictions, for the majority of management units on most soils in all catchments, a given fertiliser practice improvement incurs similar costs and delivers similar proportional reductions in DIN loss. Practice changes from Df to Cp/Cf and Bp typically deliver large reductions in average DIN loss at zero or modest cost (generally below \$10 - 12/kg DIN @ EoC). Further practice

improvement to Bf generally delivers a considerable additional reduction in DIN loss at costs in the range \$15 – 60/kg DIN @ EoC). Practice improvement to Af, however, incurs large additional transition costs and delivers only modest further reductions in DIN, resulting in DIN reduction costs that increase very rapidly from around \$50/kg DIN @ EoC to more than \$250/kg DIN @ EoC.⁵⁸ *This suggests that other approaches for DIN reduction should be considered for catchments that appear unlikely to be able to meet their End-of-Catchment Water Quality Target through full adoption of the fertiliser practice Bf* Chapters 6 and 11, respectively, explore opportunities for delivering DIN reductions by changing land use and installing constructed nitrogen treatment systems on marginal cane land, and obtaining DIN reductions by reducing the particulate nitrogen released via fine sediment erosion.

The costs of delivering End-of-Catchment DIN reductions via fertiliser practice change reported in this Chapter are compatible with previous estimates from Alluvium (Alluvium 2016, 2019) Rolfe & Windle (2016), Rolfe et al. (2018) and Star et al. (2018), after differences in the objectives, perspectives, and spatial resolutions of these studies have been considered. For example, in producing their prioritisation curve for DIN abatement via cane practice change, Star et al (2018; Fig.6 p.374) also account for likely levels of adoption (under voluntary programs), time lags in delivery of beneficial outcomes, risks of failure due to adverse weather, and sensitivity of the receiving region of the GBR to pollutant damage, as well as the on-farm costs of implementing practice change. In Project 3.1.6, adopting the perspective of the Point Source Water Quality Offsets Policy, concerns such as time lags in delivery, risks of failure, and sensitivity of the receiving environment are considered separately in Chapter 8 that addresses the roles of the delivery ratio and environmental equivalency ratio in setting an overall '*trading ratio*' to be applied when using DIN credits from diffuse sources to offset EA exceedances from point sources. A common feature across all supply curves in the literature for DIN reductions from practice change in cane production is an extremely rapid rise in cost per kilogram DIN reduction as the final steps of practice change are implemented (e.g. Rolfe et al.(2018; Figure 2, p.384), Star et al. (2018; Fig.6 p.374)).

Two major caveats apply to the results presented in this Chapter. Firstly, the supply curves produced for DIN credits assume that full adoption of each successive level of practice improvement can actually be achieved. How tenable this assumption is will depend on the mindset within the industry and the resources available to support monitoring of compliance with best management practice and whole-of-farm nitrogen plans under the Agricultural ERA Standard for sugarcane (State of Queensland 2019a). Lower rates of adoption will reduce the total DIN load reduction that can be achieved through practice change and thus likely necessitate introduction of other DIN reduction mechanisms to deliver the End-of-Catchment Water Quality Objectives in most catchments. As diffuse agricultural sources are the main contributor to DIN load in many Reef catchments (e.g. Queensland Government Department of Environment & Science n.d.), practice change in other agricultural/horticultural sectors (e.g.

⁵⁸ P2R APSIM modelling of yearly DIN losses under varying weather between 1987-2013 suggests that production risk in 1-in-10 'good years' and 'bad years' occurs mainly on todd and mari soils and only occurs for very small numbers of management units in each catchment. *Perceived*, as opposed to *actual*, production risk is, however, likely to influence landholders' minimum compensation requirements in reality. The accuracy of model-based predictions usually reduces when operating towards the extremes of a model's calibration space. Further discussions with the P2R APSIM modelling team would be useful to clarify the reliability of 2016 P2R APSIM's predictions of extreme upper and lower yields.

bananas) could be considered. Alternatively, land use change to reduce DIN losses could also be considered (see Chapter 6).

Secondly, in common with other studies in the literature (e.g. van Grieken et al., 2019), a homogeneous transport coefficient of 60% to End-of-Catchment is assumed for DIN lost through the groundwater pathway. This has a very considerable influence on the load reduction delivered to End-of-Catchment by all steps of practice change. As the proportion of DIN lost to drainage reduces slightly as fertiliser practice improves⁵⁹, the influence of this assumption for DIN transport via ground water is particularly strong for the early stages of fertiliser practice improvement e.g. Df to Cp or Cf. If DIN transport to End-of-Catchment via the ground water pathway is lower than 60%, the total DIN load reduction delivered via practice change under the Agricultural ERA Standard for sugarcane will be less than that predicted here. Again, this would likely necessitate use of additional DIN reduction approaches to achieve catchment-specific Water Quality Objectives. Conversely, if DIN transport via ground water is higher than 60%, higher load reductions than those predicted here could potentially be achievable under the Agricultural ERA Standard for sugarcane (assuming full adoption of relevant practices). *Additional research into the level of and spatial variation in DIN transport via ground water is suggested as an important priority for future research.*

A shortcoming of this Chapter's approach to predicting supply of DIN credits from practice change in fertiliser management is that it focuses solely on financial compensation as the driver of landholders' decision making. An increasing body of literature emphasises the importance of non-financial factors as key drivers of landholders' participation in voluntary mechanisms to supply a range of different public good outcomes from changes in land management and land use on their farms (Brown et al., n.d.; McGurk et al., 2020; Pannell and Zilberman, 2020 provide recent summaries and insights). This literature reveals that considerable heterogeneity in attitudes, preferences and psychosocial characteristics amongst landholders appear to influence willingness to participate in voluntary government-funded schemes to incentivise environmentally beneficial changes in land management (e.g. Christensen et al., 2011; Comerford, 2014; Greiner and Gregg, 2011). Details of scheme design, in addition to the level of financial compensation on offer, have also been shown to influence uptake rate (e.g. Beharry-Borg et al., 2013; Greiner, 2016). Rolfe et al's review of the underlying causes of low uptake rate in conservation tenders acknowledges the interplay between compensation rates, scheme design and underlying attitudinal and psychosocial predispositions (Rolfe et al. 2018a). Rolfe et al. also point out that participation (or not) in such schemes is not a single decision, but rather the outcome of a set of simultaneous decisions by the landholder regarding whether they are minded to change management practice, whether they choose to engage with a tender-based mechanism to do this, and the price level they set in their offer to participate. As the decision to submit an offer to supply nitrogen credits to a water quality trading market shares many of the characteristics of participating in conservation tenders, it is likely that participation rates in nitrogen credit supply will also be influenced by multiple financial and non-financial factors. A Q-Method study undertaken as part of Project 3.1.6 provides initial

⁵⁹ For example, for management units in the Johnstone catchment, the mean proportion of total DIN lost to drainage comprises more than 90% for fertiliser management practices Df, Cp, Cf and Bp; dropping to 87% and 86% for practices Bf and Af, respectively. The minimum proportion of total DIN lost to drainage across all management units in the Johnstone is greater than 80% for practices Df, Cp, Cf and Bp; dropping to 75% and 74% for practices Bf and Af, respectively.

insights into landholders' potential engagement with a nitrogen credit market by exploring discourses around innovative incentive approaches for land management practice change and land use change in the sugarcane sector. The Q-Method study is reported as an Addendum to this Report.

5.4 Summary

This Chapter used a bottom-up approach to produce spatially-specific estimates of sugarcane growers' minimum compensation requirements for changing fertiliser management practice as a mechanism for supplying nitrogen credits to a nitrogen trading market. Compensation requirements were calculated as the sum of four separate cost elements: opportunity cost, the compensation required for exposure to production risk, transition cost, and transaction cost. Catchment-specific supply curves for nitrogen credits were produced for the Mulgrave-Russell, Johnstone, Tully, Murray and Herbert catchments by estimating minimum compensation requirements for different levels of practice change for individual cane land management units. Stochastic simulation was used to randomise allocations of management units to farm sizes and to randomise allocations of baseline nutrient practice to management units in accordance with recently reported data for the catchments as a whole.

The nitrogen credit supply curves produced suggest that full adoption of fertiliser practice change through to practice level Bf should generate considerable DIN reductions at End-of-Catchment at per-unit costs below approximately \$60/kg DIN reduction at EoC. Based on modelling results, it appears unlikely that full adoption of nitrogen practice Bf will deliver the desired Water Quality Targets for the Johnstone and Herbert catchments. Water Quality Targets for the Mulgrave-Russell and Tully catchments may also prove difficult to achieve in full via full adoption of practice Bf. The costs of supplying DIN credits from further practice improvement increase very rapidly to more than \$250/kg DIN EoC. This suggests that, once fertiliser practice has improved to level Bf, other mechanisms such as installation of constructed treatment wetlands and treatment systems should be considered for supply of DIN credits and further reduction of nitrogen loads towards the Water Quality Targets.

6.0 NITROGEN CREDITS FROM LAND USE CHANGE AND NITROGEN TREATMENT SYSTEMS

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6. Summary

This Chapter considers the potential for supplying nitrogen credits from sugarcane catchments in the Wet Tropics through land use change (e.g. setting aside marginally profitable low-lying cane land) and construction of nitrogen treatment systems (e.g. farm-scale constructed treatment wetlands and field-scale bioreactors). The Chapter draws heavily on recently-published findings on the effectiveness of nitrogen removal by natural wetlands in the Wet Tropics (Adame et al., 2019a, 2019b) and new results from the Wet Tropics Major Integrated Project (WT-MIP) on the costs of constructing farm- and field-scale nitrogen treatment systems (Australian Wetlands Consulting 2020).

The minimum DIN credit payment requirement to persuade cane growers to set aside lower cane yield, higher DIN loss land on coom and mari soils was estimated to be approximately \$130/kg DIN reduction at End-of-Catchment, based on opportunity cost.

Based on DIN removal rates from the literature and actual construction cost data from wetlands constructed for the WT-MIP, the minimum DIN credit payment required to cover the total present value cost of constructing farm-scale embellished and landscape wetlands, bioreactor walls and bioreactors in drainage ditches was estimated to be in the range 16 – 58 \$/kgDIN over a period of 10 years. These estimated DIN credit supply costs compare favourably with the minimum cost that is likely to be required to persuade cane growers to improve fertiliser practice from level Bf to level Af. This finding suggests that constructed treatment systems appear to be worthy of further research as a potentially promising mechanism for assisting cost-effective delivery of challenging End-of-Catchment water quality targets for DIN in several major Wet Tropics catchments.

For example, following full adoption of level Bf fertiliser practice on all cane land, if constructed treatment systems deliver DIN load reductions of 0.5 tonnes per hectare per year, as assumed in this Chapter, conversion of *an appropriately situated* 0.2% of cane land in the Mulgrave-Russell to constructed treatment wetlands might be sufficient to deliver the remaining desired DIN load reduction for that catchment, for a total cost of around \$1.5 million. Equivalent calculations for the Johnstone are 0.5% of existing cane area for a cost of around \$3 million, and for the Herbert 0.4% of cane area for a cost of around \$5.5 million.

Project 3.1.6 is grateful to Fernanda Adame, Mark Bayley and Suzette Argent for sharing their expertise, and to Australian Wetlands Consulting and Terrain NRM for agreeing that their emerging results could be included in this Chapter.

Section 6.1 estimates the minimum level of compensation (\$/kgDIN reduced) that would be required to persuade canegrowers to reduce nitrogen losses by setting aside marginally productive land from cane production. Section 6.2 estimates the minimum level of payment for DIN credits (\$/kgDIN removed) that would be required to effectively incentivise installation of on-farm nitrogen treatment systems – specifically, farm-scale constructed treatment wetlands and field-scale bioreactors. Section 6.3 reflects on the potential for on-farm nitrogen treatment systems to deliver meaningful reductions in End-of-Catchment DIN loads in priority Wet Tropics catchments.

6.1 Nitrogen credits from land use change to set aside

Chapter 5 estimated the potential for supplying nitrogen credits through improvements in fertiliser management practice in cane production, across the range of management practice improvements from Df to Af. A further reduction in DIN loss below that achieved by full adoption of practice level Bf could be obtained by setting land aside from cane production entirely. This would only be a feasible proposition if landholders were to receive sufficient compensation to at least cover their foregone gross margin. This compensation could potentially be generated through sale of credits supplied to a DIN credit market.

Van Grieken et al. (2010, Table 2, p.6) calculate a representative gross margin of \$1,292/ha (when expressed in 2018 AUD\$) from cane production in the Wet Tropics under class B fertiliser management. In a companion report, van Grieken et al. quote average cane yields of between 84 to 64 tonnes/ha for four main soil types in the Wet Tropics under management practice B-level fertiliser applications (van Grieken et al., 2010; Table 8, p.6). This matches reasonably well with the P2R APSIM predictions of fresh weight cane yield from individual management units in Wet Tropics catchments reported in Chapter 4 (see Figure 6.1 below), although some management units on Todd soil are predicted to deliver lower yields.

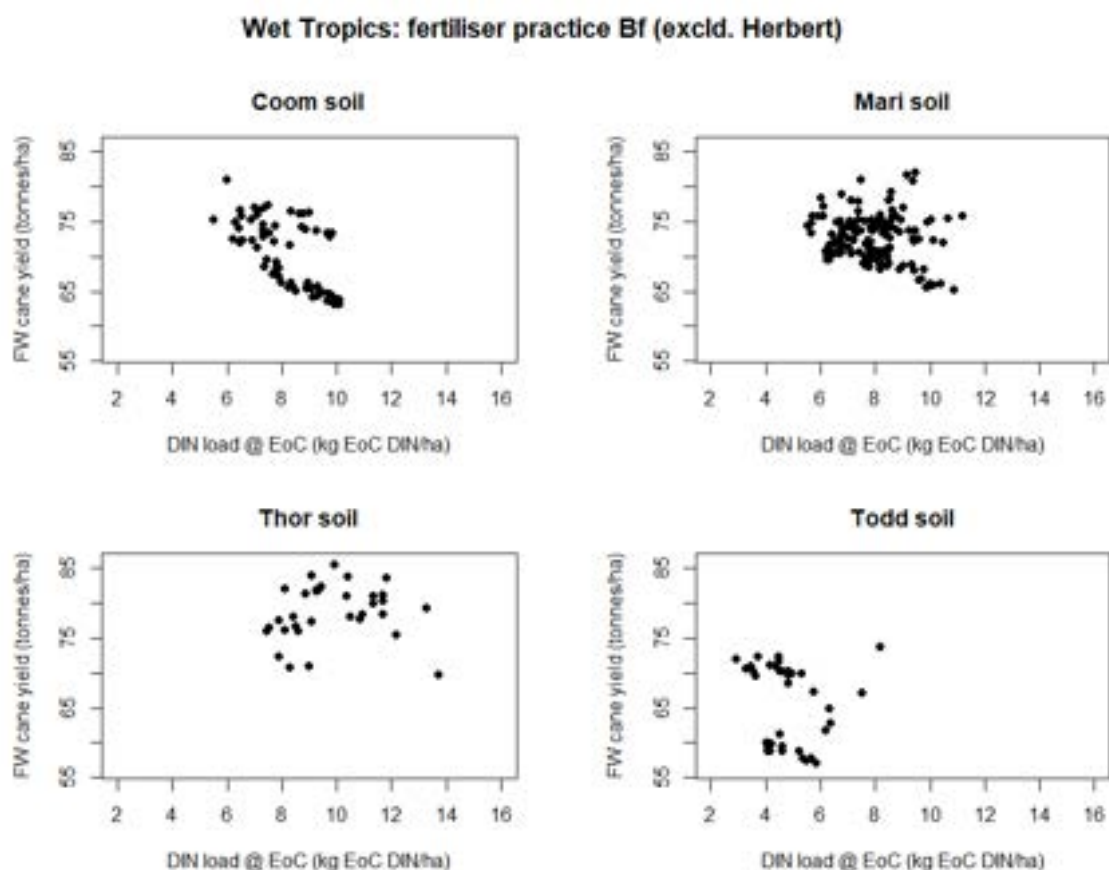


Figure 6.1: Cane yield plotted against End-of-Catchment⁶⁰ DIN load from fertiliser management practice level Bf for individual management units on the four main soil types in Wet Tropics catchments (excluding the Herbert).

Figure 6.1 also shows the associated End-of-Catchment DIN loads from practice level Bf fertiliser applications. Median, 3rd quartile and maximum End-of-Catchment DIN loads and minimum, 1st quartile and median fresh weight cane yields for individual management units on the four soil types are reported in Table 6.1. Management units that return low cane yield⁶¹ but contribute higher End-of-Catchment DIN loads are the most likely candidates for switching to set aside. Some management units on coom and mari soils have these characteristics (Figure 6.1).

⁶⁰ As explained in Chapter 4, End-of-Catchment DIN loads are heavily influenced by the DIN transport coefficient assumed for DIN lost to drainage. Following van Grieken et al. (2019), Project 3.1.6 assumes the transport coefficient to End-of-Catchment for DIN lost to drainage is 0.6.

⁶¹ Reductions in gross margins may not follow linearly with reductions in cane yield, particularly if management units that deliver lower yields also require lower fertiliser applications. For the data plotted in Figure 6.1, mean practice Bf fertiliser applications are significantly lower for coom soils than for mari and thor soils ($T = -11.25$ and -7.10 , both at $p < 0.001$, for mari and thor soils compared to coom soil), so it may not be unreasonable to attribute van Grieken et al's average gross margin to management units on coom as well as to mari soil.

Table 6.1: P2R APSIM predictions of median, 3rd quartile and maximum DIN loads, and median, 1st quartile and minimum fresh weight cane yields at fertiliser management practice level Bf for individual management units on the four main soil types in Wet Tropics catchments (excluding the Herbert).

Soil type	EoC DIN load (kg EoC DIN/ha)			Cane yield (tonnes/ha)		
	median	3 rd quart.	max	min	1 st quart.	median
coom	8.4	9.6	10.1	39.4	40.2	42.4
mari	8.0	8.5	11.1	43.8	47.2	47.9
thor	9.3	11.3	13.7	42.2	47.3	47.9
todd	4.4	5.2	8.2	37.9	39.2	41.5

To provide a conservative estimate of the approximate payment required to cover the foregone gross margin from setting aside these management units on coom or mari soil, assume that the average gross margin of \$1,292/ha quoted by van Grieken et al. (2010, Table 2, p.6) has to be recouped from supply of approximately 10 kg EoC DIN/ha (Table 6.1). The requisite End-of-Catchment DIN credit price is around \$130/kg DIN EoC. This is slightly lower than the \$145/kg DIN EoC reported in Chapter 5, Table 5.5 as the minimum required compensation for changing fertiliser practice from Bf to Af on a representative management unit in the Johnstone catchment. These comparisons are broad-brush, but they do suggest that setting aside low-yielding, high DIN loss cane land is unlikely to provide a low-cost supply of DIN credits unless gross margins from the set aside land are considerably lower than the averages estimated by van Grieken et al. (2010, Table 2, p.6).

6.2 Nitrogen credits from nitrogen treatment systems

In this Chapter, two types of nitrogen treatment systems are considered as potential sources of supply for nitrogen credits: farm-scale constructed treatment wetlands and field-scale denitrifying bioreactors. In their report on nitrogen treatment systems installed during the Wet Tropics Major Integrated Project (MIP), Australian Wetland Consulting refer to these two approaches as ‘vegetated treatment systems’ and ‘woodchip bioreactors’, respectively (Australian Wetlands Consulting 2020).

6.2.1 Nitrogen removal in wetlands

This section draws heavily on information provided by *WetlandInfo* (<https://wetlandinfo.des.qld.gov.au/wetlands/>) and the explanation of nitrogen removal processes provided by Adame et al. (2019b). Natural palustrine and estuarine wetlands⁶² remove nitrogen from incoming water flows through three processes: uptake of nitrogen by plants, accumulation of sediment into soil, and denitrification (Figure 6.2). In denitrification, nitrate (NO₃⁻) is converted to nitrous oxide (N₂O) and nitrogen gas (N₂), transferring nitrogen from the ecosystem to the atmosphere. Denitrification typically provides more than 90% of

⁶² Palustrine wetlands are vegetated non-riverine wetlands such as billabongs, swamps and bogs with more than 30% cover of emergent vegetation such as grasses, sedges or trees. Estuarine wetlands are wetlands in which saltwater is diluted with freshwater runoff from the land. Both definitions from *WetlandInfo* (Queensland Government Department of Environment & Science 2020).

nitrogen removal (Adame et al., 2020; Adame et al., 2019a). Conversion of nitrite (NO_2) and ammonium (NH_4^+) to nitrogen gas (N_2) via the anammox pathway can also remove nitrogen from wetland systems, and may play a significant role in nitrogen removal in estuarine wetlands with low oxygen concentration (Zhou et al. 2014).

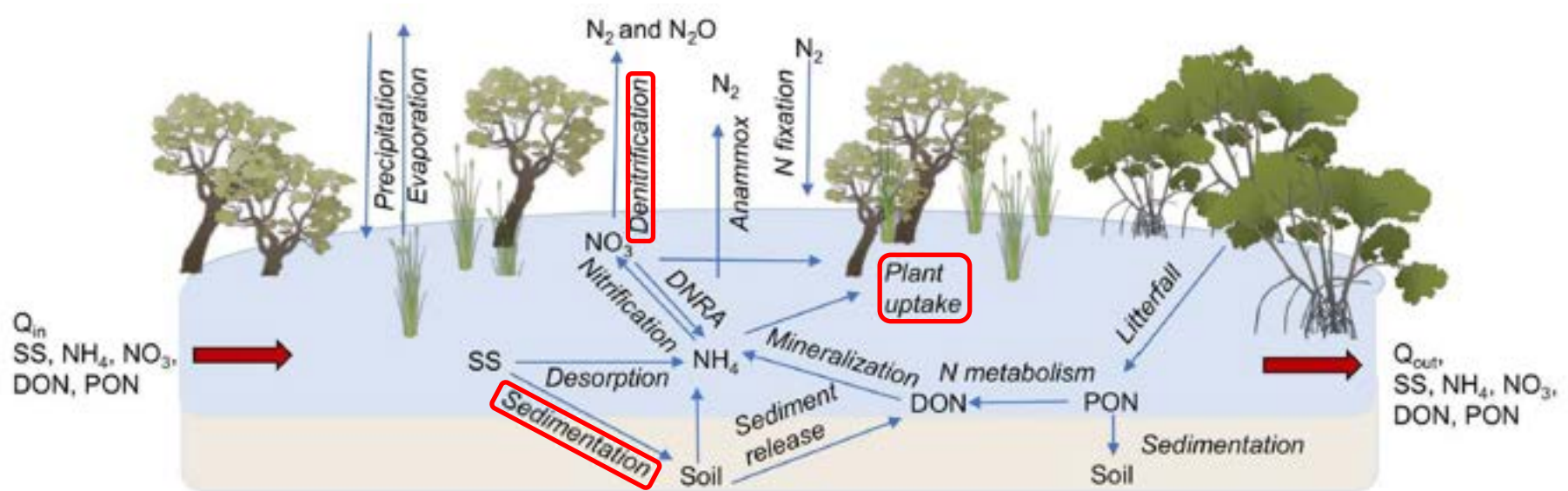


Figure 6.2: Conceptual model of nitrogen fluxes during a flood in estuarine and palustrine wetlands in tropical Australia. Nitrogen removal processes are highlighted in red. Copied from Figure 1, p.2 in Adame et al. (2019b).

Denitrifying bacteria occur in soils and water, and denitrification rates are typically highest in situations where oxygen levels are low and organic carbon levels are high – as is often the case in wetlands (Kulkarni et al. 2008). In addition to high levels of soil carbon, high rates of denitrification in tropical wetlands require high nitrate concentrations in the incoming water flow (Adame et al., 2019a). Palustrine and estuarine wetlands in Wet Tropics catchments have been shown to be capable of removing high nitrogen loads when inundated for several days with floodwaters carrying high nitrate concentrations (mean 0.06 mg L^{-1}) in runoff from cane land (Adame et al., 2019a). Mean total denitrification rates of $14 \text{ mg m}^{-2} \text{ hr}^{-1}$ were reported, with a maximum total denitrification rate of $52 \text{ mg m}^{-2} \text{ hr}^{-1}$. Denitrification rates were significantly correlated with nitrate concentration in the water in two related studies of coastal and floodplain wetlands in the Wet Tropics (Adame et al., 2019a, 2019b). In both studies, high nitrate concentrations in incoming water were very likely to be associated with fertiliser runoff from cane paddocks.

Farm-scale treatment wetlands are constructed with the aim of applying similar denitrification rates to high-concentration nitrate run-off from a farm's cane paddocks. The location of treatment wetlands in the landscape must be considered carefully to provide sufficient hydraulic retention time to remove substantial nitrogen loads, even allowing for their high potential denitrification rates. The cost-effectiveness of DIN removal by the farm-scale treatment wetlands that Australian Wetlands Consulting designed and implemented for the Wet Tropics MIP is assessed in Section 6.2.3.

6.2.2 Nitrogen removal in bioreactors

Bioreactors aim to remove nitrate in the runoff from agricultural paddocks by directing that runoff through a mass of woodchips (or another form of carbon substrate) in which microbes can denitrify the incoming nitrate to nitrogen gas (Schipper et al. 2010). Two commonly used configurations are bioreactor walls and bioreactor beds. Bioreactor walls are situated appropriately to allow a 'wall' of buried woodchips to intercept the groundwater flow path from an agricultural paddock. Bioreactor walls rely on the presence of an impervious substrate to direct groundwater flow through the woodchip wall (Figure 6.3). Bioreactor beds insert a subsurface 'bed' of woodchips in-line with existing drainage before the drain discharges to receiving waters (Figure 6.3).

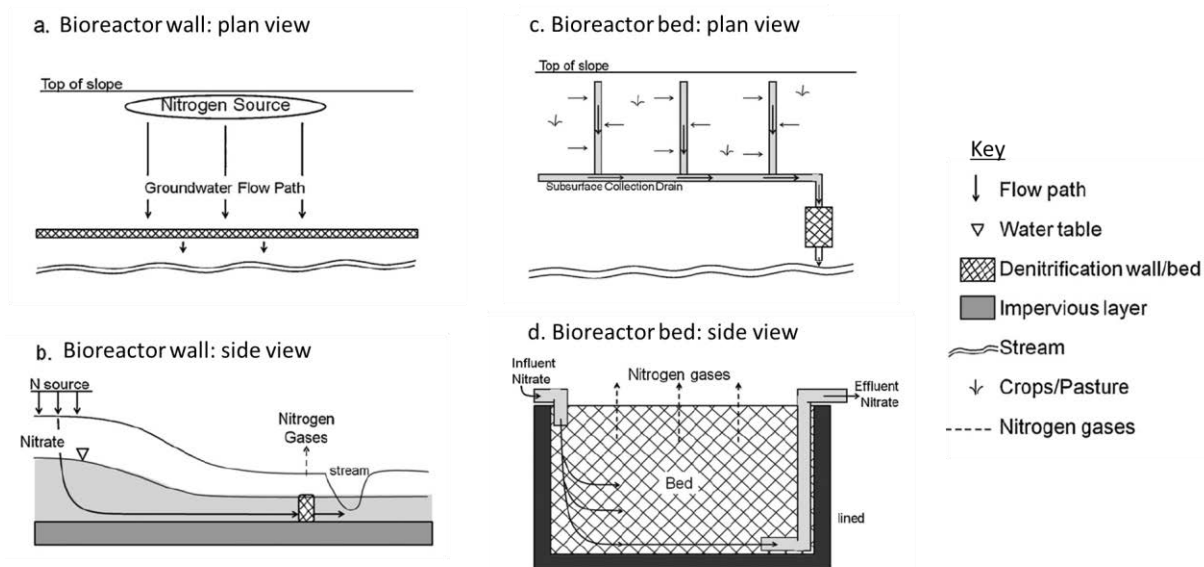


Figure 6.3: Conceptual configurations for bioreactor walls (a) and (b), and bioreactor beds (c) and (d).
Copied from Fig. 1 (p. 1534) and Fig. 2 (p.1535) in Schipper et al. (2010).

Schipper et al's review reports denitrification rates for bioreactor walls and beds of between 0.6 and 12.7 g NO₃-N m⁻³ day⁻¹ (Schipper et al., 2010; Table 3, p.1537), although these are all for installations operating at lower temperatures than would be expected for the Wet Tropics (9°C to 20°C, but with the majority at or below 15°C). A meta-analysis by Addy et al. of results from 57 field and laboratory bioreactor studies (reported in 26 different publications) found that hydraulic retention time and temperature both significantly affected denitrification rate (Addy et al. 2016). The meta-analysis also identified that nitrogen removal rates were significantly higher in bioreactor beds than in bioreactor walls. This may reflect that bed configurations can potentially be more effective at intercepting runoff.

Australian Wetlands Consulting designed and implemented various forms of bioreactor beds and walls for the Wet Tropics MIP. The cost-effectiveness of these installations for DIN removal is estimated in Section 6.2.3.

6.2.3 Cost-effectiveness of constructed treatment systems

Constructed Treatment Wetlands

The constructed treatment wetlands that Australian Wetlands Consulting designed and implemented for the Wet Tropics MIP took various forms: a *fully constructed wetland* (a formal constructed wetland involving large-scale earthworks and vegetation plantings), *landscape wetlands* (re-instated natural hydrology to create a wetland system), *embellished wetlands* (minor modification to enhance water flow through existing drainage areas on farms), and *in-drain wetlands* or '*vegetated drains*' (cessation of herbicide spraying and clearing of drains to increase native vegetation cover) (Australian Wetland Consulting, 2020; Section 2.1 p.3). Characteristics of the constructed treatment wetland installations are summarised in Section 6.2.1.

Bioreactors

Australian Wetlands Consulting designed and implemented the following types of bioreactors for the Wet Tropics MIP: standard *bioreactor walls*; standard *bioreactor beds* (both as illustrated in Figure 6.3); *in-drain bioreactor beds* (the base of an existing drain is excavated to a square cross-section, the lower half of the drain is filled with woodchip; the woodchip layer is topped with 200-400mm of rock); *agricultural pipe drain bioreactor trench* (the trench dug for an agricultural drainage pipe is back-filled with woodchip for the initial 400mm of fill) (Australian Wetlands Consulting, 2020; Section 2.2 p.5).

Denitrification rates

Dr Mark Bayley from Australian Wetlands Consulting applied the following denitrification rates to calculate the DIN load removed by the constructed wetlands and bioreactor installations for the Wet Tropics MIP. Based on Kadlec and Wallace (2009; Table 9.37, p.335), a DIN removal rates of 550 kg DIN ha⁻¹ yr⁻¹ was assumed for the landscape wetland, and removal rates of 750 kg DIN ha⁻¹ yr⁻¹ were assumed for the formal constructed wetland and embellished wetlands and the vegetated drains⁶³. A DIN removal rate of 6000 kg DIN ha⁻¹ yr⁻¹ was assumed for the bioreactors, based on Robertson (2010)⁶⁴.

Cost-effectiveness assessment

The cost-effectiveness of nitrogen treatment systems implemented for the Wet Tropics MIP was assessed using data provided on system design and implementation by Mark Bayley of Australian Wetlands Consulting and Suzette Argent, Catchment Repair Project Officer for the Wet Tropics MIP at Terrain NRM. The cost effectiveness metric reports the annualised present value cost of the installation per kilogram of DIN reduction (i.e. \$/year per kg DIN/year, or, equivalently, \$/ kg DIN).

For each treatment system, the total present value costs of construction and maintenance are calculated over a 10-year period, using a 7% annual discount rate. Construction costs include: design, earthworks and vegetation planting (earthworks and planting costs were only incurred for some systems). Maintenance costs are assumed to accrue annually, with eight maintenance visits per year (each costed at \$45/hour for 8 hours for three personnel, plus \$250 materials and operating costs: totalling \$1,330 per visit) required for the 8.5ha landscape wetland, 6 visits per year for the 1.2ha formal constructed wetland and the 1.4ha embellished wetlands, and 4 visits per year for the 0.3ha embellished wetland. One lower-intensity maintenance visit per year (costed at \$270) and a three-yearly higher intensity maintenance visit (costed at \$1,200) is assumed for the in-drain bioreactor beds and for bioreactor beds embedded around agricultural drainage pipe. An annual opportunity cost is included for the 8.5ha landscape wetland (on cane land) and the 1.2ha formal constructed wetland (on a banana farm). Opportunity costs are estimated using gross margins from van Grieken et

⁶³ These rates apply the 50th and 60th percentiles of the denitrification rate distribution reported in Table 9.37 p.335 of Kadlec and Wallace (2009).

⁶⁴ Lower end of reported rates (1-20g Nitrate N removed per m³ media per day) @ 6.5g DIN m³ media per day, with 1 m³ of media translated to effective 0.5m² of filter area (accounting for dead zones and inlet/outlet structures) at a reduced efficacy of 50% due to the low inlet DIN concentrations anticipated.

al.(2010 Table 11 p.7 and Table 24 p.15 - updated to 2019 AUD\$). The embellished wetland sites were not used for agricultural production, so no opportunity cost is included for these installations. A negative opportunity cost, i.e. a cost saving, of \$175 per year was assumed for the vegetated drains. This reflects the reduction in herbicide spraying and drain clearing effort.

Maintenance costs are discounted at discount rate r over a project lifespan, N , and added to the construction cost to arrive at the total present value cost (TPVC) of wetland or bioreactor j :

$$TPVC \text{ of wetland or bioreactor } j = C + \sum_{n=1}^N \frac{M_n}{(1+r)^n} \quad (6.1)$$

where:

C = one-off construction cost

M_n = future maintenance costs incurred post-construction in years n (for $n = 1, 2, \dots, N$)

r = annual real discount rate

The discount rate used is 7% (Rust and Star 2018; Star et al. 2018; Alluvium 2019), and an operational lifetime of 10 years is assumed.

Converting a total present value cost in absolute \$ into an annualised equivalent present value cost in \$/year enables this annualised cost to be used alongside estimates the yearly DIN load reductions delivered by the treatment systems. This produces a consistent cost-effectiveness metric (in \$ per kg DIN removed) that can be used to compare the cost-effectiveness of these wetlands and bioreactors with the cost-effectiveness of DIN credits supplied from fertiliser practice change or setting land aside.

The following formula is used to convert total present value cost (TPVC, \$) to equivalent annualised present value cost (APVC, \$/year):

$$APVC = TPVC \left[\frac{r}{1 - (1+r)^{-N}} \right] \quad (6.2)$$

where symbols are as defined for Equation (6.1).

Table 6.2: Areas, assumed denitrification rates, costs and cost-effectiveness of constructed treatment wetlands and bioreactors installed for the Wet Tropics MIP.
Data provided by Australian Wetland Consulting (Mark Bayley) and Terrain NRM (Suzette Argent).

Treatment system	Area		DIN reduction rate (kg ha ⁻¹ yr ⁻¹)	Annual DIN reduction (kg)	Total PV cost (\$)	Annualised cost (\$)	Cost- effectiveness (\$/kg DIN)
	m ²	ha					
Landscape wetland	85,000	8.5	550	4,675	532,548	75,954	16
Formal Constructed wetland	12,000	1.2	750	900	806,334	115,002	128
Large embellished wetland	14,000	1.4	750	1,050	208,951	29,801	28
Medium embellished wetland	3,000	0.3	750	225	52,866	7,540	34
Vegetated drain (no earthworks)	350	0.035	750	26.25	1,273	182	7
Vegetated drain (weirs installed)	625	0.0625	750	46.88	58,630	8,362	178
Vegetated drain (bench developed)	1760	0.176	750	132.0	38,043	5,426	41
Bioreactor wall	80	0.008	6000	48	19,319	2,755	57
Bioreactor bed: in drain	10	0.001	6000	6	20,588	2,936	489
Bioreactor bed: in drain	15	0.0015	6000	9	26,375	3,762	418
Bioreactor bed: in drain	10	0.001	6000	6	20,207	2,882	480
Bioreactor: drainage ditch	80	0.008	6000	48	19,543	2,787	58
Bioreactor: offline ag pipe	15	0.0015	6000	9	15,680	2,236	248
Bioreactor: offline ag pipe	15	0.0015	6000	9	15,630	2,229	248

At the denitrification rates assumed in Table 6.2, the landscape and embellished wetlands, and vegetated drains where construction costs are modest, offer DIN reductions in the range 16 – 58 \$/kg DIN. This cost-effectiveness is reported at the wetland site, not at End-of-Catchment; however, if, as is likely, there is a surface water connection from the wetland to End-of-Catchment then the DIN transport coefficient to End-of-Catchment for farm-scale wetlands in the Wet Tropics will be close to one. In these circumstances, End-of-Catchment cost-effectiveness will be close to the cost-effectiveness at the treatment installation. Initial results from Table 6.2 suggest that these constructed treatment systems are potentially more cost-effective sources of DIN credits than nitrogen practice improvement in cane beyond practice level Bf, or from setting aside low-productivity cane land (Section 6.2, this Chapter). Water samples from the Wet Tropics MIP wetlands are currently being analysed to determine empirical denitrification rates. It will be important to re-visit this cost-effectiveness analysis once empirical results are available⁶⁵.

For comparison, the mean denitrification rate of 14 mg m⁻² hr⁻¹ reported for natural coastal and floodplain wetlands by Adame et al. (2019a) indicates that 1ha of these natural wetlands would deliver a DIN load reduction of 550 kg over a period of 164 days (i.e. 5.5 months), or 750kg over a period of 223 days (7.4 months). These periods would be reduced by about a factor of 4 (i.e. to between one and two months) at the maximum denitrification rate reported by Adame et al. for the natural wetlands. Multiple authors have reported that denitrification rate in wetlands is closely associated with nitrate concentrations in the influent water (Adame et al., 2019b; Piña-Ochoa and Álvarez-Cobelas, 2006; Zhou et al., 2014). Bulk nitrogen load removal requires an appropriate combination of denitrification rate and hydraulic retention time. Both key factors in removal of high nitrogen loads – influent nitrate concentration and hydraulic retention time – are strongly affected by the siting of the treatment wetland within the topography and hydrology of the farm landscape. If existing wet scrapes and sumps that receive runoff and/or groundwater drainage from surrounding cane paddocks can be identified as sites for embellishment, and if the volume of these embellished wetlands is adequate to ensure that they remain wet through continuing seepage of groundwater for several months of the year, then it seems likely that these systems could make a meaningful contribution to denitrification and provide a useful source of supply for DIN credits at moderate cost⁶⁶.

The results in Table 6.2 suggest that the bioreactor wall and the bioreactor in the drainage ditch can deliver DIN credits at similar cost levels to landscape and embellished wetlands and vegetated drains. Initial results suggest that bioreactor beds are less cost-effective. Further research is required to explore the reasons for this discrepancy in performance, particularly as the meta-analysis by Addy et al. found denitrification rates to be higher for bioreactors beds than for bioreactor walls (Addy et al. 2016).

⁶⁵ Several of the constructed wetlands were not fully vegetated when water samples were taken for analysis (Mark Bayley pers. comm.). It is likely that denitrification rates in these wetlands will increase once their vegetation is fully established.

⁶⁶ An important caveat here is that this analysis assumes that the retention time of the constructed treatment wetland is adequate to allow effective denitrification of incoming nutrient loads. This may not be the case if the wetland occupies a very small proportion of its contributing catchment and/or if the majority of DIN runoff is concentrated into a single large rainfall event post fertiliser application. The levels of denitrification performance and cost-effectiveness suggested here assume that the constructed treatment wetland has been located and designed appropriately within the farm landscape. This will not be possible in all settings.

6.3 Potential for DIN credit supply from constructed nitrogen treatment systems

The stochastic simulations of Chapter 5 suggested that full adoption of the levels of nitrogen practice that are likely to be required to comply with whole-of-farm nitrogen budgets as specified in the Agricultural ERA Standard for sugarcane will be unlikely to deliver the desired End-of-Catchment DIN load targets for several Wet Tropics catchments. The supply curves of Chapter 5: Figure 5.5, Figure 5.6, and Figure 5.9 suggest that target shortfalls for the Mulgrave-Russell, Johnstone and Herbert catchments could be of the order of 30 tonnes, 60 tonnes and 110 tonnes DIN at End-of-Catchment, respectively. If constructed treatment systems can deliver DIN load reductions of between 0.5 and 0.75 tonnes per hectare per year, as assumed in Table 6.2, then 40 to 60 hectares of such systems could potentially cover the water quality target shortfall for DIN in the Mulgrave-Russell, 80 to 120 hectares in the Johnstone, and 150 to 220 hectares in the Herbert. The QLUMP spatial mapping layer indicates that there is around 24,400 hectares of cane land in the Mulgrave-Russell, 25,200 hectares in the Johnstone, and 60,200 hectares in the Herbert. Taking the denitrification rate of 0.5 tonnes per hectare per year, the comparative areas indicate that conversion of *an appropriately situated* 0.2% of cane land in the Mulgrave-Russell to constructed treatment wetlands might be sufficient to deliver the desired DIN load reduction for that catchment, for a total cost of around \$1.5 million. Equivalent calculations for the Johnstone are 0.5% of existing cane area for a cost of around \$3 million, and for the Herbert 0.4% of cane area for a cost of around \$5.5 million. These are simplistic predictions, based on actual construction costings and assumed denitrification rates; they should also be re-visited once longer-term denitrification rates have been verified empirically. They do, however, suggest that constructed treatment systems appear to be worthy of further research and investigation as a potentially promising mechanism for assisting cost-effective delivery of challenging End-of-Catchment water quality targets for DIN in several major Wet Tropics catchments.

7.0 DEMAND FOR NITROGEN CREDITS

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7. Summary

To provide a context for estimating potential demand for nitrogen credits, this Chapter summarises data collated from the Environmental Authorities Register (Sudarjanto et al. 2017) on licenced nitrogen discharge loads from point-source nitrogen emitters (principally sewage treatment plants (STPs) and aquaculture facilities) operating under environmental authorities (EAs) in Wet Tropics catchments. In total, STPs are licenced to contribute 74%, 14%, 12% and 9% of total anthropogenic DIN load in the Barron, Mossman, Mulgrave-Russell and Tully catchments, respectively. STP licenced contributions to anthropogenic DIN load in other Wet Tropics catchments are considerably lower. Aquaculture facilities are licenced to contribute 86%, 35% and 9% of the total anthropogenic DIN load in the Mossman, Murray and Johnstone catchments, respectively.

Two scenarios are explored as the likely sources of demand for nitrogen credits from STPs under the Queensland Government's Point Source Water Quality Offsets Policy: (i) 'long term offsetting' to avoid upgrading treatment to 5mgN/L discharge performance in small, rural STPs, and (ii) 'short term offsetting' to defer capacity augmentation of larger, urban, STPs in the face of population expansion. Catchment-specific demand curves for N-credits are estimated using (i) data from the Queensland Water Directorate (hereafter qldwater) on recent discharge loads and volumes to calculate STPs' discharge concentrations (qldwater 2017, 2018), and (ii) estimates of STPs' maximum willingness to pay for nitrogen credits from avoided the expenditures on capex and opex that would otherwise be incurred in upgrading treatment or expanding capacity.

This analysis suggests that in the Wet Tropics, modest 'long term offsetting' demand for nitrogen credits at prices that should be sufficient to elicit diffuse source credit supply from constructed nitrogen treatment systems potentially exists from STPs in the Mulgrave-Russell (16 tonnes of annual N-credit demand from Gordonvale and Babinda STPs in combination), Mossman (15 tonnes annually from Mossman STP), and Herbert (33 tonnes annually from Hinchinbrook Shire Council's STPs) catchments. Elsewhere along the Reef coastline, annual demand for long term offsetting could arise in Townsville (17 tonnes: Ross catchment), Mackay (35 tonnes: Plane & Pioneer catchments), Rockhampton (165 tonnes: Fitzroy catchment), and Bundaberg (25 tonnes: Burnett catchment) (qldwater 2017, 2018).

Population growth in major urban centres along the Reef coast has slowed or reversed plateaued (stat.data.abs.gov.au). This suggests that although short term offsetting to avoid capacity augmentation at large urban STPs could generate adequate willingness to pay to elicit credit supply, there is unlikely to be credit demand from this mechanism until population growth rates in the region's urban centres increase again.

7.1 Point source nitrogen emitters in Wet Tropics catchments

Information pertaining to all environmental authorities (EAs) that have been granted in Queensland is publicly available from Environmental Authorities Register website (<https://environment.ehp.qld.gov.au/env-authorities/>). Using information collated from this public register of EAs, Sudarjanto et al., (2017) created a digital and spatial database containing EA approval conditions for point sources and used available information to calculate total licenced nitrogen (TN) and total phosphorus (TP) loads from all point source emitters in Wet Tropics catchments. Estimates of TN and TP loads are only provided for point sources for which relevant information is available to enable such calculations. The Environmental Authorities Register point source database indicates that there are 51 EA-licenced discharge points that release to water in Wet Tropics catchments. These discharge points are associated with the following sectors:

- 24 aquaculture discharge points
- 23 sewage treatment plant (STP) discharge points
- 3 discharge points from mining
- 1 discharge point from seafood processing

Of the 51 discharge points identified in Wet Tropics catchments, total nitrogen load estimates (kgN/year) are available for 39 facilities or sites across the Mossman, Barron, Mulgrave-Russell, Johnstone, Tully, Murray, and Herbert catchments (Table 7.1). Information on the licenced nutrient concentration limits and/or on release flows were not included in the EA documentation for the remaining 12 sites (Sudarjanto et al. 2017). One of the seven STPs listed in the Environmental Authorities Register database as being in the Barron catchment has a release point that the *WetlandInfo*⁶⁷ website indicates to be in the Mitchell catchment, and therefore flows to the Gulf of Carpentaria rather than the GBR. Thus, Table 7.1 lists 6 EA-licenced STPs for the Barron catchment.

Figure 7.1 shows the locations of EA-licenced aquaculture and STP facilities in the Wet Tropics. If an EA holder is permitted to release nitrogen at several discharge points, each release point will have been recorded separately in the Environmental Authorities Register database. In these circumstances, it may be the case that only one of the licence discharge points is in use. For example, there are three discharge points associated with Innisfail STP in the Johnstone catchment, but a TN load estimate is only provided for one discharge point (Ninds Creek Outfall). The Cassowary Coast Regional Council report that Innisfail STP discharges to Ninds Creek (Cassowary Coast Regional Council, 2019; p13), suggesting that the other EA-licenced discharge points are currently not in use. Consequently, all relevant information regarding the total nitrogen load discharged from Innisfail STP has been captured in the database.

⁶⁷ <https://wetlandinfo.des.qld.gov.au/wetlands/assessment/monitoring/point-source-release/sewage-treatment-facilities/>

Table 7.1: Distribution of licenced discharge points across catchments in the Wet Tropics, as listed in the Environmental Authorities Register database. Numbers in brackets indicate the number of discharge points for which total nitrogen load estimates are available (data source: Sudarjanto et al. (2017)).

Wet Tropics Catchment	Number of discharge points				
	Aquaculture	STP	Mining	Seafood processing	Total
Daintree	0	0	0	0	0
Mossman	2 (2)	3 (2)	0	0	5 (4)
Barron	1 (1)	6 (6)	0	0	7 (7)
Mulgrave-Russell	4 (4)	5 (2)	0	0	9 (6)
Johnstone	14 (14)	3 (1)	0	0	17 (15)
Tully	0	1 (1)	0	0	1 (1)
Murray	3 (3)	1 (1)	0	1 (0)	5 (4)
Herbert	0	3 (1)	3 (0)	0	6 (1)
Total	24 (24)	23 (15)	3 (0)	1 (0)	50 (38)

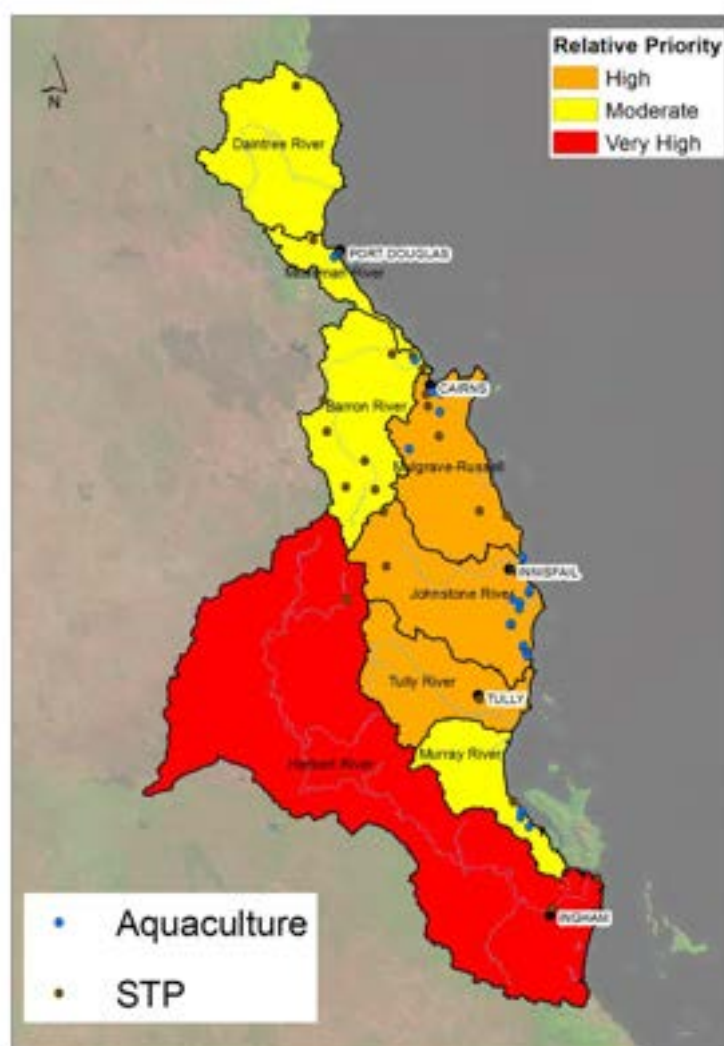


Figure 7.1: Locations of EA-licensed sewage treatment plants and aquaculture facilities in Wet Tropics catchments, as listed in the Environmental Authorities Register database. Colours indicate relative priority for nitrogen reduction in the relevant catchment. Data sourced from Sudarjanto et al. (2017).

7.1.1 Sewage treatment plants: licensed loads

The TN discharge loads for STPs located in Wet Tropics catchments, estimated from licence information (where available) in the Environmental Authorities Register database are summarised in Table 7.2. Following Sudarjanto et al. (2017; p.9) and data from a 2003 report on the chemical constituents of effluence discharged by the Ayr STP (Willett et al. 2003; Figure 4, p.8) we assume that the majority of the STP loads are dissolved inorganic nutrient (DIN), Table 7.2 lists licenced STP nitrogen loads as a percentage of the modelled DIN anthropogenic baseline loads in 2012-2013 for each Wet Tropics catchment (Brodie et al., 2017; Table 10 on p.33-34).

Based on the figures in Table 7.2, STPs in the Barron are licenced to contribute almost three quarters of the catchment's total anthropogenic DIN load. Two of the STPs in the lower Barron catchment (Cairns Northern STP and Marlin STP) are large plants that process sewage from the northern suburbs of Cairns. Sewage from southern Cairns suburbs is processed by two large STPs (Cairns Southern STP and Edmonton STP) in the Mulgrave-Russell catchment. Table 7.2 shows that STP nitrogen loads for the Barron and Mulgrave-Russell catchments are of similar magnitude. The STP load in the Barron constitutes a much larger proportion of the total DIN load from the catchment, however, because the Barron contains a much smaller area of sugarcane than the Mulgrave-Russell (Queensland Government Department of Environment & Science, n.d.). In contrast, licenced STP contributions to anthropogenic DIN loads in the Murray and Herbert catchments are well under 1%. This is because the Murray contains no large towns, and STP load from the town of Ingham in the Herbert is small compared with the very large area of sugarcane in this catchment (Queensland Government Department of Environment & Science, n.d.).

Table 7.2: Estimates of licenced total nitrogen (TN) annual load (kgN/year) released to water from sewage treatment plants in Wet Tropics catchments, calculated using information in the Environmental Authorities Register database (data source: Sudarjanto et al. 2017).

Catchment	Number of STPs	Licenced TN load discharged to water (kgN/year)	Licenced TN discharge concentration (mgN/L)	Anthropogenic baseline DIN load (tonnes/year) [‡]	Licenced STP loads as a percentage of anthropogenic DIN load
		TN _{STP}		DIN _{Anthropogenic}	% TN _{STP} to DIN _{Anthropogenic}
Daintree	0	0	-	135	0%
Mossman	2	14,600	10, 5	104	14%
Barron	6	64,082	5	87	74%
Mulgrave-Russell	2	51,100	5	423	12.1%
Johnstone	1	7,300	5	499	1.5%
Tully	1	4,563	5	50	9.1%
Murray	1	251	5	232	0.1%
Herbert	1	1,734	5	886	0.2%
Total	14	143,630		2,750	5.2%

[‡] Figures are based on modelled dissolved inorganic nitrogen (DIN) anthropogenic baseline load in 2012-2013 as reported in (Brodie et al., 2017, Table 10 on p.33-34).

7.1.2 Sewage treatment plants: reported loads

Section 7.1.1 reported the total licenced nitrogen loads from STPs in Wet Tropics catchments, as held in the Environmental Authorities Register database. Useful additional data on reported STP discharge volumes and loads are provided by qldwater (the central advisory and advocacy body for Queensland's urban water industry <https://www.qldwater.com.au>). Qldwater collate operating and reporting data from Queensland's sewage service providers via their Statewide Water Information Management (SWIM) project. In their Interim Report on estimation of point source loads from individual STPs in GBR catchments, qldwater provide SWIM data on total nitrogen discharge loads (kgN) and annual discharge volumes (ML) from STPs in catchments that drain to the GBR lagoon (qldwater, 2018; Appendices 3 & 4). STP-specific discharge load and volume data allow average annual total nitrogen discharge concentrations (mgN/L) to be calculated for individual STPs. A summary of discharge loads, volumes and total nitrogen (TN) concentrations from STPs in Wet Tropics catchments, derived from qldwater SWIM data (qldwater, 2018; Appendices 3 & 4) is provided in Table 7.3.

As noted by qldwater (2017; Section 3. p.4), citing Brodie (1994), Waterhouse & Johnston (2002), and GBRMPA (2005), although there is no specific definition of the TN discharge concentration expected from tertiary-level treatment, prior practice in GBR catchments indicates that a TN discharge concentration of 5 mgN/L appears to be the de facto expectation. The qldwater SWIM data show that, with the exception of Rockhampton, all large STPs from major cities along the Reef coast already operate with discharge concentrations of around 5mgN/L (qldwater, 2018; Appendices 3 & 4). Qldwater argue that it would be unjustifiably costly to upgrade all smaller STPs in GBR catchments to the stringent 5mgN/L discharge concentration limit (qldwater 2017). Table 7.3 shows that although most STPs in Wet Tropic catchments are already operating at 5mgN/L, four of the small/medium STPs do not currently deliver this level of performance.

Table 7.3: Discharges of total nitrogen (TN) to water from sewage treatment plants (STPs) in Wet Tropics catchments for Financial Year 2016-17, as reported to qldwater (2018; Appendices 3 & 4). Rows coloured blue indicated STPs for Cairns city and suburbs. Rows coloured grey indicate STPs for which offsetting would be required to reduce TN load emissions to an average annual concentration of 5mg/L.

Catchment	STP	Volume treated (ML/year)	Volume discharged to water (ML/year)	TN load discharged to water (kg/year)	TN average discharge conc. to water (mg/L) [‡]	Licenced TN discharge load (kg/year) ^{‡‡}	Offsetting requirement: assuming 5mg/L TN conc. limit (kg/year)
Daintree	-	-	-	-	-	-	-
Mossman	Mossman	405	405	12144	30	10402	10119
Mossman	Port Douglas	1402	532	2661	5	4198	0
Barron	Atherton	678	544	2721	5	4015	0
Barron	Cairns Northern	6731	6664	33320	5	41245	0
Barron	Kuranda	84	84	419	5	1259	0
Barron	Marlin	2605	2016	10080	5	16608	0
Barron	Tinaroo Town	26	26	131	5	347	0
Barron	Yungaburra	92	0	0	0	608	0
Mulgrave-Russell	Babinda	395	292	8760	30	-	7300
Mulgrave-Russell	Cairns Southern	7106	7088	35438	5	38325	0
Mulgrave-Russell	Edmonton	1812	1725	8625	5	12775	0
Mulgrave-Russell	Gordonvale	609	700	6999	10	-	3500
Johnstone	Innisfail	2950	2950	14749	5	14600	0
Johnstone	Malanda	93	0	0	0	-	0
Tully	Tully	860	860	4302	5	4563	0
Murray	-	-	-	-	-	-	-
Herbert	Hinchinbrook*	1119	893	26787	30	-	22325
Herbert	Ravenshoe	160	160	799	5	1734	0

[‡] Calculated from TN load discharged to water (kg/year) divided by volume discharged to water (ML/year). Both data elements from qldwater (2018; App 3&4).

^{‡‡} Data from the GBR point source database (Sudarjanto et al. 2017).

* Hinchinbrook Shire Council reported combined data for the Ingham and Lucinda STPs. Calculated offsetting requirement applies to the combination.

7.1.3 Aquaculture: licensed loads

Sudarjanto et al. (2017) calculated licenced TN loads for aquaculture facilities located in Wet Tropics catchments, as summarised in Table 7.4. In-line with Sudarjanto et al. (2017), assuming that the majority of the aquaculture loads comprise DIN, licenced loads for aquaculture facilities can be expressed as a percentage of the modelled DIN 2012-2013 anthropogenic baseline load for each catchment as reported in Brodie et al. (2017, Table 10 on p.33-34) (repeated in Table 7.4). The Johnstone catchment contains the highest number of aquaculture facilities, but they contribute less than 9% of catchment anthropogenic nitrogen load. In contrast, aquaculture in the Mossman contributes 86%, and aquaculture in the Murray contributes 35%, of the anthropogenic nitrogen loads for their respective catchments. Although TN load contributions from aquaculture are of similar magnitudes in the Mossman and Murray, the total load contribution from sugarcane is much lower in the Mossman ((The Australian and Queensland Governments n.d.), hence the difference in the proportional distribution of N load.

Table 7.4: Estimates of licenced annual total nitrogen (TN) load (kgN/year) released to water from aquaculture facilities in Wet Tropics catchments, calculated using information contained in the Environmental Authorities Register database (data source: Sudarjanto et al. (2017).

Catchment	Number of aquaculture facilities	Licenced TN load discharged to water (kgN/year)	Licenced TN discharge concentration (mgN/L) [#]	Anthropogenic baseline nitrogen load (t/year) [‡]	<u>Licenced</u> aquaculture loads as a percentage of anthropogenic DIN load
		TN _{Aquaculture}		DIN _{Anthropogenic}	% TN _{Aquaculture} to DIN _{Anthropogenic}
Daintree	0	0	-	135	0%
Mossman	2	88,969	2.5, 2.5	104	86%
Barron	1	4,654	1.5	87	5.3%
Mulgrave-Russell	4	2,998	1.3 (0.8, 2.5)	423	0.7%
Johnstone	14	44,539	1.5 (0.6, 2.5)	499	8.9%
Tully	0	0	-	50	0%
Murray	3	81,188	1.7 (1.1, 2)	232	35%
Herbert	0	0	-	886	0%
Total	24	222,348		2,750	8%

[#] Average across all facilities, with values in brackets representing minimum and maximum concentrations.

[‡] Figures are based on modelled dissolved inorganic nitrogen (DIN) anthropogenic baseline load in 2012-2013 as reported in Brodie et al. (2017, Table 10 on p.33-34).

In summary, point sources contribute modest proportions of the total anthropogenic DIN load from the Wet Tropics catchments overall; however, point sources can contribute significant proportions of total anthropogenic DIN load in catchments with lower areas of sugarcane. For example, STPs contribute a substantial proportion of anthropogenic DIN load in the Barron, and licenced aquaculture loads comprise substantial proportions of anthropogenic DIN load in the Mossman and the Murray.

7.2 Potential demand for N offsets from sewage treatment

This Section addresses potential demand for nitrogen offsets from STPs. Two scenarios are considered: 'long term offsetting' for small sewage treatment plants for which upgrading to 5 mgN/L performance would be very costly for the plant operator, and 'short term offsetting' for larger sewage treatment plants that may be approaching their capacity limits, driven by population growth.

7.2.1 Small sewage treatment plants: offsetting to avoid upgrades

Qldwater observe that cost intensity (\$/ML treated) increases rapidly as STP size reduces, particularly so for very small plants. In the absence of substantial financial support from state government, upgrading all STPs to 5mgN/L performance could impose an excessive cost burden on local ratepayers (qldwater 2017). In this situation, offsetting N load discharges in excess of the 5mgN/L by purchasing N-credits from distributed sources via a water quality (N) credit trading market under the Queensland Point Source Water Quality Offset Policy (hereafter PSWQ Offset Policy) could be a potentially attractive solution.

Table 7.1 identifies four STPs in the Wet Tropics for which N-offsetting could be used as a permanent alternative to upgrading to 5mgN/L performance: Mossman, Babinda, Gordonvale, and (in combination as Hinchinbrook Shire Council) Ingham and Lucinda. The rightmost column of Table 7.3 shows the calculated annual offsetting load (kgN/year) required to deliver net 5mgN/L discharge. The quantity of diffuse source N-credits that would have to be purchased to offset these loads depends on the delivery ratio (diffuse source credits to point source offsets) specified in the PSWQ Offsets Policy. The delivery ratio currently specified in the Policy is 1.5:1 (see Chapter 8, for further discussion). This delivery ratio is applied to the offsetting requirements shown in Table 7.3 to determine the quantity of N-credits demanded to fully offset the N concentration exceedances above 5mgN/L from these STPs.

To derive catchment-specific demand schedules for N-credits, each STP's maximum willingness to pay per N-credit has to be quantified. This is done as follows:

1. Estimate the capital expenditure (capex) that would be required to upgrade the STP from its existing discharge concentration to tertiary-level (assumed to be 5mgN/L) performance. Qldwater (2017; Figure 10, p.11) provide a relationship linking capex intensity (\$/ML) for tertiary upgrading to plant capacity (ML); see Appendix 1 for details.
2. Knowing plant-specific capex, calculate the annualised equivalent capex intensity (\$/ML per year), using the standard annualisation formula and assuming a plant lifespan of 50 years⁶⁸ and a real discount rate of 7% per annum; see Equation 6.2 in Chapter 6.
3. Estimate the plant-specific increase in annual operating expenditure (opex) intensity (\$/ML per year) following upgrading. Qldwater provide data on opex for secondary- and tertiary-level STPs (qldwater, 2017; Figure 8, p.10), from which relationships between opex intensity (\$/ML) and plant capacity (ML) can be estimated by regression; see Appendix 1 for details.

⁶⁸ For illustration, willingness to pay calculations are also undertaken for a 25-year annualisation period in Table 7.5.

4. Sum the annualised capex intensity (\$/ML per year) and the increase in annual opex intensity (\$/ML per year) following upgrading to obtain a plant-specific annualised total cost intensity of upgrading (\$/ML per year). This cost will not be incurred if upgrading can be avoided permanently by offsetting.
5. The total cost intensity calculated in Step 4 represents the maximum amount (per annum) that the STP operator would be willing to pay for the quantity of N-credits required⁶⁹ to offset the load from 1 ML of discharge at the existing discharge concentration.
6. Divide the total cost intensity from Step 4 (\$/ML per year) by the N-credit requirement per ML identified in Step 5 (kgN/ML per year), to obtain the STP operator's maximum willingness to pay for one N-credit (\$/kgN).
7. For catchments in which multiple STPs are looking to purchase offsets to avoid upgrading, maximum willingness to pay for credits (\$/kgN) are arranged in decreasing order to produce a stepwise catchment-specific N-credit demand schedule.

These steps are applied as shown in Table 7.5 to produce N-credit demand schedules from STPs seeking to avoid upgrading in the Mossman, Mulgrave-Russell and Herbert catchments (Figure 7.2).

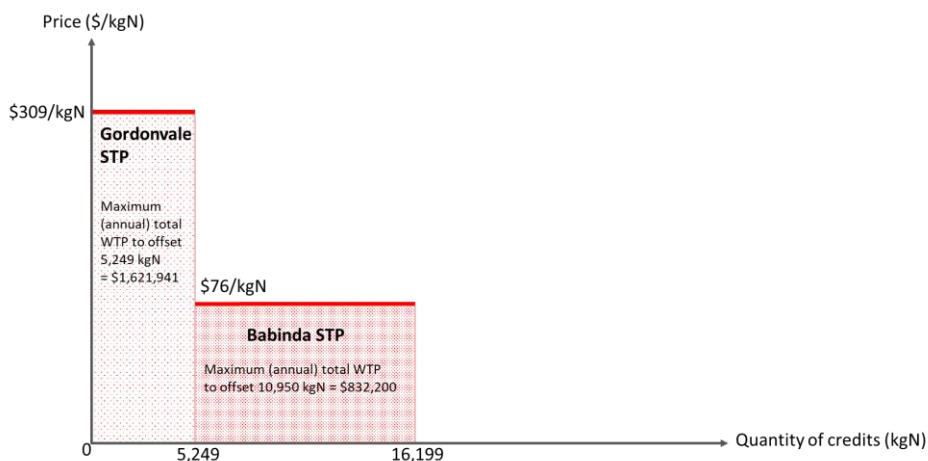
⁶⁹ Take appropriate account of the delivery ratio at this stage of the calculation.

Table 7.5: Capital expenditure intensity, annualised capital expenditure intensity, annual operating expenditure intensity on upgrading from secondary- to tertiary-level (assumed 5mgN/L) treatment, and maximum willingness to pay for N-credits for Mossman, Babinda, Gordonvale and Hinchinbrook Shire Council's STPs.
Calculated using data from Table 7.3.

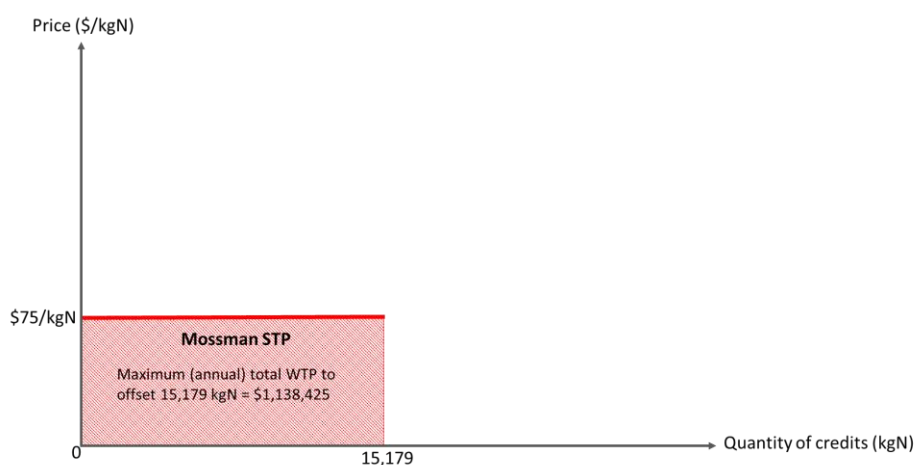
	Annual wastewater volume treated	Capex intensity for tertiary upgrade	Annualised capex intensity over 25 years	Annualised capex intensity over 50 years	Increase in annual opex following tertiary upgrade	Offsetting requirement per ML (assuming a PSWQ Offset Policy delivery ratio of 1.5:1)	Maximum willingness to pay for 1 kg of N-credit (annualisation of capex over 25 years)	Maximum willingness to pay for 1 kg of N-credit (annualisation of capex over 50 years)
STP	(ML)	(\$/ML)	(\$/ML per year)	(\$/ML per year)	(\$/ML per year)	(kgN/ML per year)	\$/kgN	\$/kgN
Babinda	395	32,124	2,757	2,328	530	37.5	88	76
Gordonvale	609	26,028	2,233	1,886	434	7.5	356	309
Mossman	405	31,736	2,723	2,300	524	37.5	87	75
Hinchinbrook*	1,119	19,366	1,662	1,403	326	37.5	53	46

* Hinchinbrook Shire Council provide amalgamated data for their two STPs (Ingham and Lucinda). Offsetting calculations are implemented for these two plants combined.

(a) Mulgrave-Russell catchment



(b) Mossman catchment



(c) Herbert catchment

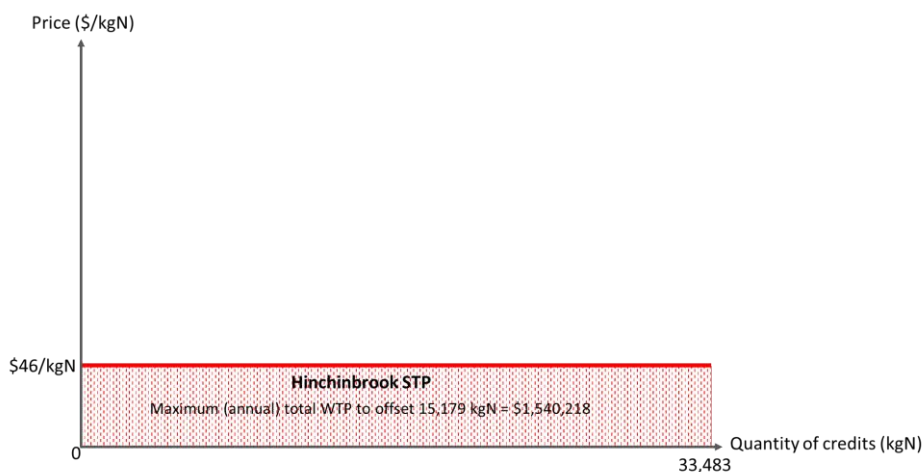


Figure 7.2: N-credit demand schedules from STPs seeking to avoid upgrading in the Mossman, Mulgrave-Russell and Herbert catchments. Quantities of credits demanded assume a delivery ratio of 1.5:1 under the Queensland Point Source Water Quality Offsets Policy. Capex intensity annualised over 50 years (see Table 7.5).

The calculation results in Table 7.5 and the N-credit demand schedules in Figure 7.2 illustrate that, all else equal, willingness to pay for a unit of N-credit increases as plant size reduces (and thus the avoided cost intensity of upgrading increases). Also – again all else equal – willingness

to pay for a unit of N-credit is lower for higher pre-upgrading N-concentrations because more N-credits will have to be purchased from the available budget per ML of discharge. Thus, in our calculations (Table 7.5), Gordonvale STP has a relatively high willingness to pay (309 \$/kgN @ a delivery ratio of 1.5:1) because although its upgrading capex intensity is relatively modest (1886 \$/ML per year, annualising over 50 years at 7% per annum), the quantity of N-credits purchased only need to offset an excess concentration of 5mgN/L. Hinchinbrook's STPs, however, have an even lower upgrading capex intensity (1403 \$/ML per year), but have to purchase sufficient N-credits to offset an excess concentration of 25mgN/L; consequently, their willingness to pay is considerably lower (46 \$/kgN @ a delivery ratio of 1.5:1).

The N-credit demand schedules in Figure 7.2 would potentially comprise the demand sides of catchment-specific N-credit markets. Chapters 5 and 6 estimated the cost of supplying N-credits from practice change in cane and installation of constructed treatment systems, respectively. Referring back to these Chapters, once fertiliser practice level Bf has been fully adopted, small volumes of N-credits from practice change in cane would typically be available for between 60 - 100 \$/kgDIN, whereas N-credits from appropriately located embellished wetlands would potentially be available for between 20 - 60 \$/kgDIN. It is likely that only small STPs (with consequently high avoided capex intensities from offsetting instead of upgrading), or small/medium STPs with discharge concentrations that are only slightly higher than the 5mgN/L target, will be willing to pay prices that are sufficiently high to incentivise credit supply. However, in either of these circumstances, the STPs would only require low quantities of credits to offset their total load exceedances; as a consequence, the demand for N-credits from small STPs that are seeking to avoid upgrading is likely to be quite limited, leading to thin markets.

Further south along the Reef coast, again drawing on data from the Environmental Authorities Register database (Sudarjanto et al. 2017) and qldwater (2018, 2017), and applying the same calculation methodology (including the impact of the currently-specified 1.5:1 delivery ratio), significant demands for DIN offsets from STPs currently discharging at concentrations above 5mgN/L could arise at the following locations:

- Townsville ~ 17 tonnes N offsets @ max WTP 53 \$/kgDIN EoC: Burdekin Region Ross catchment (63% of sediment erosion from hillslopes)
- Mackay ~ 35 tonnes N offsets @ max WTP 100 \$/kgDIN EoC: Mackay-Whitsundays Region: Pioneer catchment (90% of DIN load from sugarcane) & Plane catchment (91% of DIN load from sugarcane)
- Rockhampton ~ 165 tonnes N offsets @ max WTP 83 - 26 \$/kgDIN EoC: Fitzroy Region: Fitzroy catchment: (22% of sediment erosion from hillslopes)
- Bundaberg ~ 25 tonnes N offsets @ Max WTP 530 – 150 \$/kgDIN EoC: Burnett-Mary Region: Burnett catchment (66% of DIN load from sugarcane)
- Gympie ~ 3.5 tonnes N offsets @ max WTP 530 – 245 \$/kgDIN EoC: Burnett-Mary Region: Mary catchment (57% of DIN load from sugarcane)

Catchment DIN load baselines in the WQIPs will presumably include current levels of discharge (i.e. discharges at concentrations that are higher than 5mgN/L) from these STPs. Consequently, offsetting of discharge concentrations in excess of 5mgN/L will contribute to delivery of the DIN load reduction targets for the catchments concerned: here the Mossman, Mulgrave-Russell, Herbert, Ross, Pioneer, Plane, Burnett and Mary.

7.2.2 Large sewage treatment plants: offsetting to delay upgrades

Under another plausible scenario, larger STPs that deliver tertiary-level performance but are approaching their capacity limits due to population growth could be interested in the option of 'short term offsetting' to defer the cost of capacity augmentation. An illustrative estimate of the demand for N-credits for this scenario is produced by considering Cairns Southern STP as an example (blue row in Table 7.3).

From the qldwater data reported in Table 7.3 (qldwater 2018), in Financial Year 2016-17 Cairns Southern STP treated 7106 ML of sewage and discharged 35,438 kgN. The licenced load limit for this plant in the Environmental Authorities Register database (Sudarjanto et al. 2017) is 38,325 kgN. Cairns Southern STP is a tertiary-level plant discharging at an N-concentration of 5mgN/L. Cairns City Council report a daily wastewater flow of 270 litres per person per day⁷⁰. At a discharge concentration of 5mgN/L this per-person wastewater loading will generate an additional annual discharge load of 493 kgN from a Cairns STP for an additional 1,000 population. Using the performance data reported for FY 2016-17, 2,887 kgN of headroom remains in the Cairns Southern STP's licenced load limit. This equates to an additional population of 5,856. The population of Cairns continues to increase, albeit at a slower rate since 2010 (ABS statistics, plotted in Figure 7.3). In this context, Cairns City Council could potentially be interested in purchasing N-credits to offset licenced N-load exceedances over, for example, a 5-year period to defer the cost of augmenting the plant's treatment capacity.

Consider a tertiary-level STP with a 7000ML design capacity, and a licenced discharge concentration limit of 5 mgN/L, with a corresponding licenced annual load limit of 35,000 kgN (i.e. similar to Cairns Southern STP). As population increase pushes the plant beyond its 7000ML design capacity, each ML of additional discharge flow will contribute an additional 5 kg of discharge N load, (assuming that the plant can still maintain its discharge concentration at around 5 mgN/L). It is likely that the plant could maintain discharge concentration around 5 mgN/L for modest increases in flow beyond design capacity; this is the circumstance in which the option to offset load exceedances, and thereby defer capacity augmentation, could be financially beneficial. The N-credit demand generated in this context can be calculated as follows.

The qldwater results (Appendix 1) predict that for a 7000ML plant, the capex intensity of tertiary-level treatment is 7944 \$/ML⁷¹. Each year that capacity augmentation is deferred thus provides a present value cost saving of \$520/ML⁷². This \$520/ML cost saving is the maximum amount (per annum) that the STP operator would be willing to pay for the quantity of N-credits required⁷³ to offset the 5 kgN load exceedance from 1 ML of additional discharge. The corresponding maximum willingness to pay for N-credits is 104 \$/kgN; allowing for a delivery

⁷⁰ <https://www.cairns.qld.gov.au/water-waste-roads/water/wastewater-management/treatment-plants>

⁷¹ From the equation provided by qldwater (2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018)(qldwater 2018), see Appendix 1: $587177 \times 7000^{-0.486} = 7944 \text{ \$/ML}$.

⁷² The present value of a \$7944/ML cost deferred by 1 year at a 7% real discount rate = \$7424/ML. The 1-year deferral of expenditure thus realises a present value cost saving of \$520/ML.

⁷³ Take appropriate account of the delivery ratio at this stage of the calculation.

ratio of 1.5:1 in the PSWQ Offsets Policy, this reduces to \$69/kgN. The quantity of credits demanded in this scenario would be 493 kgN per 1000 population (or, equivalently per 0.27 ML of wastewater input flow) in excess of the plant's design capacity. In practice, it is likely that the plant would no longer be able to maintain the 5 mgN/L discharge concentration target as population increase caused flow to exceed its 7000ML design capacity. As wastewater flow and discharge concentration rise, more N-credits will have to be purchased. This will decrease the price the STP is willing to pay per N-credit, so \$69/kgN should be regarded as an upper limit for credit purchase price in this scenario.

All major cities along the Reef coast except Rockhampton already have tertiary-level STPs (qldwater 2017, 2018). The Fitzroy catchment, in which Rockhampton is situated, does not contain any sugarcane production areas. If Rockhampton's STPs sought to offset their N-loads as an alternative to upgrading, or in order to defer upgrading, N-offsets would have to be purchased from sediment reductions. This would invoke the environmental equivalency multiplier⁷⁴, as well as the delivery ratio multiplier, under the PSWQ Offsets Policy – further reducing willingness to pay per kgN credit.

With the exception of Cairns and Townsville, the population of major cities along the Reef coastline have plateaued since 2010 (Figure 7.3). This suggests that N-credit purchase to delay capacity augmentation is unlikely to generate significant demand for N-credits for the foreseeable future. This may change, however, when the north Queensland economy starts to expand again.

7.3 Potential demands for N-offsets from expansion in aquaculture

Licences recorded in the Environmental Authorities Register database (Sudarjanto et al. 2017) indicate that aquaculture facilities discharge substantial nitrogen loads because, even though their regulated nitrogen discharge concentrations are low (typically 1.1 kgN/L), their discharge volumes can be very high (e.g. ~26,000 ML/year). Expansion in aquaculture in GBR catchments will have to seek approvals from the Queensland Government and the GBRMPA under the GBR Marine Park Act. New aquaculture facilities must be configured to deliver zero net increase in sediment and nutrient into the receiving environment under the EPBC Act. This therefore entails installation of intensive on-site treatment (e.g. with intensive recirculating systems as described by Castine et al. (2013), or industrial-scale commercial algal bioremediation (e.g. <https://www.pacificbio.com.au/our-business/bioremediation/>)), or purchase of N-offsets.

A representative large aquaculture facility in the Environmental Authorities Register database would operate at a nitrogen discharge concentration limit of 1.1 kgN/L and a discharge volume of 26,000 ML/year, giving a total discharge N load of 28,600 kgN. Offsetting this nitrogen load with diffuse source sediment credits under the PSWQ Offsets Policy would require purchase of 42,900 kg of N credits annually (at a PSWQ Offset Policy delivery ratio of 1.5:1). From Chapters 5 & 6, the minimum pricing that suppliers of diffuse source credits in sugarcane

⁷⁴ Under current policy, the environmental equivalency ratio is 1:1, however, environmental equivalency of nitrogen offsets sourced from sediment is an active area of research (Franklin et al. 2018; Garzon-Garcia et al. 2018b).

catchments would be willing to accept is likely to be in the range 20 – 60 \$/kgN. At this level of pricing, the potential cost of offsetting 28,600 kgN load from a large aquaculture facility will be in the range \$858k - \$2574k annually. The recently approved Guthalungra prawn farm development (<http://pacificreef.com.au/guthalungra/>) in the Don catchment will use algal bioremediation technology, rather than N-offsetting to achieve the zero net nitrogen emissions requirement. As the cost-effectiveness of algal bioremediation technology continues to improve, it seems likely that proponents of new aquaculture facilities will embed these treatment technologies into their production systems – potentially removing the need for N-offsetting.

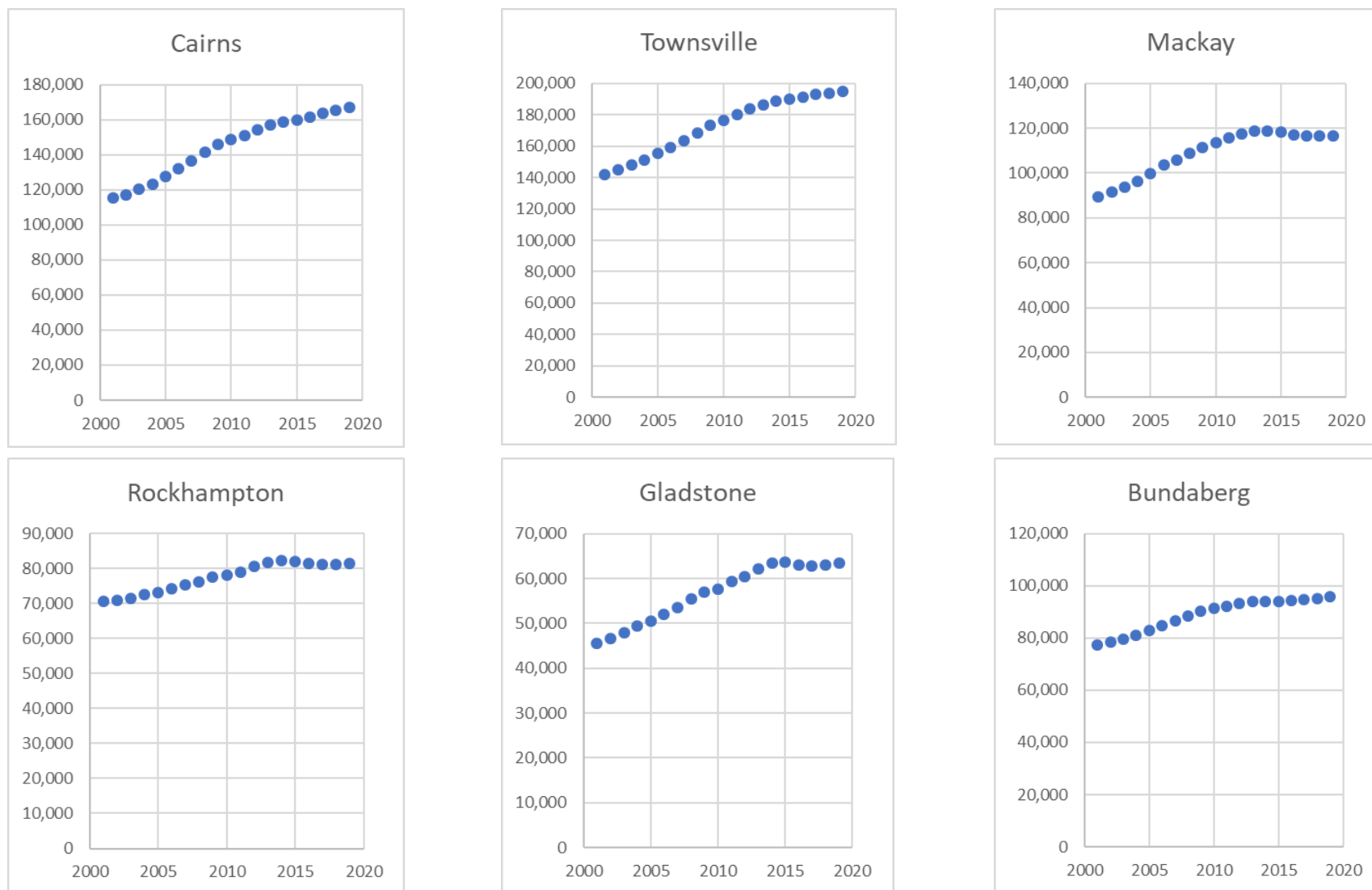


Figure 7.3: Population sizes for cities along the GBR coastline. Australian Bureau of Statistics estimated resident population by LGA: stat.data.abs.gov.au.

7.4 Summary

Licensed point source nitrogen emissions from STPs and aquaculture represent more than 10% of anthropogenic DIN load in Wet Tropics catchments. This Chapter explored scenarios in which these point sources could potentially generate demand for diffuse source N-credits.

Two scenarios were considered for STPs: small STPs seeking to offset to avoid upgrading from secondary- to tertiary-level nitrogen discharge concentrations, and larger STPs seeking to purchase N-credits as an interim measure to defer capacity augmentation. In both scenarios, STP operators' willingness to pay for credits was estimated via an avoided cost approach, using predicted capex and opex costings data from qldwater (2018, 2017). Whilst willingness to pay for credits could be sufficient to elicit supply in some catchments under some circumstances, the aggregate demand for N-credits from STPs is only likely to be modest, leading to thin markets.

Existing aquaculture facilities operate under tighter N discharge concentration limits than tertiary STPs (Sudarjanto et al. 2017); however, because their discharge volumes are typically considerably higher than STPs, the total N load discharged can still be substantial. New aquaculture developments have to achieve no net increase in N emissions as a condition of their environmental approvals. These tight environmental regulations have driven development of algal bioremediation technologies for nutrient removal. Recent approval of a large-scale prawn farm development in the Don catchment (Burdekin region) which uses this treatment technology suggests that the industry and the regulator are pursuing on-site treatment solutions in preference to offsetting.

Looking beyond the Wet Tropics, and other sections of the Reef coastline, there may well be stronger demand for N-credits from STPs in densely populated areas such as South East Queensland that are experiencing sustained population growth. In these settings N-credit demand may be sufficient to drive viable nitrogen markets. However, in many such settings, nitrogen offsets would likely have to be obtained via reductions in sediment loads. On-going research to better quantify environmental equivalency ratios for N-offsetting via sediment reduction should therefore be pursued as a priority to underwrite the environmental credibility of offsetting markets.

Appendix 7A: Capex and opex for upgrading secondary-level sewage treatment plants tertiary-Level performance

Queensland Water (qldwater 2017) provide a regression relationship to predict the capital expenditure (capex) required to upgrade sewage treatment plants (STPs) in GBR catchments from secondary-level treatment to tertiary-level treatment. Data from the same source can also be used to predict (separately) the annual operating expenditure (opex) for secondary- and tertiary-level STPs, and thus to estimate the increase in annual opex that follows from upgrading.

Capex for upgrading to tertiary treatment

A relationship between capex intensity (\$/ML) and STP load (ML) for upgrading STPs in GBR catchments to tertiary-level treatment is provided by Queensland Water (Qldwater, 2017; Figure 10, p.11), as reproduced below.

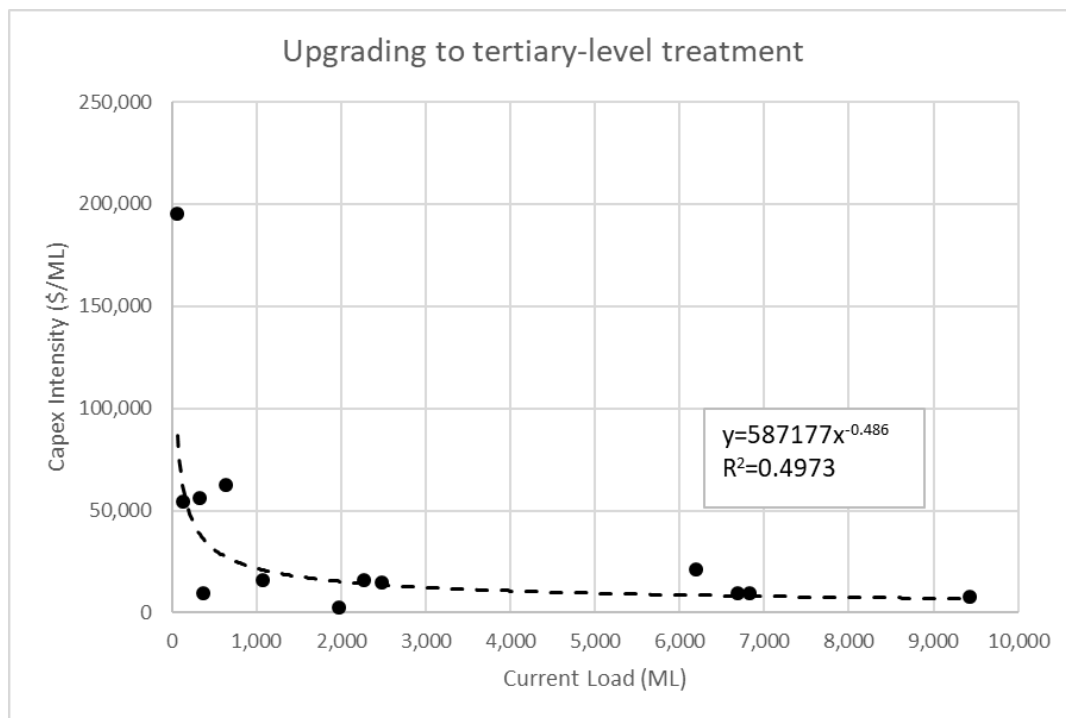


Figure 7.4: The relationship between capex intensity (\$/ML) and STP load (ML) for upgrading STPs in GBR catchments to tertiary-level treatment, as provided by Queensland Water (Qldwater, 2017; Figure 10, p.11).

The relationship is estimated to be the power law function between STP annual load capacity L (ML) and capex intensity C (\$/ML) for upgrading to tertiary-level treatment as in Equation (A1):

$$C = 587177 \cdot L^{-0.486} \quad (7A.1)$$

Increase in opex after upgrading to tertiary treatment

Queensland Water (Qldwater, 2017; Figure 8, p.10) provide unpublished data from the Water Services Association Australia (WSAA) on the annual operating cost intensity (\$/ML) in

relation to total annual treatment capacity (ML) for secondary-level and tertiary-level STPs. These data enable the relationships between opex intensity and treatment capacity to be estimated as power law functions by running linear regressions of $\log_{10}(\text{opex intensity})$ on $\log_{10}(\text{STP load})$ for both data sets separately. Regression results are shown in Table 7A. 1 and Table 7A. 2. The fitted relationships are plotted against the data in Figure 7A. 2 and Figure 7A. 3.

Table 7.6: Regression results for opex intensity (\$/ML) for secondary-level sewage treatment plants as a function of annual treatment capacity (ML). Data from Queensland Water (2017; Figure 8, p.10).

<i>Dependent variable</i> $\log_{10}(\text{opex_intensity})$			
		<i>Num. observations</i>	122 [‡]
		<i>F(1,120)</i>	88.14
		<i>Prob > F</i>	<0.001
		<i>R² (log model)</i>	0.4186
		<i>Root MSE</i>	.35767
	<i>coefficient</i>	<i>t-stat^{‡‡}</i>	<i>Prob > t </i>
constant	3.717934	-9.39	<0.001
$\log_{10}(\text{annual_load})$	-0.322269	37.46	<0.001

[‡] Data point #123 from the original dataset omitted because it has a very high Cook's Distance and significantly influences the estimated parameter for $\log_{10}(\text{Load_ML})$. This data point seems to be extremely anomalous. It reports that a very small STP (1.56ML annual load) can operate with an annual opex of \$179/ML.

^{‡‡} Regression estimated with robust standard errors.

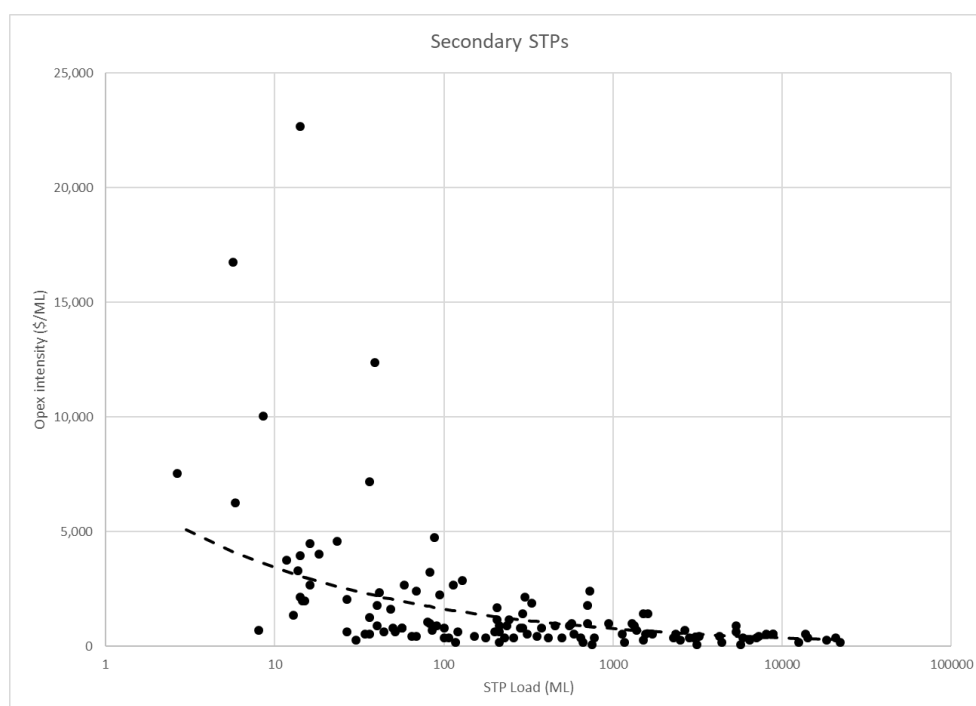


Figure 7.5: Data and fitted regression relationship linking annual opex intensity (\$/ML) to annual treatment capacity (ML) for secondary-level STPs.

Table 7.7: Regression results for opex intensity (\$/ML) for tertiary-level sewage treatment plants as a function of annual treatment capacity (ML). Data from Queensland Water (2017; Figure 8, p.10).

<i>Dependent variable</i> $\log_{10}(\text{opex_intensity})$			
		<i>Num. observations</i>	59 [‡]
		<i>F(1,57)</i>	49.53
		<i>Prob > F</i>	<0.001
		<i>R² (log model)</i>	0.5977
		<i>Root MSE</i>	.30739
	<i>coefficient</i>	<i>t-stat^{‡‡}</i>	<i>Prob > t </i>
constant	4.055061	-7.04	<0.001
$\log_{10}(\text{annual_load})$	-0.3595749	26.47	<0.001

[‡] Data point #18 from the original dataset omitted because it has a very high Cook's Distance and significantly influences the estimated parameter for $\log_{10}(\text{Load_ML})$. This data point seems to be extremely anomalous. It reports that an STP with 2358ML annual load can operate with an annual opex of \$0.25/ML.

^{‡‡} Regression estimated with robust standard errors.

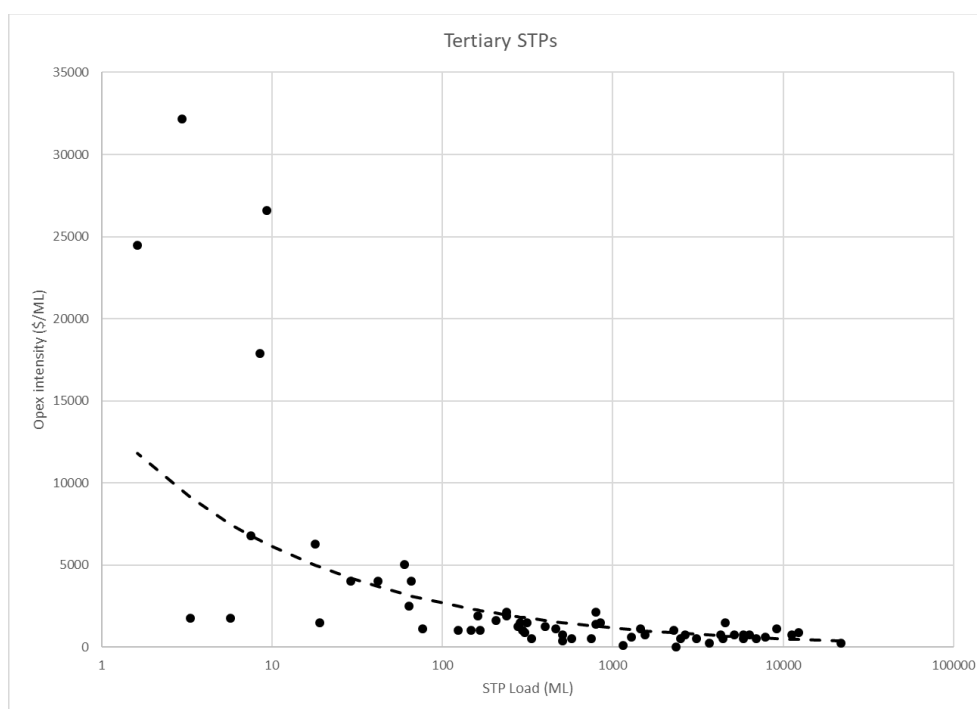


Figure 7.6: Data and fitted regression relationship linking annual opex intensity (\$/ML) to annual treatment capacity (ML) for tertiary-level STPs.

Equations from the regressions (after the inverse $\log_{10}$ transform, incorporating Duan's correction (Duan 1983)) to predict opex intensity O (\$/ML) for secondary-level and tertiary-level STPs of load capacity L (ML) are:

$$O_{secondary} = 1.838461 \cdot 5223.2 \cdot L^{-.3222688} \quad (7A.2)$$

$$O_{tertiary} = 1.239617 \cdot 11351.7 \cdot L^{-0.3595749} \quad (7A.3)$$

For an STP of a given load capacity, the difference in annual opex after upgrading from secondary-level to tertiary-level treatment can be predicted from Equation (7A.2) – Equation (7A.3).

8.0 TRADING RATIOS IN POINT SOURCE TO DIFFUSE SOURCE CREDIT TRADING

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8. Summary

The Queensland Point Source Water Quality Offsets Policy (PS WQ Offsets Policy) applies a delivery ratio and an environmental equivalency ratio when calculating the quantity of credits from a diffuse source that are required to offset point source emissions in exceedance of EA limits (State of Queensland 2019b). The *delivery ratio* (D_r) aims to account for uncertainty regarding the ability of a diffuse source offset to deliver an equivalent reduction in the mass load of the relevant pollutant in the receiving environment specified in the purchasing point source's EA. The *offset environmental equivalency ratio* (E_{er}) aims to account for uncertainty associated with the chemical form or species of pollutant reduction provided by a pollution offset relative to the chemical form or species of the point source emission exceedance that the offset is intended to negate. D_r and E_{er} are applied as multiplying factors to determine the quantity of diffuse source credits that a point source is required to purchase in order to offset a given level of EA exceedance:

Offset required = Load to be offset $\times D_r \times E_{er}$

Currently, under the PS WQ Offsets Policy, the delivery ratio is 1.5 and the environmental equivalency ratio is 1.0 (State of Queensland 2019b).

This Chapter uses P2R APSIM predicted End-of-Catchment (EoC) DIN load reductions delivered by changing fertiliser management practice from Df to Bf on 311 individual cane land management units for the years 1987 – 2013 in the Wet Tropics to investigate the delivery ratio that would be required to ensure that this practice change delivered an EoC DIN load reduction that was at least as large as the mean level expected for the soil type concerned with (i) an 80% probability, i.e. on 4-out-of-5 occasions on average, and (ii) a 90% probability (on 9-out-of-10 occasions on average).

The delivery ratio required to ensure that average offsetting performance is achieved varies across soil types. If management units on todd soil are excluded as outliers (see Figure 8.2), the required delivery ratios are between 1.6 and 2.1 to ensure average offsetting four occasions out of five, and between 2.0 and 3.2 to ensure average offsetting nine occasions out of ten. The appropriate balance between minimising point source abatement cost and safeguarding the condition of key environmental assets in receiving waters is an empirical issue for the environmental regulator. Ongoing investment in the science and modelling that underpin delivery ratio and environmental equivalency ratio should extend the knowledge base on which the environmental regulator can draw to inform that decision.

As described in Chapter 2, the Queensland Point Source Water Quality Offsets Policy (hereafter, PS WQ Offsets Policy) applies a delivery ratio and an environmental equivalency ratio (Figure 8.1) when calculating the quantity of credits from a diffuse source that are required to offset point source emissions in exceedance of EA limits (State of Queensland 2019f).

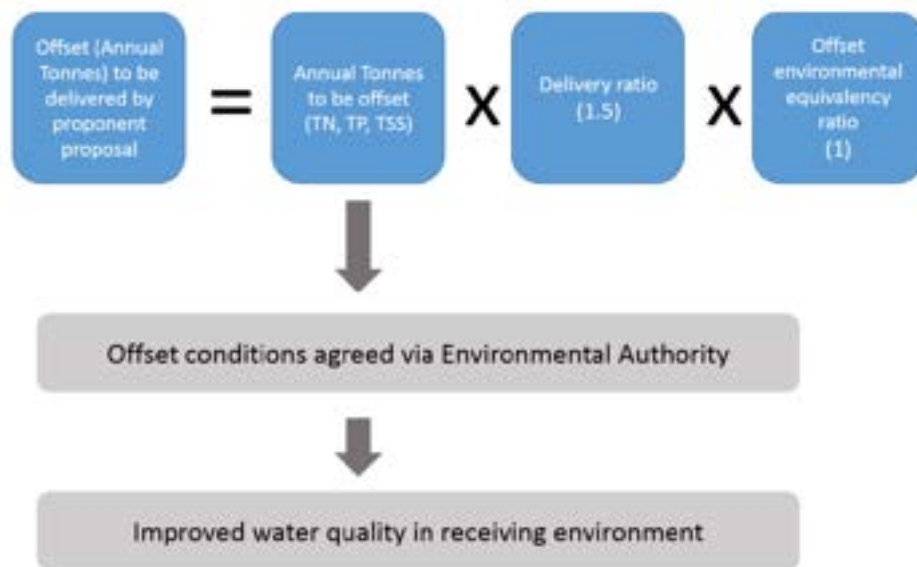


Figure 8.1: Delivery ratio and environmental equivalency ratio applied when calculating the quantity of diffuse source credits required to offset emissions from point sources. Copied from Figure 1, p3.

The *delivery ratio* aims to account for uncertainty regarding the ability of a diffuse source offset to deliver an equivalent reduction in the mass load of the relevant pollutant in the receiving environment specified in the purchasing point source's EA (State of Queensland 2019f). The *offset environmental equivalency ratio* aims to account for uncertainty associated with the chemical form or species of pollutant reduction provided by a pollution offset relative to the chemical form or species of the point source emission exceedance that the offset is intended to negate (State of Queensland 2019f).

As explained in Chapter 2, Section 2.5, the *delivery ratio* (D_r) and *environmental equivalency ratio* (E_{er}) are multiplying factors that are applied as shown in Equation 8.1 to determine the quantity of diffuse source credits that a point source has to purchase in order to offset a given level of EA exceedance.

$$\text{Offset required} = \text{Load to be offset} \times D_r \times E_{er} \quad (8.1)$$

In Equation 8.1 the offset required is expressed in units of credits (e.g. kg DIN credit or tonnes sediment credit) and the load to be offset is expressed in pollutant load at the receiving water specified in the point source emitter's EA. The delivery ratio and environmental equivalency ratio would be expected to be at least one. Currently, under the PS WQ Offsets Policy, the delivery ratio is 1.5 and the environmental equivalency ratio is 1.0 (State of Queensland 2019f).

8.1 Delivery ratio

The delivery ratio aims to account for risks and uncertainties surrounding the processes that generate pollutant reductions at the source of credit supply, and uncertainties in the transport and attenuation of pollutants between the supply source and the receiving water specified in the point source credit buyer's EA (State of Queensland, 2019; p.7). Variation in the level of DIN load reduction generated by a specific improvement in nitrogen management practice can be assessed using 1987 – 2013 yearly predictions from P2R APSIM of DIN losses from 311 individual cane land management units in the Wet Tropics under fertiliser management practices Df to Af (see Chapter 4). The predicted DIN load reductions at End-of-Catchment generated by changing fertiliser management practice from Df to Bf on each management unit are shown in Figure 8.2.

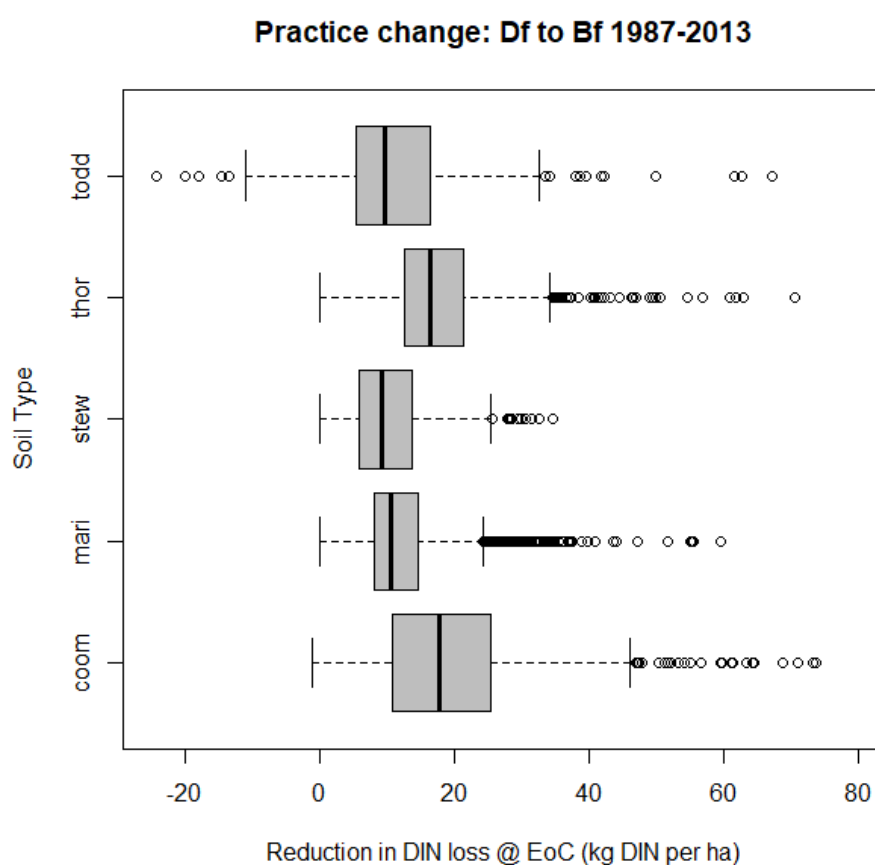


Figure 8.2: Predicted End-of-Catchment (EoC) DIN load reductions delivered by changing fertiliser management practice from Df to Bf on 311 individual cane land management practices for the years 1987 – 2013 in the Wet Tropics (excluding the Herbert catchment), grouped by soil type. Boxes denote the interquartile range between the 25th to 75th percentiles, the bold line denotes the median, whiskers span 1.5 times the interquartile range or the extent of the data (whichever is smaller).

The P2R transport coefficients for DIN in runoff for the 311 management units are all very similar⁷⁵, and a uniform transport coefficient of 0.6 is assumed for DIN lost to drainage. Consequently, the main source of the variation in the End-of-Catchment DIN load reductions

⁷⁵ Transport coefficients from P2R for the 311 individual management units: minimum=0.868, 25th percentile=0.995, 75th percentile=0.999, maximum=1.0.

shown in Figure 8.2 is variation in the DIN load reductions generated on-site by practice change Df to Bf. The spread of DIN load reductions produced by this practice change varies with soil type, being much lower for stew soil. For some management units on todd soil, in some years, P2R APSIM predicts that the Df to Bf practice change will actually increase End-of-Catchment DIN load.

An alternative presentation of the same data is provided in Figure 8.3 which shows empirical cumulative probability distribution functions (ecdfs) for the predicted End-of-Catchment DIN load reductions, grouped by soil type. The ecdfs report – for each soil type – the probability (vertical axis) of the End-of-Catchment DIN load reduction from practice change Df to Bf exceeding a given value (horizontal axis). The dotted horizontal lines denote (in order from the bottom of the plot) the 10th, 20th, 50th (median), 80th and 90th percentiles.

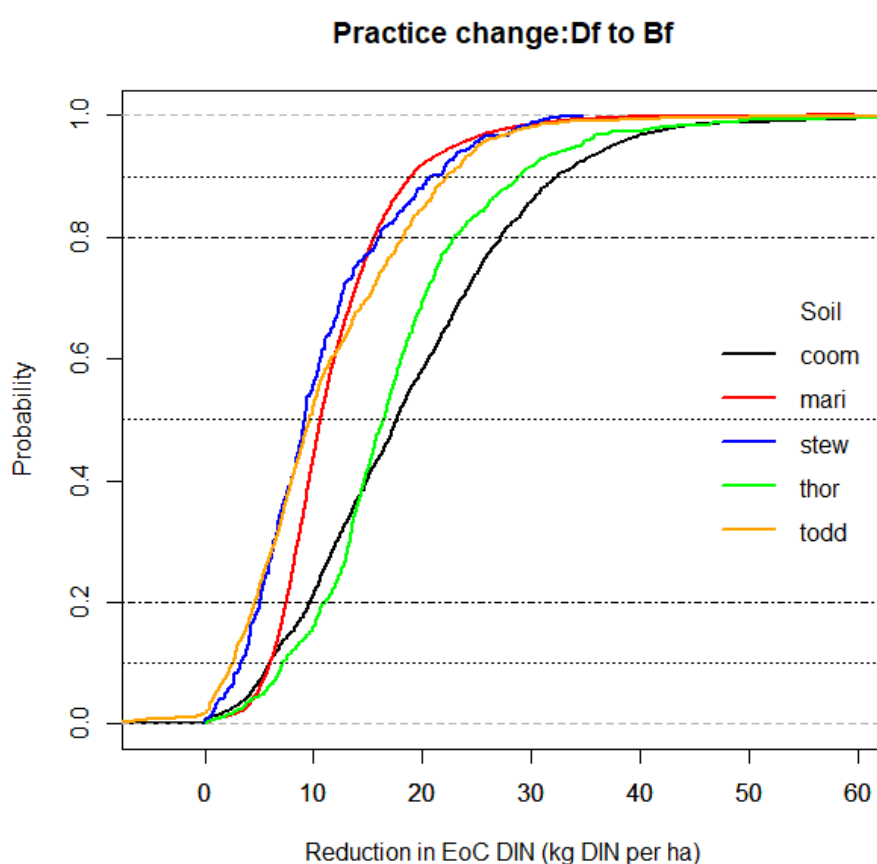


Figure 8.3: Empirical cumulative probability distributions for predicted End-of-Catchment (EoC) DIN load reductions delivered by changing fertiliser management practice from Df to Bf on 331 individual management units in the Wet Tropics (excluding the Herbert catchment), grouped by soil type. Dotted horizontal lines denote (in order from the bottom of the plot) the 10th, 20th, 50th (median), 80th and 90th percentiles.

To illustrate the potential implications of the variations in evident in Figure 8.2 and Figure 8.3 for delivery ratio, Table 8.1 reports the 10th, 20th, 50th (median), 80th and 90th percentiles, and overall mean, End-of-Catchment DIN load reductions for each soil type, predicted over the period 1987-2013.

Table 8.1: The 10th, 20th, 50th (median), 80th and 90th percentiles, and overall mean, predicted End-of-Catchment (EoC) DIN load reductions across management units for each soil type, over the period 1987-2013.

Soil type	10 th %-ile (kg DIN @EoC)	20 th %-ile (kg DIN @EoC)	50 th %-ile (kg DIN @EoC)	Mean (kg DIN @EoC)	80 th %-ile (kg DIN @EoC)	90 th %-ile (kg DIN @EoC)	delivery ratio 4-out-of-5 @mean	delivery ratio 9-out-of-10 @mean
coom	6.02	9.64	17.72	18.87	27.27	32.45	1.96	3.14
mari	6.02	7.50	10.64	11.88	15.57	18.98	1.58	1.97
stew	3.26	5.08	9.17	10.70	16.05	20.92	2.11	3.28
thor	7.13	11.01	16.45	17.63	22.93	28.90	1.60	2.47
todd	2.56	4.61	9.64	11.33	18.18	22.28	2.46	4.42
Average across all soils							1.94	3.06

Delivery ratio intends to ensure that the pollutant load reduction delivered by a diffuse source credit is sufficient to offset a given exceedance in point source emissions at the receiving water⁷⁶. Taking the Df to Bf change in fertiliser practice on management units on coom soil as the source of the diffuse credit, on average this practice change is predicted to deliver 18.87 kg of DIN load reduction at End-of-Catchment (Table 8.1). However, on coom soil, for 20% of instances, the predicted DIN load reduction will not exceed 9.64 kg DIN@EoC (20th percentile in Table 8.1). Consequently, to ensure that the credit sourced from management units on coom soil at least matches the average predicted DIN load reduction of 18.87 kg@EoC on four out of five occasions, a delivery ratio of 1.96 would have to be applied ('delivery ratio 4-out-of-5 @ mean' column in Table 8.1). To ensure that the predicted average DIN load reduction was delivered from management units on coom soil on nine out of ten occasions a delivery ratio of 3.14 would have to be applied. The delivery ratio required to ensure that average offsetting performance is achieved varies across soil types, with delivery ratios of between 1.6 and 2.5 required to ensure the 'average' offset for the soil type is delivered on four occasions out of five, and delivery ratios between 2.0 and 4.4 required to ensure 'average' offsetting on nine occasions out of ten. If management units on todd soil are excluded as outliers, the required delivery ratios reduce to between 1.6 and 2.1 for average offsetting four occasions out of five, and between 2.0 and 3.2 for average offsetting nine occasions out of ten.

The level of 'security' required for offsetting is an empirical question for the environmental regulator but is likely to depend to some extent on the vulnerability of environmental assets in the receiving water to the pollutant concerned. For the GBR, where outbreaks of coral-eating Crown of Thorns starfish and heightened susceptibility to coral bleaching have been linked to elevated nutrient concentrations (Brodie et al., 2017 p.8), it is possible that the environmental regulator may wish to decrease the risk of either of these occurrences on at least four occasions out of five, or possibly nine occasions out of ten. Both these levels of security would potentially require higher delivery ratios than the 1.5 currently applied in the PS WQ Offsets Policy (State of Queensland 2019f). Furthermore, as indicated earlier, the only source of variation considered in the delivery ratios in Table 8.1 is variation in DIN load reduction at-

⁷⁶ Environmental equivalency ratio is ignored for now, on the assumption that the diffuse source credits have the same chemical and biological impacts on receiving waters as the point source emissions. Environmental equivalency ratio is considered separately in Section 0.

source. Variations in DIN transport via the surface water and drainage pathways would likely increase the spread of DIN reductions further, necessitating increased delivery ratios to achieve a given security of offsetting.

As described in Chapter 6, recent results from Adame et al. (2019a, 2019b) indicate that the rate of denitrification in wetlands increases as the nitrate concentration in influent water flow increases. This suggests that inclusion of constructed treatment wetlands as part of the on-farm or in-catchment treatment train could be beneficial for reducing higher DIN loads – and thus alleviating the need for a higher delivery ratio that follows from those higher loads – during adverse weather situations. Use of constructed treatment wetlands in this way could therefore benefit market operation by preventing excessively high delivery ratios from restricting demand for offsets.

Random effects panel regression analysis indicates that – along with soil type, soil permeability and catchment categorisations – annual rainfall at the management unit is a significant driver of the predicted reduction in End-of-Catchment DIN load that follows from practice change Df to Bf (Table 8.2). All else being equal, DIN load reduction at End-of-Catchment from the Df to Bf change in fertiliser practice *increases* as rainfall *increases*. The influence of increasing rainfall is stronger for DIN lost to drainage than for DIN lost via runoff (Table 8.3, Table 8.4). Farm-scale modelling of interactions between increased DIN losses under higher rainfall and increased denitrification rate in wetlands under elevated influent nitrate concentrations would be a suitable subject for additional research to better quantify the potential advantages offered by including practice change (e.g. improvements in fertiliser management practice) and land use change (e.g. constructed treatment wetlands) within the same extended DIN treatment train.

Table 8.1 shows relatively close correspondence between mean and median load reductions for all soil types. Thus, the ecdfs in Figure 8.3 indicate that when delivery ratios are set to ensure that average levels of load reduction can be achieved on four out of five, or nine out of ten occasions, load reductions in excess of average will likely be achieved from close to half of offsetting occasions. Consequently, when delivery ratio is increased to secure receiving water quality against ineffectual diffuse source offsets under adverse weather conditions, receiving water quality will benefit from increased offsetting by diffuse sources when weather conditions are more favourable.

8.2 Environmental equivalency ratio

Environmental equivalency ratio aims to account for uncertainty associated with the chemical form or species of pollutant reduction provided by a pollution offset relative to the chemical form or species of the point source emission exceedance that the offset is intended to negate (State of Queensland 2019f). In the context of water quality credit trading, an environmental equivalency ratio is most likely to be invoked with regard to diffuse source sediment credits being used to offset point source nitrogen exceedances. Generation of nitrogen credits from reductions in fine sediment load is explored in Chapter 11, which focused on reductions in particulate nitrogen (PN) load as the starting-point metric for assessing the accompanying reduction in bioavailable nitrogen delivered to receiving waters. The influence of physical (e.g. granulometry), chemical (e.g. underlying geology and chemistry of the eroding soils), and transport (e.g. pedo-transfer functions) characteristics of DIN load reduction from sediment

abatement were considered through the lens of PN. Accounting for spatial variation in these characteristics produced considerable spatial variation in the level of DIN load reduction generated by sediment abatement, as shown in Figure 11.9. These findings from Chapter 11 suggest that the environmental equivalency ratio of one, as currently specified in the PS WQ Offsets Policy, is unlikely to be sufficient to account for emerging understandings of the drivers of spatial variation – and potentially also temporal variation – in the DIN load reductions that accompany sediment abatement.

Furthermore, research from Franklin et al. and Garzon-Garcia et al. (Franklin et al. 2018; Garzon-Garcia et al. 2018b) suggests that indicators constructed from metrics other than PN alone can provide a better reflection of ecosystem response to bioavailable nutrients from sediment. This continuing research should also be used to inform spatial and, if appropriate, temporal variation in environmental equivalency ratio to safeguard receiving waters from over-estimation of the reductions in bioavailable nitrogen load that accompany sediment abatement.

8.3 Conclusion

Delivery ratio and environmental equivalency ratio have important roles to play in securing the environmental integrity of receiving waters when diffuse source credits are used to offset licenced emission exceedances from point sources. Current research and modelling suggest that careful consideration should be given to increasing the delivery ratio and environmental equivalency ratio from the levels currently specified in the PS WQ Offsets Policy. Whilst such changes appear justified from the emerging science, increases in the quantity of diffuse source credits required to offset a given level of point source emissions will inevitably act to reduce point source buyers' willingness to pay for diffuse-sourced credits. However, provided that point source emissions are licenced under End-of-Catchment pollutant load caps that are policed effectively, increasing delivery ratio and/or environmental equivalency ratio should only act to increase abatement costs for point sources rather than jeopardising receiving water quality. The appropriate balance between minimising point source abatement cost and safeguarding the condition of key environmental assets in receiving waters is an empirical question for the environmental regulator. Ongoing investment in the science and modelling that underpin delivery ratio and environmental equivalency ratio should extend the knowledge base on which the environmental regulator can draw to inform that decision.

Appendix 8A: Driving factors affecting reduction in DIN loads following change in fertiliser practice from Df to Bf

The level of DIN load reduction generated by a specific improvement in nitrogen management practice can be assessed using the 1987 – 2013 yearly P2R APSIM predictions of DIN losses from 311 individual cane land management units in the Wet Tropics under fertiliser management practices Df to Af (see Chapter 4). The predicted DIN load reductions at End-of-Catchment generated by changing fertiliser management practice from Df to Bf on each management unit are shown in Figure 8.2.

Random effects panel regression analysis indicates that – along with soil type, soil permeability and catchment categorisations – annual rainfall at the management unit is a significant driver of predicted reduction in End-of-Catchment DIN load that follows from practice change Df to Bf (Table 8.2). All else being equal, DIN load reduction at End-of-Catchment from the Df to Bf change in fertiliser practice *increases* as rainfall *increases*. The influence of increasing rainfall is stronger for DIN lost to drainage than for DIN lost via runoff.

Table 8.2: Results from random effects panel regression on drivers of P2R APSIM-predicted reduction in management unit-specific annual End-of-Catchment (EoC) total DIN load for the years 1987 – 2013 following fertiliser practice improvement from level Df to level Bf.

Dependent variable: DIN load reduction at EoC from practice change Df to Bf			
		No. observations	8397
		No. management units	311
		No. years	27
		R² (within)	0.1316
		R² (between)	0.7458
		R² (overall)	0.2545
	coefficient	Robust std. error	Prob > z
Constant	15.96514	0.56754	0.000
Rainfall	0.00267	0.00019	0.000
Soil type [#]			
Mari	-4.30689	0.32675	0.000
Stew	-6.51639	0.56760	0.000
Thor	-1.32977	0.67992	0.050
Todd	-8.78762	0.39820	0.000
Soil permeability [‡]			
Low	-5.15582	0.46829	0.000
Moderate	-3.15677	0.32557	0.000
Catchment [§]			
Daintree	-2.28971	0.70250	0.001
Johnson	-4.92778	0.45337	0.000
Mossman	4.06622	0.91099	0.000
Mulgrave-Russel	-3.30121	0.48086	0.000
Murray	-2.22930	0.50424	0.000
Tully	-4.33072	0.48150	0.000

[#] Baseline soil type is Coom.

[‡] Baseline permeability is High.

[§] Baseline catchment is Barron.

Data from P2R APSIM predictions of reduction in annual 1987 – 2013 management unit-specific End-of-Catchment DIN load following practice change for 311 cane land management units in Wet Tropics catchments (excluding the Herbert).

Management unit-specific DIN transport via surface waters from paddock to End-of-Catchment varies by sub-catchment, using predictions from P2R.

DIN transport via drainage is assumed to be 0.6 for all management units (after van Grieken et al. (2019)).

Table 8.3: Results from random effects panel regression on drivers of P2R APSIM-predicted reduction in management unit-specific DIN load to runoff following fertiliser practice improvement from level Df to level Bf.

Dependent variable: DIN load reduction to surface runoff from practice change Df to Bf			
		No. observations	8397
		No. management units	311
		No. years	27
		R² (within)	0.4437
		R² (between)	0.9225
		R² (overall)	0.5717
	coefficient	Robust std. error	Prob > z
Constant	-0.63645	0.03902	0.000
Rainfall	0.00037	0.00001	0.000
Soil type [#]			
Mari	0.41530	0.01366	0.000
Stew	0.15971	0.02237	0.000
Thor	-0.06296	0.02754	0.022
Todd	0.02627	0.02408	0.275
Soil permeability [‡]			
Low	0.17378	0.01825	0.000
Moderate	0.04553	0.01551	0.003
Catchment [§]			
Daintree	-0.16685	0.04564	0.000
Johnson	-0.23913	0.03232	0.000
Mossman	0.05309	0.05553	0.339
Mulgrave-Russel	-0.16819	0.03443	0.000
Murray	-0.12229	0.03321	0.000
Tully	-0.22646	0.03281	0.000

[#] Baseline soil type is Coom.

[‡] Baseline permeability is High.

[§] Baseline catchment is Barron.

Data from P2R APSIM predictions of reduction in annual 1987 – 2013 management unit-specific End-of-Catchment DIN load following practice change for 311 cane land management units in Wet Tropics catchments (excluding the Herbert).

Table 8.4: Results from random effects panel regression on drivers of P2R APSIM-predicted reduction in management unit-specific DIN load to drainage following fertiliser practice improvement from level Df to level Bf.

Dependent variable: DIN load reduction to deep drainage from practice change Df to Bf			
		No. observations	8397
		No. management units	311
		No. years	27
		R² (within)	0.1061
		R² (between)	0.7674
		R² (overall)	0.2526
	coefficient	Robust std. error	Prob > z
Constant	27.69767	0.95850	0.000
Rainfall	0.00383	0.00033	0.000
Soil type [#]			
Mari	-7.86735	0.53724	0.000
Stew	-11.12804	0.95570	0.000
Thor	-2.11398	1.09775	0.054
Todd	-14.68769	0.64780	0.000
Soil permeability [‡]			
Low	-8.88079	0.77053	0.000
Moderate	-5.33567	0.53426	0.000
Catchment [§]			
Daintree	-3.56030	1.17474	0.002
Johnson	-7.82941	0.76883	0.000
Mossman	6.66719	1.47026	0.000
Mulgrave-Russel	-5.24149	0.80084	0.000
Murray	-3.53289	0.84593	0.000
Tully	-6.85957	0.81323	0.000

[#] Baseline soil type is Coom.

[‡] Baseline permeability is High.

[§] Baseline catchment is Barron.

Data from P2R APSIM predictions of reduction in annual 1987 – 2013 management unit-specific End-of-Catchment DIN load following practice change for 311 cane land management units in Wet Tropics catchments (excluding the Herbert).

9.0 SEDIMENT CREDITS FROM LAND REMEDIATION

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9. Summary

The 2017 Scientific Consensus Statement states that the Burdekin catchment contributes 40% of the anthropogenic total suspended sediment load to the Great Barrier Reef lagoon (Bartley et al. 2017). Noting the Burdekin catchment as the largest source of fine sediment load to the GBR (Kroon et al. 2012), and using the Bowen-Bogie sub-catchment in the Burdekin as a case study area, this Chapter presents recent estimates of sediment credit supply achieved through gully rehabilitation to reduce fine sediment generation from a highly eroding landscape (Brooks et al. 2020). A sediment credit supply curve is constructed using cost-effectiveness estimates of gully rehabilitation in the case study area.

Recent results from research undertaken by Brooks et al. (2020b), funded by the Landholders Driving Change Major Integrated Project indicate that:

- Appropriately targeted remediation of large alluvial gullies has the potential to deliver substantial reductions in End-of-Catchment fine sediment load.
- Initial results from the Brooks et al, study suggest that around 120,000 tonnes of sediment load could be removed from the Bowen and Bogie sub-catchments (in combination) for a total up-front expenditure of \$22 million.
- This equates to supply of 120,000 tonnes of fine sediment reductions (i.e. 120,000 tonnes of fine sediment credits) for no more than c. \$46 annualised cost per annual tonne of sediment abated. (Annualisation of up-front and maintenance costs at 7% real discount rate per annum over 25 years). This estimated supply of fine sediment credits will be considered in the light of potential demand for sediment credits in Chapter 10.
- The cost of supplying sediment credits will inevitably rise as the required sediment load reduction increases, but these results suggest that – when appropriately targeted by high resolution spatial prioritisation – remediation of large alluvial gullies appears to provide a valuable source of cost-effective sediment reduction.

9.1 Introduction

Declining water quality is recognised as one of the greatest threats to the long-term health and condition of the Great Barrier Reef (GBR). There is well-documented evidence that anthropogenic pollutants are contributing to the decline of aquatic ecosystem health both within the catchment and in the GBR (Brodie et al. 2008). Discharge of fine sediments (silt and clay), dissolved inorganic nitrogen (DIN), phosphorus and herbicide into the GBR have increased significantly since European settlement (Kroon et al. 2012). For fine sediment loads, evidence indicates that loads from regional catchments are approximately five times pre-European settlement loads (Kroon et al. 2012). The Johnstone, Herbert, Burdekin, O'Connell, Fitzroy, Burnett and Mary rivers have been identified as priority catchments for management interventions with fine sediment reduction targets set at 100, 99, 840, 96, 390, 85 and 130 kilo-tonnes, respectively, to be achieved by 2025 (State of Queensland 2018b). A market for sediment credits could potentially assist delivery of these targets such that load reductions are achieved at least cost to society.

This Chapter explores the supply side of a potential market in sediment credits in the GBR catchments. Hillslope, streambank and gully erosion in grazing areas are the processes that generate sediments (Thorburn and Wilkinson 2013; Bartley et al. 2015; Wilkinson et al. 2015a; NQ Dry Tropics 2016), however recent studies suggest that sediments from hillslope (i.e. colluvial gullies) and alluvial gullies dominate catchment sediment loads in savannah landscapes (Brooks et al. 2014). During rainfall events, stored fine sediments in gullies and hillslopes are transported downstream and eventually discharged at the river mouth. Sediments generate adverse impacts on the health of receiving waters and the Reef. Efforts to achieve fine sediment reduction targets in priority GBR catchments require substantial private and public investment. It is therefore critical that these societal costs are incurred in ways that generate maximum water quality benefits.

Sediment credit trading, set within an integrated water quality regulation framework, presents an opportunity to incentivise private businesses to finance sediment reduction activities that generate sediment reduction credits that can then be sold in a sediment trading market. To further investigate the feasibility of this proposition, it is necessary to obtain preliminary estimates of the quantity of credits that could potentially be generated, and the range of costs that would likely be incurred. Data collected from several gully and streambank remediation projects that have been, and are still being, funded under the Great Barrier Reef Gully and Streambank Joint Program can be used as a basis for deriving preliminary estimates of credit supply. This umbrella program of work delivers a range of scientific research programs and projects on gully and streambank protection and remediation, grazing land management improvement, Traditional Owner engagement and training, communication, and opportunities for employment and tourism (The Australian and Queensland Governments 2016). The Joint Program was established as a partnership between the Australian and Queensland governments, the Great Barrier Reef Foundation, Greening Australia, CSIRO, Griffith University, regional natural resource management bodies, philanthropic organisations, and other industry groups. As more gully remediation projects are commissioned, information on aspects of practical implementation is starting to become available. Emerging data on the costs and effectiveness of different remediation options are vital for informing future investment in land remediation. To ensure data are comparable across projects, a consistent framework should be used for data collection. This chapter proposes such a framework for

recording of raw data for future gully remediation projects. It also estimates a sediment credit supply curve from remediation of alluvial gullies in the Bowen-Bogie sub-catchments of the Burdekin region.

The 2017 Scientific Consensus Statement states that the Burdekin catchment contributes 40% of the anthropogenic total suspended sediment load to the Great Barrier Reef lagoon, with the Wet Tropics (15%), Fitzroy (18%) and Burnett Mary (15%) being the other major contributors (Bartley et al. 2017). Noting the Burdekin catchment as the largest source of fine sediment load to the GBR (Kroon et al. 2012), and using the Bowen-Bogie sub-catchment in the Burdekin as a case study area, this study presents recent estimates of sediment credit supply achieved through land (i.e. gully) rehabilitation to reduce fine sediment generation from a highly eroding landscape (Brooks et al. 2020b). A sediment credit supply curve is constructed using cost-effectiveness estimates of gully rehabilitation in the case study area. Sections 9.4, 9.5 and 9.6 of this Chapter draw heavily from Brooks et al. (Brooks et al. 2020b), adapted to suit the context of supplying fine sediment credits to a water quality credit trading market.

The Chapter proceeds as follows. Section 1 provides brief background on soil erosion on grazing land as a source of fine sediments. Section 2 describes land remediation as a potentially cost-effective option for reducing fine sediment export to the Reef. Section 3 describes a newly developed costings framework for collating data on gully remediation costs. Section 4 uses a cost-effectiveness approach to derive a supply curve for sediment credits generated from ex-ante estimates of gully-specific rehabilitation costs and predicted reductions in fine sediment following gully rehabilitation. The final section concludes.

9.2 Gullies as major sources of sediments

Livestock grazing occurs over more than 74 percent of the land draining into the GBR catchments (Rolfe et al. 2018b). A combination of grazing practices, rainfall pattern and forest clearing, has over the past 100 years reduced soil resistance to erosion during rainfall events resulting in high sediment losses from hill slopes, stream banks and gullies (Thorburn and Wilkinson 2013; Bartley et al. 2014, 2015). Grazing practices remove vegetation cover, with cattle causing physical disturbance to the soil which breaks through the surface layers to expose the sub-soil. Once exposed, the sub-soil is highly prone to rapid erosion with rainfall. This problem is more acute on sodic soils that occur throughout the region. Over time, these erosion processes have created large gully systems, many of which continue to erode today (Wilkinson et al. 2015a; Brooks et al. 2019).

Though sediments are eroded from both surface and sub-surface soil, the sub-surface sediment sources have been identified as the primary contributor to sediment delivery to the GBR (Hughes et al. 2009; Tims et al. 2010; Olley et al. 2013; Wilkinson et al. 2013; Bartley et al. 2015). Fine sediment is of particular concern because of its ability to travel further through the catchment and offshore to reach GBR marine areas. Sediment particles are generally sized using Udden-Wentworth classification for sand ($>63\ \mu\text{m}$), medium and coarse silt ($16\text{--}63\ \mu\text{m}$), fine silt ($4\text{--}16\ \mu\text{m}$) and clay ($<4\ \mu\text{m}$), (Leeder, 1982 as cited in Bartley et al. 2014a). The Reef 2050 Water Quality Improvement Plan focuses on reducing fine sediment (silt and clay $<16\ \mu\text{m}$) from entering the Reef. As described in Wilkinson et al. (2019), this fine soil fraction can be traced to subsoil erosion mainly from gullies in the Burdekin (Wilkinson et al. 2013), Fitzroy (Hughes et al. 2009) and Normanby catchments (Olley et al. 2013), which

together contribute 50% of fine sediment load to the GBR (McCloskey et al. 2017), despite gullies occupying an area that is less than 0.5% of all grazing land (Wilkinson et al. 2015a). In the Burdekin catchment, Wilkinson et al. (2013) found that between 77% and 89% of fine sediment originates from subsurface soil sources in close proximity to the drainage network. The Bowen-Bogie sub-catchments, within the Burdekin, contribute approximately 30% of fine sediment load to the GBR lagoon (Brooks et al. 2016). Furthermore, of all of the GBR regional management units, gullies in the Bowen-Bogie contribute the highest average sediment export (0.5 tonnes/ha/year) to GBR coast and estuaries, with gullies in the Bowen-Bogie comprising 16% (1,162,900 ha) of all gullies in the GBR (Wilkinson et al., 2015a, Table 3 on p.12). Given that the Bowen-Bogie sub-catchments are the single most significant source of fine sediment export to the GBR (Brooks et al. 2016), this Chapter focuses on these sub-catchments as a potential targeted area for land remediation activities to generate sediment credits.

9.3 Land remediation as a sediment abatement measure

Abatement measures to reduce fine sediment export to the Reef aim to control hillslope, streambank and gully erosion by increasing resistance and reducing the eroding forces of water flowing overland and in streams and channels (Rose 2004). Grazing management practices to control soil erosion from hillslopes, streambanks and gullies generally require maintaining and improving vegetation cover at the end of dry season through additional investment in fencing and off-stream watering infrastructure (NQ Dry Tropics 2016). From the landholder's perspective, if the private abatement costs of reducing and redistributing grazing pressure, together with costs of the requisite infrastructure, are *less* than the private benefits of abatement, then abatement is a sound business investment. As uptake of improved grazing management practice is still low in the Burdekin region (NQ Dry Tropics, 2016, Tables 5.3 and 5.4 on p.97), it can be inferred that in most cases the private benefits of sediment abatement are not preceived to be large enough to incentivise voluntary uptake of improved grazing management practices. Based on 12 years of trial results on the profitability of different grazing strategies under variable climate in the Burdekin, O'Reagain et al. (2011) conclude that grazing practices that are more sustainable (i.e. wet season spelling, matching stocking rates to available forage, reducing stocking rate to match long-term carrying capacity) yield higher net present value after 12 years compared to grazing practices with higher stocking rates. The lengthy time span required to achieve positive net present value introduces risks and uncertainties regarding costs and outcomes, and is likely to be a deterrant to adoption (Rolfe and Gregg 2015; Star et al. 2019). Furthermore, vegetation takes time to recover after stocking pressure is reduced. From a societal perspective, this time delay in securing water quality benefits is a major concern because society values earlier delivery of benefits more highly (Star et al. 2018). Therefore, alternative abatement measures, such as gully remediation, that will likely deliver sediment reductions in a much shorter time frame, are desirable. Remediation of large-scale gullies, however, incurs substantial investment cost; well in excess of the benefits that would accrue to landholders. Thus, in the absence of payments in recognition of sediment reduction (such as a water quality credit trading scheme for sediment reduction credits), farmers will not be inclined to implement remediation of large-scale gullies.

Identifying appropriate combinations of treatment intensities and target locations for gully rehabilitation as part of a sediment abatement prioritisation process has the potential to deliver the most cost-effective reductions in fine sediment loss. To produce accurate results, this

prioritisation needs to be informed by high-resolution LiDAR-based mapping of gullies in the landscape to determine:

- gully-specific morphometrics (slope, width, steepness etc.)
- appropriate categorisation of gully form to assist in determining appropriate remediation measures
- erosion rates and thus the contribution of individual gullies to sediment loads at sub-catchment scale

Using a cost-effectiveness framework supported by gully-specific, LiDAR-derived fine spatial resolution estimates of erosion rate, this Chapter quantifies the abatement costs and reductions in fine sediments associated with land remediation and rehabilitation activities on highly eroding gullies in the Bowen-Bogie sub-catchments. The Chapter focuses on potential generation of sediment credits from gully remediation, rather than from improvements in grazing management practices. Readers interested to know more about existing cost-effectiveness estimates for sediment reduction from changing grazing management practices at both paddock and catchment/regional levels in GBR catchments can refer to the comprehensive and up-to date review by Farr et al. (2019).

9.4 Cost components within an overall structure for privately financed gully rehabilitation projects

The proposed cost structure for gully rehabilitation works is developed using the Australian and Queensland governments' documentation pertaining to road infrastructure projects as a reference (Commonwealth of Australia 2017b; State of Queensland 2017). Road infrastructure projects were chosen as a point of reference because of similarities in the nature of the work involved and also in key elements of the costings (e.g. earthmoving, rock filling, soil amelioration and revegetation). The range of scales for gully remediation and rehabilitation projects is also somewhat similar to that for road repair and road construction projects, and many of these projects are also publicly funded.

As described above, the cost of gully remediation is typically too high to be borne by the landholder alone. If there is an appropriate market-driven environment for trading in water quality credits, profit-making opportunities could potentially incentivise private investment in gully remediation projects. A proposed gully remediation project requires access to a grazing property. An agreement with the landholder on the following subjects will therefore need to be obtained prior the planning stage of the project:

- Current and anticipated future sediment yield from the gully, and proposed intensity of treatment (e.g. minimal, low, medium and high) to obtain a preliminary estimate of the annual volume of credits that could be generated
- Project lifespan to determine the number of years for which sediment credits can be generated from the remediated site
- Cost share between landholder and project proponent
- Revenue share from the sale of credits between landholder and project proponent

In this costing structure, a ‘project’ refers to land remediation activities undertaken on a single gully or clusters of gullies located within the same vicinity, whereas a ‘program’ refers to two or more gully rehabilitation projects undertaken at discrete sites which could span multiple grazing properties.

The proposed structure for organising the different cost elements incurred by the project proponent and landholder in a gully rehabilitation project is shown in Figure 9.1. Costs and in-kind contributions are generally classified as *upfront* (i.e. one-off) or *on-going* (e.g. annual). In Figure 9.1, upfront costs refer to costs and in-kind contributions, that are incurred only once. These could be incurred during different phases of the project (e.g. planning, design, implementation etc.) until completion of the on-ground works. The overall cost structure centres around the project proponent (and their contractors) and landholder(s) working as separate, but integrated, counterparts since the costs of gully remediation are typically too high to be borne by the landholder alone.

A project proponent may hire a contractor to carry out the on-ground works (indicated by the leftward arrow at the top of Figure 9.1). They may also hire contractors to implement on-going operating and maintenance (O&M) after completion of the on-ground work to ensure the sediment yield from the remediated site remains at a low level. This process should be verified independently by the sediment credit market regulator via a prescribed monitoring and evaluation process. On-going costs can be incurred from completion of the on-ground works until the end of the project lifespan. Landholders may also incur upfront and on-going costs as well as providing in-kind contributions throughout the project.

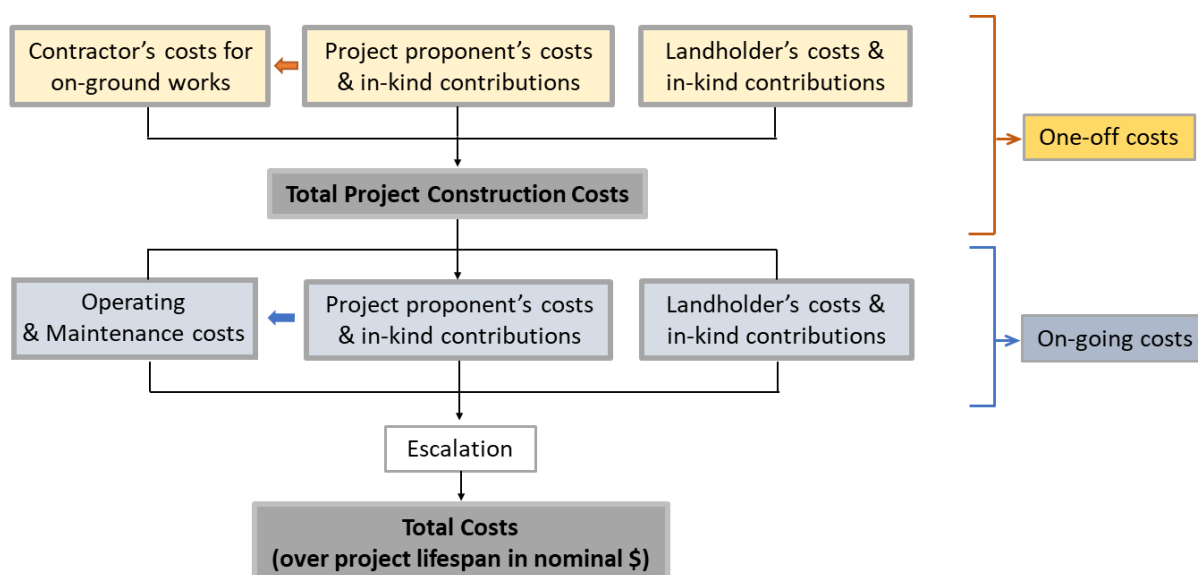


Figure 9.1: Costings structure for gully rehabilitation and remediation projects

The upfront costs and upfront in-kind contributions incurred by the project proponent and landholders together form the *total project construction costs*. An ‘escalation’ factor should be applied to all future costs to arrive at the final total costs for the project. ‘Escalation’ refers to expected inflation rate and/or any other external macroeconomic variables (e.g. exchange rates) that need to be factored into the costs. Total cost over the entire lifespan of the gully

rehabilitation project is expressed in nominal dollars (i.e. incorporating expected inflation and external macroeconomic factors [if relevant]) if escalation is applied to the total project costs.

Further breakdown of upfront cost incurred by project proponent, contractor and landholder is outlined in Figure 9.2. As shown in Figure 9.2, focusing on the *project proponent's costs and in-kind contributions* elements, a project typically goes through four phases of measured progress until full completion of the on-ground works: concept phase, development phase, implementation (i.e. delivery) phase and finalisation phase. Costs incurred by a project proponent can be project phase-specific or general, and both categories of costs must be recorded as such. The project proponent does not necessarily need to carry out all the works themselves; they can choose to pay a contractor or multiple contractors to undertake relevant work for each phase. A project proponent may instead choose to be actively involved in all phases of the project and only contract out the on-ground construction work component to a separate entity (Figure 9.1 and Figure 9.2). The *contractor's costs for on-ground works* element (Figure 9.2) is split into on-site direct and indirect costs, and off-site overheads and margins. Direct on-site costs for on-ground contractors are partitioned into broad categories of resources: labour, equipment and machinery, materials and subcontractors.

Landholder's one-off (net) costs and one-off in-kind contributions are categorised into direct and indirect costs and in-kind, net of any benefits realised from gully remediation (Figure 9.2). Direct costs to the landholder could be payments for labour, in-kind contribution of their own time towards the project, and use of the landholder's own equipment and machinery. The landholder may also incur costs in supplying materials for the gully rehabilitation works when raw materials (e.g. rocks and soil) are sourced from the property containing the remediation site. These material supply costs could therefore be regarded as an in-kind contribution from the landholder. Indirect costs to landholders comprise transaction cost, transition cost and forgone profits (the latter typically arise from changes in land use e.g. fencing livestock out of remediated areas). Any immediate one-off benefits that are realised by landholders during, or as a consequence of, gully remediation works should also be recorded, and the sum of these benefits should be subtracted from the total costs incurred.

Further breakdown of on-going cost incurred by project proponent and landholder is outlined in Figure 9.3. There are three elements in the on-going (annual or periodical) costs of a gully rehabilitation project: *project proponent's costs and in-kind*, *landholder's (net) costs and in-kind*, and *contractor's operating and maintenance costs* (Figure 9.3). The project proponent may choose to do the operating and maintenance (O&M) themselves or hire a contractor to carry out the work on their behalf. The costings structure for O&M costs is similar, in that the costs are split into on-site direct and indirect costs, and off-site overheads and margins. Finally, landholders may continue to incur annual and periodical costs and in-kind in the form of labour, equipment and machinery, and material as direct costs, as well as incurring indirect costs. Landholders may also realise on-going benefits following successful gully remediation. Long term benefits may include increases in land value, higher growth of biomass (due to improved soil health if land remediation included top and sub-soil amelioration) and greater business revenues due to higher consumer demand for food products grown on remediated (i.e. low water quality risk) properties. These on-going benefits should be subtracted from relevant components of cost to arrive at the final landholder's net costs and in-kind contribution.

If gully remediation is to be an attractive financial proposition, revenues received from the sale of sediment credits should be sufficient to cover all project costs as outlined above, plus a competitive rate of return on investment, particularly on the upfront costs. Higher-risk gully remediation projects would typically require a higher rate of return on investment. Tax incentives could be introduced to encourage participation from private investors. As for the demand side, an appropriate regulatory mechanism could be instituted to ensure sustained demand for sediment credits. Further details of the potential demand for sediment credits are provided in Chapter 10.

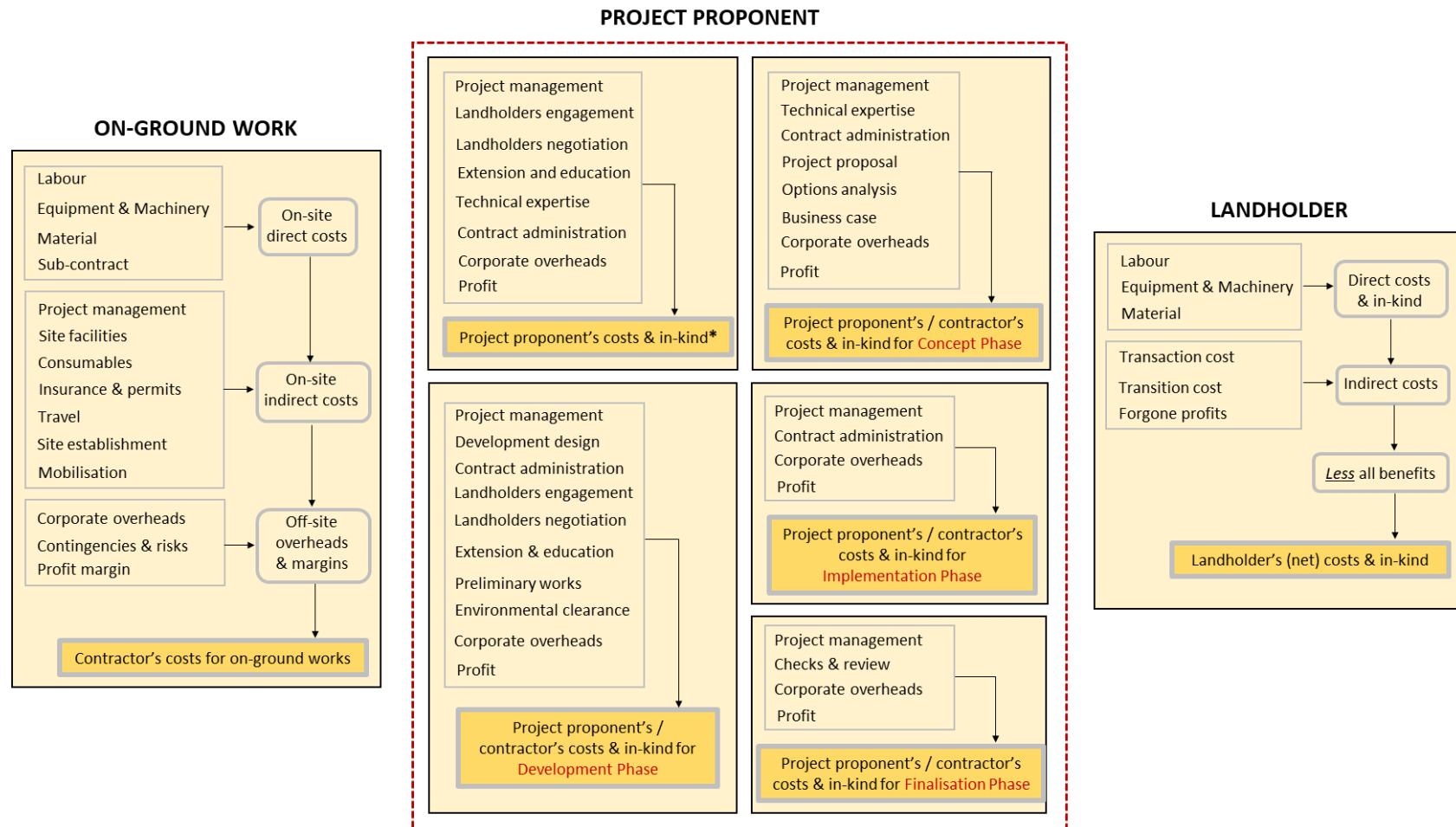


Figure 9.2: Costings structure for each element of the upfront *total project construction costs*

Note: * For costs and in-kind contributions that are not already accounted for in the Concept, Development, Implementation and Finalisation phases. The centre dotted box (in red) constitutes all costs and in-kind components that are linked directly to the project proponent, or contractors hired by the project proponent to complete the relevant phases of the work.

OPERATING & MAINTENANCE (O&M)

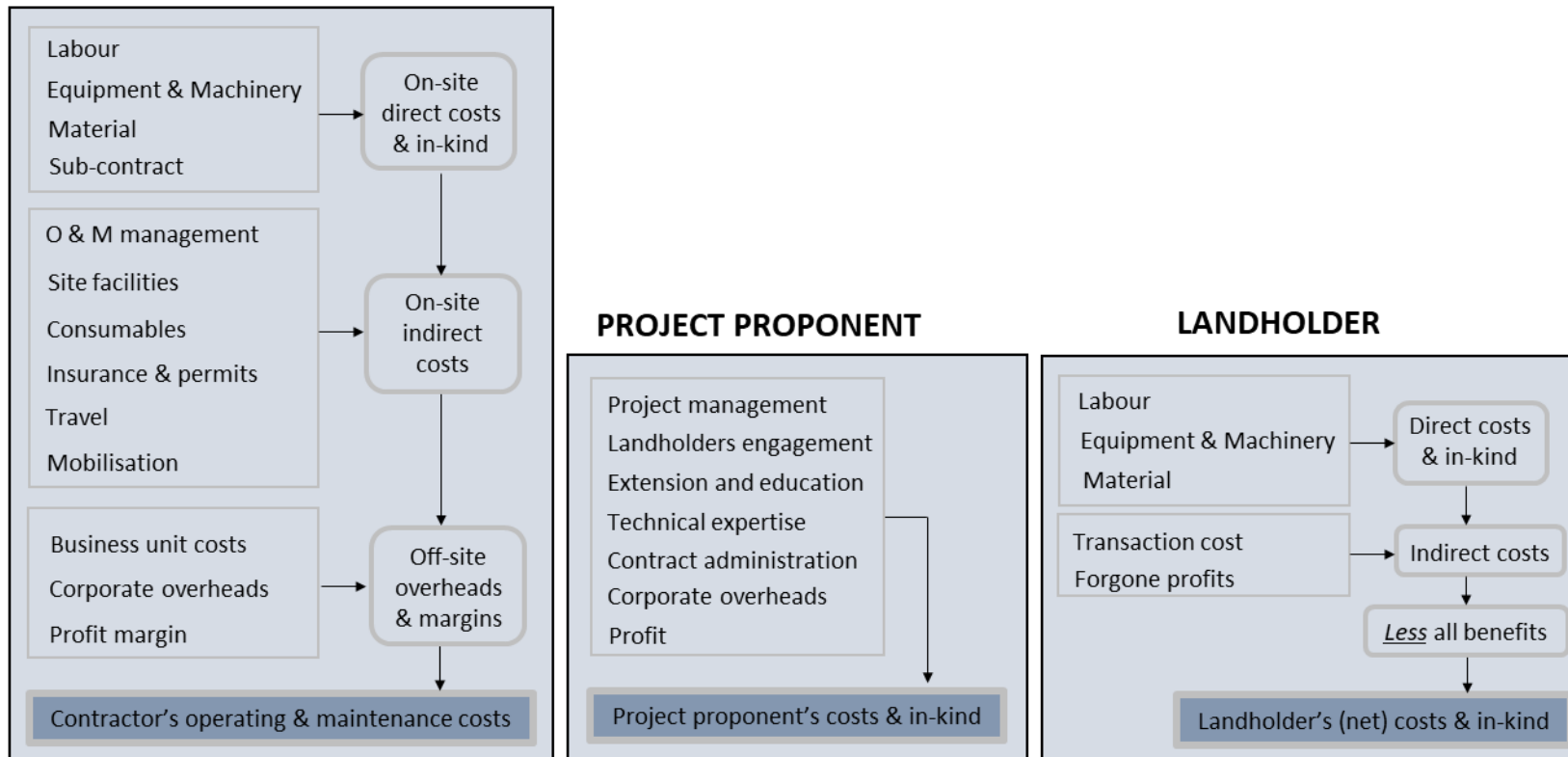


Figure 9.3: Costings structure for each element of the on-going costs

9.5 Methods

9.5.1 Cost-effectiveness and discounting

Two sets of information are required to evaluate the cost-effectiveness of a gully rehabilitation project: (i) the effectiveness of the project in terms of the amount of sediment reduced by the rehabilitation (in tonnes), specifically the fine fraction of silt and clay, and (ii) the costs incurred (in \$) to obtain that reduction. The reduction in fine sediment load for a given level of intervention and treatment effectiveness is first estimated at the remediation site. A *delivery ratio* from remediation site to end of catchment is then applied to predict the load reduction at end-of-catchment. This delivery ratio takes into account intermediate deposition of sediments within the catchment, together with rainfall patterns, intensity and duration. For some catchments, it may take several years for sediments to travel between sources and end-of-catchment, whilst in other catchments the time lag is much shorter (Walling 1983). Though the beneficial effect of sediment reduction may accrue immediately after rehabilitation is complete, it is likely that sediment reduction could continue to increase for several years as the rehabilitated site stabilises.

The sediment load reduction estimate constitutes the 'effect' component, whilst the rehabilitation cost incurred to obtain that reduction constitutes the 'cost' component in cost-effectiveness analysis. As described in Section 9.4, the total cost of gully rehabilitation is the sum of all upfront and on-going costs, including in-kind contributions, incurred by the project proponent, landholder and contractor for implementing on-ground works (Figure 9.1). All future (escalated) on-going costs need to be discounted appropriately to present value before they are added to the upfront present value cost. All future escalated costs are discounted at a nominal discount rate i over a project lifespan, T , to arrive at the total present value cost (TPVC) of remediating gully j :

$$TPVC \text{ of remediating gully } j = C_j = A_j + \sum_{t=1}^T \frac{F_{j,t}}{(1+i)^t} \quad (9.1)$$

where A_j = one-off costs incurred at the start of gully remediation project j (i.e. year $t = 0$) and $F_{j,t}$ = future annual and periodical (nominal) costs incurred post-treatment for gully remediation project j (for year $t = 1, 2, \dots, T$).

The nominal discount rate i is related to the real discount rate r , via inflation rate m , through the following equation:

$$r = \frac{i - m}{(1 + m)} \quad (9.2)$$

The real discount rate used in cost-effectiveness studies that deliver public benefits is typically 7% p.a. (Rust and Star 2018; Star et al. 2018; Alluvium 2019). Equation (9.2) indicates that for an anticipated inflation rate of approximately 2.8% p.a., a nominal interest rate of 10% will deliver a real discount rate of 7% p.a. If the inflation differs from 2.8%, nominal interest rate (i) should be adjusted appropriately via Equation (9.2) such that the real discount rate (r) remains at 7% p.a.

Cost-effectiveness can be evaluated for projects with varying project lifespans (e.g. 10 years (Rust and Star 2018); 30 years (Alluvium 2019)). The results reported in this study apply a 7% real discount rate over a 25-year project lifespan.

Combining the total present value cost in (9.1) and effect as a ratio, the cost-effectiveness metric for gully rehabilitation j (i.e. at site j) undertaken in a given catchment (i.e. Bowen-Bogie sub-catchment in this study) can be represented simply as:

$$CE_j = \frac{C_j}{E_j} \quad \text{for all } j = 1, 2, \dots, J \quad (9.3)$$

where CE_j = cost-effectiveness of gully rehabilitation j (in \$/tonne), C_j = the present value cost of rehabilitating gully j and E_j = volume of sediment reduced (tonnes) for a given treatment intensity (e.g. minimal-, low-, medium- or high-intensity) and erosion control effectiveness (expressed as a percentage), similar to that used in the Gully Toolbox (Wilkinson et al. 2015b).

Different levels of treatment intervention or intensity produce differing volumes of sediment saving over different numbers of years. Increasing treatment intensity generally also increases the effectiveness of the sediment control intervention (Wilkinson et al. 2015a, 2019; Alluvium 2016; Brooks et al. 2016). A method is therefore required to compare the economic performance of gully remediation projects with different project lifespans. Converting a total present value cost in absolute \$ into an *annualised equivalent present value cost* in \$/year enables this annualised cost to be used alongside the estimated annual sediment savings expressed in tonnes/year. This produces a consistent cost-effectiveness metric (in \$/tonne abated) that can be used to compare cost-effectiveness across all candidate gullies for the purpose of management prioritisation.

The following formula is used to convert *total present value cost* (TPVC, \$) to equivalent *annualised present value cost* (APVC, \$/year):

$$APVC = TPVC \left[\frac{r}{1 - (1 + r)^{-T}} \right] \quad (9.4)$$

where: r is the real discount rate, and T is the project lifespan in years.

If annualised present value cost (APVC) is used alongside the yearly sediment reduction (AE), Equation (9.3) can restated as follows:

$$CE_j (\$/\text{tonne}) = \frac{APVC_j (\$/\text{year})}{AE_j (\text{tonnes}/\text{year})} \quad \text{for all } j = 1, 2, \dots, J \quad (9.5)$$

9.5.2 Supply curve

For a range of different rehabilitation projects spread across a given region, heterogeneity in remediation cost per unit of sediment reduction is often observed due to human-controlled factors (e.g. intensity and scale of remediation works) and natural factors (e.g. rainfall patterns,

soil types). Arranging the cost-effectiveness metric (Equation 5) arising from these different gully rehabilitation projects from lowest (\$/tonne) to highest (\$/tonne) against cumulative sediment reduction reveals the *supply curve* for sediment reduction or 'sediment abatement' for that region. This supply curve for sediment reduction, also referred to as *marginal abatement cost for sediment*, is depicted in Figure 9.4.

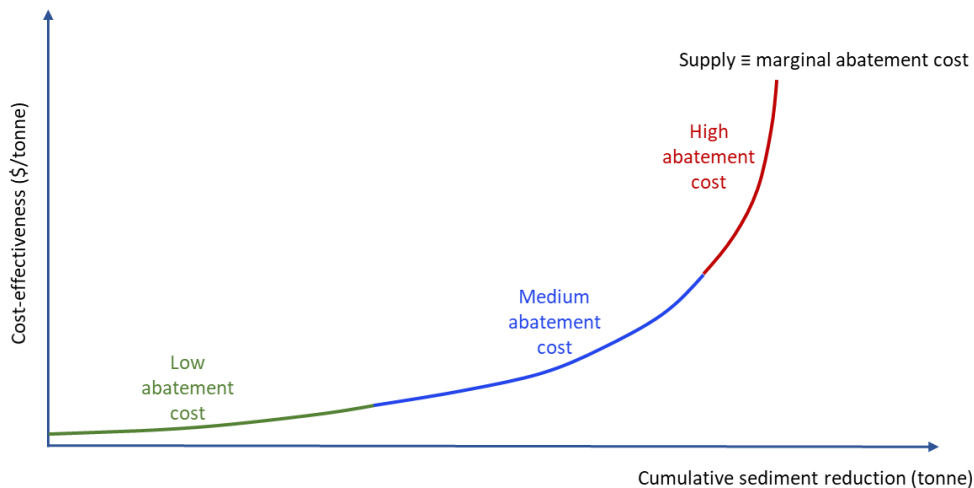


Figure 9.4: Sediment credit supply curve

The supply curve produced from the ranked project- or site-specific cost-effectiveness of gully rehabilitation (\$/tonne) can be used to provide an indication of societal cost of delivering a given total sediment reduction target.

9.5.3 Calculation of gully-specific remediation costs

The cost-effectiveness calculation process undertaken in this study is illustrated in Figure 9.5. Key information required for the cost-effectiveness calculations are: fine sediment yield, sediment reduction estimates, and gully rehabilitation costs. The gully rehabilitation cost estimate is a function of treatment intensities, feasibility of sourcing rock from an on-site quarry, and other ancillary costs. The unit costs of treatment have been compiled through direct contacts and discussions with relevant bodies and their contractors working on gully rehabilitation projects at Strathalbyn and Mount Wickham. The project at Strathalbyn Station is being undertaken over a four-year period under the Innovative Gully Remediation Project, jointly funded by the Queensland Government's Reef Innovation Fund and Greening Australia's Reef Aid Program. This collaborative project focuses on developing cost-effective and scalable options for remediating alluvial gullies on grazing land.

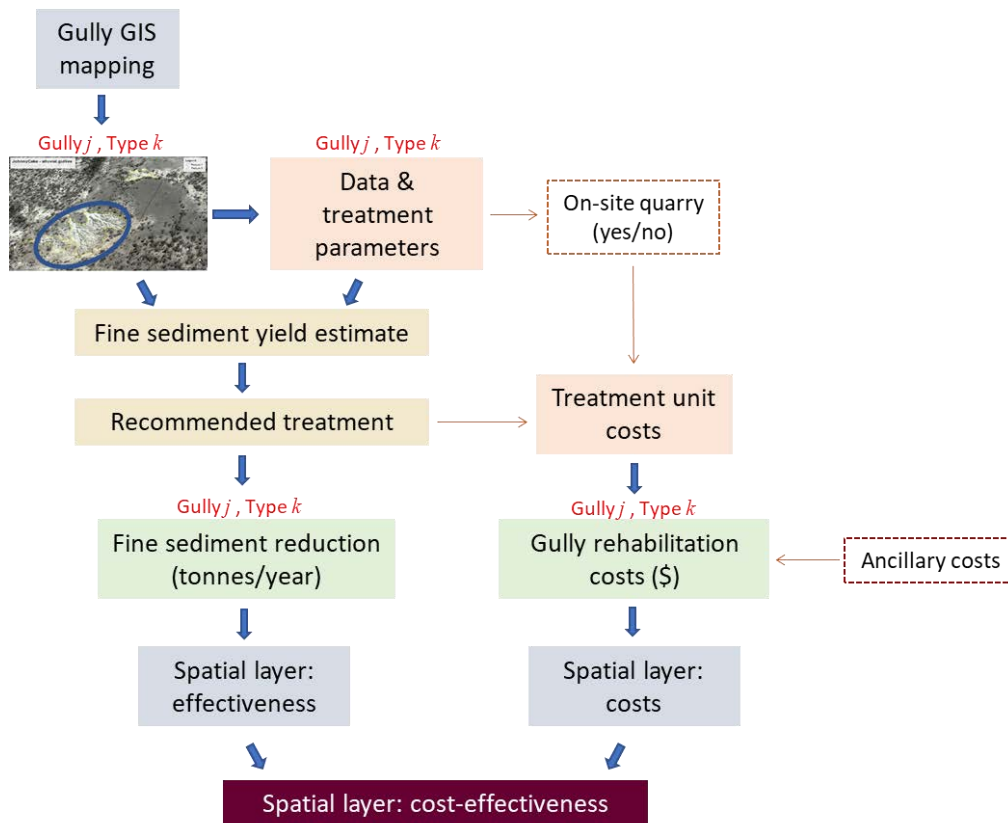


Figure 9.5: Cost-effectiveness calculation process

The gully rehabilitation project at Mount Wickham is undertaken under the North Queensland Dry Tropics Landholders Driving Change program, one of two Major Integrated Projects funded by the Queensland Government. The Mount Wickham site is characterised as a typical large-scale sodic gully making it an ideal demonstration site to trial different remediation methods for a clustered set of alluvial gullies that feature tunnel erosion. Gully remediation projects at Strathalbyn Station and Mount Wickham inform many of the indicative *base costs*, as reported in Brooks et al. (2020b). These base (unit) costs are used to estimate gully- or site-specific ex-ante costs of remediation for other alluvial gullies in the Bowen-Bogie sub-catchment.

Gully-specific on-ground rehabilitation cost is divided into *on-ground treatment costs* and *ancillary costs* (Figure 9.6). The total cost of on-ground treatment is the sum of the costs incurred from earthworks, rock procurement, work on the contributing catchment and stock exclusion fencing to protect the remediated area. Ancillary costs include soil tests, installation of site facilities and infrastructure, engineering design and maintenance. The unit costs listed in Figure 9.6 are the base costs and the corresponding biophysical quantities are gully-specific data derived from LiDAR data, followed by GIS mapping and processing (Brooks et al. 2020b). All ancillary costs, with the exception of soil tests, are taken to be proportional to the total cost of on-ground treatment (Brooks et al. 2020b).

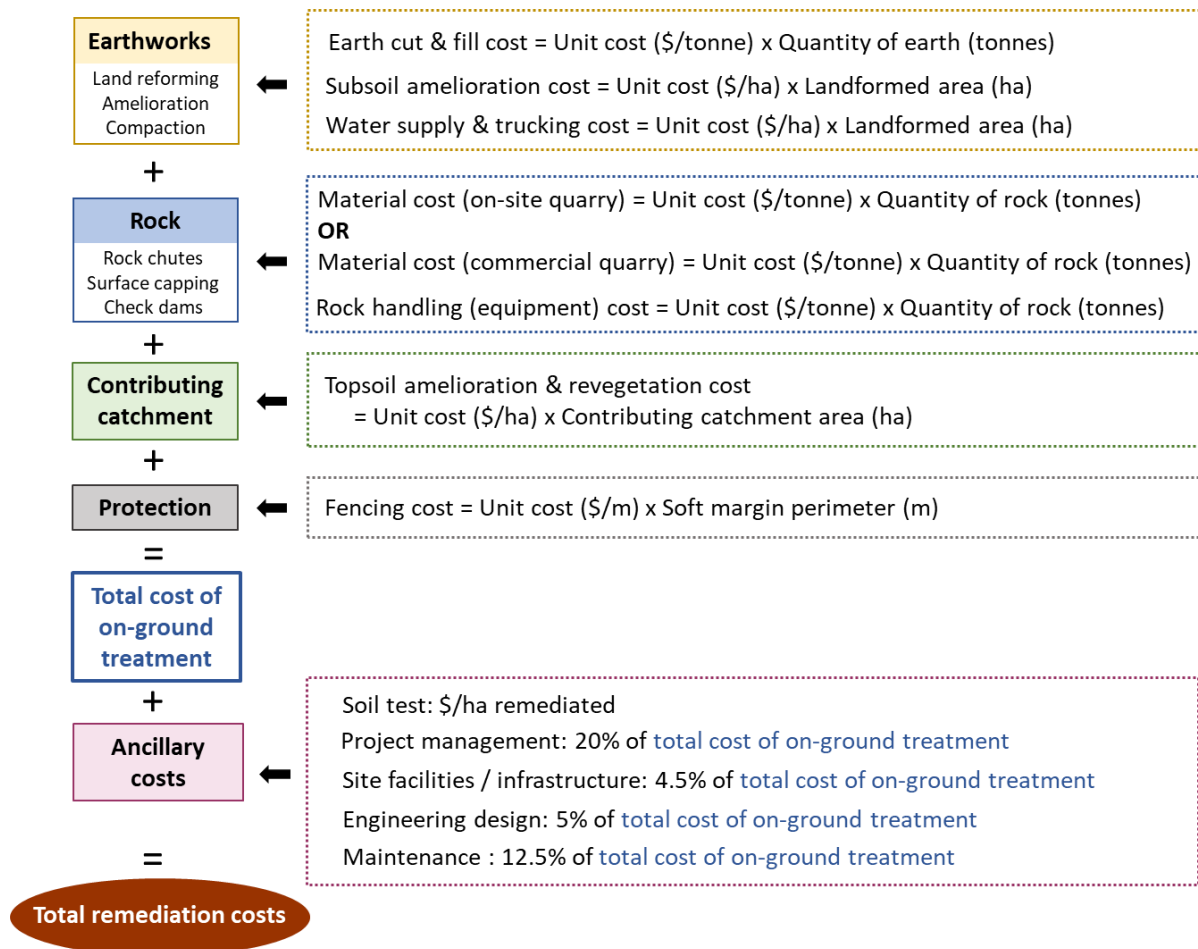


Figure 9.6: Components of gully remediation costs

9.5.4 Variation in treatment intensities

Treatments to reduce soil erosion vary according to predominant soil type, the extent of erosion, whether tunnelling is present or if there is a risk of tunnelling developing, size of the eroding site (including the area of the contributing catchment), availability of resources such as earthmoving equipment, rock and gravel within reasonable proximity (e.g. 50 km), and ease of road access, among others. The objective of remediation also needs to be considered when selecting among remediation options. For example, if the objective is to mitigate against new erosion forming on a small existing gully area, a low intensity treatment involving a combination of fencing for stock exclusion, land reforming, soil amelioration and revegetation may be appropriate. However, if the objective is to stop a large actively-eroding gully from discharging large quantities of sediment, a high-intensity treatment involving engineered structures such as surface armouring, rock chutes, and diversion bunds may be required in addition to the low intensity treatment. An example of variations in gully treatment intensities are shown in Figure 9.7.

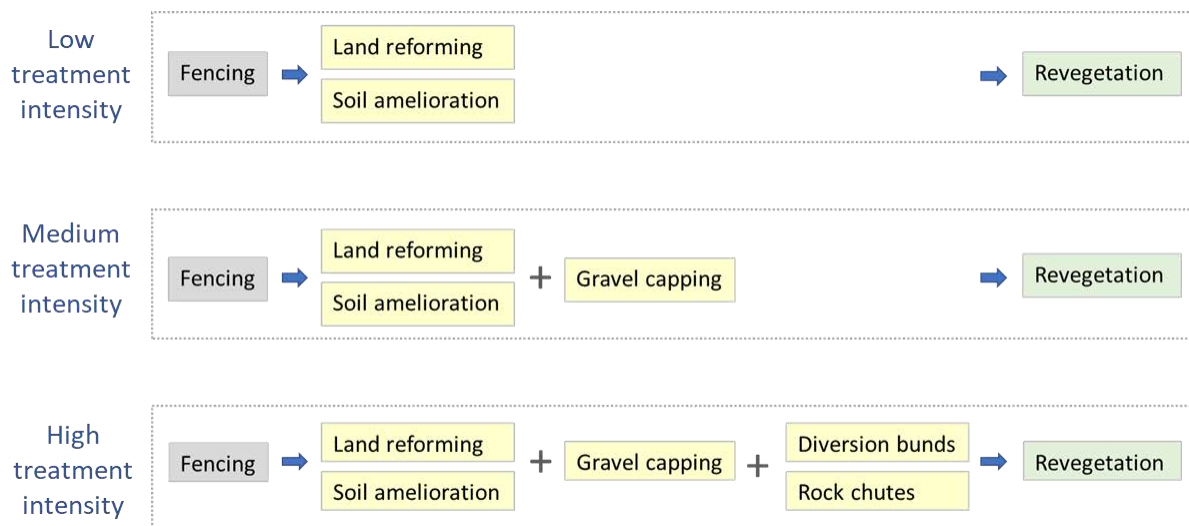


Figure 9.7: An example of variations in gully treatment intensities

9.6 Cost-effectiveness of sediment abatement from remediating alluvial gullies in the Bowen - Bogie sub-catchments in the Burdekin region

This section reports results from recent research by Brooks et al. (2020b) on the cost-effectiveness of fine sediment reduction obtained by remediating alluvial gullies in the Bowen-Bogie sub-catchments of the Burdekin. The Brooks et al. (2020b) study represents a major step forward in the resolution and precision of estimates of sediment yields and potential sediment abatement that would follow from gully rehabilitation. Brooks et al. undertook a detailed study that encompassed more than 22,000 individual gullies across sections of the Bowen and Broken sub-catchments of the Burdekin for which detailed LiDAR data were available. Brooks et al's research was funded by the Landholders Driving Change Major Integrated Project.

Combining these estimates of sediment abatement following remediation with the estimated costs of remediation (Section 9.5 above), compiled from reported expenditures on major gully remediation works completed over the past 12 months, enabled supply curves to be produced for fine sediment credits as shown in Figure 9.8, Figure 9.9 and Figure 9.10. These supply curves indicate that if a maximum budget of \$22M is available for upfront cost, a total annual fine sediment load reduction of 119ktonnes could be achieved at a maximum cost of \$46 per tonne reduction (costs annualised at 7% real discount rate over 25 years). For gullies included within this \$22M budget on upfront cost, the mean cost-effectiveness is \$36/tonne. For comparison, Star et al. (2018) report minimum and median costs of \$26/tonne and \$66/tonne, respectively, for total suspended sediment reductions from streambank and gully remediation in the Burdekin, drawing on analyses from Alluvium (2016) and Rust and Star (2018).

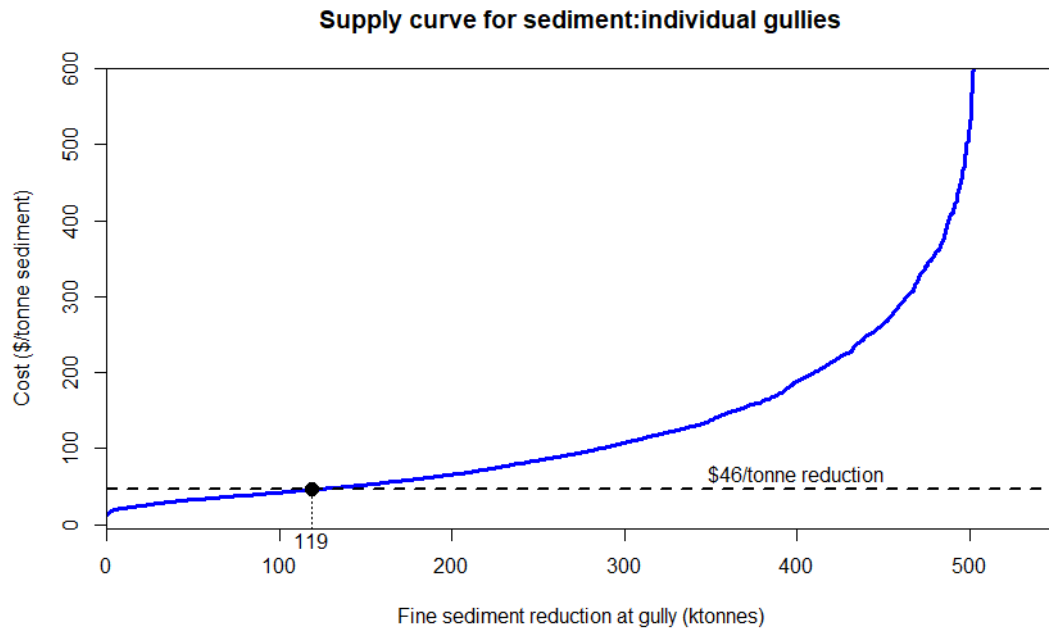


Figure 9.8: Supply curve for fine sediment reduction by remediating individual gullies. Given a \$22M budget limit on upfront cost, a total annual sediment load reduction of 119 ktonnes could be achieved at a maximum cost of \$46 per tonne reduction (annualisation of costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b)).

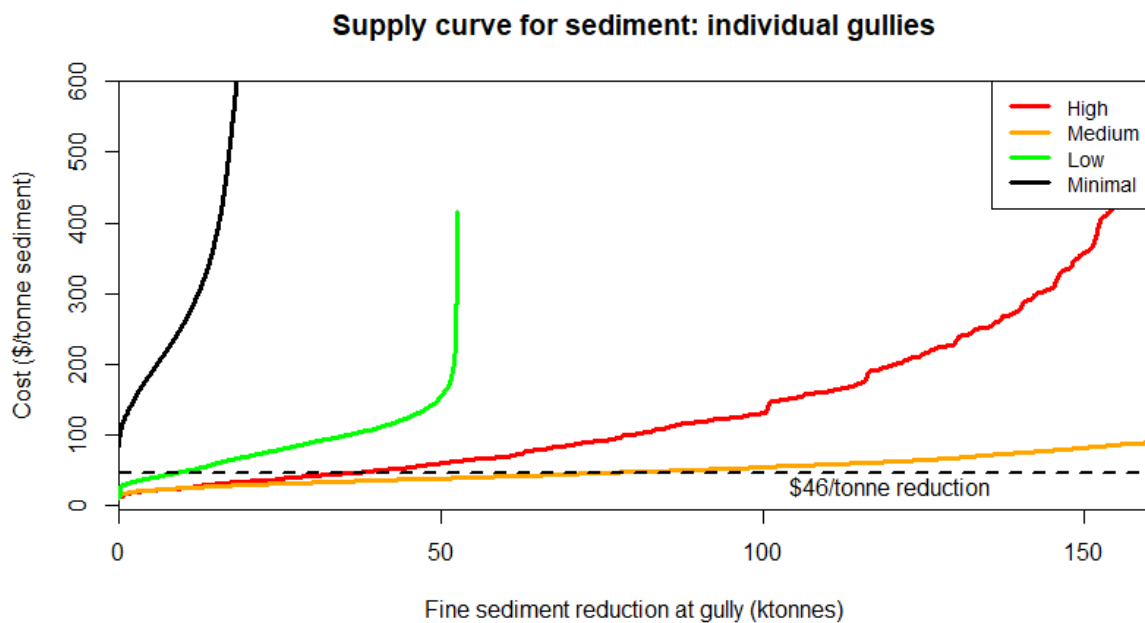


Figure 9.9: Supply curves for sediment reduction by remediating individual gullies: supply grouped by treatment intensity – high, medium, low, minimal. (Annualisation of costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b)).

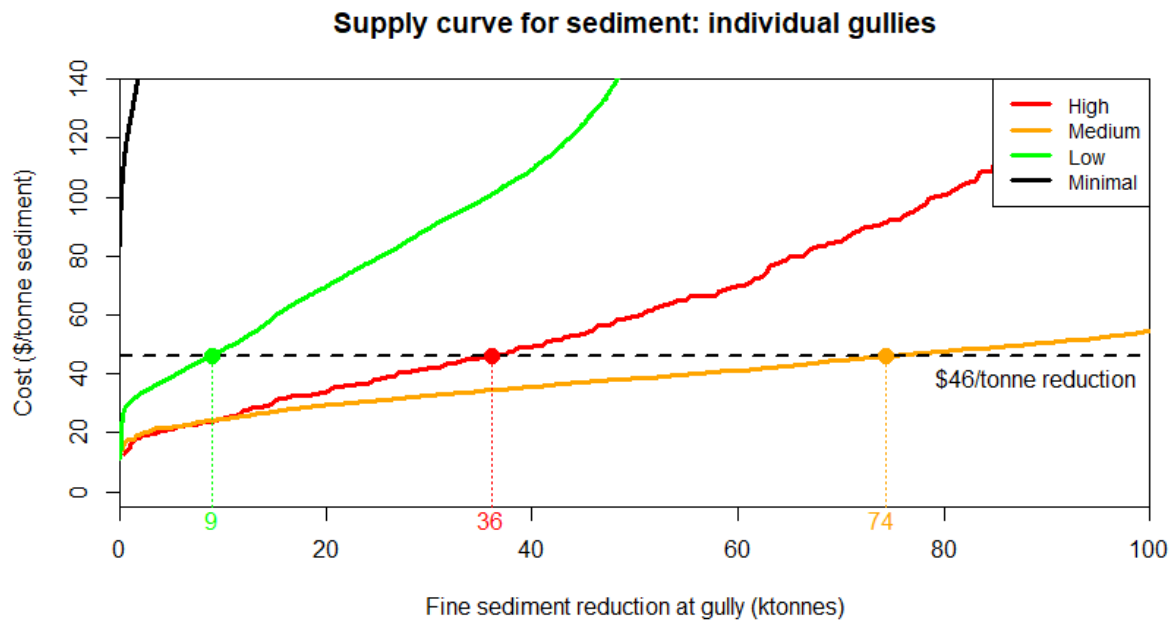


Figure 9.10: Detail of supply curves for sediment reduction by remedying individual gullies: supply grouped by treatment intensity – high, medium, low, minimal. Within a budget of \$22M on upfront cost, remediation would be pursued up to an annualised present value cost of \$46/tonne abated annually. At the upfront cost limit, 36, 74, 9 and 0 ktonnes of fine sediment abatement would be delivered by remediating high-, medium-, low- and minimal-intensity gullies, respectively (Annualisation of costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b)).

A significant innovation in the research of Brooks et al. (2020b) is the ability to identify which particular gullies – from the 22,311 gullies modelled in the Bowen-Bogie – are expected to deliver the most cost-effective sediment reductions. This fine-scale targeting is enabled by high resolution LiDAR data and supported by GIS-derived data on gully morphometrics to predict sediment yield, and GIS-derived data on distances to required resources to inform gully-specific estimates of remediation cost and the appropriate level of treatment intensity.

9.7 Summary

This Chapter has addressed the supply of fine sediment credits from reducing soil erosion in predominantly grazing landscapes, taking the Bowen and Bogie sub-catchments of the Burdekin River as case study examples.

Recent results from research undertaken by Brooks et al. (2020b), funded by the Landholders Driving Change Major Integrated Project indicate that:

- Appropriately-targeted remediation of large alluvial gullies has the potential to deliver substantial reductions in End-of-Catchment fine sediment load.
- Initial results from the Brooks et al, study suggest that around 120,000 tonnes of sediment load could be removed from the Bowen and Bogie sub-catchments (in combination) for a total up-front expenditure of \$22 million.
- This equates to supply of 120,000 tonnes of fine sediment reductions (i.e. 120,000 tonnes of fine sediment credits) for no more than c. \$46 annualised cost per annual

tonne of sediment abated. (Annualisation of up-front and maintenance costs at 7% real discount rate per annum over 25 years). This estimated supply of fine sediment credits will be considered in the light of potential demand for sediment credits in Chapter 10.

- The cost of supplying sediment credits will inevitably rise as the required sediment load reduction increases, but these results suggest that – when appropriately targeted by high resolution spatial prioritisation – remediation of large alluvial gullies appears to provide a valuable source of cost-effective sediment reduction.

10.0 DEMAND FOR SEDIMENT CREDITS

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10. Summary

The mean annual load of total suspended solids (TSS) delivered from 35 GBR catchments to the Reef lagoon has increased by 5.5 times to 17,000 ktonnes/year since European settlement Kroon et al. (2012). Accumulation of fine sediment in marine areas adversely impacts the Reef's ecosystems and water quality (Waterhouse et al. 2017), and is also likely to necessitate increased dredging activities by port operators along shipping channels. Ports along the Reef coastline have variable *maintenance* dredging requirements. Annual maintenance dredging is required for the ports of Gladstone, Townsville and Cairns. Regular maintenance dredging at between two- to five-year intervals is required at Hay Point Port and Mackay Port.

Proponents of *new* dredging projects, or *expansion* of existing dredging projects, are required to offset their residual impacts on Matters of National Environmental Significance as a condition of permit approval. The Reef Trust Offsets Program (Reef Trust, 2017) offers a mechanism whereby project proponents can choose not to undertake their own offsetting activities but instead make a payment in lieu into the Reef Trust Special Account. By paying into this fund, project proponents satisfy the condition for approval and effectively delegate their offset responsibility to the Reef Trust (Commonwealth of Australia 2017).

There is currently no requirement for ports to make payments to offset the environmental impacts of *maintenance* dredging. However, ports with frequent or regular dredging requirements can be regarded as beneficiaries of positive externalities from in-catchment sediment abatement. Government may therefore wish to consider imposing a levy on port operations and hypothecating the proceeds into a fund to finance sediment abatement actions in relevant GBR catchments. Hypothecation of 0.1% of the traded value of alumina, aluminium, coal, LNG and sugar that was traded through Reef ports with frequent (i.e. annual) or regular (i.e. two- to five-yearly) dredging requirements in the financial year 2016-17 would generate in excess of \$32 million annually. Given the costs reported in Chapter 9 for abating sediment erosion from gullies, four years of hypothecated trade revenues would be sufficient to fund in excess of 300,000 tonnes of sediment reduction via gully remediation in the Bowen/Bogie sub-catchments in the Burdekin system

The Chapter proceeds as follows. Section 1 describes the impacts of increased sedimentation on coral reef ecosystems and the pressures from dredging activities on the condition of the GBR and its biodiversity. Section 2 provides an overview of the types and extent of dredging activities undertaken by ports in Queensland. Section 3 explores the potential application of a

levy system on downstream beneficiaries to generate a resource pool to finance upstream abatement activities to reduce sediment runoff. Section 4 concludes.

10.1 Dredging activities and the Great Barrier Reef

Sediment runoff from catchments draining into the GBR has been identified as one of the factors contributing to the continuing decline in the condition of the Reef (Waterhouse et al. 2017b). Fine sediments are of particular concern because of their tendency to remain suspended, and become re-suspended, in the water column for longer periods of time. Dredging activities along shipping channels off Queensland's east coast exacerbate these problems by imposing additional stress on coral colonies through increased turbidity and sedimentation. In their investigation of the environmental impacts of dredging on corals, Erftemeijer et al. (2012) found that sensitive coral species can only survive high turbidity over several days, whilst more tolerant species can survive high turbidity for at least 5-6 weeks. Erftemeijer et al. (2012) also conclude that sensitive coral species can survive high sedimentation rates (i.e. $>400 \text{ mg cm}^{-2} \text{ d}^{-1}$) for a much shorter duration (less than 24 hours) than the highly tolerant species (more than 4 weeks).

With increased runoff from adjacent catchments, sedimentation in the GBR lagoon necessitates that port operators must dredge shipping channels regularly to maintain access. Dredging is conducted using specialised dredging vessels. It involves collecting and removing accumulated materials from one area of sea floor and transporting and dumping those materials at another site. Dredging is integral to port operations to ensure shipping channels are safe for maritime traffic.

Maintenance dredging and capital dredging are carried out by ports along the coastlines of North Queensland (Queensland Ports Association 2013). Maintenance dredging is undertaken regularly to remove siltation from *existing* and *approved* shipping channels, berths and harbour basins in order to maintain those areas at required depth. Capital dredging involves creation of *new* or *deeper* channels, waterways, harbour basins, berths or marinas, and involves dredgers excavating larger volumes of soils and virgin materials from the seabed. Capital dredging was previously referred to as developmental dredging because the process of creating a new harbour or ports begins with developmental or capital dredging followed by routine maintenance dredging thereafter (Pringle 1989). Dredging and disposal of dredged material can negatively affect coastal and marine ecosystems (e.g. mangroves, seagrass and corals) and habitats of important GBR species (e.g. turtles, dugongs, sharks and rays).

This section summarises the pressures from dredging activities on GBR ecosystem functioning and the effects of those pressures on biodiversity, as described in McCook et al. (2015). McCook et al. (2015) report findings from a 2015 synthesis of current knowledge of the biophysical impacts of dredging and disposal on the Great Barrier Reef, as determined by independent panel of experts (The Synthesis Report). The pressures posed by dredging and disposal activities on the physical and chemical environments of the GBR are shown in Figure 10.1. As described in McCook et al. (2015), the direct pressure of dredging occurs on the seabed in which the excavation takes place where silt, clay, mud, rocks, plants and animals are removed. These benthos and substrate materials are then transported to another site for disposal. The direct pressure of disposal of dredged material is burial or covering of habitats and organisms. The process of excavating and disposal alters the sediment dynamics via

resuspension, transport, consolidation and armouring of areas that could extend beyond the direct excavation footprint. The indirect pressures posed by dredge disposal include underwater noise; hydrodynamic changes; release and dispersion of suspended sediments leading to increased turbidity and deposition and emissions of nutrients, carbon and contaminants (if present). The magnitudes of the effects of dredging and disposal activities vary across space and time and are not limited to the activity footprints and timeframes. These impacts are also dependent on many factors including scale of operation; characteristics of dredged material; type of dredging equipment used; prevailing currents, tides and waves; marine water quality condition; and the spatial extent and type of habitats (McCook et al. 2015).

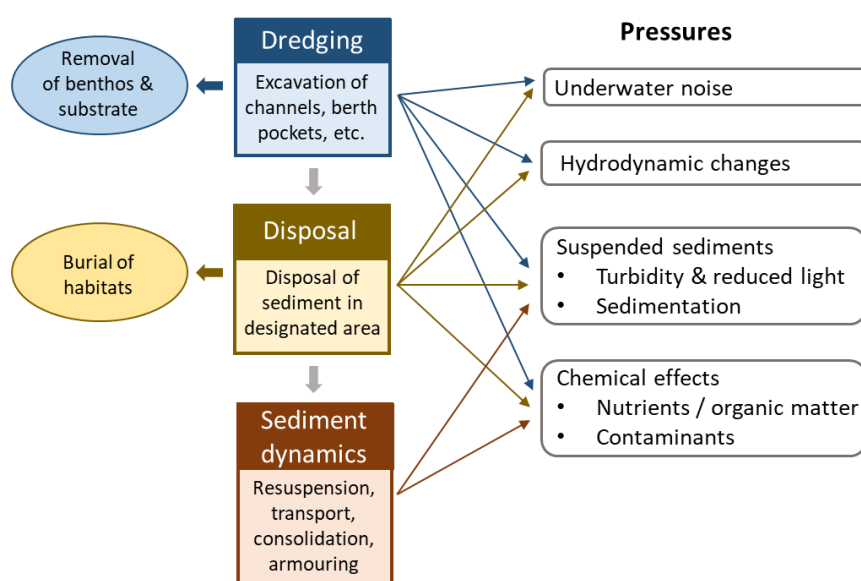


Figure 10.1: Dredging activities and dredge material disposal introduce pressures on the physical and chemical conditions and environments of the Great Barrier Reef. Adapted from McCook et al. (2015, Table 1 p.11 and Figure 5 p.12)

A summary table of the effects on biodiversity of pressures from dredging and disposal activities is provided in McCook et al. (2015, Table 5 p.44). The key biodiversity values evaluated in McCook et al. (2015) and their overall sensitivity are: coral reefs (high, variable), seagrass meadows (moderate), seafloor habitats (moderate, negligible direct evidence), pelagic habitats (moderate to high, limited direct evidence), estuaries/mangroves (moderate), fish populations (moderate), and megafauna/species conservation concern (potentially high, high sensitivity to noise). As the impacted key biodiversity values constitute Matters of National Environmental Significance (MNES) under the EPBC Act 1999 (Figure 10.2), proponents of new dredging projects, or expansion of existing dredging projects, are required to offset their residual impacts on MNES as a condition of permit approval. The Reef Trust Offsets Program offers a mechanism whereby project proponents can choose not to undertake their own offsetting activities but instead make a payment into the Reef Trust Special Account. By paying into this fund, project proponents satisfy the condition for approval and effectively delegate their offset responsibility to the Reef Trust⁷⁷ (Commonwealth of Australia 2017a) (Figure 10.3).

⁷⁷ In the terminology of Chapter 2, the Reef Trust Offsets Program is a fee in-lieu offsetting mechanism.

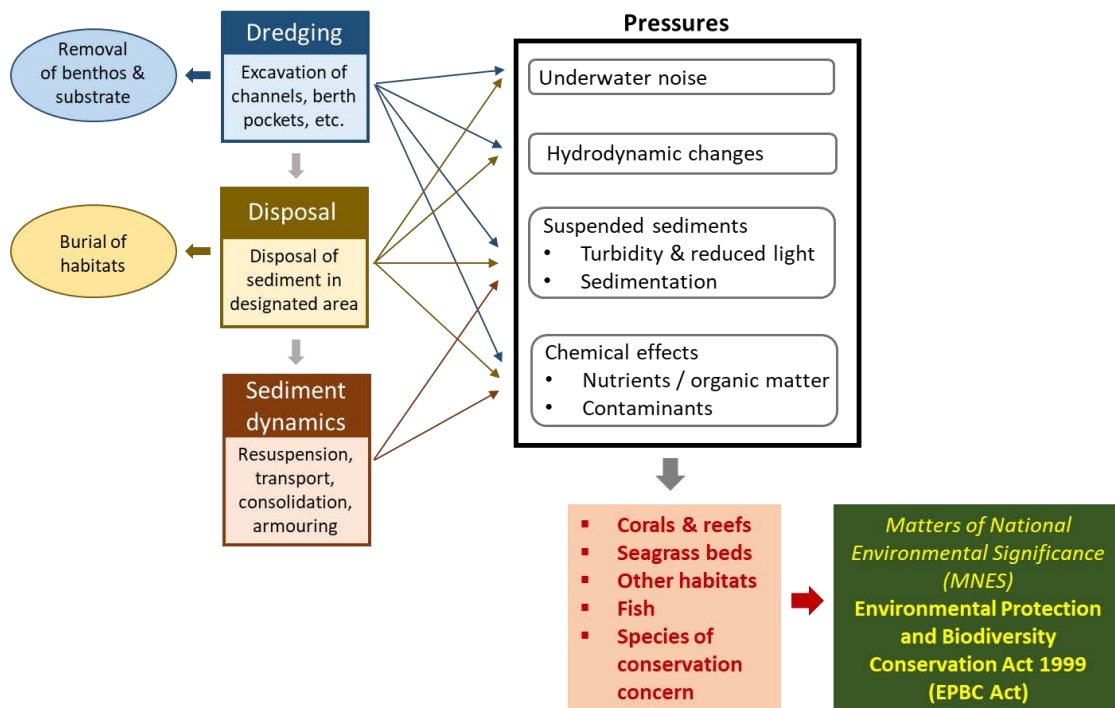


Figure 10.2: The effects on GBR biodiversity of pressures from dredging and disposal triggers application of the Environmental Protection and Biodiversity Conservation Act 1999.

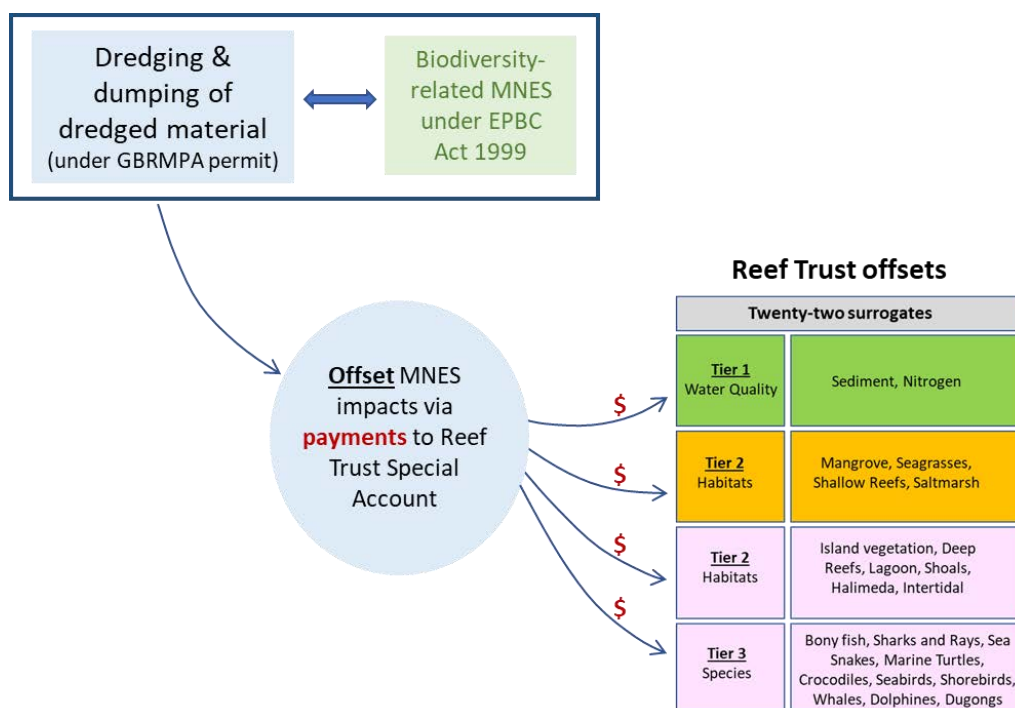


Figure 10.3: Proponents of new dredging projects can choose to offset their MNES impacts by making a specified financial contribution to Reef Trust in accordance with Tiers 1, 2 and 3 payment rates (Reef Trust 2017).

10.2 An overview of port operations in Queensland

Of the 20 recognised ports in Queensland, 12 are within or adjoin the Great Barrier Reef World Heritage Area (GBRWHA), spread over approximately 1,500km along the State's east coast. The ports located along the Great Barrier Reef coast are managed and operated by four government-owned port authorities: Far North Queensland Ports Corporation, Port of Townsville, North Queensland Bulk Ports and Gladstone Ports Corporation.

Far North Queensland Ports Corporation (Ports North) is responsible for the ports of Cairns, Cape Flattery, Cooktown, Mourilyan and Quintell Beach. Port of Townsville is responsible for the ports of Townsville and Lucinda. North Queensland Bulk Ports (NQBPs) is responsible for the ports of Abbot Point, Hay Point and Mackay. Gladstone Ports Corporation is responsible for the ports of Gladstone, Rockhampton (Port Alma), and Bundaberg. Table 10.1 collates information from the 2016-2017 annual reports of these organisations and summarises the major activities undertaken at each port.

Table 10.1: Ports operating in the Great Barrier Reef and their respective activities

Corporation	Major Ports managed	Description of Activities
Gladstone Ports Corporation (GPC)	Gladstone Rockhampton (Port Alma) Bundaberg	The Port of Gladstone handled 120.4 million tonnes of goods in 2016-2017, making it Queensland's largest port by trade. The Port of Gladstone predominantly exports coal, alumina and LNG, with other products including cement, petroleum, grain, and containers. Coal is the dominant commodity with 68.9Mt transported from Gladstone terminals in a year. The alumina/aluminium industry is also significant with 27.9 million tonnes of product being imported and exported each year. 19.4 million tonnes of LNG, worth an estimated at \$32 million, is exported from Gladstone. Throughput at the Port of Rockhampton totalled 233.3 thousand tonnes. Sugar, molasses, wood pellets, petroleum products, ammonium nitrate and silica sand are the major commodities handled. Over 35 products passed through GPC's three ports that are exported to 48 countries. The total number of cargo vessels visiting was 1,908 during the financial year 2016-17.
North Queensland Bulk Ports Corporation (NQBP)	Hay Point Mackay Abbot point	Total tonnage for all commodities through NQBP ports in 2016-17 was 170.5 million tonnes. Hay Point is one of the largest coal export ports in the world. The Port primarily exports metallurgical coal. Total throughput for Hay Point for 2016-17 was 106.4 million tonnes. A total of 1062 bulk carriers visited the Port during 2016-17. Major commodities traded through the port of Mackay include fuel, grain, raw sugar and sugar products and fertiliser. In 2016-17, the Port handled 155 ships carrying a total of 2.9 million tonnes. Abbot Point is Australia's most northerly coal export port. The Port incorporates the Adani Abbot Point Terminal with a current export capacity of 50 million tonnes per annum. In 2016-17 the Abbot Point handled 311 bulk carriers carrying a total of 25.4 million tonnes of coal.
Port of Townsville	Townsville Lucinda	The Port of Townsville is northern Australia's largest container, automotive and general cargo port. It is also Australia's largest sugar, copper, zinc, lead and fertiliser port, servicing the North West Minerals Province, the significant Burdekin sugar-growing district, major cattle and other agriculture precincts and the copper and zinc refineries located in Townsville. It is one of four designated Priority Ports in Queensland under the <i>Sustainable Ports Development Act 2015</i> (Qld). Townsville port also provides an increasingly strategic role in cruise shipping operations, and in Australia's Defence capabilities. The Port of Lucinda is situated approximately 100 kilometres north of Townsville and is primarily dedicated to the export of raw sugar from the Herbert River sugar-growing district. Sugar exports through Townsville and Lucinda ports increased 5% in 2016-2017 to almost two million tonnes, molasses exports increased 22% to 376,000 tonnes, fertiliser trade remained strong at almost one million tonnes. Almost one million tonnes of fuel and bitumen were imported for the region, mineral exports totalled more than 1.26 million tonnes, and over 600,000 tonnes of refined zinc, copper and lead was exported. 189,000 head of live cattle were exported to ASEAN countries from the Port

		of Townsville. Remaining exports include meat and meat by-products, as well as containerised legumes. Trade volumes through the Port of Townsville are forecast to treble over the next 30 years, driven by global demand for base minerals such as zinc and copper, and for agricultural products including fertilisers, sugar, rice, grains and pulses, as well as containerised cargo, fuel and motor vehicle imports to support North Queensland's growing population.
Ports North	Cairns	Ports North handle bulk shipments of sugar, molasses, silica sands, zinc, fuel, fertilisers, log products, minerals, livestock and general cargo. Ports North has extensive marina and tourism facilities, particularly in Cairns. Cairns Seaport is a multi-purpose regional port that caters for a diverse range of customers from bulk and general cargo, cruise shipping and reef passenger ferries. Cairns bulk cargo includes petroleum products, sugar, fertiliser and liquid petroleum gas. The Port has long been the natural consolidation and redistribution centre for supplies that are shipped to the coastal communities north of Cairns as well as the Torres Strait Islands and the Gulf of Carpentaria. The Port is a supply and service centre for the Freeport mine operations in Indonesia with regular mine-servicing shipping operations. Cairns Port is one of Australia's busiest cruising destinations with both major international cruise ships and several domestic cruise vessels operating out of the Port. The Cairns Marlin Marina is a 261 berth Marina accommodating a variety of cruising vessels, superyachts and reef vessel operations servicing the Great Barrier Reef. The Reef Fleet Terminal provides the gateway to the Great Barrier Reef for the one million passengers who visit the Reef from Cairns each year.
	Cape Flattery	
	Karumba	
	Mourilyan	
	Skardon River	
	Quintell Beach	
	Thursday Island,	
	Burketown	
	Cooktown	

Source: Information in this table is compiled from Gladstone Ports Corporation 2016-2017 Annual Report (Gladstone Ports Corporation 2017), North Queensland Bulk Ports Corporation 2016-2017 Annual Report (North Queensland Bulk Ports Corporation 2017), Port of Townsville 2016-2017 Annual Report (Port of Townsville 2017), Ports North 2016-2017 Annual Report (Ports North 2017).

Ports along the Great Barrier Reef are in a variety of coastal environments including estuaries, bays and open water (Department of Transport and Main Roads 2016). In the absence of dredging, waves and currents mobilise sediment particles from the seabed. As dredging is integral to port operations, dredging and disposal of dredge material have inevitably increased turbidity and sedimentation in the water column and surrounding areas. The configuration and orientation of dredged areas influences siltation. As the dredged seafloor floor is lower than the surrounding seafloor it could become a sediment sink, depending on the direction of waves and currents. Thus, configuration and orientation of each port relative to the natural sediment transport pathways plays a key role in the rate of sedimentation which occurs in artificially-deepened areas.

Ports along the coastline have variable dredging requirements ranging from frequent, regular, infrequent, episodic and not required. *Frequent maintenance dredging* is when *annual* maintenance dredging is required. Natural processes result in high sediment transport and siltation rates near ports which, when combined with the port configuration, requires annual dredging to maintain design depths. Annual maintenance dredging is required for the ports of

Gladstone, Townsville and Cairns. Sedimentation rates at these ports can be significantly increased by episodic extreme events such as tropical cyclones and large floods, hence, dredging could be undertaken more than once a year.

Regular maintenance dredging is considered as occurring every two to five years. Regular maintenance dredging is required at Hay Point Port and Mackay Port. These areas experience relatively high rates of natural sediment transport and siltation, but the dredged areas are not as effective sediment traps as those ports which require annual maintenance dredging, potentially due to the port configuration. Regular sedimentation does occur, but at a rate which typically only requires maintenance dredging every two to five years. However, these areas can also be influenced by extreme events which can significantly increase the sedimentation rates and potentially result in the maintenance dredging frequency being increased.

Episodic maintenance dredging is dredging that is undertaken after the occurrence of *extreme events* such as tropical cyclones. Episodic maintenance dredging is required at the Port of Cooktown. It is not possible to predict the frequency of these events with a reasonable level of certainty as they could occur in consecutive years or only once every few decades. *Infrequent maintenance dredging* is associated with *low sedimentation rates*. Infrequent maintenance dredging is required at Port Alma and Abbot Point Port. The natural processes at Port Alma limit sedimentation, with only coarser material accumulating over time. The natural processes at Abbot Point result in low sediment transport rates in the vicinity of the port, which, combined with the naturally deep water where the port is located, result in very low sedimentation rates and infrequent maintenance dredging.

No maintenance dredging is required at the Ports of Maryborough, Lucinda, Mourilyan, Cape Flattery and Quintell Beach. The configurations of these ports have meant that maintenance dredging is not required regardless of the natural sedimentation rates. For instance, the Port of Lucinda and the Port of Cape Flattery are located in deep water while the Port of Quintell Beach consists of a barge ramp which does not require an access channel. Table 10.2 provides lists average siltation rates at major ports along the GBR coastline and identifies the natural mechanism that causes sedimentation to occur. Table 10.3 shows ports that require frequent and regular maintenance dredging and the annual volumes dredged in 2010-2014 as reported in the Maintenance Dredging Strategy for Great Barrier Reef World Heritage Area Ports (Department of Transport and Main Roads 2016). Figure 10.4 shows the locations of ports and their dredging frequency. Annual maintenance dredging is expected to increase in the future under port expansion and capital dredging scenarios.

Table 10.2: Annual sedimentation rates at major ports in the GBR and key mechanisms causing sedimentation.

Port	Average Annual Siltation Rate (m ³ /yr)	Sedimentation mechanisms
Port of Gladstone	190,000	Waves and currents
Port Alma	6,000	Local Currents
Port of Hay Point	100,000	Waves and currents
Port of Mackay	30,000	Waves and currents
Port of Abbot Point	No data	Waves currents and cyclones
Port of Townsville	400,000	Waves and currents
Port of Cairns	400,000	Waves and currents
Port of Cooktown	5,000	Cyclones

Source: McCook et al. (2015)

Table 10.3: Ports requiring frequent and regular maintenance dredging and volume of dredging from 2010 to 2014.

Year	2010	2011	2012	2013	2014
Frequent Maintenance Dredging (annual m ³)					
Gladstone	0	309,000	150,000	0	555,107
Townsville	133,100	814,435	502,940	386,610	521,770
Cairns	314,657	439,443	246,727	421,491	574,447
Regular Maintenance Dredging (every two to five years) (m ³)					
Hay Point	216,070	0	0	0	0
Mackay	0	0	0	98,381	0

Source: Department of Transport and Main Roads (2016b)



Figure 10.4: Locations of ports along the Queensland coastline, their dredging frequency and rail connections.

Dredged spoil acquired through maintenance dredging is normally disposed of offshore within the vicinity of the ports (see Table 10.4 for a list of disposal locations for dredge spoil). Over the longer-term, offshore disposal of dredged material from maintenance dredging in the GBRWHA is expected to continue because it is a more cost-effective option than land-based alternatives (Mocke et al. 2016). As dredge spoil has poor physical properties due to its fine grain size and salinity, significant investments in further material processing would be required for its re-use in relevant industries (Mocke et al. 2016). Mocke et al. (2016) also report wider negative impacts of land-based disposal and/or land reclamation which include: loss of coastal habitats, restricted public access for recreation, reduced aesthetic appeal of waterfronts, loss of habitats for conservation species (e.g. turtle nesting areas) and loss of wetlands.

Table 10.4: Location of offshore disposal of dredge spoil.

Port	Location
Gladstone	Within port limits
Hay Point	Within Great Barrier Reef Marine Park
Mackay	Within port limits
Townsville	Within port limits
Cairns	Within Great Barrier Reef Marine Park

Source: Department of Transport and Main Roads (2016b)

10.3 Levy on downstream beneficiaries of sediment abatement⁷⁸

Large sediment loads are released annually from river mouths along the Great Barrier Reef coastline during the December to March wet season. Sediment released from rivers during high rainfall events is a key process responsible for delivery of *new* sediments to the Great Barrier Reef lagoon. Turbid flood plumes discharged from rivers during high rainfall events can transport sediment tens to hundreds of kilometres across the Great Barrier Reef lagoon (Fabricius et al. 2014). Once sediment settles out from these flood plumes, currents and waves then act as primary drivers of sediment movement regionally throughout the GBR inner-shelf.

There is some uncertainty as to what proportion of sediment re-suspended by waves and currents is derived from recent sediment input from fluvial sources and how much is already naturally present in the marine system (McCook et al. 2015). Kroon et al. (2012) estimate that the mean annual load of total suspended solids (TSS) delivered from 35 GBR catchments to the Reef lagoon has increased by 5.5 times to 17,000 ktonnes/year since European settlement. Using results reported in Kroon et al., Table 10.5 shows the pre- and post-European settlement mean annual discharge of TSS for 10 major rivers and the ports operating at each river mouth. Sediment loads have clearly increased considerably since European settlement and the commencement of cattle grazing. Abatement efforts to reduce sediment export to the Reef would, over time, be expected to reduce the quantity of material that ports would be required to dredge. Ports with frequent (i.e. annual) or regular (i.e. two- to five-yearly) dredging requirements (Figure 10.4) are therefore likely to be beneficiaries of positive externalities from in-catchment sediment abatement. It is reasonable to consider imposing a levy on operations at these ports in recognition of the likely reduction in their dredging costs⁷⁹ and hypothecating the proceeds into a fund to help finance sediment abatement actions in the catchment. This fund could function in a similar way to the Reef Trust offsets fund by supporting landholders to improve grazing management practices, and to finance gully and streambank repair to reduce fine sediment loads reaching the Reef.

The volumes and value of alumina, aluminium, coal, LNG and sugar traded through ports with frequent or regular dredging requirements (Figure 10.4) in the financial year 2016-17 are shown in Table 10.6. Hypothecation of 0.1% of the traded value in these commodities would generate in excess of \$32 million annually. Given the costs reported in Chapter 9 for abating sediment erosion from gullies, four years of hypothecated trade revenues would be sufficient to fund in excess of 300,000 tonnes of sediment reduction via gully remediation in the Bowen / Bogie catchments in the Burdekin system.

⁷⁸ An independent reviewer and reviewers from the Queensland government were averse to this suggestion. It has been retained in the text, but appears unlikely to be taken forward.

⁷⁹ It is likely that ports would still be required to dredge sediment from their shipping channels after gullies in adjacent catchment(s) had been remediated. In recognition of this, the scale of levy proposed is modest as it only seeks to recoup the *reduction* in dredging cost that would follow from gully remediation. As proposed, the levy would only be applied to ports with frequent (i.e. annual) or regular (i.e. two- to five-yearly) dredging requirements. Hypothecation would steer funding from this levy towards remediation of gullies in catchments adjacent to the ports concerned e.g. hypothecation of levy payments from the Port of Gladstone could be used to help fund gully remediation in the Fitzroy catchment.

Table 10.5: Estimates of total suspended sediment (TSS) loads, pre- and post-European settlement for 10 major rivers discharging into the GBR lagoon.

River	Nearby Ports	Current TSS discharge (tonnes/year)	Anthropogenic TSS discharge (tonnes/year)	Pre-European settlement TSS discharge (tonnes/year)
<i>Barron</i>	Port of Cairns (6 km to the south)	100,000	75,000	25,000
<i>Johnstone</i>	Mourilyan (10 km to the south)	320,000	280,000	41,000
<i>Tully</i>	Mourilyan (50 km to the north)	92,000	68,000	24,000
<i>Herbert</i>	Lucinda (6 km to the east)	380,000	270,000	110,000
<i>Haughton</i>	Port of Townsville (40 km to the north)	300,000	270,000	29,000
<i>Burdekin</i>	Abbot Point (55 km to the south); Port of Townsville (90 km to the north)	4,000,000	3,500,000	480,000
<i>Pioneer</i>	Port of Mackay (6 km to the north); Hay Point (16 km to the south)	52,000	2,000	50,000
<i>Plane</i>	Hay Point (14 km to the south)	550,000	500,000	54,000
<i>Fitzroy</i>	Port Alma (adjacent); Port of Gladstone (50 km to the south)	3,400,000	2,300,000	1,100,000
<i>Burnett</i>	Port of Gladstone (135 km to the north)	1,400,000	1,300,000	99,000
Total		10,594,000	8,565,000	2,012,000

Source: (Kroon et al., 2012, Table 1 on p.172).

Table 10.6: Volumes and values of key commodities traded through Reef coast ports that require frequent or regular dredging

Commodity	Port	Volume traded[†] in FY 2016-17 (tonnes)	Commodity price^{##} December 2016 (AUD \$/tonne)	Value Traded (AUD \$ million)
<i>alumina</i>	Gladstone	5,744,470	471	2,706
<i>aluminium</i>	Gladstone	360,151	2,665	960
<i>coal</i>	Gladstone	68,945,908	119	8,205
<i>coal</i>	Hay Point	106,454,734	119	12,668
<i>LNG</i>	Gladstone	19,392,449	364	7,059
<i>sugar</i>	Cairns	304,911	478	146
<i>sugar</i>	Mackay	643,308	478	308
<i>sugar</i>	Townsville	1,372,590	478	656
Total				32,708

[†] Traded volumes from Queensland Ports Trade Statistics Report 2017 (Queensland Government 2018)

^{##} Commodity pricing for December 2016 from <https://www.focus-economics.com/commodities>

10.4 Payments in lieu under State Planning Policy for stormwater pollutant emissions from urban development

State Planning Policy allows unmitigated post-construction pollutant emissions to urban stormwater from urban development sites to be managed by off-site implementation of appropriate catchment-based green infrastructure solutions (e.g. riverbank rehabilitation, gully remediation, wetland construction). The 2018 Draft Implementation Guidance (State of Queensland 2018a) states that post-construction emissions from an urban development site can be managed by developers making appropriate in-lieu payments to local government. Local government then have the responsibility to deliver off-site treatment solutions that provide water quality outcomes equivalent to those that would have been achieved by on-site treatment. These in-lieu payments to local government provide another potential source of demand for purchase of fine sediment credits as off-site offsets for remaining un-mitigated post-construction sediment stormwater runoff.

Alluvium estimated the extent of demand for sediment offsetting from this source (Alluvium, 2016; Appendix B-7, p.203-211). Alluvium's (2016) demand estimates were based on the Queensland Government's 2013 predictions of anticipated annual growth in the number of dwellings in urban centres along the Reef coast between 2016 and 2025 (Alluvium, 2016; Appendix B-7, Table 105, p.205). Population growth along the Reef coast has slowed since 2013, however, and the Queensland Government's most recent predictions of anticipated annual housing growth (Queensland Treasury 2018) are lower than those Alluvium used to predict demand for sediment offsetting from this source. Scaling back the Alluvium (2016) projections of demand for sediment offsets in proportion with the reduction in projected housing growth produces the predicted offset demands in Table 10.7. Alluvium report an extremely low cost-effectiveness (i.e. an extremely high price) for reducing sediment from urban stormwater runoff via on-site management (Alluvium, 2016; Table 25, p.64). This suggests that developers would likely be willing to pay the prices outlined in the sediment

supply curves shown in Chapter 9 to obtain diffuse-source sediment credits for the full extent of the sediment credit demand identified in Table 10.7.

Table 10.7: Predicted demand for sediment credits from major cities along the Reef coast. Derived from projections by Alluvium (Alluvium, 2016; Appendix B-7, p.203-211), revised to account for the Queensland Government's reduced projections of housing growth in the region (Queensland Treasury 2018).

Location	<i>Alluvium (2016) predicted demand (tonnes/year)</i>	<i>Revised predicted demand (tonnes/year)</i>
Cairns	80.4	54.7
Townsville	48.5	28.1
Mackay	33.2	16.3
Rockhampton	17.9	12.7

Although developers' willingness to pay for sediment credits is likely to be very high, the volume of credits required to offset un-mitigated post-construction sediment stormwater runoff from anticipated urban development along the Reef coast through to 2025 are very low. Consequently, it appears unlikely that urban development will provide significant demand for sediment credits unless rates of population and housing growth along the Reef coastline expand rapidly.

11.0 QUANTIFYING VARIATION IN DISSOLVED INORGANIC NITROGEN LOAD GENERATED FROM SEDIMENT EROSION FOR INCLUSION IN WATER QUALITY CREDIT TRADING

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11. Summary

Water quality credit trading has the potential to encourage a shift in agricultural intensity to areas of a catchment which minimise impacts on water quality, whilst also minimising the overall cost impact on industry. However, the cost-effectiveness of credit trading markets as a mechanism for reducing the cost of holding End-of-Catchment pollutant load targets in the face of economic expansion relies on accurate spatial representation of the physical processes and management practices that affect water quality. In water quality credit trading to date sediment and nitrogen have usually been considered separately. Recent research has demonstrated how the contribution of particulate nitrogen in sediment to End-of-Catchment nitrogen loads can be quantified (Brooks et al. 2016; Garzon-Garcia et al. 2017, 2018a).

This Chapter explores how these new understandings of nitrogen loads from sediment erosion can be used to better inform DIN attributions to practice change and gully remediation in water quality credit trading in Great Barrier Reef catchments. These insights will allow water quality credit trading to account more appropriately for the complexity of interlinked nitrogen and sediment emissions to the GBR lagoon, thus increasing the effectiveness and efficiency of this water quality credit trading as an approach for facilitating cost-effective delivery of End-of-Catchment Water Quality Objectives. Following additional investigation, validation and ground truthing this work may in the future contribute to quantifying DIN generated from sediment for the purpose of water quality credit trading.

11.1 Introduction

In addition to fertiliser application, another pathway through which nitrogen may enter the waterways of Great Barrier Reef (GBR) catchments is in association with eroded sediment. Nitrogen is a naturally occurring nutrient associated with soils and consequently with eroded sediment in catchments unaffected by human activities. However, human activities often change the severity, distribution and type of erosion processes that occur. If these changes increase sediment transport, this pathway can represent an important component of the overall impact of human activities on the flux of bioavailable nitrogen delivered to receiving

waters of the GBR catchments. Understanding of the complex linkages between soil characteristics, soil erosion and subsequent delivery of bioavailable nitrogen to receiving waters is still developing. Building on the work of Garzon-Garcia, Burton, Franklin and others (Burton et al., In prep.; Burton et al., 2015; Franklin et al., 2018; Garzon-Garcia et al., 2018a; Garzon-Garcia et al., In prep.; Garzon-Garcia et al., 2017), this Chapter describes how changes in the bioavailable nitrogen load generated from sediment following changes in land management practice can be modelled for inclusion in the nitrogen component of water quality credit trading.

Water quality credit trading aims to incentivise improvements in management practice at locations where these changes will be most cost-effective for reducing End-of-Catchment pollutant loads, whilst also reducing the economic impact of such changes on industry. Water quality credit trading is underpinned by quantified estimates of the impact of different management interventions on water quality throughout a catchment. This is achieved using water quality models. Water quality is modelled in the GBR catchments using the Paddock to Reef (P2R) framework (Figure 1.1) (Ellis and Searle, 2014; McCloskey et al., 2017; Waters et al., 2014).

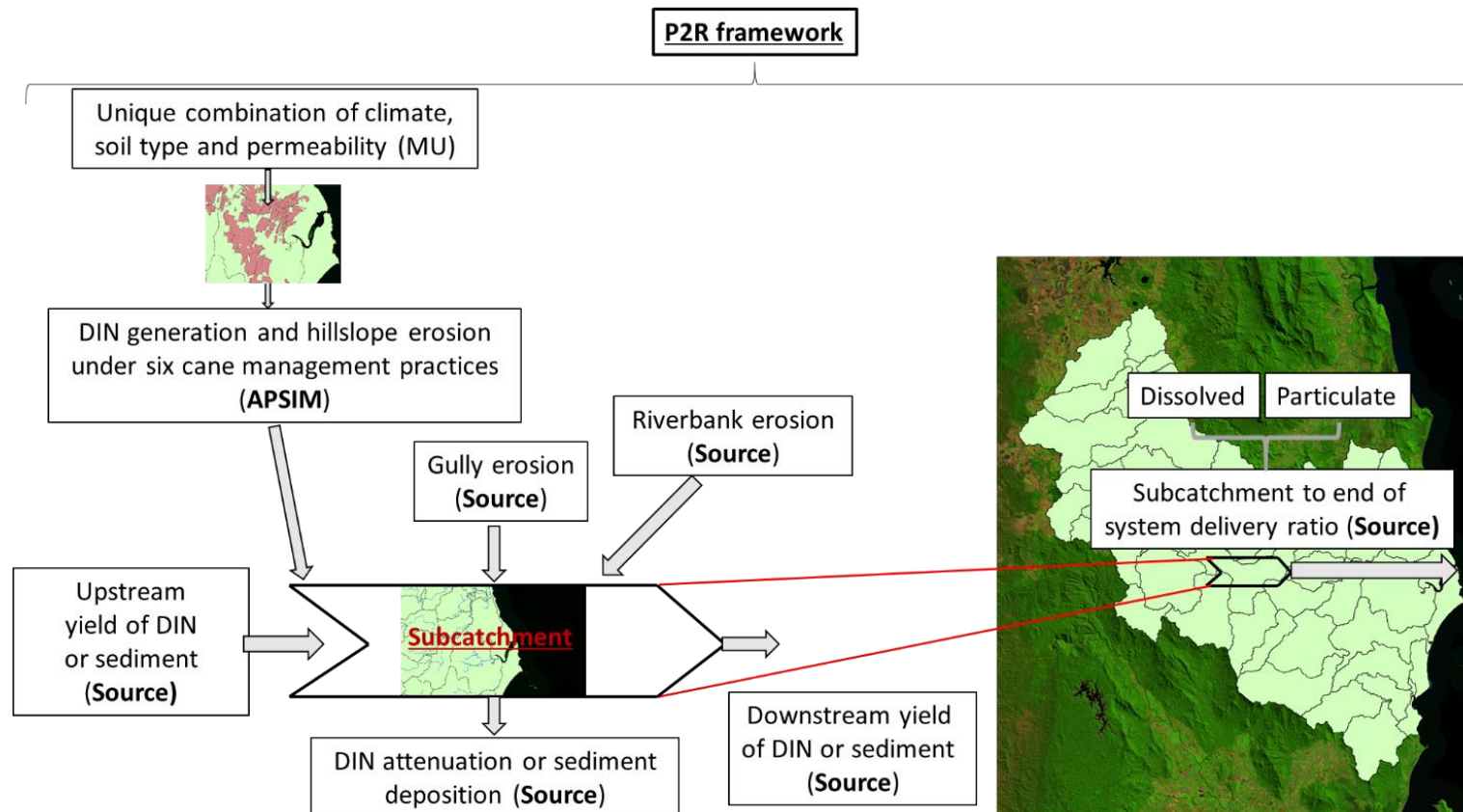


Figure 11.1: Model components within the Paddock to Reef (P2R) modelling framework. Model components within the P2R framework is indicated in brackets (APSIM or Source)

Dissolved inorganic nitrogen (DIN) and fine sediment are key water quality parameters modelled within the P2R framework. DIN is typically represented as coming from fertiliser and travelling to waterways through two pathways. The first pathway is via surface runoff and enters waterways more immediately. The second is via infiltration into groundwater and transport to waterways thereafter follows a more attenuated pathway (referred to as 'drainage'). Fine sediment is represented as being the culmination of contributions from different erosion processes, minus deposition. The erosion processes modelled in the P2R framework are hillslope, gully and riverbank erosion.

Sediment is known to contribute to nitrogen loads in water through particulate nitrogen (PN) (Mayer et al., 1998). The amount of PN in eroded sediment varies depending on the PN content of the parent soil and other characteristics (e.g. granulometry). During the process of erosion and transport to waterways, the proportion of PN typically increases. The magnitude of this increase is represented by an 'enrichment ratio'. Historically the P2R modelling framework used a uniform enrichment ratio to simulate this process. However, enrichment ratio can vary by soil type, land use and erosion process (Burton et al., 2015; Garzon-Garcia et al., 2018a).

Recent work by the Queensland Department of Environment and Science (DES) has quantified the bioavailable nitrogen associated with sediment from combinations of different land uses, soil types and sources of eroded sediment in the Johnstone and Bowen catchments (Burton et al., 2015; Garzon-Garcia et al., 2018a; Garzon-Garcia et al., 2017). In the Johnstone and Bowen catchments, sediment is eroded from surface sources primarily through hillslope erosion, while sediment is eroded from subsurface sources primarily through gully and riverbank erosion.

In the Johnstone catchment, variation in erosion of surface soil⁸⁰ from cane land due to changes in agricultural management practice is simulated through the APSIM component within P2R (Figure 11.1). P2R APSIM estimates have been provided to Project 3.1.6 for DIN load from fertiliser (in runoff and drainage) and for soil erosion for the previously identified six levels of fertiliser management practice (Af, Bf, Bp, Cf, Cp and Df), holding soil management practice constant at level Bp throughout⁸¹. Estimates of DIN load from fertiliser and of sediment load from surface soil erosion are provided for management units (MUs) that comprise unique combinations of climate, soil type and soil permeability (Figure 11.1). DIN from fertiliser and eroded sediment are transported from each MU to End-of-Catchment with different efficiencies. DIN is transported with an efficiency based on dissolved nutrients, while sediment is transported with an efficiency based on particulates (Figure 11.1).

⁸⁰ Erosion of surface soil from agricultural land is usually categorised together with hillslope erosion in the erosion processes literature (e.g. Prosser, 2018; Figure 1, p.3) because in both cases it is primarily surface soil that is eroded.

⁸¹ Sediment management practice Bp as modelled for the Wet Tropics in P2R entails: green cane trash blanketing; cowpea legume fallow; 9 tillage operations including planting the cane and cowpea crop; no tillage through ratoons (Fraser et al. 2013). Under the 2019 Reef Regulations, cane farmers in GBR catchments must implement and maintain measures to minimise soil loss and surface water runoff (State of Queensland, 2019b; Condition SC3, p.4). Maintaining adequate surface cover (e.g. via green cane trash blanketing and fallow crops) is cited as an example of a measure that can be used to comply with this aspect of the 2019 Reef Regulations (State of Queensland, 2019b; Advice to Condition SC3, p.4). These practices are included in sediment management practice level Bp. The project team gratefully acknowledges the assistance of Melanie Shaw in making these data available to the project.

While subsurface sediment can be eroded through both gully and riverbank erosion, gully erosion is a major source of sediment in GBR catchments and is a focus of interest in Project 3.1.6 (Brooks et al., 2016; Shellberg et al., 2010; Thorburn and Wilkinson, 2013; Wilkinson et al., 2015). Gully erosion is a major erosion source in GBR catchments where grazing land uses dominate and is influenced by grazing management practices (Bartley et al., 2013; Brooks et al., 2016). Quantified relationships to describe variation in gully erosion rate with changes in grazing management practice are not currently available. However, quantified estimates of gully erosion rate at subcatchment resolution are available within the Source Catchments model. Research is on-going to determine erosion rates at individual gully resolution for alluvial gullies in the Bowen catchment, but full results from this work are not yet available (Andrew Brooks, pers. comm.)

The relationships mentioned above that predict the bioavailable nitrogen contribution from sediment erosion have been used to revise P2R estimates to account appropriately for different land uses, soil types, soil characteristics, erosion sources and DIN-generating processes (Garzon-Garcia et al., 2018a). However, to utilise this new understanding in water quality credit trading, these relationships need to be applied to predict how bioavailable nitrogen from eroded sediment changes as land management practices are improved or as gullies are remediated. For example, for each of the six fertiliser management practices simulated within APSIM there is an associated estimate of surface soil loss via hillslope erosion from cane land. However, previously the extent to which hillslope erosion contributes to the total bioavailable nitrogen load generated from sediment has been unclear, as has the magnitude of its contribution to total DIN load compared to the DIN originating from fertiliser. Improved quantification of the bioavailable nitrogen load contributed by hillslope erosion from cane paddocks would further improve understanding of spatial variation in the cost-effectiveness of improvements in fertiliser management practice for reducing End-of-Catchment DIN load.

The aim of this Chapter is to use recently derived quantified data on the bioavailable nitrogen load generated via sediment erosion in relation to changes in cane management practice to inform water quality credit trading. This involves representing spatial variation in erosion magnitude for each management practice, and the associated enrichment of bioavailable nitrogen, together with efficiency of delivery to End-of-Catchment. This analysis was performed to test the hypothesis that improved understanding of such physical processes could inform a more complete representation of important considerations for integrated management of bioavailable nitrogen pathways in the GBR catchments.

11.2 Processes that generate bioavailable nitrogen associated with sediment

Four DIN-generating processes associated with soil erosion were identified by Garzon-Garcia et al. (2018a). These processes vary spatially across a catchment and in the timeframe over which they occur, however each will contribute to the total DIN load generated from sediment. These processes are:

- Easily solubilised DIN and dissolved organic nitrogen (DON) in the pore water of eroded soil and/or leached from soil organic matter (e.g. vegetation litter) entering

waterways through either runoff or drainage (referred to subsequently as DIN solubilisation);

- Mineralisation of the organic fraction of dissolved nitrogen that has been solubilised from eroded soil (referred to subsequently as DON mineralisation);
- Mineralisation of the organic fraction of particulate nitrogen associated with eroded sediment into DIN (referred to subsequently as particulate organic nitrogen, or PON, mineralisation);
- Desorption of particulate inorganic nitrogen (PIN) associated with eroded sediment through cation exchange processes (referred to subsequently as PIN desorption).

For each of the above DIN generating processes, Garzon-Garcia et al. (2018a) created a 30 x 30 m grid reflecting the DIN load (kg) generated per kilogram of eroded sediment for each soil type and land use combination. These results are available for surface soil only in the Johnstone catchment, and for surface and subsurface soil in the Bowen catchment.

The bioavailable nitrogen load generated by the DIN solubilisation, DON mineralisation and PON mineralisation processes was transported to the end of catchment with the same delivery ratio as dissolved nutrients, while bioavailable nitrogen load generated by PIN desorption was transported to the end of catchment with the same delivery ratio as particulates (Figure 11.2) (Garzon-Garcia et al., 2018a).

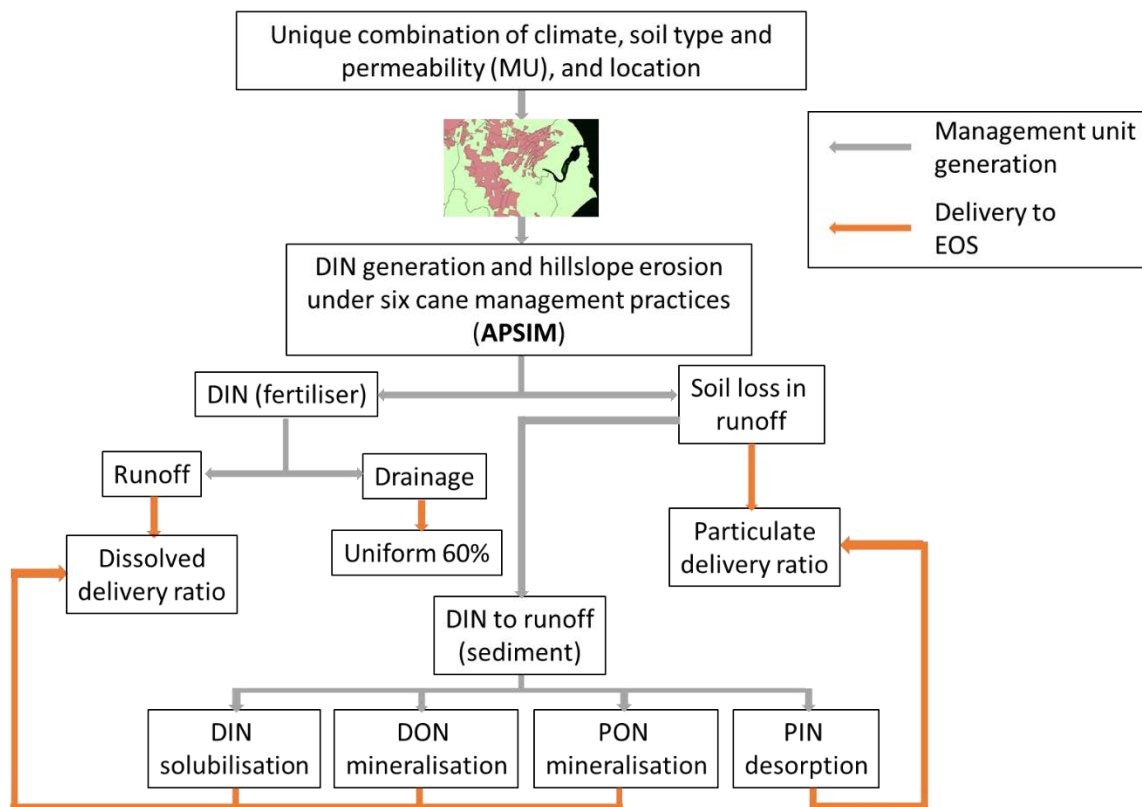


Figure 11.2: The bioavailable nitrogen generating processes associated with sediment and their delivery ratios to the End-of-Catchment in the Johnstone.

11.3 Methods

11.3.1 Johnstone

In the Johnstone catchment, the contribution of hillslope erosion from cane land to the generation of bioavailable nitrogen for each management unit (MU) and management practice modelled by APSIM was determined using the following steps:

- Surface soil loss for each MU and management practice estimated in APSIM runs provided by P2R was expressed in tonnes hectare⁻¹ year⁻¹. Soil loss rate was estimated for each MU and management practice initially assuming a constant slope of 0.2% and a slope length of 100 m. Following advice from P2R modellers (Melanie Shaw, pers. comm.), these initial estimates of soil loss rate were modified by introducing MU-specific slopes which were derived during the P2R modelling. This varying slope was capped at 8% following advice from P2R modellers (Melanie Shaw, pers. comm.). The uniform slope length of 100 m was retained for all MUs.
- Soil loss per MU in tonnes year⁻¹ was calculated by multiplying the modelled soil loss rate by the MU polygon area (in hectares).
- The 30 x 30 m rasters generated by Garzon-Garcia et al. (2018a), which reflected variation in the four DIN generating processes with spatial location and land use, were converted from kg DIN per kg of eroded soil to report kg of DIN per *tonne* of eroded soil.
- The APSIM MU shapefile was overlaid on the 30 x 30 m rasters for the four DIN generation processes. The ArcGIS Zonal Histogram tool was then applied to each MU to determine the proportions of the MU at particular DIN generation potentials for each of the four DIN generating processes.
- For each of the four DIN generating processes, the predicted soil loss from each MU (in tonnes year⁻¹) was multiplied by the proportions of the MU at particular DIN generation potentials to determine the total bioavailable nitrogen load (kg year⁻¹) generated by sediment erosion for that MU and bioavailable nitrogen generation process.
- The bioavailable nitrogen generated from each of the four processes was multiplied by the appropriate delivery ratio (dissolved for DIN solubilisation, DON mineralisation and PON mineralisation; and particulate for PIN desorption) to determine bioavailable nitrogen load delivery from that MU and process to the end of the catchment.
- The end of system bioavailable nitrogen load (represented as the most bioavailable form of nitrogen, DIN) from each process was summed to determine the total DIN load (kg year⁻¹) generated from sediment erosion for each MU under each fertiliser management practice.

11.3.2 Bowen

In the Bowen catchment, Source model estimates of the average contribution of hillslope and gully erosion to sediment yield and to DIN load were available at subcatchment resolution (Garzon-Garcia et al., 2018a). These data were used to derive the average rates of DIN generated per tonne of eroded sediment (kg DIN produced per tonne of sediment erosion) for both hillslope (surface soil) and gully (subsurface soil) erosion. Average rates of DIN

generation from erosion were derived to illustrate the potential reduction in DIN generation from sediment that could be achieved from rehabilitation measures that reduce the sediment yield from gully erosion.

11.4 Results

11.4.1 Johnstone

There was a large difference in soil erosion estimates from APSIM depending on whether a constant or varying slope was assumed per MU (Figure 11.3 and Figure 11.4). Consequently, estimates of the DIN load generated from sediment varied substantially depending on which erosion estimate was used (Figure 11.5 and Figure 11.6). For estimates derived using varying slope, some management units were clear outliers (Figure 11.6).

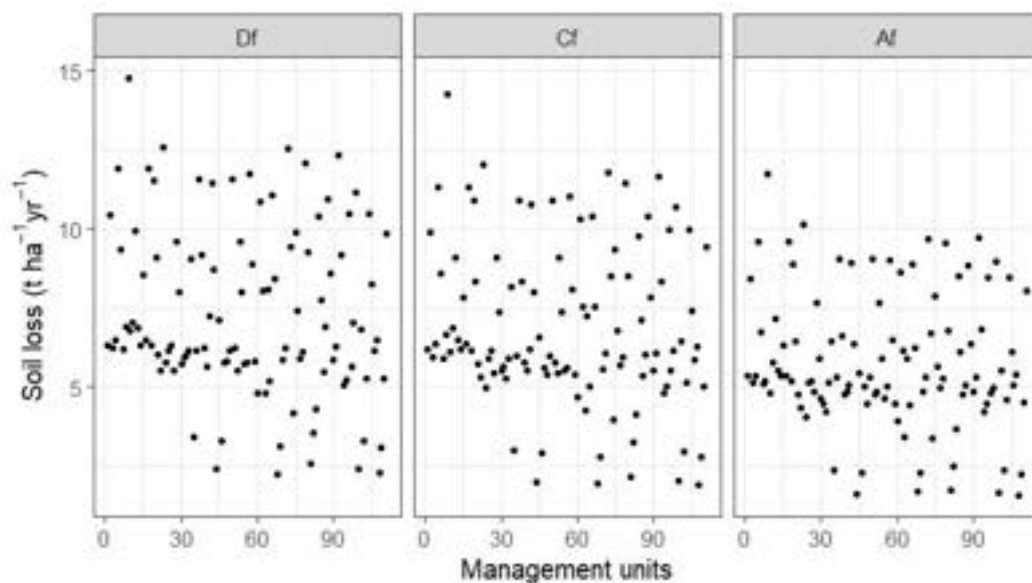


Figure 11.3: Sediment yield from hillslope erosion on cane land, assuming a constant slope of 0.2%. Variation across management units is shown for fertiliser management practices Df, Cf and Af.

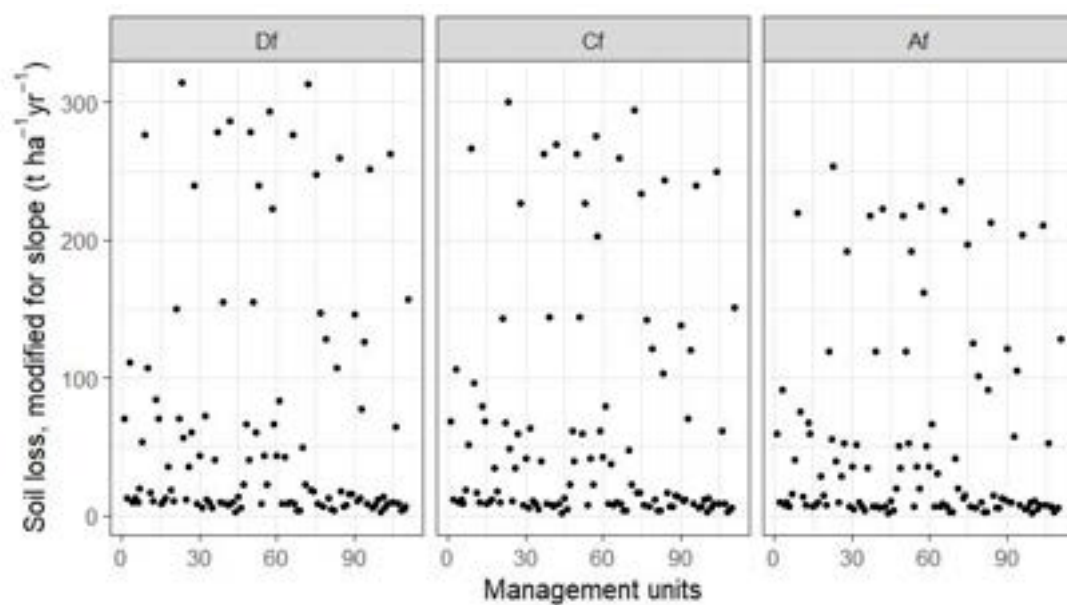


Figure 11.4: Sediment yield from hillslope erosion on cane land, assuming MU-specific slopes (capped at 8%). Variation across management units is shown for fertiliser management practices Df, Cf and Af.

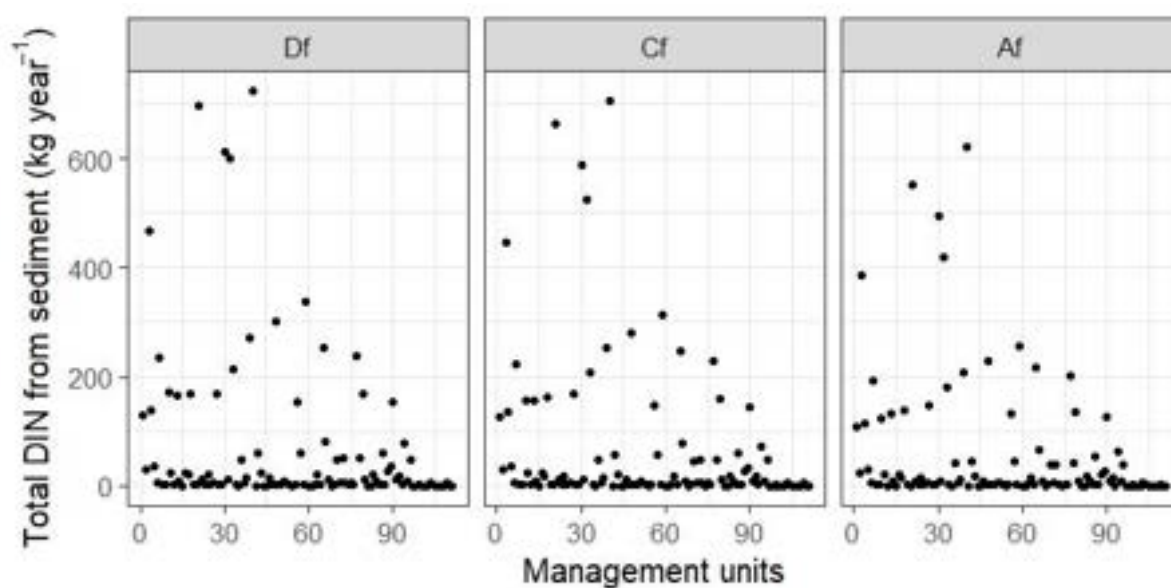


Figure 11.5: DIN generated by sediment yield from hillslope erosion on cane land, assuming a constant slope of 0.2%. Variation across management units is shown for fertiliser management practices Df, Cf and Af.

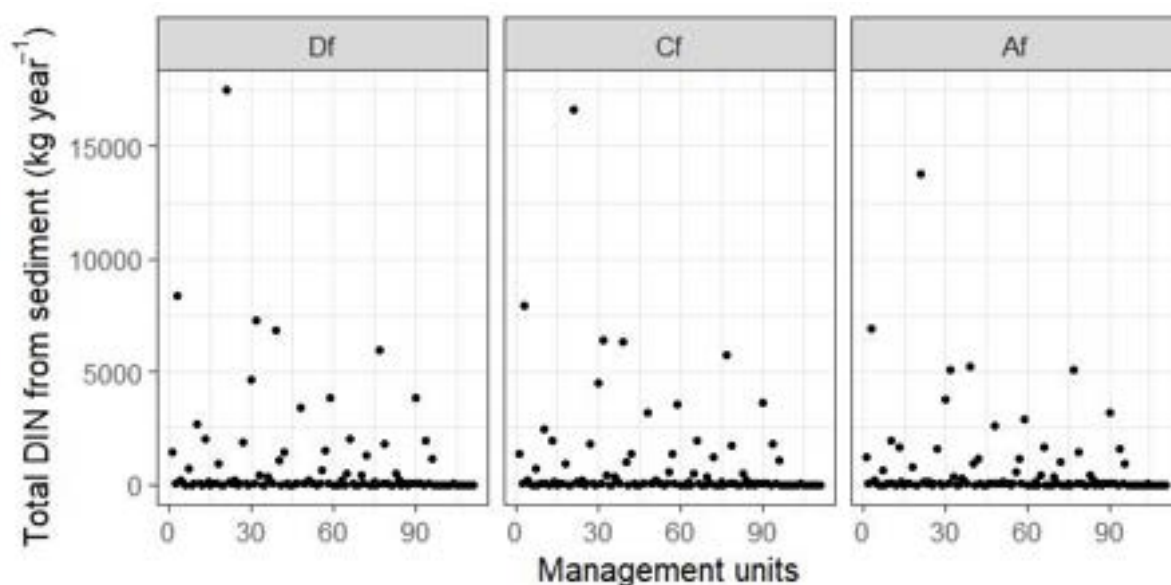


Figure 11.6: DIN generated by sediment yield from hillslope erosion on cane land, assuming MU-specific slopes (capped at 8%). Variation across management units is shown for fertiliser management practices Df, Cf and Af.

The magnitude of DIN load generated by sediment is small when compared to the total DIN load generated from fertiliser (Figure 11.7), similar to the findings of Garzon-Garcia et al. (2018a). However, in some instances, the DIN load generated from fertiliser in runoff (as opposed to the DIN load generated from fertiliser in drainage) is comparable in magnitude to the DIN load generated by sediment using MU-specific slopes (Figure 11.8 and Figure 11.6).

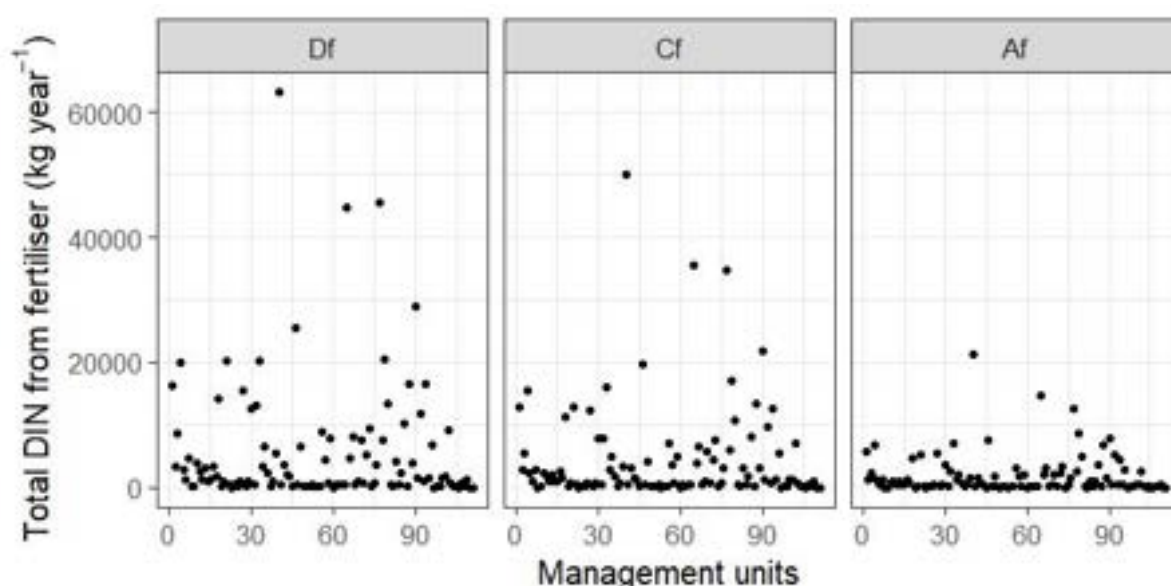


Figure 11.7: DIN load generated from fertiliser application and transported to waterways through both runoff and drainage. Variation across management units is shown for fertiliser management practices Df, Cf and Af.

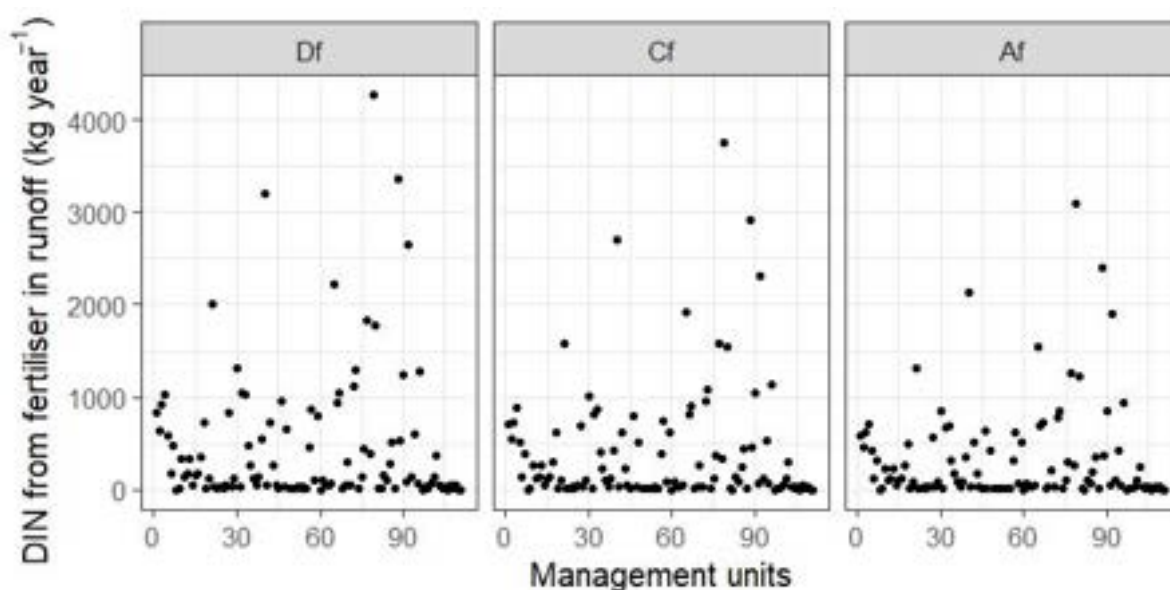


Figure 11.8: DIN generated from fertiliser application and transported to waterways through runoff. Variation across management units is shown for fertiliser management practices Df, Cf and Af.

While the DIN load generated from fertiliser in runoff and the DIN load generated by sediment (estimated using MU-specific slopes) are comparable in magnitude in some instances, the DIN load generated by sediment is much more variable across the MUs. This results in the DIN load generated from sediment consistently having a higher mean value than the DIN load from fertiliser in runoff for each management practice, but a consistently lower median value (Table 11.1).

Table 11.1: Comparison of mean and median soil loss and sources of DIN load for three fertiliser management practices (Df, Cf and Af) in the Johnstone catchment

Water quality parameter	Management practice					
	Df		Cf		Af	
	Mean	Median	Mean	Median	Mean	Median
Soil loss, constant slope (t ha ⁻¹ yr ⁻¹)	7	7	7	6	6	5
Soil loss, MU-specific slope (t ha ⁻¹ yr ⁻¹)	67	18	64	17	54	14
DIN from sediment (kg yr ⁻¹)	67	6	63	6	54	5
DIN from sediment, MU-specific slope (kg yr ⁻¹)	798	19	751	18	632	15
DIN from fertiliser (kg yr ⁻¹)	5,563	1,223	4,259	952	1,808	424
DIN from fertiliser, in runoff only (kg yr ⁻¹)	464	132	393	112	319	90

Management practices which reduce the DIN load in runoff may have more immediate impacts on reducing DIN loads entering the GBR lagoon than interventions aimed at DIN in groundwater (Brodie et al. 2012; Wang et al. 2013). Therefore management practices which reduce soil erosion may also have a more immediate impact on reducing DIN loads entering

the GBR lagoon. If management practices which reduce total DIN load over shorter timeframes are regarded as a priority (e.g. as in Star et al., 2018), the results presented here demonstrate how DIN load from fertiliser in runoff and DIN load associated with sediment could both be used to inform spatial variation in the attribution of DIN credits from particular changes in fertiliser management practice (Figure 11.9).

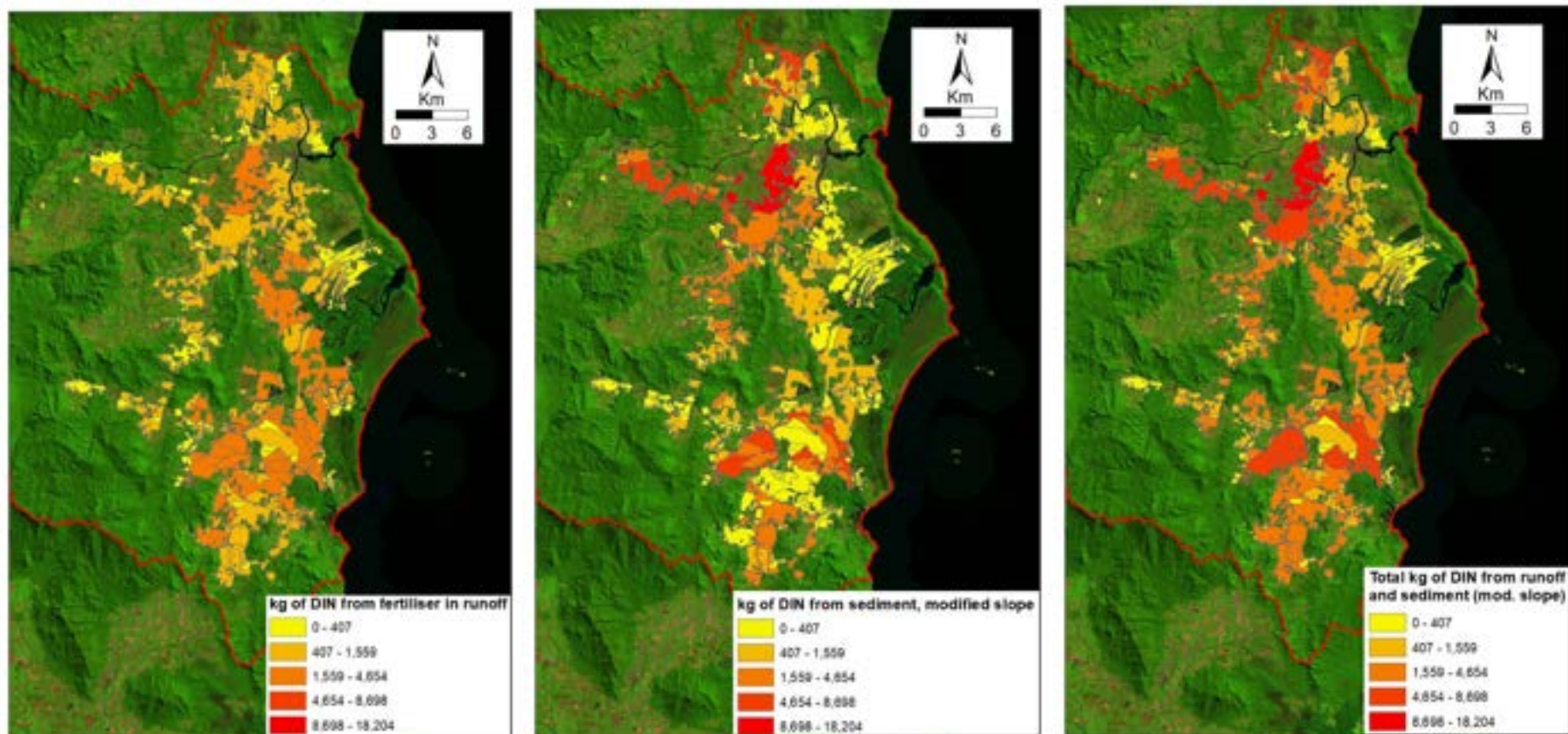


Figure 11.9: Johnstone catchment: spatial variation in DIN load (kg year⁻¹) generated from fertiliser application in runoff (left panel), from sediment erosion on cane land (centre panel), and combined (right panel) for the Cf cane fertiliser management practice.

Interestingly, while the overall magnitude of DIN loss generated from fertiliser reduces as fertiliser management practice improves, the ratio of DIN in runoff to DIN in drainage increases with improving practice (Figure 11.10). This implies that as management practices improve, for example under widespread compliance with the 2019 Reef Regulations (State of Queensland 2019d), DIN generated from sediment may become more important on a relative basis.

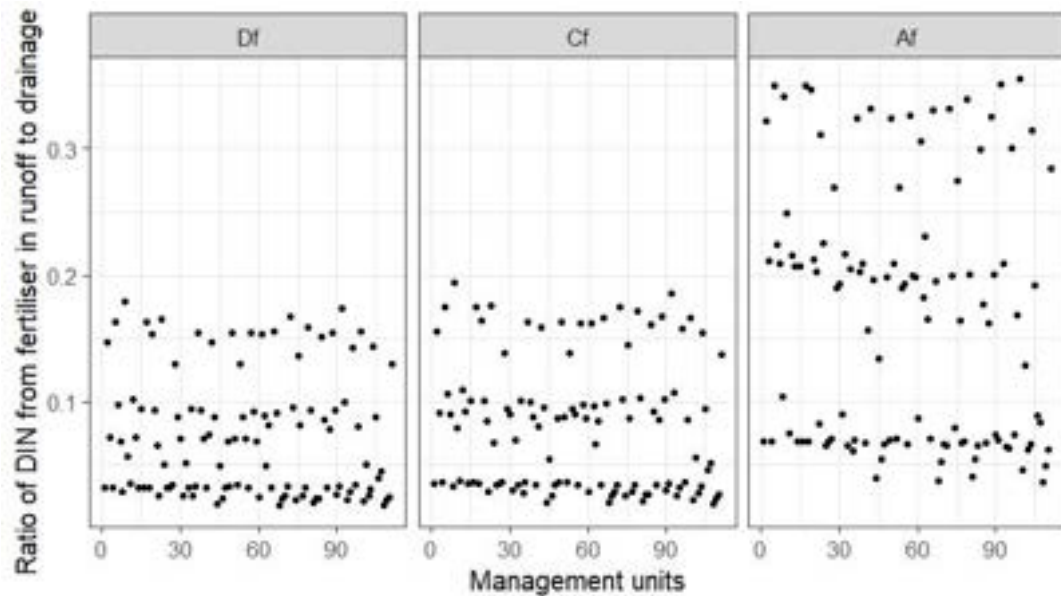


Figure 11.10: Variation in the ratio of DIN generated from fertiliser in runoff to drainage for three fertiliser management practices (Df, Cf and Af).

11.4.2 Bowen

For the subcatchments modelled by Source for the Bowen catchment in the Burdekin Region, gully erosion generally dominates sediment yield (tonnes) (Figure 11.11). However, due to surface sediment having a higher PN enrichment ratio, hillslope erosion dominates DIN load generation from sediment in the catchment (Figure 11.12). This results in the kg of DIN generated per tonne of sediment eroded being an order of magnitude higher for hillslope erosion than for gully erosion (Figure 11.13).

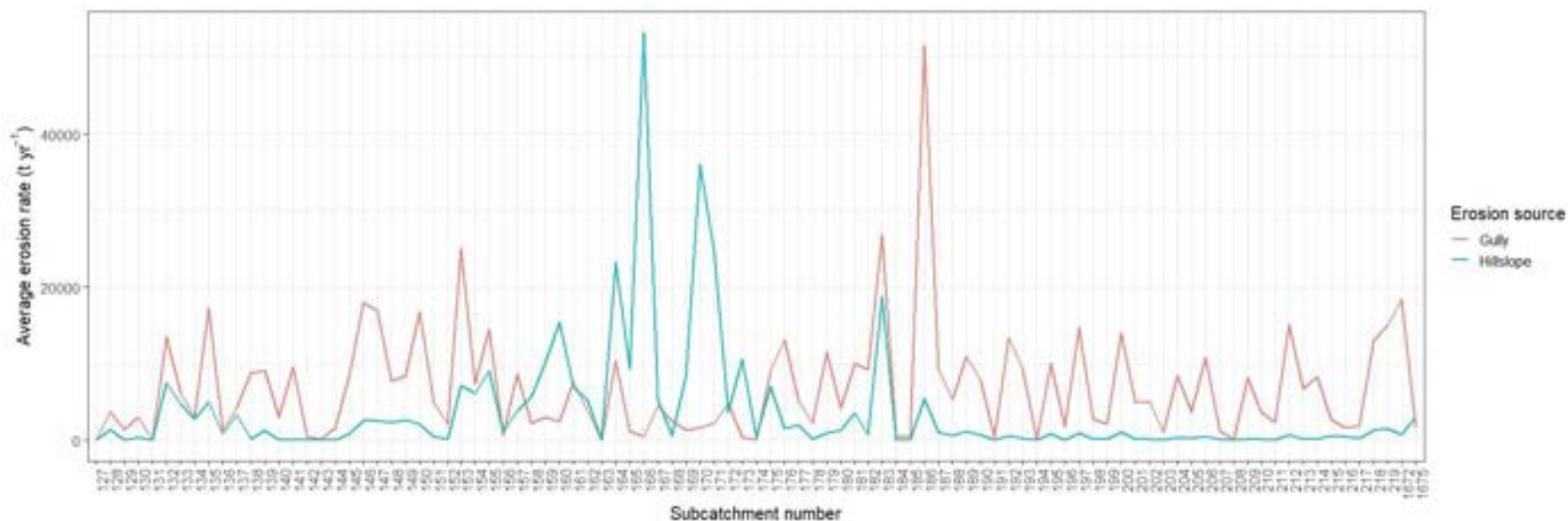


Figure 11.11: Source model estimates of average gully and hillslope erosion rate for subcatchments within the Bowen catchment

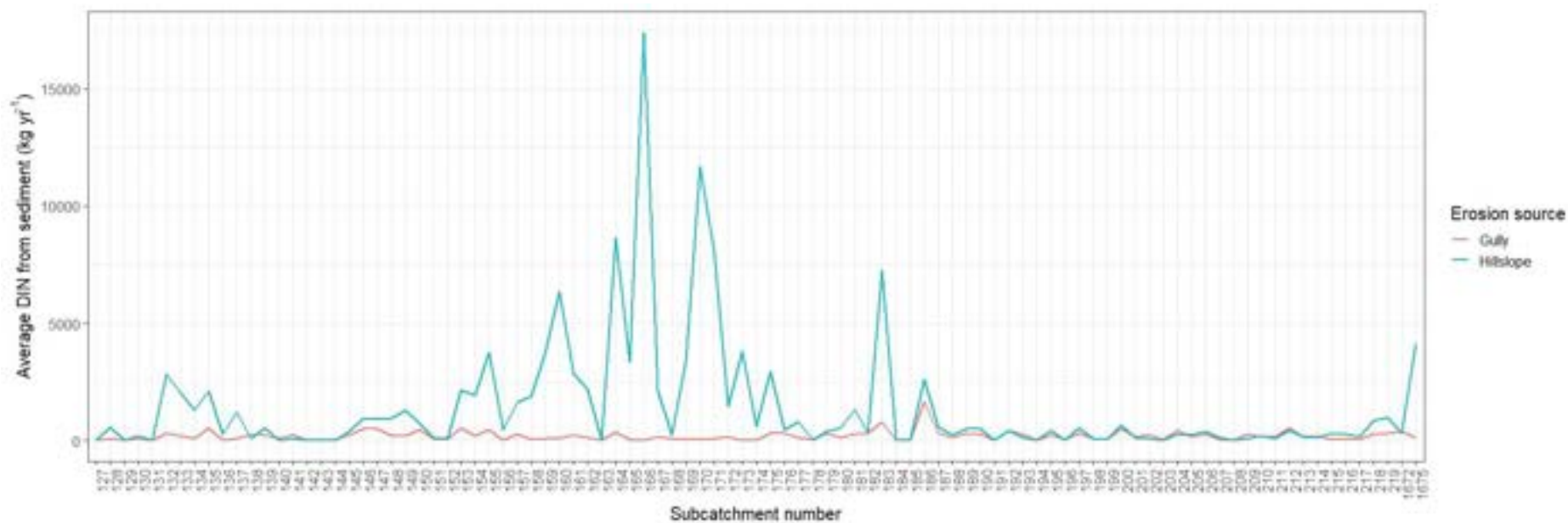


Figure 11.12: Average DIN load (kg year⁻¹) associated with sediment from gully and hillslope erosion for subcatchments within the Bowen catchment

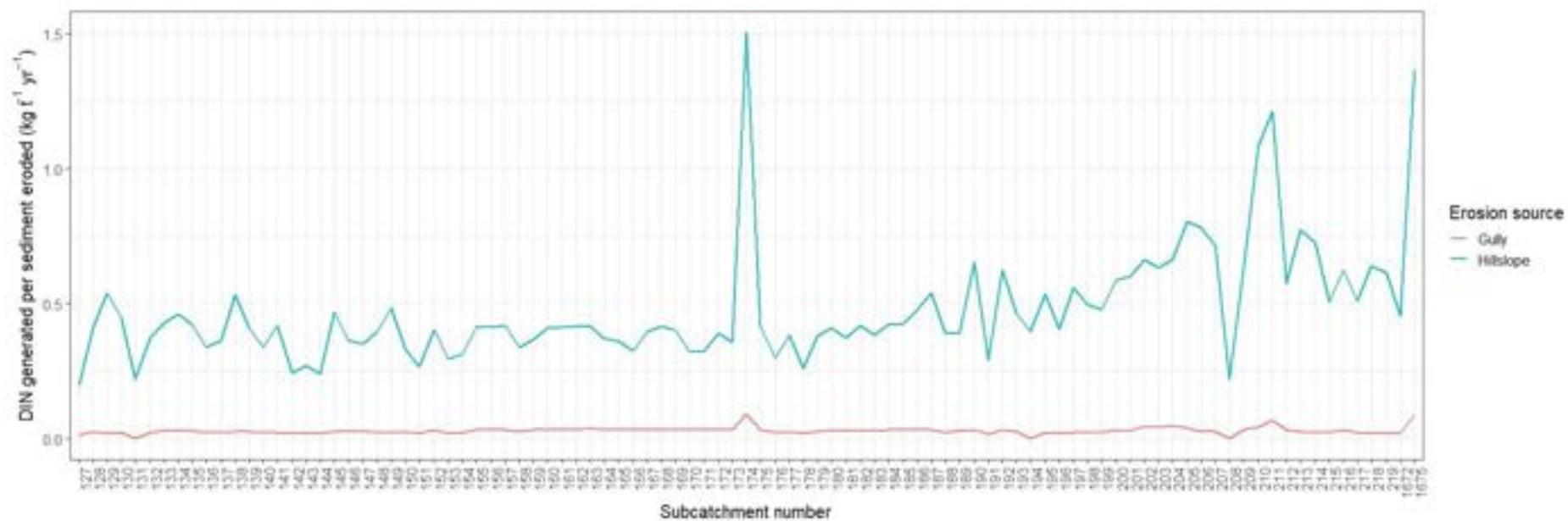


Figure 11.13: Average DIN load (kg year^{-1}) generated per tonne of sediment eroded from gully and hillslope erosion for subcatchments within the Bowen catchment.

11.5 Discussion

The data and analyses presented here demonstrate how improved understanding of spatial variation in DIN load generation, derived from understandings of dominant erosion processes and the volume of sediment contributed by those processes, can be applied in water quality credit trading (Garzon-Garcia et al., 2018b; Garzon-Garcia et al., In prep.). This improved understanding can help identify which changes in management practice and which remediation measures at which locations are likely to be most effective in reducing DIN loads from sediment erosion. More accurate attribution of sediment and DIN reductions to practice change at fine spatial scales will enable revenues from water quality credit trading to better incentivise on-ground actions in locations where specific actions are particularly cost-effective for improving receiving water quality. This should enhance the efficiency and effectiveness of water quality credit trading as a mechanism that can contribute to achieving End-of-Catchment Water Quality Objectives.

In the Johnstone catchment, the relative importance of DIN load generated from sediment erosion in comparison to that generated from fertiliser depends on the total volume of sediment eroded and the proportion of DIN from fertiliser that enters waterways through runoff or through drainage. The predicted magnitude of hillslope erosion from cane land associated with each fertiliser management practice varied substantially depending on whether a uniform slope or a MU-specific slope was used. The 25th and 75th quantiles for these estimates ranged between approximately 5 to 8 t ha⁻¹ yr⁻¹ for uniform slope, and 8 to 73 t ha⁻¹ yr⁻¹ for MU-specific slope respectively (Figure 11.3 and Figure 11.4). For MU-specific slopes, some MUs produced very high predicted erosion, in the range of 250 to 300 t ha⁻¹ yr⁻¹ (Figure 11.4). When these high sediment yielding MUs occurred in areas of the catchment with high potential to generate DIN load from sediment, the resulting predictions of DIN load generated from sediment were high (Figure 11.6). Empirical plot and Caesium-137 data from Innisfail indicate soil loss rates of up to 148 t ha⁻¹ yr⁻¹ for cropped land (including sugarcane) and up to 505 t ha⁻¹ yr⁻¹ for cultivated bare soil (Lu et al. 2003). Modelled estimates of average hillslope erosion rate range from 53 to 75 t ha⁻¹ yr⁻¹ in the Mackay Whitsunday region and 2.8 to 75.3 t ha⁻¹ yr⁻¹ for GBR catchments (Bui et al., 2010). While the soil erosion estimates using MU-specific slopes are comparable to other estimates in the literature, some are towards the upper end of reported values. Given the importance of soil erosion estimates on the modelled DIN load generated from sediment, this highlights the importance of accurately representing physical processes when using this information to inform water quality credit trading, as noted by Garzon-Garcia et al. (2018a).

If the soil erosion estimates using varying slope are taken as reasonable, then the total DIN load from fertiliser in runoff and the total DIN load contributed via sediment loss from cane land are of comparable magnitude in the Johnstone catchment. It is challenging to estimate the DIN contributed to groundwater through drainage as it is influenced by soil properties, antecedent soil moisture, flow pathways and the characteristics of individual rain events (Arnold et al. 2020). Research in other countries suggests it may take decades for nitrate to leach from soil to groundwater and surface water (Wang et al 2013). Research in the Mulgrave Catchment near Cairns suggests that such nutrient contaminants may take between 5.8 to 69 years to travel 650 metres (Rasiah et al 2011). Therefore management practice changes that aim to reduce DIN load from fertiliser in runoff and DIN load contributed via sediment loss may produce more immediate reductions in total DIN loads at End-of-Catchment than those that target reductions in DIN load to drainage. However, DIN losses to drainage still dominate the

total DIN load exported from the catchment, remain a major contributor to total nutrient loading (Rasiah et al., 2013), and are likely to represent an important component of the legacy aspect of nitrogen pollution to the GBR lagoon (Van Meter et al., 2018). Emerging data on the efficacy of actions such as land use conversion to wetlands or installation of bioreactors (Chapter 6) which could remove DIN travelling via the groundwater pathway should help to address some of these issues and inform management responses (Table 11.2).

Table 11.2: Management implications for different sources of DIN in GBR catchments

Water quality parameter	<i>Uncertainty</i>	<i>Efficacy of management action</i>	<i>Immediacy of response</i>
DIN from fertiliser in runoff	Low-Medium	Medium-High	Medium-High
DIN from fertiliser in drainage	High	Low	Low
DIN from sediment	Low-Medium	Medium-High	Medium-High

The ratio of DIN from fertiliser in runoff to DIN from fertiliser in drainage increases as fertiliser management practice improves (Figure 11.10). This is due to improved fertiliser management practices optimising fertiliser application rates to maximise uptake by crops and minimise wastage based on local soil and climate conditions (Sugar Research Australia, 2013). Consequently, the relative importance of DIN generated from sediment is likely to increase over time as compliance with the 2019 Reef Regulations becomes more widespread.

Fraser et al. (2013; Slide 23) show that green cane trash blanketing in combination with legume fallow significantly reduces sediment yield from a representative block of cane land in the Tully catchment. Given the increasing percentage contribution of DIN from sediment to total cane land DIN loss as fertiliser management practice improves, this suggests that careful consideration should be given to the potential impact of reducing trash blanket coverage on the ability to achieve DIN Water Quality Objectives, particularly once the 2019 Reef Regulations have been implemented fully.

In the Bowen catchment, based on subcatchment-resolution Source modelling of average sediment yields and PN enrichment ratios for surface and sub-surface soils, rehabilitating established gullies will likely lead to modest reductions in DIN generated from sediment (Figure 11.13). However, if management practice improvement can prevent new gullies from forming, and therefore reduce the amount of *surface* soil eroded, the reduction in DIN load generated per tonne of hillslope sediment runoff would likely be higher. Based on existing Source modelling, Garzon-Garcia et al. (2018a) reported that if gullies contributed around 93% of the total sediment load in a subcatchment then gully erosion would contribute a comparable total DIN load to hillslope erosion. Empirical evidence suggests that subsurface soil (likely sourced from alluvial gullies) is the dominant source of eroded sediment, and also of bioavailable particulate nitrogen, in the Bowen catchment during periods of high rainfall (Burton et al., In prep.). This again emphasises the importance of accurately representing contributions from different erosion processes when using results to inform credit attribution within water quality credit trading. Research focused on improving estimates of sediment yield from gullies, PN in parent soils, and PN enrichment ratios at gully-specific resolution are currently underway in

the catchment; findings from this research will improve understanding of these processes (Garzon-Garcia et al., In prep.; Andrew Brooks, pers. comm.).

Hillslope erosion is a major contributor to total sediment load in other GBR catchments. Estimates based on empirical data for cultivated areas range from a 43-50% contribution to total sediment load in the Fitzroy to a 79% contribution in the Wet Tropics (Bartley et al., 2017). Estimates based on modelling include 64%, 68% and 70% contributions for Cape York, the Mackay-Whitsundays and the Wet Tropics respectively (Bartley et al., 2017). The data presented by Garzon-Garcia et al. (2018a) indicate that sediments eroded from hillslopes have higher DIN-generating potential in the Johnstone and Bowen catchments. If this is also the case for other GBR catchments experiencing high rates of hillslope erosion, then the results presented here may also be relevant to those catchments. Given the challenge identified in Chapter 5 of meeting End-of-Catchment Water Quality Targets for DIN via improvement in fertiliser practice alone in some Wet Tropics catchments, further quantification appears warranted of DIN loads from sediment and the cost-effectiveness of practice improvement in grazing for reducing hillslope erosion.

The Scientific Consensus statement on factors influencing the health of the GBR identified the Johnstone catchment as a priority for action on both sediment and DIN, and the Bowen as a priority for sediment (Waterhouse et al., 2017). Water quality credit trading is a possible approach for incentivising improvement in land management practices and investment in gully remediation whilst also minimising the economic impacts of stricter water quality targets on local primary producers. The effectiveness of credit trading markets for encouraging cost-effective practice change and gully remediation relies on accurate representation of the underlying physical processes generating pollutants of concern. The results presented here are an important proof of concept to illustrate that the latest research on predicting the locations and magnitudes of the physical processes that underlie DIN generation from fertiliser and sediment can be integrated into water quality credit trading. This should increase the effectiveness and efficiency of water quality credit trading as a mechanism for incentivising improvements in important water quality parameters in the GBR catchments.

11.6 Conclusion

Water quality credit trading has the potential to encourage a shift in agricultural intensity to areas of a catchment which minimise impacts on water quality, whilst also minimising the overall cost impact on industry. These characteristics make them appealing to address agricultural management practices in Great Barrier Reef catchments which are affecting water quality. However, the cost-effectiveness of credit trading markets as a mechanism for reducing pollutant loads at End-of-Catchment relies on accurate spatial representation of the physical processes and management practices that affect water quality. In water quality credit trading to date sediment and nitrogen have usually been considered separately. Recent research has demonstrated how the contribution of particulate nitrogen in sediment to End-of-Catchment nitrogen loads can be quantified (Brooks et al. 2016; Garzon-Garcia et al. 2017, 2018a). The research reported in this Chapter has demonstrated how these new understandings of nitrogen loads from sediment erosion can be used to better inform DIN attributions to practice change and gully remediation in water quality credit trading in Great Barrier Reef catchments. These insights will allow water quality credit trading to account more appropriately for the complexity of interlinked nitrogen and sediment emissions to the GBR lagoon, thus increasing the effectiveness and efficiency of this water quality credit trading as an approach for incentivising delivery of End-of-Catchment Water Quality Objectives.

12.0 MARKETS FOR BUNDLED AND STACKED CREDITS

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12. Summary

This Chapter presents an overview of ‘bundling’ and ‘stacking’ of credits in water quality credit trading markets and explains how these approaches can be used to configure different forms of trading markets for different forms of traded commodity. Potential issues of financial and environmental additionality associated with credit stacking are highlighted. The comparative advantages of different credit configurations are also discussed.

Under appropriate circumstances, stacking or bundling of credits will act to increase the revenues that landholders can obtain from credit supply. Where sufficient demand for relevant types of credits (e.g. DIN, sediment, carbon etc.) exists, the higher revenues obtained from stacking or bundling may be required to elicit supply of adequate quantities of credits to maximise the cost-effectiveness of credit trading as a mechanism for facilitating economic expansion in the Reef catchment within End-of-Catchment water quality targets.

The pollutant content of credits will need to be visible to credit buyers and the environmental regulator where those credits are to be used as offsets against licenced emission exceedances under Queensland’s Point Source Water Quality Offsets Policy. Pollutant content is visible in individual, stacked and bundled differentiated credit markets, but not in bundled composite credit markets.

Transaction cost acts as a disincentive to market participation. The transaction costs that arise for credit suppliers and credit buyers under different credit configurations will depend on their objectives from market participation; consequently, no one configuration will best facilitate active market engagement in all situations.

12.1 Introduction

Conventional water quality credit trading markets as described in Chapter 3 involve trading in individual pollutants in markets that operate independently of each other, and in which suppliers of credits can only sell those credits in a *single market*. In conventional credit markets, landholders receive payments in exchange for supplying a specific type of pollutant reduction credits (e.g. DIN reductions) generated from abatement activities on their land. Buyers of these pollutant reduction credits are typically regulated point-source emitters (i.e. existing and potential new environmental authority (EA) holders) who need to offset exceedances in their emissions of a particular pollutant in excess of the discharge limits specified in the EA. This type of *point source to diffuse source trading* involving a single pollutant that is relatively homogeneous in its characteristics is called *intra-pollutant trading*

(Reeling et al. 2018). Equivalency of the traded commodity (i.e. equivalency between DIN credits generated from non-point sources and excess total nitrogen (TN) load discharged at a point source) is established using the *delivery ratio* and *offset environmental equivalency ratio* (see Chapters 2, 3 and 8). These ratios are used to safeguard environmental outcomes at End-of-Catchment but can also function as a policy lever to facilitate trading activities.

Many water pollution abatement actions deliver reductions in multiple pollutants *jointly*. For example, improved land management practices or gully remediation works act jointly to reduce fine sediment and DIN runoff (Chapter 11). Therefore, an abatement action involving a change in land use such as converting a block of low-lying cane land to a natural wetland would *jointly* deliver a *primary benefit* of reduced DIN emission and *co-benefits* such as removal of phosphorus (Vymazal 2007), sediments (Cooper 2009; Chen 2011; Hefting et al. 2013), and increased carbon sequestration (Hefting et al. 2013; Maucieri et al. 2014). These co-benefits are *positive externalities* associated with the land use change. The majority of government-funded programs to improve GBR water quality have focused on the attainment of primary benefits. For example, best management practice programs for sugarcane and grazing are focused on delivery of DIN and sediment reductions, respectively, and publicly funded gully remediation projects are designed to reduce fine sediment export to the Reef. Because of the focus on delivering primary benefits, positive externalities are typically not accounted for when land use change investment decisions are made, hence the actual level of investments in land use change tends to fall short of the socially optimal level (Perman et al. 2011).⁸²

One way of accounting for delivery of both primary benefits and co-benefits from a land use change is by crediting reductions in the different pollutants separately. These primary and secondary credits can then be ‘stacked’ or ‘bundled’ to produce different tradable commodities. Appropriately packaged mixes of credits via bundling or stacking can make the ‘products’ more appealing to different kinds of buyers, thereby increasing their value. Businesses are increasingly interested to invest in environmental improvement programs to display their green credentials to consumers and also satisfy corporate social responsibility requirements (Torabi and Bekessy 2015). This demand from ‘*non-emitters*’ adds to existing regulatory-driven credit demand from licenced entities operating within the GBR catchments (Chapters 7 and 9).

High demand for packaged (either ‘stacked’ or ‘bundled’) credits would be expected to translate into high market prices, incentivising landholders to undertake higher intensity abatement actions on existing sites or expand abatement into new areas. Higher intensity engineered abatement actions are generally more effective at removing pollutants, and usually take a shorter time for the treatment system to deliver full water quality benefits, compared to lower intensity approaches such as natural regeneration. For example, applying high intensity treatments (e.g. engineered grade control structures and livestock exclusion fencing) to remediate a gully is expected to be more effective at controlling soil erosion compared to a lower level of treatment intensity (e.g. livestock exclusion fencing with revegetation) (Wilkinson et al. 2015b). Similarly, a constructed nitrogen treatment wetland would be expected to deliver higher rates of denitrification than restored coastal wetlands (Waltham et al. 2017). Given that many publicly funded water quality improvement programs are under-subscribed

⁸² An exception to this is the evaluation process currently being applied to offers to supply carbon credits to the Queensland Government’s Land Restoration Fund where a defined set of positive externalities (termed ‘co-benefits’) are considered in the selection process.

(Commonwealth of Australia and Queensland Government 2018), and that substantial improvements still need to be made to achieve End-of-Catchment water quality targets for the GBR (Great Barrier Reef Marine Park Authority 2019), a market-based scheme involving trading in multiple dis-similar pollutants, *inter-pollutant trading*, presents a potentially attractive policy solution to achieving multi-pollutant reduction targets cost-effectively (Reeling et al. 2018).

This Chapter presents an overview of credit bundling and credit stacking and explains how they can be used to configure different traded commodities in credit trading markets. Potential issues of financial and environmental additionality associated with credit stacking are highlighted. The comparative advantages of different credit configurations are also discussed. The remainder of this Chapter proceeds as follows. Section 1 describes primary benefits and co-benefits delivered by pollution abatement actions. Sections 2 and 3 focus on credit stacking and credit bundling, respectively. Section 4 describes financial additionality and environmental additionality associated with water quality credit trading. Section 5 discusses comparative advantages of the different credit configurations from sellers' and buyers' perspectives.

12.2 Primary benefits and co-benefits from pollutant abatement actions

Options for changes in land use to improve water quality in the GBR include land retirement, land conversion from marginal cane land to constructed treatment systems (e.g. farm-scale constructed treatment wetlands and field-scale bioreactors), restoration of natural wetlands, and restoration of degraded coastal wetlands. In addition to changes in land use, landscape remediation to control soil erosion such as gully rehabilitation, streambank repair, and hillslope vegetation management on grazing properties can reduce fine sediment exports to the GBR. Each of these abatement options, via land use change or land remediation, delivers a different mix of regulating ecosystem services for different abatement cost outlays. A land use change or landscape remediation can jointly deliver reductions in a primary pollutant, referred as *primary benefits*, and multiple secondary pollutants or *co-benefits*. Models or tools such as *Paddock-to-Reef* can be used to assess the type of primary benefits and co-benefits an abatement measure delivers and estimate the number of corresponding primary and secondary credits generated, given the baseline land use. For example, retirement of sugarcane land as an abatement measure mainly acts to reduce DIN losses to surface and groundwater, hence reduction in DIN load is considered the primary credit. A secondary credit that could be attributed to sugarcane land retirement is sediment reduction. An abatement measure involving gully remediation aims to stop or significantly reduce fine sediment loss, thus sediment load reduction is the primary credit and the accompanying DIN load reduction (calculated from nitrogen bound in sediment as particulate nitrogen (PN)) is the secondary credit (see Chapter 11 on quantifying DIN load reductions from reduced sediment erosion). Primary credits (e.g. DIN credits) and secondary credits (e.g. sediment credits, carbon credits, biodiversity credits) that are *jointly generated* from the *same abatement action* on a spatially defined property can be combined via *stacking* or *bundling* to produce packaged credits tailored to suit the preferences of credit purchasers.

12.3 Credit stacking

In general, there are three forms of stacking: horizontal, vertical and temporal (Cooley and Olander 2011), and recently Duguma et al. (2018) identified a hybrid vertical-temporal stacking of ecosystem services from forested land management. Horizontal stacking occurs when a landholder, who undertakes different abatement measures on spatially distinct sites, receives a *single payment* for abatement credits generated from multiple sites (Figure 12.1). In Figure 12.1, the landholder generates sediment credits from vegetating a riparian area and restoring coastal wetland on their property, but only generates DIN and biodiversity credits from the wetland. In return for supplying these combined DIN, sediment and biodiversity credits, the landholder receives a single payment. If credits accrue yearly from these abatement measures, the single payments would also be on an annual basis.

Vertical stacking occurs when a landholder receives multiple payments for undertaking a pollution abatement measure on a spatially-defined site (Cooley and Olander 2011). Vertical stacking necessitates generating more than one credit type which are then sold in their respective markets. Temporal credit stacking is similar to vertical stacking but the payments for the different credit types are made at different points in time (Cooley and Olander 2011; Lankoski et al. 2015; Duguma et al. 2018; Motallebi et al. 2018). A hybrid vertical-temporal stacking occurs when a landholder receives multiple streams of payments over a number of years for undertaking an abatement measure on a site (Duguma et al. 2018).

A hypothetical example of vertical stacking is shown in Figure 12.2, where a landholder converted 5ha of their land into a constructed treatment wetland which generates 300 DIN credits, 100 sediment credits and 40 carbon credits. Assuming that the market prices for DIN, sediment and carbon credits are \$20, \$10 and \$5 per unit, respectively, the total revenue earned by the landholder via vertical stacking in this example is \$7,200. If this \$7,200 is a one-off payment, then this example illustrates vertical stacking, however, if the payment streams are periodical over an agreed number of years, then this becomes a hybrid vertical-temporal stacking example.

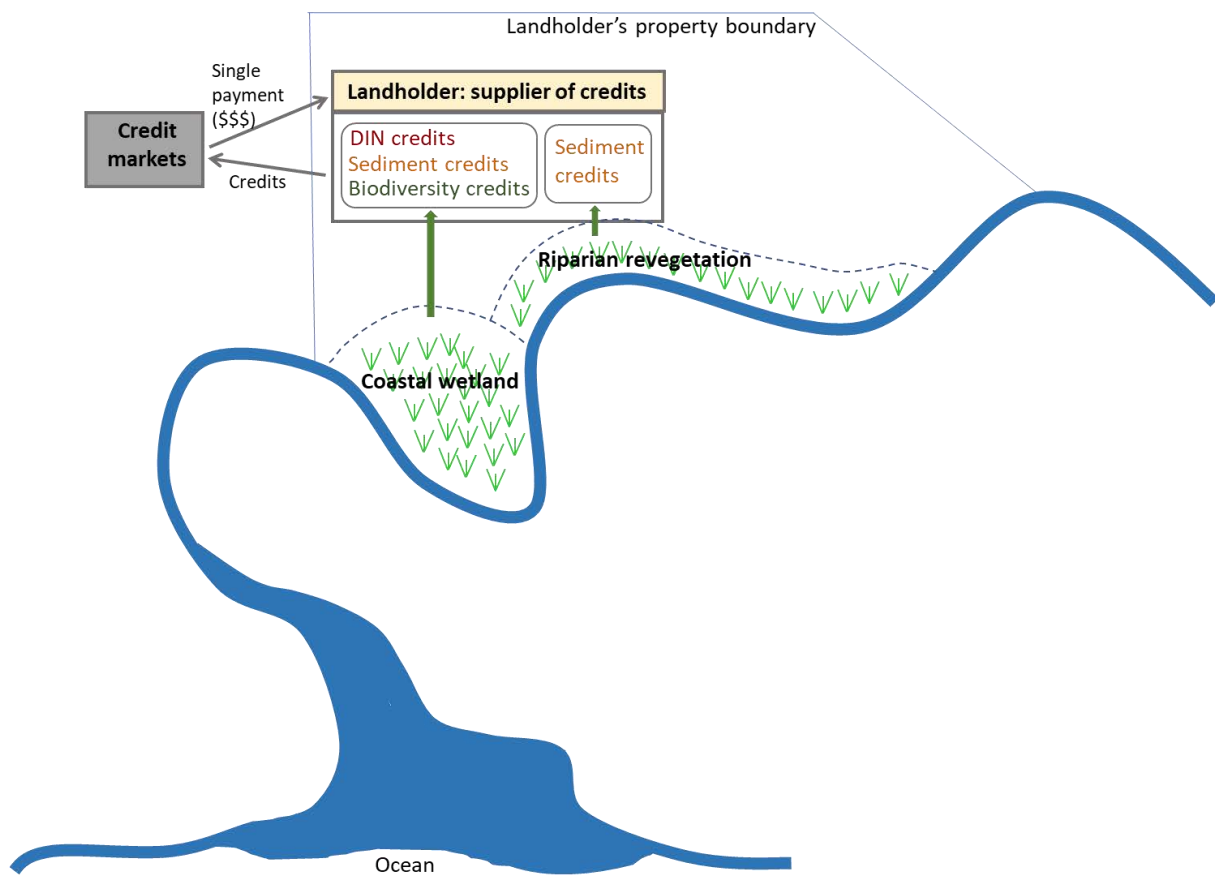


Figure 12.1: Horizontal credit stacking: landholder receives a single payment for supplying credits from two separate sites.

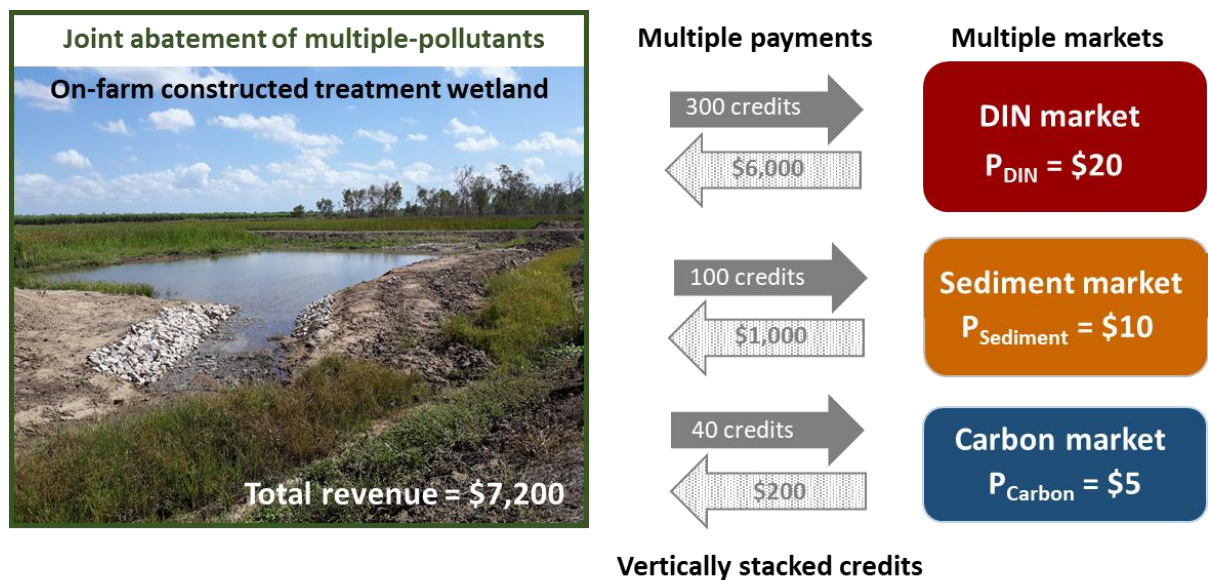


Figure 12.2: An example of vertical stacking of credits generated from an on-farm constructed treatment wetland

With reference to Queensland Point Source Water Quality Offsets Policy 2019 (State of Queensland 2019f), existing and potential new Environmental Authority (EA) holders who wish to offset their development impacts or exceedances of the discharge limit(s) specified in their EA would be able to purchase relevant credits from a landholder who undertakes a pollution abatement measure such as that shown in Figure 12.2. In this situation, the landholder vertically stacks or vertical-temporally stacks their credits to enable them to sell the DIN, sediment and carbon credits *separately* to different buyers (e.g. DIN credits to sewage treatment plants (State of Queensland 2019f), sediment credits to land developers for off-site storm water management (via the in-lieu payment fund administered by local governments) (State of Queensland 2018a) and carbon credits to power generators). Aggregate revenues received from these sales would be expected to meet the landholder's minimum willingness to accept compensation for undertaking the abatement measure. Allowing flexibility to landholders in stacking credits enables abatement actions that incur higher upfront costs, but are nevertheless still cost-effective, to be undertaken to deliver water quality improvements at End-of-Catchment. The gains from trade to credit buyers are in the form of cost savings from avoiding the need to upgrade their grey infrastructure. From society's perspective, the offset mechanism provides opportunities for establishing green infrastructure within the catchment enhancing liveability, welfare and the economy of the local communities. Other positive externalities from pollutant abatement actions upstream that accrue to other sectors include water treatment plants (via improved raw water quality), tourism (via enhanced biodiversity and liveability), ports (via reduced dredging requirement) and reduction in flood risk to downstream properties and infrastructure.

12.4 Credit bundling

Credits can be aggregated to produce bundled credits. Bundled credits can be packaged to produce *differentiated credits* or *composite credits*. The contents of a *differentiated credit* are visible to the buyers and are available in several combinations in terms of the type and quantity of each element in the bundle. In contrast, the contents of a *composite credit* are undifferentiated making the product uniform or standardised in its characteristics. Both types of bundled credits are traded in a *single market* (Deal et al. 2012) because – importantly – the separate elements within the bundle *cannot* be sold separately.

12.4.1 Differentiated credits

Differentiated credits can command different prices within the market because the content and composition of the different credit types and quantities are visible to the buyers (Figure 12.3). A seller can offer differentiated credits comprising different combinations of one, two, three or more pollutant reductions. The seller's offer price would reflect the specific contents of the differentiated credit, and buyers' bid prices would reflect the prevailing demands for the different elements included in the bundle. The level and breadth of prevailing and projected demand for differentiated credits can influence landholders' decisions regarding which abatement measures should be undertaken and where they should be sited. Market forces from a well-functioning credit market characterised by active participation of buyers from a wide range of industries, well supported by integrated environmental policies, should be able to drive significant improvements in water quality by incentivising abatement measures that produce credit bundles that are collectively of high value of society.

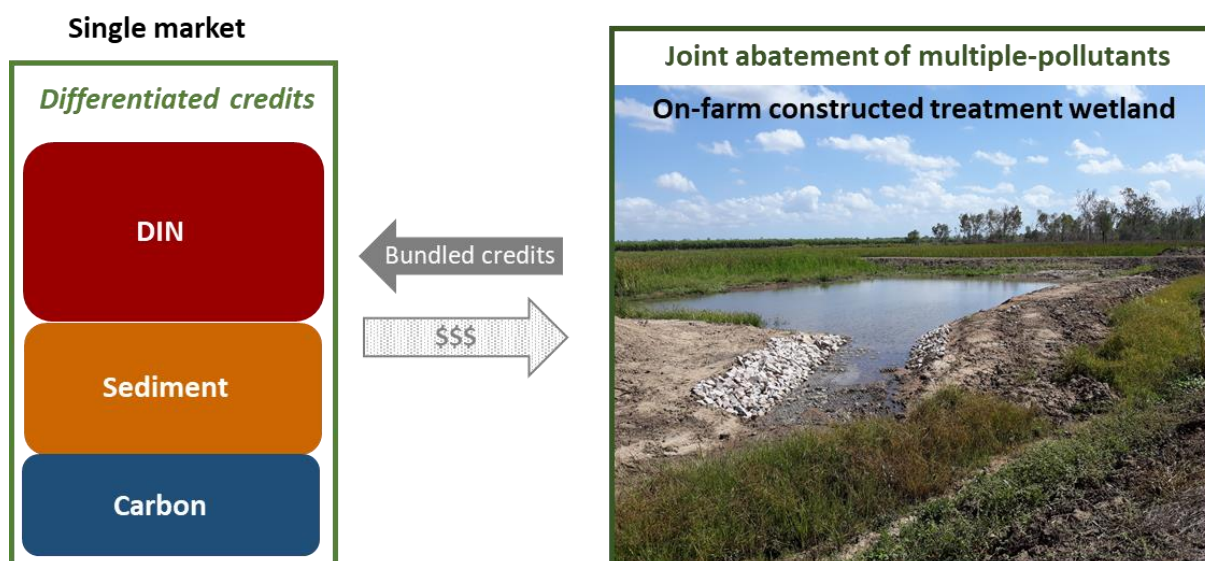


Figure 12.3: An example of bundled differentiated credits generated from an on-farm constructed treatment wetland

Buyers from a wide range of industries would likely exhibit different preferences for bundled credits, and these preferences would be reflected in their maximum willingness to pay for the different mix of elements in the credits. For example, a buyer may be willing to pay a higher price for bundled credits which delivers x kg DIN reductions, y tonnes of carbon sequestration and ‘high’ biodiversity co-benefits compared to bundled credits with the same DIN and carbon components but with ‘low’ biodiversity co-benefits. This notion of bundled-differentiated credits is similar to the description of bundled credits provided by Torabi and Bekessy (2015), who provide an example of buyers who are willing to pay a higher price for ‘premium carbon’ when carbon is sequestered along with co-benefits of biodiversity conservation.

Figure 12.4 illustrates the relationship between price per unit of differentiated credits and the quantity of credits demanded. This ‘credit differentiation curve’ in Figure 12.4 segments the tradable commodity (i.e. differentiated credits) into ‘premium’, ‘standard’ and ‘basic’ according to willingness to pay of the buyer groups⁸³. In this product differentiating curve example, the differentiating product feature is the level of biodiversity attached to the credit (high, medium and low) whilst maintaining other features as constant (i.e. along this curve, the ratio of the primary and secondary benefit quantities remains unchanged e.g. each differentiated credit contains 1 unit of DIN, 0.2 units of sediment and 0.4 units of carbon). Premium, standard and basic credits contain high, medium and low level of biodiversity, respectively. There could potentially be three types of buyers: biodiversity conservation organisations, urban developers and sewage treatment plant. For instance, ‘premium’ credits may appeal to biodiversity conservation organisation; urban developers may have a preference for ‘standard’ credits to offset their development impacts; and sewage treatment plants may find ‘basic’ credits to be sufficient to offset their licence exceedance emissions.

⁸³ This credit differentiation curve is an adaptation from an exposition by Saylor Academy 2012 under a Creative Commons Attribution-NonCommercial-ShareAlike 3.0 License, freely available at https://saylordotorg.github.io/text_developing-new-products-and-services/s08-examples-of-product-differenti.html (date accessed 11 March 2020).

The Queensland Government's Land Restoration Fund uses a differentiated credit approach to evaluate tenders that landholders submit for carbon farming projects. The level of co-benefits delivered in association with the carbon emissions reduction are considered when tenders are evaluated (<https://www.qld.gov.au/environment/climate/climate-change/land-restoration-fund>). Environmental, social and economic, and First Nations are the three categories of co-benefits included. Co-benefits considered for the environmental category include improvements in water quality, soils, wetlands and water systems, and additional habitat for threatened species (State of Queensland 2020). Tenders that deliver higher levels of co-benefits are regarded as 'premium' tenders for which the Land Restoration Fund would be prepared to pay a higher price, as shown in Figure 12.4.

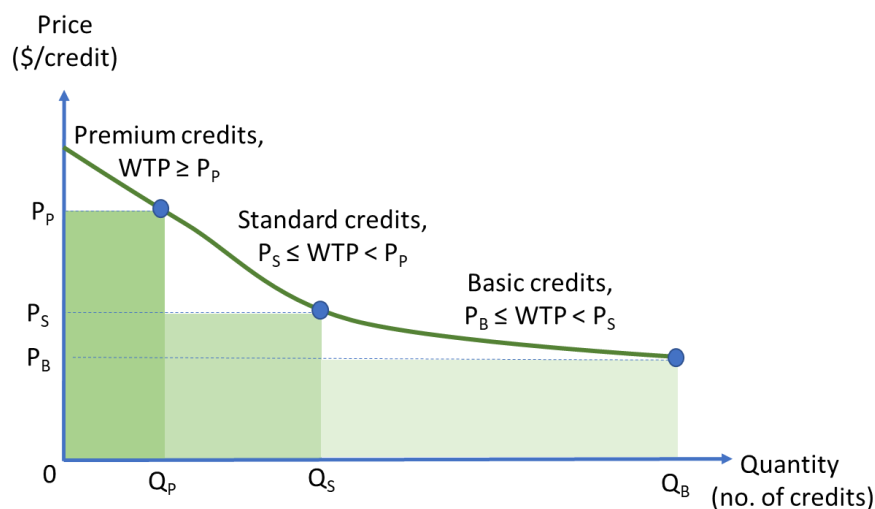


Figure 12.4: Credit differentiation curve shows segmentation of the willingness to pay (WTP) by the buyer group. Bundled credits are differentiated by their biodiversity co-benefit into premium (high biodiversity), standard (medium biodiversity) and basic (low biodiversity). The credit differentiation curve is an adaptation from an exposition of commercial vehicles given in Saylor Academy (2012).

12.4.2 Composite credits

The separate elements in a bundled *composite credit* are combined and standardised using fixed, pre-determined ratios or *exchange rates*. The separate elements in a composite credit are therefore no longer visible (Figure 12.5). For example, a composite credit can be constructed from DIN and sediment credits using these exchange rates: 1 tonne of sediment $\equiv$ 30 kg DIN. Using this fixed ratio, 1 credit = 1 tonne of sediment = 30 kg DIN = (0.5 tonne of sediment + 15 kg DIN) = (0.75 tonne of sediment + 7.5kg DIN) etc. There is only a single market price for homogenous credits because the traded commodity is an undifferentiated product as a consequence of the standardisation process.

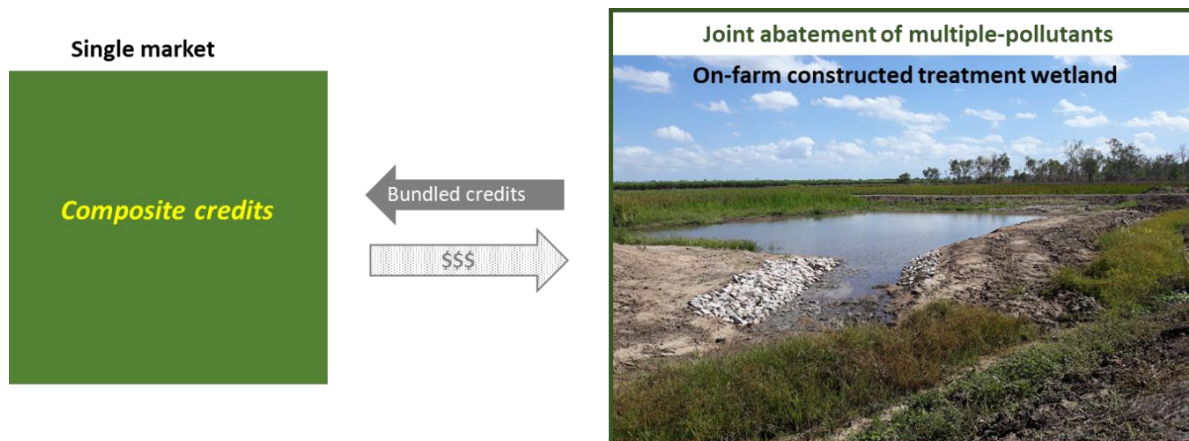
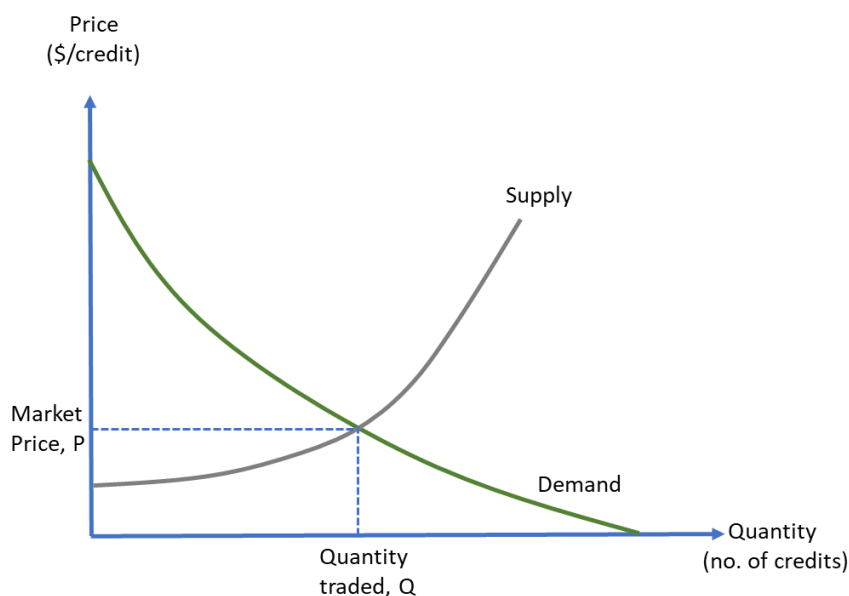


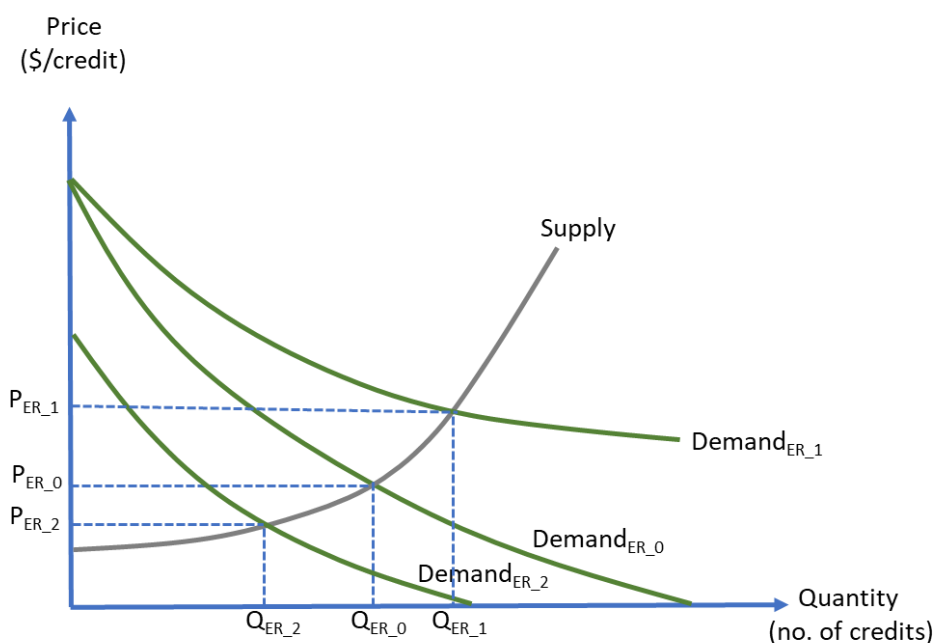
Figure 12.5: An example of composite credits generated from an on-farm constructed treatment wetland

The exchange rates could, for example, be determined based on the relative End-of-Catchment target reductions for the different pollutants, or via other mechanisms as informed by management prioritisation. Exchange rates can be set initially to encourage market participation from certain sectors and then revised accordingly as the market matures and then further adjusted to reflect the extent of the remaining reductions required to attain the End-of-Catchment targets for the separate pollutants. Figure 12.6a shows illustrative demand and supply curves, and the corresponding equilibrium price, for composite credits. For a given demand curve, the exchange rates remain fixed. When exchange rates are altered, as shown in Figure 12.6b, the demand curve shifts either upward or downward depending on the new exchange rates; given the previous supply curve, the new market price would increase or decrease accordingly.

The Reef Credit Scheme (www.reefcredit.org) is an example of trading in credits containing quantifiable amounts of nutrient, pesticide or sediment reduction generated through changes in land management practice. The exchange rate (i.e. 'conversion factor' as referred to by GreenCollar) used to bundle the sediment and DIN credits into standardised Reef Credits is 1 tonne of DIN = 538 tonnes of fine sediment, which is the ratio of DIN to sediment reduction targets to be achieved by 2025 (GreenCollar 2020). Buyers of Reef Credits seek to secure improvements in Reef resilience. The implication of the exchange rate used by GreenCollar is that a one tonne reduction in DIN load to the Reef lagoon delivers the same benefit to Reef resilience as a 538 tonne reduction in fine sediment load to the Reef lagoon. The exchange rate also implies that pro-rata combinations of sediments and DIN load reductions deliver equivalent benefits to Reef resilience. In order for this type of combined credits to be used to offset EA holders' environmental liability for exceedances of specific pollutants, the quantities of the separate elements would need to be visible to prospective buyers and the environmental regulator.



(a) Demand, supply and equilibrium price for bundled composite credits



(b) Changes in market prices under different exchange rates for bundled composite credits

Figure 12.6: The demand, supply and equilibrium market price for bundled composite credits for a given pollutant exchange rate (panel a). Altering the pollutant exchange rates (ER_1 from ER_0; ER_2 from ER_0) would shift the demand curve resulting in a new equilibrium price P_{ER_1} or P_{ER_2} .

12.5 Financial and environmental additionality

When landholders or credit suppliers are able to receive multiple payments for an abatement practice on a single site, both vertical and hybrid vertical-temporal stacking could create potential problems of “double payments and additionality” (Lankoski et al. 2015, p.9). For some abatement measures, multiple payment streams are required to cover landholders’ costs, which are framed in terms of landholders’ minimum willingness to accept compensation (WTA). As explained in Chapter 5, landholders’ minimum WTA may include opportunity costs,

transition costs, transaction costs and costs associated with increased exposure to risk and uncertainty (Rolfe and Windle 2016).

The sum of opportunity costs, transition costs and transaction costs could potentially be quite substantial. If these costs can be covered via multiple payment streams, more landholders would be inclined to participate as credit suppliers in credit trading programs. Stacking allows implementation of some larger scale but still potentially cost-effective pollution reduction measures that can abate several pollutants simultaneously. Examples include restoration of coastal wetlands instead of riparian revegetation (Cooley and Olander 2011), or remediation of gullies instead of streambank repair. If all low-cost abatement practices have already been implemented, credit stacking may be needed to appropriately incentivise additional higher cost abatement measures to deliver additional water quality improvement.

Figure 12.7 illustrates conceptually the situation with regard to the DIN reduction target for the GBR. The MAC_{stacking} cost level in Figure 12.7 represents the marginal cost of supply at which credit stacking may need to be introduced. The segment of the marginal abatement cost curve to the right of point A (coloured in red) is higher than that for abatement undertaken previously (segment shown in grey), indicating that each subsequent abatement is becoming increasingly expensive. If landholders can sell both primary (i.e. DIN) and secondary (e.g. sediment) credits, the sum of the multiple revenue streams received is more likely to cover landholders' costs (i.e. their WTA as described previously). Credit stacking is expected to bring in additional higher-cost credit supply to the market thus extending the aggregate supply curve beyond point A.

Despite these advantages, stacking also runs the risks of credit suppliers being paid more than once for an abatement action that would have been undertaken with just a single payment. In these circumstances, the extra payment provides a surplus to the credit supplier with no corresponding *addition* to the level of regulating services supplied (Bennett 2010; Cooley and Olander 2011; Robertson et al. 2014; Lankoski et al. 2015; Motallebi et al. 2018). If multiple payments are required to meet the landholder's WTA, then the *financial additionality test* (Lankoski et al. 2015) has been satisfied.

In addition to financial additionality, it is important to ensure *environmental additionality* in water quality outcomes in receiving waters. All credits that are offered for sale must correspond to real physical reductions in pollutant load *relative to the environmental baseline against which offsets would be applied*. For example, if an existing point source EA holder has a licence to discharge 5 tonnes of DIN, as part of a 50 tonnes total DIN load limit at the receiving water, then an offset purchased to cover 2 tonnes of DIN licence exceedance must be sourced from 2 tonnes of *additional* DIN reduction relative to the 50 tonnes load limit. Offsets must be *additional* to land management practices and land uses that were already in place when the 50-tonne load limit for the receiving water was set. Provided environmental additionality is in place, then all credits offered for sale will provide valid offsets of emission exceedances within the 50 tonnes load limit. This necessitates careful assessment of the environmental baseline. A baseline could be established using observed land use or land practice as a proxy for determining emissions at source (Lankoski et al. 2015), or could be determined using historical farm records. Many water quality programs, such as the Australian Government's Reef Trust Reverse Tenders for the Wet Tropics Pilot Round, require landholders to specify their baseline and proposed nitrogen application rates in their submissions for payment support

(Commonwealth of Australia 2014). Further information on how additionality can be assessed can be found in Bennett (2010), who discusses the issue in detail in the context of payment for ecosystem schemes and ecosystem markets.

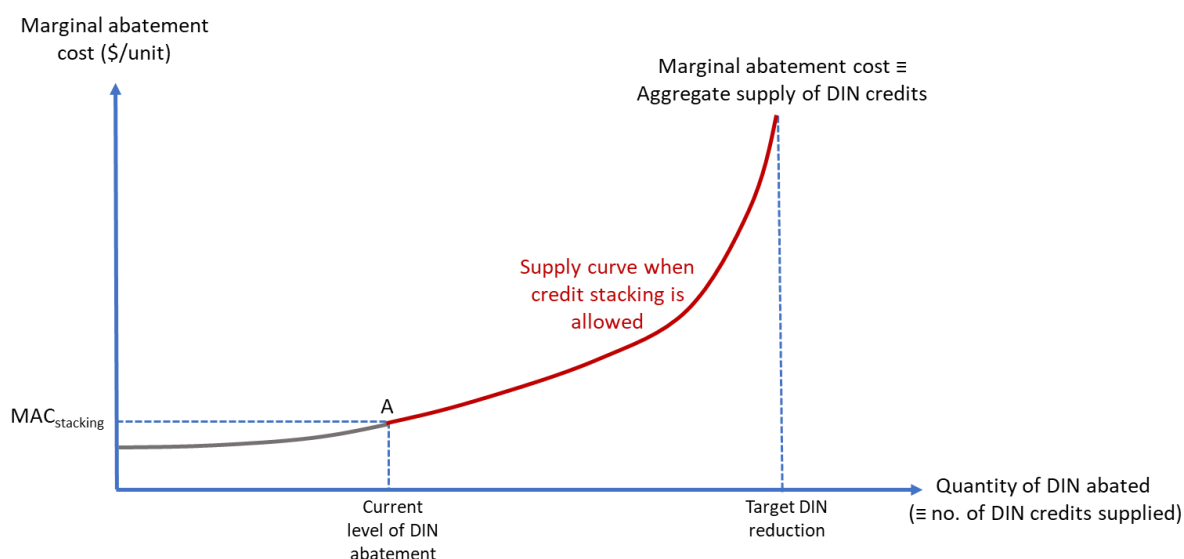


Figure 12.7: Credit stacking may need to be introduced when the marginal abatement cost (without stacking) reaches MAC_{stacking} level. The red segment of the supply curve becomes accessible through credit stacking.

12.6 Comparative advantages of different credit configurations

12.6.1 Comparative advantages in eliciting supply and demand

Different credit configurations facilitate higher levels of market engagement for different types of credit suppliers and offset buyers. Transaction costs are incurred by both suppliers and buyers in engaging with a market. If a credit supplier implements a change in land use or land management that generates multiple pollutant reductions jointly, they can either sell these credits separately on relevant markets or sell them bundled as differentiated or composite credits. The latter two options only require the supplier to engage with one market (see Figure 12.3 and Figure 12.5), reducing transaction costs compared with when selling in separate markets via stacking (see Figure 12.2). Conversely, suppliers of single pollutant reduction credits may incur lower transaction in engaging only with the market for that credit. On the demand side, buyers interested in offsetting only a single type of pollutant would probably incur lower transaction costs by only engaging with the market for that pollutant. Conversely, buyers interested in bundled credits, may prefer to engage only with bundled credit market rather than having to engage with each market separately. Thus, given the reduction in transaction costs, demand from buyers of bundled credits would be expected to be higher in the bundled credit market whereas demands from buyers of single credits would be expected to be higher in the separate markets. Higher demand would elicit a higher level of supply. This suggest that having both stacked and bundled credit products in the mix may be beneficial for eliciting the highest possible levels of market activity which translates into improved cost-effectiveness in delivery of desired water quality outcomes. As mentioned in Section 12.4.2, in order for bundled credits to be considered for potential use as offsets under the Queensland Point Source Water Quality Offsets Policy, the separate 'credit contents' will have to be visible to prospective buyers and

to the environmental regulator; consequently, there may be more limited demand for these types of credits in some situations. However, in GBR catchments with only relatively low point source loads (see Chapter 7), the attractiveness of bundled Reef Credits to non-emitting buyers may well dominate. In these situations, credit suppliers may find it advantageous to sell their credits as bundled composite products. This again emphasises the advantages of having different categories of credits or products available to provide flexible mechanisms for satisfying suppliers' and buyers' expectations and objectives.

12.6.2 Comparative advantage in the presence of intermediaries

As described in Chapter 3, an intermediary operating as a credit bank between credit suppliers and offset buyers can be regarded as a *monopoly supplier of credits*. In this configuration of market, there is effectively a monopoly supplier of DIN credits to credit buyers, with the credit bank sourcing DIN credits from landholders who undertake abatement measures involving, for example, land use change (e.g. on-farm constructed treatment wetlands, wetland restoration and streambank remediation). Indicative aggregate supply, aggregate demand and monopolist's marginal revenue in such a situation are shown in Figure 12.8. A profit-maximising monopolist would only choose to supply Q_M DIN credits and charge a P_M price for those credits (i.e. point M in Figure 12.8). Contrast this to a perfectly competitive market with many suppliers, in which the quantity of credits traded would be Q_C and the market price would be P_C (i.e. point C in Figure 12.8). At point C, there is no welfare loss to the society whereas at point M, the societal welfare loss is highest. The welfare loss increases as the degree of imperfect competition rises towards monopoly i.e. as the market moves from point C to point M. Movement towards point M may be due to the presence of an intermediary, a group of intermediaries, and/or product differentiation via bundling, injecting elements of imperfect competition into the credit trading market.

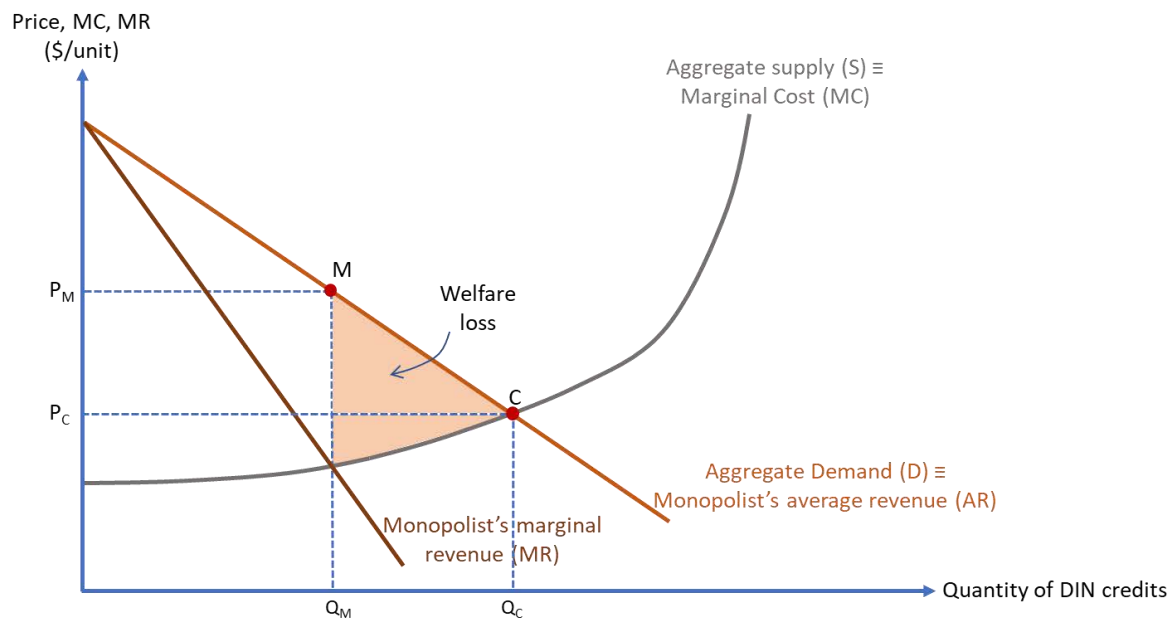


Figure 12.8: Credit market with a monopoly supplier without credit stacking

A useful further investigation, outside the scope of this project, would be to consider how the extent to which markets for the different forms of credit product (stacked, bundled differentiated and bundled composite) are robust against manipulation by monopoly credit supplier.

13.0 EMPLOYMENT OPPORTUNITIES FROM INVESTMENTS TO RESTORE THE HEALTH OF THE GREAT BARRIER REEF

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13. Summary

The Great Barrier Reef is important economic asset to the State of Queensland and Australia. Its economic contribution to the Australian economy is estimated at 6.4 billion annually and it supports 64,000 full time equivalent jobs. Tourism provides the majority of added value. The Great Barrier Reef is threatened by climate change and poor water quality runoff from catchments. The Australian and Queensland governments are investing approximately \$615 million over 5 years to address poor water quality. This investment is a key contributor to regional economic development in North Queensland, supporting a variety of agricultural, scientific, administration, extension, earthmoving, revegetation, engineering and monitoring and reporting services.

This Chapter used the Economic Impact Analysis Tool from Flinders University to estimate that investment under the Reef 2050 Water Quality Improvement Plan creates 1083 (574 direct and 509 indirect) full time equivalent positions annually. Estimation results indicate that 1 full time position is created for each \$114,000 invested in Reef Health annually. While the majority of these positions are in the agricultural and scientific sectors, other economic sectors also benefit substantially from this investment. Although outside the scope of this analysis, this investment also makes an important contribution to ongoing economic activity in North Queensland with several studies indicating that a decline in Reef health would have a detrimental impact on tourism-related activities.

13.1 Introduction

The purpose of this Chapter is to examine the economic and employment impacts of programs associated with managing, protecting and restoring the environmental values of the Great Barrier Reef^{f84}.

⁸⁴ As a consequence of the input-output based analysis approach used in this Chapter, only market-based exchange value estimates of increments to gross regional product (i.e. the increment to regional value-added) and regional employment are included. Increments to non-market values and welfare benefits are excluded.

13.2 Economic contribution of the Great Barrier Reef

The Great Barrier Reef is an area of world heritage that is recognised as one of the natural wonders of the world. Its outstanding environmental value is mirrored in its economic importance to the Queensland and Australian economies. In 2017 Deloitte Access Economics estimated the economic contribution of the Great Barrier Reef at \$6.4 billion in value added to the Australian economy, with 90% of this coming from tourism activity, of which approximately 60% is generated within Queensland (Table 13.1) (Deloitte Access Economics 2017). In terms of employment the Great Barrier Reef supported more than 64,000 full-time jobs in Australia. Approximately 33,000 of these were in Queensland, and 24,000 in the Reef regions themselves (Table 1.2) (Deloitte Access Economics 2017).

Table 13.1: Economic contribution (value added) of the Great Barrier Reef (\$ millions)

		<i>GBR Region</i>	<i>Queensland Total</i>	<i>Australia Total</i>
Value added	Tourism	2,400	3,400	5,700
	Fishing	139	140	162
	Recreation	284	296	346
	Scientific Research	155	161	183
Total value added		2,978	3,997	6,391

Source: Deloitte Access Economics (2017)

Table 13.2: Employment contribution of the Great Barrier Reef (Full Time Equivalents)

		<i>GBR Region</i>	<i>Queensland Total</i>	<i>Australia Total</i>
Employment	Tourism	19,855	28,768	58,980
	Fishing	680	690	814
	Recreation	2,899	2,964	3,281
	Scientific Research	895	914	970
Total FTE		24,329	33,336	64,045

Source: Deloitte Access Economics (2017)

In terms of scientific research and reef management, the Deloitte report identifies the Great Barrier Reef “as the largest living structure on the planet with one of the greatest diversity of species and habitats.... [it] has attracted a large number of researchers” (Deloitte Access Economics, 2017; p.24). Five research and Reef-related organisations that carry out extensive research were included in the valuation exercise: the Great Barrier Reef Foundation, the Australian Institute of Marine Science, the Great Barrier Reef Marine Park Authority, James Cook University ARC Centre of Excellence, and Lizard Island Reef Research Foundation.

In 2015-2016, \$130 million of revenue was generated by these organisations through the conduct of scientific research, reef management and related activities. The five organisations examined spent \$57 million on employment and \$65 million on intermediate inputs. The total value of the scientific research and reef management associated with the Great Barrier Reef

in 2015-2016 was estimated to contribute \$183 million to the Australian economy (Table 13.1) (Deloitte Access Economics 2017).

13.3 Great Barrier Reef health and the Reef 2050 Water Quality Improvement Plan

The 2017 Scientific Consensus Statement found that key Great Barrier Reef ecosystems continued to be in poor condition, with this largely being due to the collective impact of land run-off associated with past and ongoing catchment development, coastal development activities, extreme weather events and climate change impacts such as the 2016 and 2017 coral bleaching events (Waterhouse et al. 2017b).

The Reef 2050 Water Quality Improvement Plan (State of Queensland 2018b) seeks to improve the quality of water flowing from catchments adjacent to the Reef. Investment in the Plan over the five years between 2018 and 2022 by the Queensland and Australian governments is expected to total \$615 million (Commonwealth of Australia 2016).

13.4 The restoration economy

BenDor et al. (2015) when estimating the size and impact of the restoration economy in the United States define restoration as “any combination of activities intended to result in ecological uplift, improve ecosystem health, and result in a functioning ecosystem that provides a suite of ecosystem services” (BenDor et al., 2015; p.2). The same authors describe the restoration economy as being comprised of “the set of activities that contribute to restoration, from project planning, engineering and legal services, to intermediate suppliers of inputs, to on-the-ground earthmoving, forestry and landscaping firms that contribute to the ecological restoration process” (BenDor et al., 2015; p.2).

The Reef 2050 Water Quality Improvement Plan investment actions include investments to support delivery of minimum practice standards in agriculture, catchment restoration, science and knowledge advancement, and performance evaluation (State of Queensland 2018b). A wide range employment sectors, skills and services will need to be engaged to deliver these outcomes, including program planning, management and administration, agricultural and horticultural extension, revegetation and earthmoving contractors, engineering, science, monitoring and reporting.

Investment in Reef protection and restoration is a major form of regional economic development. The economic benefits that investment in measures to protect and restore the Great Barrier Reef can be considered as the sum of the value of the direct and indirect benefits obtained, together with the value that protection and enhancement of the Great Barrier Reef provides from an income (mainly tourism) perspective.

13.5 Estimating direct and indirect benefits

Analysis of direct and indirect benefits is undertaken using the Economic Impact Analysis Tool (EIAT). EIAT is an input-output (I-O) analysis tool developed by the Australian Industrial

Transformation Institute (AITI) at Flinders University. The tool draws on Australian Bureau of Statistics census data and the national I-O table to calculate industry multipliers which in turn provide estimates of economic impacts of regional infrastructure investment projects (Australian Urban Research Infrastructure Network 2019).

The EIAT tool then applies the regional I-O model for economic impact estimation in terms of gross regional product (GRP) and employment. GRP is a measure of the net contribution of an activity to the regional economy and is calculated as value of output less the cost of goods and services used in producing the output. Employment is a measure of the number of employees in the region in terms of the number of full-time equivalent (FTE) jobs.

To match with the parameters of the EIAT model each action identified in the Reef 2050 Water Quality Investment Plan was categorised as either 'Science and Professional Services' or 'On-ground Works'. Activities such as science, innovation, education, extension⁸⁵, monitoring and reporting and administration were categorised as Science and Professional Services. Activities such as best management practices and landscape restoration were categorised as On-ground works. This categorisation indicated that within the entire program of investment of \$614,762,536 across the five years 2018 to 2022, \$258,457,604 is being spent on activities that can be considered as scientific and professional services, with the remaining \$356,304,932 being spent on on-ground works. This translates into the EIAT model as an approximate spend of \$71M annually on agricultural-related activities and \$52M on scientific and professional services each year between 2018 and 2022.

In using the EIAT model the following assumptions were made:

- The majority of funding is spent in the Great Barrier Reef catchments
- The investment is spent evenly over the five years of the program
- The majority of on-ground works are being undertaken on existing agricultural land

Outcomes from the model based on annual investments of \$71M annually in agriculture and \$52M in scientific and professional services in the regional economy of North Queensland are shown in Table 13.3, Table 13.4 and Table 13.5.

Table 13.3: Annual Increase in Gross Regional Product from Reef 2050 Water Quality Improvement Plan investments 2018-2022. Results produced by AURIN Economic Impact Analysis Tool (2019).

<i>Gross regional product</i>	Annual increase (\$ millions)
Direct	54.8
Flow on	65.0
Total	119.8

⁸⁵ Provision of extension services was categorised as a 'Sciences and professional service' for the purposes of the EIAT model because the suite of inputs required to support supply of extension services is similar to those required to support scientific and research services. In contrast, the suite of inputs required for implementing 'On ground works' is quite different.

Table 13.4: Annual Increase in Direct and Indirect Full-Time Equivalent (FTE) employment from Reef 2050 Water Quality Improvement Plan investments 2018-2022.

<i>Employment</i>	Annual FTE positions created
Direct	574
Flow on	509
Total	1,083

Source: Output from AURIN Economic Impact Analysis Tool 2019

Table 13.5: Annual Direct and Indirect Full-Time Equivalent (FTE) positions created through Reef 2050 Water Quality Improvement Plan investments 2018-2022.

<i>Industry sector</i>	Annual direct and indirect FTE positions
Agriculture, Forestry and Fishing	356
Mining	1
Manufacturing	47
Electricity, Gas, Water and Waste Services	11
Construction	20
Wholesale trade	26
Retail trade	72
Accommodation and Food Services	44
Transport, Postal and Warehousing	39
Information Media and Telecommunications	16
Financial and Insurance Services	16
Rental, Hiring and Real Estate Services	13
Professional, Scientific and Technical Services	294
Administrative and Support Services	23
Public Administration and Safety	10
Education and Training	24
Health Care and Social Assistance	28
Arts and Recreation Services	8
Other Services	35
Total	1,083

Source: Output from AURIN Economic Impact Analysis Tool 2019

EIAT modelling results predict that investment in protection of the Great Barrier Reef through the Reef 2050 Water Quality Improvement Plan creates an estimated 574 direct FTEs and 509 indirect FTEs for a total of 1,083 FTEs annually.

Each \$114,000 of annual investment in restoration activities in the Great Barrier Reef as outlined in the Reef 2050 Water Quality Improvement Plan creates 1 equivalent direct or indirect full-time position.

13.6 Protecting the Reef as a tourism asset

The third type of value generated through restoration activities is associated with the maintenance of the Great Barrier Reef as an asset for the tourism industry. At this stage it is not possible to identify the effectiveness of investment through the Reef 2050 Water Quality Improvement Plan in terms of protecting the health of the Reef. Consequently, the economic benefits of this investment are difficult to evaluate within the scope of this Chapter. However, other studies have shown that degradation of the Great Barrier Reef will decrease the attractiveness of the destination for tourism. Tourism Australia (TA) found that 42% of international visitors rank the Great Barrier Reef as the most appealing tourist attraction in Australia (Tourism Australia 2015). Esparon et al. (2015) conducted a major survey of visitors to the Great Barrier Reef and found that:

- Water clarity (clear ocean) was deemed the most important value
- The level of satisfaction with water clarity was low
- Visitors had a stronger negative reaction to the prospect of four different types of environmental degradation (oil spills, reduced water clarity, rubbish, and reduced coral cover) than to a hypothetical 20% increase in local prices.

Esparon et al. (2015) also found that the greatest sensitivity to environmental degradation was in the Cairns/ Port Douglas region. Stoeckl et al. (2014) conclude that changes in the environment in the Great Barrier Reef would have a real impact on people. These authors further conclude that the decisions that would likely follow from these individual impacts could have a considerable cumulative impact on the regional economy. Stoeckl et al. found through a major survey of visitors and residents that:

- Degradation of environmental values are likely to have real impacts in the tourism industry including:
 - Reductions in tourist satisfaction and less repeat visitation
 - Reduced numbers of tourists visiting the region
 - Tourists staying for shorter periods of time.
- Wages may need to rise in the region due to degradation of the Great Barrier Reef, due to a loss of aesthetic and recreational benefits important in retaining the current workforce with the current wage levels.

Mustika et al (2014) examined the potential implications of environmental deterioration for business and non-business visitor expenditures in the Great Barrier Reef. Mustika et al's (2014) conclusions were:

- Nature based tourism is an important source of income for the region
- More than 90% of non-business visitors and 67% of business visitors came to the region for at least one nature related reason
- Substantial environmental degradation could reduce visitor expenditure and thus local tourism income by at least 17%

13.7 Conclusion

The Great Barrier Reef is an important economic asset to the State of Queensland and to Australia. Its economic contribution to the Australian economy is estimated at \$6.4 billion annually and it supports 64,000 full-time equivalent jobs. Tourism accounts for most of the added value generated. The Great Barrier Reef is threatened by climate change and poor water quality runoff from catchments. The Australian and Queensland governments are investing approximately \$615 million over 5 years to address poor water quality through the Reef 2050 Water Quality Improvement Plan. This investment is a key contributor to regional economic development in North Queensland, supporting a variety of agricultural, scientific, administration, extension, earthmoving, revegetation, engineering, and monitoring and reporting services. Using the Economic Impact Analysis Model from Flinders University, it is estimated that investment under the Reef 2050 Water Quality Improvement Plan creates 1083 (574 direct and 509 indirect) full-time equivalent positions annually. While the majority of these positions are in the agricultural and scientific sectors, other economic sectors will also benefit substantially from this investment. While outside the scope of this analysis this investment also makes an important contribution to ongoing economic activity in North Queensland with several studies indicating that a decline in Reef health would have a detrimental impact on tourism related activities.

14.0 ACHIEVING THE POTENTIAL – REALISING EMERGING OPPORTUNITIES

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(The water quality credit trading configuration suggested in Figure 14.8 draws on concepts proposed during discussions with Nicholas Mathiou and Steven Chambers from Griffith Enterprise, and Professor Stuart Bunn from the Australian Rivers Institute at Griffith University).

14. Summary

This Chapter summarises findings from preceding chapters and reflects on how emerging technologies and appropriate market configurations could assist in realising the opportunities that water quality credit trading markets provide for facilitating economic expansion in Reef catchments within the water quality targets of the WQIPs.

14.1 Summary of key findings

14.1.1 Supply of nitrogen credits

Nitrogen credits from fertiliser practice change in cane

Chapter 4 explains how Paddock to Reef (P2R) APSIM modelling of DIN losses from sugarcane production over the period 1987-2013 is used to determine average annual DIN losses, and individual-year-specific DIN losses, to runoff and to drainage⁸⁶, from 563 cane land management units in the Wet Tropics. Average DIN losses to runoff and drainage vary with soil type, soil permeability, climate zone and fertiliser management practice (e.g. Figure 14.1). Variations in average DIN losses to runoff and drainage by management unit within catchments for a given level of management practice vary considerably, reflecting the variation in soil type, soil permeability and climate within the catchment (Figure 14.2). These results enable the average reductions in end of catchment DIN loss that follow from step improvements in fertiliser management practice to be estimated for each of the 563 management units in Wet Tropics catchments.

⁸⁶ As the majority of DIN loss travels via the drainage pathway, the predicted reductions in DIN loss are heavily dependent on an assumption of a uniform 60% transport coefficient for this pathway.

Average DIN losses 1987-2013 Wet Tropics

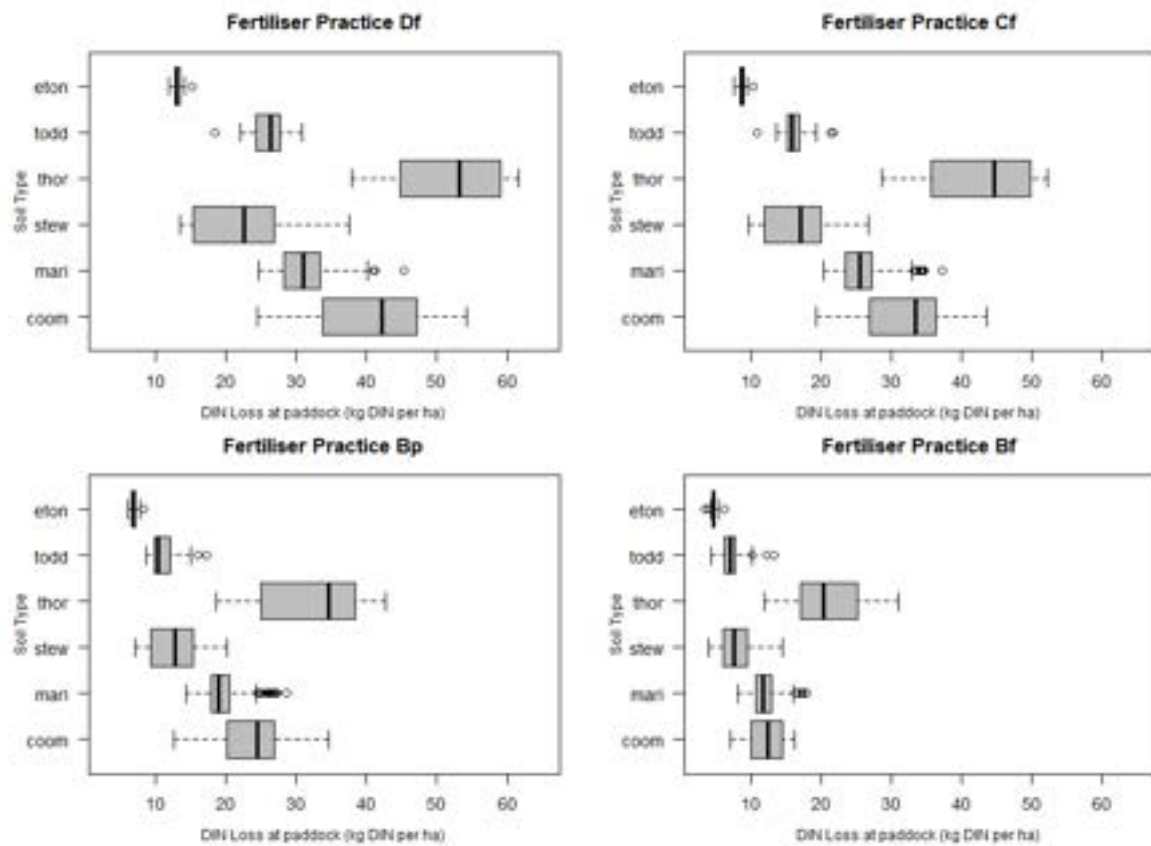


Figure 14.1: Average total annual DIN loss (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by soil type.

Average DIN losses 1987-2013 Wet Tropics

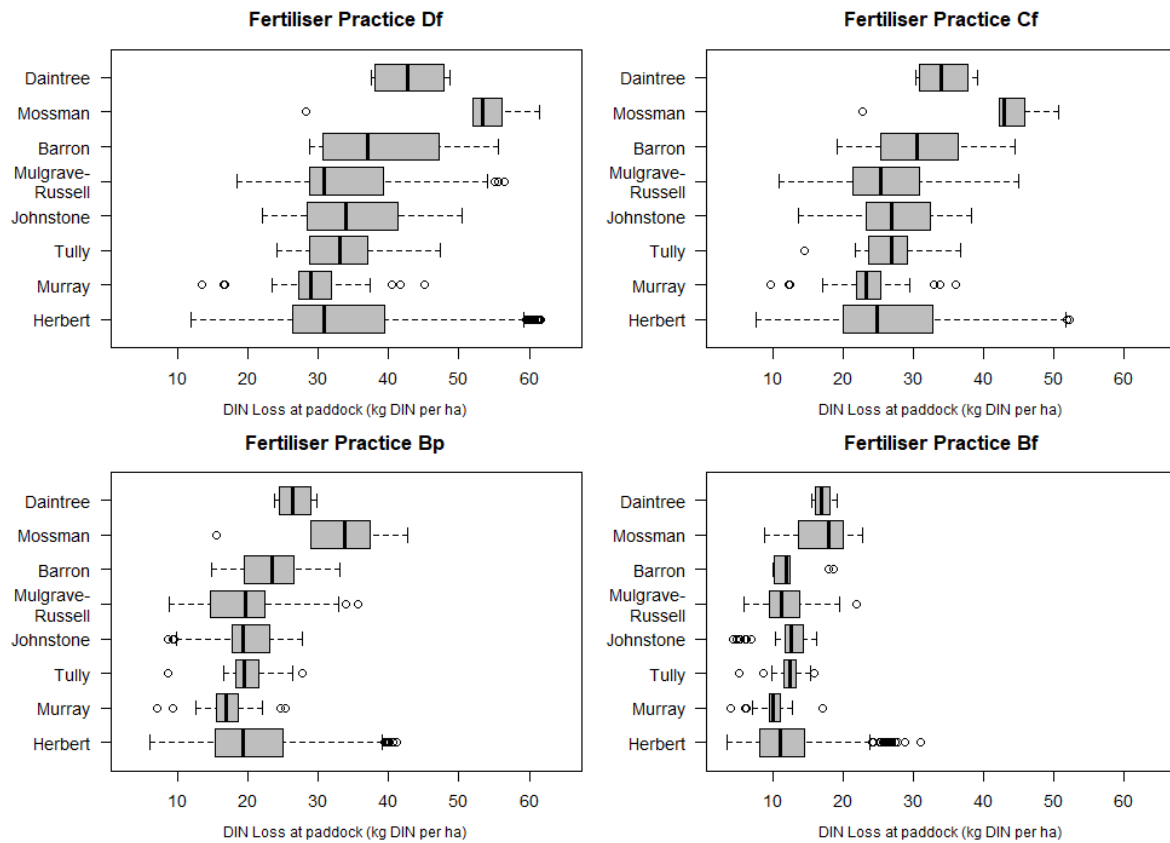


Figure 14.2: Average total annual DIN loss (1987-2013) at fertiliser management practices Df, Cf, Bp and Bf for 563 cane land management units in the Wet Tropics, grouped by catchment.

Landholders' minimum compensation requirements for undertaking step changes in fertiliser management practice are estimated in Chapter 5. Minimum compensation requirements are estimated as the sum of opportunity cost, compensation requirement for exposure to increased production risk, transition cost, and transaction cost. Per hectare opportunity cost and compensation requirement for exposure to production risk vary by management unit with changes in soil type, soil permeability, climate, and the extent of the change in fertiliser management practice. Transition cost (which largely comprises the cost of purchasing the machinery required to implement improvements in management practice) is generally the dominant element of the cost associated with improving management practice. Transition cost and transaction cost per hectare vary with farm size.

Soil type, soil permeability, and climate zone are known for each of the 563 management units modelled. Farm size and the current level of fertiliser management practice for individual management units are not publicly available. However, the overall proportions of farms in particular size classes (small, medium and large), and at particular levels of current management practice, are known for each Wet Tropics catchment. This permits stochastic simulation of management unit-specific compensation requirements for changing fertiliser management practice through randomised allocation of farm size and current fertiliser management practice across management units, in combination with P2R APSIM-derived, management unit-specific estimates of opportunity cost and compensation for exposure to increased production risk.

Stochastic simulation estimates the cost effectiveness (\$/kgDIN reduction @ End-of-Catchment) at which management unit-specific improvements in fertiliser practice generate End-of-Catchment reductions in DIN load. A supply curve for nitrogen credits can be produced from each stochastic simulation run by ordering changes in fertiliser practice from most to least cost-effective (i.e. ordering fertiliser practice changes from the lowest \$/kgDIN reduction to the highest \$/kgDIN reduction). Figure 14.3 below shows the stochastic simulations of DIN credit supply curves from improvements in fertiliser management practice in the Johnstone catchment.

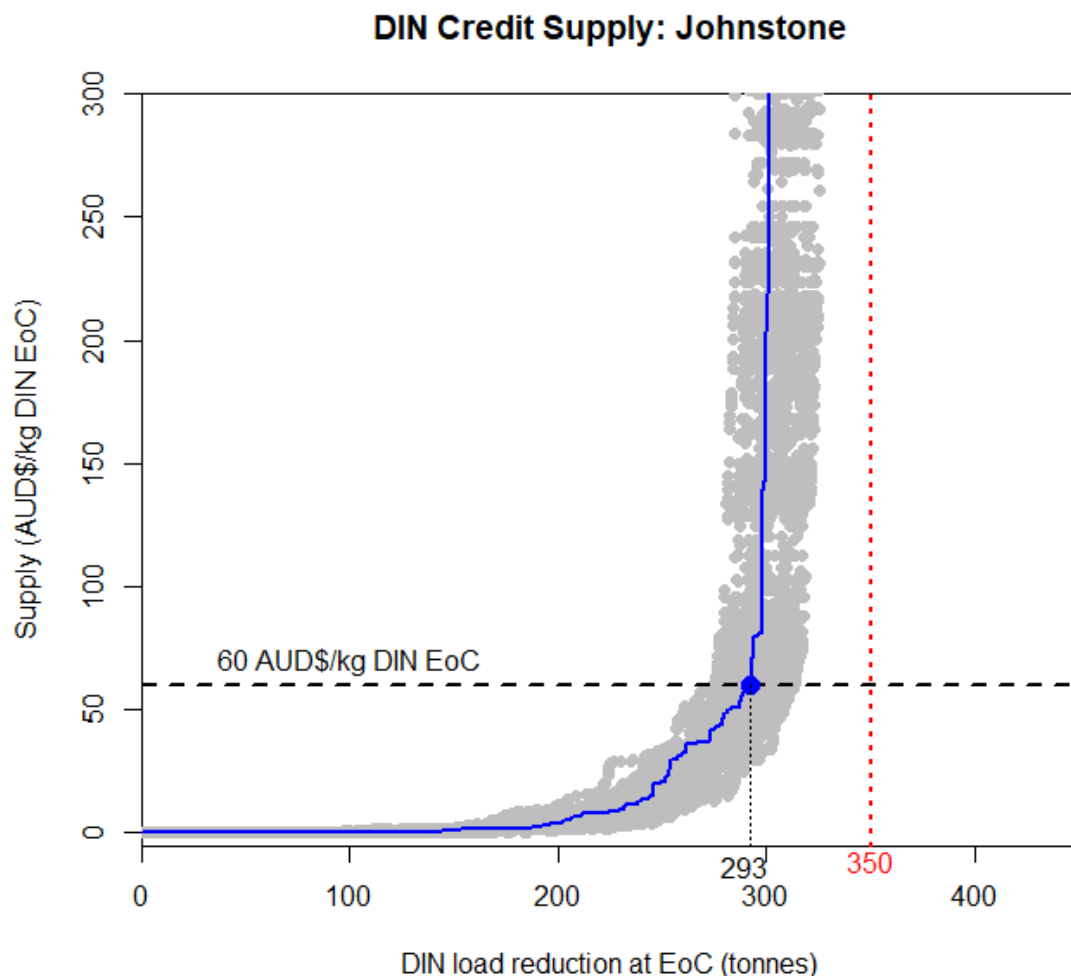


Figure 14.3: Supply curve for End-of-Catchment DIN credits from cane fertiliser practice change in the Johnstone catchment. Grey dots show 50 randomised replicate supply curves. The blue line shows a representative mid-band supply curve. The End-of-Catchment DIN load reduction target required to meet the catchment's Water Quality Objective is shown in red. Credit supply price and quantity required to achieve more than 95% of the available cumulative DIN load reduction from adoption of practice level Bf are shown in black for the mid-band supply curve.

The estimated supply curves for DIN credits from fertiliser practice change in the Johnstone are reasonably representative of those for all the main Wet Tropics catchments. DIN credits are typically available at very low cost for improving fertiliser management practice from levels Df, Cp, and Cf to Bp. Further improvement to level Bf incurs considerable additional cost, but

still delivers substantial additional reductions in DIN loss on most soils. The one remaining step of practice change – Bf to Af – entails within-block variation in fertiliser applications, based on fine-scale variation in historically-derived ‘block yield potential’. This incurs considerable additional cost but returns only very modest additional DIN reductions. Consequently, the cost-effectiveness of DIN reductions from this final step of practice change is very low, i.e. the price of DIN credits sourced from further improvement in nitrogen management practice beyond practice level Bf.

Figure 14.3 shows that near-full implementation of practice level Bf produces the outcomes indicated by the dotted black lines for the representative mid-band supply curve (shown in blue), i.e. 293 tonnes of EoC DIN load reduction at costs of no more than 60 \$/kgDIN End-of-Catchment. The width of the grey band of stochastic simulations on either side of the blue mid-band supply curve indicates that full implementation of practice level Bf could deliver 293 ± 21 tonnes EoC DIN load reduction, depending on the level of fertiliser practice already adopted by farms of different sizes⁸⁷.

Under the assumptions inherent in P2R APSIM, and the strong assumption of uniform 60% transport to End-of-Catchment for DIN travelling via the groundwater drainage pathway, catchment-specific stochastic nitrogen credit supply curves suggest that fertiliser practice change through to practice level Bf (as will likely be required to comply with a whole-of-farm nitrogen budget as specified in the Agricultural ERA Standard for sugarcane) should generate considerable DIN reductions at End-of-Catchment at per-unit costs below approximately 50 – 60 \$/kgDIN reduction at EoC. Based on modelling results, it appears unlikely that full adoption of fertiliser management practice Bf will deliver the desired Water Quality Targets for the Johnstone and Herbert catchments. Simulation results in Chapter 5 suggest that Water Quality Targets for the Mulgrave-Russell and Tully catchments may also prove difficult to achieve completely solely through full adoption of practice level Bf. The costs of supplying DIN credits from further practice improvement increase very rapidly to more than \$250/kg DIN EoC. This suggests that once fertiliser practice has improved to level Bf other mechanisms such as installation of constructed treatment systems should be considered for cost-effective supply of DIN credits and further reduction of nitrogen loads towards the Water Quality Targets.

Nitrogen credits from land use change and constructed treatment systems

Chapter 6 considers the potential for supplying nitrogen credits through land use change (e.g. setting aside less profitable cane land) and construction of nitrogen treatment systems (e.g. farm-scale constructed treatment wetlands and field-scale bioreactors). Chapter 6 draws heavily on recently published research on the effectiveness of nitrogen removal by natural wetlands in the Wet Tropics (Adame et al., 2019a, 2019b; Waltham et al., 2019) and new results from the Wet Tropics Major Integrated Project (WT-MIP) on the costs of constructing farm- and field-scale nitrogen treatment systems (Australian Wetlands Consulting 2020).

⁸⁷ If most of the large farms in the catchment have already adopted high (i.e. ‘good’) levels of fertiliser management practice, a smaller reduction in total EoC DIN load will result from improving practice on the remaining of farms. This situation will produce a stochastic (grey) DIN credit supply curve that sits to the left of the mid-band blue curve. If most of the large farms in the catchment remain at lower levels of fertiliser practice, the opposite outcome arises.

For setting aside cane land on which fertiliser practice is likely to be sufficient to comply with whole-of-farm nitrogen budgets as specified under the Agricultural ERA for sugarcane cultivation (i.e. at fertiliser management practice Bf), representative estimates for average gross margins and higher-end estimates for DIN losses for coom and mari soils suggest that End-of-Catchment DIN credits from this mechanism could potentially be supplied for c. 130 \$/kgDIN EoC.

Cost data from wetlands and bioreactors constructed for the Wet Tropics MIP, together with indicative estimates of nitrate removal rates from the literature, suggest that End-of-Catchment DIN credits could potentially be supplied from landscape and embellished wetlands for c. 16 – 34 \$/kgDIN EoC. Landscape and embellished wetlands appear to be more cost-effective sources of DIN credits than set aside because, although the per-hectare costs incurred are c. 15 times higher, the anticipated per hectare DIN removal rates are 75 times higher (c. 10 kg DIN ha⁻¹ year⁻¹ for set aside vs. c. 750 kg DIN ha⁻¹ year⁻¹ for constructed wetlands), producing a cost-effectiveness ratio of 5:1 in favour constructed wetlands. These results suggest that landscape and embellished wetlands could potentially provide the most cost-effective source of DIN credits once cane management practice is fully compliant with whole-of-farm nitrogen budgets as specified under the Agricultural ERA for sugarcane cultivation.

Results from the Wet Tropics MIP suggest that vegetated drains could potentially source DIN credits at even lower cost, but there is considerable variation in the cost-effectiveness of this option, depending on the extent of earthworks required. Initial results indicate that there is also considerable variation in the cost-effectiveness of sourcing DIN credits from Wet Tropics MIP bioreactors, with DIN credits from these sources costing between c. 50 – 490 \$/kgDIN EoC.

The cost-effectiveness outcomes quoted here use DIN removal rates from the literature. Data on actual DIN removal rates from treatment systems constructed under the Wet Tropics MIP are currently being analysed. The cost effectiveness outcomes reported here should be re-calculated once site-specific nitrate removal rates become available.

14.1.2 Demand for nitrogen credits

Chapter 7 explored potential demand for nitrogen credits from sewage treatment plants (STPs) and aquaculture facilities as point sources. Data from the Environmental Authorities Register database (Sudarjanto et al. 2017) and qldwater (2018, 2017) indicate that all large STPs, and many smaller STPs, in Wet Tropics catchments already operate with nitrogen discharge concentrations of 5 mgN/L or lower, with only a small number of older technology STPs still operating at discharge concentrations of 30 mgN/L. Based on reported discharge concentrations, Chapter 7 (Table 7.2) identified that if these older-specification STPs sought to purchase DIN credits as a long-term policy to avoid upgrading to a de facto 5 mgN/L discharge standard, this would require offsetting of around 22 tonnes, 11 tonnes and 10 tonnes of point-source STP N-discharges to water annually in the Herbert, Mulgrave-Russell, and Mossman catchments, respectively.

Chapter 7 assumes that the sum of avoided annualised expenditure on upgrading capex and avoided incremental annual opex represents the maximum amount STP operators would be willing to pay for the quantity of diffuse-source DIN credits required to offset point-source discharges to water in excess of 5 mgN/L from their older generation STPs. This enables

catchment-specific demands for diffuse-source DIN credits to be calculated for these three catchments, taking account of the 1.5:1 diffuse-source to point source delivery ratio currently specified in the PSWQ Offsets Policy. The consequent demands for diffuse-source DIN credits are⁸⁸: 33.5 tonnes at a maximum price of 46 \$/kgDIN for the Herbert, 15 tonnes at a maximum price of 75 \$/kgDIN for the Mossman, and 5 tonnes at a maximum price of 309 \$/kg DIN, plus a further 11 tonnes at a maximum price of 76 \$/kg DIN, for the Mulgrave-Russell⁸⁹. The supply of DIN credits from constructed treatment systems (Table 6.2) suggest that these DIN credit demands could be supplied by c. 44 ha, 22 ha, and 20 ha of constructed treatment wetlands in the Herbert, Mulgrave-Russell and Mossman catchments, respectively⁹⁰.

Further south along the Reef coast, again drawing on data from the Environmental Authorities Register database (Sudarjanto et al. 2017) and qldwater (2018, 2017), and applying the same calculation methodology (including the impact of the current 1.5:1 delivery ratio), significant demands for DIN offsets from STPs currently discharging at concentrations above 5mgN/L could arise in Townsville, Mackay, Rockhampton, Bundaberg and Gympie.

Chapter 7 also explored potential demand for diffuse-source DIN credits for 'short term offsetting' by STPs that already deliver 5 mgN/L discharge performance but are approaching their capacity limits due to population growth and are seeking to defer the cost of capacity augmentation. If large (c. 7000ML annual volume treated) STPs in Townsville or Cairns look to use this approach, they would generate c. 0.5 tonne of demand for diffuse-source DIN credits per 1,000 additional population, and would be willing to pay up to c. 70 \$/kgDIN per diffuse-source DIN credit⁹¹.

New aquaculture facilities were also explored as a potential source of demand for diffuse-source DIN credits. However, the recently granted approval for the new Guthalungra prawn farm suggests that high-performance macroalgal treatment, with associated 'polishing' mangrove areas, appear to be the preferred solution for this sector – from the perspective of both the industry and the regulator.

In summary, once the levels of fertiliser management practices that are likely to be required to comply with whole-of-farm nitrogen budgets under the Agricultural ERA for sugarcane cultivation have been fully adopted, constructed treatment wetlands will likely provide the most cost-effective supply of diffuse-source DIN credits. Initial results from landscape and embellished wetlands constructed under the Wet Tropics MIP suggest that DIN credits from constructed treatment wetlands could be available at prices which most STPs that are seeking to avoid upgrading to 5mgN/L performance, and STPs that are seeking to delay the cost of capacity augmentation, would be willing to pay. The current level of demand for diffuse-source DIN credits in Wet Tropics catchments appears to be quite modest, however, because most

⁸⁸ Maximum willingness to pay figures assume avoided capex is annualised over 50 years at 7% discount rate, and that the PSWQ Offset Policy delivery ratio is 1.5:1.

⁸⁹ DIN credit demand for the Mulgrave-Russell is a conservative estimate because Hinchinbrook Shire Council provide amalgamated data for their two STPs (Ingham and Lucinda). Offsetting calculations are implemented for these two plants combined, thus reducing the capex and opex intensity.

⁹⁰ These wetland areas comprise c. 0.07%, 0.09%, and 0.4% of the cane growing areas in the Herbert, Mulgrave-Russell and Mossman catchments, respectively.

⁹¹ These price and quantity estimates are only approximate. An STP-specific calculation would be required to generate more precise estimates.

STPs are already at 5mgN/L performance, and because population growth is currently relatively slow. More sizeable DIN credit demand could potentially arise from STPs further south along the Reef coast.

Findings from this Report suggest that, when economic growth in the region accelerates, DIN offsetting has the potential to be an important facilitator of cost-effective economic expansion with no net decline in water quality.

14.1.3 Supply of sediment credits

Drawing on a major new detailed study that encompassed more than 22,000 individual gullies across sections of the Bowen and Bogie sub-catchments in the Burdekin region for the Landholders Driving Change MIP, Chapter 9 explored generation of sediment credits from gully remediation. Informed by high-resolution LiDAR-based mapping of gullies, Brooks et al. (2020b) determined:

- gully-specific morphometrics (slope, width, steepness etc.) from LiDAR data
- appropriate categorisation of gully form to identify relevant remediation measures
- erosion rates, and thus the contribution of individual gullies to sediment loads at sub-catchment scale

LiDAR-derived gully-specific estimates of erosion rate, were combined with assumptions of the relative effectiveness of different intensities of remediation measures, and estimated remediation costs (based on new data from large-scale gully remediation works in the region) to estimate the cost-effectiveness of gully remediation as a source of fine sediment credits (\$/tonne of fine sediment abated). The resulting estimated supply curves for fine sediment credits from gully remediation in the Bowen and Bogie sub-catchments are shown in Figure 14.4, Figure 14.5 and Figure 14.6.

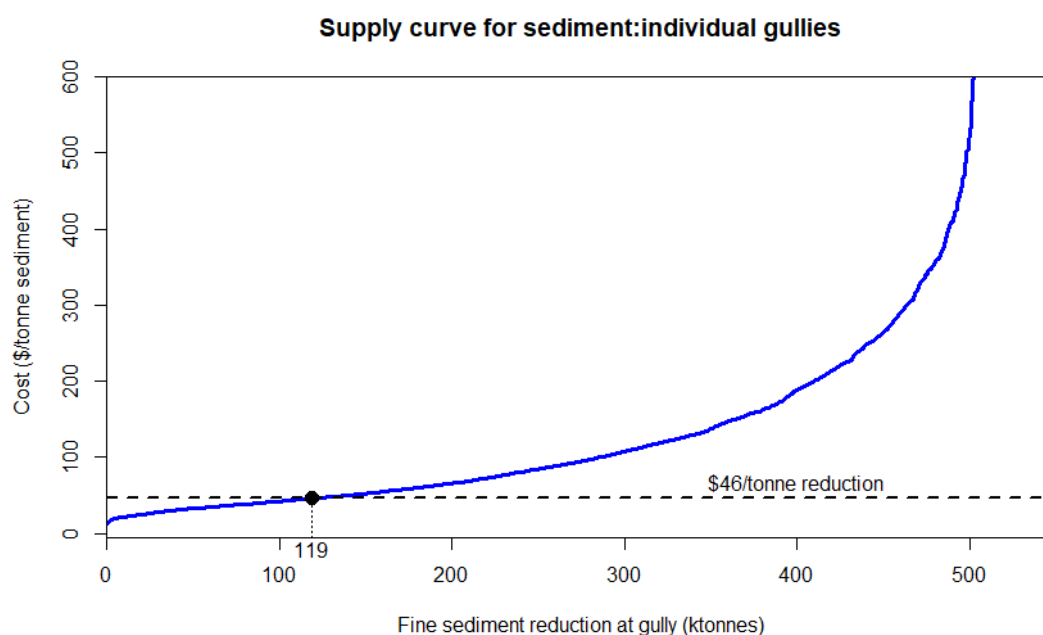


Figure 14.4: Supply curve for fine sediment reduction by remediating individual gullies in the Bowen-Bogie sub-catchments. Given a \$22M budget limit on upfront cost, a total annual sediment load reduction of 119 ktonnes could be achieved at a maximum cost of \$46 per tonne reduction (annualisation of the total present value of up-front and on-going costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b))

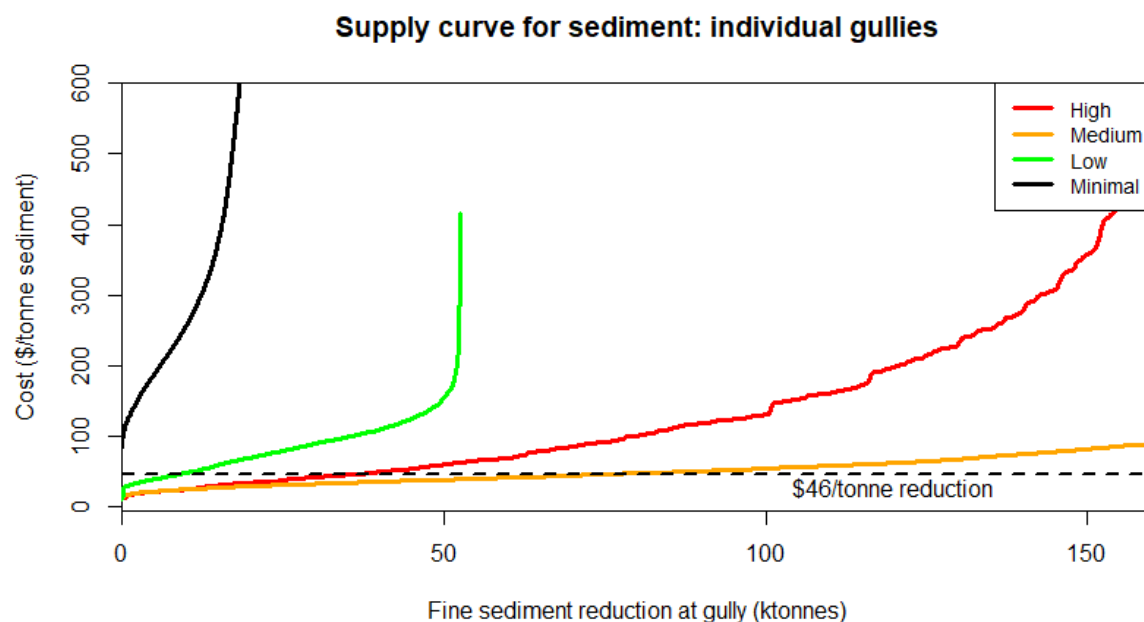


Figure 14.5: Supply curves for sediment reduction by remediating individual gullies in the Bowen-Bogie sub-catchments: supply grouped by treatment intensity – high, medium, low, minimal. (Annualisation of the total present value of up-front and on-going costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b))

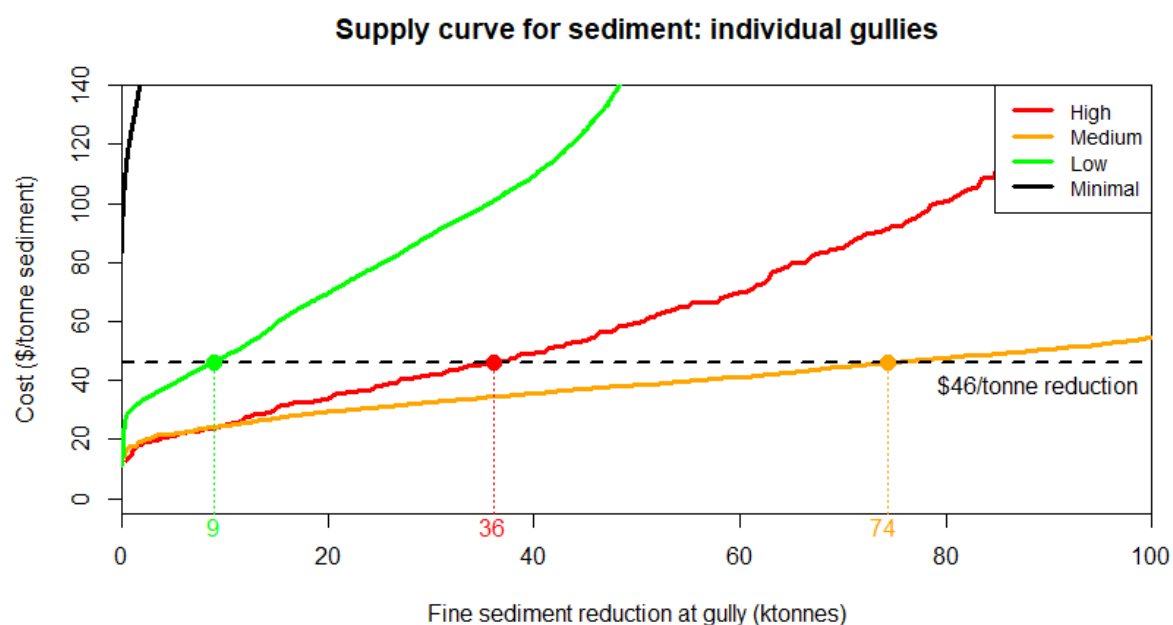


Figure 14.6: Detail of supply curves for sediment reduction by remediating individual gullies in the Bowen-Bogie sub-catchments: supply grouped by treatment intensity – high, medium, low, minimal. With a budget limit of \$22M on upfront cost, remediation would be pursued up to an annualised present value cost of \$46/tonne abated annually. At the upfront cost limit, 36, 74, 9 and 0 ktonnes of fine sediment abatement would be delivered by remediating high-, medium-, low- and minimal-intensity gullies,

respectively (Annualisation of the total present value of up-front and on-going costs at 7% real discount rate over 25 years). (Figure reproduced from Brooks et al. (2020b))

These findings suggest that:

- Appropriately targeted remediation of large alluvial gullies in Bowen-Bogie sub-catchments in the Burdekin region has the potential to deliver substantial reductions in End-of-Catchment fine sediment load.
- Initial results suggest that around 120,000 tonnes of sediment load could be removed from the Bowen and Bogie sub-catchments (in combination) for a total up-front expenditure of \$22 million.
- This equates to supply of 120,000 tonnes of fine sediment reductions (i.e. 120,000 tonnes of fine sediment credits) for no more than c. \$45/tonne based on annualised cost per annual tonne of sediment abated. (Annualisation of up-front and on-going costs at 7% real discount rate per annum over 25 years).
- The cost of supplying sediment credits will inevitably rise as the required sediment load reduction increases, but these results suggest that – when appropriately targeted by high resolution spatial prioritisation – remediation of large alluvial gullies appears to provide a valuable source of cost-effective sediment reduction.

14.1.4 Demand for sediment credits

In addition to the PSWQ Offsets Policy, two other frameworks address offsetting of emissions to water in GBR-related contexts. Both of these could potentially be relevant as sources of demand for sediment credits:

1. As described in Chapter 2, defined activities within the Great Barrier Reef Marine Park or Queensland's Great Barrier Reef Coast Marine Park can offset any impacts that remain unmitigated by on-site actions by making in-lieu payments to the Reef Trust's Special Account at rates determined by the Reef Trust Offsets Plan and Calculator (Reef Trust 2017). The specified offset payment rates allow for an estimated success rate, uncertainty regarding restoration cost, the suitability of the restoration site as a surrogate for the impacted site, and the time taken for the offset to become fully effective. The current version of the Reef Trust Offsets Plan and Calculator (Reef Trust 2017) specifies a one-off in-lieu payment of 464 \$/tonne of unmitigated sediment. This equates to an annualised equivalent value of c. \$40 year⁻¹ tonne⁻¹ when annualised at 7% real discount rate over a 25-year period (to match the annualisation parameters of the gully remediation costs in Chapter 9). Referring to the sediment credit supply curve in Figure 14.6, c. 85 ktonnes of sediment reduction could potentially be achieved by gully remediation in the Bowen-Bogie from annual expenditures of no more than \$40 tonne⁻¹ year⁻¹.
2. As also mentioned in Chapter 2 and Chapter 10, State Planning Policy allows for the management of unmitigated post-construction pollutant emissions from urban stormwater by off-site implementations of appropriate catchment-based green infrastructure solutions (e.g. riverbank rehabilitation, gully remediation, wetland construction). The 2018 Draft Implementation Guidance (State of Queensland 2018a) states that post-construction emissions from an urban development site can be managed by developers making appropriate in-lieu payments to local government.

Local government then have the responsibility to deliver off-site treatment solutions that provide water quality outcomes equivalent to those that would have been achieved by on-site treatment. In a 2016 report, Alluvium estimated the extent of demand for sediment offsetting from this source (Alluvium, 2016; Appendix B-7, p.203-211). Alluvium's (2016) demand estimates were based on the Queensland Government's 2013 predictions of anticipated annual growth in the number of dwellings in urban centres along the Reef coast between 2016 and 2025 (Alluvium, 2016; Appendix B-7, Table 105, p.205). Population growth along the Reef coast has slowed since 2013, and the Queensland Government's most recent predictions of anticipated annual housing growth (Queensland Treasury 2018) are lower than those used to predict demand for sediment offsetting from this source in the Alluvium (2016) Report. Scaling back the Alluvium (2016) projections of demand for sediment offsets in proportion with the reduction in projected housing growth produces the predicted offset demands in Table 14.1. Alluvium report an extremely low cost-effectiveness (i.e. an extremely high price) for reducing sediment from urban stormwater runoff via on-site management (Alluvium, 2016; Table 25, p.64). This suggests that developers would likely be willing to pay the prices required to obtain diffuse-source sediment credits for the full extent of the sediment credit demand shown in Table 14.1, even allowing for a delivery ratio (if applied).

Table 14.1: Predicted demand for sediment credits from major cities along the Reef coast. Derived from projections by Alluvium (Alluvium, 2016; Appendix B-7, p.203-211), revised to account for the Queensland Government's reduced projections of housing growth in the region (Queensland Treasury 2018).

Location	<i>Alluvium (2016) predicted demand (tonnes/year)</i>	<i>Revised predicted demand (tonnes/year)</i>
<i>Cairns</i>	80.4	54.7
<i>Townsville</i>	48.5	28.1
<i>Mackay</i>	33.2	16.3
<i>Rockhampton</i>	17.9	12.7

In addition to the existing offsetting mechanisms as current sources of potential demand for diffuse source sediment credits, Chapter 10 also considered the possibility of hypothecating a small proportion of the value of commodities traded from Reef coast ports that undertake frequent (i.e. annual) or regular (i.e. two- to five-yearly) maintenance dredging into a fund to implement sediment reduction measures in adjacent catchments. Based on reported volumes of key commodities shipped and pricing from FY 2016-17 (Chapter 10, Table 10.6) hypothecation of 0.1% of the traded value of alumina, aluminium, coal, LNG and sugar shipped from Gladstone, Hay Point, Mackay, Townsville and Cairns ports would raise in excess of \$30 million annually. Referring to the sediment credit supply curve for the Bowen-Bogie sub-catchments as an example (Figure 14.4), an annual funding input of this scale would be sufficient to purchase very substantial quantities of sediment reduction.

14.1.5 Trading ratios: diffuse source credits to offset point source emissions

Under the Queensland Point Source Water Quality Offsets Policy (PSWQ Offsets Policy), point source entities operating under Environmental Authorities (EAs) (i.e. "licenced limits" on

their emissions to receiving water) can offset exceedances of the annual discharge load limit stipulated in their EA by buying a sufficient number of water quality credits. Credits can be purchased from other point source entities or from diffuse sources. Where point source exceedances are offset by credits purchased from diffuse sources (either directly or via a credit trading market) the quantity of diffuse-source credits required to offset one unit of licenced point source emissions is determined by the delivery ratio and – if appropriate – the offset environmental equivalency ratio specified in the PSWQ Offsets Policy (Figure 14.7). The delivery ratio is currently set at 1.5:1; the environmental equivalency ratio at 1:1.

Taking nitrogen practice improvement in cane production as an example, Chapter 8 used year-specific P2R APSIM predictions of DIN losses to explore the delivery ratio required to ensure that, for a given level of practice improvement on each soil type, the mean level of End-of-Catchment DIN reduction for that practice improvement and soil type is achieved on average 4 years out of 5, or 9 years out of 10 (Chapter 8, Table 8.1). Results suggest that the currently specified delivery ratio of 1.5:1 should be sufficient to achieve the mean level of End-of-Catchment DIN reduction 4 years out of 5 on average for mari soil (the predominant soil type across the Wet Tropics); however, a delivery ratio of 2:1 would be required to achieve the same performance from coom soil (the second most common soil type in the Wet Tropics). To ensure mean levels of End-of-Catchment DIN reduction 9 years out of 10 on average would require a delivery ratio of 2:1 on mari soils, and 3.1:1 on coom soils.

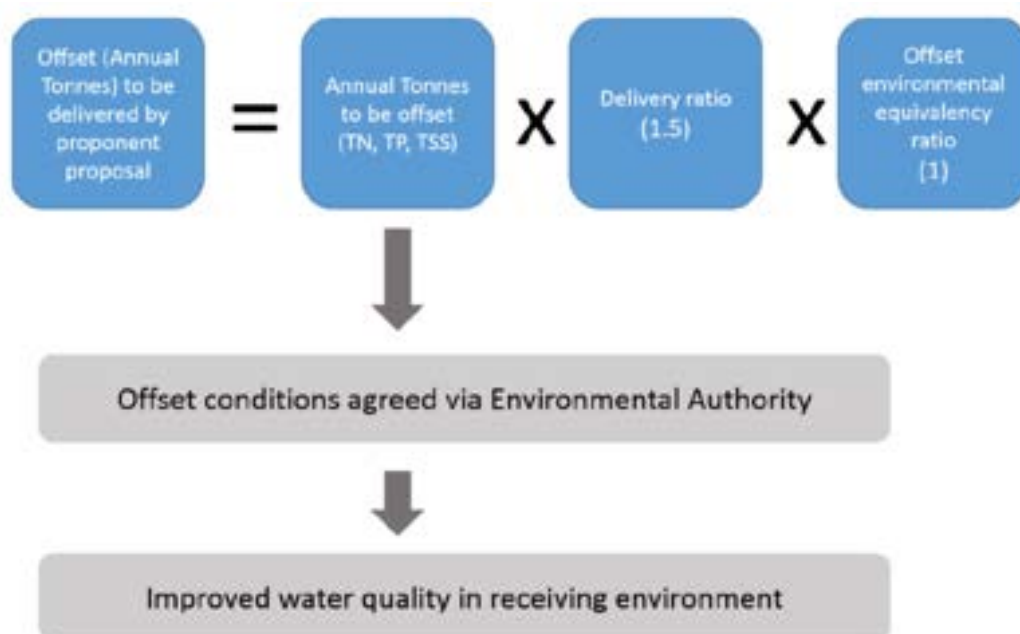


Figure 14.7: Conceptual diagram illustrating how delivery ratio and offset environmental equivalency ratio address uncertainty regarding the offsetting capacity of water quality offsets in the receiving environment to which the purchasing point source's EA conditions apply. (Copied from Point Source Water Quality Offsets Policy 2019, Figure 1, p.3 (State of Queensland, 2019f)).

Data to support an equivalent evaluation of site-to-site and year-to-year variation in the End-of-Catchment DIN removal delivered by constructed treatment systems should be regarded as a priority for further research. Under full implementation of the 2019 Reef Regulations, at the current 1.5:1 delivery ratio, landscape and embellished wetlands appear to offer the most cost-effective supply of DIN credits – at price levels that could support permanent offsetting by

smaller, older technology STPs and temporary offsetting to delay capacity augmentation by larger, tertiary technology STPs. DIN credit demand from these sources would likely be reduced considerably if delivery ratios of 2:1 or 3:1 were required to ensure the desired level of environmental integrity in receiving waters.

Determining an appropriate environmental equivalency ratio that adequately reflects site-to-site and year-to-year variation in the reduction in bioactive nitrogen generated per tonne of diffuse source sediment credit should also be regarded as a priority for on-going research in this topic area (building on e.g. Franklin et al., 2018; Garzon-Garcia et al., 2018). A substantial change in the environmental equivalency ratio from its current setting of 1:1 would likely have considerable consequences for the demand for diffuse-source DIN offsets derived from sediment reduction.

14.1.6 Employment opportunities

Chapter 13 used the Economic Impact Analysis Model from Flinders University to estimate that investment under the Reef 2050 Water Quality Improvement Plan creates 1083 (574 direct and 509 indirect) full-time equivalent positions annually. This equates to approximately 1 equivalent direct or indirect full-time position per \$114,000 of annual investment in restoration activities. Similar levels of employment opportunities could reasonably be anticipated from in-catchment works to supply water quality offsets. At the expenditure levels indicated in Sections 14.1.1 and 14.1.3 for nitrogen and sediment credits, respectively, this loosely equates to 1 equivalent direct or indirect full-time position per 2 tonnes of DIN credits, or per 2,500 tonnes of sediment credits.

Whilst throughout this Report every attempt has been made to base predictions of supply and demand for DIN and sediment credits on the most relevant and recent data, potential compounding of errors poses a challenge, as it does in all ex ante modelling, particularly of inter-connected biophysical – economic systems. In line with recommended best practice (Uusitalo et al. 2015; Guillaume et al. 2017), the following approaches were used to provide a counterbalance to the potential for compounding of errors: sensitivity analysis (e.g. regarding the allocation of sugarcane management units to farms of different sizes and to different initial levels of management practice in Chapter 5; regarding the lifetime for annualisation of the avoided costs of upgrading or expanding STPs in Chapter 7), incorporation of data that exemplify temporal and spatial variability (in the supply curves for sediment credits from gully remediation in Chapter 9), and expert judgement (in consultation with the authorship team and colleagues – throughout the Report). In addition, this Report has been independently reviewed by subject experts from the Queensland government's Office of the Great Barrier Reef, and by two independent academic reviewers to ensure that the analyses it contains can be regarded as robust, defensible and appropriate.

14.2 Realising opportunities

Findings from this Report suggest that water quality credit trading in nitrogen and sediment offsets has the potential to be an important facilitator of cost-effective economic expansion along Queensland's Reef coast, with no net decline in water quality. Water quality credit trading schemes enable point source emitters to avoid incurring high pollution abatement cost by allowing them to offset their emissions through purchase of credits. The scheme offers point

source emitters flexibility in identifying cost-effective options for managing their emission liabilities. At an aggregate level, if sufficient numbers of trades take place, credit trading schemes offer the potential to achieve water quality goals at least cost to society.

However, despite their acknowledged potential, there remain few examples worldwide for which water quality credit trading has succeeded in generating sufficient numbers of trades to achieve its objective (Ribaudo and Gottlieb 2011; Shortle 2013; Tabaichount et al. 2019). This implies that, in most instances, the expected cost savings fail to materialise (Ribaudo and Shortle 2019). Key challenges that have prevented water quality credit trading from achieving its potential can be identified from the perspectives of credit buyers, credit suppliers and the environmental regulator, as described in the following sections.

14.2.1 Challenges from the perspective of credit buyers:

The legal position regarding liability for emissions in excess of licenced limits is a key concern from the perspective of potential credit buyers. The Reef Trust Offsets Program and State Planning Policy regarding off-site solutions for un-mitigated sediment and nitrogen emissions from urban stormwater both operate as fee-in-lieu mechanisms under which credit buyers' liability ceases once the in-lieu payment has been made. Under the PSWQ Offsets Policy, however, liability for the adequacy of purchased credits as an offset for emissions in excess of licenced loads remains with the credit buyer – even if 'verified' credits have been purchased via a market. Information asymmetry (e.g. the credit buyer will not know for sure whether necessary on-ground maintenance has actually been undertaken) is thus likely to be a considerable concern for PSWQ Offset Policy buyers. This may lead to risk averse PSWQ Offset Policy buyers to purchase higher quantities of credits, which in turn will reduce their maximum willingness to pay per credit. Risk aversion is likely to be particularly prevalent among PSWQ credit buyers during the early stages of market development when credit supply actions and verification mechanisms are still becoming established.

Water quality trading markets aim to provide flexible, cost-effective options that will enable regulated point source emitters to fulfil their licence conditions. Point source emitters will only engage with credit markets if they are convinced that credit purchase provides a secure long-term option for managing their environmental liabilities. Point source buyers will therefore be unlikely to participate in credit markets if they perceive that there is a significant risk that insufficient quantities of offsets will be available at appropriate prices over the medium-long term. Concerns regarding the longer-term viability and availability of credits are thus likely to deter credit buyers from engaging in credit markets.

14.2.2 Challenges from the perspective of credit suppliers:

High transaction costs (Ribaudo and Gottlieb 2011; Coggan et al. 2014; Motallebi et al. 2017) and inadequate compensation for exposure to risk (O'Connell et al. 2017; Kandulu et al. 2018) are frequently cited deterrents to credit suppliers' participation in credit markets. Financial risk (the risk that revenues from credit sales will be insufficient to cover the costs incurred in credit supply) will likely be a major concern for suppliers – particularly during the early stages of market development before demand is well established.

Land use change and land remediation incur substantial up-front costs, but both mechanisms are likely to provide relatively cost-effective credit supply (e.g. conversion of marginal cane

land to wetlands, installation of field-scale bioreactors, remediation of rapidly eroding alluvial gullies). Large up-front costs expose credit suppliers to considerable financial risk. This is likely to be a strong deterrent to market participation, particularly in the early stages of market development before demand (and thus credit pricing) is well established.

14.2.3 Challenges from the perspective of the Environmental Regulator:

The Environmental Regulator is responsible for safeguarding the environmental quality of receiving waters. The Regulator is thus naturally risk averse with regard to environmental outcomes. Risk averse behaviour from the regulator is amplified in the face of information asymmetry and when uncertainty is pervasive. Information asymmetry regarding the adequacy of on-ground actions that generate credits, and their continuing maintenance, is therefore a concern for the Regulator as well as for credit buyers. Uncertainty regarding the reliability of an offset action may lead the Regulator to press for a higher delivery ratio and a higher environmental equivalency ratio in the PSWQ Offset Policy, and/or higher charges in the fee in-lieu schemes. Both of these responses will act to dampen credit demand. Risk aversion can also manifest as multiple interactions with the Regulator during project planning, implementation, verification, trading and on-going monitoring – increasing transaction costs for all parties.

14.3 Addressing the challenges: recommended market configuration supported by technological innovations

Recognising the challenges described in the preceding sections, the market configuration in Figure 14.8, supported by the technological innovations depicted in Figure 14.9, is recommended for implementing water quality credit trading along the Reef coastline to facilitate economic expansion whilst also maintaining water quality outcomes for the Great Barrier Reef. Key features of the suggested market configuration and associated technological innovations are outlined the following sub-sections.

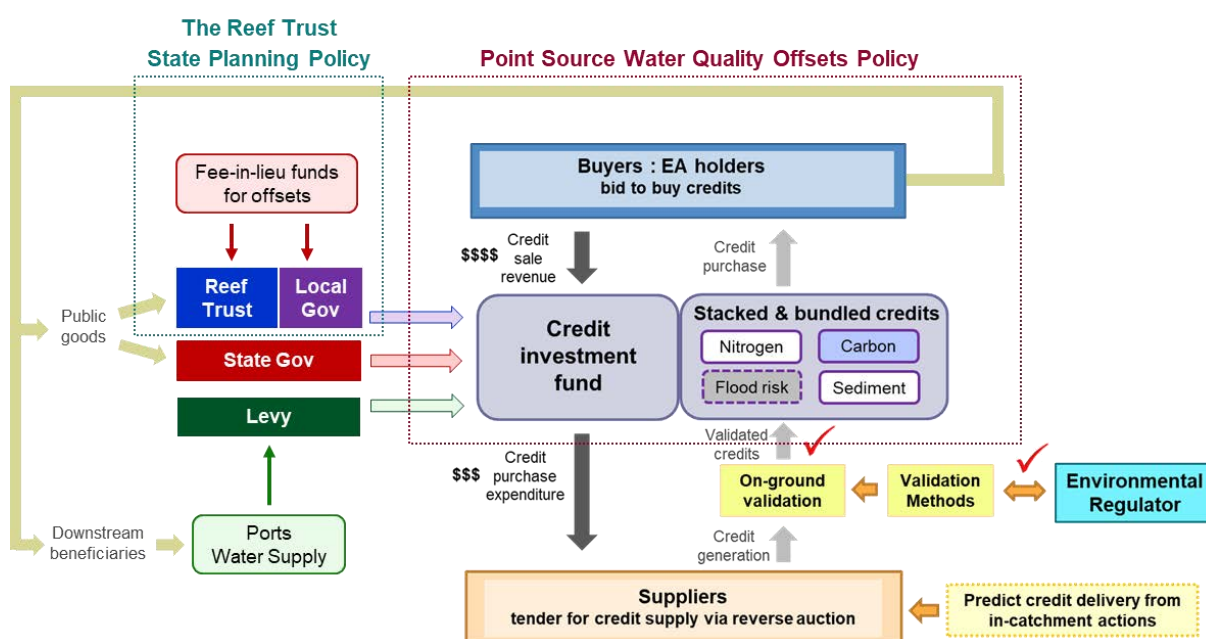


Figure 14.8: Suggested configuration for a market in water quality credits to facilitate economic expansion along the Reef coast, whilst also maintaining water quality outcomes for the Great Barrier Reef.

14.3.1 Active participation from government

Research that addresses two of the most active and effective water quality trading programs to date – the Lake Taupo Nitrogen Trading Program in Waikato, New Zealand, and the South Nation Total Phosphorus Management Program in Ontario, Canada – suggests that active participation from relevant arms of government is a key factor in their success (Tabaichount et al. 2019). Tabaichount et al. consider that active support from government in these examples creates a hybrid institutional structure that combines public and private funding sources with aspects of top-down regulation and bottom-up market trading to secure cost-effective delivery of desired water quality outcomes. Building on Tabaichount et al's insights, we suggest that combining existing elements in the configuration shown in Figure 14.8, supported by the technological innovations shown in Figure 14.9, should facilitate economic expansion along the Reef coast, whilst also maintaining water quality outcomes for the Great Barrier Reef.

In Figure 14.8 monies from the Reef Trust Offset Fund and SPP urban stormwater offsets are combined with funding from state government (e.g. potentially raised via a green bond issue), and a levy on downstream external beneficiaries of water quality improvement (e.g. water supply utilities and ports that benefit from sediment reductions), to provide capital for a water quality credit investment fund. This credit investment fund provides a pool of capital that can cover a substantial proportion of the up-front costs arising from credit generation via land use change and land remediation. Landholders and project proponents are invited to tender for offset supply, choosing offset actions from a list appropriate to their catchment and location⁹². With a substantial proportion of major upfront costs already covered by the credit investment fund, the financial risk to suppliers is considerably reduced; supplier participation should increase as a result. On-ground pollution abatement actions are validated through methodologies approved by the Environmental Regulator.

Validated stacked and/or bundled credits from those actions accrue in credit ledgers. Some credit generation actions (e.g. improvement in fertiliser management in sugarcane) will generate only a single type of water quality credit (nitrogen credits in this example), others (e.g. land use change to a constructed treatment wetland) will produce several types of credit simultaneously (nitrogen, carbon and possibly also sediment credits). The proportion of credits retained by the credit investment fund relative to the proportion assigned to the landholder/project proponent will reflect the distribution of financial risk. The majority of credits will accrue to the investment fund where the fund covers the majority of up-front costs. A proportion of credits will generally be assigned to the landholder/proponent as an incentive to continue undertaking the necessary maintenance. Periodic spot checks will be carried out for continuing validation of credits in subsequent years; credit validation will be withdrawn if the necessary maintenance has not been completed.

⁹² Tenders to supply credits will be submitted via a credit suppliers smart phone 'app' – see below for further details.

Validated nitrogen and sediment credits in the credit ledgers will be eligible for purchase as offsets under the PSWQ Offsets Policy. PSWQ Offset Policy buyers will bid for these credits in a standard auction⁹³. The revenue from credit sales will accrue to the credit investment fund and landholders/proponents in proportion to their credit holdings. If investments are focused on relevant catchments (in which an appropriate level of demand exists for the types of credit generated by the actions specified in the tender list) findings from this study suggest that credit buyers' willingness to pay should at least be sufficient to cover the pricing required to elicit credit supply, including recovery of the capital injected by the credit investment fund.

Where credit supply actions generate carbon benefits that can be verified by appropriate carbon credit validation methods (e.g. the well-established methods used Queensland's Land Restoration Fund), carbon credits could be sold separately through the carbon market.

If the list of eligible credit generating actions and locations is appropriately informed by state-of-the-art catchment modelling, actions and locations that produce nitrogen and/or sediment credits for the PSWQ Offset Policy market will also produce additional public good outcomes for downstream beneficiaries (indicated by the thick green arrow in Figure 14.8)⁹⁴. The list of credit generating actions presented to credit suppliers can be tailored accordingly to facilitate supply of marketable credits for the credit ledgers and additional public good benefits relevant to the catchment concerned. For example, sediment credits for PSWQ Offset Policy buyers, or for SPP urban stormwater offsetting, will typically also deliver downstream benefits to external beneficiaries such as water supply companies (via reduced drinking water treatment costs) and ports (through reduced dredging costs). This provides the justification for including a levy on these beneficiaries as an additional funding stream for the catchment investment fund. Other positive externalities – such as a reduction in the rate of rise and extent of flooding – will benefit multiple downstream beneficiaries (households, industry, infrastructure providers and insurers). Provision of these public goods further justifies the case for mobilising public funds to initiate the credit investment fund. Broader public good benefits would also accrue from the economic expansion that water quality credit trading facilitates whilst maintaining Reef water quality, and from the additional direct and indirect job opportunities created by the in-catchment land-based investments undertaken to produce the credits (as outlined in Chapter 13).

A major advantage of adopting the market configuration shown in Figure 14.8 is that the credit investment fund would provide the capital investment injection required to initiate water quality trading in appropriate catchments (where adequate demand exists for the relevant type(s) of water quality credits), and at sufficient scale to ensure that adequate quantities of credits would be supplied to the market. This would help overcome PSWQ Offset Policy buyers' concerns regarding inadequate credit supply – which might otherwise severely restrict credit demand. The fee-in-lieu offset funds (particularly SPP urban stormwater offsets, once population growth rate along the Reef coast increases again) would generate a 'base load' demand for sediment and nitrogen credits in catchments where urban populations are expanding. The credit

⁹³ Offers to buy credits will be submitted via a credit buyers smart phone 'app' – see below for further details.

⁹⁴ The most relevant set of public good benefits for a particular catchment can be determined by state-of-the-art catchment modelling (e.g. the integrated catchment modelling being developed in ARI's Catchment Resilience Project <https://news.griffith.edu.au/2019/10/23/launch-of-1m-griffith-led-project-to-address-impacts-on-catchments/>).

investment fund could then engage more heavily to facilitate water quality credit trading under the PSWQ Offset Policy in catchments where the need for sewage treatment plant upgrades or expansions would otherwise act as a handbrake on economic development.

Active participation from government to create the market configuration of Figure 14.8 would produce the 'hybrid institutional structure' that Tabaichount et al. (2019) consider to be a necessary condition for effective water quality credit markets. The key policy components are already in place (as indicated by the dotted boxes in Figure 14.8): the PSWQ Offsets Policy, the Reef Trust Offsets Policy and SPP urban stormwater offsets. The innovation proposed is to combine the funding streams from these policies, together with a levy on downstream beneficiaries and additional state government funding (that could potentially be recouped via revenues from PSWQ Offset Policy credit sales), into a catchment investment fund with associated ledgers of water quality credits. This water quality credit market configuration would deliver private- and public-good benefits in combination. Credit revenues would only be received for supply of validated credits, i.e. payments are output-based. Spatial targeting of credit supply via the location-specific list of appropriate credit supply actions, together with output-based payments for credits generated, should significantly increase economic efficiency compared with previous 'input-based' subsidy mechanisms. Credit market action also aligns the motivations of credit suppliers (to generate adequate revenues from credit sales) with those of credit buyers (to secure long-term credit supply at reasonable cost), since only validated credits would continue to generate revenue payments. This would act to lock in credit supply from appropriately maintained land management and land use change / land regeneration actions over the longer term and thus help to stabilise demand and supply balance in the market.



Figure 14.9: Technological innovations to support the water quality credit market proposed in Figure 14.8 by reducing uncertainty and information asymmetry. (Graphic by Andrew Buckwell).

14.3.2 Technological innovations to reduce transaction cost and information asymmetry

The technological innovations illustrated in Figure 14.9 would help to reduce transaction cost and information asymmetry and thereby facilitate market participation. Figure 14.9 illustrates typical water quality stressors (DIN losses from cane land (A), sediment erosion from hillslopes (C) and gullies (D), point source emissions (H), and dredging (I)), and potential sources of water quality credits (improvement in fertiliser management practice in cane (B) and stocking density reductions in grazing (E), gully remediation and constructed/embellished wetlands (E & G)). A distributed network of real-time water quality sensors (e.g. monitoring flow, pH, suspended sediment and nitrogen concentration) is also shown ('wifi' symbols distributed through the catchment). The information provided by this distributed water quality sensor network, together with state-of-the-art modelling of the predicted effectiveness of in-catchment actions to reduce pollutant loads, and smart phone 'apps' that simplify credit suppliers' and credit buyers' interactions with the credit investment fund and the credit ledgers, will help to facilitate water quality credit trading as follows:

- As explained in the preceding section, transaction costs are frequently cited as a reason for low participation rates in water quality trading programs (Ribaudo and Gottlieb 2011; Coggan et al. 2014; Motallebi et al. 2017). Smart phone 'apps' can be used to reduce transaction time and cost by simplifying credit suppliers' and credit buyers' interactions with the credit investment fund and the credit ledgers, respectively.
- The credit suppliers' 'app'⁹⁵ would:
 - Ask the credit supplier to identify the landholding on which credit generation actions would be implemented – using a 'Google Maps'-type interface.
 - Present the credit supplier with a list of permitted credit generation actions, appropriate to their catchment and location
 - Ask the credit supplier to select their preferred credit generation action(s) from that list (e.g. reduce fertiliser applications on specified cane paddocks by a specified amount from a given initial application rate; convert a specified area of land to a constructed treatment wetland). Locations for each credit generation action would be specified via the 'Google Maps'-style interface.
 - For each credit generation action selected, provide the supplier with the type(s) and number(s) of credits (at End-of-Catchment) that would be generated. (At this point the 'app' would link to a database holding results from state-of-the-art modelling of the predicted effectiveness of in-catchment actions to reduce pollutant loads.)
 - Ask the supplier to confirm which credit generation action(s) they wish to tender for.
 - Ask the supplier to nominate their tender price for the specified actions
 - Evaluate whether the tender price is attractive, and, if so, present a funding and credit-split proposal to the supplier via the app. For land use change-based actions, the majority of up-front capital would be provided by the credit investment fund, and the fund would retain the majority of the credits,

⁹⁵ Much of the functionality required for the suppliers' app is already resident in the GIS-driven interface of P2R Projector, and in the smart phone app that was prototyped in NESP Project 3.1.8 for use by cane farmers to estimate the changes in yield that would follow from reductions in fertiliser applications.

commensurate with the fund carrying most of the financial risk. Land management practice-based actions would typically require less up-front capital, so the credit supplier would carry a higher proportion of the financial risk. For these types of actions, a higher proportion of the credits would be retained by the landholder.

- Ask the supplier to confirm their agreement to implement the specified credit generation action(s) under the proposed funding and credit-share arrangements.
- The credit buyers' 'app' would:
 - Ask the credit buyer to provide the registration number of the Environmental Authority (EA) against which they wish to purchase credits under the PSWQ Offsets Policy.
 - Link the EA registration number to a discharge location and receiving water.
 - Ask the credit buyer how many credits of which type(s) they wish to buy.
 - Knowing the catchment and receiving water to which the buyer's EA applies and the number of credits (at that receiving water) that the buyer wishes to purchase, the app will present the buyer with a list of available validated credits from the relevant credit ledger(s). For each source of credits, the list will inform the buyer of the quantity of credits available (automatically converted into credit units (e.g. kg N) for the receiving water specified in the buyer's EA), and the mechanism that generated the credits (e.g. fertiliser practice improvement in cane, constructed treatment wetland, bioreactor, bioavailable nitrogen reduction from sediment reduction via gully remediation)
 - Having viewed the quantity and type of credits available for purchase, the buyer can construct a bid to purchase the quantity of credits they require from credit sources that best match their appetite for risk⁹⁶.
 - The buyer submits a bid to buy their chosen mix of credit type(s) and source(s) to the credit ledger via the app.
 - The app would process the buyer's bid, checking whether the submitted bid is adequate to cover the costs incurred in credit supply and ledger management. If they buyer's bid is appropriate, the app would ask the credit buyer to confirm their bid. Once confirmation has been received the app would approve the purchase and – once payment has been transferred – assign the credits to the buyer's EA.
- The preceding section also explained how asymmetric information and uncertainty regarding the efficacy of credit generation actions (particularly those for which on-going maintenance is required to ensure continuing delivery of credits) can lead a risk averse Environmental Regulator to increase transaction cost (by requiring multiple interactions with landholders/proponents during project planning, implementation, and verification), and inadvertently discourage market participation by setting overly cautious delivery ratios and environmental equivalency ratios in the PSWQ Offsets Policy. These issues can be addressed through technological innovation as follows:

⁹⁶ Recall that the credit buyer remains liable for environmental impacts on the receiving water after they have purchased validated credits to offset emission exceedances against their EA.

- The Environmental Regulator should be fully engaged in the development of the modelling tools that predict the quantity of credits that are generated by specific in-catchment actions, and how the quantity of credits generated can vary from year to year depending on weather conditions. By engaging the Regulator in the development of these tools, the Regulator effectively signs off on the validity of the tool for predicting credit generation and therefore no longer requires separate interactions with landholders/proponents to verify the credit generation potential of a particular credit supply proposal.
- Similarly, the Environmental Regulator should be fully engaged with the methods that will be used to validate credit supply, once a credit supply project has been implemented. By signing off on the validation methods, the Regulator no longer needs to validate each individual credit supply arrangement, but instead relies on independent certified verifiers implementing the agreed validation methods.
- Data supplied from the distributed water quality sensor network (Figure 14.9) can operate in combination with state-of-the-art catchment modelling to assist in addressing the environmental integrity concerns of the Environmental Regulator. Catchment modelling will predict expected nutrient and sediment concentrations at sensor locations in the catchment, with contracted water quality credit generation actions in place. Water quality monitoring from the sensor network will identify locations with significant discrepancies between predicted and observed water quality. Random post-implementation spot checks of compliance can then be targeted at credit generation projects in the vicinity of the discrepancy. This will enable limited compliance resource to be targeted effectively. It will also provide a useful input to on-going development of the in-catchment pollutant modelling. If compliance spot checks find that everything is in order this would suggest that additional modelling effort could usefully be directed towards the credit generation mechanisms prevalent in that section of the catchment – or to underlying aspects of catchment modelling (e.g. groundwater linkages) in that area.

For the reasons explained, active government participation in the institutional arrangements shown in Figure 14.8, and the technological innovations of Figure 14.9, are recommended for implementing water quality credit trading along the Reef coastline to facilitate economic expansion whilst also maintaining water quality outcomes for the Great Barrier Reef.

14.3.3 Operation alongside GreenCollar's Reef Credits

The environmental consultancy GreenCollar operate a commercial market in Reef Credits (<https://greencollar.com.au/reef-credits/> and <https://www.reefcredit.org>). Reef Credits represent validated reductions in DIN, sediment, or pesticide loads to the Reef lagoon, generated via in-catchment improvements in land management practice (e.g. reduction in fertiliser applications to cane land), land remediation (e.g. gully repair), or land use change (e.g. conversion of an area of marginally profitable cane land to constructed treatment wetland). Reef Credits differ from the PSWQ Offset Policy credits proposed in Figure 14.8 because Reef Credits are 'composite credits' (in the terminology of Chapter 12). A composite Reef Credit can thus comprise reductions in any one pollutant (DIN, sediment or pesticide), or

reductions in combinations of those pollutants, with the different constituents contributing to a Reef Credit at pre-specified 'exchange rates' or 'conversion factors' (<https://www.reefcredit.org>).

GreenCollar's Reef Credit market and the market in PSWQ Offset Policy credits proposed in Figure 14.8 serve different, but complementary, purposes, as illustrated in Figure 14.10. As explained above, the objective of the credit investment fund-enabled market in point source water quality offsets is to facilitate cost-effective economic development along the Reef coast, with no net decline in Reef water quality. This objective is pursued by ensuring – to the satisfaction of the Environmental Regulator – that credits purchased under the PSWQ Offset Policy are adequate for offsetting increased emissions from point sources, relative to their EA-licenced loads. The broader societal objective pursued by GreenCollar's Reef Credit market is to improve Reef water quality by reducing pollutant loads entering the Reef lagoon. GreenCollar's Reef Credit market pursues this objective by selling Reef Credits to non-emitting buyers (i.e. buyers who do not have catchment water quality liabilities under EAs). Non-emitting buyers may be looking to improve Reef water quality to help secure the longer-term viability of the Reef as an ecological and economic asset (e.g. GBRF, GBR tourism-related businesses, airlines, cruise ship companies etc.), and/or may be purchasing Reef Credits to further corporate social responsibility objectives. When non-emitting buyers purchase Reef Credits this produces a net reduction in pollutant loads to the Reef.

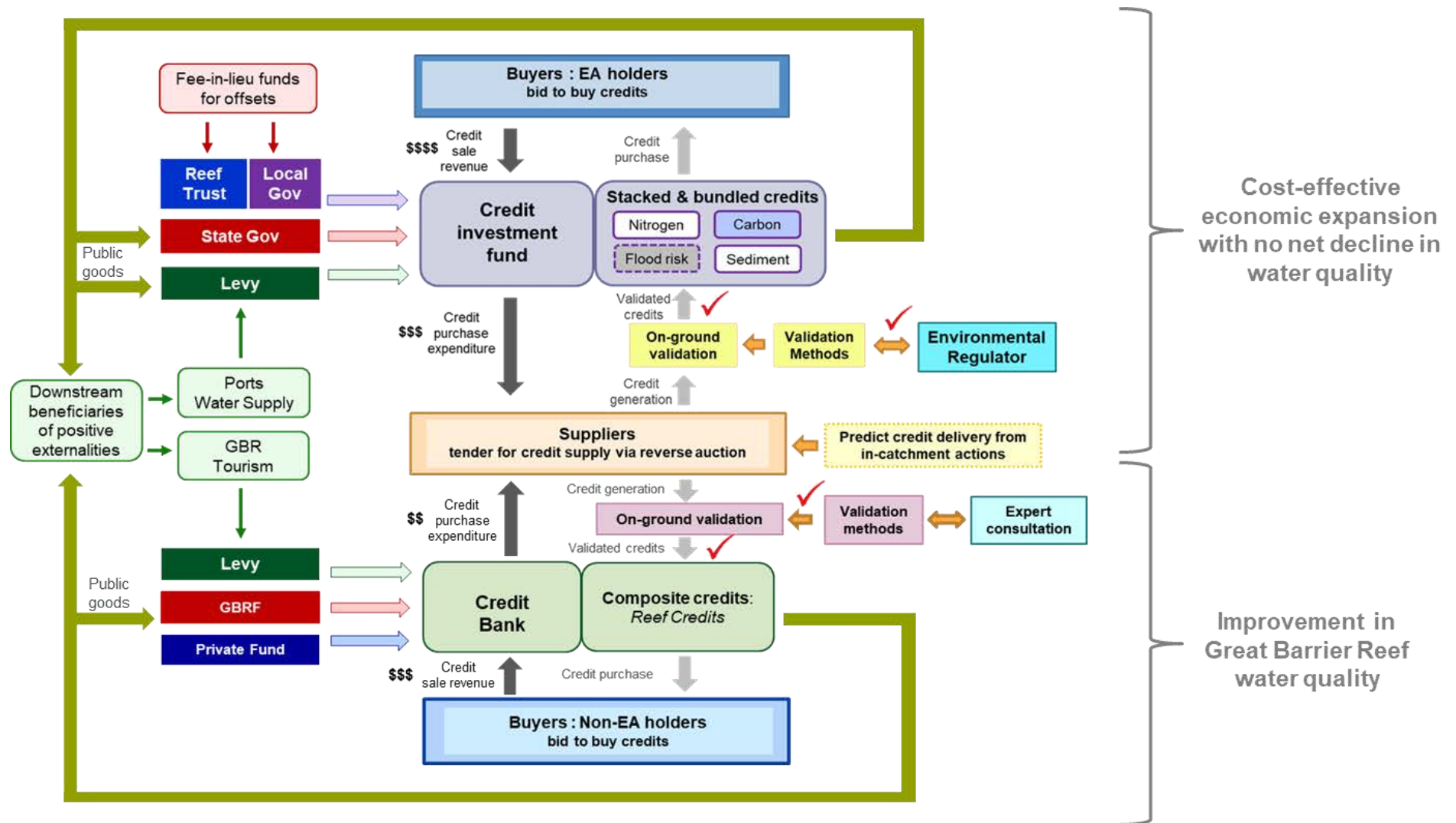


Figure 14.10: Complementary operation of a credit investment fund-based market in PSWQ Offset Policy credits and a market in GreenCollar's Reef Credits.

Figure 14.10 suggests that Reef Credits will be purchased by entities whose objectives align with increasing Reef resilience, and/or are looking to promote corporate social responsibility. Figure 14.10 also suggests that the veracity of credits in both markets would be enhanced by on-going improvements in catchment modelling (the cream box labelled 'predict credit delivery from in-catchment actions' in Figure 14.10) and validation methods (pink and yellow boxes in Figure 14.10). Since both markets will deliver public goods, it is appropriate for government to continue to invest in scientific research that underpins further improvements in catchment modelling and improves quantification of the delivery ratios and environmental equivalency ratios specified in the PSWQ Offsets Policy. Government support for smart phone 'apps' that streamline credit suppliers' and buyers' interactions with the PSWQ Offset Policy offset market would also be defensible because high levels of participation by credit suppliers are central to successful operation of both markets.

14.4 Recommendations for future research

The following recommendations for future research are proposed to facilitate successful implementation of a credit investment fund-enabled, technology enhanced market to support economic development along the Reef coast whilst safeguarding improvements in Reef water quality:

- Continuing scientific investigation of DIN transport via drainage/groundwater pathways
- Continuing scientific investigation and parameterisation of bioavailable nitrogen reductions achieved via abatement of fine sediment
- Continuing investment in improving catchment water quality modelling to incorporate the latest scientific advances in DIN transport and reductions in bioavailable nitrogen from sediment reductions.
- On-going validation of the cost-effectiveness of DIN removal by constructed treatment systems
- On-going validation of the cost-effectiveness of sediment reductions from gully remediation
- Continuing technological developments:
 - Low-cost distributed real-time water quality sensing – particularly for bioavailable nitrogen
 - Smart phone apps for credit suppliers and credit buyers

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