Cost-effectiveness of changing land management practices in sugarcane and grazing to obtain water quality improvements in the Great Barrier Reef: Evaluation and synthesises of existing knowledge

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Summary

This report aims to shape future assessments of cost-effectiveness and profitability of practice change within the Paddock to Reef Program for improved Great Barrier Reef (GBR) outcomes. A framework is provided to ensure that costs are more reconcilable and comparative. This will assist with ensuring the best return on investment is received for future government funding programs designed to address GBR water quality.

The report evaluates and synthesises peer reviewed and published research on cost-effectiveness and profitability of changing land management practices in sugarcane and grazing land production systems for water quality improvements in catchments adjacent to the Great Barrier Reef. Methodological approaches to cost-effectiveness, key determinants of cost, assumptions and limitations in bio-physical modelling, and profitability in the literature have all been examined. The scope of the literature search included all grey and published literature on international, national, and Great Barrier Reef studies on paddock/property, region/catchment, and country levels.

Past research indicates that cost and effectiveness estimates vary across different farming systems, soil types, land types, regions and catchments. Studies are often undertaken for different purposes, time scales within the regions/catchments and audiences. Such inconsistent approaches and reporting of the results mean that the estimates are not always directly comparable. The review has highlighted the importance of timescale, risk and uncertainty, integration with the production system, and links to biophysical models in estimating cost-effectiveness and profitability of land management practice change. Clearly, describing the purpose of the research study and utilising a consistent and transparent methodological approach is critical for comparative purposes and scientific rigour.

Paddock scale studies are often completed using individual farm information or representative (hypothetical) farms that reflect more broadly landholders in the region. Individual farm information is often confidential, thus, difficult to assess if the individual property results reflect the implications of other growers/landholders in the region. As such, a process of getting a number of growers together to determine a representative farm has been more commonly used. Future research, irrespective of any scale, should consider both representative and individual property information as well as consultations of an expert panel. This is particularly important for capturing the large farming area within the GBR catchments and the considerable variation in property characteristics.

Costs considered should be appropriate for the scale at which they are assessed. Common characteristics across sugarcane and grazing that influence costs, production and profitability include property size, soil fertility, land condition, and distance to processing plant or market. These aspects should continue to remain a focus of future work. In addition, transaction costs and transition time to adopt practices and for costs and benefits to accrue, program and administration costs and sediment, nitrogen and pesticide export locations should be consistently captured depending on the scale of assessment and purpose of the research study.

Risk and uncertainty is a challenge in the assessment of any future options. It is more likely to be present in the effectiveness and/or the cost of measures. Perceived risk and uncertainty in returns may increase the perceived and real private costs to producers. As such, it would be desirable to apply a range of cost and effectiveness estimates rather than single estimates. In addition, timescales should be between ten (for sugarcane) and twenty years (for grazing) to appropriately consider realistic periods for practice change. Fifteen years can also be considered, to align to the Great Barrier Reef Foundation time frames. A real discount rate should be used, which incorporates risk and inflation.

Profitability is important and should be included in the evaluation process. The most cost-effective water quality improvement practices may have substantial profitability implications for landholders and may not represent the most profitable option for adoption. Consideration of profitability of management change, in conjunction with social and other key drivers, may also offer insight into landholder adoption rates.

Better integration of economic analysis with biophysical modelling is required to improve consistency of measures (e.g. management practice change) being implemented. Future research should examine

the consequences of simplifying management practice change in bio-economic models. In addition, the limitations of respective biophysical models should continue to be carefully noted in future analysis.

Research should also consider the likelihood of combined pollutant reductions. Management actions that potentially focus on a single pollutant reduction may fail to capture cost-effectiveness reductions in other pollutants. Water quality measures vary across the studies, however, dollars per unit of mass per year appears to be the most common. This measure is also the most robust available.

The research on cost-effectiveness of water quality improvement in the Great Barrier Reef is relatively limited. More research is needed predominantly focusing on potential cost-effectiveness outcomes from particular management practices at all scales. This will assist in managing the impacts of variability and heterogeneity across the GBR catchments by identifying a range of potential costs and benefits for similar practices in different regions.

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Acronyms and Appreviations

ABARES……… Australian Bureau of Agricultural and Resource Economics and Sciences
ACC…………….Abatement Cost Curve
AE………………Adult Equivalent
AEB…………….Annualised Equivalent Benefit
AC………………Abatement cost
APSIM………….Agricultural Production Systems sIMulator
BBN…………….Bayesian Belief Network Model
BMP…………….Best Management Practice
CAPEX…………Capital Expenditure
CE………………Cost-effectiveness
CPV…………….Cumulative Present Value
CSIRO…………Commonwealth Scientific and Industrial Research Organisation
DEHP…………..Department of Environment and Heritage Protection
DEM……………Digital elevation model
DCF…………….Discounted Cash Flow
DIN……………..Dissolved inorganic nitrogen
DR…………… Discount rate
EAA…………….Equivalent Annual Annuity
EMC……………Event Mean Concentration
ERT…………….Ecologically Relevant Targets
FEAT…………...Farm Economic Analysis Tool
GIS……………..Geographical Information System
GBRMPA………Great Barrier Reef Marine Park Authority
GVP…………….Gross Value of production
HC………………Harvesting cost
GCTB…………..Green Cane Trash Blanketing
GM……………..Gross Margin
GRASP………..Grass Production Model
I…………………Investment
IC……………….Investment cost
K………………..Capital
IRR……………..Internal Rate of Return
L………………...Labour
LTCC…………...Long Term Carrying Capacity
MACC………….Marginal Abatement Cost Curve
NPV…………….Net Present Value
NRM……………Natural resource management
OC……………..Opportunity cost
PASTOR………Pasture and livestock systems technical coefficient generator
PSII……….…...Photosystem II herbicides
PV………………Present Value
RE………………Random effect
SEK…………….Swedish Krona
TC………………Transaction cost
TIC…………… Total Investment cost
TSS…………….Total suspended sediment
VC………………Variable cost
WPAC………….Water pollution abatement cost
Y………………..Yield

Abbreviations

g.m-2……………grams per square metre
ha………………..hectare
h…………………hour
kg………………..kilogram
L…………………litre
lbs……………….pound
m………………..million
N………………..nitrogen
P………………..phosphorous
t……………….. tonne
€.........................euro

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1 Introduction

The decline in health of the Great Barrier Reef (GBR) is mainly attributed to climate change impacts and poor water quality caused by pollutants in run-off from agriculture. A range of government - led programs and policies have been implemented in an attempt to improve land management practices to manage run-off water quality flowing to the GBR. To date, these programs and policies have predominantly targeted reductions of specific pollutants, primarily dissolved inorganic nutrient (DIN) derived from sugarcane farming, and fine sediments derived from extensive grazing. Smaller amounts of funding has also been allocated to the grain and banana industries' primary mechanisms. The intent is to reduce pollutant run-off in these industries and to encourage adoption of new and improved land management practices with lower risk of pollutant run-off. The Paddock to Reef (P2R) Water Quality Risk Frameworks have been developed to assign levels of risk to water quality from management practices for all the major agricultural industries in the GBR catchments (Waterhouse et al., 2017). The adoption of the lowest risk practices is most likely to lead to the greatest reduction in pollutants.

Since 2009, there have been a number of programs and policies to achieve adoption. These include cost-share incentives (Reef Rescue, Reef Programme), market-based instruments (Wet Tropics Reverse Tenders), regulation (Great Barrier Reef Amendment Act) and extension (industry and government led extension projects). Although there has been over $1 billion in investment (Thorburn & Wilkinson, 2012), the adoption levels are still insufficient to reduce the pollutant loads to the desired targets. With increased pressure for government to achieve reef outcomes, and the impacts of climate change becoming more frequent, prioritising investments through understanding the cost-effectiveness (CE) of those investments is critical.

Understanding the cost and outcome trade-offs of implementing these programs and policies has been a key question. There are various economic studies that aim to examine the cost of adoption for landholders at a paddock/farm level (Donaghy et al., 2007; Bass et al., 2013; Poggio et al., 2014; North Queensland [NQ] Dry Tropics, 2015), the costs of natural resource management groups looking to implement programs at a catchment level (van Grieken et al., 2013; Rust & Star, 2018; van Grieken et al., 2019), and the costs to achieve targets at the overarching policy level for state governments (Alluvium, 2016). Often these economic studies take costs and mechanisms from the paddock scale, re-apply, and adjust them to a catchment scale. The purpose of these analyses has also varied significantly by informing either private investment, different policy mechanisms, or programs. Reef catchment level studies build on smaller scale studies and modify them to cost a number of options to achieve a target level of pollutant load. Although there is now a growing body of cost information in the GBR catchments (Rolfe et al., 2018), there has not been an overarching coordination process. Therefore, the costs are not necessarily reconcilable, due to the different cost and benefit parameters accounted for.

This report aims to shape future assessments of cost-effectiveness and profitability of practice change for reef outcomes, providing a framework to ensure that costs and benefits are more reconcilable and comparative. This will assist with the prioritisation of future government funding programs to address reef water quality issues for a greater return on investment, along with improving the profitability and sustainability of agricultural industries.

Scope and objectives

The literature review evaluates and synthesises existing knowledge of the cost-effectiveness and profitability of changing land management practices in sugarcane and grazing to obtain water quality improvements. The scope of the literature search included all grey and published literature on international, national, and Great Barrier Reef studies on paddock, region, and country level focusing primarily on:

- methodological approaches in cost-effectiveness analysis and profitability of management practices;
- key determinants of cost including property characteristics;
- assumptions and limitations in bio-physical modelling; and
- water quality improvement indicators.

Within that scope, the objectives of the review were to:

- review and analyse the methodological approaches taken by published studies of cost-effectiveness and profitability at various scales, including a discussion on the appropriateness and limitations/assumptions of the methodologies used;
- highlight the gaps and limitations in the current body of research and make recommendations for prioritising future research on the cost-effectiveness and profitability of land management practices;
- evaluate a process for maintaining consistency and accuracy in future work, appropriate fit, and presentation of economic data to suit other costing exercises/activities.

The report is structured as follows. A brief review of the agricultural industries in the GBR is provided in the next section, followed by a review of methodological approaches (section 3) and the review of cost-effectiveness studies in the reef space (section 4). It will then provide an analysis of profitability considerations (section 5). Conclusion, recommendations and future research are provided in section 6, section 7 and section 8 respectively.

2 Background

The GBR catchments cover 423 134 km^2 of Queensland (QLD) and include a diverse mix of horticulture, sugarcane, and beef in the Burnett Mary through to extensive grazing in the northern catchments of Cape York. There are 35 major river basins within six natural resource management (NRM) regions (Waters et al., 2014) (see Figure 1).

Both tourism and agriculture are significantly contributing to the Queensland economy. Tourism is one of the major industries in the GBR and was estimated to contribute $3.4 billion to the Queensland economy in 2015-16 (Deloitte Access Economics, 2017). The gross value of production (GVP) of Queensland's sugarcane in 2018-19 was estimated to be $1.036 billion with 31.4 million tonnes crushed and with the expected total revenue in raw sugar equivalent to be $1.58 billion (QLD Government, 2018c). In 2017-18, 377,000 ha of sugarcane were harvested and 33.35 million tonnes of cane was crushed resulting in an export value of $1,687 million (ABARES, 2017; ABARES, 2018a).

Cattle grazing activities accounting for 75 per cent of land use in the GBR catchments (GBRMPA, 2014; Waters et al., 2014). The meat and livestock industry in Queensland employs over 40 000 people directly, contributing more than 20 per cent of direct employment within the Australian meat and livestock industry (Ernst & Young, 2018). Queensland's grazing industry is worth an estimated $5.05 billion in GVP (QLD Government, 2018c).

The health of the GBR is under threat from a number of pressures, including agricultural land-based run-off from the sugarcane and grazing industries (Schaffelke et al., 2017). While climate change is recognised as the most serious threat to the reef, run-off presents the largest opportunity for change in the relevant time scale for reef health outcomes (Kroon et al., 2016; Deane et al., 2018; Waterhouse et al., 2017).

The largest pollutant contributors identified by the 2017 Scientific Consensus Statement (Waterhouse et al., 2017) are:

- nutrients, which are mainly coming from 'excess fertiliser applications in the sugar industry' (Rolfe et al., 2018, p. 376) and putting an additional stress on 'many coral species, promote crown-of-thorns starfish population outbreaks with destructive effects on mid-shelf and offshore coral reefs, and promote macroalgal growth' (Waterhouse et al., 2017, p. 10);
- fine sediments, which are predominantly coming from grazing land and increasing water turbidity and reducing 'the light available to seagrass ecosystems and inshore coral reefs' (Waterhouse et al., 2017, p. 10; Fabricius et al., 2014); and

- pesticides, which are mainly from intensive cropping including sugarcane and horticulture (Smith et al., 2012) and which are posing 'a toxicity risk to freshwater ecosystems and some inshore and coastal habitats' (Waterhouse et al., 2017, p. 10).

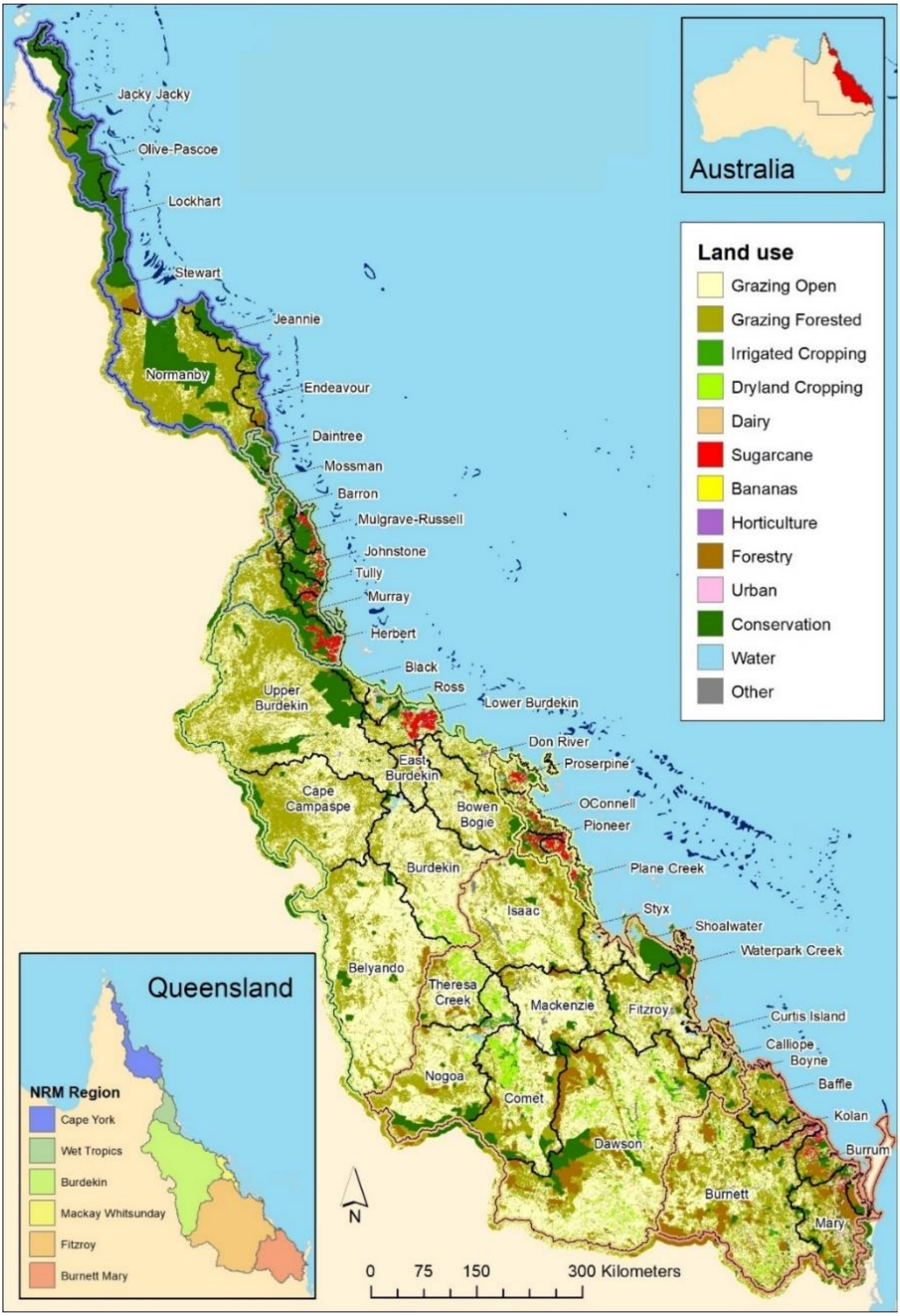

Figure 1: Great Barrier Reef catchments and industries (Source: Star et al., 2018)

To facilitate change and provide a basis for policy decisions there have now been three iterations of Reef Water Quality Improvement Plans. The latest is Reef 2050 Water Quality Improvement Plan (WQIP) 2017-2022 (QLD Government, 2018a). The plan outlines specific water quality targets to be achieved by 2025 at a region level and overarching end-of-catchment loads:

- 60 per cent reduction at the end-of-catchment in anthropogenic inorganic dissolved nitrogen (DIN);
- 25 per cent reduction at the end-of-catchment anthropogenic sediments;
- 20 per cent reduction at the end-of-catchment in anthropogenic particulate nutrient; and
- pesticide target: to protect at least 99 per cent of aquatic species at the end-of-catchments.

To achieve these pollutant reduction targets, land management targets of 90 per cent of land in priority areas under grazing, horticulture, bananas, sugarcane and other broad-acre cropping are managed using farm management practice systems for water quality outcomes (soil, nutrient and pesticides).

This review focuses on the past work completed assessing the cost-effectiveness and profitability of pollutant reductions using the P2R management practice framework to achieve DIN reductions from sugarcane and sediment reductions from the grazing lands.

2.1 Assessing farm management practices through cost-effectiveness analysis

To understand progress towards the pollutant and land management targets from the WQIP, a program called P2R aims to conduct modelling, monitoring and evaluate projects funded under the WQIP. The P2R program consists of Water Quality Risk Frameworks (QLD Government, 2013a) across all agricultural industries that provide the basis for cost-effectiveness studies. These frameworks assess the level of landholder adoption of different management practices and align on-farm management practices to a range of likely risk impacts on pollutant loads and subsequent water quality.

2.1.1 Sugarcane management practices and the basis for cost-effectiveness studies

Past cost-effectiveness studies in sugarcane used the Water Quality Risk Framework based on the ABCD classification of management (e.g. van Grieken et al., 2010a; van Grieken et al., 2013; van Grieken et al., 2014a, b; Poggio et al., 2014; Beher et al., 2016; DEHP, 2016; van Grieken et al., 2019). A comparison of the previous ABCD management practices classifications for sugarcane to current P2R Sugarcane Water Quality Risk Framework is in Table 1. Previous cost-effectiveness studies have been based around implementing these practices on farm or stepping through each classification from a landholder's perspective. This includes aspects such as the purchase of new machinery, changes to application rates, or changes to labour and efficiency (reduction in costs) or productivity (changes to yields).

Table 1: P2R classification of land management practices in sugarcane industry

P2R Water Quality Risk Framework (2013, 2018)	**Lowest risk**	**Low risk**	**Moderate risk**	**High risk**
Reef Plan 2009 'ABCD' Framework	**A**	**B**	**C**	**D**
Description	Innovative practices (expected to result in further water quality benefits, but commercial feasibility is not well understood)	Best practice	Minimum standard	Superseded or outdated

Source: Harvey et al. (2016); QLD Government (2016)

The current Sugarcane Water Quality Risk Framework aligns each management practice to varying degrees of detail based on their risk level and likelihood of achieving improved water quality outcomes (QLD Government, 2018d). The 2018 P2R Water Quality Risk Framework for sugarcane activities comprises four management sections, which align to the different pathways for pollutants (see Table 2):

- soil management;
- nutrient management;
- pesticide management; and
- irrigation management.

There are four main sugarcane-growing regions in the GBR catchment: the Wet Tropics, Burdekin, Mackay Whitsunday and Burnett Mary. There are differences between the regions due to growing conditions which are reflected in each of the regional frameworks (Shaw & Silburn, 2016). For example, Green Cane Trash Blanketing (GCTB) and no irrigation are very common in the Wet Tropics whereas crop irrigation and burning the crop prior to harvest are more common in the Burdekin.

The management practice framework weights each practice and each section based on their risk level and likelihood of achieving improved water quality outcomes (Table 2). Often the selection of sugarcane practices for cost-effectiveness analysis of different pollutant loads are based on consideration of the weightings of their importance for improved water quality outcomes (see Alluvium, 2016; van Grieken et al., 2014b). For example, Alluvium (2016) explored a target of 90% of landholders adopting nutrient management practices.

Table 2: Sugarcane management practices and weighting of significance to classification of risk impacts on pollutant loads and subsequent water quality

Soil Management	Weighting (%)
Crop residue cover	40%
Controlled machinery traffic	20%
Land management during sugarcane fallow	20%
Preparing land for planting	20%
Nutrient Management	
Matching nitrogen supply to crop nitrogen requirements	70%
Matching Phosphorus (P) supply to crop P requirements	15%
Application of mill mud or mud/ash	15%
Pesticide Management	
Use of residual herbicides in ratoons	30%
Targeting herbicide application	30%
Timing of application	20%
Pesticide selection	10%
Managing Canegrub	10%
Irrigation Management	
Calculating the timing of irrigation	20%
Calculating the volume of Irrigation to apply	35%
Minimising irrigation losses	20%
Irrigation tailwater capture and re-use	25%

2.1.2 Grazing management practices and the basis for cost-effectiveness studies

Similarly to the Sugarcane Framework, the Grazing Water Quality Risk Framework categorises a range of practices and identifies them as lowest, low to moderate, moderate, and high risk to water quality improvements (QLD Government, 2018e). The framework weighting (Table 3) aligns practices to either management area (i.e. weaner management) or erosion process (hillslope). The 2018 update added two new sections 'Managing the breeder herd' and 'Weaner management'.

The framework weights and scores each practice under each section to demonstrate the significance of risk of impact on water quality. It is this score that is used to classify landholders into adoption of high risk to low risk management.

The 2018 P2R Water Quality Risk Framework for grazing groups each erosion process under the following sections:

- hillslope;
- gully; and
- streambank.

And herd management areas of:

- managing the breeder herd; and
- weaner management.

Table 3: 2018 Grazing management practices classifications

Hillslope erosion	
Realistic long term carrying capacity for the property	10%
Realistic seasonal stocking rate for the property	35%
Monitored ground cover thresholds	30%
Land condition assessments	10%
Tailored management to encourage recovery of C or D condition land	10%
Property mapping and inventory of natural resources	5%
Streambank erosion	
Managed grazing pressure on frontage country and wetlands	100%
Managed grazing pressure on frontage country and wetlands to maintain or improve condition	100%
Gully Erosion	
Remedial actions undertaken to facilitate gully recovery	40%
Managing risk of erosion with linear features	30%
Hillslope erosion assessment	30%
Managing the breeder herd	
Appropriate nutritional management of heifers from weaning	10%
Segregated heifers from the main breeder herd	10%
Managing breeder body condition with nutrition to maintain fertility	35%
Weaning rate to females joined used as key indicator of performance	15%
Specific criteria used for culling and replacing bulls and heifers	5%
Recognise, prevent and manage fertility diseases	15%
Nutritional deficiencies can affect performance and health condition	10%
Weaner management	
Appropriate management and separation for weaning	30%
Health management strategies implemented during weaning	30%
Breeder management assessment	40%

Previously practices were ranked and termed 'A, B, C, D' as they were linked to land condition outcomes defined under the Grazing Land Management framework (Chilcott et al., 2005) (Table 4). As such, the existing cost-effectiveness studies mainly assessed management practices based on the ABCD land condition transition, rather than framework practices independently, (e.g. Star et al., 2013; East et al., 2010b; Star et al., 2009; Timms et al., 2012) with components of the two new sections of herd management implicit in the analysis (Star & Donaghy, 2009, Star el al., 2013). Ultimately, targeting the framework practices independently is more realistic and researchers can review land condition transition separately.

Table 4: P2R classification of management practices in the grazing industry

Water Quality Risk classification	**Low risk**	**Moderate risk**	**Moderate-High risk**	**High risk**
Resource condition objective	Practices highly likely to maintain land in good (A) condition and/or improve land in lesser condition	Practices are likely to maintain land in good or fair condition (A/B) and/or improve land in lesser condition	Practices are likely to degrade some land to poor (C) condition or very poor (D) condition	Practices are highly likely to degrade land to poor (C) or very poor (D) condition
Previous 'ABCD' nomenclature	**A**	**B**	**C**	**D**

3 Methodological approaches

This section provides an overview of cost-effectiveness method, followed by a review of the approaches that have been used in both the sugarcane and grazing industries. Following this, cost-effectiveness studies in the reef space and the concept of profitability will be discussed.

3.1 Cost – effectiveness analysis (CEA)

Environmental and ecological consequences of land management practices are difficult to predict due to the complexity of interaction between economic, agronomic, and hydrologic systems and the stochastic nature of a variety of factors such as soil, climate, and topographic conditions. Concerns for 'minimizing the economic burden of pollution control in agriculture makes cost-effectiveness an important consideration in the designation of best management practices (BMPs) and the development and evaluation of farm plans for meeting water quality goals' (McSweeny & Shortle, 1990, p. 95).

The cost-effectiveness approach is an applied assessment technique designed to provide 'a ranking of alternative measures on the basis of their costs and effectiveness for achieving' the environmental objectives/outcomes (Balana et al., 2011, p. 1121). Cost-effectiveness analysis (CEA) is simpler than Cost-Benefit analysis (CBA) because 'it avoids all the problems of measuring and valuing benefits' (Stiglitz, 2000, p. 294).

The main steps of conducting CEA (Balana et al., 2011; Martin-Ortega & Balana, 2012) are:

(1) exogenously determining the environmental goal to be met, evaluating potential impact/ pressures and risks, and identifying 'the mitigation options and initial assessments of measures' (Balana et al., 2011, p. 1023);
(2) identifying alternative measures to achieve the stated goals/objectives;
(3) assessing the potential effectiveness of measures such as 'defining base/reference scenario, the scale of analysis, choice of estimation methods' (Balana et al., 2011, p.1023);
(4) estimating the costs of implementing the measures (e.g. identifying data sources, aggregation level, methods of estimating costs); and
(5) assessing measures of cost-effectiveness (e.g. 'alternative options or combinations of options are assessed based on their costs per unit environmental outcome') (Martin-Ortega & Balana, 2012, p. 16).

Within the CEA, costs are usually presented as the direct financial or economic costs (e.g. foregone production) 'of implementing a proposed measure, with effectiveness measured in terms of some physical measure of environmental outcome' (Postle et al., 2004, p. 8). As such, analytical tools like cost-effectiveness can help to identify 'the least cost set of measures to reach multidimensional environmental improvements' in these environmental types of studies (Rolfe et al., 2018, p. 373; Pannell et al., 2012).

Assessment of improved environmental management is a challenge especially when multiple benefits are involved (e.g. benefits to ecosystem services, water quality, and marine areas) (Rolfe et al., 2018). The treasury guide on CBA suggests that CEA should be used as a supplementary approach to CBA in rare instances where it is difficult or impossible to monetise the main benefits. Inability to assess major benefits and the net impact on social welfare recognised as the main disadvantages of CEA. It is conceivable that 'the preferred option in a CEA could result in a net cost rather than a net benefit to society' (NSW Government, 2017, p. 64). Cost-effectiveness approach cannot be used to compare programs/projects with different outcomes that are directly incomparable (NSW Government, 2017) and may deal with issues such as the choice of discount rate and quantification of the effectiveness mainly in cases where 'measures have more than one target objective, and perform differently for these multiple objectives' (Institute for Environmental Studies, 2019, p. 5).

Although there will never be 'complete precision, especially in these hard-to-quantify areas, judgments will be made weighting various considerations, and quantifications can be a helpful step in resolving the complicated trade-offs that have to be faced' (Stiglitz, 2000, p. 295).

CEA has been applied widely around the world at different scale levels including farm/enterprise level (Bartolini et al., 2007, Italy; Fezzi et al., 2008, United Kingdom [UK]; Poggio et al., 2014, Australia; McSweeny & Shortle, 1990, United States of America [USA], river basin/catchment (Lacroix et al., 2005, France; van Grieken et al., 2013, Folkers et al., 2014, Australia), national (Brady, 2003, Sweden) and international (Elofsson, 2003, 2010, Baltic Sea; Froschl et al., 2008) levels. Measures used in cost-effectiveness studies range from a single measure (Yang & Weersink, 2004) to multiple/combinations of measures or alternative policies (Lacroix et al., 2005; van Grieken et al., 2013). A few cost-effectiveness studies entirely focus on direct measures (e.g. actions required to meet specific objectives) while others focus on policy instruments such as mechanisms to implement the measures (Schou et al., 2000) or both (Sorensen et al., 2006).

3.1.1 Measuring cost components

There are a number of critical cost components to be considered in fully assessing the cost-effectiveness of the investment. One way to consider costs is who pays for the cost, and timing of costs. The way these costs are captured in some ways depend on who is seeking to understand the trade-offs. For example, a farmer is concerned about the cost of capital, time and maintenance whereas natural resource management groups or government groups are mainly concerned about the costs of implementing the programs, administration and extension costs.

The simplest forms of private cost calculation reviewed are commonly focusing on-farm profits or private operating costs and on-farm production models, which concentrate on partial economic analysis. Changes in Gross Margin (GM), for example, can be calculated 'by determining the change in variable costs and production implications, holding most factors such as capital, prices and fixed costs constant' (Rolfe & Windle, 2016a, p. 7) while bio-economic models produce private costs focusing on changes in net farm profit (van Grieken et al., 2014 a). The total cost of change in management practices, however, may involve an additional indirect or non-financial private cost. Those costs may include cost of risk and uncertainty 'associated with increased variability or risk of production returns with changes in management', transaction and administration costs arising from finding and organising new opportunities and public costs (e.g. administration, monitoring and enforcement) (Rolfe & Windle, 2016a, p. 7).

Rolfe and Windle (2016a) identified a number of challenges when estimating private costs:

- costs of management practice changes increase with an increase in the quantity of desired changes;
- additional risk, transaction and administration costs increase the perceived and/or real private costs of producers; and
- perceived risk and/or uncertainty in returns may increase private costs.

Farm/paddock scale costs require consideration of management actions and subsequent changes to production and profitability but it can be difficult because production functions are typically complex and non-linear (Star et al., 2017, p.1986). One of the biggest issues is often related to the scarcity of information on the production implications of specific farm management practices, particularly in the case of practices that are considered to be low risk from a water quality perspective. Further complications include the heterogeneity of land and water resources, the stochastic impacts of weather variables, varying time lags, and other external forces, such as market conditions, which all influence the costs at farm scale (Star et al., 2017, p.1986).

Cost of capital and scale

Cost of capital is the cost for the required new/second hand machinery or machinery modification, the costs associated with site preparation and installation as well as costs of possible changes to 'infrastructure to implement the new management classification' (Alluvium, 2016, p. 90; Postle et al., 2004). There are two common approaches taken by researchers for calculating the cost of capital. The first approach is modelling a hypothetical farm or representative property system based on the management practice and/ or Water Quality Risk framework (see van Grieken et al., 2010a; van Grieken et al., 2013; Poggio et al., 2014). The second approach is based on actual capital costs paid by a particular grower or group of growers which often involves a capital cost to make the change (Alluvium, 2016; Poggio & Hanks, 2007; Thompson, 2014).

Capital purchases tend to be one off at the start of the project and the scale in which this is allocated can result in large differences in the overall cost-effectiveness. For example, if the capital is allocated to a large property where the pollution reduction occurs over a number of hectares this is more cost - effective than the same capital used on a smaller property. Assessment of capital required varies between the studies. Some researchers varied the capital requirements based on the different property sizes while others assume the same capital and greater economies of scale achievement with an increase in the size of the property (East, 2012; van Grieken et al., 2013; van Grieken et al., 2014a, b; van Grieken et al., 2019).

Another approach has been to consider that to achieve a management change there might be scope for a number of capital requirements depending on the existing machinery of the landholder. Thus, a best case (potentially modifications to existing machinery), most likely (what is the capital purchase that most farmers will have to implement) and what is the worst case or least likely (this may include purchasing equipment that is brand new and the largest scaled version i.e. 11m width implement as opposed to a 6m width) has also been analysed (Alluvium, 2016). Cost of capital also varies substantially depending on the degree of class level shift (e.g. D to C; B to A) (see van Grieken et al., 2010 a, c) and the management methods for the same proposed measurement (e.g. improved irrigation). Capital costs can vary over time with technology improvements and increased demand (i. e GSP units although this is hard to predict and cannot be captured in the analysis directly) (Alluvium, 2016; Poggio et al., 2014).

Cost of maintenance

Cost of maintenance refers to the cost of maintaining project works, machinery and equipment and for technical training. It may vary widely from site to site within the same measurement/scenario (Alluvium, 2016). It may also include accounting for de-silting or re-shaping of drainage ponds or recycle pits. These costs are associated with ensuring that the initial capital is functioning to its correct efficacy across the time period in the analysis.

Operating cost

Operating cost is an additional on-going cost such as raw materials costs, chemical inputs and fuel costs, disposal, soil sampling, water and pumping, and testing (Postle et al., 2004).

Opportunity cost of production

Cost of forgone production can be accounted for in two main ways. First is by assessing the production that would normally take place using bio-economic models which are relevant to the production system (Star et al., 2013). Another is to assume that the production value is reflected in the values of land prices over an investment period (Alluvium, 2016).

Implementation/Transaction cost

Implementation costs are mainly associated with the required institutional changes, learning and gain, information requirements, training, opportunities for technology, and economic incentives needed for a change such as grants, subsidies, and taxes (Watzold & Schwerdtner, 2005). Transaction cost (TC) does not directly arise from the production of a good but 'from the transfer of a good from one agent to another' (Coggan et al., 2015, p. 500; Nilsson & Sundqvist, 2007). Dahlman (1979) defined the TC concept by splitting it into three elements: (1) information gathering costs, (2) contracting costs, and (3) the cost of control.

The information gathering cost is the cost of collecting and understanding information required for appropriate decision making including information on the cost of production, scientific knowledge on natural resources, and, in the case of conflicting goals/aims, information on preference (Watzold & Schwerdtner, 2005; Birner & Wittmer, 2004, Coggan et al., 2015). The information on the cost of production required if the producer would get compensation for the cost of conservation. Decision-making costs may include coordination decision costs and planning, particularly, when a few individuals or groups are involved in making a decision. Such costs will include 'the resources spent on meetings and resolving conflicts as well as costs arising due to delayed decisions' (Watzold & Schwerdtner, 2005, p.331; Birner & Wittmer, 2004).

The private TC of involvement in improved management activities can be financial and nonfinancial. Nonfinancial cost is the cost of time spent on learning about the program and potential practice changes, how to apply, time spent on application and reviewing contracts, and time spent on planning for and implementing the change on the farm in a transaction. Non-financial costs include an additional time spent for learning about 'the activities in the context of the property (additional to time invested before the application was lodged)' as well as time for monitoring and reporting on the change in management practice to funding body (Coggan et al., 2015, p. 500). Financial private TCs include expenses associated with purchasing information, travelling to meetings and engagement of consultants (e.g. consultant fees, fuel, phone call costs), costs involving any paddock trials of practice and costs of any resources required fulfilling contractual obligations (e.g. monitoring and reporting) and cost of labour (Coggan et al., 2015; Coggan et al., 2010).

Regulatory compliance cost

Some of this type of cost could be borne by different parties including government and landholders. These costs can be hard to capture, as there are different costs associated with complying and not complying and the consequences. Landholders bear the costs of implementing the regulation in terms of the paperwork, documentation and changes; these are often direct and indirect (Alluvium, 2016). Governments typically may have different departments, which interact with the landholder to ensure compliance is met. Providing information on compliance, for example, may be a different department to the department enforcing the regulation.

Program and Extension costs

Extension cost refers to an additional cost to the grower, NRM group or government whilst a change is occurring. This includes participation in agronomic, economic, construction or maintenance technical advice or capacity building (Alluvium, 2016). It is often calculated at a property level and then 'extrapolated to a per hectare basis based on the average farm size' (Alluvium, 2016, p. 96) for using estimates at catchment and region levels. Extension effectiveness is difficult to deal with due to significant time lags in practice adoption and achievement of outcomes by the landholder (Alluvium, 2016).

3.1.2 Measuring effectiveness

Similar to cost calculation/estimation approaches, several methods can be used to estimate pollutant loads relevant to different practice changes:

- *direct measurement* - which is difficult because of 'logistics and expense, the difficulties of identifying marginal changes, and the variations in biophysical factors' (Rolfe & Windle, 2016a, p. 6);
- *paddock scale models* - which are used to estimate pollutant loads at paddock scale while most other factors are held constant; and
- *catchment models* - which are required for 'predictions of changes across different types and locations of farming operations, and where it is important to estimate changes in pollutant delivery at end of the catchment (rather than end of farm)' (Rolfe & Windle, 2016a, p. 6).

Effectiveness accounts for the likelihood that not all actions will necessarily take place or take immediate effect. Not all pollutants generated on-farm reach the end of the catchment and the marine environment, with lower transport rates for pollutants generated further upstream in catchments. As well, some sediments are only transmitted through to the marine environment over very long period of time. This means that not all pollutant reductions made on-farm will necessarily directly benefit the marine environment, while other pollutant reductions may only take effect over long time periods.

Pollutant reductions do not occur immediately when a management change is made because of factors such as long crop cycles (e.g. sugarcane) and natural processes (e.g. ground cover improvements in grazing) that delay the time the change is made or the benefits are realised. In some cases, there are significant lags to achieve benefits. Accounting for these lags avoids overstating the benefits of different actions.

This component of the function allows integration of parameters that will impact on achieving the load reductions including landholder participation, time delays to achieve practice change and pollutant reductions, and transmission factors which can limit pollutant changes between end-of-farm and end-of-catchment. The effectiveness element identifies why some actions may not occur immediately or in full, and, hence, captures the reality that not all pollutant reductions will occur immediately.

3.1.3 Paddock Scale Models

One of the most common approaches to assess cost-effectiveness is the use of bio-economic models (combinations of economic and biophysical sub-models). These models were developed to help identify the environmental outputs and the economic impacts of 'achieving pollutant reduction targets and more recently to help identify the costs involved in achieving targets in some regional WQIPs' (Rolfe & Windle, 2016a, p. 44; Fitzroy Basin, 2015). All bio-economic models are 'limited by their baseline assumptions and are restricted in the extent to which heterogeneity can be incorporated' (Rolfe & Windle, 2016a, p. 44).

The cost-effectiveness is generally estimated using bio-economic models that link pollutant generation and delivery models with farm production models. One of the main advantages of bio-economic modelling is that non-linear effects and interactions between key variables in the model can be accounted for and they can report on a range of costs of reductions and subsequent pollutant loads. Those models enable researchers to generate a cross-section of the potential cost-effectiveness of management changes' by running models at different scales and various enterprise changes (Rolfe &

Windle, 2016a, p. 6; van Grieken et al., 2010a; Star et al., 2013). Bio-economic models can be run at a paddock scale and have commonly been linked to APSIM or GRASP for the biophysical underpinning.

Agricultural Production system simulation model (APSIM)

Agricultural uses of land can employ a diversity of management practices including different ways of soil preparation and treatment of crops. The Agricultural Production Systems sIMulator (APSIM) model 'was developed to simulate biophysical processes in agricultural systems, particularly as it relates to the economic and ecological outcomes of management practices in the face of climate risk' and used to investigate opportunities and solutions for climate change adaptation and mitigation, carbon trading issues and food security (APSIM Initiative, 2019).

The APSIM is structured around the plant, soil and management modules which 'include a diverse range of crops, pastures and trees, soil processes including water balance, N and P transformations, soil pH, erosion and a full range of management controls' (APSIM Initiative, 2019). It aims to provide 'accurate predictions of crop production in relation to climate, genotype, soil and management factors while addressing the long-term resource management issues' (APSIM Initiative, 2019). The modelling framework contains a few components:

- a set of biophysical modules that simulate physical and biological processes in a farming system;
- a set of management modules that enable 'the user to specify the intended management rules that characterise the scenario being simulated and that control the simulation' (APSIM Initiative, 2019);
- several modules 'to facilitate data input and output to and from the simulation;
- a simulation engine that drives the simulation process and facilitates communication between the independent modules;
- various user interfaces for model construction, testing, and application;
- various interfaces and association database tools for visualisation and further analysis of output;
- various model development, testing and documentation tools; and
- a web based user and developer support facility that provides documentation, distribution and defect/change request tracking' (APSIM Initiative, 2019, Keating et al., 2003).

The APSIM can only be applied where the appropriate biophysical modules are obtainable (Keating et al., 2003). Capability of the APSIM has led to different scenarios including whole farm modelling (Holzworth et al., 2014). More recently, the focus of APSIM expanded to an assessment of more general N and water efficiency of intensive sugarcane (Biggs et al., 2013) and was applied within catchment scale water quality frameworks (Carroll et al., 2012).

Those models use to 'predict the water quality benefits of improved management systems' and to 'inform pollutant generation in the catchment scale models' (Shaw & Silburn, 2016, p. 2). One of the main advantages of this modelling approach is the ability to investigate the relative benefits of adopting improved management practices at a paddock scale on the end-of-catchment water quality (within the limitations and assumptions of the approach) and allow comparison between different management practices and their relative cost-effectiveness (Shaw & Silburn, 2016, p. 2).

One of the primary initiatives for developing APSIM was 'a perceived need for modelling tools that provided accurate predictions of crop production in relation to climate, genotype, soil and management factors, whilst addressing long-term resource management issues in farming systems' (Keating et al., 2003, p. 268). However, there are some potential issues with sugarcane productivity (predicted yield) modelled by APSIM which warranted a discussion.

Estimates of yield

APSIM crop models are developed and frequently tested 'against experimental trial data and with large systematic gaps often reported between experimental and farmer yields', thus, raising the question of the relevance of simulated yields to commercial yields and its accuracy (Carberry et al., 2009, p.1044). Carberry et al. (2009) collected information on APSIM performance of yield simulation over 700 commercial crops in all cropping regions in Australia including sugarcane, canola, barley, cotton, wheat, maize, chickpea, sorghum, and mung bean over the period from 1992 to 2007. Their results indicated that APSIM 'can predict the performance of commercial crops at a level close to that reported for its performance against experimental yields' but an essential requirement was 'to accurately describe the resources available to the crop being simulated, particularly soil water and nitrogen' (Carberry et al., 2009, p.1044). Sadras et al. (2003) also reported significant improvement in predicted yields when 'field determined soil water properties were used instead of estimates based on soil texture' (Carberry et al., 2009, p.1046).

However, across 14 case studies Carberry et al. (2009) found that APSIM could account for 52-98 per cent of variation measured in field yield. 'The validations demonstrated some tendency to over-predict low yields and under-predict high yields' (Carberry et al., 2009, p.1046; Robertson et al., 2000). Other researchers also observed gaps between farm actual and modelled/experimental yields (Sadras & Angus, 2006; Whish et al., 2007).

It is expected by inference that simulated yields built on small set of experimental data may considerably over predict commercial yield on farm (Carberry et al., 2009). APSIM sugarcane simulated yield reported by Stewart et al. (2006) in the Burdekin Delta was 13 per cent higher than the actual mill yield but similar for all three fertiliser treatments used in the model. Thorburn et al. (2017) also reported some slightly over predicted simulated yields with most noticeable yield differences in the Mulgrave site when investigating N use efficiency in Bundaberg, Mossman, Maryborough, Mulgrave, and Innisfail. Crop at the Mulgrave site was severely lodged by cyclone 'Larry' beyond the extent to being captured in the model. Thorburn et al. (2017) simulated yield 'to increase with increasing N fertilizer applied' but in many years 'yields reached a "plateau" and did not increase with additional N' (p. 7). In the fine textured soils, an increase in yields 'with increasing N above 150 kg ha^{-1} was small' (Thorburn et al., 2017, p. 7).

Reliability of cane yield estimates modelled by APSIM have been discussed by van Grieken et al. (2010a). They used the Nitrogen Replacement Theory developed by the CSIRO for calculating a fertiliser application rate for class *A* and noted that particularly for class *A* practices 'the cane yields modelled by APSIM may not be achievable in reality with the low rate of fertiliser application' (van Grieken et al., 2010a, p. 18).

Grass Production Model (GRASP)

The QLD Government (McKeon et al., 2000) has developed a Grass Production (GRASP) model for Northern Australian pasture-growth and soil-water simulations. The model can be calibrated to demonstrate the effects of land management practice changes based on pasture utilisation rates, and can be configured to provide outputs appropriate for use with the economic and herd simulation model Breedcow and Dynama (Jones et al., 2016; Holmes, 2005). The head per hectare, live weight gain per head and sediment run-off in tonnes are key data that can contribute to economic information from the GRASP model (Star et al., 2012).

GRASP has been designed to model perennial grass growth in Northern Australia. The quality of this data varies across land types, and modelling irrigated, fertilized, or otherwise rehabilitated pastures requires extensive alteration of parameters to ensure accuracy. The model averages parameters across heterogeneous soils, pastures, and trees; this data includes climate data, soil data, and plant data. GRASP simulates the relationships of these variables with livestock management practices at a particular point in a land type. In addition, run-off is modelled based on trial data to determine water available for plant uptake based on trial data research (Robson & Williams, 2006).

Grazing cost-effectiveness studies have seen several million hectares of land type modelled in GRASP, with some considerable effort required for each type. Expert opinion typically identifies geographical location, land type coverage as a per cent of the catchment, inherent productivity, decreasing ground cover trend, and erosion susceptibility as important criteria for selecting land types to model (Star et al., 2012). The model incorporates various types of biophysical data such as land type, Tree Basal Area, soil water, litter, cover levels, rainfall quantity and intensity, and pasture structure (Alluvium, 2016; Star et al., 2013; Star et al., 2012).

However, GRASP has a number of limitations when integrating the outcomes with economic modelling. These centre around the inflexibility of certain elements and the way in which the data is applied. For example, management practices are set from within the model and the model does not reflect a flexible response to business conditions, and cannot directly model real world business flexibility with respect to managing stocking rates. The model also drives herd modelling and pressure rather than those practices affecting the model. Extrapolation of data from the model beyond properties for which the model is programmed must be done with caution given heterogeneity of land types (Jones et al., 2016).

GRASP does not simulate a range of elements that can affect the reliability of results. Environmental attributes like 'run-on or lateral drainage', the 'complete nitrogen and carbon cycle', and 'effects of changing CO_2 concentration' are not simulated (Robson & Williams, 2006). Perhaps more significantly, the model does not simulate the effects of phosphorus on pasture growth and animal production, nor the way in which cattle select pasture for feed (Robson & Williams, 2006). Despite these limitations, pasture growth, when appropriately calibrated, tends to accurately reflect monitoring sites (Jones et al., 2016).

3.1.4 Catchment Scale Models

Catchment models aim to capture both paddock scale details and catchment information such as distance to the coast, interaction of flood plains and alluvial geomorphology. Source Catchments is the catchment model used to assess pollutant reductions in the GBR. The model accounts for a number of biophysical parameters across the catchment, such as slope, water holding capacity, nutrient pathways, soil type, soil erosivity, vegetation, pasture species and bare ground. The model allows the process to identify the sediment loads (in a unit of tonnes) from the different erosion process of hillslope, streambank, and gullies. Nutrients are also accounted for, while pesticides have been accounted for with the focus on five photosystem II (PSII) herbicides.

Although Source Catchments runs at a daily time-step, allowing the user to explore the interactions of climate and management at a range of time-steps, only the average annual catchment loads were required for the Paddock to Reef Report Card reporting purposes over a 28-year period. Models were validated against the six years of loads monitoring data collected at over 50 individual sites (12 end-of-systems and 19 sub-catchment sites) (Turner et al., 2013) and any additional data sets available to validate modelled load estimates (Dougall & Carroll, 2013).

The Source Catchment Model has been used to assess the change in loads for a number of prioritisation processes (Star et al., 2018, Alluvium, 2016). The efficacy of the pollutant reductions for the different management practices has also varied amongst studies (

Table 5) which impacts the cost-effectiveness as it is the denominator in the cost-effectiveness equation. Using Source, however, has allowed an improved understanding of the scope for improvements in targeting investments across the GBR and where the most cost-effective opportunities for investment are located.

Table 5: Efficacy or percentage of pollutant reductions from projected management changes

Catchment	Industry	Management practice change	Erosion process	Pollutant	% pollutant reduced	Source
ALL	Grazing	D to C	Hillslope	TSS	24	***Alluvium (2016)***
ALL		C to B	Hillslope	TSS	39	
ALL		B to A	Hillslope	TSS	54	
ALL		Remediation	Gully	TSS	40	***Alluvium (2016) (averaged across all approaches)***
ALL		Remediation	Streambank	TSS	20	
WT	Sugarcane	D-C	Diffuse	DIN	87	***Alluvium (2016)***
WT		C-B	Diffuse	DIN	54	
BDT		D-C	Diffuse	DIN	52	
BDT		C-B	Diffuse	DIN	76	
BM		D-C	Diffuse	DIN	75	
BM		C-B	Diffuse	DIN	65	
MW		D-C	Diffuse	DIN	65	
MW		C-B	Diffuse	DIN	63	
WT		D-C	Diffuse	TSS	95	***Rolfe et al. (2018)***
WT		C-B	Diffuse	TSS	80	
BDT		D-C	Diffuse	TSS	85	
BDT		C-B	Diffuse	TSS	76	
BM		D-C	Diffuse	TSS	72	
BM		C-B	Diffuse	TSS	62	
MW		D-C	Diffuse	TSS	95	
MW		C-B	Diffuse	TSS	86	

WT=Wet Tropics BDT= Burdekin Dry Tropics BM=Burnett Mary MW= Mackay Whitsundays
(Source: Star, 2018)

4 Review of cost-effectiveness studies in the reef space

A number of economic studies have been completed to better understand the trade-offs for changing management to reduce pollutant run-off in the reef space. These studies have varied based on the policy implementation, funding agency and the question that is posed to better understand the trade-offs. Prior to 2018, economic studies were funded through a number of different programs and there was no co-ordinated program. This resulted in the time frames, cost components, and measurement of effectiveness all varying depending on what and who the analysis was for. This combination has resulted in large variation in the costs per unit of pollutant per tonne due to the methodology and parameters under which the costs were calculated.

The capture of costs of pollutant reductions has also varied between studies with some studies considering paddock scale and others including an end-of-catchment assessment of pollutant change (Star et al., 2018, Rolfe et al., 2018). The scale to which studies have been undertaken, and the vast heterogeneity in the biophysical parameters used for analysis, have resulted in a wide variation of costs and benefits of pollutant reduction actions. This, in turn, limits the confidence of investment decisions based on these costs (Table 6). The lack of consistency in analysis has resulted in an effort to provide future analysis in a more concise and standardised format, particularly when going to a cost-effectiveness measure.

Table 6: Parameters used in economic analyses

Management Practice Framework	Scale	Bearer of costs & benefits	Policy	Time frame	Costs	Benefits	Capture of pollutants	Discount rate
2009 Management Practice framework **Updated in 2018**	-Paddock -Catchment	-Private, -NRM, - State government -Federal Government	-Extension, -Incentives, -Regulation, -Market instruments	-Production cycle, -Pollutant reduction targets	-Capital, -Opportunity, -Maintenance, -Transaction, -Program	-Production benefits, -Adoption of landholders -Pollutant reductions, Marine outcomes	-Paddock, -End-of-catchment	-Real discount rate accounts for inflation, private capital required, full investment

4.1 Parameter selection

4.1.1 Time frames

To reflect these costs and outcomes, often the whole production system has been captured and the area of management change assessed to understand the implications on the production system and overall private costs and benefits, with the time to achieve benefits framed in the context of the production cycle. Ten years has typically been used in the sugarcane industry. This allows the change to be implemented across the farm over a full crop cycle (typically 5 to 6 years) and some additional years following full implementation (van Grieken et al., 2010a; East, 2012; Poggio et al., 2014). In grazing 20 years has typically been used as this is the period described in which one land manager implements decisions (Star & Donaghy, 2010). This time period also reflects the type of investment being made by a farmer (typically machinery and equipment) and a reasonable payback period before technology becomes obsolete or management practices evolve (e.g. after 10 years the practice will likely change). Although, for some studies (e. g. Whitten et al., 2015 & Star et al., 2015) the time period reflected the time to achieve the pollutant reduction targets by 2020.

4.1.2 Discount rate

Discounting for impacts in the future should be applied to both benefits (pollutant reductions) and costs at the same rate. Benefits need to be discounted to reflect that earlier pollutant reductions are preferred over later ones, while costs are discounted to reflect that purchasing power declines over time.

Discount rates were selected based on real (accounted for inflation) discount rates over a 3-5 year period of time (i.e. Star et al., 2011, East, 2012, Poggio & Page, 2010a, b) or to align to other studies completed in the reef space (Alluvium, 2016, Star et al., 2018).

Discount rates are reflected in real terms which means that the level of inflation is taken from the cash rate at the time of the analysis. A number of studies which were completed when there were strong economic conditions have continued to be used although the economy has weakened. Sensitivity analysis was usually conducted with varying discount rates (Donaghy et al., 2007; Star et al., 2013; Alluvium, 2016; Rust & Star, 2018) and has captured some of the risk regarding economic conditions.

In relation to any scale, there is no clear justification of the most appropriate discount rate in the existing literature, and a discount rate of 6 and 7 per cent is usually applied to the cash-flow streams of grazing and cane enterprises respectively (see Table 10, 12, and 13).

4.1.3 Risk and uncertainty

Risk factors within the literature varied in the significance of consideration, usually focused on the risk to enterprise returns. The variability in prices and climates, extreme weather events, and perceived financial outcomes found to be a significant risk for landholders to adopt/continue adopting management practice change at paddock scale. Other risks to enterprise return included management strategy adherence, the success of adoption and efficacy of management practices.

When investing in costly changes in management practices facing risk and uncertainty with regard to productivity, climate and input/output prices relevant to those changes, producers may perceive relatively higher discount rate when making decisions. Their perceived transaction cost and benefits may be higher and vary significantly from the average figures as well as individual Gross Margins (GMs). Cost of capital investment during transition stage 'is likely to be a major constraint to landholders because of considerable uncertainty about actual costs of change, a high degree of heterogeneity in costs across landholders, and the potential for significant costs that are not captured' (van Grieken et al., 2013, p. 164).

Risk has also been explored for landholders participating in programs and project implementation (Star et al., 2019). They considered the risk of the project failing due to an extreme weather event or the risk or landholder time being more than initially assessed. The study found that landholders were more concerned about the risk of the project failing rather than their time. The project highlighted that risk of failure is critical to consider in costing exercise.

At the catchment and region scale, risk and uncertainty in the existing literature are relatively less considered than at the paddock scale. Cyclones and major flood events identified as significant risks to practice adoption and efficacy success (Alluvium, 2016). Financial, political, regulatory and other market risk could be also significant at this level.

At the GBR scale, the literature considered risk in more detail than previous scales. Researchers took note of variability in climate and land type and raised this as a risk to returns for enterprise operators (Beverly et al., 2016; Rolfe & Windle, 2016a; Star et al., 2012; Star et al., 2013). Risk arising from more complex management practices and requirements to manage based on climate variability were raised (Star et al., 2012). Success of adoption had limited consideration with most literature assuming enterprise operators would adopt management practice with the incentives provided, in addition, practice efficacy is not always immediately realized and poses a risk to benefits in the short term (Alluvium, 2016). Alluvium et al. (2016) pointed to the potential risks of future program costs being higher and adoption success lower. Again, Rolfe & Windle (2016a) pointed to water quality auctions as the most effective method of valuing risk, as enterprise operators determine the incentives they require themselves.

Different scenarios (DEHP, 2016) and sensitivity analysis (van Grieken et al., 2010a; Beher et al., 2016; Pallottino et al., 2005; Brouwer & De Blois, 2008; Star et al., 2017) are the most common approaches when dealing with risk and uncertainty.

4.2 Property and paddock level studies

Initially, a number of studies were completed to better understand the cost-effectiveness implications for landholders, the private costs and benefits, and were often explored at paddock or property level (Bass et al., 2013; Poggio et al., 2014; DEHP, 2016; Poggio & Hanks, 2007; Star & Donaghy, 2009; East, 2010a; East, Simpson & Simpson, 2012; Law et al., 2016). The parameters that are used to complete these analyses vary considerably and this influences the overall results.

4.2.1 Whole of System Approach or Specific Practices Approach

Property and paddock scale modelling considered any research that reported on cost-effectiveness of any pollutant (e.g. DIN, PSII) exported from a farm in the GBR catchment. The results are summarised in Table 7.

Table 7: Sugarcane cost – effectiveness studies at the paddock level (QLD, Australia)

Study/Area	Methodological approach	Practises assessed	Transition / Soil type	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Bass et al. (2013) WT region *Sugarcane*	Secondary data for estimating CE	N/A	N/A	5 year program	DIN and Pesticide (PSII) reduction CE of the Reef Rescue grant *Reported results:* ($/kg/year) CE at the end-of-paddock: • N reduction = $141/kg/year • PSII = $667/kg/year
Poggio et al. (2014) Tully, Mackay, Burdekin Delta, Burdekin BRIA *Sugarcane*	Bio-economic modelling (combination of APSIM crop model and HowLeaky pesticide model) GM analysis AEB	ABCD framework Pesticide management Two types of tillage practice (low and intensive)	*Average hypothetical farms* *Transition assessed* - C→B, C→A & B→A are only considered (transition from D to any other class was not assessed) for all four regions *Soil types* - APSIM major soil types under the 6ES	AEB, NPV 10 years DR = 6%	Pesticide reduction at the **end-of-paddock** *Reported results:* $/kg/year pesticide reduction across ABCD class practice change -$4,920/kg/year to +$4,910/kg/year pesticide reduction; average = $780/kg/year
DEHP (2016) WT & Burdekin catchments *Sugarcane & grazing*	Bio-economic modelling	ABCD framework (without specific focus on a specific management practice) Four policy scenarios	*Transition assessed* - D→C, C→B & B→A *Soil types* - no details on soil types	AEB GM MACCs	DIN reduction CE of current extension programs *Reported results:* ($/t/year, $/kg/year) Average cost of DIN abated *From D to C*: BRIA = $14,822/t/year; Delta = $13,426/t/year; Herbert = $20,174/t/year *From C to B:* BRIA = $30,747/t/year; Delta = -$289,428/t/year; Mossman = -$494/t/year; Herbert = -$957/t/year; Russell-Mulgrave = -$1,865/t/year; South Johnson = -$3,457/t/year; Tully = -$206/t/year Regulation scenario: Burdekin = $7-$10/kg of DIN; Herbert = $40/kg of DIN Extension scenario: $50/kg of DIN Incentive scenario: $0/ha abatement t of DIN *Average cost per kg of DIN abated:* BRIA = $27.57/kg; Delta = $11.22/kg; Mossman = $0.44/kg; Herbert = $1.07/kg; Russell-Mulgrave = $0.80/kg; South Johnson = $0.93/kg; Tully = $1.21/kg

A number of studies in sugarcane (see Table 7) and grazing (see Table 8) have focused on one practice or a set or practices pertaining to one of the pollutant process groupings from the Water Quality Risk frameworks, for example, either pesticide reductions, or hillslope erosion and then stepped through the different management practices. These analyses have typically captured the capital costs for making the change, maintenance of machinery and the opportunity cost for foregoing areas of land or potential income to change management (Poggio et al., 2014; Rust et al., 2017; Star et al., 2011). These studies have not captured extension or program costs, or transaction costs.

In sugarcane, few studies explored the simultaneous shift across management practices to investigate a systems change across the whole property (i.e. changing nutrient and pesticide management practices at the same time) (DEHP, 2016; East 2012; Law & Star 2015). These studies allow for improved efficiencies in farm planning, machinery acquisition and learning along with large benefits in yield and increased pollutant reductions. This is often why there are private benefits also associated with making these whole of farm management practice changes. However, in reality it is difficult for a landholder to have sufficient capital or capacity (time and money) to make a whole of system change and practice considerations should be explained in future cost - effectiveness studies.

In grazing, studies have generally focused on components of the Water Quality Risk framework. Farm scale studies detailed in Table 8 report sediment reductions exported from the paddock. These reductions can be larger than those at the catchment or region and GBR scales due to intermediate sediment deposition. That is, while sediment may be exported from the farm scale, the same quantity does not necessarily make it beyond the catchment, and that sediment that is exported from the catchment may not be exported to the GBR.

Table 8: Grazing studies that reports pollutant reductions at the paddock scale

Study	Discount Rate & time	Economic result	Sediment result	Cost-Effectiveness measure	Practice assessed	Method
NQ Dry Tropics (2015)	Not stated, 20 years	-$2,230 NPV	1,600t over 20 years	$4.68/t	Remedial actions for gully recovery, managing risk of erosion associated with linear features.	Case study and NPV
Donaghy et al. (2007)	6%, 20 years	NPV[1] $269,873 – $519,096	607 – 10,496 t over 20 years	-$20 - $445/t opportunity cost over 20 years	Seasonal stocking rate	GRASP and NPV case study
Ash et al. (2001)	10%, 100 years	-$114,000 – $57,000 in cash return per year	548 – 2,291 kg/ha/year	N/A	LTCC, seasonal stocking rates , managing land recovery, and wet season spelling.	Cash flow analysis

1 Expected NPV relative to a 10% pasture utilisation reduction scenario.

The use of opportunity cost of stock exclusion is an important cost consideration in these analyses. Rust & Star (2018) derived opportunity cost of an 18-month stock exclusion from self-reported data and Wilkinson et al. (2015) discussed the impacts of stock exclusion concluding they would likely outweigh any additional profit or productivity from gully remediation. Where data was unavailable from property managers in Rust & Star (2018) the opportunity cost was valued by determining the size of the zone, stocking density, and gross margin per adult equivalent from external data reports. While these methods simplify the calculation of opportunity cost, the business implications are more significant, with areas where stock exclusion is required often being more productive; therefore, risk-averse property managers may be deterred by high potential opportunity cost of stock exclusion (Wilkinson et al., 2015).

4.2.2 Representative farm or individual farm

Paddock scale studies have been completed using both individual farm information or representative farms that reflect more broadly growers or landholders in the region. Individual farm information is often confidential and this presents some challenges. It is also difficult to assess if the individual property results reflect the implications of other growers and landholders in the catchment. Therefore, a number of individual studies and a process of getting a number of growers together to determine a representative farm has been more commonly used.

In sugarcane, paddock scale studies have mostly been undertaken for ‘a representative farm’ across the different sugarcane growing regions (Poggio et al., 2014). Bio-economic modelling approach was used to investigate cost-effectiveness of DIN losses and reduction in pesticide emissions at the end of the paddock, where the pollutant load was captured under APSIM modelling (see Table 7). All studies assessed cost-effectiveness across various ABCD classes of practice management. For the purpose of identifying costs and benefits from practice change, farming operations were differentiated into three farm sizes: small, medium, and large (Table 9). Cost-effectiveness at this scale was modelled for an individual pollutant and the Annual Equivalent Benefit (AEB) reported as calculation of private cost (Poggio et al., 2014; DEHP, 2016) (Table 7).

Table 9: Review of farm size variance and source across sugarcane studies

Catchment	Practice change	Reference	Farm Size (ha)
Wet Tropics	All possible ABCD transitions assessed	van Grieken et al. (2010a)	120ha
	C-B	Smith et al. (2015) WQIP	Small <100 ha, >100 ha medium <250 ha; and large >250 ha
	C-B	Whitten et al. (2015)	Small (<100ha), medium (100ha-200ha) and large (>200ha)
	C-B, C-A, B-A	van Grieken et al. (2014b)	Not actually specified assumed small (<100ha), medium (100ha-200ha) and large (>200ha)
	C-B	Rolfe and Windle (2016b)	Small 150ha, medium 250ha large 930ha
	D-C, C-B	Alluvium (2016)	Average farm size (150ha)
Wet Tropics (Tully)	C-B, C-A, B-A	Poggio et al. (2014)	Small (50ha), medium (150ha) and large (250ha)
Wet Tropics (Tully-Murray)	All possible ABCD transitions assessed	van Grieken et al. (2013)	120ha
	C-B, C-A, B-A	van Grieken et al. (2019)	Small (20ha -80ha), medium (80ha -180ha) and large (180ha -1230ha)
Burdekin	C-B, C-A, B-A	van Grieken et al (2014b)	Not actually specified assumed small (<100ha), medium (100ha-200ha) and large (>200ha)
	C-B	Whitten et al. (2015)	Small (<100ha), medium (100ha-200ha) and large (>200ha)
	C-B	Rolfe and Windle (2016b)	Small 30ha, medium 297ha, large 1059ha
	D-C, C-B	Alluvium (2016)	Average farm size (106ha)
Burdekin Delta & BRIA	All possible *ABCD transitions assessed*	van Grieken et al. (2010a)	Burdekin Delta (120ha), Burdekin BRIA (240ha)
	C-B, C-A, B-A	Poggio et al. (2014)	Small (50ha), medium (150ha) and large (250ha)
	C-B	Smith et al. (2014) WQIP	BRIA maximum up to 3,500ha, average 140ha & median 94ha. Delta is max 2,000ha, average 98ha, & median 83ha
Burdekin Delta	C-B	Poggio and Page (2010a)	120ha
Burdekin BRIA	C-B	Poggio and Page (2010b)	240ha
Burnett Mary	All possible ABCD transitions assessed	van Grieken (2014a)	Small (75ha), medium (125ha) and large (250ha)
Mackay Whitsundays	All possible *ABCD transitions assessed*	van Grieken et al. (2010a)	150ha
	C-B	East et al (2010)	50ha, 150ha and 300ha
	C-B	East (2012)	240ha
	C-B, C-A, B-A	Poggio et al. (2014)	Small (50ha), medium (150ha) and large (250ha)
	C-B	Law and Star (2015)	150ha
	C-B	Rolfe and Windle (2016b)	42ha, 226ha, 490ha
	D-C, C-B	Alluvium (2016)	Average farm size (125ha)
Mackay Whitsundays (Pioneer)	All possible ABCD transitions assessed	van Grieken et al. (2013)	112ha

With large catchments understanding the cost-effectiveness implications on every property is unrealistic, therefore, a combination of both individual site or paddock scale (Rust & Star, 2018) and representative properties have been completed (Star et al., 2014). Land type differences are a critical driver of productivity, enterprise, production system implications, thus, a range of economic modelling using hypothetical properties (Ash et al., 2001; Alluvium, 2016; Star et al, 2018; Donaghy et al., 2007) have been used.

Economic modelling simplifies management practices by necessity, in order to create manageable mathematical functions to maximise profit for the landholder. In the case of Roebeling et al. (2009), Beverley et al. (2016) and Star et al. (2013), these models assigned water quality and productivity benefits to particular management practices. When examining particular management practices, there was more consideration of individual enterprise challenges with Ash et al. (2001), Donaghy et al. (2007), and Star et al. (2012) all modelling complicated stocking rate management practices. Despite this, the grazing strategies modelled in Donaghy et al. (2007) were considered 'simplistic'; this tends to occur when translating enterprise strategies to models, which are less flexible than reality. On the other hand, studies focused on gully management were much simpler economic models, with cash flow analysis being the key consideration along with up front capital investment costs (Rust & Star, 2018; Wilkinson et al., 2015; NQ Dry Tropics, 2015).

The significance of modelling at this scale should not be underestimated, with results relying on the quality of a range of data across significant areas in all literature examined. Data quality and availability was generally a concern across the research at this level. In Rolfe & Windle (2016a) and Wilkinson et al. (2015), researchers used sediment data from a range of sources, and the lack of consistency across those sources in terms of determining sediment export results makes comparison of figures difficult. Star et al. (2013) identifies a key assumption across much of the literature modelling management practice changes, which is that enterprise managers are always profit maximising and have perfect knowledge of the benefits and costs associated with change. Whilst this is often required given the scale of modelling and monitoring, it does not necessarily reflect reality.

Across the farm/paddock, dollars per kg or tonne per year appears to be the most appropriate measure of cost-effectiveness (Donaghy et al., 2007; Bass et al., 2013; Poggio et al., 2014; NQ Dry Tropics, 2015; DEHP, 2016) (see Table 7 and Table 8). This recognises the time investment required to maintain management practices and associated run-off reduction benefits. Sediment reduction, for example, is a long-term benefit, though as previously mentioned, the research does not value more immediate reductions over those in the future. Measures can also be in volume, rather than weight. Where this was undertaken, researchers used a conversion factor to return to a tonne value (Rust & Star, 2018). This indicates that researchers consider dollars per tonne per year the better measure.

4.3 Catchment/region level studies

More recently, a number of catchment level studies have been completed to understand the NRM management costs of implementing changes based on achieving adoption or water quality targets (see Table 10 for sugarcane and Table 11 for grazing). These have drawn on paddock scale costs and benefits and extrapolated them to a catchment scale accounting for further costs such as extension, program, or transaction costs (Alluvium, 2016; van Grieken et al., 2010a; van Grieken et al., 2013). They have also often adjusted the time frame to achieve change by relative to water quality improvements (i.e. time to achieve pollutant reduction targets or time within NRM Water Quality Improvement Plans) (Burnett Mary NRM Group, 2015; Cape York NRM and South Cape York Catchments, 2016; NQ Dry Tropics, 2016).

Most of the cost-effectiveness studies for sugarcane activities were undertaken at catchment/region scale and used bio-economic modelling to investigate cost-effectiveness of pollutant loads and modelled a single pollutant (e.g. DIN) (van Grieken et al., 2013; van Grieken et al., 2014b; Smart et al.,

2016) or a number of pollutants where each pollutant was modelled separately (Folkers et al., 2014; Rolfe & Windle, 2011b; Star et al., 2018) (see Table 10).

Table 10: Sugarcane cost –effectiveness studies at the paddock level (QLD, Australia)

Study/Area	Methodological approach	Practises assessed	Transition / Soil type	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Roebeling et al. (2009) Tully–Murray catchment *Sugarcane & grazing*	EESIP modelling Water pollution abatement cost (WPAC) functions	Current, minimum & zero tillage Bare & legume fallow Six N application rates (from 60 to 210 kg/ha & N replacement Single & split N application methods Current & reduced herbicide application rates Grassed headlands GCTB	*Soil types* - four soil types (no details)	N/A	DIN & TSS reduction *Reported results:* % change in pollutant at cost in AUD • Significant water quality improvement at a negative cost • Max. benefits through a reduction in TSS & DIN of ~20% & 25% respectively through the adoption of win–win practices (e.g. reduced & zero tillage) • Reductions in water pollution beyond 20% (TSS) & 25% (DIN) come at a cost • Reductions in TSS & DIN over 35% & 50% respectively come at a cost up to ~$8.1m/year for a 60% decrease in TSS & up to ~$6.2m/year for an 80% decrease in DIN
van Grieken et al. (2010a) WT, MWS, Burdekin Delta, Burdekin BRIA regions *Sugarcane*	Bio-economic modelling APSIM crop model Productivity analysis (GM) Investment analysis (NPV) Risk & Sensitivity analysis	ABCD framework Two types of fallow: • bare fallow • legume fallow grown for green manure (cowpea – WT, soybean – Burdekin, MWS)	*All ABCD transitions assessed* *Soil types:* WT - sandy loam MWS - cracking clay Burdekin BRIA - medium clay Burdekin Delta - silty clay with light clay subsoil	NPV 5 & 10 years DR = 7%	Pollutant is N/A No estimates of CE No details on emission reduction *Reported results:* NPV of practice (ABCD) change
Rolfe & Windle (2011a) Burdekin region *Sugarcane*	Inputs-based best management practice scorecard Outputs-based auction metrics	Recycle pits Water management Nutrient management Pesticide management		N/A	N & pesticide reduction CE of WQ tender (77 projects in total) *Reported results:* Average cost of N & pesticides reduction ($/kg) Auction metrics approach Average cost $/kg N reduction: Recycle pits= $7.17/kg; Water management = $5.28/kg; Nutrient management = $8.39/kg; Total = $6.27/kg Average cost $/kg pesticide reduction = $1,234.61/kg BMP score Average cost $/kg N reduction: Recycle pits= $41.82/kg; Water management = $7.88/kg; Nutrient management = $7.99/kg; Total = $13.59/kg Average cost $/kg pesticide reduction = $734.78/kg
Rolfe & Windle (2011b) 4 trials MWS – *sugarcane*	Field trials Output-based metrics Calculations	Four water quality tenders	N/A	N/A	N, P, pesticide & sediment reductions *Reported results:* ($/t, $/kg) *CE of sediment reduction:* Burdekin = $89.22/t; MWS = $4.06/t *CE of N reduction*: Burdekin = $4.56/kg; MWS = $1.40/kg

Study/Area	Methodological approach	Practises assessed	Transition / Soil type	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Burdekin – *sugarcane & grazing* BM – *dairy, horticulture*					*CE of P reduction*: MWS = $10.80/kg Substantial variations in CE between the different agricultural sectors The least cost-effective – sugarcane & grazing
van Grieken et al. (2013) Pioneer & Tully-Murray catchments *Sugarcane*	Bio-economic modelling Production analysis (GM, NPV) Hydro-logical analysis (BBN)	ABCD framework Suite of practices (fertiliser, soil & pesticide) with the same management classes (ABCD)	*All ABCD transitions assessed* *Soil types* - multiple soil types that are major soil types modelled by APSIM for each catchment	NPV 5 & 10 years DR = 7%	DIN reduction *Reported results:* NPV of practice (ABCD) change ($/t) <u>Regulations scenario</u> NPV per t of DIN reduction 5 years frame: Tully-Murray = $13,163/t; Pioneer= -$1,204/t 10 years frame: Tully-Murray = $20,880/t; Pioneer= $22,066/t <u>Aspirations scenario</u> NPV per t of DIN reduction 5 years frame: Tully-Murray = −$21,579/t; Pioneer = -$22,106/t 10 years frame: Tully-Murray = -$8,453/t; Pioneer = $11,312/t
Folkers et al. (2014) MWS region *Sugarcane & horticulture*	N/A	Grant program	N/A	7 years grant program	DIN & herbicides reduction *Reported results:* Cost per unit of load reduction ($/t/year, $/kg/year) $9.89m for DIN reduction of 194 t/year $5.99m for herbicide reduction of 916 kg/year or $6,539/kg
van Grieken, Pannell, & Roberts (2014a) BM region *Sugarcane*	Bio-economic modelling FEAT Local expert knowledge GM analysis	ABCD framework (no specific management practices assessed)	*All ABCD transitions assessed* *Two soil types:* • red dermosols • redoxic hydrosol	AEB, NPV + non-financial 10 years DR = 6% For risky *I* DR = 12%	Pollutant is N/A *Reported results:* ABCD practice class change ($/ha/year) Estimated the required level of incentives to adopt improved practices
van Grieken et al. (2014b) WT, MWS, Burdekin Dry Tropics regions *Sugarcane*	Bio-economic modelling APSIM I analysis (NPV, EAA, AEB, DCF) CE (Marginal Abatement Cost Curve (MACC)	N application rate Fallow management N application management Application method Application timing Record keeping & planning Tillage management	*Transition assessed* - A, B, C only; transition from D was not assessed *Soil types:* • Burdekin BRIA – medium clay • Burdekin Delta – silty clay loam/light clay • MWS – heavy clay loam WT (Tully) – silty clay loam/ clay loam	NPV, AEB, MACC 10 years DR = 3%, 6%, 10%	DIN reduction *Reported results:* CE of management change – AEB per unit DIN ($/ha/year); % DIN (kg/ha/year) and AEB (AU$/ha) See various estimates on pp. 41-48
BM Regional Group (no date) BM region	Bio-economic modelling CB analysis	Various sugarcane & grazing activities			DIN & PSII reduction *Reported results:* Cost per unit of load reduction ($/kg)

Study/Area	Methodological approach	Practises assessed	Transition / Soil type	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Sugarcane & grazing					Average cost per unit of load reductions: N (DIN) reduction = $135/kg Pesticide (PSII) reduction = $26,484/kg
Smart et al. (2016) Tully catchment *Sugarcane*	The spatially-specific N-market model GIS digital elevation model (DEM) GM	Different N application rates	*Soil types:* • sandy loam • medium to heavy clay • slowly drained gradational or duplex textured • poorly drained loam	N/A	DIN reduction CE of a tradeable water quality permit scheme *Reported results:* DIN losses at source, GM/Extra GM
Beher et al. (2016) Fitzroy & MWS catchments *Sugarcane & grazing*	Source Catchment model Sensitivity analysis	ABCD framework (no specific management practices assessed)	*All ABCD transitions assessed* *Soil types* – all soil types modelled by the Source Catchment model (no other details)	N/A	Sediment runoff reduction Spatial scales – end-of-catchment, end of sub-catchment & individual projects *Reported results:* ($/t) CE varies from $9/t to $71,000/t (all projects)
van Grieken et al. (2019) Tully–Murray catchment *Sugarcane*	Deterministic bio-economic modelling GM Abatement cost Local expert opinion (cost calculations)	ABCD framework (no specific management practice)	*Transition assessed:* C→B, C→A & B→A *Soil type:* • sandy loam • cracking clay • medium clay • silty clay with light clay subsoil	Net ACCs per farm type Net regional ACCs	DIN losses CE of two incentive payment policy instruments *Reported results:* Cost sharing ratios (CSRs) – (% $/crop cycle) If 100% of the total costs of moving from current to improved practices paid to farmers given current technologies, water quality would be approximately 56% (reduction of DIN). Total regional costs would be ~$ 30m over one cropping cycle of 6 years
Australian Government (2014) WT, MWS, Burdekin, Cape York, Fitzroy, BM regions *Sugarcane & grazing*	Bio-economic modelling	Reef Rescue program	N/A	5 year program	DIN, PSII & sediment reduction *Reported results:* ($/kg/year, $/t/year) *N reduction:* Cape York = $192/kg/year; WT = $47/kg/year; Burdekin = $62/kg/year; MWS = $164/kg/year; Fitzroy = $1,494/kg/year; Burnett Mary = $192/kg/year *DIN reduction:* WT = $142/kg/year; Burdekin = $124/kg/year; MWS = $157/kg/year *PSII reduction:* WT = $16,343/kg/year; Burdekin = $120,770/kg/year; MWS = $19,315/kg/year; Fitzroy = $1,060,650/kg/year; Burnett Mary = 38,153/kg/year
Alluvium (2016) Burdekin, WT MWS, BM regions *Sugarcane & grazing*	Bio-economic modelling Meta modelling AEB Total ACC MACC	WQ Risk framework aligned with ABCD framework (2013) Land management practice change	*Transition assessed* - for set 1 (sugarcane) transition from D→C and C→B are only considered (transition to A was not assessed) for all four regions and the GBR *Soil types* - no details	PV, CPV 10 years DR = 7%	DIN reduction *Reported results:* ($/t/year, $/kg/year) *DIN reduction:* WT – from $4,890/t to $14,500/t; Burdekin – from $12,300/t to $62,500/t ; MWS – from $597/t to $24,700/t; BM - $1,770/t

Study/Area	Methodological approach	Practises assessed	Transition / Soil type	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Bass et al. (2013) WT region *Sugarcane*	Secondary data for calculating CE	Reef Rescue grant	N/A	5 year program	DIN and Pesticide (PSII) reduction *Reported results:* ($/kg/year) DIN reduction = $438/kg/year; PSII reduction = $22,886/kg/year

Cost-effectiveness was mainly assessed in terms of units of pollutant reduction (see Table 10 and 11):

- $/kg (Folkers et al., 2014; Burnett Mary Regional Group, (no date; Australian Government, 2014) or $/t (van Grieken et al., 2013; Alluvium, 2016) of DIN reduced;
- $/kg of N reduction (Rolfe & Windle, 2011a, b; Australian Government, 2014);
- $/kg of pesticides reduction (Rolfe & Windle, 2011a, b; Burnett Mary Regional Group, no date; Australian Government, 2014);
- $/kg of herbicides reduction (Folkers et al., 2014);
- $/kg of P reduction (Rolfe & Windle, 2011b); and
- $/t of sediment reduction (Roebeling et al., 2009; Australian Government, 2014; Wilkinson et al., 2015; Rust & Star, 2018).

Table 11: Catchment and region level grazing cost-effectiveness studies

Study	Discount rate & time	Economic result	Sediment result	CE measure (sediment)	Practice assessed	Method
Rust & Star (2018)	7%, 10 years	$36,244.07 – $118,823.39 PV cost	126 – 822 m^3 per year	$66.3 - $516.23/t/year	Remedial actions for gully recovery (40%), managing risk of erosion associated with linear features (30%)	Case study NPV analysis
Wilkinson et al. (2015)	N/A, 10 – 40 years	$4,500 - $9,000 per km of fence	180 – 290 t/year	$81 - $217/t/year	Remedial actions for gully recovery (40%), managing risk of erosion associated with linear features (30%)	Case study
Australian Government (2014)	Not stated, 5 years	Not stated	Not stated	$106 – $1,343/t/year[1]	Whole of framework	Bio-economic model and program cost
Roebeling et al. (2009)	Not stated	$2.4 million regional agricultural income reduction	60% reduction in TSS (7,200 t)	$333/t/year[2]	Seasonal stocking rate (35%)	SedNet and PASTOR EESIP[3]

1 Varying depending on region (Cape York, Wet Tropics, Burdekin, Mackay Whitsunday, Fitzroy and Burnett Mary)

2 Not reported but calculated from $2.4 million per year to reduce 7,200 t per year (no discounting reported)

3 Environmental Economic Spatial Investment Prioritisation

4.3.1 Program and Extension costs

Cost assessments at the catchment scale incorporate program costs, extension, regulation, implementation, management and opportunity costs, as well as transaction costs based on management practices (Alluvium, 2016; van Grieken et al., 2013; van Grieken et al., 2014a; Star et al., 2017; van Grieken et al., 2019). These studies extrapolate up paddock-scale costs, usually based on the number of farms, or the proportion of a catchment at a particular level of management practice (Alluvium, 2016; Star et al., 2018).

Program costs are associated with the cost to administer programs, government administration costs, the time frames between funding allocations and program implementation, and involve the cost of multiple layers of administration (Star et al., 2017). Calculating extension costs is also more common at catchment/region scale often due to the groups funders and deliverers seeking to understand the cost to achieve their goals. Time lags in both landholder adoption and achieving outcomes, as well as the possibility of diminishing returns to extension expenditure, make these costs difficult to capture (Star et al., 2017, p. 1987). There is limited data collected by extension providers to date and previous policy mechanisms have been based predominately around incentives. Another challenging aspect of extension costing is directly linking the cost of extension to the change in management.

In grazing the majority of modelling conducted at a catchment/region scale derives from aggregating paddock scale data to a catchment/region level (Star et al., 2018, Alluvium 2016). Following the heterogeneity of costs at paddock scale between locations, types of land and climate variations, costing of different sets of management practices adoption at catchment scale is often 'explored with the outcomes of land condition used as the proxy for land management, and many of the paddock scale outcomes extrapolated to catchments' (Star et al., 2017, p. 1985; Star et al., 2015). This is a challenge, as decision makers should exercise caution when extrapolating the modelling, as at the paddock scale there are a number of parameters that may need to be considered. The costs of water quality improvement and the opportunity cost vary substantially across catchments and pollutants (Rolfe et al., 2011; Rolfe & Windle, 2011b) (see Table 10 and 11). In addition, catchment scale modelling typically applies "delivery ratio" (e.g. to sediment and deep drainage DIN) and have losses of constituents instream and in water storages. Thus, there are differences between generation rate (i.e. at edge of paddock) and export rates.

4.3.2 Transaction costs

Similarly, at the catchment scale transaction costs may deter management practice adoption. The literature broadly defines these costs as the costs of obtaining information related to a new management practice (Rolfe & Windle, 2016a, Coggan et al., 2015). While no studies at this scale attempted to value transaction costs, the research did consider them. Wilkinson et al. (2015) suggest that transaction costs reduce the desire for gully management even when presented with potential benefits. Rolfe & Windle (2016a) suggest that transaction costs are difficult to value through traditional grant mechanisms and that water quality auctions are likely to reveal these values by allowing enterprise managers to assign their own values to those costs.

4.4 Overview of reef cost-effectiveness studies

Costs and benefits at the GBR scale capture all previous scale costs and benefits as well as Government implementation cost which is a subject to political risk. Time between program implementation and administration of funds is also important. Measuring costs at this level is a challenge because they are borne by various parties including NRM groups and governments (Star et al., 2017). These costs are not well documented (Rolfe & Windle, 2016a).

Literature at the GBR scale identifies similar challenges to previous scales. Modelling conducted at a GBR scale derives from researchers aggregating paddock and region scale data to a GBR level. Program and administration costs have been included cautiously while transaction costs were not included in the analysis. Costs examined at this scale included on-ground costs such as capital costs, maintenance costs, extension costs, drought-feeding costs (Alluvium, 2016; Beverly et al., 2016; Rust & Star, 2018; Star et al., 2012; Star et al., 2013), regulation costs (Alluvium, 2016), and intangible costs like opportunity cost of stock exclusion (Rust & Star, 2018) and alternate use of funds (real discount rates in Table 12 and Table 13).

Table 12: GBR level sugarcane cost-effectiveness studies

Study	Methodological approach	Practises assessed	Transition / Soil type / Farm size	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Australian Government (2014)	Bio-economic modelling	Reef Rescue program	N/A	5 year program	DIN & PSII reduction *Reported results:* ($/t/year, $/kg/year) $63,000t/year (DIN) $3,500kg/year (PSII)

Study	Methodological approach	Practises assessed	Transition / Soil type / Farm size	Private cost/ Time frame/ DR	Water Quality improvement indicators/ CE measure
Alluvium (2016)	Bio-economic modelling Meta modelling AEB Total ACC MACC	Land management practice change	*Transition assessed* - D→C and C→B *Soil types* - no details	PV, CPV 10 years DR = 7%	DIN reduction *Reported results:* ($/t/year, $/kg/year) $390m/year to reduce DIN loads by 2,650t = $147.4/kg/year
Rolfe et al. (2018)	Index approach Disaggregation approach (CE)	WT (238 projects) Burdekin (168 projects) MW (74 projects) BM (50 projects) Not all included in the analysis	N/A		SS, DIN, PSII reduction Net changes in pollutant for each project *Reported results:* ($/unit of benefit/ year (t, kg)) <u>Disaggregation approach</u> Average cost of SS reduction = $24.96/t/year; Average cost of DIN reduction = $6.89/kg/year; Average cost of PSII = $849/kg/year

Discount rate and time frames are similar to farm and catchment/region scales. Seven per cent discount rate and time frames following implementation between 5 and 10 years in sugarcane and up to 20 years in grazing are considered as reasonable representation (Alluvium, 2016) (see Table 12 and Table 13).

Table 13: GBR level grazing cost-effectiveness studies

Study	Discount Rate & time frame	Economic result	Sediment result	CE measure	Practice assessed	Methodological approach
Rust & Star (2018)	7%, 10 years	$36,244.07 - $118,823.39 NPV	21 – 137 m^3 per year[1]	$652.44/t/year	Remedial actions for gully recovery (40%), managing risk of erosion associated with linear features (30%)	Case study
Alluvium (2016)	7%, 10 years	$7.82 bn Present Value	2,920,000 t/y	$268/t	Changing management practice	GRASP, meta-model, Source Catchments outputs
Beverly et al. (2016)	6%, 20 years	-$16.4 – -$3m NPV cost[2]	63% – 72%[3]	Ha/$/year (area transitioned)	Changing management practice	GRASP and watershed model
Rolfe & Windle (2016a)	6% – 7%, 5 – 20 years	N/A	15-20% reduction	$259/t[4]	Changing management practice	Benchmarking / synthesis review
Australian Government (2014)	Not stated, 5 years	Not stated	Approximately 8% reduction from baseline.	$130/t/year	Whole of framework	Bio-economic model and program cost
Star et al. (2013)	6%, 20 years	$295,416 – $1,213,380 NPV[5]	186 – 258 kg/y/ha[5]	$4/t to $421/t/year	Land condition and stocking rate	Case study, GRASP, NPV, spatial optimisation
Star et al. (2012)	6%, 20 years	-$28.69 – $6.97m NPV	198 – 214,729 t	-$835.08/t - $25,594.46/t	Land condition and stocking rate	GRASP and Breedcow / NPV

1 Using export delivery rate of 1 in 6.

2 Program cost to meet whole region set targets, from ecologically relevant targets (ERT) to reef plan targets (RPT).

3 Contribution of total sediment reduction to ERT and RPT from grazing.

4 Benchmark rate averaged across a range of literature (costs and reductions are not consistent).

5 Results for sediment reduction scenarios deemed most cost-effective.

Currently, the range of costs and benefits accounted for in the cost-effectiveness literature are hard to reconcile as the range of costs and benefits vary. For example, the Rolfe and Windle (2016a) study did account for administration and program costs whereas Rust and Star (2018) did not. However, Rust and Star (2018) did account for maintenance costs (Table 14). The report highlights the components of these studies, and why they have captured the data they have. One key aspect is that all studies have used the Paddock to Reef water quality risk practices as the platform to assess the management changes and subsequent costs and benefits.

Table 14: Different types of costs assessed by cost-effectiveness studies for reef projects

Costs	Alluvium (2016)	Rolfe & Windle (2016a)	Rolfe et al. (2018)	Rust & Star (2018)
Project (capital) costs		X	X	X
Opportunity costs (any losses in production)	X			X
Transaction costs	X			
Maintenance costs				X
Administration and program costs	X	X		
Other landholder costs/investment				
Average Cost / annual tonne Sediment	\$3,415	\$500	\$24.96	\$194
Average Cost / annual kg DIN	\$147	\$268	\$6.89	

Source: (Rolfe, 2018)

Investments can be better prioritised to improve the efficiency and effectiveness of practice change programs when cost-effectiveness information is well understood. The costs of meeting the water quality targets has been shown to be very high; much higher than previously thought in part due to the geographical location and heterogeneity in costs (Alluvium 2016). This is also due in part to considering all program costs and accounting for risk which have previously have not been considered at a whole of catchment scale (Star et al., 2018). As the water quality targets are approached, the costs of additional actions are likely to rise sharply (Rolfe & Harvey 2017). Better prioritisation of investments should take into account the cost-effectiveness of agricultural management options including adoption rates, costs, time lags and climatic influences, as well as risks to the marine environment (Waterhouse et al., 2017, p. 13).

4.5 Benefits from practice change for water quality improvement

As was previously mentioned, CEA avoids all issues associated with measurement and valuation of potential benefits that arise from practice change for water quality improvement but different sets of practices/ scenarios may still produce various benefits to a landholder. Ultimately the benefits to the farmer are mainly related to an improvement in profitability. These farm scale benefits, however, can be grouped into two areas either efficiency mainly related to the reduction in cost of inputs, and productivity improvements such as increased yield. Improved irrigation practices such as well-designed and managed automated furrow systems, for example, may create savings in labour in the long-term while installation of drip or overhead low-pressure irrigation systems may create farm profit and result in lower electricity and water costs due to limited or no surface irrigation runoff.

Similarly to costs, benefits may vary significantly from enterprise to enterprise indicating the heterogeneity of locations, land types and climate conditions (Rolfe & Windle, 2016a). The larger the farm the greater the accumulation of the marginal benefit. Another consideration is when specific practices are assessed (Poggio & Page, 2010 a, b; East, 2010a) rather than whole of property changes the benefits may be not correctly accumulated as only one component of management is explored. For example, efficiency benefits may be achieved from adopting variable rate nutrient application through purchasing a GPS and variable rate controller, however, the improvements to pesticide management from having the GPS may not be captured.

Benefits at the catchment/regional scale are more challenging to quantify because many of those are public benefits, which represent the priorities of the wider community for better protection of the GBR. Benefits have generally been captured as pollutant reductions using the Source model to assess the impact with the change in the level of adoption drive this reduction in pollutants. Depending on when the analysis was completed means that the baseline pollutant load and adoption levels may have changed along with the efficacy attributed to changing the management practice.

Benefits at the GBR scale typically related to particular combinations of management practice on land types. Where land productivity was low there were a number of benefits achieved from improved management practice adoption, however incentives were still required on more profitable land types (Star et al., 2012). There were also potentials for management practice improvement to align with reductions in input prices (Alluvium, 2016) though Beverley et al. (2016) found the adoption of lower risk grazing management practices was associated with much greater costs than sugarcane.

Water quality benefits are a much greater focus and are determined by either assigning a water quality outcome for particular sugarcane/grazing practices (Roebeling et al., 2009; Australian Government, 2014; Wilkinson et al., 2015), or through a regional standard calculation of gully growth (Rust & Star, 2018). In the latter case, reduction in gully growth per year was the assumed benefit once the remediation work was completed (Rust & Star, 2018). Wilkinson et al. (2015) assigned water quality benefits based on international research linking ground cover from particular management practice to a per cent improvement in water quality for that ground cover level. Roebeling et al. (2009) relied on agricultural production simulation models to determine management practice impacts on water quality.

4.6 Review of international studies

Water quality implications, because of agricultural pollutants, is not only specific to sugarcane, grazing, and the GBR, internationally there is increasing pressure to protect coral reefs from climate change, agricultural pollutant run-off and coastal development (Thia-Eng, 1993; Gibson et al., 1998; Meliadou et al., 2012; Tabet & Fanning, 2012). In relation to water quality from agricultural sources, there have been various approaches to changing farm management practices to reduce nonpoint source pollution (Logan, 1993; Ripa et al., 2006; Baumgart-Getz et al., 2012), along with methods for identifying the most cost-effective outcomes for multiple objectives and with limited budgets (Claassen et al., 2008).

4.6.1 International paddock level studies

In the United States, the importance of a broad watershed management program to reduce both point and nonpoint sources of fertilisers and other pollutants is gaining increased recognition at local, state, and federal levels. In agricultural areas, the promotion of best management practices (BMPs) has resulted in widespread acceptance of practices that reduce erosion of nutrient-laden soils (EPA, 1996). Such practices include no-tillage and conservation tillage, vegetated filter strips and grass waterways, lowering of fertiliser application rates, and proper handling of animal manures. The adoption of BMPs in the United States is voluntary although cost-sharing programs are available through federal agencies such as the Farm Services Agency and the Natural Resources Conservation Service. Section 303(d) of the Clean Water Act calls for the implementation of total maximum daily loads into streams and lakes that have low water quality, and the Clean Lakes Program provided assistance in watershed management and improving water quality in lakes prior to 1995 (Lant et al., 2005; Liu et al., 2013). Clearly, there is general recognition of the importance of watershed management in improving water quality, and the erosion of sediments into waterways has been considerably lessened. There are, however, still areas where the implementation of programs has been slow.

Examples of where water quality issues from agricultural run-off have been prominent include Chesapeake Bay, New Hampshire, Long Island Sound, and Iowa. These catchments have participated in a number of water quality improvement programs targeting nutrients predominately from poultry farms in the catchment area but also grains and grazing (Diebel et al., 2009; Maxted et al., 2009; Michaud et al., 2007). Cost-effectiveness for reductions of nitrogen has been based on a number of parameters. The water quality improvements have been to achieve standards (i.e. the standard of 5.0 mg/L for the Delaware River) overall reductions for the system and daily limits (Kauffman 2018). There is an annual EPA Water Quality Analysis Simulation program (https://www.epa.gov/ceam/water-quality-analysis-simulation-program-wasp) which assesses pollutant reductions of farm management practices across agriculture, wastewater, urban storm water and atmospheric deposition.

The key parameters used for these cost-effectiveness studies are based on the contract time frames of 15 years, however, it is acknowledged that streambank revegetation may take up to 60 years. The discount rates used vary for sensitivity testing and are between two per cent and seven per cent providing a range of costs and pollutant reductions. The costs ranged from US$2.64-$24.20 kg/N for agricultural conservation, $18.83-$174 for wastewater reductions, US$165-$639 for air born emissions and $90-$500 for urban and stormwater retrofit (Kauffman, 2018, Wieland et al., 2009, Environmental Protection Agency, 1996).

The European Water Directive Framework (WDF) explicitly integrates economics into its water policy, however, with its application in different countries under different contexts there are variances in how it is captured. The Directive's emphasis is on achieving good status of water bodies and is accompanied by the need to take action to achieve good status because such action would be disproportionately expensive. A number of approaches have been used such as bio-economic modelling (combined agronomic simulation and multi-objective programming models). The effect of agricultural policies on water application efficiency, farmers' revenue and nitrate leaching reduction, and the cost-effectiveness of various nonpoint pollution abatement policy measures have been analysed in Southern Italy with bio-economic models (Semaan et al., 2007). Similarly, farm production models, farm accounting statistics, and GIS-based spatial production structures, have been used to assess nitrate leaching and cost-effectiveness for Danish agriculture (Schou et al., 2000). Brady (2003) has applied a spatially distributed nonlinear programming model to link changes in agricultural production on crop farms and coastal nitrogen loads in southern Sweden. In Spain, the Instrucción de Planificación Hidrológica (IPH) or Water Planning Instruction is the responsible government agency in the implementation of the WDF IPH mandates the use of CEA for the Programme of Measures. Costs are supposed to be expressed as the equivalent annual cost that includes annualised investment, maintenance, and operational costs for implementing a proposed measure or combinations of measures (Balana et al. 2011).

4.6.2 International catchment/region level studies

The scope of national level studies is limited to a specific country for evaluating the relative cost-efficiency of mitigation measures or policy instruments but may have wider salience in similar countries. Using a spatially distributed nonlinear mathematical programming model, Brady (2003), for instance, investigated the relative cost-efficiency of three nitrogen pollution abatement policies in Swedish arable farming. His model linked changes in agricultural production practices on arable farming induced by the three policies to coastal nitrogen load in the Baltic Sea. Such models could not explore the potential efficiency gain from coordinated implementation at regional level.

A case study in the north-east of France examined the cost-effectiveness of six nitrate pollution reduction scenarios (Lacroix et al., 2005). The study explicitly incorporated uncertainty and climate variability in evaluating the environmental and economic impacts of the various scenarios. They concluded that in the short-run none of the scenarios tested could achieve the 50 mg N/L environmental targets. To satisfy this target the production system would have to change which may lead to a huge economic burden on farmers.

At a catchment level these paddock scale costs are used and incorporated into other models.

4.6.3 Assumptions and limitations

A large number of WDF-related CEA studies of agri-environmental measures were based on 'representative' farm types. For instance, most Defra-commissioned CEA studies followed 'stylised' farm systems classification. Such studies may fail to capture the inherent heterogeneity among real-world farms and send wrong signals to the decision making process. The actual variability of real-world farms, thus, needs to be captured if the aim of CEA studies is to support policy. It should be borne in mind, however, that this does not mean modelling each and every farm individually which appears to be practically infeasible. Our suggestion is that to better support decision-making process, CEA studies need to utilize actual farm data instead of 'stylized' farms. In this regard, for instance, Fezzi et al. (2008) assessed four proposed WDF measures using changes in farm gross margins calculated from a dataset of over 2000 farms in England and Wales. Furthermore, heterogeneity could be captured better with alternative modelling approaches such as agent-based modelling to traditional representative farm model.

Most CEA studies concentrate on the effect of a single measure (e.g., effect of a measure on P mitigation) or based on cost calculation that reflect costs which occur at the sectors directly involved in the pollutant-reduction programme (e.g. costs external to the sector and transaction costs associated with programme implementation are usually excluded). But in reality, implementing a mitigation measure for reducing a particular pollutant may generate co-benefits (e.g. enhance biodiversity) or unintended impacts (e.g. pollution swapping). Additional positive and negative impacts not related to the targeted objective or sector must also be considered in order to reflect the full impacts of measures.

Cost-effective targeting of agri-environmental measures requires an integrated approach, including understanding the bio-physical processes, landholders/users responses to policy, and social costs of measures. The effects of a measure taken are difficult to calculate due to uncertainty about sources of pressures, dose-response relationships and an absence of adequate monitoring data.

Uncertainty may affect cost and effectiveness estimates but also the definition of good status at this early stage, since the good status objectives have not yet been defined with precision in the European Water Directive Framework (WDF, 2010). There are a number of strategies for dealing with uncertainty. These range from selecting the measures where uncertainty is less (French reporting), to producing range estimates (UK reporting) or using sensitivity analysis (as in Spain reporting) and seeking to obtain more information in order to reduce uncertainty. For this latter method, which is commonly advocated, it is necessary to review the full cost of obtaining such additional information, including delayed achievement of objectives, versus benefits. Except for a few studies, for instance, Elofsson (2003) and Lacroix et al. (2005), uncertainties in both costs and effects estimates were rarely incorporated in most of the studies reviewed. Future research should pay due attention to incorporate the stochastic nature of the effectiveness of abatement measures and their impact on economic sectors for better decision support.

4.6.4 Policy setting

The use of incentives are common in environmental policy and have been used internationally to promote biodiversity in conservation (Rode et al., 2015). Generally, they take the form of tax deductions and credits, full or partial payment for conservation projects, low-interest loans, or tradeable credits (Stern, 2006). Internationally, they have included direct payment programs and cost-share projects in the United States for wildlife habitat, and the Conservation Reserve Program along with permanent easements. The European Union's (EU) agri-environment schemes (AES) which are a part of the Common Agricultural Policy, provides an annual payment to farmers to manage their land for the benefit of particular habitats, species, and water quality outcomes.

The cost-effectiveness measures in these studies represent the same issue as economic studies completed in the GBR catchments. First, there is high heterogeneity in the performance of different management practices or actions in different places based on geographic and biophysical parameters (Newburn et al., 2005; Bryan & Crossman, 2008), which underpin variations in benefits for the environmental asset per dollar spent, particularly when the time to achieve benefits is factored in

(Bainbridge et al., 2009; Star et al., 2018; Yang et al., 2010; Yang et al., 2003,). Second, the different farm level management practices and actions vary in effectiveness at reducing pollutants, costs, time lags to be effective (Meals et al., 2010; Strauss et al., 2007), adoption rates by landholders (Feather & Amacher, 1994) and risks of success or interruptions (Prokopy et al., 2008; Doole & Pannell, 2011). Third, in large and complex catchments and ecosystems the effect of reductions on the most important components of the asset varies (Waterhouse et al., 2017; de Valck & Rolfe, 2019).

The platform for policy development in Europe and North America presents different governance structures and subsequent farm policies. This makes the cost structure quite different and, therefore, it is difficult to compare the cost-effectiveness to Australian agriculture. Likewise, the biophysical parameters and interactions between geographic features, climates and unique landscapes make the pollutant loads difficult to compare.

5 Measuring profitability

5.1 Sugarcane

Profitability is 'the fundamental measure of economic performance at a farm level' and its indicators 'measure the relationship between revenues of the farm enterprise and the costs of the inputs (resources) required to produce its output' (van Grieken et al., 2014b, p. 29). The level of profitability of the sugarcane farms are affected by the size of the property, under-utilising or over-utilising labour and its cost (e.g. cost of electricity and water, the yield per ha, sucrose content in sugarcane, increased ratoons, frequency of fertilizer applications, farmer's experience, cost of seed cane, planting and replanting time, varieties planted, soil fertility, irrigation system, the distance between the farm and the mill, use of herbicides, weeding and harvesting) (Thompson, 2014; Poggio & Page 2010a, b; Alluvium, 2016).

This literature review highlighted the importance of considering farm profitability when evaluating cost-effectiveness of management practice change. The change may be ranked as the most cost-effective but it doesn't necessarily mean that it will be the most profitable to a farmer or profitable at all. Ghebremichael et al. (2013), for example, illustrated that management changes aiming to decrease P losses did not result in a positive impact on enterprise profitability and those negative consequences to a farmer can be only revealed by assessing management changes at an enterprise level. It is particularly important for small scale sugarcane farms which dominate the Australian sugar farming industry (Sugar Research Australia & QLD Government, 2015). Enterprises with sugarcane planted in less than 50 ha have cash operating margins close to zero. Nearly 30 per cent of small farm operators are only marginally profitable and do not have sufficient resources to allocate to management practice changes (Deane et al., 2018). There are always trade-offs between water quality and profitability in agricultural production (Contant et al., 1993), thus, accounting for profitability is vital when evaluating cost-effectiveness and ranking practices change options.

The Farm Gross Margin (GM) per hectare is commonly used as a proxy for farm profit (Hirth et al., 2001; Hajkowicz et al., 2005; Masuku, 2011). While GM is useful when assessing the financial impact of adjustments to a farming system, it does not account for overhead costs and, therefore, must be transparent about how the benefits and costs have been captured in the GM (van Grieken et al., 2014b).

5.2 Grazing

Profitability in grazing is particularly driven by live weight gain (LWG) per head of cattle. LWG is dependent on climate sequence, land productivity and stocking rate, while transport costs, labour costs, supplement costs and distance to processing plant or market, all impact on profitability (East, 2010b; Moravek & Hall, 2014; O'Reagain & Bushell, 2015; Jones et al., 2016; Star et al., 2017). By improving land condition through changing management practices, land productivity increases; this can result in greater LWG and lower sediment run off (East, 2010b; Gowen et al., 2012; O'Reagain & Bushell, 2015;

Jones et al., 2016). However, the length of time to improve condition and management practice change impacts can be heavily dependent on rainfall (Star et al., 2017). Other practices driving farm enterprise profitability (linked to higher Return on Assets and Return on Equity) include the average age of turnoff, weaning, and mortality rates (Rolfe et al., 2016). By maximising enterprise profitability in a sustainable manner there is an implicit benefit realised through use of resources in a more efficient manner.

While the grazing industry recognises profit and cost drivers, recording and understanding of the economic implications are less comprehensive (Rolfe et al., 2016). Costs and benefits can vary widely from property to property and are indicative of the heterogeneity of land types, locations, and climate sequences (Rolfe & Windle, 2016b). Inputs to farm production models should incorporate cattle sale prices, capital costs, depreciation, maintenance and salvage values, time frames, and land productivity (Star et al., 2017). Literature at the individual farm scale typically assesses these tangible inputs, as business practices that are unique to land type and condition (Rolfe et al., 2016). Less clear at this scale is intangible transaction cost (i.e. the cost of acquiring information necessary to implement a practice change), opportunity cost (benefits foregone in practice change), and the response times of land productivity to management practice change and other time lags (Star et al., 2016).

Profitability at the farm scale typically relates to discounted cash flow returns in comparison to initial capital outlays for the change in management practice, usually in the form of a net present value (NPV) (NQ Dry Tropics, 2015; Donaghy et al., 2007). While Ash et al. (2001) reported a cash return per year figure, rather than NPV, this was still a discounted cash flow analysis. There is some evidence that improved management practices can return a benefit and reduce sediment run-off. In these situations education and extension support is required for the realisation of these benefits. Though, as previously mentioned, heterogeneity of variables between properties mean incentives are just as likely to be required for benefits to be realised at this scale.

Profitability was less of a focus at the catchment level. As previously mentioned, neither Wilkinson et al. (2015) nor Rust & Star (2018) identified significant benefits to the enterprises. Similarly, Roebeling et al. (2009) found meeting sediment targets typically only came at a cost to grazing, even when maximising regional grazing income for those targets. Roebeling et al. (2009) also captured positive production impacts on the cost side of the equation (a negative cost). As a result, the use of cash flow analysis was less prominent at the catchment and region scale. Wilkinson et al. (2015) used a direct on ground, one off cost and Roebeling et al. (2009) derived an input-output model of agricultural production that produced a figure of total regional grazing income examining the reduction in income from meeting water quality targets. Rust & Star (2018) used a cash flow analysis over ten years to discount costs at a real discount rate of 7 per cent. Only the cash flow analysis would consider the ongoing cost at a farm scale.

At the GBR scale, despite a focus on results of sediment delivered to the GBR, there was considerable awareness of profitability at the farm scale. This is likely due to the tendency to apply farm scale data at an aggregate level to achieve GBR results. Researchers identified farm size, where used to assess the GBR scale sediment export, as relevant, with larger farms tending to benefit from transition to improved management practices (Alluvium, 2016; Beverly et al., 2016). In addition, land type inherent productivity was particularly relevant, along with associated factors like land condition or tree basal area (Star et al., 2012). Star et al. (2012), similar to Roebeling et al. (2009) found negative costs through profits occurred in some instances, while Beverley et al. (2016) incorporated profitability in reported abatement cost figures. Where the literature assessed gully rehabilitation there may be some private benefits as it becomes easier to manage timing of pasture use (Wilkinson et al., 2015). When literature focused on region or basin targets, using targets at the smaller scale tended to provide more profitable farm enterprise outcomes (Alluvium, 2016; Beverly et al., 2016; Rolfe & Windle, 2016a).

6 Conclusion

This literature review has examined research on cost-effectiveness and profitability of changing land management practices in sugarcane and grazing associated with water quality improvement, primarily within the GBR catchments, but also relevant international studies. Methodological approaches to cost-effectiveness, key determinants of cost, assumptions and limitations in bio-physical modelling, and water quality improvement indicators have all been reviewed across the literature.

A number of past studies have used individual level data to develop case study analysis. For future research it is recommended that a panel of industry representatives, agronomist and extension staff are brought together to develop case study properties. This will provide a basis for the latest Water Quality Risk frameworks to be considered against. Both individual studies and representative property analysis provides equally important insights into the profitability for adoption of management and then the effectiveness of potential policy investments.

Cost and effectiveness figures vary across different countries, regions or catchments within a country, between types of land and farming systems. Cost-effectiveness studies are often undertaken for different purposes, time scales within the catchments and audiences. Estimates usually depend on an implementation of specific measures, environmental conditions, spatial and temporal scales, baseline/reference scenario, types of land use and management practices. An inclusion of different costs and/or elements of costs significantly contribute to variations. Such inconsistent approaches and reporting of the results mean that the estimates are not directly comparable and 'do not accurately account for some of the cross sector and regional heterogeneity in abatement costs' (Rolfe & Windle, 2016a, p. 51).

There were a range of identified measures that heavily influenced cost and profitability at the paddock scale. Property characteristics identified and common across sugarcane and grazing production include: property size, soil fertility, land condition, and distance to processing plant or market. Future research should continue to focus on these characteristics along with industry specific characteristics when determining costs and profitability at the paddock scale.

Selected scales and potential costs that should be included in the analysis to improve consistency and accuracy in cost-effectiveness exercises are provided in Table 15. The availability of data, however, is a key factor in whether or not these recommendations on costs and scales can be considered and/or achieved.

Due to relatively limited research on transaction cost within the GBR catchments, many researchers anticipated transaction cost to be zero or excluded from the analysis. However, there is strong evidence that transaction cost should not be ignored because it can be substantial to the landholder and/or other parties implementing practices/programs. Such an exclusion may have implications for cost-effectiveness estimates.

Uncertainty, timescale of effectiveness, and obtaining precise cost measures over a period are the key challenges to reduce pollution from agricultural activities. To some extent, difficulties with assessing the cost-effectiveness measures are due to spatial scale issues (e.g. while measures are implemented at paddock/enterprise level, environmental targets are established at catchment or sub-catchment scales. Achievement of 'good ecological status over short time-scales appears problematic due to lags in water quality and ecosystem responses' (Balana et al., 2015, p. 163; Bouraoui & Grizzetti, 2014; Meals et al., 2010). Realistic time periods for practice change implementation, and benefit realisation, allows better consideration of potential cash flow changes.

Profitability is the fundamental measure of economic performance at a paddock level and this literature review highlighted the importance of considering farm profitability when evaluating cost-effectiveness of management practice change. The change may be ranked as the most cost-effective but it does not necessarily mean that it will be the most profitable to a landholder which is particularly important for small scale enterprises.

Analytical differences in cost-effectiveness studies also contribute significantly to variations in cost-effectiveness results. Assessment and modelling are 'further complicated by the jointness of many pollutants and corresponding management actions' (Rolfe & Windle, 2016a, p. 7) which may be closely linked. Improving sugarcane management practices may result in a reduction in fine sediment while improving grazing management practices may result in reduction in DIN. As such, it is important to consider the effect of changes in management practices on other related pollutants, specifically for DIN, 'where reductions in particulate nitrogen may be just as important in reducing overall DIN loads because of the transformation of particulate nitrogen to DIN in some systems' (Alluvium, 2016, p. 75). Moreover, some actions in particular regions are significantly more effective than in other regions suggesting that a whole reef CEA 'should be conducted to identify if trade-offs between regions could be undertaken, including the impacts that such trade-offs could have on local stream and reef areas' (Alluvium, 2016, p. 75). Modelling and assessment should account for those complexities.

Water quality measures did vary across the studies, however, dollars per unit mass per year appears to be the most common. This measure is also the most robust available, acknowledging the importance of time in realising benefits and that it is a reduction in total mass of pollutant moving to a receiving water body that we wish to achieve. For specific pollutants, common measures are tonne per year reduction for sediment, and dollars per kg per year for pesticides and DIN.

7 Recommendations

Representative vs. individual property information

Both individual and representative farm properties are important. To assess the economic implications of the new water quality risk frameworks expert panels should be used to develop representative properties.

Scale appropriate costs captured

Improved consistency across studies capturing all relevant costs (see

Table *15*) including transaction costs are required to comprehensively consider the cost-effectiveness of different investment options. For each study future costs should be transparent.

Table 15: Recommended costs and pollutants to be assessed by future cost-effectiveness studies

Costs and pollutant measure	Paddock Scale	Catchment or Region Scale	GBR Scale
Project (capital) costs	X	X	X
Based on selected scale of work and potential costs that should be included	X	X	X
Transaction costs / Administration and program costs	X	X	X
Maintenance costs	X	X	X
Other landholder costs/investment	X		
Pollutant exported from the paddock	X		
Pollutant exported to the end of catchment		X	
Pollutant exported to the GBR			X

Risk and Uncertainty

Climate change is one of the key contributing factors to the decline in health of the GBR, however, it is currently poorly captured in cost-effectiveness studies. Future studies should at a minimum consider reasonable time frames for landscape or production responses with 10 years in sugarcane and 20 years in grazing capturing the production aspects with 15 years capturing the Great Barrier Reef Foundation timeframes. Along with this, sensitivity analysis at a minimum is required and where improved biophysical information is available this should be implemented to capture the climate effects. Modelling beyond those time frames starts to grapple with obsolescence of technology or practice.

Risk and uncertainty is a challenge in any assessment of future options, and within the CEA it is more likely to be present in the effectiveness of measures and/or the cost of measures. Perceived risk and/or uncertainty in returns may increase the perceived and real private costs to producers. As such, it would be desirable to apply a range of effectiveness and cost estimates rather than point estimates.
In addition, researchers should report the time-value of money and justify selection. This allows ease of comparison and demonstration of potential risks through sensitivity analysis. Particularly, future work should use real discount rates along with any basis for decision on final rate chosen.

Private and public benefits

Assessing the profitability of an individual to adopt the different management practices needs to be considered in an overall cost-effectiveness analysis. This provides the basis for landholder adoption and insights into the policy mechanism that would be most suitable to support adoption.

Cumulative and diminishing points of return

To consider the full range of cost-effective options, where possible, multiple pollutant reductions should be examined. Past studies have tended to focus on management practices linked to one specific pollutant (e.g. DIN in sugarcane) and this then ignores the fact that potentially other cost effective options can be captured (e.g. sediment reductions in sugarcane).

8 Future research

While research at the paddock scale is limited across sugar cane and grazing, cost-effectiveness studies are limited in general. Researchers should focus future work in determining potential cost-effectiveness outcomes from particular management practices at all scales. This will help with managing the impacts of variability and heterogeneity across the GBR catchments by identifying a range of potential costs and benefits for similar practices in different catchments.

While modelling, it is necessary to extrapolate costs across large areas. Future research should consider the consequences of simplifying management practice change within a bio-economic model. Not only are costs and benefits of practice change subject to time lags, they are also subject to cash flow and capacity restrictions at the paddock scale. Where producers receive relatively low returns and rely heavily on time invested in earning off-farm income, management practice change that significantly affects cash flow in the short term involves considerable risk. Despite this, those management practices may register as the most cost-effective investment at a program level; a potential solution is to look at areas for pollutant reduction that provide the most stability in implementing practice change.

While profitability is one of the main drivers of management practice change, other factors such as information and awareness (Rezvanfar et al., 2009; Rolfe & Gregg, 2015), financial incentives (Januchowski-Hartley et al., 2012; Rolfe & Gregg, 2015), social norms (Läpple & Kelley, 2013), climate change and extreme weather conditions (Liu et al., 2018), uncertainties regarding market price (Pannell et al., 2014), farmers' demographics, knowledge, and attitudes (Lamba et al., 2009; Greiner et al., 2009), risk and time preferences (Pannell et al., 2014; Prokopy et al., 2014), farmer's environmental consciousness (Ulrich-Schad et al., 2016), and farm characteristics (Pannell et al., 2014; Gedikoglu & McCann, 2012) may encourage or discourage adoption. Since better understanding of those factors can be crucial in explaining practice change behaviour, further and more comprehensive investigation on their role in decision-making process should benefit the research.

The use of representative farms within the modelling is integral to determining water quality impact across a large area. However, there should be acknowledgement of the heterogeneity of real world property characteristics. This is especially relevant when extrapolating a practice change to commercial scale. The use of trial and monitoring data to reinforce modelling is, therefore, vital to delivering realistic outcomes.

There is a need for targeting investment, with the literature potentially identifying greater pollutant reductions through spatial targeting and focusing on areas that produce the majority of a pollutant (Star et al. 2012). The research highlighted a need for further study on catchment specific characteristics, such as the rate of intermediate sediment deposits (Rust & Star, 2018). There is also identified potential to expand the research conducted to capture regional variations (Star et al., 2013).

Appendix 1

Sugarcane growing industry

Sugarcane in Australia is mainly growing along coastal areas with high rainfalls. Raw sugar production in QLD is accounting for approximately 95 per cent of Australia's raw sugar production and a large proportion of this product is sold on the world market (Harvey et al., 2016; QLD Government, 2018c). In 2017-18, 377,000 ha of sugarcane were harvested and 33.35 million tonnes of cane crushed resulting in an export value of $1 687 million. Australia was the ninth largest world sugar producer at 4.6 million tonnes of cane production (ABARES, 2017; ABARES, 2018a). The gross value of production (GVP) of Queensland's sugarcane in 2018-19 estimated to be $1.036 billion with 31.4 million tonnes crashed and with the expected total revenue in raw sugar equivalent to be $1.58 billion (QLD Government, 2018c).

Over the past 20 years, sugarcane industry in QLD encountered numerous pressures that significantly impacted farm environmental conditions and profitability. Those pressures included extreme weather events, changes in regulations and rules, increasing costs of inputs, outbreaks of diseases and pests and international competition. Combination of relatively weak and long-lasting world sugar prices and these internal factors required the industry to develop more innovative land management practices for improving farm productivity and profitability (Harvey et al., 2016).

Currently increased sugarcane production from India and Pakistan and support policies influencing export decisions in Thailand and India are expected to have a significant influence on world sugar price volatility in 2018–19 (QLD Government, 2018c). ABARES (2018a) forecasts that sugar prices may fall as much as to a 10-year low due to high export availability. It is expected that world sugarcane import demand will be significantly reduced due to many suppliers in countries such as China, US and EU which are the major consumers of sugar product. As such, returns to sugarcane growers in Australia predicted to be lower because of fall in world sugar prices. Also, import tariffs on sugar in China designed to protect its sugarcane domestic industry, 'expected to direct world exports to other Asian and Middle Eastern markets' and redirect export to the Republic of Korea, Japan and Indonesia which may put even further pressure on return to Australian cane growers (ABARES, 2018a, p. 27).

Sugarcane has been a dominant crop in GBR catchment for more than 150 years (Deane et al., 2018) and 85 per cent of sugar production in QLD occurs in the Wet Tropics (WT), Burdekin and Mackay Whitsunday (MWS) regions (Harvey et al., 2016). Cane growing activities have been identified as the largest contributor of DIN and pesticides to the GBR catchment area. The WT, Burdekin and Fitzroy regions contribute most of the river pollutant loads while the Mackay Whitsunday and Burnett Mary (BM) regions are also significant contributors highlighting the importance of 'identifying management priorities at the catchment or finer scale' (Waterhouse et al., 2017, p. 11).

Catchment modelling shows significant increase in 'mean-annual …nutrient and pesticide loads delivered' to the GBR lagoon 'since pre-development conditions' (Waterhouse et al., 2017, p. 11) which include

- approx. 2.0 fold increase in DIN ranging from 1.2 to 6.0 fold depending on the region with the exception of Cape York and with the most significant exposure from the Herbert, Johnstone, Tully, Haughton, Murray, Mulgrave-Russell, and Plane catchments;
- approx. 1.5 fold increase in particulate nitrogen (N) ranging from 1.2 to 2.2 fold depending on the region; and
- approx. 2.9 fold increase in particulate phosphorus ranging from 1.2 to 5.3 folds (Waterhouse et al., 2017).

Pesticide loads are present in all regions and estimated to be approximately 12,000kg per annum across the entire GBR. The Mackay Whitsunday and the Lower Burdekin regions pose high to moderate risk to ecosystems from pesticides (Waterhouse et al., 2017).

Grazing industry

The QLD grazing industry represents 43 per cent of the Australian cattle herd, and covers approximately 75 per cent of the area within the GBR catchment (Ernst & Young, 2018), (Shaw & Silburn, 2016). Approximately 47 per cent of Queensland's cattle herd is represented in the GBR catchments along with 52 per cent of all grazing businesses (ABS, 2018). The meat and livestock industry in QLD employs over 40,000 people directly, contributing more than 20 per cent of direct employment within the Australian meat and livestock industry (Ernst & Young, 2018). Queensland's grazing industry is worth an estimated $5.05 billion in GVP (QLD Government, 2018c).

Financially, QLD beef enterprises are close to Queensland's ten-year average equity ratio of 88.6 per cent, at 88.2 per cent in the 2016-17 financial year (Meat and Livestock Australia, 2018). This is consistent with safe levels of equity in the long term, which is approximately 85 per cent, below which debt servicing becomes challenging for the business (McLean et al., 2014). Despite this, the national ten-year average remains higher at 89.6 per cent (

), and at 89.4 per cent for the 2016-17 financial year (Meat and Livestock Australia, 2018). This may be indicative of relatively higher debt levels compared with the industry nationally. A small number of beef producers providing the majority of beef in Northern Australia tend to skew data (McLean et al., 2014).

demonstrates the increasingly leveraged nature of these high performing enterprises (the top 25% by rate of return) as well as Queensland's industry on average.

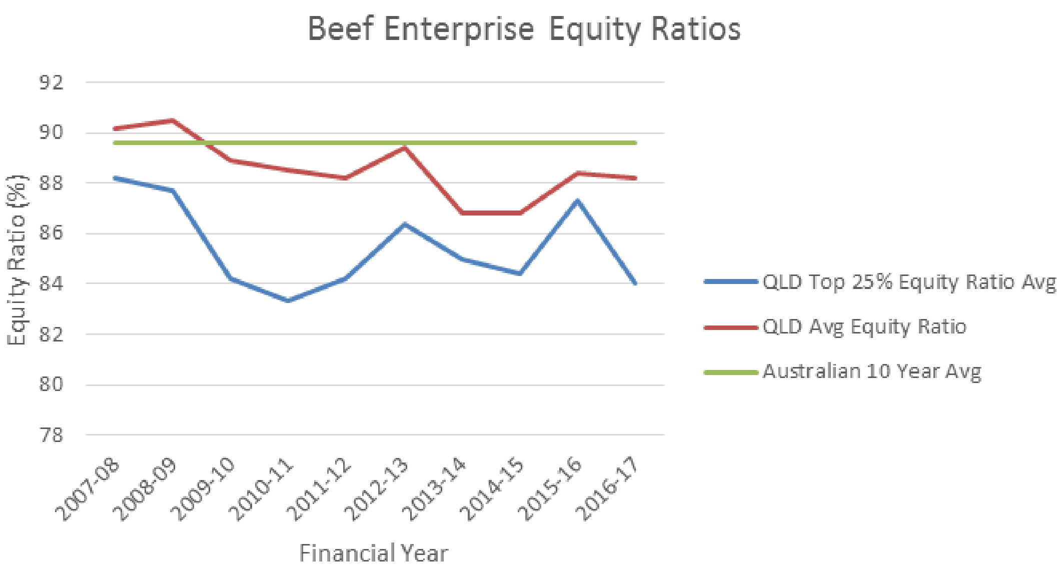

Figure 2: Beef Enterprise Equity Ratios
(Source: Meat and Livestock Australia, 2018)

Preliminary figures from the Australian Bureau of Agriculture and Resource Economics and Sciences (ABARES) indicate rate of return on capital for Australian beef businesses is currently projected to be 2.5 per cent in the 2017-18 financial year, down from 2.7 per cent the previous year (ABARES, 2018b), (Meat and Livestock Australia, 2018). This is still well above the average rate of return between 2001 and 2017 of 1.2 per cent, with Northern Australian beef enterprises consistently outperforming those in Southern Australia (ABARES, 2018b). Figure 3 excludes 2018 preliminary figures but indicates a slight upwards trend after strong returns in the 2015-16 and 2016-17 financial years. This aligns with a significant increase in price per head of cattle received in those years with growth in prices received of

38 per cent and 11 per cent respectively in 2017-18 dollars. The higher rates of return in the top 25 per cent tend to have larger farm size and more debt (Meat and Livestock Australia, 2018).

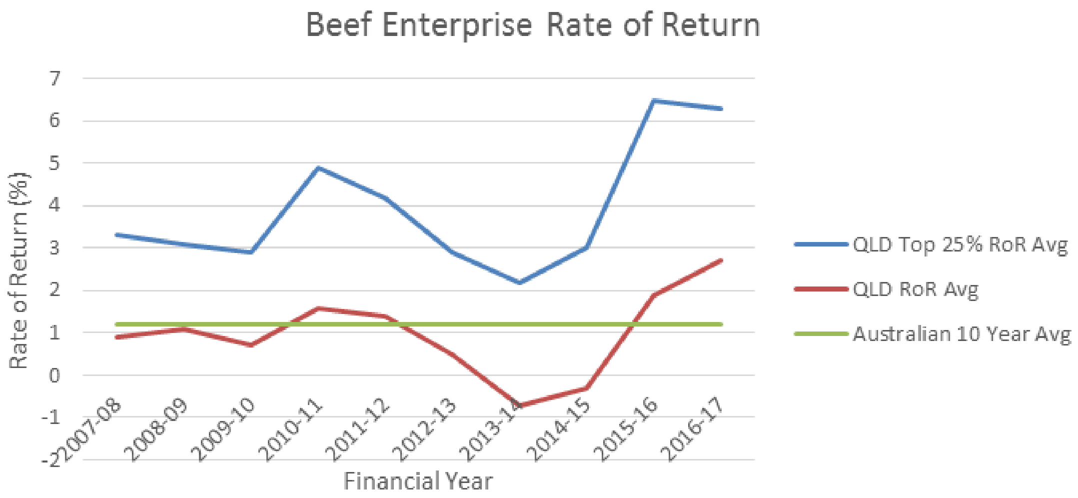

Figure 3: Beef Enterprise Rate of Return
(Source: Meat and Livestock Australia, 2018)

Challenges for the Northern Australian beef industry moving forward include recent trends of declining terms of trade, increasing debt levels and declining profitability with no change in productivity. Terms of trade conditions mean that costs have remained the same in the face of declining prices received in real terms (McLean et al., 2014). While total farm debt (average per farm) in the decade to the 2017-18 financial year has seen a slight decline, it is still high historically considering productivity has remained relatively unchanged (ABARES, 2018b; Boult et al., 2016). Total factor productivity for Northern Australia over the 38 years to 2015-16 financial year averaged 1.2 per cent per year, 0.3 per cent higher than Southern Australia's 0.9 per cent (Boult et al., 2016). This is in part due to a slight year-to-year reduction in input use averaging negative 0.2 per cent for Northern Australia over that time, indicating enhanced efficiency (Table 16).

Table 16: Total factor productivity for Northern Australia (reproduced from Boult et al., 2016)

Region	TFP (%)	Output (%)	Input (%)
Northern	1.2	1	-0.2
Southern	0.9	1.2	0.3

Source: Boult et al. (2016)

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