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Transferring and extrapolating estimates of cost-effectiveness for water quality outcomes: Challenges and lessons from the Great Barrier Reef

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ABSTRACT

In recent decades the declining health of the Great Barrier Reef has led to a number of government policies being implemented to reduce pollutant loads from the adjacent agricultural-based catchments. There is increasing use of cost-effectiveness measures to help prioritise between different programs and actions to reduce pollutants, given limited resources and the scale of the issues. However there are a small number of primary studies available, and the consistency of cost-effectiveness measures and their application is limited, particularly given the various uncertainties that underlie the measures. Unlike Europe and the United States of America water policy or benefit transfer approaches, there are no procedural guidance studies that must be followed in the context of the Great Barrier Reef catchments. In this study we review the use of cost effectiveness estimates for pollutant reduction into the Great Barrier Reef in the context of a benefit transfer framework, where estimates of costs from a particular case study are transferred to various scenarios within different catchments. The conclusions suggest a framework be developed for the Great Barrier Reef, which is consistent, transparent, and rigorous.

1. Introduction

Internationally there has been substantial policy attention on addressing the impacts of poor water quality from agriculture on estuarine ecosystems, with cost-effectiveness employed to improve the allocation of funds, assess likelihood of adoption of particular management practices and evaluate past investments (Fröschl et al., 2008; Gooday et al., 2014; Elofsson, 2010). In Australia the declining health of the Great Barrier Reef (GBR) has been attributed to a number of factors, including pollutant runoff from agricultural-based industries (Brodie et al., 2012; Brodie et al., 2017; Kroon et al., 2016). The pressure on the GBR has led to a large number of policies and investments to improve water quality, many of which are focused on improving agricultural management practices (The State of Queensland, 2017). The Reef 2050 plan has a target of 90% of land managers operating low risk water quality practices (previously termed best management practices) by 2050 (The State of Queensland, 2017). Given the variation in biophysical, climate, agricultural and management factors, there has been increasing focus on assessing the cost effectiveness of different measures so as to guide the allocation of public funds and select options that would maximise outcomes and participation by landholders.

The initial focus of cost-effectiveness studies in the GBR was to identify costs of landholders changing management practices to reduce pollutants leaving agricultural lands (East and Star, 2010, Van Grieken et al., 2010), but cost information has also become important to help understand adoption barriers to management change (Rolfe and Gregg, 2015) and to help evaluate different policy mechanism designs (Rolfe and Windle, 2011). For these purposes, cost-effectiveness has been measured, which is essentially the ratio of costs involved to achieve pollution changes. While the initial focus was on farm level studies, in recent years there has also been increasing interest in predicting the total costs of achieving water quality targets and prioritising between actions for the whole GBR, which involves greater modelling and extrapolation of available estimates (Alluvium, 2016, 2019; Beverly et al., 2016; Star et al., 2018).

Policy frameworks for the GBR do not require strict assessments of the benefits and costs, unlike in Europe, where the Water Framework directive explicitly provides guidance regarding the application of economic principles, tools, and instruments (Balana et al., 2011; Carvalho et al., 2019; Hanley et al., 2006; Lam et al., 2011; Martin-Ortega, 2012),

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or the United States, where detailed cost-benefit assessments are required under the Clean Water Act (Keiser and Shapiro, 2018). Although there is no strict requirement for cost-effectiveness in policy settings for the GBR, use has been increasing for two broad reasons. On the supply side there is increasing availability of detailed data from various trials and management options about the pollutant reductions achieved and the costs involved that have allowed more estimates to be generated. On the demand side the requirements to meet ambitious pollutant reduction targets with set funding caps have focused greater attention on where activities should be prioritised.

Information about costs helps to identify where investments are best made and also identifies viable options for landholders to change management practices. However, the assessment of costs relating to agricultural water management is complex. Keiser et al. (2019) identifies that estimation of primary costs is difficult, because costs are difficult to apportion, cost signals are distorted because of market power issues, and taxes and regulations distort real costs. Other problems are that it can be difficult to measure physical changes such as pollutant reductions, and agricultural systems are often stochastic because of climate and market variations (Star et al., 2018). These issues about the accuracy of a primary study are often referred to in terms of the reliability of the original estimate. The small number of primary studies available in the GBR means that estimates of cost effectiveness are routinely transferred between case studies and extrapolated to different scales (Star et al., 2018), raising issues of whether there are issues of validity generated by the transfer processes.

The transfer of cost-effectiveness estimates to various situations in the GBR are examples of benefit transfer, where a value estimate from a primary case study, typically referred to as a 'study site', is transferred to another similar situation, typically referred to as a 'policy site' (Loomis and Rosenberger, 2006; Johnston et al., 2015). Different types of transfers can be used, including transfers of single unit values, benefit functions, or a meta-analysis summarises values across a number of studies. It is also possible to transfer the source values as they are, or to adjust them to account for variations in time and differences between the study and policy sites (Johnston et al., 2015). Substantial effort has been devoted to improving the reliability and accuracy of benefit transfer methods to value benefits of water quality improvements (Bateman et al., 2011; Johnston et al., 2015). While most attention has been on transferring benefits, including non-market values, similar challenges of ensuring reliability and validity also apply to transfers of cost estimates.

The aim of this paper is review the use of cost-effectiveness estimates for improving water quality in GBR catchments, particularly in reference to the widespread transfer and reuse of a small number of primary study values. The next section outlines the concepts underpinning cost-effectiveness, followed in Section 3 by identifying some of the important challenges in generating estimates in GBR contexts. An overview of the availability and use of cost effectiveness estimates in GBR catchments is provided in Section 4, and an analysis of consistency between estimates follows in Section 5. Discussion and conclusions follow in the final section.

2. Cost-effectiveness

Cost-effectiveness analysis is an applied economic assessment technique that compares the outputs generated by an action relative to the costs involved and is often used to prioritise actions in natural resource management (Balana et al., 2011; Duke et al., 2013; Boerema et al., 2018). It allows different management actions and policy options to be evaluated relative to the outcome in terms of the costs involved, which are important criteria for policymakers (Balana et al., 2011). Cost-effectiveness analysis is simpler to apply than cost-benefit analysis, as it avoids the often-contentious monetary valuation of benefits involved in the latter approach. However a disadvantage of cost effectiveness analysis is that it is often difficult to compare analyses between actions because units may vary (Boerema et al., 2018).

Cost-effectiveness is commonly used for evaluating options to improve environmental outcomes (Claassen et al., 2008; Duke et al., 2013; Wätzold and Schwerdtner, 2005), including pollution reduction or improving environmental quality in water bodies (Cools et al., 2011; Doole, 2012; Lise and van der Veeren, 2002). In these scenarios, the environmental goals have already been identified, so the analyst can then concentrate on finding the least cost way of meeting the environmental objectives. While there are some subtle differences in the way that cost-effectiveness analysis is conducted, each involves at least four key components (Boerema et al., 2018):

1. Collecting data on the cost of the management measures;
2. Quantifying the effect of each management measure on the different environmental target of interest;
3. Calculating the average cost of each measure for environmental target of interest, and
4. Selecting the most cost effective strategy.

Drawing on Rolfe et al. (2018), the cost-effectiveness measure relevant to the pollutant categories for water quality in the Great Barrier Reef can be defined in the following way:

$$CE_{abc} = C_{mi} / B_{pin} \quad (1)$$

where CE = cost-effectiveness, C = present value of project costs, B = project benefits, subscripts a, b and c refer to farm, grower and catchment respectively, subscript m refers to the management or intervention, i refers to the discount rate used, while subscripts P refer to the pollutants being assessed. Pollutants are typically assessed either as the annual change in emissions or the total reductions over the time span being considered (n). When the stream of pollutant reductions are uneven over time, or differ between projects, then they should be discounted like costs to a present value equivalent so that different projects can be compared.

Brouwer and De Blois (2008) identify the practical steps in completing a cost-effectiveness analysis as:

1. Identify the environmental objective/s involved (target situation);
2. Determine the extent to which the environmental objective/s is/are met;
3. Identify sources of pollution, pressures and impacts now and in the future over the appropriate time horizon and geographical scale (baseline situation);
4. Identify measures to bridge the gap between the reference (baseline) and target situation (environmental objective/s);
5. Assess the effectiveness of these measures in reaching the environmental objective/s. Assess the direct (and if relevant indirect) costs of these measures;
6. Rank measures in terms of increasing unit costs; and
7. Determine the least cost way to reach the environmental objective/s based on the ranking of measures.

Important challenges are to deal with uncertainties around the measured costs and their effectiveness and to identify what components of costs should be included (Brouwer and De Blois, 2008). While some analyses include only direct program costs, Duke et al. (2013) recommend that all costs should be monetised and included in an assessment, but suggest that the subset of costs falling on landholders should be separately identified because of the direct impacts on program participation.

Other challenging aspects of conducting a cost-effectiveness analysis are the measures of the physical changes involved and the integration of ecological knowledge with economic analysis (Duke et al., 2013; Wätzold and Schwerdtner, 2005), and dealing with the risks and uncertainties associated with outcomes, particularly when there are variations over temporal and spatial scales (Brouwer and De Blois, 2008; Duke et al., 2013; Glenk et al., 2020; Jung et al., 2020). Balana et al. (2011) reviewed a range of approaches that can be taken to predict outcomes when risk and uncertainty are present, including mathematical programming, bioeconomic modelling, regression models, Bayesian belief networks and simulation and optimization models.

3. The great barrier reef context

The GBR covers two-thirds of the coast of Queensland, involving six major Natural Resource Management (NRM) regions (Fig. 1). Initially, the Great Barrier Reef Water Quality Protection Plan (Reef Plan) identified priority pollutants and industries to target based on loads entering the marine environment (The State of Queensland, 2013). Under Reef Plan 2013, a number of targets were set, which include a 20% reduction in Total Suspended Sediments (TSS) and a 40% reduction in pesticides and nutrients (specifically Dissolved Inorganic Nitrogen (DIN)), and an allied target of 90% of land managers to be using best management practices by 2018. In 2017, Reef 2050 Plan (The State of Queensland, 2017) was developed, and a number of catchment specific targets have been set along with overarching targets, including a 25% reduction in Fine Suspended Sediments (FSS), 20% reduction in particulate nutrient loads, 60% reduction in Dissolved Inorganic Nitrogen (DIN), and an allied target of 90% of land managers to be using best management practices by 2025.

Changes in the definition of best management practices have also

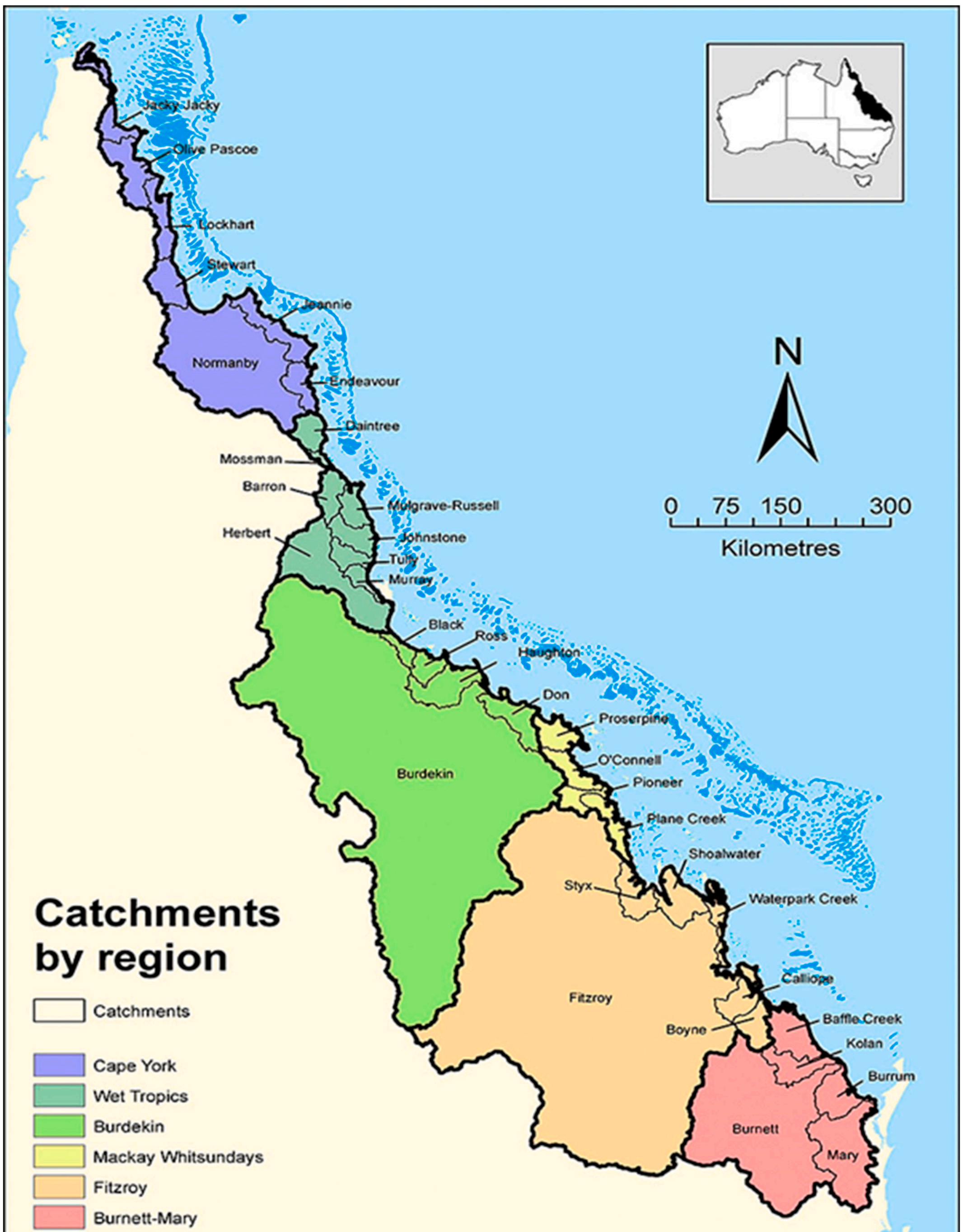

Fig. 1. Natural Resource Management regions in the Great Barrier Reef.

changed over time; initially, management practices were classified relative to their impact on land condition and subsequent water quality outcomes ranging from D - dated, or degraded through to medium risk (C), low risk (B - best management) and extremely low risk (A – aspirational best practice) (Government, Q, 2013). In 2017-18 these were then updated to a water quality risk framework, with practices classified as A, B, C, D from “A” very low water quality risk through to “D” high water quality risk, with “B” described as current best management. The classification of management is also separate from the grazing A, B, C, D land condition classification, which is the state of land classified by bare ground, pasture species and woody weeds (Karfs et al., 2009; Scarth et al., 2006).

The current progress towards the Reef Plan targets has been tracked through the Reef Plan report cards, underpinned by the Paddock to Reef Monitoring and Modelling (P2R) program to capture progress towards the targets (Carroll et al., 2012). The program monitors adoption and ground cover, along with river flows and water quality monitoring sites across the GBR catchments. These monitored parameters are collated into a Source Catchments model, accounting for the biophysical parameters and geographical features, which then allow the end-of-catchment pollutant reductions to be predicted for different farm management changes (Carroll et al., 2012).

To achieve this level of landholder change, the Australian and Queensland governments have utilised different policy and program mechanisms, including incentives (Bainbridge et al., 2009), extension and education (Barbi et al., 2015), market-based instruments (Great Barrier Reef Water Science Taskforce, 2016; Rolfe et al., 2011; Smart et al., 2016), regulation (*Great Barrier Reef Protection Amendment Bill 2009* (Qld) and conservation management land purchases Fig. 1.

The application of economic analysis to these major policy and funding initiatives in the GBR has been limited. Initial interest focused on valuing benefits generated by the GBR, including both use and non-use benefits, with De Valck and Rolfe et al. (2018) identifying 48 primary studies from 1983 to 2019. Multi-criteria analysis was applied to prioritise funding allocations to regional areas in the GBR (Cotsell et al., 2009) and the relative impact of diffuse source pollution across river basins using bio-physical, ecological and socio-economic information (Greiner et al., 2005). There has also been application of conservation planning tools for zoning decisions in the GBR, but consideration of socio-economic impacts was very broad brush and not well documented (Ban and Klein, 2009). Other areas of focus have been to identify the farm level costs of making management changes (East and Star, 2010; Rolfe and Gregg, 2015; Van Grieken et al., 2010), mechanism design (Greiner, 2015; Rolfe et al., 2011; Rolfe and Windle, 2011; Smart et al., 2016), adoption issues (Rolfe and Gregg, 2015; Rolfe and Harvey, 2017), and investment prioritisation (Alluvium, 2016; Star et al., 2019; Star et al., 2015; Star et al., 2018).

The initial focus of efforts to measure both the benefits and the costs of water quality improvements was piecemeal and uncoordinated, perhaps consistent with a ‘wicked problem’ where the complexity of issues and the ways to evaluate and address them only become apparent over time (Peters, 2017). On the costs side, the earlier work was conducted by agronomists and other production specialists, who considered what the biophysical effects of practices such as over-grazing and over-fertilising would be on both pollutants and farm productivity (Ash et al., 1995; Landsberg et al., 2002; Roebeling et al., 2009; Mallawaarachchi et al., 2002). Over time, economists began to analyse how more risky practices such as overgrazing would impact profitability, helping to incorporate offsetting impacts of productivity changes into cost-effectiveness assessments of management changes (Macleod and Johnston, 1990; MacLeod and McIvor, 2008). Productivity analysis from rangelands grazing systems was used to point out that grazing land in better condition, with lower sediment emissions, generated higher returns to producers (Macleod and Johnston, 1990; McIvor et al., 1995), while analysis in sugarcane production focused on identifying how excessive levels of fertiliser application and other outdated management practices had sub-optimal economic and ecological outcomes (e.g. Schroeder et al., 1998; Thorburn et al., 2005, 2010). Over time there was increased evidence of positive economic tradeoffs with improved management in both grazing lands (MacLeod and McIvor, 2006, 2008) and sugarcane production (Thorburn et al., 2003; Bell, 2014).By 2010, analysis began to emerge that went beyond costing farm level changes to showing the relationships between investment costs, pollutant reductions and production tradeoffs expected from programs and subsequent actions designed to improve water quality from agricultural operations (East and Star, 2010; Star and Donaghy, 2010; Van Grieken et al., 2010, 2013). These estimates were slow to emerge because bioeconomic modelling was required to account for the complex interplay between production and environmental outputs when management or inputs change, partly because of stochastic impacts of factors such as weather and lagged relationships. It took time to develop many of the biophysical and agronomic relationships that underpinned those bioeconomic models. There was increased use of production models such as GRASP (for grazing) (Whish, 2012) and APSIM (for cropping) (McCown et al., 1996) as core components of bioeconomic modelling.

4. Challenges in estimating cost-effectiveness for pollutant reductions in the GBR

4.1. Elements of costs to include for primary studies

There are a number of cost elements associated with changing management practices or implementing remediation changes. Measures of cost-effectiveness may vary because of differences in the elements of costs that are included, the period of analysis and the risks considered. In addition to the direct financial costs of proposals or management changes, other costs that may be considered include additional program costs (operation, maintenance and decommissioning costs), additional costs incurred by landholders (private capital costs, opportunity costs and transaction costs) (McCann et al., 2005), and broader organisational costs (program and administration costs) (Coggan et al., 2015; Duke et al., 2013).There are often large non-financial costs involved, and costs may be non-linear with increases in the quantity of desired changes. The primary assessment of these costs is critical to understand the financial impacts and the adoption implications and to be transferred for further catchment prioritisation. An indicative typology of the types of costs to consider is shown in Fig. 2.

4.2. Consistency of measurement units

Cost-effectiveness may also vary because of measurement units being different across studies. Some projects will measure cost-effectiveness by comparing costs to annual reductions in pollutants, whereas others will assess costs against total savings in pollutants. Where costs are incurred over time, there may be variations in the discounting process to calculate present values. Balana et al. (2011) recommend that costs should be expressed as the equivalent annual cost that includes annualised investment, maintenance, and operational costs for implementing a proposed measure or combination of measures.

A major challenge in measuring cost effectiveness is to ensure that the physical units are assessed in a consistent manner. Star et al. (2018) note that assessing the outcomes of agricultural practice changes at a catchment scale is complex because it is important to consider not just the practice change but also:

- The effectiveness of the practice change in reducing emissions.
- The time lags until the change is effective.
- The proportion of farmers who will adopt the new practice.
- The transmission losses between the generated pollutant and the receiving environment.
- Risks of variations because of weather and other factors, and
- Varying effects on the marine environment.

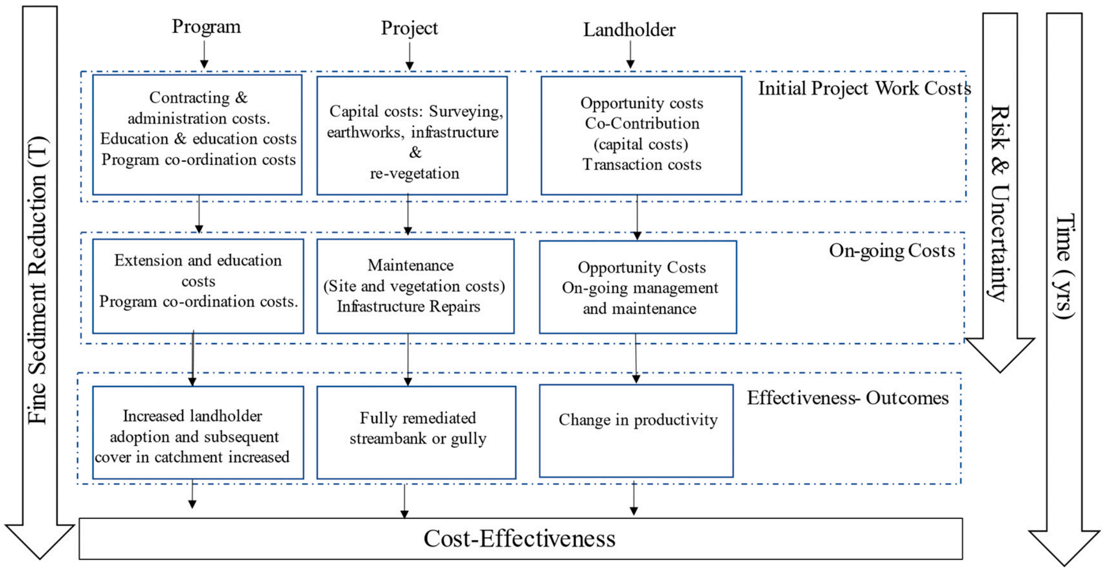

Fig. 2. Example of indicative typology example for gully or streambank cost components over the life of the remediation for incentive mechanisms.

In the GBR cost-effectiveness is typically measured as the costs of achieving pollutant reductions at the end-of-catchment, but it would be more appropriate to assess it as costs relative to changes in the marine environment (Star et al., 2018).

4.3. Timing of costs and benefits

There are often significant time lags in the GBR between an action and the response to that action (Bartley et al., 2020; Mcdowell and Laurenson, 2014; Meals et al., 2010), which impacts on the effectiveness benefits. In modelling for the GBR report cards and a number of bio-economic modelling approaches (Alluvium, 2019), once an action has been assigned, it is modelled to achieve an immediate reduction. However, the more significant the time lag, the more that cost-effectiveness is over-estimated (i.e. reported more favourably than it is in reality). In contrast, Star et al. (2018) accounted for time delays to achieve practice change and pollutant reductions at end-of-catchment to differentiate between projects that had faster or slower response times. This is separate to the time lags in the ecological response which also experience time lag effects (Laukkanen and Huhtala, 2008). Discounting is an effective way of standardising variations in cost and benefit streams over time, however often only costs are discounted.

4.4. Risk and uncertainty

A major challenge in assessing cost-effectiveness is that changes in outputs can rarely be predicted with certainty (Christianson et al., 2013). There are two broad groups of reasons for this. The first reflects the heterogeneity in land and water resources, production enterprises and management systems (Baumgart-Getz et al., 2012), while the second reflect the stochastic impacts of weather variables and biological processes (Brouwer and De Blois, 2008). These factors make it difficult to predict costs (which is why bioeconomic models are often required) and may also impact landholder perceptions about the outcomes of management changes. While some studies have incorporated elements of heterogeneity (van Grieken et al., 2010) and weather variability (Star et al., 2015) into cost-effectiveness measures for the GBR, the treatment of these factors is not consistent.

5. Identifying cost-effectiveness studies in the GBR

To identify cost-effectiveness studies relevant to the GBR, a formal search of published literature was conducted. The Scopus database and the Queensland Government Publications database were searched in August 2020 along with relevant references of these articles for estimates of cost-effectiveness. The criteria used in the search string contained keywords and boolean operators, including: *economics, management practice changes, water quality, Great Barrier Reef, grazing economics, sugarcane economics, cost-effective*.

The initial search results yielded 157 papers. Papers were evaluated through stages to select only those that reported estimates of cost-effectiveness (Fig. 3). Each article was then reviewed for relevance to the study, and current Reef Plan (2009-2017) policy context.

There were 20 articles and reports remaining at the end of the selection process that could be classified into three broad categories by purpose: costs, evaluation or prioritisation. The evaluation studies were largely based at the farm or project scale. One was a case study that focused on actual site works (Rust and Star, 2018), others reported bioeconomic modelling to summarise farm-level costs for specific catchments and industries (Star et al., 2013; Van Grieken et al., 2010), and some evaluated the investments and outcomes from previous investment programs (Rolfe et al., 2018). In contrast, the prioritisation studies were focused at larger scales. Five of the prioritisation studies involved water quality improvement plans for the Natural Resource Management groups in the major catchments, while others pooled available data across catchments and industries to generate cost-effectiveness estimates so that actions could be prioritised (Alluvium,

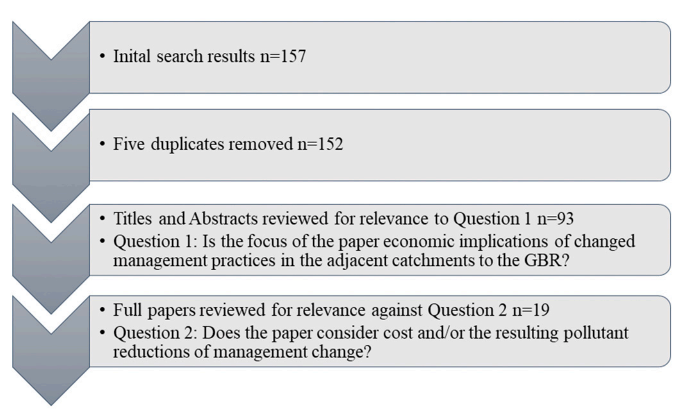

Fig. 3. Process of selecting studies in the systematic review.

2016; Star et al., 2018).

The 20 studies varied across a large number of factors (Table 1), including the scale of the analysis, the pollutants considered, and the land uses, mechanisms and management changes involved. The methodology used for assessment, time frames and discount rates also varied.

The small number of available cost-effectiveness studies explain why benefit transfer methods have been widely used to predict costs across catchments and to extrapolate up to lager scopes (e.g. Alluvium, 2016, 2019; Star et al., 2018). There is a substantial literature around benefit transfer and protocols to ensure that transferred estimates meet reliability and validity standards and that applications follow appropriate guidelines (Johnston et al., 2015, 2020). Using the framework from Johnston et al. (2020) the cost-effectiveness studies identified from the review process above are considered in the context of (i) value definition and valuation context, (ii) theoretical foundation, (iii) selection of study sites and information, (iv) data adjustments and selection of transfer method, (v) auxiliary data, data analysis and robustness analyses, and (vi) aggregation and reporting, reviewing what has been done and further improvements that can be made.

5.1. Value definition and context

Value definition and context is described by Johnston et al. (2020) as the delineation of relevant features such as the policy change in question, in this case a pollutant change, the increment of change to be valued. Along with the geographic location of the policy site, geospatial features of the site, quantities/qualities of substitutes and complements, market conditions such as relevant prices and incomes, and the composition and size of the affected population. In the context of the GBR, the most important value definition relates to the pollutant type, with different cost-effectiveness estimates reported for key pollutants such as fine sediments, nutrients and pesticides.

Changing requirements for cost-effectiveness estimates have resulted in variations in value definition and value context over time. Initially, the value definition was based around the private costs and benefits for landholders to change management actions (Star et al., 2011a; Van Grieken et al., 2010). This later shifted to evaluating programs from a public good perspective (Rolfe and Windle, 2011; Rolfe et al., 2018; Smart et al., 2016) and then to allocating funds towards achieving a pollution target (Alluvium, 2016, 2019). These different definitions and contexts mean that the underlying scope of cost-effectiveness measures has changed from private cost tradeoffs to public cost tradeoffs.

The simplest forms of cost assessment have focused on changes to on-farm profits or private operating costs from management changes performed through partial economic analysis. Changes in Gross Margin (GM), for example, can be calculated by determining the change in variable costs and production implications from an enterprise variation, holding most factors such as capital, prices and fixed costs constant (Law et al., 2016). These analyses employ farm production models to predict the changes in costs or net returns from changes in management actions, which can then be combined with estimates of pollutant reduction to estimate cost-effectiveness. A model of production on an 'average' farm in a district is typically assembled, and then changes in inputs are used to estimate flow-through effects to operating costs and revenues and net returns (Rust et al., 2017; Van Grieken et al., 2010).

Other evaluation approaches have used data from funding programs to generate cost-effectiveness by comparing the costs of the investments (both public and private) with measured or modelled reductions in pollutants. While these evaluation approaches provide feedback about cost-effectiveness at the farm level, they are also useful in evaluating the policy mechanism's effectiveness (Rolfe and Windle, 2016).

5.2. Theoretical foundation

The differences between theoretical definitions of the value measure include if the purpose is to generate public benefits or producer benefits. For example, the producer surplus realised by a landholder changing management practices or the consumer surplus from the public achieving pollutant reductions have different theoretical foundations, although both can be derived from changing management practices. Johnston et al. (2020) note that pooling divergent welfare constructs within valuation meta-data may be useful for analysing a topic, but to transfer these estimates between each other results in inconsistent analysis and poor prediction of a well-defined economic value (Johnston et al., 2020).

The theoretical foundations of the cost-effectiveness studies completed in the GBR have been in a similar context. However, a notable finding is that there is only limited consistency in methodology between studies, even within sub-groups of studies involving farm level studies for a particular pollutant. Table 2 summarises key factors for studies that assessed on-farm cost-effectiveness estimates for sediment reduction, identifying differences in the types of costs included, the practices assessed, and the treatment of discount rates and time periods. These methodological variations in conjunction with different theoretical frameworks make it difficult to compare the resulting cost-effective estimates.

5.3. Selection of study sites and information

Depending on the type of analysis to be employed, the study site should closely align with policy site values, and for meta-equations, the study sites should collectively provide data to calibrate the transfer to policy site conditions. Johnston et al. (2020) recommend that for a data selection process, four different general steps are: (i) identification of potentially relevant sites, (ii) evaluation and screening of studies for transfer suitability, (iii) identification of relevant study site data, and (iv) supplementation of study site data with information from external sources.

Studies sites can allow for variations to be understood and calibrated. As there is such a variety of management actions and strategies available to reduce emissions into waterways in the GBR catchments, it is normal to report cost-effectiveness measures (cost per pollutant reduction) than costs per se. However, transferring these values then presents challenges. Large variations in pollutant reductions underpin major heterogeneity in cost-effectiveness estimates (Alluvium, 2019; Star et al., 2016a). In the GBR catchments, these variations are caused by heterogeneity in geography, soils, climate and waterway transmission, amongst other factors, as well as varying scale, time lags in pollutant reductions and transmission (Bartley et al., 2020; Darr, 2017; Doriean et al., 2020; Packett et al., 2009; Scanlan et al., 1996; Silburn, 2011). This means that it is important to understand the context when transferring estimates of cost-effectiveness.

There have also been different use of data models for pollutant run-offs and efficacies to estimate the pollutant changes underpinning cost-effectiveness estimates. Paddock Scale models such as GRASP (Whish, 2012) and How Leaky (Ghahramani et al., 2020) were initially used to capture local case study site characteristics (East and Star, 2010; Star et al., 2011b). The Source Catchment Model has been used to assess the annual average change in loads over 30 years for a number of management changes at the sub-catchment level (Star et al., 2018; Alluvium, 2016). Using an annual average load avoids spikes in loads due to wet or dry years, which may not match with the incidence of production and opportunity costs. Other important sources of variations in pollutant estimates are that the efficiency of practice changes in generating pollutant reductions are assumed constant over time, despite limited research to justify this (Liu et al., 2017).

5.4. Data adjustments and selection of transfer method

Johnston et al. (2020) highlight that the transfer method should be selected based on: (a) data availability, (b) steps required to harmonize study site estimates with policy site conditions, and (c) the intended uses

Table 1
Results from review of cost-effectiveness studies and parameters.

	Scale	Pollutants	Landuse	Mechanisms	Management	Data	Methodology	Time Frame (yrs)	Discount rate	Observations
Costs only										
Eccels and Star et al. (2015)	Paddock Scale	DIN	Sugarcane	Incentives	Makay/ Whitsunday	C-B	Investment Analysis	10	7	1
Poggio and Page (2010)	Paddock Scale	DIN	Sugarcane	Incentives	Burdekin Delta	ABCD	Investment Analysis	10	7	1
Poggio and Page (2010)	Paddock Scale	DIN	Sugarcane	Incentives	Burdekin BRIA	ABCD	Investment Analysis	10	7	1
East and Star (2010)	Paddock Scale	DIN	Sugarcane	Incentives	Mackay/ Whitsunday	ABCD	Investment Analysis	10	6	1
Law et al. (2016)	Paddock Scale	DIN	Sugarcane	Incentives	Mackay/ Whitsunday	C-B	Investment Analysis	10	6	1
East and Star (2010)	Paddock Scale	TSS	Grazing	Incentives	Mackay/ Whitsunday	ABCD	Investment Analysis	20	5	2
Bass et al. (2013)	Paddock Scale	DIN,PS11	Sugarcane	Incentives	P2R Management Practices 2014	ABCD	Index Approach	1	N/A	
Prioritisation										
Alluvium (2016)	Catchment	DIN, FSS	Sugarcane, grazing, gully streambank, wetland, land use change,urban storm water management	Incentives, Extension, Regulation	P2R Management Practices 2014	Secondary -Derived from other studies	Bio-economic modelling	10	7	94
Alluvium (2019).	Catchment	DIN, FSS	Sugarcane, grazing, streambank, gully, land use change, bananas	Incentives	P2R Management Practices 2018	Secondary - Modelled	Bio-economic modelling	5, 15,30	7	98
Star et al. (2016b)	Neighbourhood catchment	TSS	Grazing, gully, streambank	Incentives, Extension	P2R Management Practices 2014	Primary -past projects and Secondary-modelled	Bioeconomic Modelling	5	7	192
Star et al. (2013)	Paddock Scale	TSS	Grazing	Incentives Opportunity costs	Land condition	Secondary- GRASP Modelled	Bioeconomic Modelling	20	6	360
Beher et al. (2016)	Paddock Scale	TSS	Sugarcane, grazing	Incentives (opportunity costs not accounted)	P2R Management Practices 2014	Primary	Bioeconomic modelling	1	0	296
Beverly et al. (2016)	Paddock Scale	TSS,DIN, PS11	Sugarcane, grazing	Incentives, Extension,	P2R Management Practices 2014	Secondary	Bioeconomic modelling	10	6	
Evaluation										
Rolfe et al. (2018)	Paddock Scale	TSS, PSII, DIN	Sugarcane	Incentive, Extension	P2R Management Practices 2014	Primary	Index Approach CE	1	N/A	337
Rust and Star (2018)	Paddock Scale	TSS	Grazing	Incentives	Specific Remediation approaches	Primary	Investment Analysis Bioeconomic modelling	10	7	6
Star and Donaghy (2010)	Paddock Scale	TSS	Grazing	Inventive	Land Condition	Secondary -GRASP Modelling	Bioeconomic Modelling	20	5	960
van Grieken et al. (2014)	Paddock Scale	DIN	Sugarcane	Incentives	P2R Management Practices 2014	Secondary - Modelled	Bioeconomic Modelling	10	6	432
Whitten et al. (2015)	Sub-Catchment Scale	DIN	Sugarcane	Regulation Incentives, Extension	P2R Management Practices 2014	Secondary - Modelling	Bioeconomic Modelling	7	6	63
Star et al. (2015)	Sub-Catchment Scale	TSS	Grazing	Incentives, Extension	Land Condition	Secondary - Modelled	Bioeconomic Modelling	7	6	411
Smith (2015)	Paddock Scale	TSS, PSII, DIN	Sugarcane	Incentives, extension	P2R Management Practices 2014	Primary	Bioeconomic Modelling	7	10	910

Table 2
Differences between cost-effective studies for sediment reductions.

Study	Discount rate & time (yrs)	Cost-effectiveness measure	Practice assessed	Costs included	Theoretical framework
North Queensland Dry Tropics (2016)	Not stated, 20	$4.68/t	Remedial actions for gully recovery, managing risk of erosion associated with linear features.	Capital, maintenance, opportunity	Producer surplus
Rust and Star (2018)	7%, 10	$652.44/t/year	Remedial actions for gully recovery (40%), managing risk of erosion associated with linear features (30%)	Capital, opportunity, maintenance,	Mixed- Producer Surplus (opportunity cost) and consumer surplus(grant funds)
Alluvium (2016)	7%, 10	$268/t	Changing management practice	Capital, opportunity, program	Mixed- Producer Surplus and consumer surplus (grant funds)
Rolfe and Windle (2016)	6%–7%, 5–20	$259/t	Changing management practice	Capital and program	Consumer Surplus
Star et al. (2013)	6%, 20	$4/t to $421/t/year	Land condition and stocking rate	Capital, opportunity	Mixed- Producer Surplus
Star and Donaghy (2010)	6%, 20	-$835.08/t - $25,594.46/t	Land condition and stocking rate	Capital, opportunity, extension, program.	Mixed- Producer Surplus and consumer surplus (grant funds)
Star et al. (2018)	7%, 7	$3.09/t-$2398/t	Sugarcane and grazing all sub-catchments all management level changes.	Capital, opportunity, maintenance,	Mixed- Producer Surplus and consumer surplus (grant funds)
Alluvium (2019)	7%, 5	$28/t-$492,674/t	Sugarcane, bananas, grazing all sub-catchments all management level changes.	Capital, opportunity, extension, program.	Mixed- Producer Surplus and consumer surplus (grant funds)

of the resulting information. Step (c) is, in part, why there are such different approaches to ranges and approaches to studies completed in the GBR to date.

Johnston et al. (2020) suggest the following questions be considered in the transfer of values. How similar are study and policy sites across relevant dimensions? Is there a single study that provides information sufficient to transfer the study site estimate directly, perhaps with adjustments (e.g. income), to the policy site? Alternatively, are there several study sites providing values that, when averaged and perhaps adjusted, provide an accurate and credible estimate of the policy site value? Does the weight of evidence on site similarity support a value transfer over other transfer methods that allow for greater calibration of study site estimates to match policy site conditions?

Prioritisation studies such as van Grieken et al. (2014), Whitten et al. (2015), Alluvium (2016, 2019) and Star et al. (2018) have generally calculated marginal abatement cost curves in order to summarise and prioritise the information from multiple transfers. However, there is a significant variation in the quality and treatment of the data that underlies these cost summaries. Van Grieken et al. (2014) essentially generate all their cost-effectiveness estimates from their own studies through a mix of farm economic analysis and detailed farm modelling and then summarise these into marginal abatement cost curves. In contrast, Alluvium (2016) use transfer approaches to harvest values from other primary studies and then combine them in a modelling framework at a sub-catchment scale to generate predicted marginal abatement cost curves.

Cost heterogeneity refers to the variation in costs that are presented in studies. A notable outcome of the assessment is that most studies report large variations in cost-effectiveness. For example, van Grieken et al. (2014) reported that farm enterprise models showed that DIN abatement costs for medium-sized cane farms in the Mackay Whitsunday region ranged across multiple potential management changes from -$7.90/kg to +$16.70/kg. Rolfe et al. (2018) analysed the cost-effectiveness generated by Reef Rescue grants, showing that while the first quartile of grants generated DIN reductions for $1.17/kg, costs were much higher for the second, third and fourth quartiles at $19, $55 and $203/kg, respectively.

Some of the heterogeneity in estimated cost-effectiveness stem from underlying variation in the management actions considered and the size of the enterprise involved. This is demonstrated in Table 3, where factors underpinning the studies analysing the costs of changing from C to B nutrient management categories are summarised, and then cost-effectiveness estimates are reported in Fig. 4. There is a large variation in the cost estimates, but it is difficult to identify the extent to which these are driven by variations between management actions, enterprises or regions and where the data adjustments based on the number of study sites influence the range in costs.

5.5. Auxiliary data, data analysis and robustness analyses

Auxiliary data that is consistently available across study sites and that can enhance transfer accuracy should be used when available (Johnston et al., 2020). Auxiliary data of this type is frequently derived from GIS data layers on a myriad of environmental, landscape and population characteristics, thereby providing a source of consistently measured and often quality-controlled information that may be applied to all observations in the metadata.

Any overall assessment of water quality improvement costs for the GBR relies on the transfer and reuse of cost estimates simply because of the small number and limited diversity of primary studies. However, there are likely to be significant errors involved in both the cost transfer and cost aggregation stages, which are currently not assessed in these studies. Some indication of the extent of those potential errors can be shown by comparing the results of Star et al. (2018), and Alluvium (2019) for sediment reductions across the GBR as different auxiliary data has been employed for risks, large effects, and marine exposure

Table 3
Studies assessing the private (landholder) costs and benefits of shifting from medium risk (C) management to low risk (B) nutrient management.

Catchment	Primary source	Capital equipment assessed for changing management	Farm size (ha)
Wet Tropics	Whitten et al. (2015)	Most farmers would already have suitable equipment although there would be a capital cost of approximately $10,000 for those who did not, along with some reduction in fertiliser application	Small (<100 ha), medium (100 ha-200 ha) and large (>200 ha)
Wet Tropics	Terrain (2015)	Stool splitter fertiliser box * costing based on van Grieken et al., 2014	Small (<100 ha), medium (100-250 ha); and large(250 ha)
Wet Tropics	Rolfe and Windle (2016)	Modification of variable rate stool splitting sub-surface fertiliser applicator	Small (150 ha), medium (250 ha), large (930 ha)
Wet Tropics	Catalyst Growers Forum (2015)	Modify stool split fertiliser box	A 120 ha grower
Wet Tropics	Van Grieken et al. (2014)* capital items costed 2012	Stool splitter fertiliser box, harvester modifications	Small (<100 ha), medium (100 ha-200 ha) and large (>200 ha)
Burdekin	Smith (2015)	Zonal ripper/rotary hoe; wavy discs; double-disc open planter (stool splitter fertiliser box); GPS, flow rate monitor, harvester modifications	BRIA maximum up to 3500 ha, average 140 ha and median 94 ha and the delta is max 2000 ha, average 98 ha, and median 83 ha
Burdekin	Rolfe and Windle (2016)	Wavy disc cultuers, GPS, bed former, variable rate fertiliser box	Small 30 ha, medium 297 ha, large 1059 ha
Burdekin	Whitten et al. (2015)	We understand that most farmers would already have suitable equipment although there would be a capital cost of approximately $10,000 for those who did not, along with some reduction in fertiliser application	Small (<100 ha), medium (100 ha-200 ha) and large (>200 ha)
Burdekin	Poggio and Page (2010)	Stool splitter fertiliser box	Farm size 120 ha
Burdekin	Poggio and Page (2010) BRIA	Stool splitter fertiliser box bed renovator	Farm size 240 ha
Burdekin	Van Grieken et al. (2014)	Stool splitter fertiliser box, harvester modifications	Not actually specified assumed small (<100 ha), medium (100 ha-200 ha) and large (>200 ha)
Burnett Mary	van Grieken et al. (2014)	Change fertiliser box and tillage equipment, zonal till implements	75 ha, 125 ha, 250 ha
Mackay	Rolfe and Windle (2016)	Nutrient management plans, variable rate controller	42 ha, 226 ha, 490 ha
Mackay	East et al. (2012)	Bed renovator, GPS, modification to double disc planter.	240 ha
Mackay	East et al. (2011)	Variable rate controller	50 ha,150 ha and 300 ha
Mackay	Law et al. (2016)	GPS, bed renovator and ripper, rate controller, SMS software, widen existing equipment	150 ha

(Table 4). Both studies assess the same actions and rely on cost transfer to source multiple primary studies, but then have different value adjustment processes and incorporate different auxiliary data. The resulting cost estimates are very different, with the results from Star et al. (2018) five times higher than those from Alluvium (2019).

The incorporation with supplementary data can help explain the variation in costs noted on project sites in the GBR (Beher et al., 2016) and highlight why targets may take longer to achieve than the policy guidelines or why adoption levels may be lower than expected (Star et al., 2019). These aspects are critical for consideration in value transfers.

5.6. Aggregation and reporting

The key aspect of the reporting is the documentation of all key components of the transfer exercise. This should include reporting on the key study and policy site characteristics, data used in the transfer, transfer procedures, analyst assumptions and resulting value predictions in line with a credible scientific investigation (Johnston et al., 2020). This has been completed inconsistency across studies in the GBR; with some approaches aggregating to adoption levels, others have aggregated to pollutant reduction targets. This results in confusion regarding the value definition, the theoretical framework, the time period and per unit change.

6. Discussion and conclusion

This paper has reviewed the availability of studies assessing the costs of generating water quality improvements into the Great Barrier Reef. These are normally presented as the cost-effectiveness of pollutant reductions, with some evaluation and prioritisation studies summarising a variety of cost-effectiveness estimates into marginal abatement cost curves. In many cases, the prioritisation studies do not report any primary assessments of cost-effectiveness, instead of harvesting and extrapolating data from other primary costing studies through benefit transfer processes. We reviewed the relevant literature in the context of benefit transfer processes to highlight an improved approach and summarised several key findings as follows.

First, the number of primary studies is limited. While some studies report a large number of cost estimates across different factors such as management changes, farm sizes and catchments, there are also a number of scenarios where no cost estimates are available. This limits the understanding of both landholder costs and public costs and therefore the ability to analyse either the private or public trade-offs is missing. This limited number of primary studies results in challenges to transfer cost values for prioritisation across catchments. Poor evaluation also hinders an improved understanding of maintenance costs time frames, effectiveness, lag-effects and risks and uncertainties impacting the cost-effectiveness and limiting future prioritisation and program design.

These risks and uncertainty are often not reported in studies. Evaluation of past projects and programs would allow an understanding of their role on both the costs and effectiveness, providing an improved understanding of scale impacts and allowing a more diverse collection of primary studies to provide insights in the range of costs and outcomes of different treatments based on different biophysical and management approaches. This would also provide more context regarding the value definitions and adjustments required to transfer values for prioritisation.

Second, there is limited consistency in the way that different estimates of cost-effectiveness are constructed. There are many variations in the type of costs included if it is based on consumer or producer surplus, the calculation of pollutant reductions, and the adjustments made to account for factors such as project efficiency, time lags and delivery rates. The interactions between these variables are critical for further program design and project evaluation. These issues mean that care needs to be taken in comparing cost estimates between studies. Better

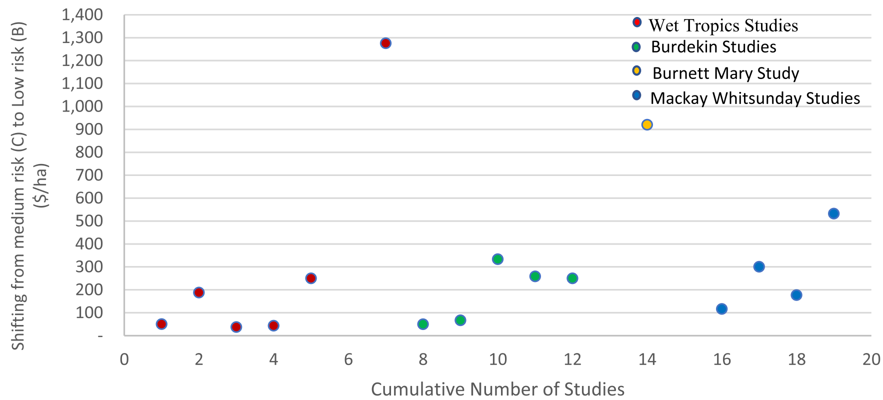

Fig. 4. Variation across NRM regions in paddock scale capital costs per hectare for shifting from medium risk (C) to low risk (B) nutrient management.

Table 4
Different data set inputs used in prioritisation at a sub-catchment level.

	Ground cover	Adoption and participation	Risks	Lag effects	Costs	Pollution loads and efficacy for remediation	Marine exposure
Star et al., 2018	✓	✓	✓	✓	✓	✓	✓
Alluvium, 2019		✓			✓	✓	

protocols about calculation of cost-effectiveness would limit this variability and provide a framework for further program assessments and design.

Third, the processes used to reuse and extrapolate primary studies to fill data gaps in policy sites are very poorly documented. There are likely to be significant errors involved in transferring and extrapolating cost estimates from primary and paddock scale studies to regional or whole GBR levels, and over different time periods to achieve different objectives, but currently, these are not recognised or understood. The comparison of values across multiple studies confirms that there are considerable variations in the actions to improve water quality. An important driver of this cost heterogeneity, apart from differences in projects and biophysical processes, are highly varied impacts on farm productivity and opportunity costs. Therefore, a better understanding of cost-effectiveness is important to improving prioritisation processes and program efficiency and increasing the interest of farmers and adoption rates for improved management practices.

In conclusion, this review has provided clear consideration for future studies in capturing the cost elements that are required for the different uses at a primary study level. Capturing these and transparently reporting them in primary studies allows for cost transfers to be applied more accurately by the analyst and to understand the value definition, capacity to make data adjustments, use of auxiliary data and approach to aggregate and report the analysis. Further work is required to develop a framework for cost-effectiveness in across the GBR catchments, this application would be required to be developed in conjunction with policymakers, which is robust, transparent and rigorous. The framework would also have to consider the different benefits and further time lags for reef health as opposed to only water quality outcomes these are required to be identified and considered in future studies, and how dual pollution sources would be dealt with in the framework (Balana et al., 2011).

Declaration of competing interest

We wish to confirm that there are no known conflicts of interest associated with this publication and there has been no significant financial support for this work that could have influenced its outcome.

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