

Benchmarking costs of improving agricultural water management in GBR catchments

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ACRONYMS

DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved inorganic phosphorus
DOE	Department of the Environment
EBI	Environmental Benefits Index
GBR	Great Barrier Reef
NESP	National Environmental Science Programme
NRM	Natural Resource Management
PN	Particulate nitrogen
PP	Particulate phosphorus
RRRC	Reef and Rainforest Research Centre Limited
TSS	Total suspended solids
TWQ	Tropical Water Quality
WQIP	Water Quality Improvement Plan

ABBREVIATIONS

Aust. Govt.....	Australian Government
kg	kilogram
t	tonne
yr	year

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EXECUTIVE SUMMARY

- Estimates of cost-effectiveness to reduce key pollutants into the Great Barrier Reef has been collated from a number of sources, including evaluations of Reef Rescue grants, water quality tenders, water quality improvement plans and bioeconomic modelling.
- Cost effectiveness is a key criteria for judging both the performance of past investments and guiding the allocation of future investments. A simple definition of cost-effectiveness is that it is the ratio of water quality improvements (such as reductions in sediment or nutrient loads) to the costs of achieving the change.
- Some estimates of cost-effectiveness are not directly comparable. Key differences include whether (a) emissions are estimated at farm boundary or end-of-catchment, and (b) whether costs are estimated as changes in net farm enterprise returns or also include indirect and involvement costs.
- There is very limited data available on cost-effectiveness of Reef Rescue grant programs, even though returns on investment should have been key criteria for funding allocations. In part this is because modelling information to predict pollutant reduction has been limited for NRM groups when allocating project funds.
- Results show substantial variation (heterogeneity) in cost estimates, even after allowing for systematic differences in the estimation of both costs and emissions. This confirms that cost-effectiveness should be a key criteria for project prioritisation and funding evaluation.
 - At the NRM/pollutant level, variations in cost-effectiveness provide some indication about where overall funds can be better invested.
 - At the farm/project level scale, variations mean that there are substantial benefits in prioritising projects within a regional area.
- The summary estimates at the NRM/pollutant level have been reported in two blocks, depending on whether loads are modelled at end-of-paddock or end-of-catchment. The latter costs are more expensive, in part because of the transmission losses. Costs may be also higher because of program design factors (reliance on grant programs).
- Estimates of cost effectiveness at end-of-paddock are much lower. This is in part because water quality tenders may have selected the lowest cost projects, and some estimates of pollution reductions may be less accurate. However these low estimates match results of bioeconomic models that indicate many improvements can occur at very low cost.
- The estimated costs are averages, so many higher cost projects have been funded within programs. There will be more variation at the farm or project level.
- Benchmarks for cost-effectiveness are a mechanism to set thresholds or caps for funding. Approximate cost-effectiveness thresholds can be set at the average of achieved and predicted costs for end-of-catchment loads, as follows :
 - Sediment: \$259/tonne
 - Nitrogen (DIN): \$150/kg
 - Pesticide (PSII): \$8,351/kg
- These benchmarks should be treated as average thresholds for allocating project funds, both between and within regions. It is likely that lower cost projects can be achieved through more targeted program design and better assessment of cost-effectiveness. As both modelling and cost information improves, more accurate benchmarks can be established.

- The summary below provides a guide to the cost-effectiveness of various schemes by NRM group and regional area. Estimates need to be treated with some caution because the way that costs have been estimated and pollution reductions modelled are not necessarily consistent across regions and programs, and over time.

Region	Sediment \$/tonne	Nitrogen \$/kg	Nitrogen (DIN) \$/kg	Pesticide (PSII) \$/kg
End of catchment loads				
Whole GBR				
Achieved: Aust Govt 2014	130		63	3,500
<i>Predicted: Reef Regions 2015</i>	<i>61</i>	<i>21</i>	<i>34</i>	<i>3,945</i>
Cape York				
Achieved: Aust Govt 2014	297	192		
Wet tropics				
Achieved: Aust Govt 2014	375	47	142	16,343
Achieved: Bass et al. 2013	255		438	22,886
<i>Predicted: WQIP(70:30 funding split)</i>			<i>239</i>	<i>3,076</i>
Burdekin				
Achieved: Aust Govt 2014	106	62	124	120,770
Mackay Whitsunday				
Achieved: Aust Govt 2014	987	164	157	19,315
<i>Predicted: WQIP</i>	<i>263</i>		<i>68</i>	<i>10,178</i>
Fitzroy				
Achieved: Aust Govt 2014	513	1,494		1,060,650
<i>Predicted: Star et al. 2015b WQIP</i>	<i>277</i>			
Burnett Mary				
Achieved: Aust Govt 2014	1,343	192		38,153
<i>Predicted: WQIP (50:35:15 funding split)</i>	<i>237</i>		<i>142</i>	<i>11,918</i>
Whole GBR				
Recommended Benchmark	\$259/t		\$150/kg	\$8,351/kg
End of paddock loads				
Wet tropics				
Achieved: Bass et al. 2013	25	141		667
Burdekin				
Achieved: Burdekin Tender 2011	104	5		1,976

- It is recommended that:
 - Tools to model pollution changes at the project selection point be developed
 - Data on cost-effectiveness should be automatically collated at the project level when predictions of improvements are made and funding is allocated.
 - There should be greater emphasis on cost-effectiveness in project selection.
 - Program design should be adjusted away from a reliance on simple grant mechanisms to processes that optimise price discovery and project selection.
 - Benchmarks be applied in program funding to set limits on program and project funding levels, and help in prioritisation of projects.
 - Benchmarks be further developed to identify lower, average and upper ranges for expected cost-effectiveness benchmarks.

1.0 INTRODUCTION

The Australian and Queensland Governments have allocated significant public funds to improve water quality from agricultural land management in catchments draining into the Great Barrier Reef (GBR) lagoon. Over the past two decades the funds have been allocated across a range of different programs and initiatives, largely through the six Natural Resource Management (NRM) regional bodies. Reef Plan Targets (Australian and Queensland Governments 2013) indicate the following water quality reduction targets to be achieved by 2018.

- At least a 50% reduction in anthropogenic loads end-of-catchment dissolved inorganic nitrogen loads in priority areas;
- At least a 20% reduction in anthropogenic end-of-catchment loads of sediment and articulate nutrients in priority areas; and
- At least a 60% reduction in end-of-catchment pesticide loads in priority areas

Cost effectiveness is a key criteria for judging both the performance of past investments and guiding the allocation of future investments (Pannell 2015). A simple definition of cost-effectiveness is that it is the ratio of water quality improvements (such as reductions in sediment or nutrient loads) to the costs of achieving the change. More cost-effective investments generate larger improvements per allocated dollar of investment. Both Governments are exploring new investment approaches to generate more targeted and effective outcomes, through the creation of programs such as Reef Trust.

There are two key reasons why it is important to identify the cost-effectiveness of investments to improve agricultural water quality:

- It is a key criteria to prioritise between different proposals to improve water quality,
- It can demonstrate the value and efficiency of investment of public funds.

Some higher level estimates of cost-effectiveness have been included in the design and assessments of programs, where the total funding allocated to different programs and NRM groups have been matched with the projected reduction in pollutant emissions. Much of this work has been summarised in the GBR report cards and in submissions for funding to the Australian and Queensland Governments, where projected program costs and outputs are summarised.

However there has been limited evidence to date that funds allocation in the GBR has been cost-effective. Initial allocations of funds to programs were driven by science priorities and expert opinion, with multi-criteria analysis used in some cases (Australian and Queensland Governments 2013), while programs have mostly allocated funds through fixed rate grants. Pannell and Roberts (2010) raised concerns that these types of arrangements would lead to inefficient allocations of funds, while at the program level Rolfe and Windle (2011a) demonstrated for the Burdekin catchment that traditional grants mechanisms had the potential to be inefficient.

The complexity of the GBR system makes prioritisation and evaluation very difficult. Poor water quality from agriculture is an externality problem characterised by asymmetric and incomplete information, and complicated by multiple pollutants, a focus on input measures

rather than environmental outcomes, and a complex set of drivers for farmers to make practice changes (Brodie et al. 2013).

There has been substantial attention paid to reporting the outcomes of various GBR programs. The health of the reef is now summarised annually in a Great Barrier Reef Report Card, which also includes estimates of improvements in land management in catchments and the modelling reductions in pollution loads across catchments (GBR Report Card 2014). The outcomes of funding programs such as the Reef Rescue program have been reported by the NRM bodies and the Australian and Queensland governments (e.g. Bass et al. 2013; Australian Government 2014). In addition returns on investment and cost-effectiveness are becoming more explicit in planning for new programs and funding allocations (e.g. Addison and Walshe 2015), in modelling the tradeoffs that farmers face to adopt new management practices (e.g. Star et al. 2013, van Greiken et al. 2013), or from limited trials of water quality auctions and reverse tenders (e.g. Rolfe and Windle 2011b).

A key challenge is to reconcile the differences in cost-effectiveness that can be measured from various programs and modelling. Estimates of cost-effectiveness can be systematically different for several reasons:

- The environmental benefits may not be consistent
- The assumptions about future farm management and practice adoption may vary
- Different components of cost may be included
- Accuracy of estimates for either the costs or environmental benefits may vary
- The mechanisms to select projects or programs for inclusion or funding may differ.

This means that it is often difficult to determine whether variations in the cost-effectiveness of programs and projects are related to variations in efficiency or systematic differences in the way that cost-effectiveness has been assessed.

The purpose of this report is to summarise and evaluate available data on the cost-effectiveness of improvements in agricultural water quality within GBR catchments, identifying how differences in estimates might be reconciled. A key aim is to provide benchmarks on cost-effectiveness across the key pollutants to help prioritise future investments in water quality.

The report is structured as follows. A summary of key concepts relating to cost-effectiveness is provided in the next section, followed by a review of historic data in the next two sections: the Reef Rescue grants in section 3 and competitive tenders in section 4. This is followed by a summary of predictive modelling: the regional Water Quality Improvement Plans are covered in section 5, and the most important of the bioeconomic models in section 6. Final conclusions follow in section 7.

2.0 CONCEPTS

Cost effectiveness in the context of agricultural water quality improvements in the GBR is the ratio of water quality improvements (such as reductions in sediment or nutrient loads) to the costs of achieving the change. There is normally significant variation (heterogeneity) in the costs of making water quality improvements. This variation in cost effectiveness occurs across different scales and parameters because both costs and emissions are driven by a number of factors that vary between and across catchments, between farms, and within farming operations. This is demonstrated in the figure below, where potential projects / regions are ranked from lowest to highest cost effectiveness, and the area under the cost line represents the costs of achieving the actions. If there is significant variation in costs, prioritisation and selection of projects limit the average costs of improvements and generate larger environmental benefits for a given budget allocation.

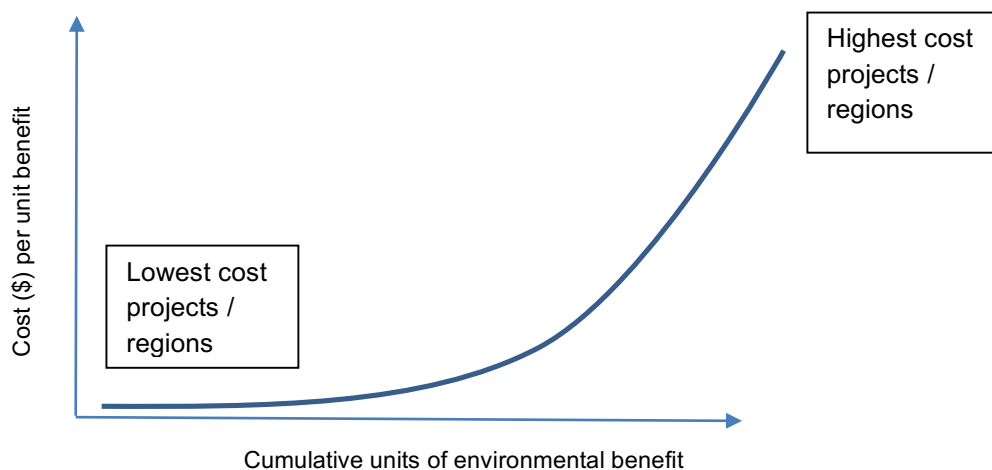


Figure 1: Heterogeneity in cost-effectiveness

However there are a number of complexities in assessing cost effectiveness. These include:

- The types of benefits involved
- The scale at which benefits are assessed
- Missing or limited information
- Challenges in modelling or measuring both environmental and cost variables.

Measures of cost-effectiveness focus closely on the damaging environmental **outputs** (e.g. total nitrogen emissions) or **outcomes** (e.g. area of healthy reef) rather than **inputs** (e.g. change in farm machinery) which are the focus of most existing approaches. They are typically structured to take account of marginal changes, such as the **additional** costs and reduced pollution emissions associated with a farm management change, rather than the **total** or **average** farm management costs and pollutant emissions. Many grant programs do not assess benefits in these ways, focusing on changes in inputs and grouping farm management into A,B,C,D classifications, making it difficult to assess exactly what improvements in water quality will result.

Problems of information asymmetry also limit information about cost effectiveness. Farming operations are private enterprises, so the information about the costs of making enterprise changes to improve management actions are available to the enterprise managers, not to government and agencies. Similarly, information about the emissions generated on-farm and transferred into waterways and GBR lagoon is typically not available to farmers. Incomplete information about both the costs of enterprise changes and the environmental benefits generated have largely been responsible for a focus on input mechanisms and practice changes that maintain productivity.

There are several approaches available to estimating the costs of a farm management change:

- Farm production models that estimate changes in revenue, costs and profits with management / activity changes, typically based on data of multiple farm operations,
- Incentives paid or required by farmers to make the necessary management changes,
- Expert opinion about the average cost within a sector or industry to make changes.

The modelling and expert opinion approaches typically estimate costs for an average farm operation, so limit heterogeneity in cost estimates. In contrast, information about the incentives paid to farmers, particularly those revealed through competitive tenders, are more accurate in identifying the variation in cost tradeoffs.

In a similar way, there are different approaches to estimating pollutant emissions associated with different management changes:

- Direct measurement: challenging because of logistics and expense, the difficulties of identifying marginal changes, and the variations in climate and other factors.
- Simple models: these are often used to estimate emissions at the farm level where most factors can be held constant.
- Catchment models: these are necessary to make 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 catchment (rather than end of farm).

These models are typically for an average land type and farming operation within a catchment area, which have the effect that they limit heterogeneity in estimates of off-farm emissions. As well, the estimates can be for changes in pollutant loads at the farm level or at the end-of-catchment (delivery point to the GBR lagoon).

The cost-effectiveness of management changes can be estimated through bioeconomic models, which link farm production models with pollutant generation and delivery models. These models have the advantage that the interactions between key variables and non-linear effects can be accounted for, but mostly report average costs of emission reductions. However by running bioeconomic models at different scales and enterprise changes, it is possible to generate a cross-section of the potential cost effectiveness of management changes (e.g. van Grieken et al. 2010, 2014).

Modelling and assessment are further complicated by the jointness of many pollutants and corresponding management actions. Assessment and reduction measures are focused on the following key pollutants:

- Total suspended solids (TSS), with increasing focus on the fines within this group
- Dissolved inorganic nitrogen (DIN)
- Particulate nitrogen (PN)
- Dissolved inorganic phosphorus (DIP)
- Particulate phosphorus (PP)
- PSII herbicides (PSII)

Agricultural activities and changes in management actions will often affect multiple pollutants (e.g. sediments and phosphorus are closely linked), so modelling needs to take account of these complexities. In some cases environmental benefits are summarised together into an Environmental Benefits Index (EBI) so that the costs of action can be compared to the 'package' of environmental improvements. One example of this approach being applied in a water quality tender in the Burdekin is provided in Rolfe et al. (2011). Alternatively, and more commonly, cost-effectiveness is modelled for each pollutant separately.

Predictions of changes in emissions are further complicated by discharge through groundwater in some catchments and the challenges of estimating transmission factors when there are variations in stream length, frequency and intensity of flow events, and barriers such as recycle pits, weirs and dams. Bioeconomic models often require pollutant generation functions to be linked with models of water flows, pollutant transmission and interception functions to provide accurate estimates of the net changes and delivery of pollutants, particularly to end-of-catchment scenarios.

Estimates of environmental benefits are particularly challenging for programs focused on changing farm inputs, such as grants to purchase farm machinery. Estimates of the environmental benefits rely on a series of assumptions about how the inputs translate to outputs. Typically these assumptions are about:

- The rate of improvement generated by an input
- The area of the farm to which it is applied
- The length of time that the improvement occurs
- The propensity of the landholder to adopt further improvements.

It is not only the environmental factors where predictions vary across several dimensions; costs also vary depending on the scope of the tradeoffs being considered. The simplest types of cost estimates focus on changes in enterprise profits or returns (private operating costs). These are often generated by farm production models that focus on partial economic analysis; e.g. changes in gross margins can be estimated by changing the variable costs and production of an operation, holding most factors such as capital, prices and fixed costs constant. Most bio-economic models (e.g. Star et al. 2011, van Greiken et al. 2014) generate estimates of private costs that focus on changes in net economic surplus (net farm profit).

However the full costs of changes in management practices may include additional private and public costs, particularly where management changes are being targeted through programs. These may involve indirect or non-financial costs. Extra components of private costs may involve:

- Capital costs (where not included in the enterprise changes)

- Risk and uncertainty – associated with increased variability or risk of production returns with changes in management
- Transaction and administration costs – the costs of finding and organising new opportunities

Public costs of conducting a program to encourage changes in landholder management may include:

- Direct incentive payments
- Administration costs
- Other costs, such as monitoring and enforcement.

The challenges of estimating private costs is summarized in the figure below, where supply curve S1 represents the net change in enterprise production returns for new management practices, and supply curve S2 represents those same management practices with the additional indirect and involvement costs included. The diagrams show that:

- Costs of making changes increase with the quantity of desired changes
- Some management changes may involve almost zero costs in terms of net production returns
- Additional risk / transaction / administration costs increases the real or perceived private costs of landholders
- The increase in costs can be much higher if there are perceptions of risk or uncertainty in returns.

Farm production models and bioeconomic models tend to measure only net production costs (e.g. curve S1). However costs that are revealed by farmers' involvement in programs (e.g. grant mechanisms) or by farmers setting the cost of management changes (e.g. in water quality tenders) usually incorporate the full costs of involvement (e.g. curve S2). This means that estimates of costs generally vary according to whether they are modelled or revealed, as different components of costs are included.

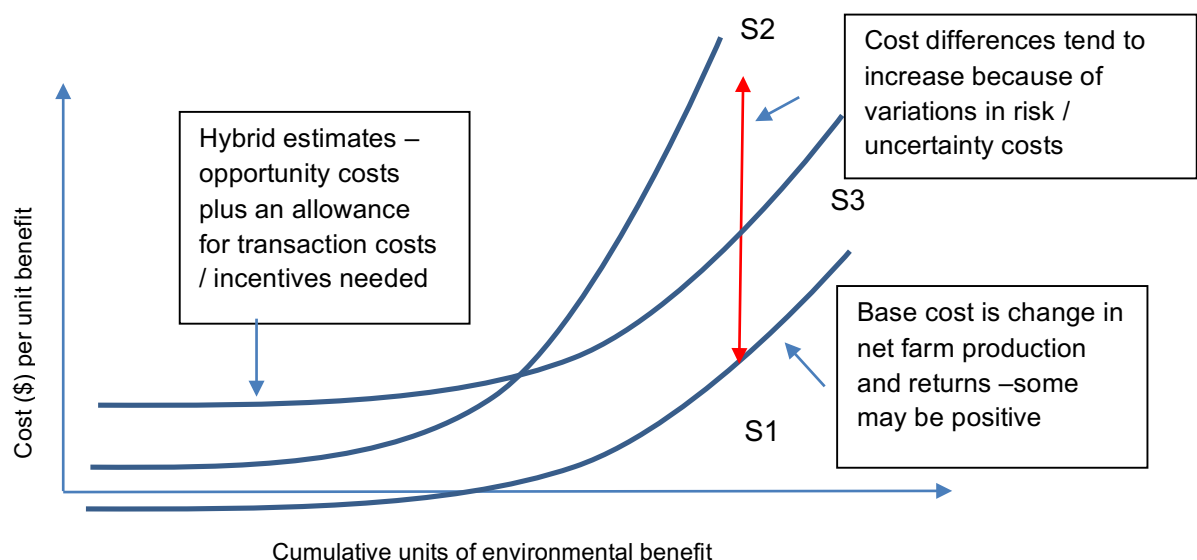


Figure 2: Components of private enterprise cost curves

Mechanisms to predict costs deal with the challenge of estimating indirect costs in different ways. Many grant programs estimate payment rates for actions on the average cost for a practice change and allowing for some nominal administration costs (e.g. Bass et al. 2013); these then become a flat-rate payment irrespective of underlying landholder costs. A similar approach was taken by Pannell (2015); low opportunity costs for practice change were supplemented with (large) allowances for the incentives needed to encourage landholders to change practices (\$25/ha/year). These approaches are represented in the diagram as curve S3 (a vertical shift of S1). A comparison shows that costs estimated with the hybrid approach (curve S3) will over-estimate costs for lower levels of environmental change and over-estimate costs for higher levels of change (full costs represented in S2).

Based on this analysis, a matrix of the key options to generate an estimate of cost-effectiveness from a site or regional area can be identified according to whether (a) emissions are estimated at farm boundary or end-of-catchment, (b) costs are estimated as changes in net farm enterprise returns, or also include all indirect and involvement costs, or some allowance to generate farmer involvement. While estimates may vary by other factors, such as the quality of the modelling and the scale of the environmental improvements involved, Table 1 summarises the key factors that limit the comparison of cost-effectiveness estimates in GBR catchments, and categorises the programs that are reviewed in the report by those key factors.

Table 1: Matrix of cost-effectiveness estimates by key factors

		Predictions of emissions	
		<i>On –farm</i>	<i>End-of-catchment</i>
Costs of action	<i>Operating costs (Curve S1)</i>	Bio-economic models (various)	Bio-economic models (various)
	<i>Operating + risk & transaction/admin. Costs (Curve S2)</i>	Water quality tenders (various)	
	<i>Operating costs + allowance for admin/incentives (Curve S3)</i>	Local Assessment (Terrain)	Reef Rescue grants WQIP (various)

3.0 PUBLIC FUNDING: GRANTS PROGRAMS 2008-2013

The Australian and Queensland Governments introduced the Reef Plan in 2003 to improve the quality of water entering the reef and the resilience of the reef to poor water quality (Queensland Audit Office 2015). There have been several programs and funding initiatives to achieve this, most notably after the Reef Plan was revised in 2009 to provide specific targets for best practice adoption in agriculture and reduced emission loads for key pollutants.

The Queensland Government committed \$175 million in a five year program to achieve the Reef Plan from 2009 to 2014. Activities have included the industry-led Best Management Practice programs, extension and education activities, catchment monitoring and modelling, and research. The Australian Government invested \$200 million in the Reef Rescue Programme (2008-2013) to reduce nutrient, pesticide and sediment discharge into the Great Barrier Reef from broadscale landuse. Funding included \$158 million to cover water quality grants for on-ground works and funding for developing industry partnerships and community participation. In 2015, each Government committed an additional \$100 million towards reef protection activities, with the funding still to be specifically committed.

A range of reports have summarized the results and achievements of the Reef Rescue Program, but few sources provide information about load reduction cost estimates where information is provided about both program costs as well as load reductions. For example, the Queensland Audit Office (2015) identified that a small component of Queensland Government expenditure had been on investment programs through the Regional NRM bodies (\$4.3 million in 2013-14), but that practice changes are poorly quantified.

The most informative data comes from the analysis of the Australian Government investment into water quality grants, as summarized in the following sub-sections. However the insights provided are limited. One problem is that the available information is limited to averages by NRM region. Ideally data would be available at an individual project level so that funding allocations could be matched to projected reductions in emissions. While there is potential for Paddock to Reef modelling to retrofit emissions predictions to individual grants, this data is not automatically available.

A second problem is that the Reef Rescue grant program, and NRM grants more generally, do not function well as a cost revelation mechanism. This is because the funding levels for particular activities tend to be set in advance through the program design, based either on some expert opinion about average costs of performing an activity and funding levels necessary to attract participation, or some level of cost-sharing. The outcome is that costs tend to be uniform for specific activities, rather than tailored to individual operations. The emphasis on the use of fixed grants to allocate funds has disguised the variations in private costs by landholders.

3.1 Reef Rescue Achievements 2008 – 2013 (Aust. Govt. 2014)

A summary of the Reef Rescue Programme achievements across the regions is provided by the Australian Government (2014). The report provides broad level estimates of the incurred cost of pollutant load reductions over the life of the program. Load reductions were estimated

from bio-physical modelling, conducted through the joint Australian and Queensland Government Paddock to Reef Programme, and based on the reported improvements in land management practices collected by the natural resource management (NRM) regions. The modelled Reef Rescue pollutant load reductions and total costs for the five year program were used to estimate approximate costs of sediment, dissolved inorganic nitrogen (DIN) and of photosystem II (PSII) pesticide reductions. The reported costs represented the approximate costs to the Australian Government for the annual average, **end-of-catchment**, load reduction, as follows:

- **Suspended (fine) sediment reduction:** approximately **\$130 per tonne per year** from the adoption of improved grazing land management practices.
- **Dissolved inorganic nitrogen (DIN) reduction:** approximately **\$63,000 per tonne per year** from the adoption of improved fertiliser management practices in sugarcane.
- **Photosystem II (PSII) inhibiting pesticides reduction:** approximately **\$3,500 per kilogram per year** from the adoption of improved pesticide management practices in sugarcane.

These estimates of cost-effectiveness are limited in several ways. First, it is unclear what the cost base is, i.e. total program costs (\$200 million), water quality grants (\$158 million) or some other variant. Second, not all benefits were included in the modelling, such as improvements from the horticulture and dairy industries. Third, the modelling is underpinned by a number of assumptions about the extent and duration of practice change, which may be over-optimistic; typically model estimates have become more conservative over time as model accuracy has improved¹. As well the assumptions may vary between regional groups, creating some variation in the estimates of pollution change. It is also important to note that the grants programs involved generally relied on a fixed cost or subsidy for a mechanism change plus some allowance for administration costs, so did not function as a price revelation system.

The regional level details provided in the program summary reveal considerable regional heterogeneity in the estimated costs of load reductions (Table 2). The cost of sediment reductions ranged from \$106 per tonne in the Burdekin to \$1,343 in the Burnett Mary region, more than a 10 fold increase. The cost of nitrogen reductions varied by a factor of more than 30 and pesticide reduction by a factor of 65.

Table 2: End-of-catchment load reductions and cost estimates by NRM region

	Funding	Sediment reduction			Nitrogen reduction			DIN reduction			Pesticide PSII reduction		
	2008-13	%	tons	\$/ton	%	tons	\$/kg	%	tons	\$/kg	%	kg	\$/kg
Cape York	4.05M	8	13,639	\$297	8	21	\$192						
Wet Tropic	33.70M	12	89,792	\$375	11	710	\$47	12	237	\$142	24	2,062	\$16,343
Burdekin	32.43M	12	306,334	\$106	9	525	\$62	14	261	\$124	13	269	\$120,770
Mky Whit	32.00M	9	32,421	\$987	11	195	\$164	24	204	\$157	42	1,657	\$19,315
Fitzroy	30.63M	4	59,717	\$513	2	21	\$1,494				5	29	\$1,060,650
Burn Mary	16.19M	3	12,050	\$1,343	6	84	\$192				28	424	\$38,153

Data sourced from Australian Government (2014).

¹ Comment by Kevin McCosker, Paddock to Reef program.

The costs of achieving a sediment reduction were much higher in the Mackay Whitsunday and Burnett Mary regions, where the main focus has been on nutrient reductions. The lowest cost provider was the Burdekin (Figure 3). The Wet Tropics and the Burdekin were the lowest cost providers of Nitrogen reduction (Figure 3), while the cost of DIN reductions was similar across the three reporting regions (Table 2). The Fitzroy region was not a low cost provider of any pollutant reduction and particularly not for pesticide reductions (Table 2), although these were not program priorities in the catchment.

There is an inconsistency between the estimates reported for the whole GBR by the Australian Government (2014) and the average of the NRM costs, even though approximately the same data should be involved:

- Sediments: whole GBR cost (\$130/t) < average NRM cost (\$290/t)
- Nitrogen: whole GBR cost (\$63/kg) < average NRM cost (\$96/kg)
- Pesticides: whole GBR cost (\$3,500/kg) < average NRM cost (\$48,465) (excludes Fitzroy)

It is likely that the cost data is similar in both the Whole GBR and NRM Region analysis, so it appears that the estimates of environmental benefits are much larger for the whole GBR.

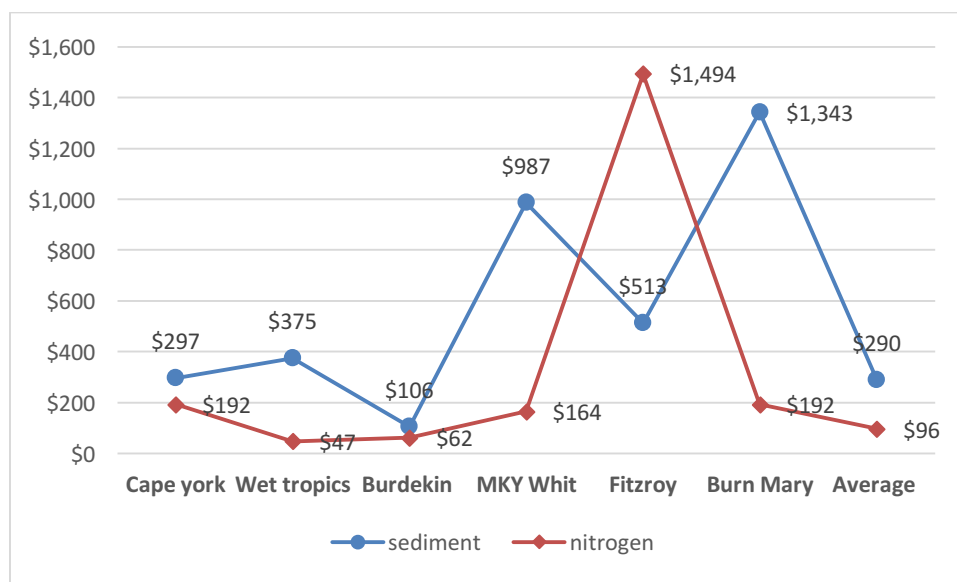


Figure 3: The average cost of sediment (\$/t) and nitrogen (\$/kg) reduction across the regions

3.2 Regional Assessment (Bass et al. 2013)

Terrain NRM conduct their own annual assessment to identify the impacts of the Reef Rescue grants in the Wet Tropics region. In the most recent summary, reporting on the 2011-12 period, Bass et al. (2013) apply the best secondary data estimates available to estimate the load reduction from the different type of activities funded across different industry sectors. However, in some cases load reductions estimates were relatively subjective and attributed to an input with an unpredictable output or outcome. The total **end of paddock** load reduction estimates are reported as 180,575 t of sediment; 315 t of nutrients and 6.9 t of

pesticide and **the end of catchment** load reductions as 18,057 t of sediment; 105 t DIN and 201 kg PSII herbicides.

In Bass et al. (2013) the cost of achieving these reductions (\$4.6 million) is more precisely defined (compared to the Reef Rescue Achievements report) as the water quality incentive grants to the agricultural industries: dairy, cane, wet and dry grazing, banana, papaw, multicrops, forestry and tree crops. The estimated costs of load reduction are outlined in Table 3 and compared to those reported for the region in Table 2 above. The variations are difficult to reconcile: the end-of-catchment estimates are essentially for the same program so estimates should be closer, while variations between end-of-paddock and end-of-catchment appear unrealistic (farms in the Wet Tropics region are very close to the coast so transmission rates should be high).

Table 3: Cost of pollutant reductions in the Wet Tropics region

	Sediment (\$/tonne)	Nitrogen (\$/kg)	DIN (\$/kg)	PSII (\$/kg)
End of paddock (Bass et al. 2013)	\$25	\$141		\$667
End of catchment (Bass et al. 2013)	\$255		\$438	\$22,886
End of catchment (Table 2 [Aus Gov 2014])	\$375		\$142	\$16,343

4.0 COST PREDICTIONS FOR FUTURE PROGRAMS

There has been increased attention on predicting the costs and pollution reductions from future grants programs. Concerned that current investment would not meet the Long Term sustainability Plan vision for a sustainable GBR region, Reef Regions (2015) developed an investment plan to provide a realistic estimate of abatement costs, based on nine years of experience and two generations of Water Quality Improvement Plans. It was estimated that continued improvement in agricultural management practices across the Great Barrier Reef would cost \$175 million for the period 2015 to 2020. It was predicted that such expenditure would result in the following **end-of-catchment** reductions (incorporating existing achievements (2008 – 2013):

- 15-20% reduction in total suspended solids
- 30-35% reduction in dissolved inorganic nitrogen
- 15-20% reduction in particulate nitrogen
- > 90% reduction in PSII herbicides

The approximate load reduction costs are outlined in Table , using data from the 2014 GBR Report Card to estimate annual loads. It is not possible to identify how the \$175M would be allocated between pollutants, so estimates are based on allocating \$175M equally across the three pollutant types (sediment, nutrients and pesticides), or split 50:35:15 (broadly in line with the Reef Catchments Water Quality Improvement Plan (WQIP)).

Table 4: Estimated end-of-catchment abatement costs for the whole GBR 2015-2020

	Sediment	Nitrogen	Nitrogen (DIN)	Herbicide (PSII)
Load reduction %	15-20%	15-20%	30-35%	90%
1% equivalent ¹	54,465.34t	162.24t	52.58t	164.28kg
Cost@\$58.3M per pollutant	\$61/t	\$21/kg	\$34/kg	\$3,945/kg
Cost split 50:35:15 by pollutant	\$91/t	\$22/kg	\$36/kg	\$1,775/kg

¹Based on information about end of catchment load reduction provided in the GBR Report Card, 2012-2013 (A&QG 2014) and Waters et al. (2014).

The estimates of future program costs in Reef Regions (2015) were based on details from Water Quality Improvement Plans prepared by each of the NRM groups, with reports available from the Wet Tropics (Cairns region), Reef Catchments (Mackay Whitsunday region), Fitzroy Basin and Mary Burnett. Estimates of both costs and emission reductions have been generated through a variety of approaches, including the use of bioeconomic modelling, expert opinion and value transfer.

There is considerable variation in the economic analysis applied in the development of the WQIPs and future funding cost estimates, however most provide estimates of the predicted costs of achieving change and the anticipated percentage change in pollutants at the end-of-catchment. The reported percentage reductions were converted into absolute quantities applying information from the Great Barrier Reef report cards and Waters et al. (2014). The resulting estimates of cost-effectiveness are summarised in Table 5; further details are available in Appendix 1.

Table 5: Summary of cost estimates for future funding (WQIPs)

Region	Sediment	Nitrogen	Nitrogen (DIN)	Pesticide (PSII)
End of catchment loads				
Whole GBR (Reef Regions 2015)				
Load reduction	15-20%	15-20%	30-35%	90%
Public Cost @ \$175M (equal allocations)	\$61/t	\$21/kg	\$34/kg	\$3,945/kg
Public Cost @ \$175M (50:35:15)	\$92/t	\$22/kg	\$32/kg	\$1,776/kg
Wet tropics				
Load reduction	na		19%	63%
Public Cost @ \$13M			\$239/kg	\$3,076/kg
Mackay Whitsunday				
Load reduction	15%		12%	15%
Public Cost @ \$46M inc extrn	\$263/t		\$68/kg	\$10,178/kg
Burnett Mary				
Load reduction	20%		50%	60%
\$32.5M split 50:35:15 by pollutant	\$238/t		\$142/kg	\$11,917/kg
Fitzroy				
Load reduction	20%			
Private cost	\$277/t			

Data sourced from Reef Regions (2015) and WQIPs.

Results show that cost estimates of emission reductions across the whole Great Barrier Reef (Reef Regions 2015) are systematically about three to five times lower than the average costs from the WQIPs, even though the information base should be the same.

- Sediments : whole GBR cost (\$92/t) < average WQIP cost (\$259/t)
- DIN : whole GBR cost (\$32/kg) < average WQIP cost (\$150/kg)
- Pesticides : whole GBR cost (\$1,776/kg) < average WQIP cost (\$8,390)

It is likely that the differences are generated by overly optimistic predictions of pollution reductions in Reef Regions (2015). The WQIPs are generally more conservative in modelling predictions, possibly because (a) modelling is becoming more accurate and (b) estimates of on-farm changes are becoming more conservative.

The costs estimates from both the past Reef Rescue grants and the future anticipated program costs are summarised in Figure 4 below (future costs are circled blue).

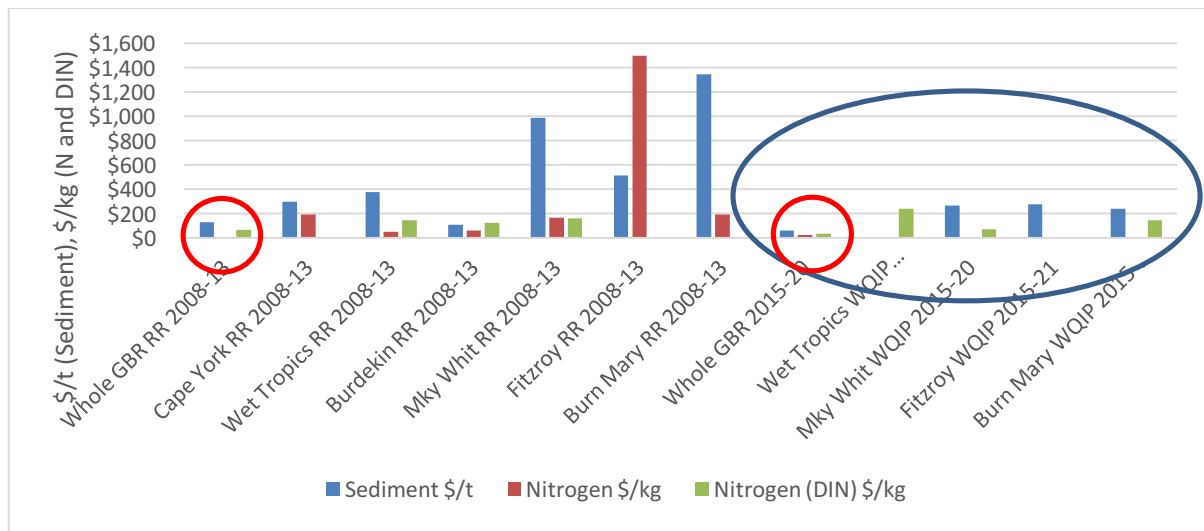


Figure 4: The average cost of sediment (\$/t) and nitrogen (\$/kg) reduction across programs

While comparisons are limited, they reveal that heterogeneity in costs is predicted to be lower in future programs than in past ones. Costs in future programs (circled blue) are predicted to be much lower than over past grants (average decreases are 64% in Sediment, 11% in DIN and 46% in PSII); this is despite more accurate modelling generally reducing predictions of pollution changes². The data also illustrates how estimates for costs across the whole GBR (circled red) are systematically lower than costs at the regional level, whether the estimates are for the 2008-13 Reef Rescue program or the 2015-20 program.

² Comments from Kevin McCosker (Paddock to Reef program)

5.0 ESTIMATING COSTS: WATER QUALITY AUCTIONS

Information about cost-effectiveness can also be sourced from water quality auctions (also known as competitive tenders, reverse auctions or procurement auctions). These are market based approaches where governments purposefully design allocation mechanisms for public funding, rather than allocating funds through flat-rate grants (Rolfe and Windle 2011a,b).

The intention is to engage the competitive power of market processes to improve the efficiency of public funding allocation. Rather than paying a fixed price for a fixed action, a tendering mechanism exploits the heterogeneity in both inputs and outputs. Landholders face different opportunity costs in providing environmental improvements, and the bio-physical outputs vary across the landscape. In a competitive tender, landholders nominate the management practice changes they are prepared to implement and the price they are prepared to accept to provide the changes.

Water quality auctions provide much more accurate information than grant mechanisms about the private costs of changing management practices, for two key reasons:

- Landholders reveal their individual incentives required, so costs discriminate between landholders, and are net of most private benefits that might be generated
- Landholder costs include all costs of involvement, include considerations of capital costs, risks and transaction costs.

However the estimates of environmental benefits tend to be less accurate, as the primary purpose of assessment is to rank proposals (time constraints often preclude the accurate predictions of emission changes). In most water quality tenders, proposals have been evaluated only in terms of end-of-paddock changes using quite simple models of the relationship between practice adoption and change in pollutants.

A number of competitive tenders, designed to achieve water quality improvements, have been implemented in Great Barrier Reef catchments and provide information about the revealed costs of emission reductions (Rolfe and Windle 2011b). The most relevant tenders was a water quality auction conducted in the Burdekin region in 2008 (Rolfe et al. 2008, Rolfe and Windle 2011b), and the Wet Tropics Reverse Auction conducted in 2014 (results are still confidential) (Table 6).

Insights into cost heterogeneity are provided from the Burdekin tender in Rolfe et al. (2011), where there were 87 bids submitted in the tender but only 33 funded. The data highlight the significant heterogeneity in supply costs, even across a relatively small number of landholders. In both the sugarcane and grazing sectors there were some very high cost bids that provided little environmental improvement (Figure 5), but also a number of lower cost proposals that generated large environmental benefits. This heterogeneity is essentially hidden in flat-rate grant schemes.

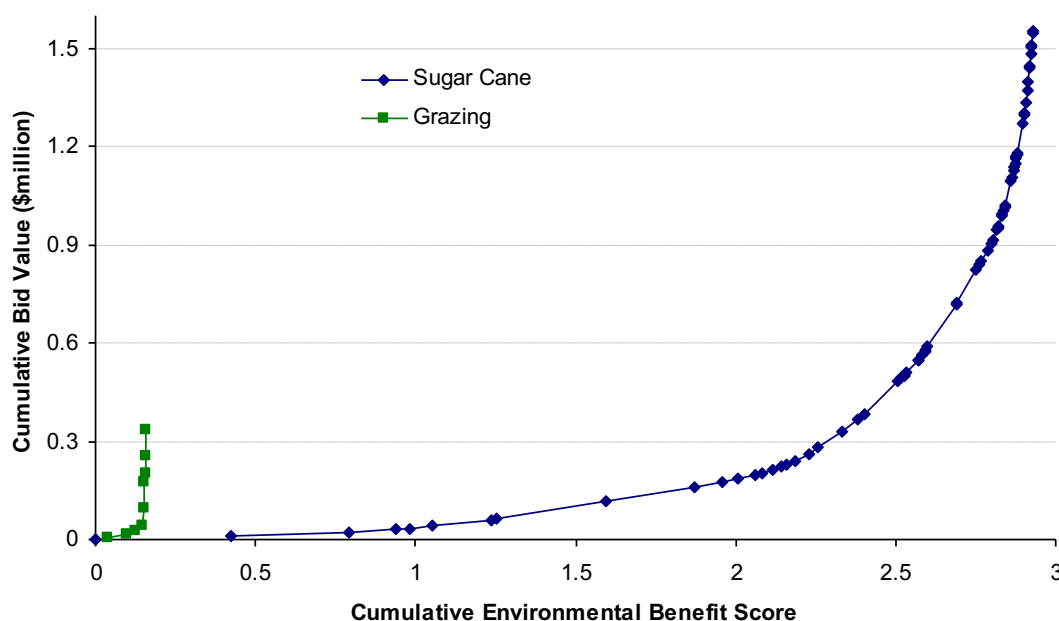


Figure 5: Cumulative bid curve for all bids (87) in the Burdekin tender (Rolfe et al. 2011: Figure 4)

Table 6: Summary statistics for the water quality tenders (successful bids only)

2015 dollar values	Burdekin:
End of paddock reductions	Cane + Grazing 2008
Total cost	\$707,850
Sediment	(grazing only)
Sediment reduction (tons)	492
Sediment reduction cost	\$91,885
Av. Reduction cost (\$/ton)	\$104.39
Reduction price range (\$/ton)	\$34.88 – \$244.69
Nitrogen	(cane only)
Nitrogen reduction (kg)	96,016
Nitrogen reduction cost	\$512,021
Av. Reduction cost (\$/kg)	\$5.34
Reduction price range (\$/kg)	\$0.76 – \$20.46
Pesticide	(cane only)
Pesticide reduction (kg)	55.5
Pesticide reduction cost	\$109,710
Av. Reduction cost (\$/kg)	\$1,976
Reduction price range (\$/kg)	\$860 - \$19,379

The evidence from the water quality auctions is that reductions in pollutants from agriculture are available at a range of costs, including many very low cost proposals. The schemes that have been implemented appear to be lower cost than flat rate grant schemes. However comparisons with the Reef Rescue costs are complicated because the tender estimates are

end-of-paddock, limited to the length of the contracted arrangement, and estimation of emissions are often simplistic.

As an example, the cost-effectiveness of the Nitrogen emissions from the tenders might be converted to end-of-catchment emissions at a factor of one-third. On this basis, the average end-of-catchment costs of annual change in Nitrogen emissions would be \$16/kg in the Burdekin tender, which are lower than both the historic and forward NRM cost estimates for the Burdekin reported in Tables 2 and 5. This comparison is based on an assumption that both types of allocation mechanisms generate benefits of equal duration and effect; variations in those assumptions will impact on the relative cost-effectiveness of either approach.

6.0 ESTIMATING COSTS: BIO-ECONOMIC MODELLING

The more standard approach in economics to predicting costs of making management changes is to employ bio-economic models. These are essentially economic production models for an average enterprise unit in an industry, where potential interactions with biological factors are added to the model. Changes in production systems and intensity can be represented in the models, with outputs showing both changes in production and profit (economic measures) as well as changes in environmental factors (such as emissions of pollutants). The model outputs allow predictions to be made about the tradeoffs between net private returns (both positive and negative) and changes in environmental outputs.

Many of the earlier studies relevant to the GBR have been focused on case studies and end-of-paddock improvements (e.g. Star et al. 2011), but more recent studies have involved broader applications and more detailed modelling. Some of the more sophisticated analyses are summarised in Table 7, together with the key variations in pollutant, catchment and modelling approaches that limit the comparison of estimates. Note that many of the predictions in the WQIPs are based on bioeconomic modelling. Further details are provided in Appendix 2.

Two important conclusions can be drawn from the bio-economic modelling. First, there are large variations in private costs, confirming that heterogeneity is important. Second, there are many situations where there are potential private returns from improving management practices that also deliver environmental improvements (win-wins); in these situations the financial enterprise costs of making management changes are effectively zero. These conclusions are demonstrated in Figure 6 for the costs of pesticide reduction, where the variation in costs are shown across catchments and practice changes. Here the tradeoffs range from -\$4,920/kg (a positive benefit to growers) to \$4,920/kg (a cost to growers), averaging \$780/kg.

A comparison of the bioeconomic modelling results (Table 7) to the previous costs and projected future costs of funding programs (Tables 2 and 5) reveals that the bio-economic modelling estimates of costs are a magnitude lower. For example, the estimates of costs for Nitrogen reduction from van Greiken et al. (2013), which are estimated for end-of-catchment loads in the Tully and Pioneer catchments, range from -\$22/kg to +\$22/kg, compared to an average of \$47/kg and \$164/kg for previously funded Reef Rescue projects in the Wet Tropics and Mackay Whitsunday regions (Australian Government 2014), and \$185/kg across the GBR for future projects estimated from Reef Regions (2015).

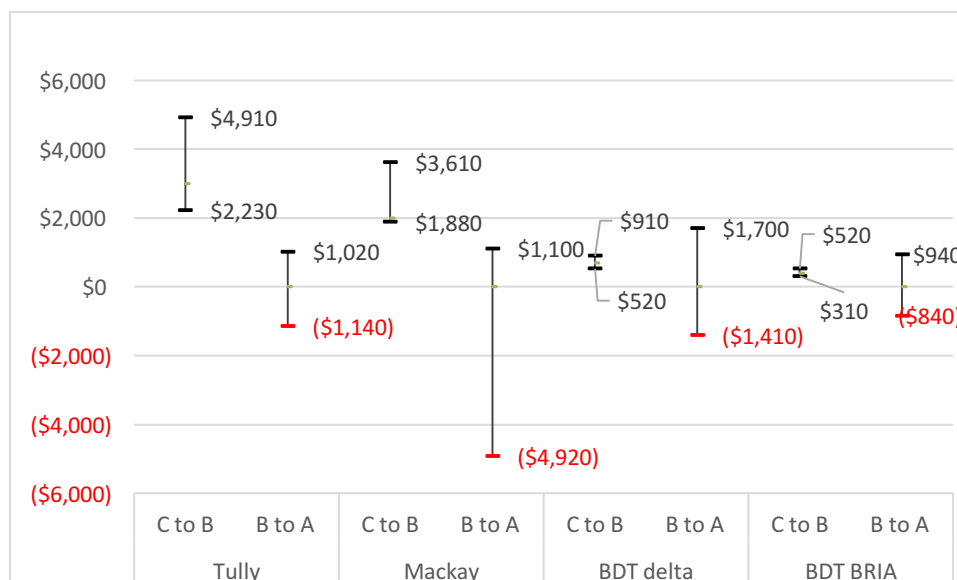
Similarly, the average cost of \$277/t to achieve a 20% reduction in sediment in the Fitzroy (Star et al. 2015) is much lower than \$513/t for previously funded Reef Rescue projects in the Fitzroy region (Australian Government 2014), and \$550/t across the GBR for future projects estimated from Reef Regions (2015). The estimated cost curves from Star et al. (2015) are replicated in the Figure below; these capture opportunity costs and an allowance for incentive/transaction costs, so should be directly comparable to Reef Rescue cost and Reef Regions estimates.

Table 7: Overview of the outputs from bio-economic modelling studies

	Pollutant	Private cost	Location	Reported results
Sugarcane				
Poggio et al. 2014	Pesticide End of paddock	AEB (10yr @6%)	Tully; Mackay; Burdekin (x2)	-\$4,920/kg to +\$4,910/kg pesticide reduction across C-B, C-A and B-A class practice change, av. = \$780/kg
Van Grieken et al. 2013	DIN End of catchment	NPV (gross margins + capital) (10 + 5 yr @ 7%)	Tully Mackay	-\$22/kg (5yrs) to +\$22/kg (10yrs) -\$22/kg (5yrs) to +\$22/kg (10yrs) NPV of practice (ABCD) change
Grazing				
Star et al. 2011, 2013, 2015a	Sediment End of paddock	NPV (20yr @6%)	Fitzroy Basin	\$4/t to \$421/t sediment reduction across 5 land types and 3 land condition classes
Star et al. 2015b	Sediment End of catchment	Capital + operating	Fitzroy WQIP	\$19/t to \$9,799/t across land types and land conditions in 93 sub-catchments; average cost to achieve 20% reduction is \$277/t
Pannell et al. 2014	n/a	Capital + operating + non-financial	Burnett Mary WQIP	\$10 - \$160/ha/yr for practice (ABCD) change across different practices, land types & farm size (includes \$25/ha for incentives to change practices)

AEB = Annual equivalent benefits (capital costs are included)

ABCD = four classes of management practices, ranging from A = aspirational to D = dated.



Data sourced from Poggio et al. (2014).

'C-B' and 'B-A' represent shifts from Common to Best and Best to Aspirational management practices

Figure 6: Private benefit/cost (\$/kg) variation for pesticide reduction from practice change

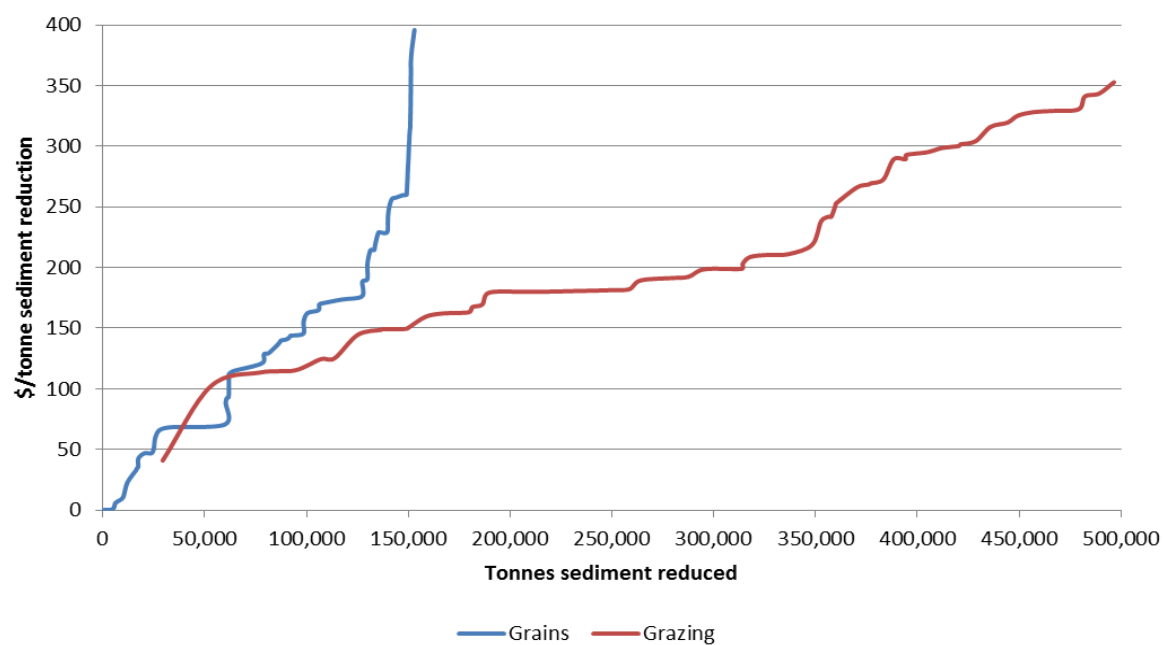


Figure 7: Costs of sediment reductions in the Fitzroy Basin (Figure 16 in Star et al. 2015)

7.0 SETTING BENCHMARKS FOR COST-EFFECTIVENESS

The data reviewed for this report reveals significant variation in cost-effectiveness. Many estimated costs are averages, so many higher cost projects have been funded within programs. The potential benefits of setting benchmarks for cost-effectiveness are that they provide:

- A guide for project selection and funding allocations
- An incentive to ensure that cost-effectiveness is estimated for each program and project
- Stronger guidelines to ensure that public funds are expended in an efficient manner.

Key options for setting appropriate benchmarks for each pollutant are:

- An average of historical grant programs (e.g. Reef Rescue 2008 – 13)
- The estimated average across the GBR 2008 – 13 (Australian Government 2014)
- The estimated future cost across the GBR 2015-20 (Reef Regions 2015)
- Averages estimated from the WQIPs across the NRM regions

While the evidence from some bioeconomic modelling and water quality auctions indicates that costs may be lower, the lack of cost estimates at end-of-catchment precludes these estimates at this point. The choice of benchmarks for sediment reductions is shown in the Figure below, with the following possibilities:

- Average of Reef Rescue programs by regions 2008 – 13 = \$597/t
- Australian Government estimate for all GBR 2008 – 13 = \$130/t
- Reef Regions (2015) estimates for future costs across the GBR 2015-20 = \$61/t
- Averages estimated from the WQIPs across the NRM regions = \$259/t

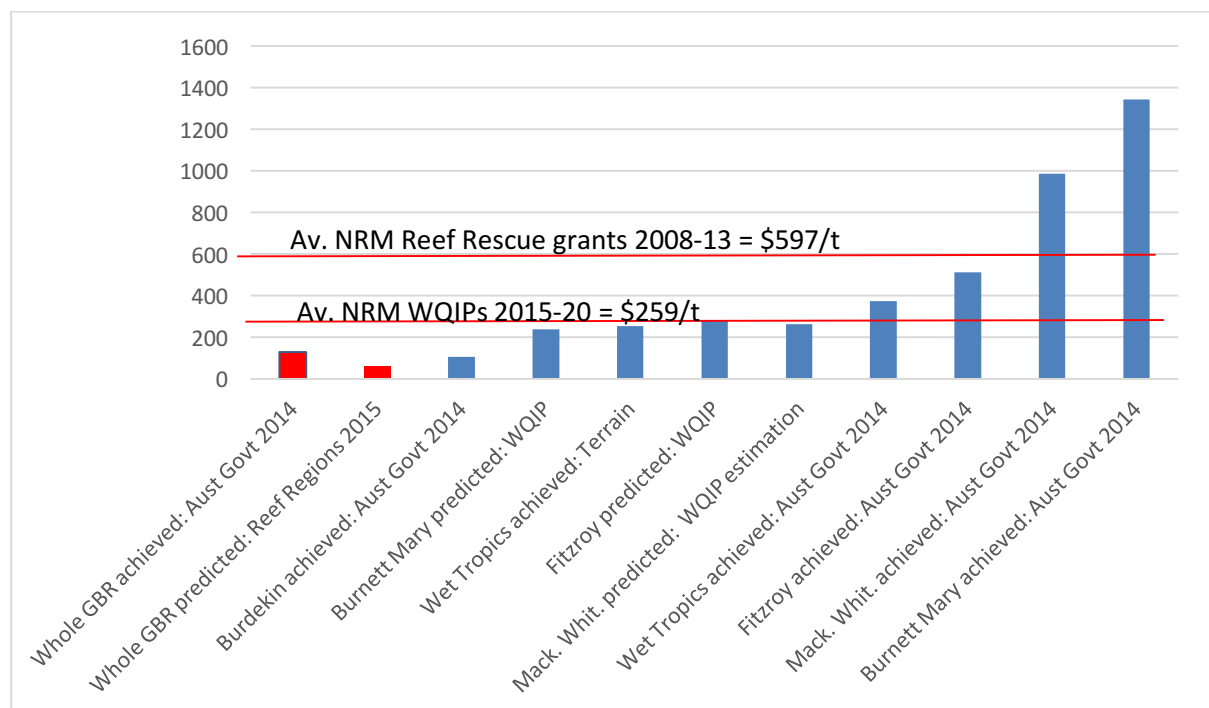


Figure 8: Summary and thresholds for cost-effectiveness of sediment reductions

The estimates derived from the WQIPs are likely to reflect the most accurate modelling of both costs and environmental benefits. **Therefore \$259/t is recommended as the benchmark level for investment into sediment reductions.**

A similar analysis of potential benchmarks can be conducted for N and DIN, as follows:

- N Average of Reef Rescue programs by regions 2008 – 13 = \$91/kg
- N Australian Government estimate for all GBR 2008 – 13: Not available
- N Reef Regions (2015) estimates for future costs across the GBR 2015-20 = \$63/kg
- N Averages estimated from the WQIPs across the NRM regions: Not available
- DIN Average of Reef Rescue programs by regions 2008 – 13 = \$172/kg
- DIN Australian Government estimate for all GBR 2008 – 13 = \$63/kg
- DIN Reef Regions (2015) estimates for future costs across the GBR 2015-20 : not available
- DIN Averages estimated from the WQIPs across the NRM regions = \$405/kg

The estimates derived from the WQIPs are likely to reflect the most accurate modelling of both costs and environmental benefits, noting that evidence from bio-economic modelling and water quality auctions is that costs may be even lower. **Therefore \$150/kg is recommended as an interim benchmark level for investment into DIN reductions.**

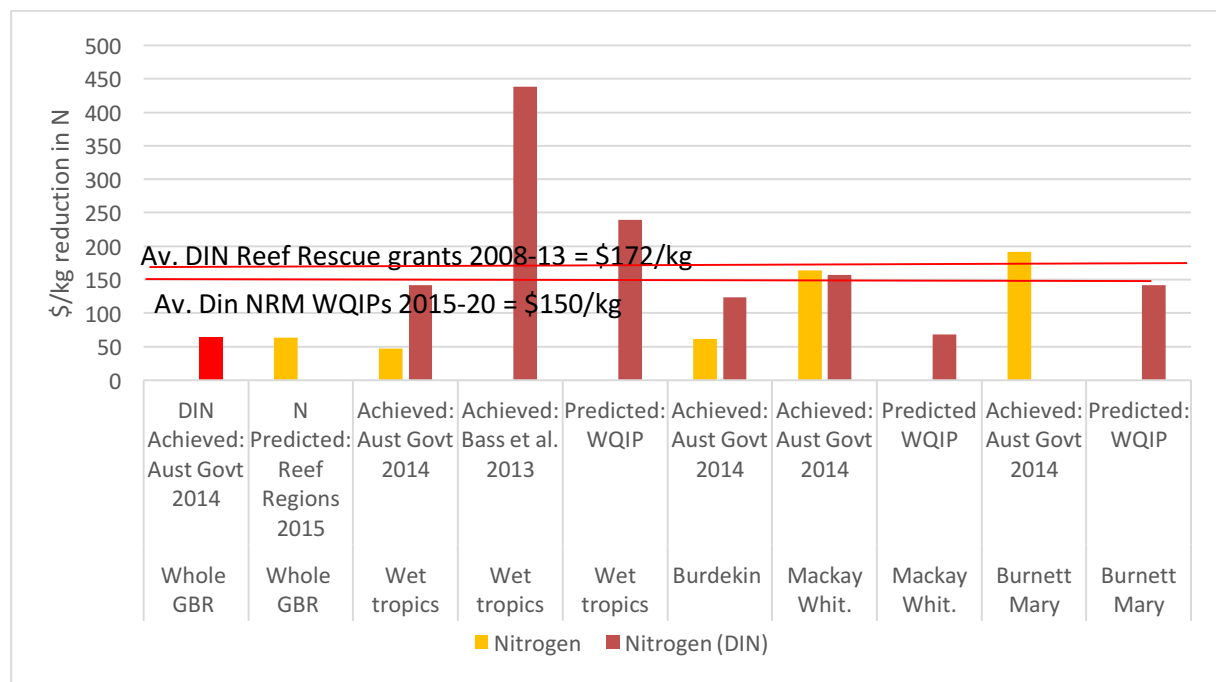
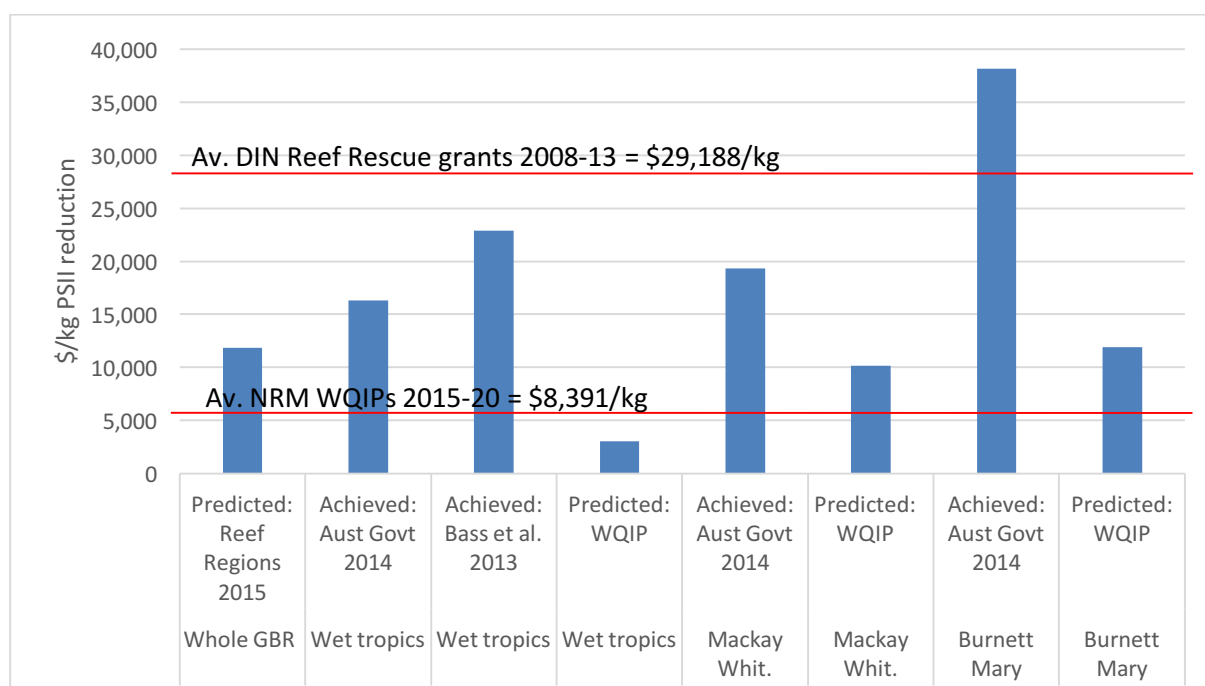


Figure 9: Summary and threshold for cost-effectiveness of nitrogen reductions

A similar analysis of potential benchmarks can be conducted for pesticides, as follows:

- Average of Reef Rescue programs by regions 2008 – 13 = \$6,393/kg
- Australian Government estimate for all GBR 2008 – 13 = \$3,500
- Reef Regions (2015) estimates for future costs across the GBR 2015-20 = \$11,836/kg
- Averages estimated from the WQIPs across the NRM regions = \$8,391/kg

The estimates derived from the WQIPs are likely to reflect the most accurate modelling of both costs and environmental benefits, noting that evidence from bio-economic modelling and water quality auctions is that costs may be even lower. **Therefore \$8,391/kg is recommended as an interim benchmark level for investment into PSII reductions.**



Note: Reef Rescue cost for Fitzroy (2008-13) excluded from analysis, and Reef Rescue cost for Burdekin (2008-13) included but not shown in Figure.

Figure 10: Summary and threshold for cost-effectiveness of pesticide reductions

The research revealed that data on cost-effectiveness was generally difficult to source at the project level, even though estimates of water quality improvements should be a key component of evaluation in each project approval. **It is recommended that:**

- **Data on cost-effectiveness should be automatically collated at the project level when predictions of improvements are made and funding is allocated.**
- **Modelling tools be developed to estimates of pollutant changes at the project level**

Estimates of cost-effectiveness appear to be much higher in grant programs than predicted by bioeconomic modelling or achieved through water quality tenders (noting some

confounding between end-of-paddock and end-of-catchment modelling). **It is recommended that:**

- **There should be greater emphasis on cost-effectiveness in project selection.**
- **Program design should be adjusted away from a reliance on simple grant mechanisms to processes that encourage better price discovery and project selection.**

8.0 RECOMMENDATIONS AND CONCLUSION

The estimates of cost effectiveness have been summarised in Table 8, by end-of-catchment and end-of-paddock loads. Results from bioeconomic models that are not captured in the WQIPs have not been included because of the complexity of the data, although it should be noted that those models generally predict much lower costs.

Table 8: Summary of Cost estimates

Region	Sediment \$/tonne	Nitrogen \$/kg	Nitrogen (DIN) \$/kg	Pesticide (PSII) \$/kg
End of catchment loads				
Whole GBR				
Achieved: Aust Govt 2014	130		63	3,500
<i>Predicted: Reef Regions 2015</i>	<i>61</i>	<i>21</i>	<i>34</i>	<i>3,945</i>
Cape York				
Achieved: Aust Govt 2014	297	192		
Wet tropics				
Achieved: Aust Govt 2014	375	47	142	16,343
Achieved: Bass et al. 2013	255		438	22,886
<i>Predicted: WQIP(70:30 funding split)</i>			<i>239</i>	<i>3,076</i>
Burdekin				
Achieved: Aust Govt 2014	106	62	124	120,770
Mackay Whitsunday				
Achieved: Aust Govt 2014	987	164	157	19,315
<i>Predicted: WQIP</i>	<i>263</i>		<i>68</i>	<i>10,178</i>
Fitzroy				
Achieved: Aust Govt 2014	513	1,494		1,060,650
<i>Predicted: Star et al. 2015b WQIP</i>	<i>277</i>			
Burnett Mary				
Achieved: Aust Govt 2014	1,343	192		38,153
<i>Predicted: WQIP (50:35:15 funding split)</i>	<i>237</i>		<i>142</i>	<i>11,918</i>
Whole GBR				
Recommended Benchmark	\$259/t		\$150/kg	\$8,351/kg
End of paddock loads				
Wet tropics				
Achieved: Bass et al. 2013	25	141		667
Burdekin				
Achieved: Burdekin Tender 2011	104	5		1,976

The initial research question was to determine if variations in cost components and modelling approaches could explain differences in modelling, and to identify if recommendations for benchmarks could be set.

The results of this review reveal significant differences in cost estimates for each of the key pollutant types, with most estimates available at end-of-catchment to facilitate comparison:

- Costs vary across regions (see Table 2 and Figure 2) where cost effectiveness of NRM grants vary by at least a factor of 10 across regions. This indicates that cost-effectiveness can be significantly improved by the allocation of funds across regions.
 - Costs within regions (see Figures 4, 5 and 6), reflecting significant heterogeneity between landholders and projects. This variation is largely disguised in the grants programs because funding rates are often fixed. This heterogeneity means that cost-effectiveness can be significantly improved by the allocation of funds within regions.
 - Projected costs of grant projects appear to be decreasing over time (Figure 3). Costs in future programs are predicted to be much lower than over past grants (average decreases are 64% in Sediment, 11% in DIN and 46% in PSII); this is despite more accurate modelling generally reducing predictions of pollution changes and thus increasing estimates of cost-effectiveness.
 - Cost estimates vary by estimation approach, with the economic tools of bio-economic modelling and water quality auctions predicting much lower levels of cost-effectiveness than what is revealed through grant programs. Bio-economic modelling indicates that there are many practice changes that involve little or no opportunity costs; even after making allowances for incentive and transaction costs the costs of making changes is low (Table 7).
 - Costs vary by allocation approach, with costs revealed through the Burdekin tender significantly lower than costs identified with grants programs:
 - For Sediments, average grant costs (\$536/t) > five times tenders (\$105/t)
 - For Nitrogen, average grant costs (\$359/kg) > 71 times tenders (\$5/kg)
 - For Pesticides, average grant costs (\$39,107/kg > 20 times tenders (\$1,976)
- While some of the differences are because of confounding between end-of-paddock (tenders) and end-of-catchment modelling, the size of the differences indicates that allocation methods have an impact on costs.
- Summary estimates of costs at the whole GBR level appear to be systematically lower than when costs are analysed at the regional level.

These results indicate that better information about cost-effectiveness should be used to improve prioritisation of funding allocations and project selection, and that large gains in the effectiveness of public funding are possible.

The research revealed that data on cost-effectiveness was generally difficult to source at the project level, even though estimates of water quality improvements should be a key component of evaluation in each project approval. It is recommended that data on cost-effectiveness should be automatically collated at the project level when predictions of improvements are made and funding is allocated.

Estimates of cost-effectiveness appear to be much higher in grant programs than predicted by bioeconomic modelling or achieved through water quality tenders (noting some confounding between end-of-paddock and end-of-catchment modelling). It is recommended that:

- There should be greater emphasis on cost-effectiveness in project selection
- Tools to model pollution changes at the project selection point be developed

- Program design should be adjusted away from a reliance on simple grant mechanisms to processes that encourage better price discovery and project selection.

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APPENDIX 1: ESTIMATING FUTURE COSTS IN GRANT PROGRAMS

Four updated regional WQIPs (Wet Tropics, Mackay Whitsunday, Fitzroy and Burnett Mary) are discussed in this appendix. The WQIPs vary in content and only summary details are presented below, although more details about the bio-economic modelling are presented in the following section. In terms of emission reductions, the plans include reference to both Reef Plan Targets as well as Ecologically Relevant Targets, but only the former are discussed here. Reef Plan Targets (RPTs) indicate the following water quality reduction targets to be achieved by 2018.

- At least a 50% reduction in anthropogenic loads end-of-catchment dissolved inorganic nitrogen loads in priority areas;
- At least a 20% reduction in anthropogenic end-of-catchment loads of sediment and articulate nutrients in priority areas; and
- At least a 60% reduction in end-of-catchment pesticide loads in priority areas

In all cases the WQIPs have outlined detailed information for the sub-catchments as well as for the region generally. Significant sub-catchment heterogeneity in emissions and load reduction costs is evident within the regions, but for brevity only broad regional level-information is presented below.

Two further reports are discussed in this section with both reports providing information about future funding: The recently published Reef 2050 Long-term Sustainability Plan (Reef Regions 2015) and a cost effectiveness assessment to prioritise Reef Trust Phase 2 investment (Addison and Walshe 2015).

A1.1 Terrain: Wet Tropics region WQIP

The WQIP for the Wet Tropics region (Terrain 2015) covers the period 2015-2020. It is estimated that that an investment of at least \$6 million per year or **\$30 million over 5 years** would be required to fully implement the actions identified in WQIP. This includes both the cost of incentive payments for on-ground works as well as program delivery costs.

Terrain contracted external expertise to help develop their WQIP. The Investment Framework for Environmental Resources (INFFER™) was used to assess the benefits and costs of different management scenarios to achieve water quality improvements in the sugarcane industry. This analysis helped determine the most suitable policy mechanism to achieve the selected improvements (such as incentives or extension) and the full costs of program implementation were then estimated.

Three improvement scenarios were highlighted (more scenarios were evaluated) with the on-farm costs estimated using currently available bio-economic modelling results (Poggio et al. 2014; van Grieken et al. 2010; 2014b) and expert opinion. Evidence indicated that financial incentives would be required for growers to shift permanently from B to A practice (to cover the on-farm costs involved). In contrast, moving to B class practices was shown to be profitable, but a small incentive scheme was included in the program costs to encourage adoption. It was estimated that even if all growers converted to A class practices (at an annual cost over \$6.6

million over five years) the Reef Plan target of a 50% reduction in nitrogen loads (DIN) would not be reached and the benefits would not be greater than the costs (Table A1.1).

Table A1.1: Program costs and outcomes for three improvement scenarios (Sugarcane)

	Annual funding required (\$M) in the first 5 years	Benefit: Cost Ratio	Predicted DIN reduction RP target 50%)	Predicted PSII reduction RP target 60%)
1. All A practice	6.6 (3.66 +3.0) ¹	0.49	~30%	~95%
2. 50% A:50% B	5.8 (2.82 +3.0)	0.54	~24%	~79%
3. All B practice	2.6 (0 +2.6)	1.3	~19%	~63%

Source: Wet Tropics WQIP: Table 4.2

¹ Values in parenthesis are the annual incentive payments followed by the annual extension and program delivery costs) outlined in Park and Roberts 2014: Table 11)

The estimated program delivery costs (\$13 million) represented 42% of the estimated budget for the WQIP (Terrain 2015: Table 8.17) and 62% of the total budget if the cost is estimated across all agricultural industries and not only for sugar cane (Table A1.2). Based on this, the costs of achieving all 'B' class practice are estimated at \$30.935M, 50%B50%A at \$41.935M, and all A practice at \$50.935M.

Table A1.2: Summary of preliminary cost estimates to implement the Wet Tropics WQIP over the first 5 years. (Terrain 2015: Table 8.17)

Item	Cost per annum	Total cost over first 5 years
Leadership, governance, project management	\$427,000	\$2,135,000
Communication and Engagement	\$160,000	\$800,000
Compliance and Auditing – stewardship payments	\$120,000	\$600,000
Urban investigations and planning	\$100,000	\$500,000
Shifting to All B practices in sugar cane	\$2,600,000	*\$13,000,000
Shifting to All B practices in other agricultural industries	\$1,250,000	**\$6,250,000
System Repair actions	***\$200,000	***\$1,000,000 (TBD)
Monitoring and Evaluation	\$500,000	\$2,500,000
Priority research themes to fill knowledge gaps (preliminary assessment):	\$580,000	\$2,900,000
• Identification of small scale pollutant hotspots that can be feasibly managed • Quantifying the environmental benefits and costs of ecological restoration projects • Evaluating the effectiveness of alternative management scenarios • Understanding the socio economic characteristics of practice change in agricultural industries • Regionally specific climate change scenarios Synthesis of supporting knowledge		
Synthesis of supporting knowledge	\$100,000	\$500,000
Estimated sub total	\$6,037,000	\$30,185,000
Contingency: Supports emerging priorities or measurement of episodic events	\$150,000	\$750,000
Total	\$6,187,000	\$30,935,000

* Sourced from INFER analysis

**Estimated from investment through the Wet Tropics Reef Rescue Water Quality Grants 2008 to 2013 in other agricultural industries (bananas, dairy, forestry, dry grazing, wet grazing, multi-crops, pawpaw and tree crops).

***Estimate to support planning and development of regional system repair strategy only. Does not include cost of any on-ground works.

However, information about the improvements in pollutant load reductions that can be expected from this level of practice change is only presented in terms of percentage changes. There is no detailed information about the quantity of pollutant load reduction that could be expected. In order to estimate an approximate cost per unit of load reductions, information has been applied from the GBR Report Card, 2012-2013 (A&QG 2014) which relates percentages to quantities. As no information is provided in the WQIP about how costs are allocated between pollutants, a 70:30 budget allocation is adopted (broadly in line with the breakup provided in the Mackay Whitsunday WQIP).

The results are outlined in Table A3 and account for the level of load reduction improvement already achieved as reported in the 2014 GBR Report Card. The budget estimates for the WQIP were based on the third scenario of moving to all B class practice. The cost of reaching a 19% DIN reduction was approximately **\$239/kg** and a 63% pesticide reduction was approximately **\$3,076/kg**. Although the cost of moving to all B class practice was lower (\$31 million including \$19.23M in practice change costs) as no incentive payment were required, the low level of load reduction improvement meant the cost of achieve DIN reduction was high.

Table A1.3: Average costs of achieving end-of-catchment load reductions in the Wet Tropics

	Nitrogen (DIN)	Pesticide	Nitrogen (DIN)	Pesticide	Nitrogen (DIN)	Pesticide
	<i>All A class practice</i>		<i>50% A class: 50% B class</i>		<i>All B class practice</i>	
1% load reduction ¹	19.77 t	85.92 kg	19.77 t	85.92 kg	19.77 t	85.92 kg
Estimated reduction	30%	95%	24%	79%	19%	63%
2014 achievement	14.5%	28.5%	14.5%	28.5%	14.5%	28.5%
% reduction required	15.5%	66.5%	9.5%	50.5%	4.5%	34.5%
Assumed funding (\$M)	\$35.3	\$15.1	\$29.0	\$12.4	\$21.3	\$9.1
Avg cost	\$115/kg	\$2,646/kg	\$154/kg	\$2,862/kg	\$239/kg	\$3,076/kg

¹Based on information that equated percentage to absolute quantities provided in the GBR Report Card, 2012-2013 (A&QG 2014)

A1.2 Reef Catchments: Mackay Whitsunday region WQIP

The Mackay Whitsunday WQIP (Folkers et al. 2014) covers the period 2014-2021. The cost of working towards achieving the WQIP targets across the region was estimated at \$46 million by 2021, comprising:

- **\$36 million** in direct costs for on ground work (sugarcane/horticulture \$20m and grazing \$16m), plus
- **\$10 million** for consultation and extension

There was no supporting economic analysis or technical information in the WQIP to outline how these cost estimates had been determined.

Load reduction targets were developed from a combination of modelling and water quality data, and the likely reductions that could be expected from widespread adoption of land management practices. The following WQIP (2021) targets (**end-of-catchment loads**) were set (which are lower than the Reef Plan targets):

- **Sediment: 14-16% reduction** from a 9% improvement in Sugarcane and Horticulture and 13% improvement in grazing management practices
- **Nitrogen (DIN): 12% reduction** from a 16% improvement in Sugarcane and Horticulture management practices
- **Herbicide: 10-20% reduction** from a 10% improvement in Sugarcane and Horticulture management practices

The expected cost to achieve these load reductions (over a period of seven years [2014-2021]), excluding the urban sector, was reported at:

- \$20.3 million for sediment reduction of 89,960 tonnes/year
- \$9.89 million for nitrogen (DIN) reduction of 194 tonnes/year
- \$5.99 million for herbicide reduction of 916 kg/year

This equates to an approximate reduction cost over 7 years of:

- \$225 per tonne for sediment reduction
- \$51 per kg for nitrogen(DIN) reduction
- \$6,539 per kg over 7 years for herbicide reduction

Reduction costs increase when the \$10 million extension cost is included (Table A1.4).

Table A1.4: End-of-catchment emission reduction costs for the Mackay Whitsunday region

	Sediment	nitrogen(DIN)	Herbicide
1. Apportioned cost and loads			
Apportioned cost ¹	\$20.3 million	\$9.89 million	\$5.99 million
Load reduction (7 yrs) ²	89,960 tonnes	194 tonnes	916 kg
Reduction cost	\$225 per tonne	\$51 per kg	\$6,539 per kg
Reduction cost inc \$10M extension ³	\$263 per tonne	\$68 per kg	\$10,178 per kg

¹ WQIP Table 66

² WQIP Table 68

³ Cost of 10 million apportioned equally across the three pollutants

A1.3 Fitzroy Basin Association: Fitzroy Region WQIP

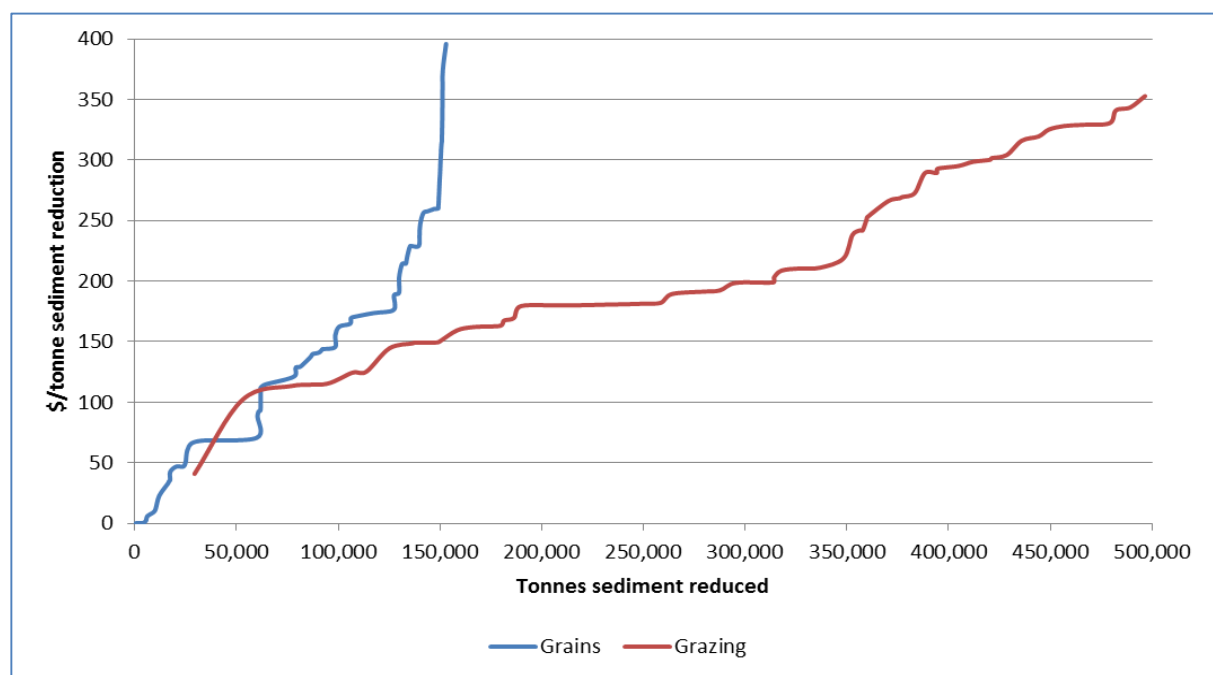
The WQIP for the Fitzroy region has yet to be finalised and only the synthesis report (Waterhouse et al. 2015) and supporting reports are currently available. There is no information about the specific objectives and targets to be achieved in the WQIP or the estimated cost of achieving them. Consequently, it was not possible to infer the load reduction costs as estimated for Wet Tropics, Mackay Whitsunday and Burnett Mary regions.

The economic analysis to support the WQIP was conducted by Star et al. (2015b). The authors applied a bio-economic modelling approach aimed at prioritising locations (neighbourhood catchments) within the Fitzroy Basin where the most cost effective sediment reductions could be targeted. The focus was on sediment reductions within the grazing sector. Key assessment criteria included sediment loads, ground cover, management effectiveness as well as costs. The cost estimates included **up-front capital costs and opportunity costs** associated with the adoption of B class management practice. Further details are provided in Section 5. Different variables, including cost, were normalised to effectively combine them within the assessment function. This meant that details of the cost

effectiveness, in terms of cost per unit of sediment reduction, was not provided for the different modelled scenarios and only summary details were presented. The reported **end-of-paddock** costs ranged significantly **from \$18.83 per tonne to \$9,779 per tonne sediment reduction**.

The cost to achieve the Reef Plan sediment target of a 20% reduction (390,200 tonnes) over a 10 year period was estimated at \$108 million **or an average load reduction cost of \$277 per tonne sediment reduction**. The cost curves differed significantly across the grains and grazing sectors (Figure A1).

Figure A1.1: End-of-paddock sediment reduction cost range across industry sectors (Star et al.2015b: Figure 16)



A1.4 Burnett Mary Regional Group: Burnett Mary region WQIP

The WQIP for the Burnett Mary region (BMRG n.d) does not cover any specified period of time. The cost of working towards achieving Reef Plan Targets across the region was estimated at \$32.5 million over the first five years, comprising:

- **\$27.1 million** in direct costs for on ground work, plus
- **\$5.4 million** for the indirect cost including program delivery and extension.

The Burnett Mary Regional Group contracted external expertise to help develop their WQIP. The Investment Framework for Environmental Resources (INFFER™) was used to assess the benefits and costs of different management scenarios to achieve water quality targets in both the sugarcane and grazing sectors. New bio-economic modelling was also commissioned for the grazing and sugarcane sectors and a wide range of potential improvement scenarios were evaluated. A summary of the economic analysis is provided below.

To meet the Reef Plan targets across the whole region, it was estimated that:

- All sugarcane land management needs to move to A or B class practice (9,431 ha cane to A class practice level and 41,010 ha to B class practice).
- All grazing land management needs to move to A class practice across 131,349 ha, from areas currently in B, C and D class practices.

Given the overall cost of \$32.5 million, this would equate to a cost of \$179/ha. However, information about the improvements in pollutant load reductions that can be expected from this level of practice change, is only presented in terms of percentage changes. There is no detailed information about the quantity of pollutant load reduction that could be expected. In order to estimate an approximate cost per unit of load reductions, information has been applied from the GBR Report Card, 2012-2013 (A&QG 2014) which relates percentages to quantities. The results are outlined in Table A and account for the level of load reduction improvement already achieved as reported in the 2014 GBR Report Card. Assuming the Reef Plan targets are met in the five year funding cycle of the WQIP and that funding is allocated equally across sediments, nutrients and pesticides, it is estimated the average cost of load reductions would be:

- \$159 per ton of sediment reduction
- \$135 per kg of nitrogen (DIN) reduction and
- \$26,484 per kg of pesticide (PSII)

If funding ratios were 50: 35: 15 for Sediments, Nutrients and Pesticides, the average costs of load reductions would be:

- \$238 per ton of sediment reduction
- \$142 per kg of nitrogen (DIN) reduction and
- \$11,918 per kg of pesticide (PSII)

Table A1.5: Average costs of achieving end-of-catchment load reduction targets

	Sediment	Nitrogen (DIN)	Pesticide
1% load reduction ¹	4016.67 tonnes	4.34 tonnes	15.15 kg
RP target	20%	50%	60%
2014 achievement	3%	31.5%	33%
% reduction required	17%	18.5%	27 %
Avg cost over 5 years	\$159/t	\$135/kg	\$26,484/kg

¹Based on information that equated percentage to absolute quantities provided in the GBR Report Card, 2012-2013 (A&QG 2014)

A1.4.1 Economic analysis for grazing

Pannell et al. (2014) conducted the bio-economic modelling and economic analysis for the grazing industry with input from local experts. All practice changes across all farm sizes were estimated to come at a cost, once non-financial related barriers were considered. They estimated the required level of incentives to adopt improved practices, based on three components:

1. the up-front costs required for adoption
2. changes in the annual profitability of land use following adoption
3. non-profit barriers to adoption

A1.4.2 Economic analysis for sugarcane

Van Grieken et al. (2014a) conducted the bio-economic modelling and economic analysis for the sugarcane industry. They estimated the required level of incentives to adopt improved practices, based on the same three components as for grazing (capital costs, profitability and non-financial costs). However in this case, two separate categories (transaction costs and 'other' non-profit costs) were estimated to account for the non-profit barriers to adoption.

1. the up-front costs required for adoption
2. changes in the annual profitability of land use following adoption
3. non-profit barriers to adoption
 - a. transaction costs
 - b. 'other' non-profit costs

A1.5 GBR Long term Sustainability Plan 2050

Concerned that current investment would not meet the Long Term sustainability Plan vision for a sustainable GBR region, Reef Regions (2015) developed an investment plan to provide a realistic estimate of abatement costs, based on nine years of experience and two generations of Water Quality Improvement Plans. It was estimated that continued improvement in agricultural management practices would cost \$175 million for the period 2015 to 2020. It was predicted that such expenditure would result in the following **end-of-catchment** reductions (incorporating existing achievements (2008 – 2013)):

- 15-20% reduction in total suspended solids
- 30-35% reduction in dissolved inorganic nitrogen
- 15-20% reduction in particulate nitrogen
- > 90% reduction in PSII herbicides

The approximate load reduction costs are outlined in Table A1.6. It was also estimated that further expenditure of \$180 million would provide accelerated outcomes but no further details were provided (Reef Regions 2015: Table 1). As no information was provided on how investments was to be allocated across pollutants, it is assumed for this analysis that the ratio would be 50:35:15, broadly consistent with the Reef Catchments WQIP.

Table A1.6: Estimated end-of-catchment abatement costs for the whole GBR 2015-2020

	Sediment	Nitrogen	Nitrogen (DIN)	Herbicide (PSII)
Load reduction %	15-20%	15-20%	30-35%	90%
1% equivalent ¹	54465.34t	162.24t	na	164.28kg
Cost allocation of \$175M	\$87.5M	\$61.25M	\$61.25M	\$26.25M
Cost per pollutant	\$92/t	\$22/kg	\$36/kg	\$1,775/kg

¹Based on information about end of catchment load reduction provided in the GBR Report Card, 2012-2013 (A&QG 2014).

A1.6 Cost effectiveness assessment to prioritise Reef Trust Phase 2 investment

The Australian Institute of Marine Science developed a cost-effectiveness estimation process to prioritise specified management interventions for investment under the second phase of Reef Trust (Addison and Walshe 2015). Nine management interventions were nominated by the Department of the Environment for consideration. The costs of the interventions were estimated by personnel from Reef Trust and included the estimated cost of administration as well as the cost of any associated incentive payments. The main focus of the report was on estimating the environmental benefits of the interventions using expert judgement to develop preferential weights in a structured decision making approach. The authors noted the assessment was limited with insufficient background information (i.e. no estimated rates of adoption) to make a meaningful assessment.

The nine interventions were ranked in order of their cost effectiveness as outline below:

1. Rehabilitation of coastal ecosystems (\$1 million): Burdekin
2. Erosion control in priority grazing landscapes (gully restoration) (\$5 million): Cape York, Burdekin, Mackay Whitsunday, Burnett Mary
3. Smartcane best management practice (BMP) accreditation (\$0.52 million): Burdekin
4. Farmer led innovative cane management practices (game changer) (\$4 million): Wet tropics, Burdekin, Mackay Whitsunday, Burnett Mary
5. Expansion of holistic grazing pilot project (\$0.9 million): Burdekin
6. Reef Bonus trial (\$2.98 million): Wet Tropics, Burdekin
7. Reverse tender for nitrogen use efficiency of sugar cane farms (\$2.98 million): Burdekin
8. One million trees for the reef (\$7.55 million): Wet tropics, Burdekin, Mackay Whitsunday, Fitzroy, Burnett Mary
9. Crown-of-thorns starfish (COTS) tactical (\$6.97 million): Wet Tropics

The results favoured the low cost projects (smaller denominator) and those with greater spatial implementation (larger numerator). The cost effectiveness rankings provided no information about environmental outputs or the relative cost of pollutant load reductions which might be used for comparison elsewhere.

A1.7 Summary

There is considerable variation in the economic analysis applied in the development of the WQIPs and future funding cost estimates. There is variation all three key components of analysis, including:

1) estimating the improvements in land management practices required

- a) In some cases, the focus on management practice improvements is based very broadly on the benefits of moving to a higher class of management practice under the ABCD framework (e.g., Wet Tropics and Burnett Mary WQIPs).
- b) In other cases (e.g., Fitzroy WQIP) the focus is on more specific and detailed management practices.
- c) In other cases (e.g. Mackay Whitsunday WQIP) specific details are not provided.

- d) Most studies relate to changes in the sugarcane and grazing sector, with few details from other sectors (e.g. Grains: Star et al. 2015b).

2) determining the costs of achieving practice change

- a) In most cases the cost of practice change is based on **expert opinion** (including best available secondary sources) and includes an assessment of **upfront capital costs and operating costs**.
- b) Variation occurs in the inclusion of additional **non-financial costs** such as transactions costs and other barriers to adoption.
- c) Cost estimates vary in precision depending on the practice change. Estimates associated with changing management classes under **ABCD framework** are less precise than those associated with a very specific and **well defined practice**.
- d) In some studies cost information is disaggregated with details provided about the private costs and the public costs (costs of program implementation) of achieving the stated objectives. In other cases only very broad level costings are provided and in these cases load reduction cost estimates are likely to be more expensive as cost estimates may include unrelated costs.

3) estimating load reductions associated with practice change

- a) Most load reduction estimates refer to **end of catchment** reductions but some refer to **end of paddock** reduction (e.g., Fitzroy WQIP)
- b) Some load reduction estimates are based on **expert opinion** (including best available secondary sources) while other estimates have been generated from specially commissioned **bio-physical modelling**
- c) In many studies detailed information about load reductions estimates is not provided and is only reported in broad **percentage terms**. In such cases, estimates of **absolute quantities** have been inferred from information provided in the Great Barrier Reef report cards.

The most revealing example of where analytical differences can impact on the results is provided by the Mackay Whitsunday WQIP (Table A1.4). The WQIP provided information about the estimated absolute quantity reduction for each pollutant as well as the cost of reduction for each pollutant. This was the only WQIP where both costs and load reduction were apportioned in such a way. When the addition costs of extension were included, load reduction costs were estimated at \$80/t, \$10/kg and \$1,454/kg for sediment, DIN and pesticide respectively.

A second estimated process was performed to try and match the limited information available in some of the other WQIPs (i.e. Wet Tropics and Burnett Mary). This time the reported percentage reductions were converted into absolute quantities applying information from the Great Barrier Reef report cards. As well, the single total cost estimate was applied. In this case the load reduction costs were estimated at \$851/t, \$451/kg and \$77,745/kg, representing a 10-fold, 45-fold and 53-fold increase for sediment, DIN and pesticide respectively.

A summary of the abatement costs estimated from information provided in the WQIPs is outlined in Table A1.7. However, unless more accurate load reduction costings and information becomes available, the true underlying regional and sectoral heterogeneity is not revealed.

Table A1.7: Summary of cost estimates for future funding (WQIPs)

Region	Sediment	Nitrogen	Nitrogen (DIN)	Pesticide (PSII)
End of catchment loads				
Whole GBR (Reef Regions 2015)				
Load reduction	15-20%	15-20%	30-35%	90%
Public Cost @ \$175M (equal allocations)		\$21/kg		
	\$61/t		\$34/kg	\$3,945/kg
Public Cost @ \$175M (50:35:15)	\$92/t	\$22/kg	\$32/kg	\$1,776/kg
Wet tropics				
Load reduction	na		19%	63%
Public Cost @ \$13M			\$239/kg	\$3,076/kg
Mackay Whitsunday				
Load reduction	15%		12%	15%
Public Cost @ \$46M inc extn	\$263/t		\$68/kg	\$10,178/kg
Burnett Mary				
Load reduction	20%		50%	60%
\$32.5M split 50:35:15 by pollutant	\$238/t		\$142/kg	\$11,917/kg
Fitzroy				
Load reduction	20%			
Private cost	\$277/t			

APPENDIX 2: ESTIMATING FUTURE COSTS WITH BIO-ECONOMIC MODELLING

Economic modelling can help provide insights into the cost benefit trade-offs associated with changes in land management practices. Until recently, economic modelling has tended to focus on gross margin analysis and identifying the farm level financial costs and benefits of specific management practices. The majority of these studies have been focused on the sugarcane industry and have been conducted by staff from the Queensland Department of Agriculture and Fisheries (e.g., East et al. 2012; Edwards et al. 2013). Advances are now being made in the use of bio-economic modelling (combining economic and biophysical sub-models) which helps identify the environmental outputs as well as the economic impacts. New bio-economic models have been developed with funding from the Reef Rescue program to help identify the economic impacts of achieving pollutant reduction targets and more recently to help identify the costs involved in achieving targets in some regional WQIPs.

All models are limited by their baseline assumptions, and are restricted in the extent to which heterogeneity can be incorporated. Authors are generally very thorough in outlining the assumptions made in their models, but by necessity, end-users tend to report the modelled outputs without further reference to the underlying assumptions (as the case in this report). After years of development and refinement, some bio-physical models are becoming more robust and better able to predict outputs that align with real-world monitored records. Other newly developed bio-physical models are less accurate have to be further refined. Economic modelling is less well developed and models remain limited in the extent to which known socio-economic heterogeneity has been incorporated.

In this section, further summary details are provided about some bio-economic modelling that was developed either through research funding under the Reef Rescue Programme, or to help inform the updated WQIPs. The focus has been on the sugarcane and grazing sectors.

A2.1 Sugarcane

Recent bio-economic modelling results reveals that private farm-level economic tradeoffs from moving to better management practices can vary from strongly positive to strongly negative.

Poggio et al. 2014

Poggio et al. 2014 combine economic analysis with two biophysical models (APSIM, a crop model and HowLeaky, a pesticide model), to estimate the cost effectiveness of reducing **end of paddock** pesticide emissions. They calculate the economic benefit or cost of emission reductions as the ratio of the economic benefit or cost (\$/ha/year) to the total reduction of herbicides.

The analysis was conducted across:

- two types of tillage practice (low and intensive)
- three farm sizes (50, 150 and 250ha)

- four regional areas (Tully, Mackay and the Burdekin Delta and Irrigation (BRIA) areas).

The abatement costs are summarised in Table A2.1 and Figure 6. Across all scenarios the private benefits of reducing pesticide loads by improving management practices vary 70-fold from \$70/kg to \$4,910/kg reduction, and the costs vary more than 80-fold from **-\$60** to **-\$4,920/kg** reduction.

Table A2.1: Farm-level benefits/costs (\$/Kg) of pesticide reductions reported in Poggio et al. (2014)

	Tully			Mackay			Burdekin Delta			Burdekin BRIA		
Farm size (ha)	50	150	250	50	150	250	50	150	250	50	150	250
Bare & High Tillage												
C- to B-Class	\$2,230	\$2,470	\$2,280	\$1,880	\$2,250	\$2,090	\$750	\$910	\$910	\$360	\$440	\$440
C- to A-Class	\$770	\$1,500	\$1,540	-\$60	\$1,430	\$1,850	\$200	\$650	\$750	\$70	\$380	\$340
B- to A-Class	-\$870	\$500	\$750	-\$4,920	-\$550	\$1,100	-\$610	\$300	\$520	-\$280	\$300	\$220
Legume & Low Tillage												
C- to B-Class	\$4,440	\$4,910	\$4,530	\$3,020	\$3,610	\$3,370	\$520	\$820	\$880	\$310	\$480	\$520
C- to A-Class	\$1,220	\$2,390	\$2,440	\$0	\$2,180	\$2,760	-\$60	\$980	\$1,250	-\$60	\$560	\$720
B- to A-Class	-\$1,140	\$680	\$1,020	-\$740	-\$410	\$1,510	-\$1,410	\$1,070	\$1,700	-\$840	\$590	\$940

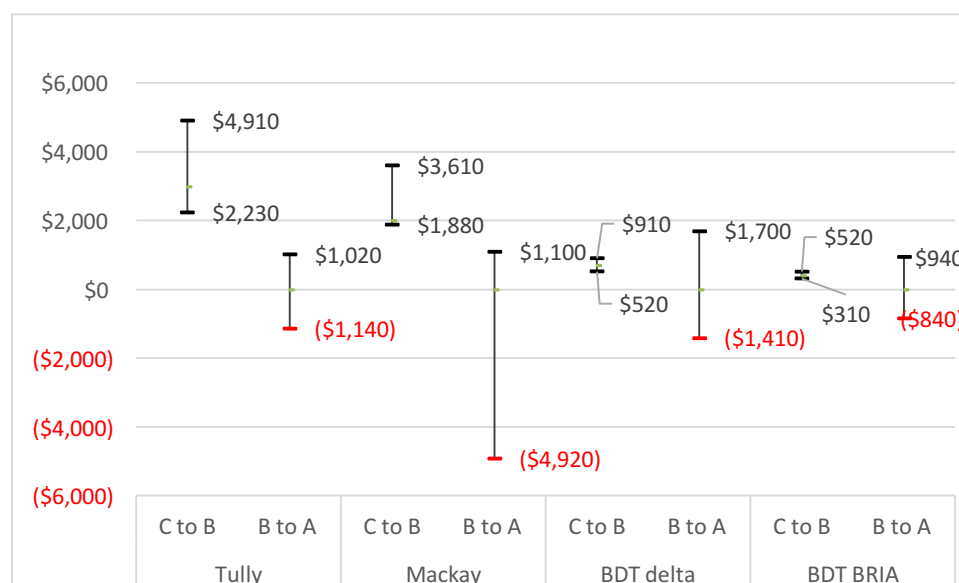


Figure A2.1: Private benefit/cost (\$/kg) variation for pesticide reduction from practice change

Van Grieken et al. 2010, 2013

Bio-economic modelling conducted by van Grieken and others, combined economic analysis with a bio-physical crop model (APSIM) and other bio-physical models. The results, also

provide evidence of wide ranging positive and negative returns associated with improving land management practices.

Van Grieken et al. (2010) provide details of positive and negative economic impacts of moving from D to C; C to B and B to A class practices, in the same four regions covered in Poggio et al. (2014). However no details of emission reductions were provided and no inference can be made about relative abatement costs.

Van Grieken et al. (2013) estimate and compare the net private benefits or costs of achieving **end of catchment** dissolved inorganic nitrogen (DIN) loads in the Tully and Mackay regions. Under a scenario where all farmer move to either A or B class practice, in the short run (5 years) it is more cost-effective to reduce DIN in the **Tully-Murray catchment (\$13,163/t)** than in the **Pioneer (\$-1,204/t)**, but with little difference between the regions in the long run (10 years) (Table). Under a more stringent scenario where all farmer move to A class practice, in the short run the costs of additional DIN reductions are similar, but In the long run, it is considerably more cost-effective to reduce an additional tonne of DIN in the **Pioneer catchment (\$11,312/t)** than it is in the **Tully-Murray (\$-8,453/t)** (Table A2.2).

Table A2.2: Net private benefit of reducing (end of catchment) DIN loads

	Tully-Murray catchment (Tully)		Pioneer catchment (Mackay)	
<i>Time frame</i>	<i>5 yrs</i>	<i>10yrs</i>	<i>5 yrs</i>	<i>10yrs</i>
All farms are either in management practice class A or B	\$13,163/t	\$20,880/t	\$-1,204/t	\$22,066/t
All farms in management practice class A	\$-21,579/t	\$-8,453/t	\$-22,106/t	\$11,312 t

Van Grieken et al. 2014a (Burnett Mary WQIP)

Van Grieken et al. (2014a) conducted the bio-economic modelling and economic analysis for the sugarcane industry to inform the Burnett Mary Regional WQIP. They combined previous Farm Economic Analysis Tool (FEAT) modelling, a crop production model (APSIM) and local expert knowledge. The analysis was conducted across:

- three farm sizes size (small 75 ha; medium 125 ha; large 250 ha) and two soil types (well drained and less well drained) for sugarcane
- changes from D-C, C-B and B-A class practices (without specific focus on a specific management practice)

The authors estimated the required level of incentives to adopt improved practices, based on three cost components (capital costs, profitability and non-financial costs), with two separate categories (transaction costs and 'other' non-profit costs) to account for the non-profit barriers to adoption.

Non-financial costs

To estimate the transaction cost (non-profit) barriers to adoption, it was assumed that transaction costs for the WQIP would be similar to the values estimated by Coggan et al. (2013) for Reef Rescue (\$9,026 per farm). These results were extrapolated to determine a 'medium' level of non-profit barrier of \$120/ha for small farms, \$72/ha for medium and \$36/ha

for large farms. Values were then adjusted to account for different non-profit barrier ratings and with different rates applied to account for transaction costs and 'other' non-financial reasons (Table A2.3).

Table A2.3: Non-financial barrier estimate adjustments for sugarcane

Non-profit barrier category	Transaction costs	'other'
• Low non-profit barriers	Medium –20%	Medium –40%
• Medium to high	Medium +10%	Medium +20%
• High	Medium +20%	Medium +30%
• Very High	Medium +30%	Medium +40%

Details of some practice improvements are summarised below (more scenarios are modelled in the report) in Table A2.4 with full information presented in Van Grieken et al. (2014a: Table 7).

Table A2.4: Net annual equivalent benefits (AEB) of improving management practices (\$/ha/yr)

<i>CHANGE (over 10 years at 6%)</i>	Soil type:	Red Dermosol (well drained)			Redoxic Hydrosol (less well drnd)		
	Farm size:	Small	Medium	Large	Small	Medium	Large
D to C Class							
AEB (\$/ha/year)		\$190	\$218	\$223	\$154	\$181	\$187
Annual non-profit costs (\$/ha)		\$36	\$22	\$11	\$36	\$22	\$11
Transaction costs (\$/ha at year 1)		\$96	\$58	\$29	\$96	\$58	\$29
Net AEB (\$/ha/year)		\$141	\$189	\$209	\$105	\$152	\$172
AEB reduct re non-financial costs		35%	15%	7%	47%	19%	9%
C to B Class							
AEB (\$/ha/year)		\$4	\$100	\$105	\$75	\$172	\$176
Annual non-profit costs (\$/ha)		\$60	\$36	\$11	\$60	\$36	\$11
Transaction costs (\$/ha at year 1)		\$120	\$72	\$29	\$120	\$72	\$29
Net AEB (\$/ha/year)		-\$72	\$54	\$90	-\$1	\$126	\$161
AEB reduct re non-financial costs			85%	17%		37%	9%
B to A Class							
AEB (\$/ha/year)		-\$208	-\$125	-\$109	-\$211	-\$127	-\$111
Annual non-profit costs (\$/ha)		\$84	\$50	\$22	\$84	\$50	\$22
Transaction costs (\$/ha at year 1)		\$144	\$86	\$40	\$144	\$86	\$40
Net AEB (\$/ha/year)		-\$312	-\$187	-\$136	-\$314	-\$189	-\$138
Net cost increase non-financials		50%	50%	25%	49%	49%	24%

The modelling results indicated that on sugarcane farms, moving from D to C and/or B class practice was profitable on farms of all sizes (small, medium, large), although low levels of adoption would be likely on small farms. Adoption of A class practice generally incurred a cost.

The inclusion of non-financial costs has a significant impact on the economic viability of practice change. When moving from D to C class practice, including non-financial costs reduces the profitability by an average of 22% across all farm sizes and soil types. The net impact is stronger, particularly on small and medium sized farms, when moving from C to B

class practice. The inclusion of non-financial costs, increases the net loss associated with moving from B to A class practice, by an average of 41%.

A2.2 Grazing

Star et al. 2011, 2013, 2015a

Star et al. (2011) explored the net private benefit regenerating land condition in grazing enterprises from 'D' condition to 'B' condition (land condition classes are not the same as management practice classes referred to elsewhere), on two land types in the Fitzroy catchment. The results indicated that land regeneration was an economically feasible option on a Brigalow-Blackbutt land type, although the benefit cost ratios and sensitivity testing demonstrated that in some cases net private returns were negative. In contrast, land regeneration on a Narrow-leaved Ironbark land type (with the potential to achieve the largest reduction in sediment runoff and therefore greatest water quality improvements), was not found to be profitable.

The economic analysis was then extended to include five land types and combined with GRASP, a pasture utilisation model, so that the potential sediment reductions could also be estimated (Star et al. 2013). The results indicated that the **cheapest options** for sediment reductions were from the **lower productivity land types** and from farms in **poor (C) condition** (Table A2.5). The marginal costs of sediment reduction increase sharply as higher levels of reduction are sought.

The costs of achieving sediment reductions varied by more than 100 times over the selected case studies, with opportunity costs very sensitive to the existing land condition. The cost of reducing sediment emissions varied from a low of **\$4 per tonne** for the Silver-leaved Ironbark on duplex soils land type in poor condition to a high of **\$421 per tonne** for the Brigalow-Blackbutt land type in very good condition (Table A2.5).

Table A2.5: Cost effectiveness (\$/t) of end-of-paddock sediment reduction across land types

Land type	Land condition		
	A	B	C
Brigalow Blackbutt	421	391	59
Narrow leaved Ironbark Woodlands	6	21	6
Coolibah Floodplains	362	291	80
Narrow Leaved Ironbark Ranges	12	7	7
Silver leaved Ironbark – Duplex	25	21	4

Additional layers of cost heterogeneity were revealed when the models were modified by including different 20-year periods for a climate cycle. Two 20-year climate periods were selected for comparison with one cycle starting in 1912 and the other in 1981. Significant variations in net present value over the 20-year weather period were recorded for the two weather cycles (Star et al. 2015a).

Star et al. 2015b (Fitzroy Basin WQIP)

The main aim of the study by Star et al. (2015b) was to allocate investments spatially by prioritising neighborhood catchments (sub-locations) within the Fitzroy region to achieve the most environmental benefit and cost effective outcomes. This approach involved modelling climate, biophysical and economic data to predict where estimated pollutant reductions could be made at the lowest cost. Four key variables and data layers that were incorporated in the process:

1. **Sediment loads** (SOURCE modelling data)
2. **Land condition** (Residual cover data)
3. **Management effectiveness** (Management practice survey data is collected currently in the grains BMP, or through the Queensland Department of Agriculture and Forestry Paddock to Reef monitoring and evaluation program)
4. **Cost** data (up-front capital and opportunity costs based on expert opinion)

The **up-front capital costs or incentive costs** were assigned a nominal value of \$5,000 for grazing and \$2,000 for cropping, based on a review of the past on-ground investments under Reef Rescue. This covered activities such as fencing off affected sites, revegetation, contour banks and/or drop structures. It was assumed that these management practices would likely be appropriate for most sites regardless of soil erosion type, (e.g., gully, hillslope or streambank erosion), erosion severity and site-specific characteristics.

The **opportunity costs** were the estimated changes in gross margins (over ten years) based on a 5,000 ha finishing enterprise which had been estimated at \$218 per adult equivalent adjusted for the Eastern Young Cattle Index as of June 2015 to \$240 (Star et al. 2011). The marginal change in production was informed by the bio-economic outcomes and estimated to be 27% on average across the catchment (Star et al. 2013).

For cropping land, contour banks were the main erosion control considered, with the opportunity cost estimated to be approximately 20% of gross margins. Gross margins was averaged and weighted based on the combinations of crops and the ratio planted over a 10-year timeframe, which was based on the Paddock to Reef modelling.

However, as the cost data information presented in the report had been normalized, and the reported results do not provide estimates of the cost per unit of pollution reduction which might then be used for comparison elsewhere. The results are specific to the situation for which they were commissioned.

Pannell et al. 2014 (Burnett Mary WQIP)

Pannell et al. (2014) conducted the bio-economic modelling and economic analysis for the grazing industry to inform the Burnett Mary regional WQIP. Results indicated that all practice changes across all farm sizes were estimated to come at a cost, once non-financial barriers were considered.

The results were presented on a cost per hectare basis and no details were provided about the unit cost of pollutant emission reduction.

The analysis was conducted across:

- three farm sizes size (small 288ha, medium 880 ha, large 4134 ha) and land productivity classes (high, medium and low)
- four specific management practice changes
 - Practice 1. Average stocking rates imposed on paddocks are consistent with district long-term carrying capacity benchmarks for comparable land types, current land condition and level of property development
 - Practice 2. Retention of adequate pasture and groundcover at the end of the dry season (pasture assessment and stock management) informed by (a) knowledge of groundcover needs and (b) deliberate assessment of pasture availability in relation to stocking rate in each paddock during the latter half of the growing season or early dry season
 - Practice 3. Strategies implemented to recover any land in poor or very poor condition (C or D class)
 - Practice 4. The condition of selectively grazed land types is effectively managed
- changes from D-C, C-B and B-A class practices

The authors estimated the required level of incentives to adopt improved practices, based on three components with input from local experts:

1. the up-front costs required for adoption
2. changes in the annual profitability of land use following adoption
3. non-profit barriers to adoption

The **non-profit barriers to adoption** were estimated by applying the subjective opinion of local experts. First, each practice shift was assigned a non-profit barrier rating of low, medium, high or very high. Equivalent dollar values for non-profit barriers were estimated subjectively for each rating, and for each level of practice profitability. These dollar values were anchored by values of \$75 per ha per year for very high non-profit barriers under low profitability, and zero for low non-profit barriers under high profitability. The \$75 value was chosen to be similar to a typical level of annual profit in this farming system. Incentive payments were scaled to zero where there was estimated to be high profitability and low non-profit related barriers.

Estimated incentive payments for adoption of improved practices **ranged from \$10 to \$160 per hectare per year**, across different practices in different circumstances. Required payments per hectare were highest for small farms relative to medium and large farms. They were also higher for low-productivity farms. In general, the highest incentive payments were required for practices that involve high up-front costs (practice 3, shift from D to B; practice 4, shift from C to B) and those with very high non-profit barriers (practice 1, shift from D to C; practice 2, shift from C to B).

The results of Pannell et al. 2014, accounting for non-profit barriers and the lower financial capacity of small farms adopting practices which involve high upfront costs, suggest that for some practices, incentive payments need to be **higher on low productivity land types**. This is in direct contrast to the findings of Star et al. (2013) which indicated that **the most cost-effective sediment reductions** might be achieved on **low productivity land types**. This difference in results of the two studies highlights the care that needs to be taken when applying the results from bio-economic modelling studies, and the importance of understanding the underlying components and assumptions in the modelled results. For

example, Star et al. (2013) estimate the costs of sediment reduction (\$/t) from different land types and land condition classes, whereas Pannell et al. 2014 estimate the cost (\$/ha/yr) of changes in management practice (ABCD) change and do not report the corresponding load reductions. Star et al. (2013) do not attempt to estimate the non-financial barriers to adoption, which are known to be important, and which are included in the Pannell et al. analysis.

A2.3 Summary of bioeconomic modelling

There is no one correct way to estimate the costs of achieving emission load reductions. However, the lack of consistency in approach and reporting has meant that none of the results from the different studies outlined above (Table 7) are directly comparable and do not accurately reveal some of the cross sector and regional heterogeneity in abatement costs.

Table A2.6: Overview of the outputs from bio-economic modelling studies

	Pollutant	Private cost	Location	Reported results
Sugarcane				
Poggio et al. 2014	Pesticide End of paddock	AEB (10yr @6%)	Tully; Mackay; Burdekin (x2)	\$/kg pesticide reduction across ABCD class practice change
Van Grieken et al. 2010	n/a	NPV (10 + 5 yr @ 7%)	Tully; Mackay; Burdekin (x2)	NPV of practice (ABCD) change
Van Grieken et al. 2013	DIN End of catchment	NPV (10 + 5 yr @ 7%)	Tully; Mackay	NPV of practice (ABCD) change
Van Grieken et al. 2014	n/a	AEB (10yr @6%) + non- financial	Burnett Mary WQIP	\$/ha/yr for practice (ABCD) change
Grazing				
Star et al. 2011, 2013, 2015a	Sediment End of paddock	NPV (20yr @6%)	Fitzroy Basin	\$/t sediment reduction across land types and land condition classes
Star et al. 2015b <i>Includes grains</i>	Sediment End of catchment	Capital + operating	Fitzroy WQIP	Numeric Score WQIP reported \$/t sediment reduction.
Pannell et al. 2014	n/a	Capital + operating + non-financial	Burnett Mary WQIP	\$/ha/yr for practice (ABCD) change

The private benefit/cost of **end of paddock Pesticide reduction** (Poggio et al. 2014) ranged from:

- a private cost of **-\$4,920/kg** (moving from B to A class practice in the Mackay region) to a **benefit of \$4,910/kg** (moving from C to B class practice in the Tully region).

The private benefit/cost of **end of catchment DIN reduction** (Van Grieken et al. 2013) ranged from:

- a private cost of **-\$22,106/t (-\$22/kg)** (all farms moving to A class practice in the Mackay region within 5 years) to a **benefit of \$22,066/t (22/kg)** (all farms moving to either A or B class practice in the Mackay region).

The private cost of **end of paddock sediment reduction** (Star et al. 2013) ranged from:

- a private **cost of \$421/t** (from Brigalow Blackbutt land type in A condition) to a cost of **\$4/t** from Silver-leaf Ironbark land type in C condition) in the Fitzroy region.

The private cost of **end of catchment sediment reduction** (Star et al. 2015b) ranged from:

- a private **cost of \$9,779/t** to **\$18.83/t** in the Fitzroy region, with an **average cost of \$277/t** to achieve a 20% reduction.

