Adoption of agricultural management for Great Barrier Reef water quality improvement in heterogeneous farming communities¹

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Abstract

There is growing recognition that coastal water quality is interdependent with agricultural management in coastal catchments. Economic-incentive-based instruments can be used to internalize the negative externalities from coastal water pollution. Bio-physical and socio-economic heterogeneity across farms is expected to be an important factor in explaining differing rates of adoption of management practices. This paper hypothesises that: i) different types of farmers are likely to respond differently to incentive payments that promote the adoption of management practices for Great Barrier Reef water quality improvement, and ii) if policy makers account for heterogeneity, cost-effectiveness of incentive payments will increase. Results show that if government paid farmers 100% of the transition costs of moving from their current practices to improved practices, and given current technologies, water quality improvement for the case-study region would be approximately 56% (measured as a reduction of dissolved organic nitrogen at the end of river). Total costs for the region would be almost AU\$ 30M over the planning horizon of one cropping cycle of 6 years. Results furthermore show that as the policy seeks more ambitious land-use changes, from common practice to improved practice to aspirational practice, the public cost of incentive payments increase at an exponential rate. Larger farms make the shift sooner as they are able to offset fixed investments against increased revenues over greater areas.

Keywords

Land use, land management, heterogeneity, farming community, natural resource policy, agri-environmental policy, Great Barrier Reef

1. Introduction

Coastal and marine ecosystems around the world are threatened by disturbances caused by human activity (Brodie et al. 2012, Burke et al. 2011). Population growth, climate change and the intensification of industrial and agricultural activity are likely to increase eutrophication of estuarine and coastal waters (Rabalais et al. 2009). For example, nonpoint-source pollution from agricultural activities is linked to the degradation of coastal ecosystems in the World-Heritage-listed Great Barrier Reef (GBR) in Australia (Kroon 2011, Kroon et al. 2011). Sources that contribute to the pollution of the GBR lagoon have been identified and include suspended sediment from soil erosion in cattle grazing areas; nitrate run-off from fertiliser application on crop lands; and herbicide run-off from various land uses (Brodie et al. 2012). To address the issue of water pollution, improved management of resources in coastal catchments is needed (Doney 2010, Smith and Schindler 2009).

A range of agricultural management practices has been developed to address water pollution from sugarcane farms while maintaining farm production (Roebeling et al. 2009a, Schroeder et al. 2009, Thorburn et al. 2011). Examples include reductions in tillage operations, reductions in fertiliser application rates, the use of legume crops and the use of global positioning system (GPS) guidance for machinery (Van Grieken et al. 2010b). Adoption of these and other agricultural management practices and associated costs and benefits are impacted by the level of bio-physical and socio-economic heterogeneity within the community. While the bio-physical heterogeneity in the GBR catchment area is well understood and modelled (Armour et al. 2009, Thorburn et al. 2011), socio-economic heterogeneity has only been addressed qualitatively (Bohnet 2008, Emtage et al. 2006). Landholder classification can, however, be used to better understand how heterogeneity in socio-economic circumstances influence the costs and benefits of adoption of new technologies as well as the success of agri-environmental policy (Doole and Pannell 2011).

Pannell (2008) categorises policy mechanisms for pollution mitigation into *positive incentives* (financial or regulatory instruments to encourage change), *negative incentives* (financial or regulatory instruments to inhibit change), *extension* (e.g. technology transfer, education), *technology development* (e.g. strategic or participatory research and development to generate improved technologies) and *no action* (or informed inaction). He develops a framework for selecting policy instruments to encourage the adoption of

environmental practices, with the recommended policy instrument depending on the levels of private and public net benefits. Other factors influencing the choice of policy instruments in this framework included the learning costs associated with adopting new technologies and transaction costs involved with the implementation of a policy instrument. Pannell and Wilkinson (2009) build on this framework to account for the differences between lifestyle landholders and commercial farmers. These differences (e.g. smaller properties, lack of management skills and lack of time for land-management activities) likely change the private and public net benefits and, hence, the choice of effective policy instruments.

Doole and Pannell (2011) explore the effect of landholder heterogeneity on the spatial targeting of policies for control of agricultural nonpoint-source pollution. They conclude that a differentiated policy instrument (targeting farms with low abatement costs) results in cost savings relative to a uniform standard (all farms treated equally), but also report that potential cost savings from a differentiated policy could easily be outweighed by the associated transaction or policy costs. With regards to market-based instruments, the level of heterogeneity in abatement costs among landholders determines the level of potential cost savings from targeting (Newell and Stavins 2003). If the cost of pollution control varies little amongst landholders, a uniform policy may be favourable due to lower administration and transaction costs, while targeted policy actions may be more favourable if pollution control costs vary considerably. Furthermore, Villegas-Palacio and Coria (2010) show that under a differentiated policy instrument (tradable emission permits) the incentive to comply increases with the rate of adoption whereas under a uniform policy (taxes) this remains the same.

Where previous research to estimate the potential socio-economic and environmental impacts of economic instruments for improving water quality has focussed on incorporating bio-physical heterogeneity (Roebeling et al. 2009a, 2009b, 2014; Van Grieken et al. 2011a, 2011b) as well as enterprise-structural heterogeneity (Doole and Pannell 2011), this paper contributes to existing literature by exploring the implications of both bio-physical and socio-economic heterogeneity, including differences in soil type, property size, labour availability and transition costs. Hence, the objective of this study is to quantify the consequences of heterogeneity for the performance of an economic-incentives-based policy instrument for water-quality improvement. We model how different types of farmers (upstream users) are likely to respond to the economic-incentives-based approach where transition-cost-sharing ratios are varied between zero cost sharing

(0%; upstream polluter pays) and full cost sharing (100%; downstream beneficiary pays). We define transition costs as the upfront capital investments required to adjust the current farming system to the improved farming system.

This approach provides policy makers with important information regarding the cost-effectiveness of the policy instrument, but also on likely farm-type-specific economic indicators, such as adoption rates. It will furthermore shed light on the effectiveness of the instrument at varying levels of cost-sharing, while providing an indication of the private and public costs involved in the program. As the information provides an indication of potential public funds required to achieve a water quality improvement target (and/or associated adoption rates among the landholders), policy makers can discuss the pathways to acquiring these funds. This could be public money or, alternatively, it could be raised from the downstream beneficiaries of the improved water quality in the GBR lagoon (e.g. via tourism taxes).

2. Case study

The Tully–Murray catchment (approximately 18°S 146°E, Figure 1) in the GBR region has been identified as an important source of nutrients and sediment (Armour et al. 2009, Kroon 2009). This catchment in the Wet Tropics bioregion is characterised by high annual average rainfall (4100 mm, Bureau of Meteorology, 2010). Within the vicinity of the Tully-Murray flood plume there are 37 coral reefs and 13 seagrass meadows (Devlin and Schaffelke 2009).

Figure 1: The Tully-Murray catchment in tropical North Queensland, Australia.

Sugarcane production in the catchment contributes almost 45% to regional agricultural income (Roebeling et al. 2009a). The majority of sugarcane grown in GBR catchments is located on coastal floodplains. Water draining from sugarcane farms can reach the GBR lagoon quickly, leaving little opportunity for in-river processes to remove pollutants from the flow (Furnass and Mitchel 2001). Nitrogen (N) losses in water from sugarcane production occur through two pathways: leaching via deep drainage below the root zone and/or in surface runoff. Reported losses via deep drainage have ranged from less than 10, to greater than 70 kg N/ha/yr. In the Wet Tropics, Prove et al. (1994) reported losses of 62 kg N/ha/yr, Moody et al. (1996) reported losses averaging 30-50 kg N/ha/yr, and Webster et al. (2012) reported losses between 11 and 26 kg N/ha/yr.

There is sparse data reporting N losses in surface runoff from sugarcane production, ranging from a few to approximately 20 kg N/ha/yr (Bengtson et al. 1998, Ng Kee Kwong et al. 2002). In the Wet Tropics, N losses in runoff have been measured at less than 10 kg N/ha/yr (Webster et al. 2012). Webster et al. (2012) reported that N losses from surface water run-off and deep drainage are reduced when N application rates are reduced. They propose that the rate of N application is the most important determinant of N pollution, being more important than other management practices such as surface versus sub-surface fertiliser application.

Most of the N lost from sugarcane farms in surface water will reach the GBR lagoon (Furnass and Mitchell 2001, Mitchell et al. 1997). However, pathways of sub-surface flow are highly variable, with N lost via deep drainage having the potential to move to surface water (Rasiah et al. 2010), or to remain deep in the soil profile (Rasiah et al. 2003a, Rasiah et al. 2003b). Webster et al. (2012) hypothesized that more than 60% of the N lost through deep drainage reaches surface water via lateral flow, consistent with the findings of Rasiah et al. (2005) that N loss via deep drainage is the main source of anthropogenic N reaching the GBR.

3. Methods

In this paper we use a bio-economic modelling approach (for example Janssen and Van Ittersum, 2007, Kanellopoulos et al. 2010 and Kanellopoulos et al. 2014), that combines financial-economic and environmental analyses of farming systems at the paddock scale, differentiates three farm typologies at the farm scale, and aggregates results to the catchment scale, to explore the cost-effectiveness of improving catchment water quality via the adoption of changed land management practices in a heterogeneous farming community. We model how different types of farmers are likely to respond to an economic incentives-based approach where the private cost of transitioning to improved management practices is shared between the landholder (upstream beneficiary) and the public (downstream beneficiaries). We thereby explore the extent to which differentiating incentive payments between farm types leads to more cost-effective policy. Transition costs are represented by the upfront capital investments required to establish the new management practice.

3.1 Farm enterprise typology

A classification of sugarcane farmers in the Tully-Murray catchment was based on Van Grieken et al. (2009). The resultant landholder profile comprised three types of sugarcane farming agents, broadly based on farm size:

- Landholders with large sugarcane farms (180-1230 ha; farm type 1) who are willing to adopt improved management practices in the case where it will provide them with financial benefits. They have a knowledge-seeking attitude and from an economic point of view, their behaviour could be described as profit-maximising. In response to policy interventions, landholders with large sugarcane farms have a positive attitude towards incentive schemes such as payments for pollution control. They look closely at how the business operates and have a reasonable knowledge of their costs of production.
- Landholders with medium-sized sugarcane farms (80-180 ha; farm type 2) who have a more cautious attitude to the adoption of new farming systems. Of the three landholder types in this study, landholders with the medium sized sugarcane farms experience the largest trade-off between labour and leisure time and, hence, sacrificing leisure time to change farming systems is seen as a constraint. Landholders with medium-sized sugarcane farms tend to be traditionalists. Their most important source of information is other cane farmers.
- Landholders with small sugarcane farms (20-80 ha; farm type 3) who are tightly constrained by their availability of capital, labour and time, and who are least likely to adopt new practices. Most of these farmers work off farm so they need to hire-in farm labour, which comes at a substantial cost. Landholders with small sugarcane farms are likely to sell their land if restrictions are too binding. Because landholders with small sugarcane farms work mainly off-farm, exposure to extension material or time to learn about better farming techniques is limited.

For the Tully-Murray case study, this landholder profile is matched with geographical information, through intersection of farm property information from the Digital Cadastral Data Base (DCDB), land use and soil type data layers from Roebeling et al. (2009a), using the geographical information system software ArcGIS® by ESRI. This allows for the identification and characterisation of 418 heterogeneous agents according to their

agricultural production system, economic features (farm size) and spatially explicit agro-ecological conditions (soil typology). Table 1 shows the number of farms for each farm type in the case study area.

Table 1: Farm size (in hectares) for the different farm types in the Tully-Murray catchment.

3.2 Management classes for water quality improvement

The management classes represented in the model have been adopted from the ‘ABCD’ framework of farming system classification used by natural resource management (NRM) regional bodies for the Reef Rescue program (Higham et al. 2008). The ‘ABCD’ framework provides a standardised farming systems framework adopted in all regions. The ‘ABCD’ framework describes management practices that range from degrading (D), common (C), ‘best’ (B), through to aspirational or cutting edge (A) regarding water-quality improvement. Examples of A-class practices are reduced nitrogen application rates, legume fallow management and zero tillage. Practices are allocated to these classes depending on 1) the resource condition achieved by adopting the management class in the short, medium and long term; 2) the acceptability of the management class to the community; and 3) the feasibility of achieving wide adoption of the management class in the short, medium and long terms.

3.3 Productivity and water quality indicators

Sugarcane input-output data for different management practices were generated using the APSIM cropping systems model (Keating et al. 2003). The APSIM model was used because it is capable of modelling N-cycling in sugarcane production systems, e.g. Thorburn et al. (2011), and has been used previously in bio-economic modelling in the Tully region (Roebeling et al. 2009a, Van Grieken et al. 2011). APSIM was used to predict regional soil-type-specific productivity (sugarcane yield) and the environmental indicator for water quality (total available dissolved inorganic N (DIN) to leave a paddock) (Van Grieken et al. 2011a, 2011b). The APSIM cropping system simulator provides plot-scale DIN in runoff and DIN leached below the root zone for each management practice. In our framework we assume that 100% of DIN in runoff reaches the GBR lagoon (Furnas and Mitchell 2001) and 60% of N in deep drainage reaches the GBR lagoon (Webster et al. 2012).

3.4 Transition costs

In this paper, we refer to transition costs as a one-off upfront capital investment for new and or modified equipment. Information on upfront capital investment specific to the Wet Tropics region was obtained from research by Van Grieken et al. (2010a) and adjusted by local experts to account for variations in farm size. Table 2 presents investment costs for farms that change management classes from C-class management to B-class, B-class to A-class and C-class to A-class. Examples of capital investments required to move from C-class to B-class are the purchase of a stool splitter fertiliser box and modifications to the sprayer, harvester and farm tractor (Van Grieken et al., 2010a).

Table 2: Investment costs associated with changing management classes across farm types in the Tully-Murray catchment (based on: Van Grieken et al., 2010a).

Finally, each management class produces a farm gross margin (FGM) on each of the dominant soil types represented in the region. Parameterisation of these soil types can be found in Thorburn et al. (2011). FGM is calculated as the result of cane revenues minus the cost for growing and harvesting the cane, and presented in dollars per hectare per year (based on Van Grieken et al., 2010a). Table 3 presents the FGM across all farm and soil types in the region, as well as the distribution of the soil types (%) in the region.

Table 3: FGM across farm and soil types and soil type distribution in the Tully-Murray catchment (based on: Van Grieken et al., 2010a).

4. Framework

To explore the future level of adoption of environmentally preferred management classes in the Tully-Murray catchment under varying cost-sharing ratios, and to determine the cost-effectiveness of the policy, we use a linear constrained optimisation farm model in which sugarcane producers are characterised according to their specific objectives, production systems, and agro-ecological as well as economic constraints (see Section 3). By adjusting the investment constraints of the agents within the model, symbolising the sharing of costs by the public sector, we assess how different types of sugarcane producers in the Tully-Murray catchment are likely to respond to the cost-sharing policies that aim to promote the adoption of new management classes.

Each farm enterprise is taken to maximize income, which is defined as the gross value from sugarcane production minus the costs of production – corrected for private costs and/or benefits from policy interventions. The most important constraints faced by each farm type are related to production systems and land. As stated before, the management classes available to agricultural producers are sugarcane management according to the ‘ABCD’ framework. As we estimate short-term responses to policy instruments, land use is constrained to the currently available agricultural area. The model simulates production, income, resource use, employment and water pollution at the farm and, through aggregation, at the regional level for identified sugarcane farms in the study area. The model, which is solved using GAMS 22.7 (Brooke et al. 1998), represents a production cycle of six years (corresponding to one full sugarcane cropping cycle). Discounted values have been used for the objective function to maximize the Net Present Value. The model is structured as follows (while disregarding time subscripts).

The model represents sugarcane production by all farms (428; indexed by id) of various farm types (large, medium-sized and small sugarcane farms) in the region. Each farm comprises an area a_{id} (in ha) with a specific distribution of soil types (4; indexed by s), and $X_{id,s,c}$ is a set of decision variables that indicates the area per soil type (s) over which each farm operates using a particular management class (c ; representing options from the ABCD framework). Hence, the land balance is given by:

$$\sum_c X_{id,s,c} \leq a_{id,s} \quad (1)$$

Total farm production costs C_{id} (in AU\$/year) are now determined by the sum of growing, harvesting and transition/investment costs, such that:

$$\begin{aligned} C_{id} = & \sum_{s,c} p^{Gro} \gamma_c X_{id,s,c} + \sum_{s,c} p^{Har} \eta_{s,c} X_{id,s,c} \\ & + (1 - CSR_{id}) \sum_{s,c} p^{Tra} \tau_{id,c} X_{id,s,c} \end{aligned} \quad (2)$$

where γ is the sugarcane grown (per management class; in units/ha/year), η the sugarcane yield (per soil type and management class; in t/ha/year) and τ are the investment requirements (per farm type and management class; in units/ha/year) at corresponding input prices p^{Gro} (in AU\$/unit), harvesting machinery prices p^{Har} (in

AU\$/t) and transition/investment prices p^{Tra} (in AU\$/unit), Note that $p^{Tra}\tau$ is shared between landholders and the downstream beneficiaries, depending on the cost-sharing ratio (CSR ; 0% refers to no cost-sharing and 100% refers to full cost-sharing).

Total farm benefits B_{id} (in AU\$/year) are determined by the sale of sugarcane, such that:

$$B_{id} = \sum_{s,c} p^{Sug} \eta_{s,c} X_{id,s,c} \quad (3)$$

where η is the sugarcane yield (per soil type and management class; in t/ha/year) and p^{Sug} the sugarcane price (in AU\$/t).

Net farm income π_{id} (in AU\$/year) equals total farm benefits B_{id} (in AU\$/year) minus production costs C_{id} (in AU\$/year). Hence:

$$\pi_{id} = B_{id} - C_{id} \quad (4)$$

The objective function is to maximise regional income Π (in AU\$/year):

$$Max \Pi = \sum_{id} \pi_{id} \quad (5)$$

such that all farms (id) contribute to regional income.

Finally, water pollution ρ (in t DIN/ha/year) resulting from sugarcane production is specific per soil type (s) and management class (c). Regional water pollution P (in t DIN/year) is given by:

$$P = \sum_{id,s,c} \rho_{s,c} X_{id,s,c} \quad (6)$$

5. Results

5.1 Base scenario

The base scenario determines the profit-maximising distribution of management classes across the landscape, accounting for characteristics such as farm size, gross margins per hectare, investment costs and labour

availability, and assuming that the government does not pay any share of the cost of land-use change. These results have been validated with local experts consisting of industry, government and research. In the base scenario (see Table 4), it is optimal to manage 9% of the available sugarcane land using aspirational (A) class practices, such as reduced nitrogen application rates, legume fallow management and zero tillage. Best-practice (B) management is seen on 66% of the sugarcane land, and 25% of the land is managed using common (C-class) practices..

5.2 Management class adoption and distribution

Table 4 shows the distribution of management classes per farm type as a percentage of total land under sugarcane, and is presented for cost sharing ratios (CSRs) between zero and 100 per cent in steps of 10%. Large farms (Farm type 1) initially operate in a mix of B-class and A-class management and start to shift more to A-class management from a CSR of 20% onwards. A minimum CSR of 90% is required to obtain full (100%) adoption of A-class management across the large farm type. Medium sized farms (Farm type 2) operate in B-class management and slowly change towards A-class management. Up to a 60% CSR, no A-class management is adopted. Only with a CSR of 90% is more land managed using A-class management than B- or C-class management. A CSR of 100% is needed to obtain full (100%) adoption of A-class management across the medium-size farm type. Finally, small farms (Farm type 3) operate mainly in C-class management. They shift to B-class management as soon as some of the transition costs are covered by public funds. At a CSR of 20%, about half of the land is operated under B-class management and the remainder is still operated under C-class management. At a 60% CSR, all land is operated under B-class management. As with the medium sized farms, not until full cost sharing is offered, land is fully operated under A-class management.

At the regional scale, management shifts from C and B-class towards A-class as costs are increasingly covered by the downstream beneficiaries (public funds). At a CSR of 50%, A-class management is adopted on 19% of the sugarcane land, and the majority of sugarcane land is still cultivated using B-class management (78%), with some C-class management (3%). This results in a decrease in water pollution by only 11%. At the highest CSR (100%), all capital and other transition costs are covered by public funds. In this case, all sugarcane land is cultivated using A-class management as it provides largest mean gross margins per hectare

(including the public payments) for all three farm types. In this case, water pollution decreases by 56% relative to the base case.

Table 4: Management class distribution (% of total land under sugarcane) and water quality improvement (WQI) per cost share scenario (CSR) at the farm and regional level.

5.3 Abatement benefits and costs

Model-estimated public abatement costs aggregated at the regional scale ($[IT]_{\text{Base}} - [IT]_{\text{CSR}}$), as well as water quality improvement relative to the base scenario (given in per cent reduction in DIN reaching the end of river; $([P]_{\text{CSR}} - [P]_{\text{Base}}) / [P]_{\text{Base}}$), are presented in Figure 2 for each of the three farm types.

Figure 2: Net abatement cost curves per farm type

Cost-effective policies require the adoption of management practices with lowest marginal costs within and/or across agricultural sectors (Roebeling et al. 2009a, 2009b). Up to approximately 12% water quality improvement, marginal abatement costs are lowest for farm type 1. Between 12% and 27% water quality improvement, marginal abatement costs are lowest across farm type 1 and 2. Water quality improvements beyond 27% are, finally, most cost-effectively achieved through adoption of management practices by farm type 3.

These insights warrant an exploration of the cost-effectiveness of two different incentive payment policy instruments: i) a uniform policy where all farm types receive equal CSRs, and ii) a differentiated policy where abatement reduction is subsidized per unit of abatement. By providing incentive payments per unit of reduction, thereby accounting for heterogeneity in the ability to reduce pollution at varying cost levels, net abatement costs are expected to be lower. These abatement cost curves, presented for both the uniform policy (where CSRs are provided irrespective of farm type) as well as the differentiated policy (where CSRs are prioritized based on farm type), are presented in Figure 3.

Figure 3: Net regional abatement cost curves for the uniform and differentiated incentive-based policy instrument.

Overall, abatement costs increase exponentially with increases in the level of water quality improvement. The largest differences in abatement costs, between the uniform and differentiated incentive-based policy instrument, are observed at water quality improvements of between 25% and 30%, with abatement costs being about 3.0 million AU\$ or 32% lower for the differentiated policy. For both policy options, a maximum water quality improvement of 56% can be obtained using A-class management, which comes at a net cost to the region of just under 30 million AU\$.

Discussion and conclusion

A bio-economic model of a heterogeneous cane-farming region was used to demonstrate that i) different types of farmers are likely to respond differently to incentive payments aiming to promote the adoption of management practices for GBR water quality improvement, and ii) if policy makers account for heterogeneity the cost-effectiveness of policy instruments is expected to increase. For the Tully-Murray catchment adjacent to the GBR lagoon, adoption rates were estimated by predicting how different types of farmers are likely to respond to financial incentive payments, where the payments are based on a specified share of the costs of transitioning from current to improved management classes. Furthermore, regional environmental-economic consequences of implementing the policy were estimated, such as changes in the supply of dissolved inorganic nitrogen at the end of the catchment and the net abatement costs required to meet water quality improvement targets.

If government paid farmers 100% of the transition costs of moving from their current practices to improved practices, and given current technologies, water quality improvement for the case-study region would be approximately 56% (measured as a reduction of dissolved organic nitrogen at the end of river). Total costs for the region would be almost AU\$ 30M over the planning horizon of one cropping cycle of 6 years. Results furthermore showed that as the policy seeks more ambitious land-use changes, from common practice to improved practice to aspirational practice, the public cost of incentive payments increase at an exponential rate, consistent with previous studies (e.g. Roberts et al. 2012). Larger farms make the shift sooner as they are able to offset fixed investments against increased revenues over greater areas.

In line with Pannell and Wilkinson (2009) we found that small and medium sugarcane farmers experience higher transition costs per hectare than do large farmers. They are furthermore less capable of offsetting any fixed costs due to smaller size properties. This results in varying levels of adoption of improved management at various levels of cost sharing. We conclude that heterogeneity is indeed an important factor in explaining differing rates of adoption of improved management classes in our case study area. In line with Doole and Pannell (2011) we conclude that a differentiated policy instrument, accounting for heterogeneity, may result in cost savings relative to a uniform standard. Newell and Stavins (2003) suggest that if the cost of pollution control varies little amongst landholders, a uniform policy may be favourable due to lower administration and transaction costs, while targeted policy actions may be more favourable if pollution control costs vary considerably. In this paper we have seen that, depending on the budget and/or water quality improvement target, abatement costs vary substantially between a differentiated and a uniform policy. Abatement costs differ to a maximum of 3.0 million AU\$ for a differentiated policy instrument (26% of the cost) at a water quality improvement of between 25% and 30%, while noting that this difference is more than 2.5 million AU\$ for water quality improvements between 17% and 40%. The Reef Water Quality Protection Plan (2009) defines a (minimum of) 50% reduction in nitrogen loads at the end of catchments. A 50% water quality improvement would result in a net abatement cost saving of 1.4 million AU\$ for the Tully-Murray catchment for a differentiated policy comparing to a uniform policy.

The analysis could be expanded to examine a greater range of management practices, such as nitrogen replacement and riparian vegetation. Further extensions could include regional constraints on the minimum level of sugarcane supply to the Tully mill (Van Grieken et al. 2011a) or the inclusion of downstream benefits for the tourism and fishery industry resulting from terrestrial water quality improvement and marine resource appreciation (Roebeling et al. 2009b). Secondly, policy instrument transaction costs could be included. Recent work from Coggan et al. (2014, 2016) indicates that private transaction costs (to the landholder) for a cost sharing policy in the GBR catchment area are substantial (both financial and time related) for all participants, unrelated to the size of the business. This may suggest an upward rotation of our predicted abatement cost curves, suggesting higher costs to achieve the same water quality improvement.

A number of biophysical and socio- and financial-economic caveats of this study must be mentioned. Firstly, abatement costs are based on the management practices assessed and, thus, do not include any alternative technologies that are currently under development. Secondly, given that our focus on one crop cycle, abatement costs are based on current land-use patterns and, consequently, gains from land-use change between industries are not taken into account. We judge that this is a reasonable simplification, since, even with subsidies (cost-sharing), expansion of sugarcane area is unlikely, as the area is already mostly cultivated under sugarcane. Thirdly, sugarcane productivity (yield) is modelled using APSIM and discriminates between soil types but not between farm types. This could potentially differ and needs to be investigated further. Small farmers may not be as productive per hectare as their larger counterparts. Fourthly, in this study it is assumed that labour is freely available for hire – which is not always the case in small towns or during peak labour periods. Fifthly, transaction costs of the incentive payment program are not included in the analyses, which could have implications for cost-effectiveness, especially on the adoption and delivery process. Pannell and Wilkinson (2009), for example, found that transaction costs per hectare of land are likely to be higher for small scale (lifestyle) farmers than for larger scale (commercial) farmers. However, as mentioned before, recent work from Coggan et al. (2014, 2016) suggests private transaction costs (to the landholder) for a cost-sharing policy in the GBR catchment area are unrelated to the size of the business. Sixthly, non-profit related drivers of adoption are not included in this analysis, which would likely affect the rate of adoption. Finally, the approach used here is deterministic and thus does not account for risk or uncertainty. Consequently, care should be taken when using the figures presented in this study for policy and planning purposes.

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Figures

Figure 1: The Tully-Murray catchment in tropical North Queensland, Australia.

Figure 2: Abatement cost curves per farm type

Figure 3: Regional abatement cost curves for the uniform and differentiated incentive-based policy instrument.

Tables

Table 1: Farm size (in hectares), mean farm size (in hectares) and the number of farms in the catchment for the different farm types in the Tully-Murray catchment.

	<i>Farm size (hectares)</i>	<i>Mean farm size (hectares)</i>	<i>Number of farms in the catchment</i>
<i>Farm type 1</i>	180 - 1230	428	22
<i>Farm type 2</i>	80 - 180	115	74
<i>Farm type 3</i>	20 - 80	40	322

Table 2: Investment costs associated with changing management classes across farm types in the Tully-Murray catchment (based on: Van Grieken et al., 2010a).

	<i>Investments farm type 1 (AU\$)</i>	<i>Investments farm type 2 (AU\$)</i>	<i>Investments farm type 3 (AU\$)</i>
<i>Management class change</i>			
<i>C-class to A-class</i>	203,500	147,000	116,500
<i>B-class to A-class</i>	128,000	88,000	66,000
<i>C-class to B-class</i>	75,500	59,000	44,250

Table 3: FGM across farm types and soil types in the Tully-Murray catchment.

<i>Soil type</i>	<i>A-Class FGM</i> <i>(AU\$/ha/yr)</i>	<i>B-Class FGM</i> <i>(AU\$/ha/yr)</i>	<i>C-Class FGM</i> <i>(AU\$/ha/yr)</i>
<i>Soil type 1</i>	1073	1022	813
<i>Soil type 2</i>	854	816	619
<i>Soil type 3</i>	810	772	587
<i>Soil type 4</i>	629	617	417

Table 4: Management class distribution (% of total land under sugarcane) and water quality improvement (WQI) per cost share scenario (CSR) at the farm and regional level.

	<i>Farm type 1</i>			<i>Farm type 2</i>			<i>Farm type 3</i>			<i>Region</i>			<i>WQI</i>
<i>CSR</i>	<i>A</i>	<i>B</i>	<i>C</i>	<i>A</i>	<i>B</i>	<i>C</i>	<i>A</i>	<i>B</i>	<i>C</i>	<i>A</i>	<i>B</i>	<i>C</i>	
Base	9%	22%	-	-	28%	-	-	17%	25%	9%	66%	25%	-
10%	9%	22%	-	-	28%	-	-	18%	24%	9%	68%	23%	0%
20%	13%	18%	-	-	28%	-	-	22%	20%	12%	68%	20%	3%
30%	13%	18%	-	-	28%	-	-	28%	14%	12%	74%	14%	5%
40%	18%	13%	-	-	28%	-	-	33%	9%	18%	74%	8%	9%
50%	19%	12%	-	-	28%	-	-	39%	3%	19%	78%	3%	11%
60%	21%	10%	-	1%	27%	-	-	42%	-	22%	78%	-	13%
70%	26%	5%	-	5%	23%	-	-	42%	-	31%	69%	-	18%
80%	29%	2%	-	12%	16%	-	3%	39%	-	44%	56%	-	26%
90%	31%	-	-	24%	4%	-	18%	24%	-	72%	28%	-	42%
100%	31%	-	-	28%	-	-	42%	-	-	100%	-	-	56%