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Integrating economic drivers of social change into agricultural water quality improvement strategies

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ABSTRACT

Water quality improvements achieved through changes in agricultural land use and management practices can impose a number of trade-offs on communities. Impacts such as reduced regional productivity may, for example, threaten farm profitability as well as the viability of the whole agricultural value chain in the region. This paper expands on existing studies, by quantifying not only the private (farm-scale) but also the likely direct social (regional-scale) impacts of water quality improvement – thus informing policy by identifying potential social consequences of private actions required to achieve sustainable load targets. We use the Environmental Economic Spatial Investment Prioritization model to explore the medium to long term private and social impacts of land use and management practice change for water quality improvement. By incorporating these economic drivers of social change we avoid underestimating the impact to the region, which could have severe consequences for local communities and policy makers. For a case study of the sugarcane industry in the Tully–Murray catchment in the Wet Tropics of Queensland (Australia), results show that long term regional productivity losses associated with desired water quality improvement targets increase the risk of sugar mill closure. This impact would likely have severe social and economic flow-on effects, such as decreased regional agricultural income and increased regional underemployment. These detrimental regional economic effects of water quality improvement plans have not been accounted for in previous studies, which have therefore underestimated the potential direct social costs and decrease in community welfare associated with these plans.
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1. Introduction

Internationally there is growing recognition that coastal water quality is interdependent with agriculture practices in coastal catchments (e.g. Elofsson et al., 2003; Brodie et al., 2005; Kroon, 2009; Maier et al., 2009). Water quality improvement (WQI) research for the agricultural sector has focussed primarily on the cost-effectiveness of management practices for WQI (e.g. Higham et al., 2008; Roebeling et al., 2009a; Van Grieken et al., 2010a), the cost-effective location of management practice implementation for WQI (e.g. Khanna et al., 2003; Yang et al., 2005; Roebeling et al., 2009a), the cost-effectiveness of WQI policy (e.g. Van Grieken et al., 2013), efficient rates of WQI (e.g. Goetz and Zilberman, 2000; Hart and Brady, 2002; Roebeling, 2006; Roebeling et al., 2009b) and private-economic consequences of WQI (Rigby and Young, 1996; Helming and Reinhard, 2009). Regional level aggregated WQI impact estimates in these studies are based on private (farm-scale) impacts – none of these studies, however, account for the likely direct social (regional-scale) impacts of aggregate private actions at the community level. This study contributes to existing studies in that it quantifies not only the private (farm-scale) impacts of WQI, but also the likely direct social (regional-scale) impacts, thus aiming to better inform policy on the socio-economic consequences of setting WQI targets. Underestimating the WQI impacts due to ignoring the social consequences of these private economic drivers, could have negative consequences for the community and policy makers (Thomas et al., 2012; Miles et al., 2009).

In order to adequately support landholders that are adopting effective management strategies for WQI, the economic and social consequences related to this change need to be understood at both the landholder and community level. For instance, management practices that improve water quality require different levels of labour input as well as additional effort in the transition phase (e.g. time for training on new farming techniques, workshops and extension meetings). Increased labour requirements influence private wellbeing either via a reduction in leisure time or through increased production costs (hiring labour) – these changes in availability of leisure time and/or financial resources can act as disincentives for

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farmers to adopt new practices. On the other hand, if agricultural land is taken out of production to meet environmental targets, the decreased labour requirements influence both private as well as social wellbeing through increased levels of regional underemployment. An additional factor is if the agricultural enterprise is part of a 'tightly connected' value chain, as is the case with sugarcane (Archer et al., 2009). In order for a sugar mill to be viable, a minimum level of throughput (tonnes of cane) is required. The mill could cope financially with one or two 'bad years', where production is below this minimum level and where mill operating costs are higher than returns, but such a situation is not sustainable in the medium to long term. Regulatory policies that enforce water quality targets could, therefore, threaten a mill's viability if regional production was generally reduced below this minimum throughput. The result would be mill closure, and the risk of severe social and economic flow-on effects such as decreased regional agricultural income and increased regional underemployment, and reduced community wellbeing (Thomas et al., 2009, 2012).

Coastal and marine ecosystems are of high biological and esthetical value due to their richness in biodiversity and productivity (Koop et al., 2001). In addition, they are of great economic value (Gordon, 2007; Roebeling et al., 2009a). Regions like the Great Barrier Reef (GBR) and the Caribbean reef systems contribute billions of dollars to their local economies (Koop et al., 2001) through tourism and fisheries. These ecosystems are adversely affected by diffuse source pollution from agricultural activities in coastal river catchments (Fabricius et al., 2005; DeVantier et al., 2006; Brodie et al., 2005; Wooldridge and Done, 2009).

In addition, recent evidence suggests that high dissolved inorganic nitrogen (DIN) concentrations exacerbate coral bleaching risk under climate change (Wooldridge, 2009; Wooldridge et al., in press) – i.e. reductions in DIN loads can reduce the coral bleaching risk to near-shore coral reefs. Current CO_2 emission and stabilisation trajectories now exceed all IPCC-SRES targets (Nakicenovic and Swart, 2000) as well as most other baselines that have been established (Raupach et al., 2007). Global climate policies aim to prevent warming from exceeding 2 °C relative to pre-industrial levels, however it is now likely that this level will be transiently exceeded (Vaughan et al., 2009). Consequences of such exceedance for the GBR are serious – CO_2 levels above 450 ppm could reduce most coral reefs to crumbling frameworks with few hard corals (Hoegh-Guldberg et al., 2007; Silverman et al., 2009). Reductions in end-of-river DIN required to offset the impacts of these temperature increases, are of the order of >50–80% (Wooldridge, 2009).

In 2003 the Australian Federal Government approved the Reef Water Quality Protection Plan ("Reef Plan"), with the goal to "*halt and reverse the decline in water quality entering the Reef within 10 years* (*2013*)" and with the objective to "*reduce diffuse source pollutants entering the Reef through implementation of sustainable land management practices and better land use decisions, and to rehabilitate and conserve areas of the reef catchment that have a role in removing water borne pollutants*" (Anon., 2003). This led to the development and implementation of water quality improvement Plans (WQI Plans) for individual coastal catchments (e.g. Drewry et al., 2008; Kroon, 2008; Dight, 2009). WQI Plans identify catchment specific critical water quality issues, provide estimates and comparisons of current and sustainable pollutant loads, and quantify the most cost-effective measures and management actions to achieve progress towards sustainable loads (Kroon, 2008; Brodie et al., 2009). The Queensland government has also designed and implemented a regulatory regime designed to reduce the water quality impacts of broad-acre land management for the first time in Australia (Great Barrier Reef Protection Amendment Act 2009).

Using the exploratory Environmental Economic Spatial Investment Prioritization (EESIP) model (Roebeling et al., 2009a), we assess and compare the medium to long term private (farm-scale) and likely direct social (regional-scale) impacts of land use and management practice change for WQI. Corresponding impacts are assessed using regional environmental (water pollution), social (employment) and economic (agricultural production and income) indicators. By combining these three measures, we obtain insight into whether and to what extent WQI impacts have been underestimated in previous research.

The structure of this paper is as follows. In the next section we describe the study system, the sugarcane production and processing industry in the Tully–Murray catchment in the Wet Tropics of Queensland, Australia. In Section 3 we describe the modelling framework used to investigate the problem. We then present the results in Section 4, followed by a discussion and conclusion.

2. The Great Barrier Reef

The World Heritage listed GBR is valued for its biodiversity, recreation, fisheries and other use and non-use values (UNESCO, 2007). The biggest threat to these values is climate change (Hoegh-Guldberg, 1999), where the risk of reduced health and resilience of the GBR under global climate change can be mitigated by improvements in GBR lagoon water quality (Wooldridge and Done, 2009). Besides the increasing risk of climate change consequences, elevated levels of pollutant loads in the GBR lagoon have resulted in reef degradation (Fabricius et al., 2005), overall reduced coral biodiversity (DeVantier et al., 2006), damage from Crown-of-Thorns starfish outbreaks (Brodie et al., 2005) and coral bleaching (Wooldridge and Done, 2009).

The Tully–Murray catchment in the Wet Tropics of Queensland (Australia) (Fig. 1) is one of 35 catchments flowing into the lagoon of the GBR, and was identified as a high risk catchment under the Reef Plan because of declining water quality in an environment with high annual average rainfall (Anon., 2003; Armour et al., 2009; Kroon, 2009). The catchment covers 2787 square kilometres, and within the vicinity of the Tully–Murray flood plume there are 37 coral reefs and 13 seagrass meadows (Devlin and Schaffelke, 2009). The primary land cover is native tropical rainforest (71%), and agricultural production is dominated by sugarcane (13%), grazing (5%) and horticulture (bananas; 3%) (Armour et al., 2009).

The sugarcane industry in the Tully–Murray catchment contributes almost 45% to agricultural income in the area (Roebeling et al., 2009a), and dominates not only economic but also water quality issues in the catchment (Armour et al., 2009; Roebeling et al., 2009a). Approximately 80% of all DIN water pollution in the Tully–Murray catchment comes from agriculture, sourced predominantly from sugarcane and, to a lesser extent, from banana production (Armour et al., 2009; Brodie et al., 2009). Fertiliser is the dominant source of DIN water pollution (Armour et al., 2009), and between 1990 and 1998 fertiliser application increased by 118%.

Fig. 2 shows a conceptual model of the sugarcane production system, which emphasizes the prioritisation of land management practice change for WQI. It can be applied at four spatial scales: farm level, individual catchments, Wet and Dry Tropics catchments and GBR Catchment Area. As a proof of concept we focus on the scale of individual catchments (Tully–Murray) described by the three lower boxes and their inter-relationships, namely 'Pollution', 'Production' and 'People', with our main interest directed towards the implications of change on individual and community wellbeing.

In general, changes in (agricultural) production can affect pollution levels as well as people's wellbeing. Although changes in pollution can also affect people's wellbeing, this is outside the scope of our study. Each relationship (arrow) in Fig. 2 will be addressed

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