

Water policy impact assessment – combining modelling techniques in the Great Barrier Reef region

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Abstract

The Reef Water Quality Protection Plan (Reef Plan) defined a landmark for policy in the Great Barrier Reef (GBR) region. It identifies actions, mechanisms and partnerships and builds on existing government policies, industry and community initiatives for the purpose of “halting and reversing the decline in water quality entering the Reef within 10 years” through “reducing the load of pollutants from diffuse sources in the water entering the Reef”. A range of different indicators proposed for the nine strategies of the plan define policy goals that require an integrated assessment of the Great Barrier Reef region.

In this context and under given uncertainties in regard to policy outcomes, decision support systems can help simulate the impact of potential policy options. Policy options involving water quantity and water quality questions and the underlying context of land use require an understanding of environmental, economic and social consequences.

This paper presents an approach developed for the GBR region, which employs a computable general equilibrium (CGE) model and an agent-based model (ABM) for integrated policy impact assessment. This applied modelling approach shows that strengths of different modelling techniques can be combined to support water policy decision making more effectively. This paper is focused on *integration* in the form of policy process and research response regarding model design on two system scales that provide cross scale decision support.

Keywords: Agent-based models; Computable general equilibrium models; Great Barrier Reef; Integrated assessment and modelling

1. Introduction

The global sustainability development agenda created challenges and opportunities for policymakers in a number of domains. Prior to this change, formal indicators of policy performance and investment prioritisation were easily derivable as they represented mostly just the economic dimension (e.g. income

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per capita). Sustainability focused policy requires multi-dimensional indicators (Parker *et al.*, 2002; Gilmour *et al.*, 2005). Integrating multiple dimensions raises the question of how to combine understanding from different disciplines.

Integrative assessment and modelling (IAM) (Rotmans & van Asselt, 1996; Parker *et al.*, 2002) brings understanding from different disciplines and at different scales into a process, in which stakeholders inform their decision-making using models. The model development is therefore integrated in the decision-making process that links scientists and policymakers. Wastney *et al.* (1998) defines models as “tools that are used to predict the structure and the behaviour of a system”. Argent *et al.* (1999) state that integrated models have to include the system components and the interactions between them.

In IAM the whole decision support process becomes important and modelling is not isolated from stakeholders (Jakeman & Letcher, 2003). From a policymaker’s perspective, Caminiti (2004) argues that “if models or decision support systems are used to help develop priorities, then sufficient understanding of the processes and input is required to have confidence in the outputs on which the priorities are based”. In order to develop decision support systems (DSS) for sustainability-focused policies, science in IAM departs from an isolated position in the decision development process and from single-disciplinary modelling approaches towards a participatory model development and assessment process.

Van Daalen *et al.* (2002) identified four roles for models in environmental policy; as eye-openers, as arguments in dissent, as vehicles to create consensus and as models for management. Models that are focused on problem identification (eye-opener) are likely to be developed effectively even in isolation from stakeholders’ participation. Yet the more the role departs from being an eye-opener and the closer it comes to the management position, the more clearly an integrated approach seems to be needed. Without an integrated process, decision makers do not have trust in the DSS and therefore are unlikely to use the models.

Rizzoli & Davis (1999) argue that if models are used in the decision-making process, they are mostly just used once: “A broader application requires at least object oriented technology, flexibility in terms of space and time scales and ability to implement context-specific models”. Examples for the implementation of such features are the integrated catchment management system (ICMS) (Rahman *et al.*, 2004) and the catchment modelling toolkit (www.toolkit.net.au).

Generally, DSS elements can be grouped into three levels. There is the level of the core model(s), secondly the modelling environment and thirdly the decision-making process (see Figure 1). This paper argues that the development of a DSS that is solely focused on the level of model development is likely not to be used by decision makers. Essentially a modelling environment is required that supports decision makers by a stakeholder tailored interface and with the flexibility to include contextual aspects of the decision-making situation. This can be achieved if the process of DSS development

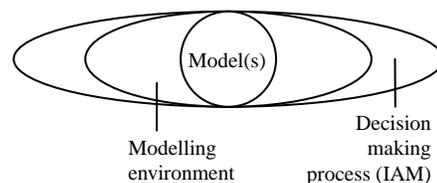


Fig. 1. Conceptualisation of three levels in integrated modelling: the core model development, the development of a modelling environment and the decision-making process.

involves decision makers and gives them an early opportunity to influence DSS design and to build trust in the DSS.

Marston *et al.* (2002) and Brinsmead (2005) give excellent overviews of what we defined as level 1. It becomes obvious that most modelling approaches are very limited in the way they integrate different dimensions. This identifies the necessity for a modification of existing modelling techniques. While in past decades, disciplines developed modelling techniques that allowed researchers to focus on specific and discipline-related questions, the current emphasis on sustainability widens the focus to encompass problems of population growth, economic growth and environmental pressures. This opens two options: (1) the development of flexible modelling environments for adapting and/or integrating existing (disciplinary) models; or (2) the development of new models (and underlying methodology) that are integrated from the start. This paper argues that both are necessary and focuses on the second point: methodology.

This paper describes the situation in the Great Barrier Reef (GBR) region and develops the first steps of an IAM approach to the GBR with a methodological focus. This paper discusses integration as a process that includes stakeholders from different levels; policymakers who define rules within catchment boundaries and land managers who operate on a farm or property level. Addressing the policy process on the catchment level and providing decision support based on average values or totals may leave the questions of land managers unanswered with regard to how policies might affect them individually. This paper therefore develops a methodology for integrated modelling on a macro and a micro scale in order to allow a more coherent approach that integrates stakeholders from different levels in the development of decision support tools.

2. Integrated process in the GBR

The GBR marine and near-shore ecosystems have a complex interdependent relationship with the adjacent river systems, which comprise some 30 major rivers and hundreds of small streams that drain into the GBR lagoon. Declining water quality, principally from agricultural land use, threatens the viability of downstream marine based activities in the GBR, in particular tourism, which contributes Australian dollars AU\$4,300 million to the state and regional economies and exceeds the contribution that agricultural activities make to the economy (AU\$3,200 million) (Productivity Commission, 2003). These economic values only partially reflect the broader value of the GBR to Australians as a cultural icon, as a place of high biodiversity and world significance and as a provider of ecosystem services such as clean freshwater for coastal communities.

This complex and interconnected ecosystem is managed through an equally complex array of legislation and policy, spanning both Queensland and Commonwealth jurisdiction. The GBR is also a World Heritage Area with international obligation regarding management.

In recognition of the water quality issues for the GBR, the Australian and Queensland Governments established the Reef Water Quality Protection Plan (Reef Plan) (Queensland Government, 2003). The Reef Plan identifies actions, mechanisms, partnerships and builds on existing government policies, industry and community initiatives for the purpose of “halting and reversing the decline in water quality entering the Reef within 10 years”, through “reducing the load of pollutants from diffuse sources in the water entering the Reef” and “rehabilitating and conserving areas of the reef catchment that have a role in removing water borne pollutants”.

Nine strategies are proposed for implementation by governments, community and industry: self-management approaches; education and extension; economic incentives; planning for natural resource management and land use; regulatory frameworks; research and information sharing; partnerships; priorities and targets; and monitoring and evaluation. The implementation of the Reef Plan is funded primarily through two national environmental programmes, the Natural Heritage Trust (NHT) and the National Action Plan on Salinity and Water Quality.

Given the links between increases in nutrient loads entering the GBR lagoon and land management practices in the catchment (Furnas, 2002; Brodie et al., 2003), the land use planning process needs to be approached from a whole-of-system perspective. The scope of the Reef Plan actions indicates the complexity of individual sector responsibilities. Overcoming the fragmentation of these responsibilities for catchment management requires information to be provided at a local, regional and whole-of-system scale and for the heterogeneity of the systems to be adequately accounted for.

3. Integrated modelling in the Great Barrier Reef

3.1. Policy perspective and decision support system

The need for an integrated approach stems from policy requirements to consider environmental, economic and social goals simultaneously. Policy is often focused on system boundaries that result from formal responsibilities. This means that a natural resource manager is mainly interested in decision support at the scale at which he or she operates. Often the indicators used in such a decision-making process are highly aggregated. For instance, a policymaker is interested in increasing income levels for the whole region but much less in income levels of each and every family. This emphasises the relevance of mean values across the population. On the other hand, social reasons often require a disaggregated view, like unemployment peaks in a focus area. Environmental problems in the GBR region require both types of simulation, for example, the net run-off into the GBR lagoon on an average level, but also spatially explicit dynamics to identify increasing risks for biodiversity.

In other words, policy requirements demand “big picture” information and more specifically the next question is “If this happens, *where* does it happen?”. Modelling can explore two other domains of policy question likely to follow in an integrated assessment process: “If this simulation shows what is likely to happen, what is a ‘better’ or an ‘optimal’ path?” and “How likely are the simulated and the ‘optimised’ paths?” Although our IAM approach covers all three questions – (1) What will happen? (2) What should happen? and (3) How likely are these trajectories? this paper is focused solely on the first question on two scales of aggregation. We used a computable general equilibrium (CGE) to integrate dynamics on a catchment scale. CGE modelling is used to quantify cross-sectoral dynamics assuming highly aggregated sectoral production functions. We used agent-based modelling (ABM) to model paddock scale dynamics. Thus, the CGE model captures the mean or the total of an indicator and the ABM quantifies the spatial distribution.

The development of both models is based on many discussions that were triggered by the requirements of the Reef Plan. The design of the CGE model was developed in workshops by research and policy experts. The CGE model is expected to be used by catchment authorities to simulate impacts or policy decisions and climate change scenarios within and across catchments. The agent-based model is focused on two types of landscape, the grazing dominated rangelands and the floodplains, which are mostly used

level, fertiliser input (pntr) enters as an input factor, followed by labour (pl) and intermediates (pa). For non-irrigation sectors, formula (1) simplifies to three inputs, pk, pl and pa.

Traditional CGE models (Ginsbergh & Keyser, 1997) explain shifts from one equilibrium to a new equilibrium by prices that signal changes in production constraints. Such a price–signal-based approach works very well in a (highly competitive) setting where only prices give signals. Natural resources – such as water in Equation (1) – can be added into a CGE as an input factor to simulate the shadow price for changing constraints. The ability to emit greenhouse gases is another domain of application (Nordhaus, 1993; Bernstein *et al.*, 1999) in which CGE models simulate carbon trading according to different emissions constraints and returns likely prices for tradable CO₂ quotas, as Smajgl (2002) and Hillebrand *et al.* (2003) illustrate for an emissions trading scheme in the European Union.

Also considering output-sided consequences such as the impact of a water trading scheme on biodiversity requires non-market values. Such “hybrid” CGE models will have to go a step further and allow responses within a non-market system. Water-related problems in the GBR require the simulation of flow-on effects within the hydrological system and the ecological system. This means that in addition to economic production functions with input and output variables that are coordinated by prices in markets, “production functions” for groundwater and surface water have to be implemented.

Existing water-focused CGE models (i.e. Horridge *et al.*, 1993; Goldin & Roland-Holst, 1995; Decaluwé *et al.*, 1997; van der Mensbrugge *et al.*, 1998; Seung *et al.*, 2000) are similar to climate change CGE models in that they implement water as an input factor without including non-market systems such as hydrology or ecology (Berck *et al.*, 1991).

Smajgl (2006) demonstrates the approximation of a water cycle on a catchment level in a CGE model. Crucial indicators for an integrated assessment are the remaining water volumes in streams and in aquifers. Irrigation from surface water, see Equation (2), is based on the input components “water in streams” (psw) and “rain” (fswf). The output side is defined by outtake (pwi) and recharge that adds to the groundwater table of the next period (pgwtl). Additionally, the option for introducing quotas as permits for pumping ground water (pgwp) and surface water (pswp) are considered:

$$\left(\text{pwi}_{\text{IRR},r}^{\rho_{\text{SWO}}} + \text{pgwtl}_{\text{IRR},r}^{\rho_{\text{SWO}}} \right)^{1/\rho_{\text{SWO}}} = \left(\left(\text{pswp}_{s,r}^{\rho_{\text{SWQ}}} + \text{psw}_{s,r}^{\rho_{\text{SWQ}}} \right)^{\rho_{\text{SW}}/\rho_{\text{SWQ}}} + \text{fswf}_{s,r}^{\rho_{\text{SW}}} \right)^{1/\rho_{\text{SW}}} \quad (2)$$

$$\left(\text{pwi}_{\text{IRR},r}^{\rho_{\text{GWO}}} + \text{pgwtl}_{\text{IRR},r}^{\rho_{\text{GWO}}} \right)^{1/\rho_{\text{GWO}}} = \left(\left(\text{pgwp}_{s,r}^{\rho_{\text{GWQ}}} + \text{pgw}_{s,r}^{\rho_{\text{GWQ}}} \right)^{\rho_{\text{GW}}/\rho_{\text{GWQ}}} + \text{pgwt}_{s,r}^{\rho_{\text{GW}}} \right)^{1/\rho_{\text{GW}}} \quad (3)$$

Groundwater irrigation has the groundwater (pgw) and the aquifer at the beginning of the period (pgwt) on the input side. The output side has the outtake (pwi) and the remaining groundwater table for the next period (pgwtl). In Equations (2) and (3) the option of permits is implemented on the input side. This allows the assessment of water trading schemes or water restrictions as well as climate change-based reductions in rainfall.

While the impacts on production add up to a change in gross regional product (GRP), an important and highly aggregated political indicator, changes on the non-market side are also taken into account. Smajgl & Hajkowicz (2005) demonstrate how water-related benefits in the GBR are structured by using multi criteria analysis (MCA). Most water issues have an impact on human well-being through ecosystem

services. Therefore, PIA defines a series of response functions for ecosystem services and species based on MCA results. Data on fish were obtained from catch statistics and calibrated based on Ecopath (Smajgl & Gehrke, 2007). Hydrological data was sourced from expert consultation and economic data was derived from input–output tables developed for the catchment level of the GBR region. Feedbacks from ecological variables to economic activities such as tourism are derived from expert consultation and several studies that have been undertaken in the GBR region (Productivity Commission, 2003).

The inclusion of nutrients demonstrates the hybrid nature of the PIA model. The application of fertiliser is economically driven, as shown in formula (1). But the application of fertiliser (pftl) “consumes” a virtual variable (fnm), which indicates water quality, shown in Equation (4):

$$\text{pntr}_{s,r} = \left(\text{pftl}_{s,r}^{\rho_{\text{NTR}}} + \text{fnm}_{s,r}^{\rho_{\text{NTR}}} \right)^{1/\rho_{\text{NTR}}} \quad (4)$$

The consumption of fnm changes the quality available for other processes such as seagrass. As seagrass is a food source for other species like dugongs, there is an impact on their habitat. Furthermore, tourism operators need not only capital (e.g. buses and boats) and labour (e.g. guides), but also the tourist attraction (e.g. dugongs). If the attractiveness of the tourism attraction declines for quantity or quality reasons, tourism-based revenues are also likely to decrease.

$$\text{fsg}_r = \left(\text{fpp}_r^{\rho_{\text{SGR}}} + \text{fnm}_r^{\rho_{\text{SGR}}} \right)^{1/\rho_{\text{SGR}}} \quad (5)$$

$$\text{fdu}_r = \alpha_{\text{dug}} \cdot \text{fsg}_r \quad (6)$$

$$\text{py}_{\text{TOU},r} = \left(\left(\text{pk}_{\text{TOU},r}^{\rho_{\text{KL}}} + \text{pl}_{\text{TOU},r}^{\rho_{\text{KL}}} \right)^{\rho_{\text{Y}}/\rho_{\text{KL}}} + \text{pa}_{\text{ss},r}^{\rho_{\text{Y}}} + \text{fdu}_r^{\rho_{\text{Y}}} \right)^{1/\rho_{\text{Y}}} \quad (7)$$

The CES production functions (5) to (7) (for sectoral production py of the tourism industry TOU) integrate seagrass (fsg), phytoplankton (fpp) and dugongs (fdu). The seagrass-dugong link is an example. The full list of variables and simulation results is provided in Smajgl (2006).

Following the MCA results (Smajgl & Hajkowicz, 2005) the most relevant non-market values are linked to abundance and diversity of fish. The following example summarises some selected results from a simulation run that is based on a combination of climate change and policy response assumptions. For this scenario, it is assumed that the annual rainfall between 2007 and 2013 drops by 5% owing to climate change. Further it is assumed that the Burdekin catchment authority, which is responsible for natural resource management of the Burdekin catchment, restricts surface water use for irrigation purposes by 10% in order to maintain environmental flows. Additionally, it is assumed that the Burdekin catchment authority introduces a water trading scheme. Furthermore, it is assumed that a quantity cap for fertiliser use is put in place for the Burdekin catchment. These assumptions combine mechanisms regarding water quantity and water quality that are currently discussed by catchment authorities in the GBR region as options beyond voluntary agreements. In contrast, none of the other four catchment authorities changes its policy, including the Wet Tropics catchments authority. Table 1 summarises some selected results of

Table 1. PIA model simulation results for the Burdekin catchment and the wet tropics regions. Assumptions: 5% less rainfall for the whole GBR, Burdekin constrains surface water use for irrigation by 10% and introduces a trading scheme for water quotas and introduces a cap for fertiliser of 5%. The Wet Tropics is assumed to put no policies in place.

Years	Regional GDP (%)	Sum of fish populations (%)	Beef production (%)	Dairy production (%)	Horticulture (%)	Sugarcane (%)
Burdekin catchment						
2007	106	105	94	108	103	104
2008	106	105	94	111	106	106
2009	106	105	94	113	108	107
2010	106	105	94	114	109	108
2011	106	105	94	115	109	109
2012	106	105	94	114	110	109
2013	106	105	94	116	110	109
Wet Tropics catchments						
2007	95	95	94	94	91	92
2008	95	95	94	94	91	92
2009	95	95	94	94	91	92
2010	95	95	94	94	91	92
2011	95	95	94	94	91	92
2012	95	95	94	94	91	92
2013	95	95	94	94	91	92

the PIA model. The results represent relative changes in the Burdekin and the Wet Tropics region compared with a business as usual case for these two areas.

The policy in the Burdekin catchment avoids a significant reduction in regional GDP by allowing market mechanisms to maintain water in highly efficient sectors such as horticulture and sugarcane. The rainfall-driven decrease in production in the other catchments (i.e. the Wet Tropics) increases prices for, for instance, horticulture products in the Burdekin catchment. Therefore, economic figures capture the catchment-focused policy change, the climatic constraints and the cross catchment market dynamics. In addition to these traditional economic features of this hybrid CGE, the net impact of water quantity and water quality changes is quantified for fish populations. The reduction of fertiliser use leads to an increase in fish biomass of 5% compared with a business as usual case while the figure for the Wet Tropics reduces by 5%.

This discussion shows how a hybrid CGE model allows the assessment of policy options taking not only market-based values into account but also non-market aspects. As these indicators are highly aggregated, crucial questions about spatial and temporal distributions are not included. Stakeholders in our workshops demanded such a function as the spatial location of impact is critical for water quality aspects. Therefore, the integrated CGE model PIA is developed in combination with an agent-based model, the single entity policy impact assessment (SEPIA). Technically, these two models (PIA and SEPIA) are not linked. Instead, parameters that are defined outside of the case study location, such as prices for goods, can be manually entered into the SEPIA model. Behavioural data that is defined within the spatial focus of SEPIA can inform parameters in the hybrid CGE model providing more flexibility to specify model parameters at both scales than if the link between both scales was automatic. The next section provides a description of the SEPIA model.

3.3. Integrated modelling on a farm level

The agent-based model SEPIA simulates land use and water use-related decision making in the GBR region. Agent-based models allow the analysis of interactions in a spatial distribution and are therefore very promising in the complex domain of agro-ecological systems (Parker *et al.*, 2003). As Berger (2005) points out, agent-based models can also provide a collaborative learning framework for policymakers and scientists (also Roeling, 1999; Hazell *et al.*, 2001; van Paassen, 2004).

SEPIA does not follow an optimisation approach. Instead, global dynamics emerge from individual decisions of agents and their interactions which are governed by rules in the model. Technically, these rules are structured in “if-then-else” formulations (see for instance Epstein & Axtell, 1996). Land users, for instance, make decisions depending on perceived values of certain indicators, such as market prices and rainfall. Within a GIS based visualisation, the spatial distribution of a decision can be analysed and linked to a wider range of bio-physical and socio-economic indicators.

The SEPIA model simulates land use decision making enacted by agricultural agents. Agents’ cognitive processes are mental models of decision making (Smajgl, 2004), resulting in the enactment of one of a number of possible land use options. The effect of these land use decisions in turn has an effect on conditions on the ground and on the payoff that agents receive from agricultural production. Environmental impacts are then estimated based on the new conditions.

Agent decision making is a composite of both market and non-market conditions. Hence information for both is available to agents and decision making is based on a non-market utility function and on costs and revenues-based economic payoff functions. As outcomes are uncertain, agents perceive a random variation of system variables such as rainfall and develop their expectations for non-market returns and economic profits.

On the non-market side of decision making, we understand that several streams of benefits that are not represented in markets accrue to individuals, which are derived from the MCA, mentioned above. Agents also maintain a threshold for non-market values. If the utility threshold is not met, agents flag their dissatisfaction. This becomes visible to other agents and if the community of dissatisfied agents dominates the population, it is assumed that the strategy that leads to a negative impact on agents’ utility function is de-activated for all agents.

Market-based conditions associated with agricultural production were drawn from a number of secondary literature sources. From these, costs and revenues associated with production of agriculture commodities were obtained. The inclusion of some costs and revenues and their amounts obviously change depending on local and individual conditions such as market access, environmental limitations, government assistance and regulations and other conditions.

Agents have the ability to alter the set of inputs into production, based on expectations of return in the changing conditions of the marketplace and farm level productivity. System dynamics are illustrated within a spatial landscape of GIS vector polygons. This allows for the specific identification of where particular impacts occur. Indicators are measured for cross-disciplinary metrics and displayed using a GIS user interface. Indicators of interest include: adoption of land use practices, production and financial returns and sediment and nutrient contributions.

The following example shows how the SEPIA model – implemented in the open source Recursive Porous Agent Simulation Toolkit (Repast) (Collier *et al.*, 2003) – visualises results of an incentive scheme for fencing riparian zones. Figure 2 shows the Bowen Broken sub-catchment, which flows into the Burdekin River and was identified by stakeholders as a major contributor to sediment delivery into the GBR lagoon.

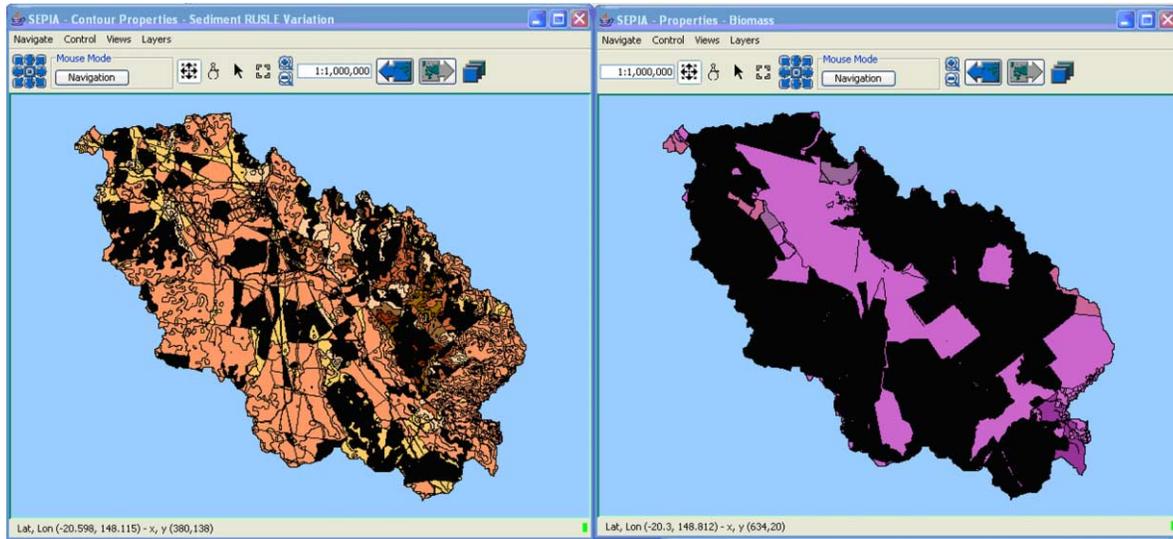


Fig. 2. Graphical user interface for the Bowen Broken catchment, showing on the left erosion dynamics and on the right accumulated biomass.

The map on the left shows potential sediment generation with increases represented by darker colours, while the map on the right shows the accumulation of pasture, with increases represented by darkness in colour.

Comparing such maps for various policy mechanisms allows the stakeholder to assess policy impacts according to spatial distribution. Additionally, figures such as total sediment delivery or farm income in the region can be quantified and compared across simulations. A more detailed description of the SEPIA model design and its simulation results are presented in Smajgl *et al.* (2007).

4. Conclusions and recommendations

The GBR region faces major challenges with regard to water quality and stakeholders such as catchment authorities have to respond to the Reef Water Quality Protection Plan, which demands improvements in water quality by 2013. Outcomes of water-related policy and management decisions are uncertain and system responses highly complex. Models and decision support systems may improve the understanding of decision makers but model development faces methodological impediments owing to disciplinary roots. Hence, the modelling component requires not only improvements on the side of modelling environments that allow combining different models, but also modelling techniques themselves that have to be modified in order to capture multi-disciplinary system descriptions.

The GBR focused project described in this paper broadens existing CGE modelling in order to integrate non-market values and their bio-physical dynamics, instead of purely market driven dynamics, which operate on a macro-scale simulating average values and totals. A process-oriented integration of stakeholders in the development of tools also involves understanding on a micro-scale, targeting land managers. As a result the agent-based model makes simulations more spatially explicit. Future steps will be focused on updating model design in response to the needs of regional stakeholders such as catchment authorities and implement further aspects of climate change, which define the conditions stakeholders expect and which they want to anticipate.

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