CSIRO PUBLISHING

www.publish.csiro.au/journals/mfr *Marine and Freshwater Research*, 2009, **60**, 1141–1149

Target setting for pollutant discharge management of rivers in the Great Barrier Reef catchment area

Jon Brodie[A,D], *Stephen Lewis*[A], *Zoe Bainbridge*[A], *Alan Mitchell*[A], *Jane Waterhouse*[B] *and Frederieke Kroon*[C]

[A]Australian Centre for Tropical Freshwater Research, James Cook University, Townsville, Qld 4811, Australia.
[B]CSIRO Sustainable Ecosystems, Davies Laboratory, Aitkenvale, Townsville, Qld 4814, Australia.
[C]CSIRO Sustainable Ecosystems, Atherton, Qld 4883, Australia.
[D]Corresponding author. Email: jon.brodie@jcu.edu.au

Abstract. Water Quality Improvement Plans (WQIPs) are being developed for individual river basins on the Great Barrier Reef (GBR) catchment associated with the GBR Water Quality Protection Plan. Within each WQIP, marine ecosystem targets are linked to end-of-river pollutant (suspended sediments, nutrients and pesticides) load targets and to farm level management practice targets. The targets are linked through quantitative models; e.g. one model connects GBR chlorophyll concentrations (marine target) to end-of-river nitrate loads, a second connects the end-of-river nitrate loads to fertiliser management targets in the catchment, whereas a third model links fertiliser application to nitrate loss at the farm scale. The difficulties of applying these linked models to derive credible and practical management targets are great, given the high degree of uncertainty in each model. Our understanding of the generation of suspended sediments, nutrients and pesticides in catchments and the relationship to on-farm management, the transport of these materials to the ocean, their transport in coastal waters and their effects on marine ecosystems is incomplete. The challenge is to produce estimates from the models, with known levels of uncertainty, but robust enough for management purposes. Case studies from the Tully–Murray basin and the Burdekin basin in north Queensland are discussed.

Additional keywords: Burdekin River, modelling, monitoring, Tully River, Water Quality Protection Plan.

Introduction

The Great Barrier Reef World Heritage Area (GBRWHA) is located along the north-eastern Australian coast (see fig. 1 in Kroon 2009) and consists of a diverse range of ecosystems including coral reefs, seagrass meadows, mangrove forests and open water communities. On its western boundary, 35 basins discharge into the GBRWHA over ~2000 km of Queensland coastline. Loads of pollutants discharging from these basins have increased greatly with the development of the river catchments for agriculture over the past 150 years (Furnas 2003; McKergow *et al*. 2005*a*, 2005*b*). Pollutant loads have increased by up to five times for suspended sediment from some rivers (McKergow *et al*. 2005*a*), and up to six times for nitrate in others (Hunter and Walton 2008; Mitchell *et al*. 2009). Moreover, considerable quantities of pesticides are now discharged that would have been completely absent before the 1950s (Johnson and Ebert 2000; Mitchell *et al*. 2005). Impacts of this increased pollutant loading on coral reefs in the central part of the GBRWHA has resulted in reef degradation in the wet tropics coastal area (Fabricius *et al*. 2005) and overall reduced coral biodiversity between Townsville and Cooktown (DeVantier *et al*. 2006). In the area adjacent to the Tully River, a reduction in species richness of 40 species, compared with the expected value, is evident (DeVantier *et al*. 2006). Reefs in the central GBR are also subject to damage from crown-of-thorns starfish outbreaks, most likely related to nutrient enrichment (Brodie *et al*. 2005). Additional impacts that affect GBR ecosystem health include coral bleaching and ocean acidification effects associated with increased atmospheric carbon dioxide and other greenhouse gases (Lough 2008) and fishing pressures (Pandolfi *et al*. 2003). The combination of these impacts has resulted in reef degradation similar to that seen in other parts of the world (Bruno and Selig 2007), although the levels of degradation are considered to be less than in many other reef systems (Pandolfi *et al*. 2003).

In response to the threats posed to the GBRWHA ecosystems from land-based pollution (Brodie *et al*. 2001*a*), a joint Australian and Queensland State Government GBR Water Quality Protection Plan ('Reef Plan') was developed (Anonymous 2003). As part of the implementation of Reef Plan, regional Water Quality Improvement Plans (WQIPs) are being developed and implemented for priority regions in the GBR catchment area (see Kroon 2009). One of the requirements of WQIPs is that a range of targets need to be developed that will protect critical waterway assets, in this case the GBRWHA. Specifically, the end-of-system (generally a GBR ecosystem) targets for water quality are linked to targets for management action in the GBR catchments. Here,

 10.1071/MF08339 1323-1650/09/111141

targets are defined as 'quantifiable performance levels or changes in level to be attained at a specific future date'.

Targets in the WQIP process are required to justify the level of investment on the basis of a known 'required' level of pollutant reduction to meet the ecosystem requirements of the GBRWHA. Historically, although targets were set (e.g. Brodie *et al.* 2001*b*), the process was quite *ad hoc* and lacked scientific transparency. The current target-setting process attempts to provide more scientifically justified targets using linked models from paddock to reef. The process follows the principle of SMART (Specific, Measurable, Achievable, Relevant and Timed) targets developed for the GBR catchments by McDonald and Roberts (2006). This process also allows analysis of management options by running scenarios and can assess potential progress towards scientifically valid targets for various management options. The end-of-river load target for specific pollutants resembles to some extent the Total Maximum Daily Load (TMDL) concept used in the USA (e.g. Karr and Yoder 2004) but is only one component of our target setting. Our target-setting process also tries to closely link river loads with marine ecosystem objectives, which greatly increases its complexity, but is known to be an important component to produce realistic targets (e.g. Borsuk *et al.* 2004; Karr and Yoder 2004).

Here, we describe how the development of targets was attempted for the Tully WQIP (Kroon 2008) and the Burdekin WQIP (Dight 2009). Our study only analyses the process used to set the biophysical part of the targets. A financial–economic model overlies the target setting process and is described in Roebeling *et al.* (2009). We attempt, albeit qualitatively (on a ranked four-level subjective scale), to assess the levels of uncertainty in each step of the model chain, which links GBR ecosystem targets to land-use management targets.

Model components of target setting

The models used in the target-setting process for the Tully and Burdekin WQIPs include SedNet and ANNEX (e.g. Kinsey-Henderson *et al.* 2007; Armour *et al.* 2009) and ChloroSim (Wooldridge *et al.* 2006). SedNet/ANNEX (Sediment River Network/Annual Nutrient Export) (Armour *et al.* 2009) estimates a long-term, annual average load, rather than predicting short-term events. The SedNet model (e.g. McKergow *et al.* 2005*a*) estimates sediment loads in catchments by constructing material budgets that account for the main sources and stores of sediment. The model makes estimates of erosion rates (gully, bank and hillslope) for available climate, soil, topography and land-use data as well as information relating to the catchment's hydrological processes (mean annual flow, extent of floodplain, channel dimensions). The model uses simple conceptualisations of hydrological transport and sediment deposition processes. The contribution of sediment from each subcatchment to the river mouth can be traced back through the system. ANNEX is an addition to SedNet used to estimate speciated nutrient loads. A weakness of SedNet/ANNEX is that there is an inadequate understanding of the model uncertainty and parameter sensitivity (but see Newham *et al.* 2003). The SedNet/ANNEX models do not effectively take into account the large variation in flow, suspended sediment and nitrogen discharge from year to year, and this introduces a large degree of uncertainty into the relationships for any one year.

ChloroSim is a combined hydrodynamic/chlorophyll-*a* (chl-*a*)–nitrate correlation model for the GBR (Wooldridge *et al.* 2006). This model links a quantitative river discharge parameter (i.e. dissolved inorganic nitrogen (DIN) concentration in event flows) with a quantitative indicator of health in the marine environment (i.e. chl-*a* concentration). This relationship has been confirmed for the GBR north of the Burdekin River, where observed summer chl-*a* concentrations in the inner-shelf areas increase significantly with the export of elevated DIN from the adjacent river catchments (Wooldridge *et al.* 2006).

A transparent process of uncertainty analysis is important for stakeholders involved in target load setting processes (DePinto *et al.* 2004), including all inputs and assumptions. However, both SedNet/ANNEX and ChloroSim are deterministic models and, while it is generally understood that these models have high degrees of uncertainty, quantitative uncertainty estimates are not part of either models' output. Uncertainty in water quality model predictions is inevitably high owing to model equation error, parameter error and boundary condition problems (McIntyre and Wheater 2004). Moreover, errors leading to target uncertainty propagate through the chain of models we have used to set the targets. Some uncertainty analysis has been studied for the SedNet/ANNEX model in a general setting, focusing on inputs such as soil nutrient data (Sherman and Read 2008), hydrology and other inputs (Newham *et al.* 2003), erosion source inputs (Herr and Kuhnert 2007), vegetation cover and gully density assumptions (Dougall *et al.* 2007), and comparing model outputs to equivalent monitoring results (Bartley *et al.* 2007; Sherman *et al.* 2007; Armour *et al.* 2009). Thus, for SedNet/ANNEX, it is possible to make some semi-quantitative estimates of model uncertainty for specific model runs. No model uncertainty studies have been conducted for ChloroSim, and estimates of uncertainty for this model are based on an 'expert judgement' approach. In the present paper, we estimate uncertainty for the various model steps on a semi-quantitative basis on a scale of four points – low, moderate, high and extreme.

Target setting for the Tully–Murray basin for nitrate

The Tully–Murray basin covers an area of 2787 km^2, with main land uses comprising natural forest (71%), sugarcane (13%), grazing (5%), plantation forestry (4%), banana and other horticulture (3%) and urban (1%) (Armour *et al.* 2009). The remaining areas in the basin are occupied by waterways (3%). The landscape of the basin has been altered extensively since European settlement (Furnas 2003), including reduction in (i) area of floodplain vegetation (~80%, to 20.8 km^2), (ii) riparian area (~60%, to 59 km^2) and (iii) wetland area (~69%, to 72.5 km^2). These floodplain alterations reflect exploitation for grazing and timber, and clearing for agricultural development that have resulted in changes in hydrology and drainage. During flood events, ~2000 km^2 of marine receiving waters are influenced by the basin's discharge, with the exact area depending on the volume and duration of flow, as well as the direction of currents and winds (Devlin and Schaffelke 2009). This receiving water body is also affected by flood plumes from both the Herbert and Burdekin Rivers. Recent satellite imagery shows that plumes from wet tropics rivers, including the Tully, can extend eastwards across the entire reef shelf and beyond into the Coral Sea (Devlin and Schaffelke 2009).

The reduced coral biodiversity off the Tully–Murray basin (DeVantier *et al*. 2006) is ascribed to the effects of poor water quality, compared to analogous reef areas further north (adjacent to Cape York) where water quality is better (Fabricius *et al*. 2005). The poorer state of this water quality is quantified through a water quality index, which includes measures of nutrient and suspended sediment concentrations (Fabricius *et al*. 2005). The level of uncertainly on our attribution of nutrient excess causing coral biodiversity loss is moderate (Table 2). While pollution effects on coral reefs at local scales are well understood, links at regional scales between increasing sediment, nutrient and pesticide loads in rivers, and the broad-scale degradation of coral reefs, have been more difficult to demonstrate (Fabricius *et al*. 2005). This is due to a lack of large-scale historic data and the confounding effects of other disturbances such as coral bleaching, tropical cyclones, fishing pressure and outbreaks of the coral-eating crown-of-thorns starfish (*Acanthaster planci*) and is further complicated by the naturally high variability in monsoonal river flood events. In addition, the relationship between macroalgal proliferation, nutrient enrichment and the abundances of grazers (fishes and invertebrates) is complex (Fabricius 2007) and far from understood, and subject to scientific debate (McCook *et al*. 2001; Littler *et al*. 2006; Hughes *et al*. 2007). The full extent of organism responses are poorly understood, as each of the numerous inshore species has their own tolerance limit at every life stage, and interactions between the organisms add to the complexity.

As an indicator of water quality, in this case phytoplankton biomass, chl-*a* is widely used as a proxy for nutrient availability (Brodie *et al*. 2007). Chl-*a* concentrations in the waters off Cape York average $0.2\,\mu g\,L^{-1}$, whereas in the area of coral reef biodiversity loss (wet tropics coast) concentrations average $0.7\,\mu g\,L^{-1}$ (Brodie *et al*. 2007). The difference is ascribed to the increased nutrient discharge from rivers such as the Tully caused by increased erosion and fertiliser loss (Mitchell *et al*. 2001, 2009; Furnas 2003; McKergow *et al*. 2005*a*; Brodie *et al*. 2007). Water quality trigger values for the GBRWHA have been set for chl-*a* at $0.6\,\mu g\,L^{-1}$ for inshore waters (Moss *et al*. 2005) and later to $0.5\,\mu g\,L^{-1}$ (GBRMPA 2008; since then reduced to $0.45\,\mu g\,L^{-1}$), and this value ($0.5\,\mu g\,L^{-1}$) was used as the GBR target for the Tully WQIP. The level for uncertainty in the value of $0.5\,\mu g\,L^{-1}$ as a guideline value is moderate because of high regional variability in nutrient dynamics and the ability of different ecosystems to withstand adverse conditions (Table 2).

To connect this chl-*a* target to river discharge targets, the ChloroSim model estimated the degree of improvement in river water quality (i.e. % reduction in DIN concentration) that is necessary to ensure that chl-*a* reaches $<0.6\,\mu g\,L^{-1}$ for all locations within the northern GBR lagoon. Specifically, to estimate sustainable nitrate loads for the Tully River, the ChloroSim model used: (i) modelled spatial extent of run-off–seawater dilution ratios; (ii) observed run-off-induced lagoon chl-*a* concentrations (Brodie *et al*. 2007); and (iii) observed flood-induced river nutrient concentrations (Furnas 2003; Wooldridge *et al*. 2006). The observations are based on monthly summer records of chl-*a* concentrations across lagoonal waters since 1992 (Brodie *et al*. 2007), and NO_x (nitrate and nitrite) data collected over 12 years (1988–2000) at the Tully River (Euramo) site (Mitchell *et al*. 2001; Furnas 2003). To achieve a chl-*a* target of $<0.6\,\mu g\,L^{-1}$ in waters off Tully, the model requires a reduction in DIN (nitrate plus nitrite plus ammonium) loading in all the adjacent rivers (including Tully, Herbert and Burdekin) of at least 80%. Our confidence level in this model step is moderate (Table 2), because the model relationship is confined to one form of nutrient correlation and ignores many other factors that may add to the cause of phytoplankton growth (Devlin and Brodie 2005). In addition, the different end-point chlorophyll values used in Chlorosim – $0.6\,\mu g\,L^{-1}$ (derived from Moss *et al*. 2005), compared with the later (in time) trigger values of $0.5\,\mu g\,L^{-1}$ and $0.45\,\mu g\,L^{-1}$ derived for marine waters (GBRMPA 2008), introduce further uncertainty to the target estimation process.

Our current estimates of DIN loads are based on modelling using the SedNet/ANNEX model (McKergow *et al*. 2005*b*; Armour *et al*. 2009) and long-term monitoring data (Mitchell *et al*. 2001, 2009; Furnas 2003). The most recent estimate from ANNEX modelling for the Tully basin is an average annual load of 1160 t DIN $year^{-1}$, which makes the target load 232 t (i.e. 20% of current) (Table 1). Both modelling and monitoring have demonstrated that ~80% of the DIN exported to the river mouth is derived from fertiliser loss in sugarcane and banana cultivation (Armour *et al*. 2009; Bainbridge *et al*. 2009*b*). Hence, this is where reductions in loss must be targeted if large reductions in end-of-river discharge are to be achieved.

A significant source of error in the estimates of contaminant loads discharged to the GBR is due to the inadequacy of sampling sites and sampling methods in existing monitoring programs. For instance, in the Tully catchment, Wallace *et al*. (2009) showed that a large proportion of the total load of suspended sediment and nitrogen was present in waters in overbank flow on the floodplain and this was not included in the load calculation made at the gauging station (Euramo) in the river channel. Similarly, it is clear that much of the nitrate lost from sugarcane fertiliser in the lower Burdekin reaches the GBR via small stream discharge and possibly groundwater discharge and is thus not included in loads measured at Home Hill in the Burdekin River (Brodie and Bainbridge 2008). Similar 'missing' loads are obvious in the Mackay Whitsunday region through small stream network discharge (Rohde *et al*. 2008) and are likely in other WQIP regions.

The level of uncertainty in the load estimates for the Tully basin is moderate (Table 2), because (i) modelled and monitored DIN concentrations at six sampling locations were in general agreement (Armour *et al*. 2009); (ii) compared to pre-1850 loads, the current total nitrogen (TN) load appears to have increased by a factor of around 3 (McKergow *et al*. 2005*b*), which corresponds to the estimated 80% reduction required in DIN concentrations at the end-of-river site for the Tully River (Euramo); (iii) the Tully River has a low level of interannual variation compared with most Australian rivers (Mitchell *et al*. 2009); and (iv) much of the load is missed at Euramo due to the overbank flow components discussed above.

To achieve the required estimated potential reductions in nitrate loads at the end-of-river, future scenarios were modelled that specifically targeted improvement of nitrogen management in sugarcane. Potential management scenarios

Table 1. Current river loads and potential reductions based on land management scenarios, for (*a*) dissolved inorganic nitrogen (DIN) in the Tully–Murray basin and (*b*) suspended sediment for the Burdekin, derived from SedNet/ANNEX models (from Kinsey-Henderson *et al.* 2007; Armour *et al.* 2009)
n/a, not applicable

(*a*) Scenarios	Fertiliser target (kg N ha^{-1} $year^{-1}$)	Reduction (%)	DIN load (t $year^{-1}$)
Current	150	n/a	1200
Sustainable load	–	>80	230
BSES '6 Easy Steps'	140	23	900
CSIRO 'Nitrogen replacement'	110	45	640
Reghenzani 'Nitrogen fixation'	30	66	390
(*b*) Scenarios	Erosion target	Reduction (%)	Total suspended solids load (million tonnes $year^{-1}$)
Current	–	0	4
Sustainable load	–	63	1.4
50 year (60% reduction)	• 70% pasture cover everywhere • Gully erosion decreased by 50% everywhere • Riparian vegetation restored to 95% everywhere	60	1.6
12 year (14% reduction)	• 70% pasture cover on priority subcatchments (8 listed) • 50% reduction in gully erosion on priority subcatchments (4 listed) • Riparian restoration to 95% on priority subcatchments (4 listed)	14	3.4
5 year (8% reduction)	• 70% pasture cover on all priority sub-catchments (8)	8	3.7
5 year (13% reduction)	• 50% pasture cover everywhere	13	3.5
5 year (13% reduction)	• Riparian vegetation restored on 95% everywhere	23	3.1
5 year (13% reduction)	• Gully erosion decreased by 50% everywhere	8	3.7

Table 2. Qualitative estimates of uncertainty for target setting, and associated modelling processes, for the Tully–Murray basin and the Burdekin
L, low; M, moderate; H, high; E, extreme; TSS, total suspended solids; n/a, not applicable. See text for details

	Tully–Murray			Burdekin		
	Target	Model	Uncertainty	Target	Model	Uncertainty
Great Barrier Reef	Chl-$a = 0.5\,\mu g\,L^{-1}$	ChloroSim	L/M	TSS = 1.5 mg L^{-1}	n/a	M
			M			E
End-of-river	230 t DIN $year^{-1}$	SedNet/ANNEX	L	1.1 million tonnes TSS $year^{-1}$	SedNet/ANNEX	M
			L			M
Land management	100% 6ES		L	Hillslope		M
	100% NR		L	Gully		H
	100% NF		L	Streambank		M
Overall			L/M			M

include 100% adoption of (i) 'BSES Six Easy Steps' (6ES, Schroeder *et al.* 2006), which uses soil tests to work out optimal fertiliser rates, (ii) CSIRO Nitrogen replacement (NR; Thorburn *et al.* 2005), which is an emerging methodology that applies N in the order of 1 kg N for every tonne harvested in the previous crop, and (iii) nitrogen fixation (NF; J. Reghenzani, pers. comm.), which utilises free-living, nitrogen-fixing bacteria to supply most of the crop's nitrogen needs. Current nitrogen fertiliser rates in the Tully area are ~150 kg N ha^{-1} $year^{-1}$ (McMahon 2007), whereas under 6ES rates would be 140 kg N ha^{-1} $year^{-1}$, under NR 110 kg N ha^{-1} $year^{-1}$ and under NF 30 kg N ha^{-1} $year^{-1}$ (Kroon 2008). The scenarios estimated that the average total DIN load per year to end-of-river would be reduced by 29%, 59% and 86%, respectively (Armour *et al.* 2009). Thus, uptake of 6ES or NR, which are currently acceptable to industry, will produce substantial reductions in DIN load in the river, and the use of these management regimes can be set as interim targets. The level of uncertainty in the effectiveness of these measures is low because they have been tested through field trials, monitoring and modelling (Table 2).

Although the current modelled scenarios indicate significant water quality improvement towards sustainable loads, 6ES and NR do not achieve the required >80% reduction in nitrate concentration that the ChloroSim model estimates. This may be due in part to uncertainties in the models and the linkages between them, and the predicted effects of approved management actions will need to be validated in future monitoring programs. Additional proposed management actions, such as large-scale riparian rehabilitation in denitrification 'hotspots' (Rassam and Pagendam 2009), are expected to contribute to a reduction in current loads, but quantification of their effects has not been possible with available models. Alternatively, if current recommended management actions do not achieve the >80% reduction in nitrate concentrations, changes in land use may need to be assessed to further reduce DIN loads to achieve a sustainable targets load.

Target setting for the Burdekin basin for suspended sediment

The Burdekin River drains a large catchment area (~130 000 km^2) where the dominant land use is rangeland beef grazing. Since the introduction of beef cattle 150 years ago, erosion rates in the catchment have risen greatly (McCulloch *et al*. 2003; Lewis *et al*. 2007) and suspended sediment loads are now estimated to be five times greater than the loads before beef grazing commenced (Furnas 2003; McKergow *et al*. 2005*a*). The increased suspended sediment loads are believed to cause increased turbidity in marine waters, with adverse effects on coral reefs through loss of light and sedimentation (Philipp and Fabricius 2003; Fabricius 2005). The potential area of influence of increased suspended sediment loads from the Burdekin River extends widely over the GBR lagoon, with transport of material as far north as Cairns (Devlin and Brodie 2005).

An acceptable water quality guideline for turbidity for coral reefs is highly controversial. This is because the depth of water, physical factors such as clouds and tides (Anthony *et al*. 2004), the autotrophic–heterotrophic balance of the coral feeding (Anthony and Fabricius 2000), the nature of the particulate material (Fabricius *et al*. 2003; Weber *et al*. 2006) and other factors all interact to cause adverse effects (Cooper *et al*. 2007, 2008). Currently, the best estimate for a trigger value for total suspended solids (TSS) in coastal and inshore waters of the GBR is 1.5 mg L^{-1}, based on studies correlating reef condition with TSS concentrations (De'ath and Fabricius 2007; GBRMPA 2008). At present, mean TSS concentrations in coastal waters off the Burdekin River are 5.5 mg L^{-1} (SE = 0.4) (De'ath and Fabricius 2007), which exceed the trigger value by 3.6 times. These turbidity relationships and guidelines have moderate uncertainty (Table 2), because single values have been established for large areas of the GBR and regional variation is most likely significant but not fully allowed for in the trigger values.

The relationship between TSS loads from rivers discharging to the GBR, such as the Burdekin, and long-term regional turbidity is not fully understood, much less quantified. Turbidity in the inshore and coastal waters of the GBR is primarily driven by resuspension (Larcombe *et al*. 1995), in depths of 10 m or less, associated with the south-easterly wind regime and tidal currents. However, sediment supply to cause turbidity may not be limited by sediment supply from the rivers, and hence increased sediment loading may not cause increased turbidity (Larcombe and Woolfe 1999). Alternatively, each river sediment discharge event leads to the formation of a 'more resuspendable' benthic sediment layer, and hence results in a period of higher turbidity until the layer is dispersed or compacted. Evidence for this second scenario is currently limited but is the topic of current research (Wolanski and Spagnol 2000; Wolanski *et al*. 2005, 2008). For the purposes of target setting in the Burdekin, we made the assumption that coastal turbidity is directly proportional to river suspended sediment load (i.e. if load has doubled then the turbidity in coastal waters will double). This assumption is obviously of extreme uncertainty to the extent it may be completely incorrect (Table 2).

Current estimates of average annual TSS loads from the Burdekin River range between 3.5 million tonnes from modelling using SedNet (McKergow *et al*. 2005*a*; Fentie *et al*. 2006; Kinsey-Henderson *et al*. 2007) to 4.6 million tonnes estimated from monitoring (Mitchell *et al*. 2006; Bainbridge *et al*. 2007). However, the annual flow statistics of the Burdekin River demonstrate the extremely variable nature of this catchment, which influences TSS loads. The annual discharge for the Burdekin River has ranged from 250 (1930–31) to 54 000 (1973–74) GL, with a mean of 8400 GL over the period 1922–2005 (Bainbridge *et al*. 2007). Therefore, the sediment and nutrient loads exported from the catchment would also reflect this extreme variability. Assuming the event mean concentration (EMC) is consistent across flood events, our current best estimates of 'average' loads suggest an EMC for TSS lies between 420 and 550 mg L^{-1}. This equates to a range in the Burdekin River sediment load from 0.10 to 30 million tonnes per year in the period 1922–2005. This extreme range highlights the difficulty of applying 'averages' to the Burdekin River and the significant challenge of setting water quality targets for this catchment. The average annual TSS loads estimated by modelling and monitoring data are both based on long-term averages (models: 30 years; monitoring: 8 years) and are also based on mean annual discharge. Therefore, these estimates account for intra- and interannual variations as well as rainfall variability within the Burdekin River catchment. Thus, our best estimate of current TSS load is assumed to be 4.0 million tonnes, with a moderate level of uncertainty (Table 2) because of the large year-to-year variability but relatively good agreement between modelling and monitoring estimates (Bartley *et al*. 2007; Sherman *et al*. 2007). To reach the turbidity target of 1.5 mg L^{-1}, Burdekin River TSS loads need to be reduced by 3 or 4 times. Thus, the suggested 'aspirational' target (the Total Maximum Yearly Load – TMYL) becomes 1.4 million tonnes of TSS per year.

Modelling scenarios have been developed to examine the reduction in end-of-catchment sediment loads for the Burdekin River, with changes in ground cover, gully density and riparian condition (Kinsey-Henderson *et al*. 2007). Even though the models have many deficiencies, they provide the only tool available to assess relative changes in sediment loads. Currently, the SedNet model appears to overestimate the proportion of hillslope erosion in the Burdekin subcatchments where field observations suggest that gully erosion makes up an important contribution (Bartley *et al*. 2007; Kinsey-Henderson *et al*. 2007). Therefore, the scenarios designed to reduce hillslope erosion (i.e. increase in ground cover) were seen to result in the highest reduction of sediment loads at the end of the Burdekin River (Kinsey-Henderson *et al*. 2007). End-of-catchment sediment loads were reduced by 8% when ground cover was improved to 70% in eight Burdekin subcatchments identified by the SedNet model as high contributors of hillslope erosion (Kinsey-Henderson *et al*. 2007) (Table 1). A reduction in gully erosion in four priority catchments reduced end-of-catchment sediment loads by 2%, whereas an improvement in riparian zone condition at four subcatchments reduced the end-of-catchment load by 4%. The model findings suggest that catchment-wide improvements in groundcover, gully density and riparian condition in the Burdekin River catchment would reduce end-of-catchment sediment loads by 60% (Kinsey-Henderson *et al*. 2007).

Modelling of various land-use scenarios on the Burdekin catchment, involving improvements in pasture cover, reductions in gully erosion and reductions in streambank erosion, then allow estimates to be made of how far these management actions fall short of the TMYL target (Kinsey-Henderson *et al*. 2007). The scenarios are built around three forms of erosion management, including hillslope erosion, which can be managed through increasing pasture cover, gully erosion, for which management responses are currently highly uncertain, and streambank erosion, for which the management remedy is improved riparian vegetation. Management can be targeted at 'priority' subcatchments where SedNet modelling shows which form of erosion is most significant (Kinsey-Henderson *et al*. 2007). Overall, it appears that an extensive management program (50-year scenario, Table 1) in the Burdekin rangelands, with gully erosion reduction, riparian vegetation restoration and high pasture cover on priority subcatchments, leads to an outcome close (1.6 million tonnes) to the sustainable load for suspended sediment (1.4 million tonnes) for the system.

SedNet modelling predicts that high levels of suspended sediment trapping will occur in dams such as the Burdekin Falls Dam (Fentie *et al*. 2006). This has significant implications for the management of different parts of the catchment when considering overall sediment delivery; for example, management could be targeted in catchment areas below the dam wall. However, recent studies using monitoring data suggest that trapping in the Burdekin Falls Dam is lower than that in modelled estimates (average 60% instead of the modelled 80%; Lewis *et al*. 2009), and therefore careful consideration is required regarding location of management efforts.

Although the approach to the TMYL target is set at a 50-year time frame, intermediate targets can be planned over shorter time frames (e.g. 5 and 12 years, examples in Table 1). The quantitative basis of the scenarios has moderate uncertainty because quantitative information on the effectiveness of the current grazing-land management practices, e.g. the Grazing Lands Management (GLM) package, to prevent hillslope erosion is at best moderately certain because of the lack of experimental verification, including problems with long lag times before improvements are evident (O'Reagain *et al*. 2005). Mechanisms to control gully erosion are highly uncertain because these have not yet been trialled successfully compared with riparian vegetation restoration as a means of reducing streambank erosion, which is of low uncertainty as this is a tested and proven method in many parts of the world.

Discussion and conclusions

We are thus faced with a linking set of models from the ecosystem end-point target to the management action target. The severe lack of quantitative knowledge between river pollutant loads and reef ecological effects makes target setting an uncertain process in this environment. The uncertainty of each link of the models ranges from low to extreme, whereas the uncertainty of the whole chain of quantitative causation is low to moderate (Tully) and moderate (Burdekin) (Table 2). The difference between the Burdekin and Tully is primarily due to one link in the chain, the river sediment load–turbidity relationship. Although still uncertain, this process of target setting is preferable to previous processes of nitrogen target setting for GBR rivers (e.g. Brodie *et al*. 2001*b*), which were much more '*ad hoc*', with no attempt to connect a marine ecosystem end-point target to the river load. The process also allows management scenarios to be run using the models and a comparison of the results to fixed end-point targets to be made. The major challenge is to improve modelling such that we can have a greater degree of confidence that the level of management is adequate to provide the ecosystem protection level we require.

Another approach being trialled is the use of Bayesian Belief Networks (BBN) as a model integration tool (see Thomas *et al*. 2005; Shenton *et al*. in press). Bayesian techniques have been used in water quality modelling for some time in systems such as the eutrophied Neuse River in the USA (Borsuk *et al*. 2004), where they have been shown to be highly applicable to analysis of systems, with complex causal chains linking the watershed to estuarine and marine waters. The techniques are also ideal for dealing with multiple data sources such as process-based models, statistical and regression models, long-term monitoring data and expert opinion (Borsuk *et al*. 2004). The use of the Bayesian approach in the Tully allows us to have one model linking paddock to reef and use the individual models mentioned in the target setting process as components to potentially populate a BBN.

As a result of limitations in monitoring and modelling capacity (Brodie *et al*. 2008), water quality targets are presently largely driven by an understanding of what is an achievable water quality improvement within current land-use systems and practices. The environmental tradeoffs in setting targets have not yet received much attention because of the low confidence in our understanding of what is actually being discharged from catchments and how this relates to requirements to sustain healthy GBR ecosystems. The implications for ecosystems of not meeting targets, and the lag time to change practices and realise water quality benefits for the target adopted have high levels of uncertainty. This is apparent in the marine environment, where relationships between water quality parameters and the resilience of GBR ecosystems are still emerging (Brodie *et al*. 2008). The establishment of the GBRMPA (2008) Water Quality Guidelines and the supporting science documentation (De'ath and Fabricius 2007) provide a substantial advancement towards setting marine water quality targets; however, considerable work is required to define and measure desired water quality outcomes in the relatively short policy timeframes.

Monitoring progress towards targets of this type is difficult because of time lags in the system response, interannual climate and hydrological variability, and spatial and logistical constraints (Bainbridge *et al*. 2009*a*). As a result of these difficulties, monitoring activities often also end up being dependent on modelling activities (Bainbridge *et al*. 2009*a*) in a similar way to target setting. The factor of improved system understanding with time and related better modelling systems and data inputs also means that we have to be careful we are monitoring success against fixed targets in a consistent way, without new knowledge shifting the targets. Improved knowledge through a time-dependent management process can be captured using an adaptive management approach as has been developed for the Tully WQIP (Eberhard *et al*. 2009). Adopting this approach allows us systematically to take advantage of new knowledge, e.g. models, while keeping track of which models and data have been used in each step of the management process.

Acknowledgements

This study was supported by the Terrain NRM Tully Water Quality Improvement Plan and the Burdekin Dry Tropics NRM Water Quality Improvement Plan (Ian Dight in particular) through the Department of the Environment, Water, Heritage and the Arts. Funding was also provided by CSIRO's Water for a Healthy Country Flagship Program and the Australian Government's Marine and Tropical Sciences Research Facility.

References

Anonymous (2003). Reef Water Quality Protection Plan: for catchments adjacent to the Great Barrier Reef World Heritage Area. Queensland Department of Premier and Cabinet. Brisbane. The State of Queensland and Commonwealth of Australia. Available at http://www.reefplan.qld.gov.au [verified 4 October 2009].

Anthony, K. R. N., and Fabricius, K. E. (2000). Shifting roles of heterotrophy and autotrophy in coral energetics under varying turbidity. *Journal of Experimental Marine Biology and Ecology* **252**, 221–253. doi:10.1016/S0022-0981(00)00237-9

Anthony, K. R. N., Ridd, P. V., Orpin, A. R., Larcombe, P., and Lough, J. (2004). Temporal variation of light availability in coastal benthic habitats: Effects of clouds, turbidity and tides. *Limnology and Oceanography* **49**, 2201–2211.

Armour, J., Hateley, L., and Pitt, G. (2009). Catchment modelling of sediment, nitrogen and phosphorus nutrient loads with SedNet/ANNEX in the Tully–Murray basin. *Marine and Freshwater Research* **60**, 1091–1096.

Bainbridge, Z., Lewis, S., and Brodie, J. (2007). Event-based community water quality monitoring in the Burdekin Dry Tropics region: 2002–2007 (Vol 2). Australian Centre for Tropical Freshwater Research, James Cook University, ACTFR Report No. 07/22 for BDTNRM, Townsville, Qld. Available at http://www.actfr.jcu.edu.au/idc/groups/public/documents/technical_report/jcudev_016591.pdf [verified 4 October 2009].

Bainbridge, Z., Brodie, J., Lewis, S., Waterhouse, J., and Wilkinson, S. (2009*a*). Utilising catchment modelling as a tool for monitoring Reef Rescue outcomes in the Great Barrier Reef catchment area. In '18th World IMACS Congress and MODSIM09 International Congress on Modelling and Simulation', July 2009. (Eds R. S. Anderssen, R. D. Braddock and L. T. H. Newham.) pp. 3301–3307. (Modelling and Simulation Society of Australia and New Zealand and International Association for Mathematics and Computers in Simulation.) Available at http://www.mssanz.org.au/modsim09/I4/bainbridge.pdf [verified 4 October 2009].

Bainbridge, Z., Brodie, J., Faithful, J., Sydes, D., and Lewis, S. (2009*b*). Identifying the land-based sources of suspended sediment, nutrients and pesticides discharged to the Great Barrier Reef from the Tully–Murray Basin, Queensland, Australia. *Marine and Freshwater Research* **60**, 1081–1090.

Bartley, R., Post, D., Kinsey-Henderson, A., and Hawdon, A. (2007). Estimating sediment loads in Great Barrier Reef catchments: balance between modelling and monitoring. In 'Proceedings of the 5th Australian Stream Management Conference, Thurgoona, May 2007'. (Eds A. L. Wilson, R. L. Dehaan, R. J. Watts, K. J. Page, K. H. Bowmer and A. Curtis.) pp. 13–18. (Institute for Land, Water and Society, Charles Sturt University: Albury, Vic.)

Borsuk, M. E., Stow, C. A., and Reckhow, K. H. (2004). A Bayesian network of eutrophication models for synthesis, prediction and uncertainty analysis. *Ecological Modelling* **173**, 219–239. doi:10.1016/J.ECOLMODEL.2003.08.020

Brodie, J., and Bainbridge, Z. (2008). Water quality targets for the Burdekin WQIP. Australian Centre for Tropical Freshwater Research, James Cook University, ACTFR Report 08/05, Townsville, Qld.

Brodie, J., Christie, C., Devlin, M., Haynes, D., Morris, S., Ramsay, M., Waterhouse, J., and Yorkston, H. (2001*a*). Catchment management and the Great Barrier Reef. *Water Science and Technology* **43**, 203–211.

Brodie, J., Furnas, M., Ghonim, S., Haynes, D., Mitchell, A., *et al.* (2001*b*). Great Barrier Reef Catchment Water Quality Action Plan. Great Barrier Reef Marine Park Authority, Townsville, Qld.

Brodie, J., Fabricius, K., De'ath, G., and Okaji, K. (2005). Are increased nutrient inputs responsible for more outbreaks of crown-of-thorns starfish? An appraisal of the evidence. *Marine Pollution Bulletin* **51**, 266–278. doi:10.1016/J.MARPOLBUL.2004.10.035

Brodie, J., De'ath, G., Devlin, M., Furnas, M., and Wright, M. (2007). Spatial and temporal pattern of near-surface chlorophyll a in the Great Barrier Reef lagoon. *Marine and Freshwater Research* **58**, 342–353. doi:10.1071/MF06236

Brodie, J., Binney, J., Fabricius, K., Gordon, I., Hoegh-Gulberg, O., *et al.* (2008). Scientific consensus statement on water quality in the Great Barrier Reef. The Reef Water Quality Protection Plan Secretariat, Queensland Department of Premier and Cabinet, Brisbane.

Bruno, J. F., and Selig, E. R. (2007). Regional decline of coral cover in the Indo-Pacific: Timing, extent and subregional comparisons. *Public Library of Science ONE* **2**(8), e711.

Cooper, T. F., Uthicke, S., Humphrey, C., and Fabricius, K. E. (2007). Gradients in water column nutrients, sediment parameters, irradiance and coral reef development in the Whitsunday Region, central Great Barrier Reef. *Estuarine, Coastal and Shelf Science* **74**, 458–470. doi:10.1016/J.ECSS.2007.05.020

Cooper, T. F., Ridd, P. V., Ulstrup, K. E., Humphrey, C., Slivkoff, M., and Fabricius, K. E. (2008). Temporal dynamics in coral bioindicators for water quality on coastal coral reefs of the Great Barrier Reef. *Marine and Freshwater Research* **59**, 703–716. doi:10.1071/MF08016

De'ath, G., and Fabricius, K. (2007). Water quality of the Great Barrier Reef: Distributions, effects on reef biota and trigger values for conservation of ecosystem health. Australian Institute of Marine Science, Final report to the Great Barrier Reef Marine Park Authority, Townsville, Qld.

DePinto, J. V., Frredman, P. L., Dilks, D. M., and Larson, W. M. (2004). Models quantify the total maximum daily load process. *Journal of Environmental Engineering* **130**, 703–713. doi:10.1061/(ASCE)0733-9372(2004)130:6(703)

DeVantier, L., De'ath, G., Turak, E., Done, T., and Fabricius, K. E. (2006). Species richness and community structure of reef-building corals on the nearshore Great Barrier Reef. *Coral Reefs* **25**, 329–340. doi:10.1007/S00338-006-0115-8

Devlin, M. J., and Brodie, J. (2005). Terrestrial discharge into the Great Barrier Reef Lagoon: nutrient behaviour in coastal waters. *Marine Pollution Bulletin* **51**, 9–22. doi:10.1016/J.MARPOLBUL.2004.10.037

Devlin, M., and Schaffelke, B. (2009). Spatial extent of riverine flood plumes and exposure of marine ecosystems in the Tully coastal region, Great Barrier Reef. *Marine and Freshwater Research* **60**, 1109–1122.

Dight, I. (2009). 'Draft Burdekin Water Quality Improvement Plan.' (Burdekin Solutions Ltd: Townsville, Qld.)

Dougall, C., Carroll, C., Herring, M., and Trevithick, R. (2007). SedNet modelling in the Fitzroy Basin; spatially variable ground cover and revised gully layers can potentially generate significant changes in erosion sources and patterns. In 'Proceedings of the International Congress on Modelling and Simulation (MODSIM 2007) Conference, Christchurch, December 2007'. (Eds L. Oxley and D. Kulasiri.) pp. 881–887. (Modelling and Simulation Society of Australian and New Zealand: Christchurch, New Zealand.) ISBN: 978-0-9758400-4-7. http:/www.mssanz.au/modsim07/Papers

Eberhard, R., Robinson, C., Waterhouse, J., Parslow, J., Hart, B., Grayson, R., and Taylor, B. (2009). Adaptive management for water quality planning – from theory to practice. *Marine and Freshwater Research* **60**, 1189–1195.

Fabricius, K. E. (2005). Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. *Marine Pollution Bulletin* **50**, 125–146. doi:10.1016/J.MARPOLBUL.2004.11.028

Fabricius, K. E. (2007). Conceptual Model of the effects of terrestrial runoff on the ecology of corals and coral reefs of the GBR. Australian Institute of Marine Science, Final Report to the Australian Government's Marine and Tropical Sciences Research Facility, Townsville, Qld.

Fabricius, K. E., Wild, C., Wolanski, E., and Abele, D. (2003). Effects of transparent exopolymer particles and muddy terrigenous sediments on the survival of hard coral recruits. *Estuarine, Coastal and Shelf Science* **57**, 613–621. doi:10.1016/S0272-7714(02)00400-6

Fabricius, K. E., De'ath, G., McCook, L. J., Turak, E., and Williams, D. (2005). Coral, algae and fish communities on inshore reefs of the Great Barrier Reef: Ecological gradients along water quality gradients. *Marine Pollution Bulletin* **51**, 384–398. doi:10.1016/J.MARPOLBUL.2004.10.041

Fentie, B., Duncan, I., Sherman, B. S., Read, A., Chen, Y., Brodie, J., and Cogle, A. L. (2006). Sediment and nutrient modelling in the Burdekin NRM region. Vol. 3. In 'The Use of SedNet and ANNEX Models to Guide GBR Catchment Sediment and Nutrient Target Setting'. QNRM06138. (Eds A. L. Cogle, C. Carroll and B. S. Sherman.) pp. 3.1–3.29. (Department of Natural Resources Mines and Water: Brisbane.)

Furnas, M. (2003). 'Catchments and Corals: Terrestrial Runoff to the Great Barrier Reef.' (Australian Institute of Marine Science: Townsville, Qld.)

GBRMPA (2008). Water Quality Guidelines for the Great Barrier Reef Marine Park (Draft). Great Barrier Reef Marine Park Authority, Townsville, Qld. Available at http://www.gbrmpa.gov.au/corp_site/key_issues/water_quality/draft_water_quality_guidelines [verified 4 October 2009].

Herr, A., and Kuhnert, P. M. (2007). Assessment of uncertainty in Great Barrier Reef catchment models. *Water Science and Technology* **56**, 181–188. doi:10.2166/WST.2007.450

Hughes, T. P., Rodrigues, M. J., Bellwood, D. R., Ceccerelli, D., Hoegh-Guldberg, O., McCook, L., Moltchaniwskyj, N., Pratchett, M. S., Steneck, R. S., and Willis, B. L. (2007). Phase shifts, herbivory, and the resilience of coral reefs to climate change. *Current Biology* **17**, 360–365. doi:10.1016/J.CUB.2006.12.049

Hunter, H. M., and Walton, R. S. (2008). Land-use effects on fluxes of suspended sediment, nitrogen and phosphorus from a river catchment of the Great Barrier Reef, Australia. *Journal of Hydrology (Amsterdam)* **356**, 131–146. doi:10.1016/J.JHYDROL.2008.04.003

Johnson, A. K. L., and Ebert, S. P. (2000). Quantifying inputs of pesticides to the Great Barrier Reef Marine Park – A case study in the Herbert River catchment of north-east Queensland. *Marine Pollution Bulletin* **41**, 302–309. doi:10.1016/S0025-326X(00)00130-2

Karr, J. R., and Yoder, C. O. (2004). Biological assessment and criteria improve Total Maximum Daily Load decision making. *Journal of Environmental Engineering* **130**, 594–604. doi:10.1061/(ASCE)0733-9372(2004)130:6(594)

Kinsey-Henderson, A., Sherman, B., and Bartley, R. (2007). Improved SedNet modelling of grazing lands in the Burdekin catchment. CSIRO Land and Water, Science Report 63/07 for Burdekin Dry Tropics NRM, Townsville, Qld.

Kroon, F. J. (2008). Draft Tully Water Quality Improvement Plan. CSIRO Sustainable Ecosystems, Report to Terrain NRM, Atherton, Qld.

Kroon, F. J. (2009). Integrated research to improve water quality in the Great Barrier Reef region. *Marine and Freshwater Research* **60**, i–iii.

Larcombe, P., and Woolfe, K. (1999). Increased sediment supply to the Great Barrier Reef will not increase sediment accumulation at most coral reefs. *Coral Reefs* **18**, 163–169. doi:10.1007/S003380050174

Larcombe, P., Ridd, P. V., Prytz, A., and Wilson, B. (1995). Factors controlling suspended sediment on inner-shelf coral reefs, Townsville, Australia. *Coral Reefs* **14**, 163–171. doi:10.1007/BF00367235

Lewis, S., Shields, G., Kamber, B., and Lough, J. (2007). A multi-trace element coral record of land-use changes in the Burdekin River catchment, NE Australia. *Palaeogeography, Palaeoclimatology, Palaeoecology* **246**, 471–487. doi:10.1016/J.PALAEO.2006.10.021

Lewis, S. E., Sherman, B. S., Bainbridge, Z. T., Brodie, J. E., and Cooper, M. (2009). Modelling and monitoring the sediment trapping efficiency and sediment dynamics of the Burdekin Falls Dam, Queensland, Australia. In '18th World IMACS Congress and MODSIM09 International Congress on Modelling and Simulation', July 2009. (Eds R. S. Anderssen, R. D. Braddock and L. T. H. Newham.) pp. 4022–4028. (Modelling and Simulation Society of Australia and New Zealand and International Association for Mathematics and Computers in Simulation.) Available at http://www.mssanz.org.au/modsim09/I14/lewis.pdf [verified 4 October 2009].

Littler, M. M., Littler, D. S., and Brooks, B. L. (2006). Harmful algae on tropical coral reefs: bottom-up eutrophication and top-down herbivory. *Harmful Algae* **5**, 565–585. doi:10.1016/J.HAL.2005.11.003

Lough, J. M. (2008). A changing climate for coral reefs. *Journal of Environmental Monitoring* **10**, 21–29. doi:10.1039/B714627M

McCook, L. J., Jompa, J., and Diaz-Pulido, G. (2001). Competition between corals and algae on coral reefs: a review of evidence and mechanisms. *Coral Reefs* **19**, 400–417. doi:10.1007/S003380000129

McCulloch, M. T., Fallon, S., Wyndham, T., Hendy, E., Lough, J., and Barnes, D. (2003). Coral record of increased sediment flux to the inner Great Barrier Reef since European settlement. *Nature* **421**, 727–730. doi:10.1038/NATURE01361

McDonald, G., and Roberts, B. (2006). SMART water quality targets for Great Barrier Reef catchments. *Australasian Journal of Environmental Management* **13**, 95–107.

McIntyre, N. R., and Wheater, H. S. (2004). A tool for risk-based management of surface water quality. *Environmental Modelling & Software* **19**, 1131–1140. doi:10.1016/J.ENVSOFT.2003.12.003

McKergow, L. A., Prosser, I. P., Hughes, A. O., and Brodie, J. (2005*a*). Sources of sediment to the Great Barrier Reef World Heritage Area. *Marine Pollution Bulletin* **51**, 200–211. doi:10.1016/J.MARPOLBUL.2004.11.029

McKergow, L. A., Prosser, I. P., Hughes, A. O., and Brodie, J. (2005*b*). Regional scale nutrient modelling: exports to the Great Barrier Reef World Heritage Area. *Marine Pollution Bulletin* **51**, 186–199. doi:10.1016/J.MARPOLBUL.2004.11.030

McMahon, M. (2007). Summary statistics of sugarcane industry best practice management. Bureau of Sugar Experiment Stations, Report to Terrain NRM, Tully, Qld.

Mitchell, A., Reghenzani, J., and Furnas, M. (2001). Nitrogen levels in the Tully River – a long-term view. *Water Science and Technology* **43**, 99–105.

Mitchell, A., Furnas, M., De'ath, G., Brodie, J., and Lewis, S. (2006). A report into the water quality condition of the Burdekin River and surrounds based on the AIMS end-of-catchment sampling program. Australian Centre for Tropical Freshwater Research, James Cook University, ACTFR Report No. 06/06 for BDTNRM, Townsville, Qld.

Mitchell, A., Reghenzani, J., Faithful, J., Furnas, M., and Brodie, J. (2009). Relationships between land use and nutrient concentrations in streams draining a 'wet-tropics' catchment in northern Australia. *Marine and Freshwater Research* **60**, 1097–1108.

Mitchell, C., Brodie, J., and White, I. (2005). Sediments, nutrients and pesticide residues in event flow conditions in streams of the Mackay Whitsunday Region, Australia. *Marine Pollution Bulletin* **51**, 23–36. doi:10.1016/J.MARPOLBUL.2004.10.036

Moss, A., Brodie, J., and Furnas, M. (2005). Water quality guidelines for the Great Barrier Reef World Heritage Area: A basis for development and preliminary values. *Marine Pollution Bulletin* **51**, 76–88. doi:10.1016/J.MARPOLBUL.2004.10.052

Newham, L. T. H., Norton, J. P., Prosser, I. P., Croke, B. F. W., and Jakeman, A. J. (2003). Sensitivity analysis for assessing the behaviour of a landscape-based sediment source and transport model. *Environmental Modelling & Software* **18**, 741–751. doi:10.1016/S1364-8152(03)00076-8

O'Reagain, P. J., Brodie, J., Fraser, G., Bushell, J. J., Holloway, C. H., Faithful, J. W., and Haynes, D. (2005). Nutrient loss and water quality under extensive grazing in the upper Burdekin River catchment, north Queensland. *Marine Pollution Bulletin* **51**, 37–50. doi:10.1016/J.MARPOLBUL.2004.10.023

Pandolfi, J. M., Bradbury, R. H., Sala, E., Hughes, T. P., Bjorndal, K. A., *et al*. (2003). Global trajectories of the long-term decline of coral reef ecosystems. *Science* **301**, 955–958. doi:10.1126/SCIENCE.1085706

Philipp, E., and Fabricius, K. (2003). Photophysiological stress in scleractinian corals in response to short-term sedimentation. *Journal of Experimental Marine Biology and Ecology* **287**, 57–78. doi:10.1016/S0022-0981(02)00495-1

Rassam, D., and Pagendam, D. (2009). Development and application of the Riparian Mapping Tool to identify priority rehabilitation areas for nitrogen removal in the Tully–Murray basin, Queensland, Australia. *Marine and Freshwater Research* **60**, 1165–1175.

Roebeling, P., van Grieken, M. E., Webster, A. J., Biggs, J., and Thorburn, P. (2009). Cost-effective water quality improvement in linked terrestrial and marine ecosystems: a spatial environmental–economic modelling approach. *Marine and Freshwater Research* **60**, 1150–1158.

Rohde, K., Masters, B., Fries, N., Noble, B., and Carroll, C. (2008). Fresh and marine water quality in the Mackay Whitsunday region 2004/05 to 2006/07. Queensland Department of Natural Resources and Water for MWNRM, Mackay, Qld.

Schroeder, B. L., Wood, A. W., Moody, P. W., Panitz, J. H., Agnew, J. R., Sluggett, R. J., and Salter, B. (2006). Delivering nutrient management guidelines to growers in the central region of the Australian sugar industry. *Proceedings Australian Society Sugar Cane Technologists* **28**, 142–154.

Shenton, W., Hart, B. T., and Brodie, J. (in press). A Bayesian network model linking nutrient management actions in the Tully catchment (north Queensland) with reef condition. *Marine and Freshwater Research*, in press.

Sherman, B. S., and Read, A. M. (2008). Uncertainty in Great Barrier Reef catchment soil nutrient data – implications for land use management. In 'Proceedings of iEMSs 4th Biennial Meeting of the International Congress on Environmental Modelling and Software, Barcelona, July 2008'. (Eds M. Sanchez-Marre, J. Bejar, J. Comas, A. Rizzoli and G. Guariso.) pp. 566–573. (International Environmental Modelling and Software Society.)

Sherman, B., Brodie, J., Cogle, L., and Carroll, C. (2007). Appropriate use of catchment models for water quality target setting and land-use management. In 'Proceedings of HS2005 at IUGG2007 Symposium on Water Quality and Sediment Behaviour of the Future: Predictions for the 21st Century, Perugia, July 2007'. Publication No. 314. (Eds B. W. Webb and D. De Boer.) pp. 239–250. (IAHS Press: Wallingford, UK.)

Thomas, C., Hart, B., Nicholson, A., Grace, M., Brodie, J., and Pollino, C. (2005). Development of criteria for simplifying ecological risk models. In 'MODSIM 2005 International Congress on Modelling and Simulation, Melbourne, December 2005'. (Eds A. Zerger and R. M. Argent.) pp. 772–778. (Modelling and Simulation Society of Australia and New Zealand.)

Thorburn, P. J., Meier, E. A., and Probert, M. E. (2005). Modelling nitrogen dynamics in sugarcane systems: Recent advances and applications. *Field Crops Research* **92**, 337–351. doi:10.1016/J.FCR.2005.01.016

Wallace, J., Stewart, L., Hawdon, A., Keen, R., Karim, F., and Kemei, J. (2009). Flood water quality and marine sediment and nutrient loads from the Tully and Murray catchments in north Queensland, Australia. *Marine and Freshwater Research* **60**, 1123–1131.

Weber, M., Lott, C., and Fabricius, K. E. (2006). Sedimentation stress in a scleractinian coral exposed to terrestrial and marine sediments with contrasting physical, organic and geochemical properties. *Journal of Experimental Marine Biology and Ecology* **336**, 18–32. doi:10.1016/J.JEMBE.2006.04.007

Wolanski, E., and Spagnol, S. (2000). Pollution by mud of Great Barrier Reef coastal waters. *Journal of Coastal Research* **16**, 1151–1156.

Wolanski, E., Fabricius, K., Spagnol, S., and Brinkman, R. (2005). Fine sediment budget on an inner-shelf coral-fringed island, Great Barrier Reef of Australia. *Estuarine, Coastal and Shelf Science* **65**, 153–158. doi:10.1016/J.ECSS.2005.06.003

Wolanski, E., Fabricius, K. E., Cooper, T. F., and Humphrey, C. (2008). Wet season fine sediment dynamics on the inner shelf of the Great Barrier Reef. *Estuarine, Coastal and Shelf Science* **77**, 755–762. doi:10.1016/J.ECSS.2007.10.014

Wooldridge, S., Brodie, J., and Furnas, M. (2006). Exposure of inner-shelf reefs to nutrient enriched runoff entering the Great Barrier Reef Lagoon: Post-European changes and the design of water quality targets. *Marine Pollution Bulletin* **52**, 1467–1479. doi:10.1016/J.MARPOLBUL.2006.05.009

Manuscript received 11 December 2008, accepted 1 September 2009