2013 Scientific Consensus Statement

Chapter 5
The water quality and economic benefits of agricultural management practices

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This document was prepared by an independent panel of scientists with expertise in Great Barrier Reef water quality. This document does not represent government policy.

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Executive summary

There has been a substantial increase in knowledge about the water quality and economic outcomes of agricultural management practices on farms. There are a number of potential actions that can be undertaken to reduce major pollutants such as sediments, nutrients and pesticides, and better information is becoming available about the benefits and tradeoffs of these reductions, as summarised below. The question of the extent to which adoption of these field-scale management actions will meet water quality improvement targets is also starting to be considered.

Sediments

- It is well established that sediment loads are reduced by maintaining ground cover and forage biomass to enhance infiltration, and redistributing the pressure of agricultural activities away from areas vulnerable to erosion.
- In grazing lands, these aims will be achieved by:
  - setting stocking rates that maintain ground vegetation cover and biomass (including during droughts and at the end of the dry season) and vegetation diversity (including maintaining some tree cover, especially in riparian areas)
  - reducing or excluding stock from, and increasing groundcover in riparian and frontage country, rilled, scalded and gullied areas, and wetlands.
- Adopting these practices in grazing lands will improve soil condition and reduce sediment loads, especially where land condition has been rundown (so infiltration is low and groundcover hard to maintain). The response time varies depending on starting vegetation and land condition, from several years to several decades, and the degree to which soil condition can be improved is uncertain. Including physical remediation works may hasten recovery of degraded and highly-eroding areas, although the success and costs of this are not well documented.
- In cropped lands, management systems that reduce or eliminate tillage and maximise soil cover (via crop residue retention and grassed inter-rows) have been demonstrated to reduce soil loss in most cropping systems of the Great Barrier Reef. Controlled traffic, opportunity cropping and contour embankments also reduce runoff and soil loss.
- Targeting practice improvement to areas contributing most to soil loss, considering erosion rates, soil texture and location of sediment traps including reservoirs, can increase the effectiveness at the Great Barrier Reef scale.

Nutrients

- Losses of nitrogen, both dissolved and total, are related to nitrogen fertiliser applications and the nitrogen surplus (i.e. the difference between nitrogen inputs and nitrogen in crop off-take) at both the field and whole-Great Barrier Reef scales, and to soil erosion. The same principle applies to phosphorous.
- Nitrogen surpluses are high (e.g. 100-250 kilograms per hectare per year) in many intensively managed crops. However, phosphorus surpluses are generally lower (e.g. less than 70 kilograms per hectare per year). Where surpluses are high, nutrient loads can be reduced by reducing nutrient inputs and reducing the surplus. Nutrient management recommendations generally aim to supply nutrients for potential (as opposed to actual) crop yields with the result that nutrients are over-applied where crop growth does not reach that potential. Aligning nutrient inputs to actual crop yields will reduce nutrient loads.
- As nutrient application rates are reduced, the risk that crop yields will be limited by nutrient supply increases. The risk can be managed by determining the minimum nutrient surplus needed to maintain maximum yields (or profitability). That information will also indicate the maximum water quality improvement that can be achieved in cropped lands of the Great Barrier Reef.

- For nitrogen, management tactics such as splitting or altering the timing of fertiliser applications, using 'enhanced efficiency fertilisers or burying fertiliser also help manage the risk of nitrogen supply limiting yield. However, these tactics are unlikely to be effective unless they are accompanied by reduced application rates. They are most likely to be beneficial where nitrogen surpluses are low, but have yet to be well evaluated in these situations.
- Soil management practices that reduce runoff and sediment movement (described above) also reduce loads of particulate and total nutrients in runoff. However, for dissolved nutrients, especially nitrogen, reducing losses in runoff may increase losses by other pathways (leaching or denitrification) and so may not give an overall environmental benefit.
- Irrigation is generally believed to exacerbate nutrient losses from cropping. However, this is not the case in sugarcane, probably because of the high rainfall received by many rain fed sugarcane crops. The situation is unclear for horticultural crops and crops grown in drier areas (e.g. cropping lands in the Fitzroy catchment). As well, on many farms irrigation tail-water is captured and reused, reducing potential for off-farm losses. It is clear is that increasing irrigation efficiency (i.e. reducing over-application of irrigation) reduces nutrient losses. Efficiencies can be increased either by better managing irrigation within a given system, or moving from systems with lower (e.g. furrow) to higher (e.g. trickle) efficiency.
- Nutrients input to fields include sources other than fertiliser, such as nitrogen from legumes, and nitrogen and phosphorus from mill mud in sugarcane areas. If these inputs are not accounted for in the nutrient management of crops, inputs via fertilisers will increase nutrient surpluses and are likely to have water quality impacts. It is uncertain whether the resultant water quality impacts will occur as local 'hot spots', or contribute substantially to regional problems. Ideas on managing fertiliser to overcome these increased surpluses are still evolving and their effects of water quality have not been established. There is little information on nutrient losses from, or nutrient management in fertilised grazing lands. However, nutrient management guidelines for grazing lands will still be underpinned by the same principles as those guiding nutrient management in cropping lands.
- Targeting management practice improvement to areas contributing most to soil loss will be effective in reducing loss of particulate nutrients. However, the evidence of benefits from targeting management is less clear for dissolved nutrients, and is most likely predicated by the need to reduce nutrient inputs.

Pesticides

- Soil management practices that reduce runoff and sediment movement (retention of crop residues, controlled traffic, etc. as described above) also reduce pesticide runoff.
- Managing pesticide input timing (increasing the time between application and runoff), as well as amount, placement and application method (e.g. banded spraying) also reduce pesticide runoff.
- Applying products with rapid degradation rates (e.g. some 'knockdown' herbicides) will reduce pesticide concentrations and loads in runoff.
- Changes in pesticide management should be undertaken within the context of integrated pest or weed management, managing resistance and considering ecotoxicology.
- There is considerable dilution of pesticide concentrations over relatively short distances after they leave fields. Thus, it may be important to target management action to fields close to receiving creek systems. Managing pesticide in irrigation areas may be most important in the dry season when dilution from rainfall runoff is absent.

Water quality targets

- The universal adoption of 'B-Class' practices is unlikely to meet water quality improvement targets for fine sediments, total nitrogen or total phosphorus, but may for photosystem II inhibiting herbicides.
- Management of agricultural lands will need to move beyond current industry accepted practices to more 'aspirational practices' if water quality targets are to be met.
- The unproven nature of these aspirational practices will slow the rate of their adoption, as they will need considerable testing and development before being recommended.

Wetlands

- Even with best management, some pollutants will leave fields and enter the broader landscape. Wetlands can have a role in improving the quality of this water by reducing or assimilating nutrient, removing solids, changing chemical parameters such as biological oxygen demand, and removing contaminants. However, wetlands should not be used to replace good land management practices.
- There have been few studies that measure the efficacy of natural wetlands in Great Barrier Reef catchments on improving water quality. Wetland efficacy depends on a range of characteristics, including position in the landscape, vegetation type, water type, water regime, hydrological processes and residence times. Thus, not all wetlands will improve the quality of water flowing through them. Due to these complexities, results for particular wetlands might not be generally applicable.
- Wetlands need to be managed in accordance with best management practice to ensure their multiple values are realised. Management may include periodic removal of excess nutrients and weed germplasm to ensure that the wetlands retain and/or improve their capacity to assimilate and cycle nutrients and sediments and break down contaminants. If not managed appropriately, these accumulated materials may be 'flushed out' of the wetland and flow into the river network to the Great Barrier Reef during flood events.
- Wetlands have much broader values in the landscape than just water quality improvement, including, reducing water velocity and flood mitigation, storing and transferring water, nutrient cycling, supporting biodiversity, connectivity, ecological processes such as breeding and recruitment of fish, carbon storage and local climate regulation. Poor quality water flowing into freshwater wetlands can impair these functions and so have serious consequences for wider landscape health and productivity.
- Constructed wetlands are designed to mimic the role of natural wetlands, in particular palustrine wetlands. They are built to enhance water quality through nutrient cycling, absorb metals and other toxicants and break down pesticides, and so can be effective in improving water quality.

Economics

- Landholders do not simply follow short term profit signals, but make land management decisions according to a complex mix of drivers, including historical patterns, their ability to adapt to changed conditions, and their personal characteristics and circumstances.
- Costs of changing management to improve water quality vary greatly (e.g. by two orders of magnitude) between different agricultural enterprises on a per-hectare basis. Variations in costs occur because of differences in natural resources, farm operations, property scale, landholder knowledge and attitudes, and the transaction costs in making changes. Therefore, it is difficult to assume a single economic outcome of changed management.
- Improving farm management to meet an industry-based Best Management Practice often gives positive economic benefits in the long-term. Nevertheless, the economic benefits may not be large enough to drive adoption of all best management practices, especially after transaction costs are considered. External support may be needed to encourage some changes. However, economic studies, like those on nutrient management, suggest that universal adoption of current industry based best management practices will not be enough or occur fast enough to meet water quality targets.
- Making further changes to management to improve water quality 'beyond' industry-based best management practices are likely to come at a cost to farmers. The costs may also have broader regional impacts if, for example, they threaten the viability of a sugar mill and local businesses.
- The transaction costs associated with changing management and the risk-averse profiles of farmers are factors that can prove significant barriers to adoption of improved practice. However, these are not well understood.
- General understanding of the economics of changing management practices in agriculture is limited, and the extension of this information to farmers needs to be improved.

Introduction

The iconic Great Barrier Reef marine ecosystems are interconnected with freshwater wetlands and the wider landscape of the Great Barrier Reef catchments. The impacts of poor water quality on both marine and freshwater ecosystems can have far reaching impacts such as on the food chain, nutrient cycling, species distribution, abundance, population size, growth and reproduction.

Increasing pressure from human activities and climate change continues to have an adverse impact on the quality of water entering freshwater wetlands in the Great Barrier Reef catchments as well as the lagoon itself, particularly during flood events. The relationships between land use, catchment and wetlands management, declining water quality and Great Barrier Reef ecosystem health are now better understood. There is well-documented evidence of the damaging impacts of terrestrial pollutants on the condition of coastal and inshore Great Barrier Reef ecosystems (Brodie *et al*., 2012) and the extent of change required in the management of agricultural lands is better understood (Thorburn and Wilkinson, 2013).

In this chapter, we examine the potential for management interventions to reduce pollutant losses from agricultural lands in Great Barrier Reef catchments, the extent to which these interventions may meet water quality targets and the costs and benefits associated with these measures. The interventions considered not only focus on those practiced as the field scale (i.e. the source control phase of the 'pollutant treatment train' concept), they include the role of natural and constructed wetlands in remediation of pollutants after they enter the broader landscape (i.e. treatment phase of the 'pollutant treatment train' concept). We focus on information gained since the time of the last Consensus Statement (2008). We also consider a wider range of agricultural activities (e.g. grains and cotton cropping) than in the Scientific Consensus Statement 2008.

The specific conclusion of the Scientific Consensus Statement 2008 relevant to this chapter was: *Current management interventions are not effectively solving the water quality problem in the Great Barrier Reef*.

Synthesis process

Chapter 5 of the 2013 Scientific Consensus Statement reviews and consolidates current knowledge about the effectiveness of management practices for improving water quality and on their economic benefits or tradeoffs. This knowledge encompasses practices both in agriculture and grazing practices as well the benefits of both natural and constructed wetlands in improving water quality. The overarching and central question to this chapter is: *What are the water quality and economic benefits of particular management practices?*

More specifically, the chapter addresses the following questions:

- What are the potential actions that can be undertaken to improve water quality and how effective are they? What are the different management practices we can take to reduce risk? How effective are they?
- What are the water quality and economic benefits from improving management systems and practices?
- When and where are these best implemented spatially to get maximum water quality benefit at minimum cost?
- How effective are wetlands (both natural and constructed) and riparian areas in reducing pollutants to the Great Barrier Reef Marine Park?

A working group (Table 1) was created to consolidate and review the evidence on management practice effectiveness. Its members' expertise spanned a range of fields including the biophysical, agricultural, social and economic, and freshwater ecosystem sciences relevant to the Great Barrier Reef. Evidence was sought from peer-reviewed literature, both scientific journals and publicly available technical reports. The working group consisted of two teams; a writing team and a review team. Discussions were held in meetings and drafts were circulated to the group at a number of stages.

Table 1. Details of the working group associated with this chapter.

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Previous Consensus Statement findings

The broad principles for effective management of sediment, nutrient and pesticide generation are generally known, and this was acknowledged in the 2008 Scientific Consensus Statement. Many of these principles were being drawn upon in the development of Water Quality Improvement Plans that were moving toward finalisation around the release of the Scientific Consensus Statement 2008 (e.g. Dight *et al*., 2009; Drewry *et al*., 2008; Kroon, 2008). Development of Water Quality Improvement Plans was an important process spurring research on the effectiveness of management practices in improving quality of water leaving agricultural lands. Other benefits came from development of Water Quality Improvement Plans. One from the Mackay Whitsunday Plan (Drewry *et al*., 2008) was categorising agricultural management practices to facilitate communication about the different levels and standards of management practice within an industry (Higham *et al*., 2008). Management was summarised by four broad classes, A, B, C and D; equivalent to Class D being 'Dated Practices', Class C 'Common Practices', Class B 'Best Practice', and Class A 'Aspirational Practices'. This ABCD framework subsequently became the basis for prioritising grants to farmers to improve management practice (Thorburn *et al*., 2011d) and for evaluating progress towards government water quality targets (Carroll *et al*., 2012).

Prior to development of Water Quality Improvement Plans, much water quality research had focussed on determining linkages between regional water quality problems and agricultural activities (Bramley and Roth, 2002; McKergow *et al*., 2005a,b; Mitchell *et al*., 2009; Packett *et al*., 2005; Rohde *et al*., 2006; Thorburn *et al*., 2003a) or characterising water quality 'signatures' from different land uses (e.g. Hunter and Walton 2008; Stewart *et al*., 2006; Thorburn *et al*., 2011a; Stork *et al*., 2003; Stork and Lyons, 2012). However, there were some exceptions. Prior to development of Water Quality Improvement Plans, there was research in Great Barrier Reef catchments on managing erosion in sugarcane (Prove *et al*., 1995), grains (Carroll *et al*., 1997), grazing (McIvor *et al*., 1995b; Scanlan *et al*., 1996) and cotton (Carroll *et al*., 1995; Silburn and Glanville, 2002) cropping, and managing pesticide losses from cotton (Silburn *et al*., 2002). Notwithstanding this progress, in 2008 there were still considerable gaps in the knowledge of management of sediment, and particularly, nutrient and pesticide generation in agricultural lands. A re-cap of the broad state of knowledge about the effectiveness of management practices for improving water quality expressed in the Scientific Consensus Statement 2008 follows.

By 2008, it was well established that sediment discharge in the Great Barrier Reef originated mostly from grazing lands of the Burdekin, Fitzroy and Burnett Mary regions (Bartley *et al*., 2007; Brodie *et al*., 2003; McKergow *et al*., 2005a,b). There are also contributions from horticultural crops, such as bananas, and possibly sugarcane, but these are minor contributions compared to grazing. Hillslope, streambank and gully erosion dominate the sediment delivery processes. In sugarcane, trash blanketing, which was introduced to improve logistics of cane harvesting in wet conditions (Wood, 1991), had the incidental benefit of reducing soil erosion (Prove *et al*., 1995). In grains (Carroll *et al*., 1997) and cotton (Carroll *et al*., 1995; Silburn and Glanville, 2002) cropping, crop residues retention, reduced tillage and controlled traffic were also shown to reduce erosion. Sediment management principles in grazing lands relate to applying appropriate utilisation rates of vegetation through better management of stocking rates, wet season spelling to improve pasture condition, forage budgeting to ensure cover levels are adequate from year to year and preventing selective overgrazing of preferred areas in the landscape (MacLeod and McIvor, 2006).

In addition, it was known that significant nutrient loads originated from grazing lands in particulate form, sourced from soil erosion (McKergow *et al*. 2005a,b). Thus, these loads can be managed through soil erosion control. Importantly, there was little direct quantification of the water quality outcomes of different management practices in grazing and cropping lands.

Dissolved nutrients in the Great Barrier Reef catchments predominantly come from sugarcane, cropping and horticulture (Brodie *et al*., 2003). These are largely associated with fertiliser application in these land uses (Bramley and Roth, 2002; Mitchell *et al*., 2009; Thorburn *et al*., 2003a). New ideas were emerging on nutrient management in sugarcane (Wood *et al*., 2003; Thorburn *et al*., 2003c, 2007, 2011b; Park *et al*., 2010; Schroeder *et al*., 2005, 2010) and bananas (Armour *et al*., 2009, 2013) suggesting there was substantial potential for reducing application rates of nitrogen. However, there was little direct quantification of the water quality outcomes of different nutrient management practices in cropping lands of Great Barrier Reef catchments. Most work had focussed on leaching of nitrogen below the root zone (Armour *et al*., 2013; Stewart *et al*., 2006; Thorburn *et al*., 2005) in response to the realisation that nitrate was accumulating in soils and groundwaters of some cropped lands (Rasiah *et al*., 2003; Thorburn *et al*., 2003a). Knowledge of managing nutrient loads in runoff was generally derived from modelling studies (e.g. Armour *et al*., 2009).

Herbicides in the Great Barrier Reef are mostly derived from applications in sugarcane production (Rohde *et al*. 2006a), with some contributions in the Fitzroy catchment from grains cropping (Packett *et al*., 2005) and woody weed control in grazing lands. Herbicide management was focused on better and more effective delivery techniques that reduce losses and on integrated pest management programs focused on reducing use. However, as with nutrient management there was little direct quantification of the water quality outcomes of different pesticide management practices in grazing and cropping lands, an exception being research in cotton cropping (Silburn *et al*., 2002).

Riparian and wetland areas were recognised as potentially playing an important role in trapping pollutants leaving agricultural lands. The functions and capabilities of riparian and wetland areas may differ depending on their position in the landscape (Hunter and Hairsine, 2002). Where hillslopes drain directly into streams without the presence of a floodplain, the riparian zone will act to reduce sediment loads and associated contaminants carried by overland flow (Hunter *et al*., 2006). In smaller, frequently ephemeral, streams the emphasis is on filtering of overland flow, while in larger streams riparian vegetation has a major role in stabilising stream banks (Hunter *et al*., 2006).

The socio-economic aspects of water quality improvement were also starting to be considered by 2008. Economic barriers to management practice change include the net private costs of changing practices, and the capital costs of upgrading infrastructure and machinery. Social barriers also exist, such as attitudes towards particular practices, skills required and attitudes towards risk (Cary *et al*., 2002; Pannell *et al*. 2006). Rates of adoption vary widely. Some improved farming practices, such as green cane trash blanketing in the sugar industry, have achieved widespread adoption within a decade, while other practice changes are slow to occur. In sugarcane for example, there appeared to be clear profitability and environmental ‘win-wins’ in adopting industry-backed nutrient management best management practices, (e.g. Roebeling, 2006; Roebeling *et al*., 2009) yet regional average nitrogen application rates exceed these levels in the majority of sugarcane producing regions (Thorburn and Wilkinson, 2013).

Scientific consensus 2013 – management practice effectiveness

This section is divided into six sub-sections. The first three deal with managing losses of those pollutants, i.e. sediments, nutrients and pesticides, from farm fields. The fourth reviews the extent to which management interventions at the field-scale may meet water quality targets. Pollutants will be lost from even well managed fields, and so the fifth sub-section considers the role of wetlands in reducing pollutant concentrations and/or loads in the broader landscape. The final sub-section considers the socio-economic dimensions of managing pollutants on agricultural lands.

Sediments

General principles

Management practices reduce erosion of fine sediment by:

1. Maintaining the groundcover and forage biomass of vegetation, including through droughts, to protect the soil surface from rain splash and rill erosion (Freebairn *et al*. 1996; Rosewell, 1993; Silburn *et al*. 2011) as well as gully incision (Prosser and Slade, 1994). Cover and biomass also reduce overland runoff in smaller events (McIvor *et al*., 1995b) through increasing evapotranspiration and slowing runoff, thereby reducing sediment transport capacity. In grazing land, ground cover and biomass are managed by setting stocking rates based on consumption of 10–30 per cent of available forage. In the long term, such management results in better land condition (capacity to produce forage) than if grazing pressure is heavier. In cropped lands, groundcover is managed by reduced tillage, and avoiding bare fallows by retaining crop residues after harvest, selecting crop that provide good ground cover where possible, maintaining a high crop frequency and/or planting fallow crops, depending on the cropping system.
2. Improving soil condition to reduce runoff. In grazing land this involves reducing forage utilisation (i.e. grazing pressure) and retaining vegetation diversity (Ash *et al*., 2011; Dawes-Gromadzki, 2005; Roth, 2004). Some tree cover can also enhance runoff infiltration (Ellis *et al*., 2006; Leguédois *et al*., 2008). In cropping land improving soil condition involves reducing tillage and using controlled traffic farming to prevent soil compaction in the crop row (Murphy *et al*., 2013) and avoiding bare fallow periods (Freebairn *et al*., 1996).
3. Redistributing the pressure of agricultural activities away from areas vulnerable to erosion. In grazing lands this involves fencing and additional water points to separate soil types with preferred pasture (Chilcott *et al*., 2003; Gordon and Nelson, 2007), and riparian or frontage country (Hunt *et al*., 2007). It also involves remediating rilled and scalded areas (Bartley *et al*., 2010a), and gullied areas (Thorburn and Wilkinson, 2013). In cropping areas this involves avoiding use of steep land, using contour banks to retain runoff (Murphy *et al*., 2013), and using buffer strips in riparian areas and drainage lines (McKergow *et al*., 2004).

Grazed lands

Processes of sediment loss

Approximately three-quarters of the fine sediment, approximately equivalent to silt and clay sized particles (less than 63 micrometres), delivered to the Great Barrier Reef is derived from grazing lands (Chapter 4; Thorburn and Wilkinson, 2013). Sediment tracing indicates that subsoil loss from hillslope rilling and scalding, gully and streambank erosion, dominates grazing land fine sediment yield where gullies are present (Hughes *et al*., 2009; Wilkinson *et al*., 2013). Similarly, recent field monitoring found that more than 90 per cent of sediment lost from hillslopes to streams was derived from less than 10 per cent of the catchment area (Bartley *et al*., 2010b). Gully erosion is extensive, with approximately 80,000 kilometres of gully network in Great Barrier Reef catchments (Thorburn and Wilkinson, 2013). Limited studies in Great Barrier Reef catchments indicate that grazing and cropping land under current practices has at least twice the runoff of non-grazed native vegetation (Hawdon *et al*., 2008; Thornton *et al*. 2007), increasing sediment mobilisation on hillslopes and in the drainage network.

Principles of grazing management

The current understanding of grazing practices and sediment loss in Great Barrier Reef catchments is that the most important influence of grazing practices is through managing grazing pressure at levels that

- maintain vegetation cover to protect the soil surface from surface erosion (McIvor *et al*., 1995b; Rosewell, 1993) and drainage incision by rills or gullies (Prosser and Slade, 1994; Valentin *et al*., 2005)
- reduce surface runoff by maintaining pasture basal area (Roth, 2004), maintaining evapotranspiration, and soil infiltration capacity (McIvor *et al*., 1995b; Orr and Carling, 2006; Thorburn and Wilkinson, 2013).

To reduce runoff, grazing practices should improve land condition including soil infiltration capacity and pasture productivity. This requires higher levels of cover, lower levels of forage utilisation and more diverse pasture species composition than are required simply for increasing cover levels to locally control sheetwash erosion (Roth, 2004).

Grazing practices which minimise sediment loss are:

1. Forage utilisation less than 25–30 per cent of annual maximum biomass (lower in drier areas, or if calculated as a proportion of biomass production) is required to sustain pasture productivity and land condition (Ash *et al*., 2011; Landsberg *et al*., 1998; McIvor *et al*., 1995; Orr *et al*., 2010;), and thus to minimise erosion. It is important to set grazing pressure based on available forage rather than visible erosion because degradation can be well advanced before decline in pasture productivity is detected (MacLeod *et al*., 2004). Lower forage utilisation enables the re-introduction of fire to control woody weeds and improve forage quality in native pastures (Landsberg *et al*., 1998).
2. To address the spatial concentration of erosion in rilled, gullied and riparian areas, grazing pressure should be redistributed away from these areas (Bartley *et al*., 2010b; Hunt *et al*., 2007; Thorburn and Wilkinson, 2013).
3. Pasture condition can be enhanced by pasture resting during growth or seeding phases (Ash *et al*., 2011; Bartley *et al*., 2010a).
4. Retaining some tree cover is highly efficient at trapping upslope runoff and sediment, even when interspersed with open ground (Ellis *et al*., 2006; Leguédois *et al*., 2008). Tree cover is especially important in riparian areas where it stabilises streambanks (Simon and Colliston, 2002).
5. Sowing exotic pasture species is not necessarily as effective at reducing sediment loss as reducing forage utilisation. In particular, grasses with low basal area such as Buffel grass (*Cenchrus ciliaris*) provide less protection of the soil surface (Barai *et al*., 2009; Chunale, 2004), and monoculture Buffel pastures tend to have high erosion rates under heavy grazing (Valdez-Zamudio and Guertin, 2000). Buffel grass is more productive in an ideal growing environment where moisture and nutrients are not limiting, however the production benefit may be small in the long term due to nutrient rundown (Landsberg *et al*., 1998).
6. Including some engineering structures in gully remediation can achieve larger magnitude reductions in gully yield than does increasing ground cover alone, and can be targeted to reaches controlling base level (Heede, 1979). A study across larger catchments found that significant reductions in sediment yield were more likely with treatments involving engineering works than vegetation cover improvement alone (Rustomji *et al*., 2008).

Effectiveness of grazing practices

There is evidence from within Great Barrier Reef catchments that reduced forage utilisation and wet-season spelling of C-condition land resulted in a 10 to 15 per cent increase in vegetation cover relative to neighbouring properties (Bartley *et al*. 2010b), and greater than 60 per cent reduction of erosion rates on hillslopes (Bartley *et al*., 2010a). A separate grazing trial found that moderate and rotational stocking rates associated with B land condition have much higher vegetation cover levels than heavy stocking rates (C/D-practice) associated with C/D land condition (Orr and O'Reagain, 2011).

Some gully remediation and streambank protection projects have been trialled in Great Barrier Reef catchments. While they potentially offer a highly targeted approach to reducing sediment loads, the cost-effectiveness of and sediment load reductions from these works have not yet been tested quantitatively in Great Barrier Reef catchments. However, international studies consistently demonstrate that comprehensive programs of gully remediation and riparian protection in grazing lands deliver large reductions in erosion rates (Thorburn and Wilkinson, 2013). Considering the large contribution to Great Barrier Reef sediment loads from gully networks and streambanks (see Chapter 2), it will be difficult to achieve load reduction targets without substantively addressing these erosion processes.

The magnitude of practice change required to achieve the Great Barrier Reef sediment load reduction target (20 per cent) has been estimated using a conceptual framework including hillslope, gully and streambank erosion processes, as requiring a 20 per cent increase in ground cover over approximately half of the grazed area in Great Barrier Reef catchments, particularly areas with lower levels of ground cover (Thorburn and Wilkinson, 2013). Further testing is required to determine whether a 20 per cent increase in cover can be achieved within Great Barrier Reef grazing lands, as it is approximately double that measured to date in commercial grazing trials (e.g., Bartley *et al*., 2010b).

There are numerous examples globally of changes in land management leading to sediment yield reductions at small catchment scale. However, there are few examples of sediment load reduction through agricultural management across contiguous areas of comparable size to the Great Barrier Reef catchments. The Loess Plateau in China is a notable exception, where catchment revegetation and engineering works resulted in large reductions in loads and concentrations over an area comparable to the Great Barrier Reef catchments (Rustomji *et al*., 2008).

Time-scales of response

Significant reductions in erosion rates in response to reduced forage utilisation and increases in mean vegetation cover can be detected within years for hillslopes with existing C-condition grass cover (Bartley *et al*. 2010a). Reductions in surface runoff can be achieved in as little as several years, if native perennial vegetation is intact (although overgrazed), forage utilisation is minimal, and at least average rainfall occurs (Connolly *et al*., 1997), but requires between one and several decades in degraded land where changes in vegetation species composition is required and some grazing continues (Bartley *et al*., 2010a; Wilcox *et al*., 2008). The rates of rill, gully and riverbank erosion, and catchment sediment yield will also respond over longer time-frames (Bartley *et al*., 2010b), since these erosion processes require improvements in soil properties and runoff, or woody vegetation cover. Thus, the limited number of catchment-scale studies of improved grazing practices in the Great Barrier Reef has not yet resulted in statistically well-quantified estimates of sediment yield reductions at catchment scale.

Cropped lands

Cropping is generally associated with higher rates of sediment loss per hectare than grazing, and this has been seen in the Great Barrier Reef (Carroll *et al*., 1997; Hughes *et al*., 2009; Murphy *et al*., 2013; Prove *et al*., 1995). In Great Barrier Reef catchments management systems that reduce or eliminate tillage and maximise soil cover (through crop rotations and the retention of crop residues) reduce soil loss in a wide variety of cropping systems (Carroll *et al*., 1997; Prove *et al*., 1995; Sallaway *et al*., 1990; Thorburn, 1992).

As well as these practices, controlled traffic is effective in reducing runoff and soil loss in sugarcane farming (Masters *et al*., 2013) and row cropping (Silburn *et al*., 2013). Contour embankments are essential for reducing soil loss from cropping lands in large storms (Murphy *et al*., 2013). In addition, fallows with low surface cover represent a major erosion hazard, and greater than 30 per cent soil cover should be maintained during fallows through retention of crop residues and/or planting cover crops to manage erosion. Clear examples of the efficacy of these practices come from the Fitzroy basin. In the central highlands area, when zero tillage resulted in high soil cover, erosion rates were 75 per cent lower than from traditional cropping practices (Carroll *et al*., 1997). Sediment deposition rates indicate that maintaining ground cover is also a factor in reducing sediment yields, with sediment yields under reduced tillage and increased soil cover being less than half the sediment yield of traditional practices (Hughes *et al*., 2009).

Areas for targeted management

From the above sections it is clear that all erosion processes can be effectively managed at property scale. Targeting practice improvement first to areas of the Great Barrier Reef catchments which have higher sediment contribution rate can be expected to reduce sediment loads more than non-targeted practice improvements (Lu *et al*., 2004). The regional-scale gradients in sediment contribution rate result from variations in erosion rates across grazing and cropping lands that are independent of management practices, and associated with gradients in the environmental drivers of erosion, including rainfall and topography. Other factors causing locally higher sediment contribution include occurrence of cropping land use as described above, and fine-textured soils such as basalt derived Vertosols in the Fitzroy catchment (Smith *et al*., 2008; Packett *et al*., 2009). Gullied areas inherently deliver several times more sediment per hectare than non-gullied areas, and streambanks are also areas of locally intense erosion which can provide larger reductions in sediment loads per hectare of treatment. Hotspot areas of hillslope erosion will also be priorities for efficiently reducing sediment loads where they are well-connected to river outlets. The rates of contribution to Great Barrier Reef sediment loads are lower for areas upstream of large impoundments, which trap the majority of fine sediment delivered from upstream areas (Lewis *et al*., 2013).

Further research

- The water quality effectiveness and costs of specific grazing practices need to be verified using local investigations within priority, highly-eroding areas of Great Barrier Reef grazing lands. Practices which are priorities for further study include riparian grazing management, reducing or removing grazing pressure from gullied areas, and remediation of gullies and other erosion features using physical works.
- There have been many more studies of grazing land degradation than of land condition improvement, and yet the latter is the objective for reducing sediment losses, as well as enhancing pasture productivity. More local studies are required of the processes, time frames and water quality effectiveness of recovery in land condition following improved grazing practices, including soil and vegetation properties, water and pollutant fluxes. Priority areas for such studies would be areas of high erosion rates, low vegetation cover and biomass, and fine textured and sodic soils.
- Systematic methods need to be developed for ensuring that tools for applying forage budgeting, forage condition assessment and climate forecasts are available across the grazing industry. These methods then need to be applied to determine stocking rates. Anecdotal information indicates that most graziers do not currently use quantitative methods for setting stocking rates.
- Pasture improvement is a common practice but there is very little evidence about the soil loss benefits relative to reduced utilisation of native pastures, or information on the soil loss under different pasture species being used.

Nutrients

Many nutrients are critical for plant growth, and hence crop and pasture production. However, only two, nitrogen and phosphorus, have been identified to date as having major ecological impacts on Great Barrier Reef ecosystems. Both are influenced by agricultural management practices. Therefore, this section focuses on nitrogen and phosphorus.

Nutrients exist in particulate and dissolved form. There are similarities, but also differences in the management outcomes for these two forms, which will be considered.

General principles of nutrient management

1. Inputs of nitrogen into cropping systems can have several fates; namely uptake by crops, storage in the soils (in both mineral and organic forms) and losses to the environment. At steady-state, soil storage is not significant (Janssen and De Willigen, 2006) so the difference between nitrogen inputs and crop nitrogen uptake, i.e. the nitrogen surplus, is an indicator of environmental losses over the long-term (Buczko *et al*., 2010; Sieling and Kage, 2006). Surplus nitrogen can be lost to the environment through various pathways, namely leaching, runoff, soil erosion and atmospheric losses (denitrification and volatilisation). The partitioning of these losses will depend on the soil, climate and management (Thorburn and Wilkinson, 2013). Thus linking nitrogen surpluses to a particular loss pathway, e.g. runoff or deep drainage, will be problematic (Buczko and Kuchenbuch, 2010). However, as a first principle, we expect changes in nitrogen surpluses to be reflected in relative changes in nitrogen losses in all pathways. Indeed, nitrogen surpluses have been correlated with anthropogenic dissolved inorganic nitrogen loads to the Great Barrier Reef at the regional level (Thorburn and Wilkinson, 2013), and with total nitrogen lost in runoff at the field scale (Webster *et al*., 2012).
2. The general principles for phosphorus are similar to those for nitrogen. Phosphorus inputs into cropping systems can be taken up by crops, stored in the soils and lost to the environment. And, as for nitrogen, at steady-state, soil storage is not significant (Janssen and De Willigen, 2006) so the difference between phosphorus inputs and crop phosphorus uptake, i.e. the phosphorus surplus, may be an indicator of environmental losses over the long-term. However, because phosphorus is bound to the soil the soil acts as a finite phosphorus sink and the assumption of steady-state soil phosphorus is less applicable than it is to nitrogen. For example, decades of over-application of phosphorus fertiliser to sugarcane crops has resulted in positive phosphorus surpluses and substantial increases in soil phosphorus concentrations in some regions (Bloesch *et al*., 1997). High soil phosphorus concentrations themselves are a fundamental driver of phosphorus losses to the environment (Moody, 2011).
3. Managing loss of particulate nutrients is achieved through managing loss of fine sediments (discussed in the above sub-section). The majority of particulate nutrients lost from Great Barrier Reef catchments come from grazed lands where pastures are not fertilised (Chapter 4; Thorburn and Wilkinson, 2013). Practices for managing particulate nutrient losses from grazing lands are thus addressed by those for managing fine sediment loss and will not be discussed further in this section.

Particulate and dissolved forms

The majority of dissolved nutrients are soluble forms of nitrogen which come from cropping lands (Chapter 4; Thorburn and Wilkinson, 2013), although high concentrations of soluble phosphorus have been found in groundwaters in two cropped catchments in the Wet Tropics (Rasiah *et al*., 2011). Application of fertiliser plays an important role in generating these dissolved nutrient losses, and so nutrient management is a major factor in mitigating losses of soluble nutrients to the environment.

Particulate nutrients are contained in fine sediments, so they are mainly lost through erosion. For example, more than 60 per cent of total nitrogen and 80 per cent of total phosphorus is delivered to the Great Barrier Reef attached to fine sediment, and grazing lands contribute approximately three quarters of the total fine sediment to the Great Barrier Reef (Chapter 4). Another factor in the generation of particulate nutrients is the concentration of the nutrient in the sediments. Phosphorus is sorbed to the soil, so phosphorus concentrations increase as a result of application of phosphorus fertilisers (Moody, 2011). Nitrogen is contained in soil organic matter, a constituent of fine sediments. Like phosphorus, application of nitrogen fertilisers increases total nitrogen concentrations in soils (Cong *et al*. 2012), although the process is complicated by net sequestration or decomposition of soil organic matter. Thus, management of nutrient application can potentially affect generation of particulate nutrients. The principles for managing nutrients to reduce their losses from Great Barrier Reef catchments apply to dissolved and particulate nutrients in fertilised crops and pastures. However, although the concepts have generally been developed and tested in the context of managing losses of dissolved nutrients from crops because of the widespread use of fertiliser in crop production and common adoption of erosion control measures in cropped lands.

Nutrient surpluses

Nitrogen

Nitrogen surpluses have been determined in experiments on sugarcane (Thorburn *et al*., 2011b) and bananas (Prove *et al*., 1997) in Great Barrier Reef catchments. These field level surpluses are significantly (r = 0.83, phosphorus less than 0.001) correlated with nitrogen fertiliser applications (Thorburn and Wilkinson, 2013). This occurs because yield and nitrogen off-take in the crops in these experiments were not responsive to nitrogen fertiliser application at the rates applied (Thorburn *et al*., 2011b; Armour *et al*., 2013; Prove *et al*., 1997). Where nitrogen limits crop growth, we would expect surpluses to be relatively small and possibly less dependent on nitrogen fertiliser inputs. It is relevant to consider the minimum nitrogen surplus needed to maintain maximum yields. It has been suggested that the minimum nitrogen surplus is approximately 50 kilograms per hectare per crop for sugarcane (Thorburn *et al*., 2011a) and other intensively managed crops in Great Barrier Reef catchments (Thorburn and Wilkinson, 2013).

At a broader scale, average nitrogen surpluses have been estimated from nitrogen fertiliser applications and estimates of crop nitrogen off-take for all major crops in Great Barrier Reef catchments (Thorburn and Wilkinson, 2013). For sugarcane, average nitrogen surpluses vary between regions, from 98 kilograms per hectare per year in the Wet Tropics, Herbert and Burnett Mary regions, to approximately 135 kilograms per hectare per year in the Burdekin and Mackay Whitsunday regions. Nitrogen surpluses also vary between different crops, from 10 kilograms per hectare per year in cereal farming systems to 250 kilograms per hectare per year in banana production. Surpluses for irrigated cotton, horticultural small crops and trees crops are more consistent, ranging from 94 to 142 kilograms per hectare per year, with rain fed cotton having a surplus of 46 kilograms per hectare per year. Yield and nitrogen fertiliser applications vary through time, so these estimates provide only a guide.

Fertiliser is not the only source of nutrient inputs to crops, and it is necessary to consider the effects of these other sources on nutrient losses. For nitrogen, biological nitrogen fixation in legumes can represent a significant nitrogen input to cropping systems. In sugarcane production, legumes have been widely promoted as a fallow crop. Certain legumes, especially if grown as ley crops rather than grain crops, can contain substantial amounts of nitrogen (e.g. 300 kilograms per hectare; Schroeder *et al*., 2005). These nitrogen inputs increase the nitrogen surplus in and hence potential nitrogen losses from subsequent crops (Bell *et al*., 2010; Park *et al*., 2010). In some circumstances nitrogen inputs from legumes can be taken up by subsequent sugarcane crops and so lessen the amount of fertiliser nitrogen needed by these crops. The effect is greatest in sugarcane plant crops, but may last for one (Bell *et al*., 2010) or more (Park *et al*., 2010) ratoon crops. Thus balancing nitrogen inputs from legumes with lower nitrogen fertiliser applications is one strategy for minimising nitrogen surpluses in, and nitrogen losses from sugarcane crops following legumes. Nitrogen fertiliser recommendations for sugarcane recognise the contribution of nitrogen from fallow legumes, advising an equivalent reduction in nitrogen applied to plant crops (Schroeder *et al*., 2005). However, legumes can commonly contain more nitrogen than that normally applied to plant crops. In these situations, reduced nitrogen fertiliser cannot ‘balance’ legume nitrogen inputs and nitrogen recommendations will be conservative. As well, there are situations where nitrogen from legumes appears to be lost to the environment before it can be taken up by crops (Bell *et al*., 2010) creating the uncertainty about productivity of crops following legumes if nitrogen fertiliser applications are reduced. Thus, it is likely that nitrogen surpluses and nitrogen losses will be exacerbated in sugarcane crops following a legume crop (Thorburn and Wilkinson, 2013). However, there are no empirical data to test this proposition.

While at an individual field scale, legume nitrogen inputs may cause a 'nitrogen loss hot spot', the effect at a whole catchment scale is uncertain. The uncertainty arises because not all fallow legumes contain large amounts of nitrogen (depending on the species or if grain is harvested), not all sugarcane farmers grow fallow legumes, and only a minority of fields (e.g. approximately 20 per cent) are in fallow at any time. Thus, at a whole catchment scale the net input of legume-sourced nitrogen in sugarcane systems may only be a few kilograms nitrogen per hectare (Thorburn and Wilkinson, 2013). Consequently, the effect of fallow legumes on dissolved inorganic nitrogen losses at a catchment scale is not yet understood.

Sugarcane cropping systems are not the only ones in the Great Barrier Reef where legumes are grown: In grains production systems legume grain crops are often grown in rotation with cereals. It can be argued that legume crops in these production systems will only have a small impact on nitrogen surpluses, and hence water quality (Thorburn and Wilkinson, 2013). This is because little or no nitrogen fertiliser is applied to either the legume or cereal grain crops in these systems, since their nitrogen requirements are largely met through legume nitrogen fixation (e.g. Huth *et al*., 2010). However, substantial amounts of dissolved inorganic nitrogen were measured in runoff from a sorghum crop in central Queensland (Murphy *et al*, 2013). The reasons for the high loss of dissolved inorganic nitrogen from this site, and how representative these results are of the general situation in grains production areas are unclear.

Mill mud is another potential source of nutrients in sugarcane production areas. It is a by-product of sugarcane milling that mainly consists of ground sugarcane stalks and leaves, soil, and lime added during clarification. Nitrogen concentrations in mill mud are low (e.g. approximately 1.5 per cent on a dry weight basis: Barry *et al*. 1998). However, while the sugarcane and soil in mill mud are 'collected' over the whole harvest region, mill mud is traditionally disposed of on only a small proportion (e.g. 5 per cent) of the harvested area (Barry *et al*. 1998), so the constituents of mill mud are effectively concentrated in that small area. This creates a situation where approximately 400 kilograms per hectare of nitrogen may commonly be applied in mill mud, considerably greater than the recommended application rates for a sugarcane crop (Wood *et al*. 2003; Schroeder *et al*. 2005). Unless fertiliser nitrogen is reduced by an equivalent amount, which cannot be achieved in a single sugarcane crop, mill mud managed in this manner potentially exacerbates losses of nitrogen to the environment. The recommendations to farmers on how to adjust their nitrogen fertiliser applications following mill mud are poorly developed, making it more likely that mill mud applications increase nitrogen losses. However, there is little direct information on the impact of mill mud on nutrient losses, or its management. A simulation study (Thorburn *et al*., 2008) illustrates the potential problem; reductions in nitrogen fertiliser after application of mill mud can only 'offset' half the nitrogen applied in mill mud. The problem posed by disposing of mill mud on farmers' fields is made more manageable if it is disposed of over greater areas, so reducing the loadings of nitrogen on an area basis. Systems have been developed in the Mackay region to do this (Markley and Refalo, 2011).

Phosphorus

In Great Barrier Reef catchments, phosphorus surpluses are generally lower than nitrogen surpluses. In horticultural crops, phosphorus surpluses are 40-70 kilograms per hectare per crop in many crops (Moody and Aitken, 1996), although higher surpluses (i.e. greater than100 kilograms per hectare per crop) have been found in specific experiments in bananas (Prove *et al*., 1997). Surpluses tend to be lower in some tree crops, e.g. less than 10 kilograms per hectare per crop in mango and coffee (Moody and Aitken, 1996) and sugarcane (Bloesch *et al*., 1997; Prove *et al*., 1997). Although these phosphorus surpluses in sugarcane are small, they have still resulted in a substantial build-up of phosphorus in soils over decades in some regions (Bloesch *et al*., 1997). This build-up may mean that sugarcane crop yields do not respond to additions of phosphorus fertiliser and little phosphorus needs to be applied to sugarcane crops. Low phosphorus applications will reduce phosphorus surpluses and potential losses to the environment, albeit over a long time period because of the small removal of phosphorus in harvested cane. Soil testing is an effective way of determining phosphorus needs in sugarcane, and is a recommended practice (Wood *et al*., 2003).

Mill mud contains phosphorus, and approximately 250 kilograms per hectare of phosphorus may be applied to sugarcane crops in mill mud (Barry *et al*., 1998). As with nitrogen, this amount of phosphorus is considerably above recommended application rates for a sugarcane crop (Wood *et al*., 2003) and cannot be offset by reduced phosphorus fertiliser in a single crop. As with nitrogen, disposing of mill mud over greater areas, so reducing the loadings of phosphorus on an area basis, will reduce the over application of phosphorus from use of mill mud.

Fertiliser recommendations and nutrient surpluses

Most industry fertiliser recommendations aim to ensure that crop yields are not constrained by nutrient supply, so they are developed to supply sufficient nutrients to achieve potential crop yields. Where the yields actually achieved by farmers are below potential, the excess nutrient applied inflates the nutrient surplus. This situation is exemplified in the sugarcane industry, which has the most clearly developed nutrient management recommendations (Wood *et al*., 2003; Schroeder *et al*., 2010) of crops in the Great Barrier Reef. Sugarcane nutrient recommendations aim to supply adequate nutrients to meet the “district yield potential”, which is defined as 120 per cent of the “estimated highest average annual district yield”. On a mill region basis, the estimated highest average annual district yield is achieved in approximately 10 per cent of years, and the district yield potential is seldom or never achieved (Schroeder *et al*., 2010). Thus at this scale, sugarcane crops managed according to industry recommendations are being over-fertilised by more than 20 per cent, a situation consistent with the high nutrient surpluses described above.

At a district or single field scale, the potential yields are achieved more commonly, although still in a minority of cases (Schroeder *et al*., 2010). Thus, following these industry recommendations will result in the majority of fields being over-fertiliser to ensure the minority are not nutrient limited. Sugarcane yields and their response to fertiliser applications can vary considerably from year-to-year (e.g. Thorburn *et al*., 2003b, 2011b), as well as from field-to-field, prompting management systems to target the maximum possible yields. However, the average over-application of fertiliser that results from this approach can lead to accumulation of nutrients in the soil supporting sugarcane. Examples include the build up of phosphorus (Bloesch *et al*., 1997) and high concentrations of mineral nitrogen in soils (e.g., more than or equal to 100 kilograms per hectare: Thorburn *et al*. 2011b), even in soils of the wet tropics in years of high rainfall (Thorburn and Goodson, 2005). Accumulation of nutrients in the soil makes sugarcane crops less responsive to fertiliser in the short-term, i.e. over one or two crops for nitrogen (Thorburn *et al*., 2003b, 2011b) or longer for phosphorus (Bloesch *et al*., 1997). The actual risk of nutrient supply limiting crop yields, and how that risk varies between the short- (e.g. seasons) and long-term (e.g. decades) is poorly understood.

Alternative management systems based on production in an individual field, rather than regional targets, have been identified for nitrogen in sugarcane (Thorburn *et al*., 2011b). These management systems result in lower nitrogen-surplus and lower dissolved inorganic nitrogen losses (Webster *et al*., 2012). While data are not as readily available for other crops, a similar situation is likely to be occurring in bananas (Armour *et al*., 2013) and cotton (Rochester, 2010), but not grains crops (Thorburn and Wilkinson, 2013).

Effects of nutrient applications on losses

Nitrogen

Despite the clear relationship between fertiliser surpluses and losses of nutrients to the environment, there are few studies quantifying this relationship in Great Barrier Reef cropped lands. However, the studies undertaken clearly show that reducing nitrogen fertiliser applications reduce nitrogen losses in both runoff and deep drainage. For example, reducing nitrogen fertiliser applications to sugarcane crops by approximately 47 per cent reduced dissolved inorganic nitrogen in runoff from those crops by approximately 60 per cent in the Wet Tropics (Webster *et al*, 2012) and approximately 50 per cent in Mackay (Agnew *et al*., 2011). In the Wet Tropics study, the reduction in total nitrogen was approximately 30 per cent. These results are consistent with studies in more temperate environments in the USA (Bengtson *et al*., 1998). Reducing nitrogen fertiliser applications has likewise reduced nitrogen leached from both sugarcane (Webster *et al*., 2012) and bananas (Armour *et al*., 2013). As expected, reducing nitrogen inputs reduces nitrogen surpluses, so that total nitrogen in runoff and nitrogen surpluses are positively correlated (Webster *et al*., 2012).

Based on these principles, and on those underpinning management of fine sediment losses (discussed above), Thorburn and Wilkinson (2013) estimated the improvement in dissolved and total nitrogen discharged from Great Barrier Reef catchments under different management practices. They predicted that universal adoption of industry-backed best practices for nitrogen management (equivalent to B-Class practices; Higham *et al*. 2008) in cropped lands would reduce dissolved inorganic nitrogen loads by approximately 12 per cent. Coupling these practices with increased cover in grazing lands (to reduce fine sediment losses) was predicted to reduce total nitrogen by approximately 15 per cent. Further reductions could be achieved if nitrogen surpluses were further lowered and cover increased.

Phosphorus

Few studies in the Great Barrier Reef characterising pollutant loads in runoff or deep drainage from cropped lands consider phosphorus. In one that did (Prove *et al*. 1997), there was an interaction between cultivation and phosphorus fertiliser inputs: phosphorus losses were lower in crops with no phosphorus applied compared with 31 kilograms of phosphorus per hectare applied. Losses were also lower with conventional cultivation than with minimum tillage in the fertilised crops. Other studies considering phosphorus focus on practices to control runoff and erosion (Agnew *et al*., 2011; Masters *et al*., 2008; Murphy *et al*., 2013) rather than the relationship between the phosphorus fertiliser applications and phosphorus losses. Because of the direct linkage between extractable soil phosphorus and dissolved phosphorus concentration in the soil solution (Moody, 2011) and runoff (Burkitt *et al*., 2010), the high extractable phosphorus status of many sugarcane soils (Bloesch *et al*., 1997) is likely to result in concentrations of dissolved phosphorus in runoff being higher than background concentrations in unfertilised soils. Enhanced extractable soil phosphorus status has also been shown to be associated with dissolved phosphorus enrichment of groundwater in the wet tropics (Rasiah *et al*., 2011).

Other in-field interventions to reduce nutrient losses

Retention of nitrogen in the soil

Dissolved inorganic nitrogen concentrations in runoff water and deep drainage are greatest immediately after nitrogen fertiliser application, then decline through time (Stewart *et al*., 2006; Thorburn *et al*., 2011a; Webster *et al*., 2012; Armour *et al*, 2013). This has led to a focus on timing of nitrogen fertiliser applications in relation to runoff and/or leaching events, promoting applications of nitrogen fertiliser before the start of the 'wet season' (Moody *et al*., 2008; Armour *et al*., 2013), or delaying irrigation after nitrogen applications, to reduce dissolved inorganic nitrogen concentrations in the soil when runoff and/or leaching events occur.

Other practices that may increase retention of nutrients in the soil are:

- splitting nitrogen applications (also seen as a way of reducing dissolved inorganic nitrogen concentrations in the soil when runoff and/or leaching events occur; Thorburn *et al*., 2011c);
- burying nitrogen fertiliser, which reduces nitrogen losses through runoff soon after nitrogen applications (Cowie *et al*., 2012); and
- applying 'enhanced efficiency fertilisers', which either release nitrogen slowly or inhibit the transformation of fertiliser nitrogen to nitrate (Chen *et al*., 2008). The implicit assumption in these recommendations is that these practices will retain more nitrogen in the soil plant system, thereby reducing losses to the environment.

However, the efficacy of these practices for improving water quality in cropped lands of the Great Barrier Reef has not been widely studied. Most studies have been undertaken at a single rate of nitrogen fertiliser, commonly rates at or above those recommended in nutrient best management practices. The crops studied did not reach potential yields, and hence were over fertilised relative to the actual yields achieved. In these situations, crops will have acquired all the nitrogen they require and so there is no potential for these practices to give greater crop nitrogen uptake and better water quality. As nitrogen applications are reduced, the risk that crop yields will be limited by nitrogen availability increases. In these situations, practices that retain more nitrogen in the soil plant system may be effective and reduce the risk of crop nitrogen stress. Testing the efficacy of these practices is needed over a range of nitrogen application rates.

Furthermore, there are limits to nitrogen storage in soils or uptake by crops and the long-term fate of this retained nitrogen has not been considered: once storage and/or uptake stop, the nitrogen will 'bleed' out of the system over a number of years (Thorburn *et al*., 2003b, 2011b). Thus, these practices will not overcome nutrient losses caused by excessive fertiliser applications. There are several examples that support this conclusion. Soil dissolved inorganic nitrogen concentrations in water leached from the root zone of bananas at a site in the wet tropics were impacted equally, or more by the amount of nitrogen fertiliser applied than by the time elapsed since the last application of fertiliser (Armour *et al*., 2013). In a simulation study on sugarcane in Tully (Thorburn *et al*., 2011c), the effect of splitting on cane yields and nitrogen losses was less than nitrogen fertiliser application rates. Similarly, burying nitrogen fertiliser has given higher losses of nitrogen in runoff over a whole sugarcane crops (Prove *et al*., 1997; Webster *et al*., 2012), which contrasts the short term effects observed in rainfall simulator studies (Cowie *et al*., 2012). The whole-crop results potentially came about because nitrogen was lost from the surface-applied nitrogen through volatilisation, but not with buried nitrogen (Prove *et al*., 1997), and the net input of nitrogen to the soil was greater with buried nitrogen (Thorburn and Wilkinson, 2013). This analysis illustrates the 'systems nature' of nutrient management in cropping systems, and the potential for unexpected outcomes as the dominant nitrogen loss pathway changes as a consequence of management interventions.

Reducing runoff and/or sediment loss

Runoff water contains both dissolved and particulate nutrients. Thus, practices that reduce runoff and/or sediment loss will reduce losses of nutrients via this pathway. In Great Barrier Reef catchments, management systems that reduce or eliminate tillage, reduce soil compaction (e.g. controlled traffic), maximise cropping opportunities and soil cover (by crop residues retention) reduce nutrient losses in a wide variety of cropping systems, including grain (Thomas *et al*., 1990), cotton (Silburn and Hunter, 2009) and sugarcane (Agnew *et al*., 2011; Masters *et al*, 2008). In sugarcane however (Masters *et al*., 2008), controlled traffic was less effective at reducing nitrogen in runoff (11 per cent less than from conventional tillage) than phosphorus in runoff (32 per cent), possibly because of the greater proportion of nitrogen in dissolved form. Contour embankments are essential for reducing loss of sediments and associated particulate nutrients from cropping lands in large storms, particularly in rain fed cropping (Murphy *et al*., 2013).

However, for dissolved nutrients, particularly nitrate, reducing losses through runoff may increase losses through another pathway, i.e. leaching or denitrification (Thorburn and Wilkinson, 2013). This situation is exemplified by the discussion above on burying nitrogen fertiliser. Given that nitrate can move from groundwater to streams in the Great Barrier Reef (Rasiah *et al*., 2013) and denitrification produces the potent greenhouse gas nitrous oxide (Thorburn *et al*., 2010), simply changing the pathway by which nutrients are lost to the environment rather than reducing the total loss of nutrients is an undesirable outcome.

Irrigation

Irrigation may increase nutrient losses relative to rain fed crops through several mechanisms: Higher yields of irrigated crops are often associated with larger fertiliser applications, increasing the chance of nutrient losses (Randall *et al*., 2008). Additionally, application of irrigation water itself can exacerbate nutrient loss processes (Kruse *et al*., 1990; Randall *et al*., 2008), especially when applied in large amounts via systems (e.g. furrow irrigation) that have low application efficiencies and hence substantial runoff and/or deep drainage (McHugh *et al*., 2008; Thorburn *et al*., 2011a).

Despite the importance of irrigation in the production of horticultural, sugarcane and cotton crops, there are few studies on the effect of irrigation management on nutrient losses in Great Barrier Reef catchments. In sugarcane, nitrogen losses from irrigated crops in the Burdekin (Thorburn *et al*., 2011a) and Mackay (Agnew *et al*., 2011) regions were not substantially different from those found in rain fed crops of Mackay (Masters *et al*, 2008) or the Wet Tropics (Prove *et al*., 1997; Webster *et al*., 2012). Thus, irrigation itself is not associated with increased nitrogen losses in sugarcane, although there is scope to reduce losses through irrigation management (discussed below). Moving from irrigation systems with low efficiency (i.e. furrow irrigation) to those with high efficiency has been shown to reduce loss of nutrients in Great Barrier Reef catchments. Subsurface drip irrigation reduced losses of nitrogen and phosphorus by an order of magnitude compared with furrow irrigated cotton (McHugh *et al*., 2008). In sugarcane, overhead low pressure irrigation also reduced nitrogen losses compared with furrow irrigation (Attard and Inman-Bamber, 2009).

Management of irrigation within a given application system can also reduce nutrient losses. With drip irrigated cotton, nitrogen and phosphorus losses were generally reduced when lower amounts of water were applied (McHugh *et al*., 2008), although yields were reduced at very low water applications. Also, there may be substantial opportunity for reducing nitrogen losses through irrigation management in furrow irrigated sugarcane. Across three soil types, reducing total application of irrigation water from approximately 4000 to approximately 2000 millimetres per year was predicted to reduce nitrogen losses by runoff or deep drainage without reducing crop yield (Thorburn *et al*., 2011a). Results varied across soils, and were more pronounced in the most permeable soil. Interestingly, nitrogen losses (and yield) were mainly affected by the total amount of irrigation water applied to the crop and not frequency of irrigation or the amount applied per irrigation (although the variations in these were kept within practical limits).

A complication with nutrient losses from irrigated fields is that on many furrow-irrigated farms irrigation tail-water can be collected in small on-farm dams and reused (Thorburn *et al*., 2011a; Cotton Australia, www.cottonaustralia.com.au/cotton-growers/mybmp), reducing the potential for off-farm losses. It has been estimated that tail-water is captured on 30 per cent of farms in the southern Burdekin irrigation area (Davis *et al*., 2013). On sugarcane farms however, the dams are relatively small and water can be flushed from the dams during large rainfall events, moving nutrients (and sediments and other chemicals) off farms, reducing the efficiency with which dams keep nutrients on farms. The frequency with which water is flush out of dams and into local water courses is not known. High concentrations of pesticides have been found in creeks draining the southern Burdekin irrigation area (Davis *et al*., 2013), suggesting substantial release of tail-water from farms in that region consistent with the proportion of farms (30 per cent) capturing irrigation tail-water. Capturing and recycling of all tail-water and some rainfall runoff is considered good practice in irrigated cotton and grains in central Queensland (e.g. Cotton Australia, www.cottonaustralia.com.au/cotton-growers/mybmp). It is believed to be widely practiced (APVMA, 2012), although no adoption data were available.

Areas for targeted management

Given that nutrient surpluses are an indicator of nutrient losses to the environment, cropping industries and/or regions with the highest surpluses could be targeted for improved management. Thus, based on its high nitrogen surpluses, banana production may be a priority for action. However, the area under sugarcane is an order of magnitude higher than bananas, so while nitrogen surpluses in sugarcane are around half those of bananas per unit area, the total mass of surplus nitrogen from sugarcane may be five times that of bananas. Nevertheless, nutrient surpluses provide a framework for assessing the potential risk of nutrient losses, concepts that could be incorporated into ecological risk frameworks.

Another factor to consider in prioritising management is the mode of transport of the nutrient, which has possible implications for spatially targeting management. Nutrients sorbed to the soil move in particulate form with sediments, while dissolved nutrients move with water. Issues related to targeting management of sediment were discussed above, and areas closer to the coast and down stream of water impoundments should be higher priority for managing particulate nutrients. However, the effectiveness of impoundments in major floods, which transport the majority of sediment, may be questionable (Lewis *et al*., 2013), so the evidence base for spatially targeting management of particulate nutrients is not firm.

Soluble nutrients move with water and so are less affected by the processes that slow or halt sediment movement. Still, there are processes that may reduce the amount of dissolved nutrients in stream, such as denitrification of dissolved inorganic nitrogen, and fields further up-stream in catchments might be seen as lower priority for management than those close to marine ecosystems. However, there is currently no firm basis for spatially targeting management of dissolved nutrients, beyond spatial variations in surpluses. Losses of dissolved inorganic nitrogen through denitrification in rivers are very small or negligible relative to total fluxes according to catchment scale modelling (McKergow *et al*., 2005b). This is supported by in-stream monitoring of a lagoon reach of the Tully River, which found denitrification to be negligible relative to the total flux (McJannet *et al*., 2012). In the Mackay Whitsunday region also, there is reasonable agreement between the mass of dissolved inorganic nitrogen predicted to leave fields and that measured exported from rivers and creek in the region (Biggs *et al*. 2013). .

Further research

- It would be valuable to determine the minimum nitrogen surplus needed to maintain crop yields (or profitability) for the crops grown in the Great Barrier Reef, particularly intensive crops such as sugarcane, bananas and cotton.
- The relationship between phosphorus surpluses, soil phosphorus concentrations and phosphorus lost to the environment in both particulate and dissolved forms needs to be determined.
- More information is needed on the risk of nutrient supply limiting crop yields, and how that risk varies between the short- and long-term is poorly understood. Crop yield responses to nutrient supply should be examined in terms of the probability of a limitation, as opposed to using fixed functions describe yield response to nutrient inputs or targeting yield potential. Such knowledge may help farmers make more informed decisions regarding the production and profitability outcomes of their nutrient management.
- The contribution of organic sources of nutrients (e.g., nitrogen from legumes, nitrogen and phosphorus from mill mud) to nutrient losses (both dissolved and particulate) needs to be determined at both the field and regional scale. If the contribution is significant, methods to manage those losses (e.g., better managing supplementary fertiliser in these situations) need to be developed.
- The long-term efficacy of practices such as splitting nitrogen applications, burying nitrogen fertiliser, and applying ‘enhanced efficiency fertilisers’ needs to be determined across soil types, climates and, importantly, nitrogen application rates.
- More information is needed about the magnitude and, possibly, management of nutrient losses from grains production areas.
- There is little information on nutrient losses from, or nutrient management in, fertilised grazing lands.
- The potential for improved irrigation management and water use efficiency to reduce losses of nutrients from fields needs to be further established.

Pesticides

Pesticides include insecticides, herbicides and other pest control chemicals. Currently, herbicides are the most widely used, most commonly occurring and ecologically significant pesticides found in the Great Barrier Reef marine environment (Kennedy *et al*., 2012; Lewis *et al*., 2009). Herbicides that inhibit functioning of photosynthesis at photosystem II in plants (PSII herbicides) have been identified as a concern (Jones, 2005). In contrast, insecticides are rarely detected in the Great Barrier Reef waters (Kennedy *et al*. 2012). Thus, most research on managing pesticide losses from Great Barrier Reef catchments has focussed on herbicides. However, the principles of managing loss of herbicides are general to all pesticides (that is, the active ingredient and in some cases its active breakdown products). The photosystem II inhibiting herbicides are soil residual herbicides including atrazine, ametryn, diuron, hexazinone and tebuthiuron, which have moderate half-lives (30-100 days; Shaw *et al*., 2011) and are typically weakly sorbed to soil (soil sorption coefficient Koc less than five; Weber *et al*., 2004). Loss or transport in runoff refers to total loss, in water and sediment, unless otherwise specified.

General principles

Pesticides have a wide range in chemical properties, which affect their behaviour and fate in the environment. These properties affect their persistence in various compartments (soil, water bodies etc), washoff from plants and crop residues, leaching and runoff, and their transport in either sediment (as an adsorbed phase) or water (dissolved phase). Management practices for reducing pesticide runoff (Connolly *et al*., 2002; Silburn *et al*., 2013) include:

1. Soil management practices that reduce runoff and sediment movement, e.g. reduced tillage, retention of surface stubble or trash cover and controlled traffic farming. Compounds strongly sorbed to soil and sediment will be managed by practices that reduce sediment losses. Compounds that are not strongly sorbed will be managed by practices that reduce runoff losses or increase infiltration.
2. Managing the source, i.e. amount and type of active ingredient used, timing of pesticide applications, its placement and application method, or alternatively replacement with a pesticide with properties that lead to less loss in runoff or less ecological impact while still obtaining the required weed control. There are strong relationships between the amount of pesticide applied, the amount in the soil surface and the amount lost if a runoff event occurs (Silburn and Kennedy, 2007). Because pesticides dissipate (loss or breakdown by any process) reasonably rapidly (weeks to months), the greatest losses will occur in the first few runoff events after application, and losses in these events will dominate the total seasonal losses (Wauchope 1978). Pesticides that have greater persistence (i.e., longer half-lives) have greater potential to be lost in runoff, and will also often have greater sorption to soil/sediment.
3. Management of runoff water after it leaves the field, e.g. practices such as silt traps, vegetative filters, storages and constructed wetlands (as discussed below).

Here we review the results from studies relevant to the Great Barrier Reef in the context of these general principles of management of pesticide runoff and in relation to previous work.

Reduce runoff and sediment movement

Retention of crop residues and reduced tillage

Conservation tillage (i.e. retaining a minimum of 30 per cent crop residue cover after planting) generally reduces runoff losses of pesticides (mostly herbicides) compared with bare, tilled soil (Fawcett *et al*. 1994). In a rainfall simulator study on cotton in the Great Barrier Reef, increasing cover reduced runoff loads and concentrations of more sorbed insecticides and herbicides, and reduced the loads but not the concentrations of the less sorbed herbicide prometryn (Silburn *et al*., 2002). Total event loads were reduced for all compounds in this study, because cover had a large effect in reducing both runoff and soil loss. In another rainfall simulator study in sugarcane, presence of sugar cane trash reduced runoff losses of residual photosystem II inhibiting herbicides by 40 per cent on average and of the knockdown herbicides by 26 per cent, compared with bare soil (Cowie *et al*., 2012).

However, these effects of conservation tillage are not always consistent for pesticides that are poorly sorbed to soil. In some cases, retention of crop residues is found to cause increased herbicide losses in runoff (Shipitalo *et al*., 2008). Greater runoff losses from conservation than conventional tillage are often related to large runoff events occurring in the first week or two after application, which will dominate the seasonal losses (Shipitalo *et al*., 2008). In these large events, herbicides incepted on crop residues that have not had time to dissipate are easily washed off by rainfall. In smaller rainfall events, infiltration that does not cause runoff moves the herbicide deeper into the soil, away from the runoff producing zone. Thus, retaining crop residues will be more effective in reducing herbicide runoff when there is rainfall before the main runoff event. At longer times after application, runoff losses will be reduced due to more rapid dissipation on crop residues than on soil (Shaw *et al*., 2011). These complex interactions between pesticide properties, behaviour, climate and crop residues are yet to be fully understood. However, in the majority of studies, retaining crop residues reduced pesticide runoff (Fawcett *et al*., 1994).

Controlled traffic

Runoff, sediment and residual pesticides loads and concentrations from row crops in fields under controlled traffic were lower than conventionally trafficked fields on contrasting soils; a black Vertosol (Silburn *et al*. 2013) and a brown Chromosol (Masters *et al*. 2013). The Vertosol site was in a condition prone to runoff, with the soil bare, crusted and moist. Controlled traffic reduced runoff by 37 per cent (36.4 to 23 millimetres), and soil loss by 59 per cent (5.0 to 2.1 tonnes per hectare). In contrast, the Chromosol site favoured good infiltration, with almost complete soil cover from a cane trash blanket and low antecedent soil moisture. Under these conditions, controlled traffic reduced runoff by 38 per cent (27.4 to 17.1 millimetres) and soil loss by 35 per cent (29 to 19 kilograms per hectare). On the Vertosol, non-trafficked furrows had 29–35 per cent lower mean concentrations in runoff and 55 per cent lower total losses than wheel traffic furrows, for the pesticides pyrithiobac sodium, metolachlor and diuron. On the Chromosol, controlled traffic reduced loads of ametryn, atrazine, diuron and hexazinone by 60, 55, 47 and 48 per cent, respectively. Rohde *et al*. (2011) found on average 15 per cent less runoff, for controlled traffic (wide rows with aligned wheel tracks) compared to narrow rows with unmatched wheel traffic, from sugar cane plots over three years with greater than average rainfall.

Managing the source

Application timing

Timing of pesticide application is important because concentrations of pesticide in soil (which affect concentrations in runoff) decrease through time after application, so delays in runoff after application reduce runoff concentrations and loads. This effect has been found in both rain fed (Masters *et al*., 2013; Murphy *et al*., 2013; Silburn *et al*., 2013) and irrigated (Davis *et al*., 2013) crops in Great Barrier Reef catchments. For example, concentration and losses of metolachlor declined from an event mean concentration of 360 micrograms per litre (equivalent to 4.3 per cent of the applied active ingredient) at 19 days after application (first event) to an event mean concentration less than 40 micrograms per litre after 36 days (Murphy *et al*. 2013). In that study, the half-life of the runoff event mean concentration was 7.2 days, showing how rapidly risk of herbicide loss diminishes after application. In another study, concentrations and losses of three residual herbicides in runoff were more than 70 per cent lower 34 days after application compared with two days (Silburn *et al*. 2013). Similarly, Masters *et al*. (2013) found event mean concentration of atrazine and ametryn in runoff were approximately eight-fold lower at 21 days after application compared with one day, although reduction in event mean concentration for diuron and hexazinone were only 1.6–1.9 fold suggesting longer persistence of these chemicals. These lower concentrations lead to lower loads. For example, Rohde *et al*. (2011) found that over 90 per cent of herbicide was lost in the first 11 millimetres of runoff when runoff occurred within 20 days of application.

In general, the combination of early application before rain or delaying irrigation after applications, with band spraying and controlled traffic is most effective in reducing pesticide losses in runoff than any one practice.

Banded spray application

Band spraying reduces the total amount of pesticide applied to a given area (i.e. a field) compared with blanket application, which in turn reduces pesticide runoff losses. In an experiment on a black Vertosol, total concentrations of pyrithiobac sodium, metolachlor and diuron in runoff were reduced with banded applications (40 per cent band) by 41, 32 and 54 per cent, respectively, and total losses in runoff were reduced by 38, 22 and 50 per cent (Silburn *et al*., 2013). Banding was more effective for diuron, a somewhat more sorbed chemical. Likewise, Masters *et al*. (2013) found losses were reduced by 32 to 42 per cent when applications were banded (50–60 per cent coverage) on a brown Chromosol. Band spraying on the hills (approximately 40 per cent band) was very effective for furrow irrigation, with concentrations in runoff of pyrithiobac sodium reduced by 75 per cent and losses by 88 per cent (Silburn *et al*. 2013). In both these studies, the effects of band spraying in reducing runoff losses diminished over time after application, likely due to movement of herbicides from the sprayed band into the unsprayed area, but it was effective at controlling losses at the time of greatest risk.

Band spraying and controlled traffic combined give greater reductions in pesticide runoff than either practice alone (Masters *et al*., 2013; Silburn *et al*., 2013).

Choice of product

The dissipation of pesticides in the environment varies, with more rapidly dissipating pesticides having lower risk of ecological impact. For example, one day after applications Masters *et al*. (2013) found the loss in the dissolved phase was in the order of hexazinone (9.8 per cent), greater than atrazine (7.9 per cent, greater than) diuron (6.1 per cent) greater than ametryn (5 per cent). Losses at day 21 were hexazinone (5.9 per cent), great than diuron (2.8 per cent), greater than atrazine (0.7 per cent), greater than ametryn (0.4 per cent), indicating that the latter two herbicides dissipated more rapidly. However, the choice between herbicide classes with contrasting ecotoxicology, for instance residuals versus 'knockdowns' in the case of protecting the Great Barrier Reef, is a more relevant issue for management. In direct comparisons, runoff concentrations of some 'knockdown' products were lower than for residual herbicides (Cowie *et al*. 2012; Murphy *et al*. 2012, 2013; Shipitalo *et al*. 2008). These differences in herbicide runoff potential are also reflected in waterway monitoring studies. In drains and creeks in the lower Burdekin region, residual chemicals have been detected at higher frequencies (e.g. great than 50 per cent of samples) than knockdowns (less than 33 per cent) (Davis *et al*. 2013).

Even though knockdowns dissipate more rapidly and are more sorbed to soil/sediment than residual herbicides, they still should be managed carefully, with due consideration of integrated weed management, weed resistance and ecotoxicology.

Areas for targeted management

Pesticide concentrations decrease substantially, i.e. an order of magnitude (Davis *et al*. 2013), between fields and nearby receiving creek systems, potentially due to considerable dilution that takes place over relatively short distances. This result suggests that it is important to target management action to fields close to receiving creek systems. Pesticide risk modelling suggested concentrations generated by irrigation in the dry season, when dilution from rainfall runoff is absent, posed considerable ecological risk to aquatic ecosystems (Davis *et al*. 2013).

There is potential for breakdown and/or losses of pesticides in streams and rivers, although there is little information on these degradation processes for the Great Barrier Reef. In the areas with greatest generation rates of herbicides, i.e. the Wet Tropics and Mackay Whitsunday regions, transit times in streams and rivers may be short, so that the pesticide load delivered to the Great Barrier Reef may be similar to that entering the stream. Thus, it is not clear to what extent there should be a spatial prioritisation for pesticide management. Further study of herbicide breakdown during in-stream transport is needed to determine the potential for prioritising management spatially.

Further research

- Pesticide persistence on crop residues and soils in tropical and sub-tropical environments needs to be better characterised for the wide range of compounds used in the Great Barrier Reef catchment, so that their fate and risk can be more clearly defined.
- It is still uncertain if retaining cover consistently reduces pesticide runoff and under what circumstance it does/does not.
- Improved weed management and herbicide use guidelines are needed in conservation cropping systems where stubble or trash blankets are retained on the surface.
- The effects of pesticides on in-stream aquatic species are poorly defined in the Great Barrier Reef, and additive effects need to be better defined.
- Further study of herbicide breakdown and sorption/desorption during in-stream transport is needed to determine the potential for prioritising management spatially.
- A wide range of new and emerging herbicides are being used in cropping systems; much less is known about their behaviour and fate than for older herbicides.

Can improved agricultural management meet water quality targets?

Two studies have recently addressed the question of the extent to which water quality will be improved by the adoption of improved agricultural management practices, and whether the water quality targets (Table 2) can be reached. The first study (Thorburn and Wilkinson, 2013) used an empirical approach to develop frameworks linking land management to pollutant export from Great Barrier Reef catchments based on past studies. The second (Waters *et al*., 2013) used coupled field- and catchment-scale mechanistic and conceptual models to predict the effect of changed management on the export of pollutants from Great Barrier Reef rivers (Carroll *et al*., 2012).

Water quality frameworks

Thorburn and Wilkinson (2013) developed conceptual frameworks linking exports of dissolved inorganic nitrogen and fine sediment from agricultural lands to Great Barrier Reef rivers. These pollutants were chosen because they are both ecologically important, but have contrasting properties, dissolved inorganic nitrogen being a dissolved pollutant and fine sediments a particulate one. Thus an approach that can describe the behaviour of each pollutant is a more generally applicable approach. This generality was illustrated by combining the two frameworks to examine total nitrogen, a pollutant with both dissolved and particulate properties. The frameworks used nitrogen surpluses is the primary driver of dissolved inorganic nitrogen losses from agricultural land to rivers. Likewise, land condition, ground cover and riparian management, which are products of recent climate and grazing practices, were used to define sediment losses from hill slopes, gullies and stream banks.

From these frameworks, relationships between firstly, estimated nitrogen surplus and dissolved inorganic nitrogen exports, and secondly field-scale erosion and river fine sediment exports were developed to examine how dissolved inorganic nitrogen and fine sediment exports to the Great Barrier Reef may respond to a range of management scenarios for (1) reducing nitrogen inputs, and (2) increasing groundcover and improving riparian management. For total nitrogen, it was assumed that the response of dissolved nitrogen in total nitrogen would follow that for dissolved inorganic nitrogen, likewise the response of particulate nitrogen would be the same as for fine sediments.

It was predicted that universal adoption of best management practices, equivalent to B-Class practices, would only reduce dissolved inorganic nitrogen loads by 12 per cent, fine sediment loads by 15 per cent and total nitrogen loads by 14 per cent (Table 2), and thus not meeting water quality improvement targets set (or implied for dissolved inorganic nitrogen) by governments for these pollutants. In comparison, complete adoption of the most extreme practices, termed agri-environmental management practices (approximately equivalent to A-Class practices), were predicted to reduce dissolved inorganic nitrogen loads by 59 per cent, fine sediment loads by 19 per cent and total nitrogen loads by 24 per cent, thus generally meeting water quality improvement targets.

Table 2. Predicted improvement (percentage) in total anthropogenic pollutant loads from Great Barrier Reef rivers due to the universal adoption of B- and A-Class management practices. (nd = not determined)

Study / practice	Pollutant				
	Fine sediments	Total phosphorus	Total nitrogen	Dissolved inorganic nitrogen	Photosystem II inhibiting (PSII) herbicides
Department of the Premier and Cabinet (2009)					
Target reduction	20	50	50	50#	50
Thorburn and Wilkinson (2013)					
All B-Class*	15	nd	14	12	nd
All A-Class*	19	nd	24	59	nd
Waters et al. (2013)					
All B-Class	13	22	17	27	62
All A-Class	25	33	24	34	91

There is no stated target for dissolved inorganic nitrogen, but the target for total nitrogen implies a similar target for dissolved inorganic nitrogen (Thorburn and Wilkinson, 2013).
* Thorburn and Wilkinson (2013) defined scenarios based on best management practices and agri-environmental practices, approximately equivalent to B- and A-Class practices, respectively.

Mechanistic modelling

The mechanistic modelling couples one-dimensional models of soil-plant-atmosphere interactions with catchment scale models to predict the effect of field-scale agricultural management on end-of-catchment pollutant export (Carroll *et al*., 2012). The one-dimensional models include the processes involved in the generation and transport of the pollutants of interest in the Great Barrier Reef. These models are driven by soils, climate and, importantly, management practices, and so predict the generation and transport of pollutants in response to variations in these three factors. The outputs from the one-dimensional models are input into the catchment models which 'move' pollutants from the field to streams, where they are subject to various processes that alter their concentrations in water, and account to water and pollutant diversions and hence the loads exported from the catchments (Waters *et al*. 2013).

The mechanistic modelling framework was developed to evaluate the effect of management practice changes resulting from Reef Rescue grants to farmers and Queensland Government regulations (Carroll *et al*. 2012), both policies aiming to improve water quality in the Great Barrier Reef. However, the framework can also be used to predict the water quality outcomes of management scenarios. Two such scenarios are the universal adoption of B- or A-Class management practices. In the first scenario, fine sediment loads are predicted to be reduced by 13 per cent, whereas loads of photosystem II inhibiting herbicides are predicted to be reduced by 62 per cent, with the other pollutants intermediate between these two values (Table 2).

Universal adoption of A-Class management practices is predicted to reduce loads further compared to B-Class, i.e. reducing total phosphorus loads by 33 per cent and photosystem II inhibiting herbicides loads by 91 per cent, with the other pollutants intermediate between these two values. The mechanistic modelling thus predicts that universal adoption of A-Class management practices will meet water quality improvement targets for fine sediments and total phosphorus, but not for total nitrogen or dissolved inorganic nitrogen. This modelling also predicted the responses to management are spatially variable. Targets may be met with a lower level of practice change in some sub-catchments.

Conclusions

The two methods employed to assess the relationship between agricultural management and pollutant loads use contrasting approaches, yet result in generally similar predictions. Both predict that universal adoption of B-Class management practices will not meet water quality targets for fine sediments, total nitrogen or total phosphorus, but may for photosystem II inhibiting herbicides. Conversely, universal adoption of A-Class practices may also meet targets for fine sediments, but still not total nitrogen or total phosphorus. For dissolved inorganic nitrogen, Thorburn and Wilkinson (2013) predicted that agri-environmental practices would meet targets for this pollutant, although this was not the case for A-Class practices with the mechanistic modelling of Waters *et al*. (2013). This difference may have arisen because the agri-environmental practices proposed by Thorburn and Wilkinson (2013) are different from A-Class practices defined within Reef Rescue evaluation program (Carroll *et al*., 2012), especially for nitrogen where they are applied to a more extensive range of crops and are have smaller nitrogen-surpluses than A-Class practices currently defined.

Given these differences in practices definitions, methodologies and other uncertainties in these two studies, the general similarity in the results provides some confidence in the predicted outcomes. The message from both studies is clear: B-Class practices are unlikely to meet water quality targets, and management of agricultural lands will need to move beyond current industry accepted practices to more 'aspirational practices' if water quality targets are to be met. Whether or not these practices are the same as the agri-environmental practices proposed by Thorburn and Wilkinson (2013), or the A-Class practices developed in conjunction with regional bodies and used in the mechanistic modelling is uncertain. However, the need to go beyond B-Class practices, which reflect the bulk of experience in past research and development activities, is clear. The unproven nature of these aspirational practices will slow the rate of their adoption. They will need considerable testing and development before they would be confidently recommended.

Wetlands

The previous sections on sediments, nutrients and pesticides dealt with reducing losses of those pollutants from farm fields, i.e. source control. However, even with best management pollutants will leave fields and enter the broader landscape. When pollutants flow into wetlands they can be detained, allowing the possibility of storage, transformations, etc, reducing their concentrations in the landscape. This can be considered a form of 'treatment' of polluted waters. The role of wetlands in reducing concentrations and/or loads of pollutants in the landscape is considered in this section.

Guiding principles

- Natural wetlands including riparian vegetation can play a role in water quality improvement by nutrient reduction/assimilation, removal of solids, changing chemical parameters such as biological oxygen demand, and contaminant removal (Daniel and Greenway, 1995; Faithful, 1997; Finlayson, *et al*. 1993; Fisher and Acreman, 2004; Greenway and Daniel, 1997; Lukacs, 1998; Millennium Ecosystems Assessment, 2005; Reedy and Gale, 1994; WetlandCare Australia, 2008; Lizotte et.al. 2012).
- The impact of poor quality water flowing into freshwater wetlands can have a detrimental impact on the **values** of the reef. For example, the Great Barrier Reef is highly prized by recreational and

commercial fisherman with many fish species such as barramundi and mangrove jack utilising both freshwater and marine ecosystems throughout their life cycle (GBRMPA, 2012). Nutrients flowing into freshwater wetlands in farm runoff causes infestation of weeds, possibly blocking fish passage and preventing species from completing their life cycles.

- Wetlands have much broader values in the landscape than just water quality improvement including; reducing water velocity and flood mitigation, storing and transferring water, nutrient cycling, supporting biodiversity, connectivity, ecological processes such as breeding and recruitment of fish, carbon storage and local climate regulation. The impact of poor quality water flowing into freshwater wetlands can therefore have serious consequences for wider landscape health and productivity.
- As a general principle, wetlands are connected to ecosystems across the landscape in a variety of ways, at many different scales. For example, water flows provide connections between land, river and sea, facilitating habitat connectivity for plants and animals, as well as providing for nutrient cycling and food webs. Given the interconnectedness of terrestrial and aquatic ecosystems, it is necessary to think about all ecosystems in a complementary fashion, and to consider wetlands and marine ecosystems from a whole-of-landscape perspective.
- Not all types of wetland function in the same way with respect to water quality improvement, and some may not have a role at all (Clayton, 2009). This is because capacity of a wetland to undertake water quality improvement is dependent on a range of characteristics including; position in the landscape, vegetation type, water type, water regime, hydrological processes and residence times (Davis et. al., 2007; Clayton, 2009; Greenway and Daniel, 1997; McJannett et. al., 2012).
- Wetlands need to be managed in accordance with best management practice to ensure their multiple values are realised. Their capacity for maintaining water quality and minimising the quantity of undesirable accumulated materials (sediment, excess nutrients, weed germplasm) from being 'flushed out' of wetlands into the river network and the Great Barrier Reef during flood events needs to be addressed in this context.
- Constructed wetlands are built to enhance water quality through nutrient cycling (Boulton and Brock, 1999; DEEDI, 2011), trapping sediments (DEEDI, 2011; Mailliard et. al., 2012), absorbing metals and other toxicants and breaking down pesticides (DEEDI, 2011; Hunter and Lukacs, 2000; Maillard et. al., 2012; Marecik et. al., 2012). Constructed wetlands are designed to mimic the role of natural wetlands, in particular palustrine wetlands.
- Constructed wetlands should not be used to replace good land management practices. Likewise, natural wetlands should not be considered the primary mechanism for water purification. This approach would likely have a high risk to broader wetland and catchment values (Clayton, 2009).

Water quality

The spatial and temporal influence of wetlands on water quality has been the subject of interest but of limited scientific study in the Great Barrier Reef catchments (Clayton, 2009). There is strong evidence from research in other areas that many wetlands perform a role in water quality improvement (Daily, 1997; Davis *et al*., 2007; Evans *et al*., 1996), although not all wetlands will perform this function similarly, equally or in some cases at all (Davis *et al*., 2007; McJannet *et al*., 2012). A recent Queensland study assessed the ability of a tropical river lagoon to improve water quality and found it to be neither a sink nor source of sediments or nutrients (McJannet *et al*., 2012). The findings from this case study cannot be extrapolated to lacustrine or palustrine wetland types within the Great Barrier Reef catchment however, as the characteristics of the lagoon are not the same as these other wetland systems. This is because water quality improvement is dependent on a range of landscape and wetland characteristics including; position in the landscape, vegetation, water type, water regime, hydrological processes and residence times (Clayton, 2009; Daniel and Greenway, 1997; Davis *et al*., 2007; Greenway and Daniel, 1997; McJannet *et al*., 2012). It is these exact characteristics which can be manipulated to develop constructed treatment wetlands specifically designed for water quality improvement.

Constructed wetlands

Artificial wetlands can be designed and constructed to maximise capacity for runoff capture and treatment. Constructed wetlands have been established to treat a wide range of polluted waters, including primary and secondary municipal sewage, landfill leachate, industrial waste waters and agricultural runoff or irrigation tail-water (DEEDI, 2011).

Constructed wetlands have been demonstrated to cycle nutrients (Boulton and Brock, 1999; DEEDI, 2011), trap sediments (DEEDI, 2011; Maillard *et al*., 2012), absorb metals and other toxicants, and break down pesticides (DEEDI, 2011; Hunter and Lukacs, 2000; Maillard *et al*., 2012; Marecik *et al*., 2012). A study undertaken in tropical El Salvador provides strong evidence to suggest that constructed wetland performance is reliable under extreme tropical conditions, such as the peaks of the wet and dry season (Katsenovich, 2008). Greenway and Woolley (1999) examined the performance efficiency of constructed wetlands in Queensland in terms of nutrient and bioaccumulation, concluding that constructed, free water surfaces wetlands are a viable option for improving the quality of secondary sewage effluent.

Constructed wetlands have been established throughout Great Barrier Reef catchments to treat runoff from urban and agricultural land uses, e.g. the Gaia Banana Farm (Queensland Wetlands Program, 2011). These have been designed and built in a variety of forms and for different purposes, such as water quality improvement, increasing biodiversity and water capture and reuse. The key design elements vary depending on the primary desired function of the constructed wetland. When designing constructed wetlands for a water quality improvement function, a holistic 'treatment train' approach is required, whereby pollutant runoff is minimised upstream of the wetland, enabling the wetland to provide a final 'polish' function. It is essential that good land management practices are adopted in the upstream catchment that minimise the potential for nutrient, chemical and sediment loss. A treatment train approach also calls for structures to slow water velocity and capture coarse sediments (i.e. vegetated swales and sediment traps) prior to entering the constructed wetland. Vegetated swales improve the treatment efficiency and lifespan of constructed wetlands (DEEDI, 2011). The effect of adding swales to the treatment train is to provide a secondary treatment process prior to the final polishing that occurs within constructed wetlands (DEEDI, 2011). Sediment traps are most effective when they are positioned ahead of constructed wetlands. Sediment traps facilitate sedimentation by enlarging the channel so that water velocities are reduced to the point that sedimentation can occur (DEEDI, 2011). They have an important role in water quality improvement because excessive amounts of sediment can rapidly change wetland depth, which can then create ideal conditions for weed infestations that smother habitat, reduce oxygen levels, reduce treatment efficiency and reduce wetland lifespan (DEEDI, 2011). Research is currently being undertaken on constructed wetlands on farms in north Queensland to better understand the ability of these structures to capture and treat runoff.

Constructed wetlands should always be located to have minimal impact on natural drainage and therefore the environmental flows necessary to maintain healthy aquatic ecosystems. It is important to avoid locating constructed wetlands in natural drainage lines and the riparian zone of wetlands to minimise any impacts on downstream flow volume and natural flow patterns. Poorly designed and maintained wetlands can:

- reduce natural stream connectivity
- impact on fish passage
- increase downstream flooding
- concentrate flows thereby increasing erosion potential
- capture overland flows which could upset downstream water users and breach legislation
- increase the spread of wetland weeds.

It is therefore important to consider the implications that a constructed wetland will have on the wider landscape. Details on the design and location of constructed wetlands are given in the Wetland Management Handbook (DEEDI, 2011).

Maintenance of wetlands

Wetlands need to be maintained to ensure proper functioning. Inspections should be carried out as part of an overall system maintenance program. Along with providing a general indication of the health and diversity of the vegetation, inspections provide the opportunity to detect specific site problems. These can include the accumulation of sediment, plant debris, litter or oils; infestation of weeds; pigs, mosquitos and other pest problems; algal blooms and scouring. Inspections will also help assess the wetland systems' performance in achieving its stated objectives.

Since many weeds are transported during flood events, a key focus of wetland maintenance is the eradication of noxious or nuisance species (DEEDI, 2011). Large areas of weed matting have been observed in lagoons in the Great Barrier Reef catchments. The growth of the weed mats is facilitated by nutrients washed into the lagoons during rainfall events. During extreme flooding events, there is a risk that weed mats will be flushed into the Great Barrier Reef lagoon, carrying nutrients, sediments and pesticides with them and thereby directly impacting reef water quality. The risk of this type of pollutant dumping in to the Great Barrier Reef is likely to be exacerbated by a greater frequency of extreme flood events predicted as a result of climate change.

A case study for Healey's Lagoon, Burdekin Shire, describes how a sudden flood event flushed out almost the entire weed mat from the lagoon (Queensland Wetlands Program, n.d.). Healey's Lagoon is a freshwater system five kilometres long, surrounded by intensive sugarcane production. Prior to the sudden flood event, the lagoon was infested with aquatic weeds, mainly water hyacinth which was having a detrimental impact on the lagoon system. As with many coastal lagoons in Queensland, Healey's Lagoon has a modified hydrology and is also used to irrigate sugarcane. This pressure combined with irrigation tail-water in flows, resulted in high nutrient levels that facilitated the growth of floating weed mats (as well as weeds attached to the substrate). It is the floating weeds however, that pose the greatest risk to the water quality of the Great Barrier Reef because of the potential to be flushed to the Great Barrier Reef lagoon during flood events.

Further research

While knowledge of the fundamentals needed for wetland management in the Great Barrier Reef catchments has been enhanced by the Queensland Wetlands Program since 2003, significant questions concerning natural wetland processes at both the site level and landscape scale in the Great Barrier Reef catchments remain unexplored (Davis *et al*., 2007). A greater understanding of wetland processes is needed to better understand biotic and abiotic dependencies within wetlands and the relationship between wetlands and non-wetland ecosystems to inform landscape management and planning decisions. To fill these gaps priority should be given to:

- targeted site specific and landscape investigations of the hydrology, ecology and function of the range of wetland types in Great Barrier Reef catchments
- on ground management of palustrine wetlands coupled with evaluation of impacts on wetland values and functions including water quality and landscape functions
- targeted site specific and landscape investigations of pesticides and sediments associated with palustrine wetland types
- comprehensive monitoring and assessment of wetland condition, extent and pressures to identify priority sites for management intervention and inform ongoing management of wetlands and catchment landscapes in the Great Barrier Reef catchment
- research to identify the best locations and designs for constructed wetlands to maximise water quality benefit with minimal impact on natural landscapes
- identify and map Great Barrier Reef lagoons with weed mat infestation and quantify the impacts of weed mat pollution on the Great Barrier Reef lagoon as an outcome of flood events
- research to better understand the water quality improvement capacity of natural and constructed wetlands in Great Barrier Reef catchments.

Economic benefits of improving water quality

General economic framework

There are five major contributions that economic analysis brings to understanding the issues involved in changing agricultural management practices:

1. Costs of making management changes vary widely between different practices and across landholders. This makes it difficult (and sometimes misleading) to generalise about the desirability of particular practice changes. The challenge is to find the management practices that deliver the largest biophysical outcomes and involve the lowest costs, as these are the most cost-effective solutions.
2. The benefits of reducing emissions vary across pollutants and geographic areas. The proportion of pollutants transmitted and the time lags involved varies within and across catchments, so the benefits making reductions at the farm level also varies. This means that there is a challenge to find the most efficient solutions that will deliver improvements where they achieve the largest benefits.
3. Agricultural practices, and changes to them, generate a variety of both public and private costs and benefits. Agriculture generates private benefits (e.g. crop production) as well as public benefits to society, but also public costs in the form of environmental damages. Improved management practices will typically involve private costs to make the changes, as well as private benefits from higher agricultural returns and public benefits from reduced environmental impacts. Some management changes, such as reductions in over-grazing or fertilizer applications, generate win-win outcomes for both landholders and the environment, while others involve mixed outcomes.
4. Economic analysis can be used to provide rigour in individual project selection, particularly where public funds are involved, using tools such as cost benefit analysis. At the macro level the key economic question is whether there are net economic benefits in reducing water quality impacts on the Great Barrier Reef. That involves comparing the economic benefits of improved protection against the costs involved in achieving that protection across all stakeholders (Rolfe and Windle 2011a). At the micro level the task is to identify for each project proposal what the projected improvements in water quality will be, what the costs of improvements will be at the farm level, and to evaluate projects by the cost effectiveness of outcomes.
5. The factors influencing the choices that landholders make about management actions are complex. There are several reasons why adoption of different management practices may be slow, such as there may be few or no net benefits at the enterprise level, there may be large risks or uncertainties involved, or there may be other barriers to change management practices. In some cases institutional mechanisms and policy settings may need to be adjusted to overcome barriers. Identifying strategies that improve adoption of preferred management practices and the underlying economic and social drivers is critical.

Many of the benefits of improved water quality are public benefits, representing the preferences of the broader community for better protection of the Great Barrier Reef. For example, reductions in fertiliser applications can limit nutrient losses into waterways, generating public benefits in the form of improved environmental outcomes. However, the costs of additional protection involve both public and private costs. The private costs to landholder can include both financial costs involved in changing management practices, as well other non-financial tradeoffs. In the case of reducing fertiliser applications, there may be private benefits in terms of reduced input costs, private costs in terms of reduced crop yields (or risks of lower yields), and public costs for programs to reduce fertiliser inputs. Landholders have little incentive to provide public benefits that incur some private cost in excess of private benefits. Increasing adoption of better management practices therefore involves an understanding of the private and public costs and benefits involved, and how landholders respond to varying financial and non-financial drivers.

Private and public benefits

There are different groups of economic benefits provided by the Great Barrier Reef. The most visible are the direct commercial benefits to the fishing, tourism and other industries specifically involved. A series of reports by Access Economics (2005 – 2007) and other sources have identified the size of industry turnover and regional contribution. The economic contribution is measured by producer surplus (roughly equivalent to industry profits).

Direct use values accrue to consumers, in particular recreational users and recreational fishers. These benefits are large and diverse, but difficult to measure because values are not traded in markets. A number of studies have estimated recreation values, using specialist non-market valuation techniques (e.g. Kragt *et al*., 2009; Prayaga *et al*., 2010; Rolfe and Gregg, 2012; Stoeckl *et al*., 2011). Given the number of international and domestic visitors to the Great Barrier Reef, and the variety of recreational uses, these direct use values are high.

An important category of economic benefits associated with reef protection are the non-use values, which are protection values held by the community, even if they do not directly use the asset. There is emerging evidence that substantial protection values exist. For example, Rolfe and Windle (2012) demonstrate that Australian households would on average be prepared to pay $21.68 per household per annum for five years for each one percent improvement in reef health, with slightly higher values held by households close to the Great Barrier Reef.

Private tradeoffs of improving water quality

There are a variety of costs involved in improving water quality from agricultural and other sources. These are difficult to predict, for three important reasons:

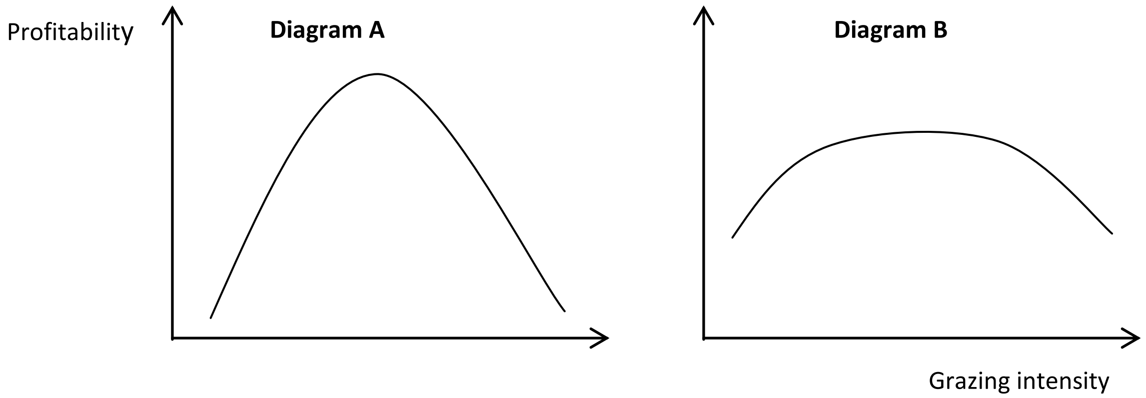
- Reductions in sediments, nutrients and agricultural chemicals can typically be achieved by changing farm operations or ceasing some practices in particular areas. The tradeoffs between changes in farm operations and the environmental outcomes generated are complex.
- Most costs of changes in operations are not directly visible to external policy makers, making it difficult to set policy with certainty about outcomes. In some cases, landholders are unsure of the full costs involved in changing management practices until trials of practice change are held.
- The costs of improving water quality through changed agricultural management practices vary substantially across producers, agricultural sectors, and catchments (Rolfe *et al*., 2011; Rolfe and Windle, 2011b).

Differences in landholder costs of changing practices occur for a number of reasons, including variations across management actions and costs, the differences in profitability and returns across industries, differences in the ecological responses, and the different time lags until actions take effect. For example Rolfe and Windle (2011b) identify that the cost of reducing nitrogen emissions in a water quality tender varied from the most efficient price of $0.03 per kilogram to the most expensive price of $17.49 per kilogram, a 600-fold difference, in only the proposals that were viable to fund.

There are different cost tradeoffs that landholders face for improving water quality. At the private business unit, an important consideration is how changes in management practices will affect net profitability to the business. For example, landholders with grazing operations face trade-off curves where net returns increase, peak and then decline with increasing levels of an intervention (Figure 1). For landholders with grazing pressures in excess of the peak profitability, it is financially attractive to reduce grazing pressures to the peak where maximum returns are gained. Reducing grazing pressures below this point generates a cost to landholders in the form of reduced returns. Thus decreasing grazing intensity to improve water quality can have varying impacts on private returns.

In some situations (e.g. Figure 1, Diagram A), the optimum point to produce is very clear because the production tradeoffs have a sharp peak. It also means that there are large costs for changing management intensity, whether positive or negative. Conversely, there may be a flatter trade-off function (Diagram B) where the tradeoffs in changing management intensity are much smaller, whether positive or negative. It is much easier to make changes in agricultural management when trade-off functions are very flat compared to when they are steep because the impacts on profitability are smaller (Pannell, 2006). These simple examples help demonstrate why the tradeoffs might vary so much within the agricultural sector, depending on the production functions and the point where landholders operate. Actual production functions vary between industries, farms and over time, making actual tradeoffs difficult to measure.

Figure 1. Two hypothetical examples of the change in profitability (vertical axis) for increasing intensity of grazing pressure (horizontal axis).



Grazing

There are net economic benefits to the grazing industry of maintaining natural resources in good condition. In the long term, higher pasture productivity from maintaining resource condition is approximately 20–30 per cent more profitable than continuous heavy stocking and subsequent declines in land condition (Ash *et al*., 1995; McIvor *et al*., 1995; O'Reagain *et al*., 2011; Star *et al*., 2012). Research has also identified that there are long term economic benefits from reducing utilisation rates during droughts to maintain the resource base (Landsberg *et al*., 1998; O'Reagain *et al*., 2009, 2011). This means that where over-grazing is occurring, there should be both private benefits and public benefits from reducing stocking rates.

Evidence from economic analysis of grazing operations (Macleod and McIvor, 2006; Star *et al*., 2011, 2012) are that economic tradeoffs vary across different land types and pasture utilization (stocking) rates. Star *et al*. (2012) modelled the economic tradeoffs to reduce grazing pressures and sediment movements from five land types in the Fitzroy catchment, identifying that the lower productivity land types presented the cheapest option for sediment reductions. In the modelled results, the costs of achieving sediment reductions varied by more than 100 times over the selected case studies, with opportunity costs very sensitive to the existing land condition. The cost of reducing sediment emissions varied from a low of $4 per tonne for the Silver-leafed Ironbark on duplex soils land type in poor condition to a high of $421per tonne for the Brigalow-Blackbutt land type in very good condition. The results also demonstrate that marginal costs of sediment reduction increase sharply as higher and higher levels of reduction are sought.

Macleod and McIvor (2006) examined reasons why cattle producers do not lower stocking rates to minimize risks of sediment runoff. They identified that relatively small changes to herd numbers can have large impacts on short-term net profit or loss for cattle producers when existing profit margins are low. In many cases poor economic prospects and a range of practical management issues are apparent barriers to changing present practices on private land. This study also identified that progressive changes in practices to better consider ecological sustainability are possible, and indeed an imperative to sustain productivity in the longer term. House *et al*., (2008) showed that there can be substantial income losses from applying conservation-based scenarios in mixed beef-grain production systems, and that there are limited opportunities to offset these with changed farming practices that do not create other environmental problems. The results of Star *et al*., (2012) demonstrate that while reduced grazing pressure can involve negative financial tradeoffs for graziers if very low utilisation rates are adopted, those tradeoffs tend to be very small, or are positive, for graziers with poor land condition, or whose current stocking rates are in excess of optimal levels.

While there is now better understanding of the structure and diversity of tradeoffs facing grazing operators, many knowledge gaps remain. Detailed analysis has only been conducted for some land types and grazing operations in Great Barrier Reef catchments and the net private benefits of making different management changes is poorly understood. Better information is needed about optimal stocking rates and tradeoffs from reducing grazing pressure across different land types, and about the net private costs (after any benefits are factored in) of different management actions.

Sugarcane

Evidence from economic analysis of sugarcane operations (Roebeling *et al*., 2009; van Grieken *et al*., 2011a, 2011b) are that economic tradeoffs from moving to better management practices can vary from strongly positive to strongly negative.

Van Grieken *et al*., (2011a) estimated the cost-effectiveness of a range of scenarios for the Tully-Murray catchments, where reductions in dissolved inorganic nitrogen loads reaching the end of the river system could be achieved by either improving management or taking land out of sugarcane production. Improvements in management were summarised by the ABCD framework (Higham *et al*., 2008). Nitrogen fertiliser usage reduces with higher management class, with rates predicted at 68-93, 120-140, 150 and 180 kilogram of nitrogen per hectare for Classes A, B, C and D respectively. It was assumed in the baseline scenario that 40 per cent, 53 per cent and seven per cent of cane land was being managed under Classes B, C and D respectively, with improvements available by changing land management to Class B (Class A was not included as commercial viability had not yet been proven). Results indicate that improving management actions can generate some positive returns to cane farmers, with improved landholder profits predicted for 10 per cent and 20 per cent end-of-river reductions in dissolved inorganic nitrogen. Further improvements in water quality would begin to generate net costs to farm operations, as more expensive management changes were made and land was retired from sugarcane production. Improving water quality by more than 30 per cent was predicted to lead to a reduction in regional income (due to reduced cane production) and a decrease in regional employment (due to the adoption of less labour intensive management classes and/or taking sugarcane land out of production). Reducing regional dissolved inorganic nitrogen loads from sugarcane production by 80 per cent, as indicated by the Tully–Murray catchment sustainable load target, may not achieve the minimum average annual cane production of two million tonnes given current proven technologies. This means that major reductions in sugarcane production, with flow-on effects into regional economies, may be required to achieve an 80 per cent dissolved inorganic nitrogen load.

A similar study was performed by van Grieken *et al*., (2011b) for the Tully and Pioneer catchments. That study also examined the costs involved for sugarcane growers to change management practices in the ABCD framework. Based on an average implementation cost (capital investment) per farm, the costs of changing practices were estimated as: D- greater than C equals $11,250; C- greater than B equals $63,000; B- greater than A equals $76,500. While the costs of improving from D Class to C Class are low and net financial benefits are generally positive, there is limited scope for improvement because most farmers are already at C Class practice standards or above. The results also identify that the average farm should generate net economic benefits over time from moving from C to B Class practices, indicating that the production benefits from moving to adopt recommended practices should outweigh the costs involved. There was some variation in tradeoffs between catchments, and results indicated that larger water quality improvements that would require changes to Class A practices or land being taken out of production would come at a net cost to farmers. The results of this study indicate that the cost-effectiveness of all sugarcane growers adopting category Class-B practices would be $5.97 million for the Tully-Murray and $349,059 for Pioneer over five years, and $18.39 million and $12.78 million respectively over 10 years. However, the net present value of adopting category A practices is significantly negative for the Tully-Murray catchment in both the five and 10 year scenarios, and only positive for the Pioneer catchment in the 10 year scenario.

The biophysical modelling undertaken with the study found that even if all sugarcane growers adopted category B Class practices, the long-term annual end of catchment load of dissolved inorganic nitrogen will only decrease by 13 and 16 per cent for the Tully and Pioneer catchments respectively (van Grieken *et al*., 2011b). This is well short of the Reef Plan 50 per cent reduction target for nitrogen by 2013. It implies that the Reef Plan targets have been set without a clear understanding of the management changes required to achieve the targets and the economic and social costs involved.

Four key messages come from the modelling results of van Grieken *et al*., (2011b):

1. Changes to higher management classes involve significant capital costs, and several years of operations are required before the change generates positive values.
2. The positive net present value of moving to Class B means that landholders have large enough private net benefits to do this without being regulated – although the study did not incorporate all transaction costs or risk preferences.
3. Only modest improvements in water quality can be generated by farmers moving to recommended management actions; larger improvements will require changes that involve net private costs to farmers.
4. There is heterogeneity in abatement costs, such that even prioritising by catchment is likely to be too large a scale for regulations to achieve cost-effective improvements to water quality relative to more flexible voluntary mechanisms.

Other industries

Other agricultural industries also exhibit variations in management practices and costs involved to achieve water quality benefits. Strahan and Hoffman (2009) summarise the net returns to broad acre cropping farmers of changing from conventional tillage to zero till or controlled traffic farming systems, and the potential associated reductions in sediment and nutrient movement. Their analysis demonstrates, similar with other industry results, that some advanced management practices that are being adopted can have both financial and environmental benefits. Rolfe and Windle (2011b) used the results of water quality tenders to estimate the private costs of improving water quality in the dairy and horticultural industries in selected Great Barrier Reef catchments, showing that the variation and scope of costs was similar to other industries.

Regional (cross-industry) costs

Roebeling *et al*., (2009) estimated for the Tully-Murray catchments the costs of sediment and nutrient reductions with an environmental/economic land use model. They identified that while some improvements in water quality can be obtained at no additional cost, or even benefits to, the agricultural industry, larger water quality improvements come at a significant cost because of costs and/or negative impacts on production. Large cost differences were identified between industries in those catchments.

In grazing production, all improvements in water quality came at a significant cost to the grazing industry because of reductions in both grazing area and stocking rates necessary to reduce sediment and nutrient movement. Estimated costs were about $1.2 million and up to $2.5 million per year for a 30 per cent and 60 per cent decrease in total suspended solids delivery respectively, and about $2.3 million and $6.6 million per year for a 35 per cent and 80 per cent decrease in dissolved inorganic nitrogen delivery respectively. It is likely that some improvements in grazing management can generate both increased profits and environmental benefits, however the modelling results from Roebling *et al*. (2009) demonstrate that major environmental improvements from grazing changes will involve substantial private costs.

In sugarcane, Roebeling *et al*., (2009) identified that maximum benefits were expected to be obtained through a reduction in total suspended sediments and dissolved inorganic nitrogen water pollution of approximately 20 per cent and 25 per cent, respectively, facilitated through the adoption of management practices that can improve net profitability (reduced tillage and zero tillage; economic optimum rates of fertiliser application, nitrogen replacement and split nitrogen application). Reductions in water pollution below 20 per cent (total suspended solids) and 25 per cent (dissolved inorganic nitrogen) would generate net benefits to canegrowers because of associated management improvements, but reductions above these levels would come at a cost to the sugarcane industry, particularly for reductions in total suspended solids and dissolved inorganic nitrogen delivery of over 35 per cent and 50 per cent, respectively.

Net benefits of additional protection

The comparison of costs and benefits confirm that many, but not all, initiatives to improve water quality may be justifiable. These include initiatives to reduce agricultural emissions, improve infrastructure, or better utilise natural systems such as wetlands. The net costs to landholders of making practice change have been shown to vary widely (e.g. Roebeling *et al*., 2009; Rolfe *et al*., 2011; Rolfe and Windle, 2011a,b; van Greiken *et al*., 2011a,b), helping to explain why there has been limited change in some farming and management practices. As well, landholders are sometime reluctant to change practices because of additional work, capital costs or risks involved, even for practice changes that may be more profitable in the medium to longer term.

Rolfe and Windle (2011a) assess the marginal public benefits of each one percent improvement in water quality ranged between $66.7 million and $102.4 million. This translates to a total annual benefit of between $19.9 million and $23.6 million for an annual reduction of 100,000 tonnes of sediment, plus 200 tonnes of nitrogen, and 46 tonnes of phosphorus. By comparison, Rolfe and Windle (2011b) identified from a small number of water quality tenders that the costs of annual reductions in sediment ranged from $1.62 to $89.22 per tonne, nitrogen from $0.23 to $4.56 per kilogram, and phosphorus from $1.78 to $10.80 per kilogram. The marginal costs for an annual reduction of 100,000 tonnes of sediment, plus 200 tonnes of nitrogen, and 46 tonnes of phosphorus range from $34.3 million to $145.6 million. The results indicate that while there is potential for water quality improvements to generate net benefits, high cost water quality improvements are generally uneconomic.

The results of Rolfe and Windle (2011b) and Star *et al*. (2012) demonstrate that a simple focus on management actions that achieve the largest reductions is not efficient. When the economic costs of making management changes is overlaid with the reduction in pollution expected, the results show large variations in the costs per unit of pollution achieved – up to more than 100 times. This means that the effectiveness of private and public funding to improve water quality can vary dramatically according to how well investments are targeted. One implication for policymakers is that funding benchmarks for major pollutants should be set below the unit funding levels reported by Rolfe and Windle (2011b):

- Sediment reduction: between $1.62 and $89.22 per tonne
- Nitrogen reduction: between $0.23 and $4.56 per kilogram
- Phosphorus reduction: between $1.78 and $10.80 per kilogram
- Pesticide reduction: up to $1689 per kilogram.

Another implication is that all proposals for public investment should be both evaluated and monitored by the expected reductions in pollutants achieved and the costs involved. Better targeting of management actions for cost-effectiveness of outcomes will ensure that more is achieved with public funding.

Improving adoption of best practices

Understanding the drivers for landholders to change management practices is an essential part of reducing agricultural impacts on water quality. Reasons for adoption or non-adoption include both economic and social drivers. Researchers such as Pannell (2006) have identified that a practice is unlikely to be adopted if landholders are not convinced that it advances their goals sufficiently to outweigh its costs and that broad based extension programs alone are unlikely to convince landholders unless the innovation is 'adoptable'.

Landholders do not simply follow short term profit signals, but make land management decisions according to a complex mix of drivers, including historical patterns, their ability to adapt to changed conditions, and their personal characteristics and circumstances. This implies the ability of landholders to adopt new practices, may be determined or constrained by a number of factors such as barriers to change (e.g. high debt levels), trialability, risks of adoption, and landholder capacity to assimilate new concepts and information. In many cases landholders have non-financial motivations for farming and adoption (Landsberg *et al*., 1998; Greiner and Gregg, 2011; Marshall *et al*., 2011).

Many studies in the Great Barrier Reef region about the adoption of best management practices have consistently found that apart from cost considerations, there is substantial variation in motivations for farming, perceptions of risk, and attitudes to trialling and adopting best management practices (e.g. Greiner *et al*., 2009; Greiner and Gregg 2011). Research is beginning to identify and quantify some of this heterogeneity, and to understand how landholder perceptions of benefits include both financial and non-financial rewards. Constrained economic conditions in rural industries can limit the ability of landholders to trial new practices or make the investments needed to change management operations.

Marshall *et al*., (2011) found that social factors rather than technical factors explained why graziers in the Burdekin catchment adopted or did not adopt the use of seasonal climate forecasts. Greiner *et al* (2009) note that the variation in drivers for farming practices and adoption mean that incentives need to appeal to the range of motivational profiles and risk perceptions of farmers, which may vary between industries and regions. They also note that best management practices have been adopted particularly by those graziers who pursue lifestyle and conservation goals and are intrinsically motivated to adopt conservation practices. An implication of this finding is that policy mechanisms should be designed in ways that do not generate disincentives to these landholders.

Some changes in management practices appears to be easier to facilitate, such as by lowering the costs of trialling and adoption. Capital investments and subsidies for practice change are likely to be successful where new practices are marginally unprofitable. Other practices require more targeted support, particularly those that involve increased risks of reduced profits or where changes are needed in whole farm systems.

Further research

In the past decade there have significant advances in the knowledge of the social and economic drivers of landholders adopting new management practices that deliver improved water quality outcomes, and some of the cost and other tradeoffs that inhibit change. The results identify a number of issues requiring attention.

A key problem revealed by the economic analysis is that there is no potential for low-cost solutions to achieve the desired targets of major reductions in sediments, nutrients and pesticides. Only modest improvements in water quality can be generated by farmers moving to recommended management actions; larger improvements will require changes that involve net private costs to farmers. While some zero and low cost solutions exist, the scale of the desired targets means that some management changes involving high private costs, with potential for regional economic impacts, will need to be found. Gaps in knowledge about the costs of making management changes means that targets have been set with little economic analysis of the tradeoffs involved.

A second issue is that low-cost solutions are rarely common across industries, regions and farms because of substantial variations in geography, climate and enterprises. Better understanding is needed about which low-cost solutions exist within industries, sub-catchments and across enterprises, and what are the factors that explain the heterogeneity in costs.

A third issue is that factors influencing landholder adoption of better management practices are complex and poorly understood. While there is awareness that factors such as knowledge and awareness, attitudes, impacts on production and costs, transaction costs of making changes, and the risks and uncertainties of different outcomes are influencing adoption, limited work or case studies have been conducted in Great Barrier Reef settings. There needs to be better understanding of both (a) the non-cost-based impediments to adoption and (b) systemic cost-based drivers of adoption or non-adoption in ways that allow for them to be targeted by new or novel adoption approaches.

A fourth issue is that poor understanding exists about the scope for low cost, medium cost and high cost solutions to deliver pollutant reductions across the Great Barrier Reef catchments. While this is partly because of the variability in costs involved and likelihood of adoption, the lack of a general framework to identify the potential practice changes and likely costs involved in achieving adoption is needed to evaluate the appropriateness of the targets that have been set, the necessary investments to achieve these targets, and the industry or regional scale impacts of changes.

A fifth issue is that current investments are being poorly targeted because of a failure to consider what the cost-effectiveness of management changes will be at the farm level. Heterogeneity in both the costs involved at the farm level, and the pollutant reduction achieved, mean that broad targeting of management changes is unlikely to be very efficient. Instead, the investments should be targeted by the cost-effectiveness of individual practice changes. Better mechanisms or benchmarks for project selection are required to generate higher returns for public investment.

A sixth issue is that the institutional framework needs attention. Different mechanisms to encourage practice change are needed according to (a) the mix of private and public benefits and costs that are generated, (b) the factors that landholders will respond to (c) the transaction costs involved. Many of the enterprise tradeoffs involved and incentives required to make changes will remain hidden to policy makers, so mechanisms are needed that will select for the most cost-effective options when information is incomplete, and will update programs and targets as better information is revealed.

Conclusions

Since the Scientific Consensus Statement 2008, there has been a substantial increase in our understanding of the water quality and economic benefits of agricultural management practices. There is clear, field-scale evidence relating a number of practices to water quality improvement. Examples include reduced application of nitrogen fertiliser in intensive cropping, reduced tillage and controlled traffic farming, banded spraying of herbicides and improvements in land condition in grazing enterprises. These maintain or increase productivity, reduce input costs, and so give economic benefits. As well, there is clearer recognition of the role played by natural and constructed wetlands in improving water quality.

Agricultural activities in Great Barrier Reef catchments are characterised by diversity of situations, climate and operations. This means that the most effective actions in terms of pollutant reductions to undertake can vary across farms, industries and districts. While there have been improvements in the extent of adoption of best management practices, aided through a number of support mechanisms, potential remains for better take-up of best management practices.

The costs of reducing pollutant emissions can also vary widely, as do the financial benefits that may be available. The variation in net benefits means that some actions may be profitable for landholders to undertake, while others involve net private costs. Examples of the former include changing management from traditional practices to industry recommended practices, while examples of the latter may include trialling aspiration practices or removing land from production. In some cases financial assistance may be needed to address capital or operating cost constraints, while in others information and encouragement mechanisms may be required to improve adoption.

Reference list

Access Economics, 2007. Measuring the economic & financial value of the Great Barrier Reef Marine Park, 2005-06. Research publication No.88, Great Barrier Reef Marine Park Authority, Townsville.

Access Economics, 2008. Economic Contribution of the GBRMPA, 2006-07. Report for the Great Barrier Reef Marine Park Authority, Townsville.

Agnew, J.R. Rohde, K., Bush, A. 2011. Impact of sugar cane farming practices on water quality in the Mackay region. Proc Aust Soc Sugar Cane Technol 33, 9 pp.

APVMA, 2012. Diuron review. Volume 2 of 4 Supplemental Environmental Assessment Report - Runoff Risk Assessment. Report was prepared for the APVMA by Department of Sustainability, Environment, Water, Population and Communities. Australian Pesticides and Veterinary Medicines Authority. ISBN 978-1-922188-01-4. www.apvma.gov.au. pp 175.

Armour, J.D., Nelson, P.N., Daniells, J.W., Rasiah, V., Inman-Bamber, N.G. 2013. Nitrogen leaching from the root zone of sugarcane and bananas in the humid tropics of Australia. Agriculture Ecosystems and Environment, in press http://dx.doi.org/10.1016/j.agee.2012.05.007

Armour, J.D., Hateley, L.R., Pitt, G.L., 2009. Catchment modelling of sediment, nitrogen and phosphorus nutrient loads with SedNet/ANNEX in the Tully–Murray basin. Mar. Freshwater Res. 60, 1091–1096.

Ash, A.J., Corfield, J.P., McIvor, J.G., Ksiksi, T.S. 2011. Grazing management in tropical savannas: utilization and rest strategies to manipulate rangeland condition. Rangel. Ecol. Manag., 64: 223-239. DOI: 10.2111/REM-D-09-00111.1.

Ash, A.J., McIvor, J.G., Corfield, J.P., Winter, W.H. 1995. How land condition alters plant-animal relationships in Australia's tropical rangelands. Agriculture Ecosystems & Environment, 56: 77-92.

Attard, S.J. and Inman-Bamber, N.G. (2009). Use of overhead irrigation in the lower Burdekin uses less water and does not penalise yield. Proc Aust Soc Sugar Cane Technol 31, 10 pp.

Barai, V.N., Akurde, A.B., Tandale, A.B. 2009. Effect of different soil covers on runoff and soil loss under simulated rainfall. International Journal of Agriculture Environment & Biotechnology, 2: 476-480.

Barry, G.A., Price, A.M. and Lynch, P.J. (1998) Some implications of the recycling of sugar industry by-products. Proc Aust Soc Sugar Cane Technol, 20, 52-55.

Bartley, R., Corfield, J.P., Abbott, B.N., Hawdon, A.A., Wilkinson, S.N., Nelson, B. 2010a. Impacts of improved grazing land management on sediment yields, Part 1: hillslope processes. Journal of Hydrology, 389: 237-248. DOI: 10.1016/j.jhydrol.2010.05.002.

Bartley, R., Wilkinson, S.N., Hawdon, A.A., Abbott, B.N., Post, D.A. 2010b. Impacts of improved grazing land management on sediment yields, Part 2: catchment response. Journal of Hydrology, 389: 249-259. DOI: 10.1016/j.jhydrol.2010.06.014

Bartley, R., Hawdon, A., Post, D.A., Roth, C.H., 2007. A sediment budget for a grazed semi-arid catchment in the Burdekin Basin, Australia. Geomorphology, 87: 302-321.

Bell, M.J., Garside, A.L., Halpin, N., Salter, B., Moody, P.W., Park, G, 2010. Interactions between rotation breaks, tillage and N management on sugarcane grown at Bundaberg and Ingham. Proc. Aust. Soc. Sugar Cane Technol. 32, 119-139.

Bengtson, R.L., Selim, H.M., Ricaud, R., 1998. Water quality from sugarcane production on alluvial soils. Transactions of the American Society of Agricultural Engineers 41, 1331-1336.

Biggs, J.S., Thorburn, P.J., Crimp, S.J., Masters, B., Attard, S.J., 2013. Interactions between climate change and sugarcane management systems for improving water quality leaving farms in the Mackay Whitsunday region, Australia. Agric. Ecosyst. Environ., doi:10.1016/j.agee.2011.11.005

Bloesch, P.M., Rayment, G.E., Pulsford, J.S. 1997. Regional total phosphorus budgets for sugar production in Queensland. Proc Aust Soc Sugar Cane Technol 19, 213-220.

Boulton, A., Brock M., (1999) Australian Freshwater Ecology Processes and Management; Cooperative Research Centre for Freshwater Ecology

Bramley, R.G.V., Roth, C.H., 2002. Nitrogen levels in the Tully River - a long-term view. Mar. Freshwat. Res. 53, 931-940.

Brodie, J., McKergow, L.A. Prosser, I.P., Furnas, M., Hughes, A.O., Hunter, H. 2003. Sources of sediment and nutrient exports to the Great Barrier Reef World Heritage Area. Australian Centre for Marine and Tropical Freshwater Research Report No. 03/11, James Cook University, Townsville.

Brodie, J.E., Kroon, F.J., Schaffelke, B., Wolanski, E.C., Lewis, S.E., Devlin, M.J., Bohnet, I.C., Bainbridge, Z.T., Waterhouse, J., Davis, A.M., 2012. Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses. Marine Pollution Bulletin 65, 81-100.

Buczko, U., Kuchenbuch, R.O., 2010. Environmental indicators to assess the risk of diffuse nitrogen losses from agriculture. Environ. Manage. 45, 1201-1222.

Buczko, U., Kuchenbuch, R.O., Lennartz, B., 2010. Assessment of the predictive quality of simple indicator approaches for nitrate leaching from agricultural fields. J. Environ. Manage. 91, 1305-1315.

Burkitt, L.L., Dougherty, W.J., Carlson, S.M., Donaghy, D.J. 2010. Effect of variable soil phosphorus on phosphorus concentartions in simulated surface runoff under intensive dairy pastures. Soil Research 48, 231-237.

Carroll, C., Halpin, M., Burger, P., Bell, K., Sallaway, M.M., Yule, D.F., 1997. The effect of crop type, crop rotation, and tillage practice on runoff and soil loss on a Vertisol in central Queensland. Aust. J. Soil Res. 35, 925–939.

Carroll,C., Halpin, M., Bell, K., Mollison, J. 1995. The effect of furrow length on rain and irrigation induced erosion on a vertisol in Australia. Australian Journal of Soil Research. 33: 833-50.

Carroll, C., Waters, D., Vardy, S., Silburn, D.M., Attard, S., Thorburn, P.J., Davis, A.M., Schmidt, M., Wilson, B., Clark, A., 2012. A paddock to reef monitoring and modelling framework for the Great Barrier Reef: paddock and catchment component. Mar. Pollut. Bull. 65, 136-149.

Cary, J., Webb, T., Barr, N. 2002. Understanding landholders' capacity to change to sustainable practices: insights about practice adoption and social capacity for change. Bureau of Rural Sciences, Canberra.

Chen, D., Suter, H., Islam, A., Edis, R., Freney, J.R., Walker, C.N. 2008. Prospects of improving efficiency of fertiliser nitrogen in Australian agriculture-A review of enhanced efficiency fertilisers. Aust J Soil Res 46, 289–301.

Chilcott, C.R., McCallum, B.S., Quirk, M.F. and Paton, C.J. 2003. Grazing Land Management Education Package Workshop Notes – Burdekin. Meat and Livestock Australia Limited, Sydney, Australia.

Chunale, G.L. 2004. Evaluation of different grass species for soil binding and soil aggregation properties under sub-montane zone of Maharashtra. Indian Journal of Soil Conservation, 32: 24-27.

Clayton, P. (2009). Natural Wetlands and Water Quality: The Role of Wetlands in Water Quality Improvement, Report to the Department of Environment and Natural Resources. Conics, Toowoomba

Cong, R.H. Wang, X.J., Xu, M.G., Zhang, W.J., Xie, L.J., Yang, X.Y., Huang, S.M., Wang, B. R. 2012. Dynamics of soil carbon to nitrogen ratio changes under long-term fertilizer addition in wheat-corn double cropping systems of China. European Journal of Soil Science, 63, 341–350.

Connolly, R.D., Ciesiolka, C.A.A., Silburn, D.M, Carroll, C., 1997. Distributed parameter hydrology model (Answers) applied to a range of catchment scales using rainfall simulator data. IV Evaluating pasture catchment hydrology. Journal of Hydrology, 201: 311-328.

Connolly, R.D., Silburn, D.M., Freebairn, D.M., 2002. Transport of sediment and pesticides in surface water. In: Kookana, R., Sadler, R., Sethunathan, N., Naidu, R. (Eds.), Environmental Protection and Risk Assessment of Organic Contaminants. Science Publishers Inc., Enfield, New Hampshire, pp. 143–167.

Cowie, B., Shaw, M., Di Bella, L., Benson, A., Nash, M., Davison, L., Tang, W. 2012. Runoff Loss of Herbicides Applied to Cane Trash and Bare Soil A Rainfall Simulation Study. Proc Internat Soc Sugar Cane Technol.

Daniel, A.J. and Greenway, M. 1995. Nutrient dynamics in two urban Melaleuca wetlands. Proceedings of National Conference on Wetlands for Water Quality Control, James Cook University, Townsville, 25-29 September, 343-360

Daily, G.C. 1997. Societal Dependence on Natural Ecosystems, Island Press, Washington DC.

Davis, L., Negus, P., MacKenzie-Smith, F. 2007. Understanding ecological and biophysical processes in Queensland wetlands: a literature review and gap analysis, Department of Natural Resources and Water.

Davis, A.M., Thorburn, P.J., Lewis, S.E., Bainbridge, Z.T., Attard, S.J., Milla, R., Brodie, J., 2013. Herbicide run-off dynamics of furrow irrigated sugarcane farms and associated drainage systems on the Burdekin River Floodplain, north-eastern Australia. Agric. Ecosys. Environ. doi:10.1016/j.agee.2011.06.019.

Dawes-Gromadzki, T. 2005. Bugs beneath the surface: the functional significance of soil macro-invertebrates to landscape health in Australia's tropical savannas. Insect Science 12 (4), 307–312.

DEEDI 2011. Department of Employment, Economic Development and Innovation, Wetland Management Handbook: Farm Management Systems (FMS) guidelines for managing wetlands in Intensive agriculture, Queensland Wetlands Program, Brisbane www.wetlandinfo.ehp.qld.gov.au

Dight, I., 2009. Burdekin Water Quality Improvement Plan, NQ Dry Tropics, Townsville http://www.nqdrytropics.com.au/burdekin-water-quality-improvement-plan-1

DPC, 2009. Reef Water Quality Protection Plan, 2009, For the Great Barrier Reef World Heritage Area and adjacent catchments. Queensland Department of Premiers and Cabinet. The State of Queensland and the Commonwealth of Australia. www.reefplan.qld.gov.au.

Drewry, J., Higham, W., Mitchell, C., 2008. Water Quality Improvement Plan: Final report for Mackay Whitsunday region. Reef Catchments Mackay Whitsunday Inc, Mackay (http://reefcatchments.com.au/wp-content/uploads/2012/05/MWNRM-WQIP-total-report-ultra-lo-res.pdf).

Ellis, T.W., Leguédois, S., Hairsine, P.B., Tongway, D.J. 2006. Capture of overland flow by a tree belt on a pastured hillslope in south-eastern Australia. Australian Journal of Soil Research, 44: 117-125, doi: 110.071/SR05130

Evans, R., Gilliam, J.W., Lilly, J.P. 1996. Wetlands and Water Quality. North Carolina Cooperative Extension Service, AG 473-7.

Faithful, J.W. 1997. The fate of phosphorus in wetlands: a review. Australia Centre for Tropical Freshwater Research.

Fawcett, R.S., Christensen, B.R., Tierney, D.P., 1994. The impact of conservation tillage on pesticide runoff into surface water: a review and analysis. Journal of Soil and Water Conservation 49, 126-135.

Finlayson, C.M., Cowie, I.D., and Bailey, B.J., 1993. Biomass and litter Dynamics in a Melaleuca forest on a seasonally inundated floodplain in tropical north Australia, Wetlands Ecology and Management, 56 (3), 269-77.

Fisher, J., and Acreman, M.C., 2004. Wetland nutrient removal: a review of the evidence. Hydrology and Earth System Science, Vol. 8, no. 4, pp. 673-85.

Freebairn, D.M., Loch, R.J., Silburn, D.M. 1996. Soil erosion and soil conservation for vertisols. Developments in Soil Science 24, 303-362.

Gordon, I.J., Nelson, B. 2007. Reef Safe Beef: Environmentally sensitive livestock management for the grazing lands of the Great Barrier Reef catchments. In: Swain, D.L., Charmley, E., Steel, J.W. & Coffey, S.G. (eds) Redesigning Animal Agriculture; the Challenge of the 21st Century. Pp. 171-184. CABI Publishing, Wallingford, UK.

Great Barrier Reef Marine Park Authority. 2012. Informing the outlook for Great Barrier Reef Marine Park Authority, Townsville.

Greenway, M., Daniel, A., 1997. Melaleuca wetlands for water storage and nutrient removal. Australian Water and Waste Water Association, Queensland Branch Meeting, Hervey Bay.

Greenway, M., Woolley, A., 1999. Constructed wetlands in Queensland: Performance efficiency and nutrient bioaccumulation. Ecological Engineering, 12, pp39-55.

Greiner, R., Gregg, D. 2011. Farmers' intrinsic motivations, barriers to the adoption of conservation practices and effectiveness of policy instruments: Empirical evidence from northern Australia. *Land Use Policy* 28(1), 257-265.

Greiner, R., Patterson, L., Miller, O. 2009. Motivations, risk perceptions and adoption of conservation practices by farmers. *Agricultural Systems* 99(2-3), 86-104.

Hawdon, A., Keen, R., Post, D., Wilkinson, S., 2008. Hydrologic Recovery of Rangeland following Cattle Exclusion. In: J. Schmidt *et al*. (Editors), Sediment Dynamics in Changing Environments (Proceedings of a symposium held in Christchurch, New Zealand, December 2008). IAHS Publ. 325. IAHS Press, pp. 532-539.

Heede, B.H., 1979. Deteriorated watersheds can be restored: A case study. Environmental Management 3, 271-281.

Higham, W., Drewry, J., Mitchell, C. 2008. Development of 'ABCD' nutrient management framework for water quality improvement at regional scale. Proc. Aust. Soc. Sugar. Cane Technol. 30, 2 pp.

House, A.P.N., MacLeod, N.D., Cullen, B., Whitbread, A.M., Brown S.D., McIvor, J.G. 2008. Integrating production and natural resource management on mixed farms in eastern Australia: The cost of conservation in agricultural landscapes. *Agriculture, Ecosystems and Environmen*t 127: 153–165.

Hughes, A.O., Olley, J.M., Croke, J.C., McKergow, L.A., 2009. Sediment source changes over the last 250 years in a dry-tropical catchment, central Queensland, Australia. Geomorphology, 104(3-4), 262-275.

Hunt, L.P., Petty ,S., Cowley, R., Fisher, A., Ash, A.J., MacDonald, N. 2007. Factors affecting the management of cattle grazing distribution in northern Australia: preliminary observations on the effect of paddock size and water points. Rangeland Journal, 29(2), 169-179.

Huth, N.I., Thorburn, P.J., Radford, B.J., Thornton, C.M., 2010. Impacts of fertilisers and legumes on N_2O and CO_2 emissions from soils in subtropical agricultural systems: a simulation study. Agric. Ecosys. Environ. 136, 351–357.

Hunter, H.M., Hairsine, P.B. 2002. Riparian and wetland buffers for water quality protection, Proceedings 2nd National Conference on Aquatic Environments: Sustaining Our Aquatic Environments – Implementing Solutions, Townsville, Australia, 20–23 November, 2001.

Hunter, H., Lukacs, G. 2000, LWRRDC project QPI nutrient control in irrigation drainage systems using artificial wetlands.

Hunter, H.M. 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 356, 131– 146

Hunter, H., Fellows, C., Rassam, D., DeHayr, R., Pagendam, D., Conway, C., Bloesch, P., Beard, N. 2006. Managing riparian lands to improve water quality: optimising nitrate removal via denitrification. Technical Report No. 57, Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management, Indooroopilly. http://www.coastal.crc.org.au/Publications/download-57Riparian.asp

Janssen, B.H., De Willigen, P., 2006. Ideal and saturated soil fertility as bench marks in nutrient management. 1. Outline of the framework. Agric. Ecosys. Environ. 116, 132–146.

Jones, R., 2005. The ecotoxicological effects of photosystem II herbicides on corals. Mar. Pollut. Bull. 51, 495–506.

Katsenovich, Y.P., Hummel-Batista, A., Raveinet, A.J., Miller, J.F. 2009. Ecological Engineering, Vol. 35 pp. 1529-1537.

Kennedy, K., Schroeder, T., Shaw, M., Haynes, D., Lewis, S., Bentley, C., Paxman, C., Carter, S., Brando, V., Bartkow, M., Mueller, J., 2012. Photosystem-II herbicides on the Great Barrier Reef- results from up to five years of monitoring and a preliminary comparison with remote sensing derived water quality parameters. Mar. Pollut. Bull.

Kragt, M. E., Roebeling, P. C., Ruijs, A. 2009. Effects of Great Barrier Reef degradation on recreational reef-trip demand: a contingent behaviour approach, *The Australian Journal of Agricultural and Resource Economics* 53: 213-229.

Kroon, F. 2008. Draft Tully Water Quality Improvement Plan. Report to Terrain NRM, CSIRO Water for a Healthy Country.

Kruse, E.G., Bucks, D.A., Von Bernuth, R.D. 1990 Comparison of Irrigation Systems. In: Irrigation of Agricultural Crops. No 30 in the Agronomy Series Am Soc of Agron pp 475-508.

Landsberg, R., Ash, A., Shepherd, R., Mckeon, G. 1998. Learning From History to Survive in the Future: Management Evolution on Trafalgar Station, North-East Queensland. The Rangeland Journal, 20: 104-118. DOI: http://dx.doi.org/10.1071/RJ9980104.

Leguédois, S., Ellis, T.W., Hairsine, P.B., Tongway, D.J. 2008. Sediment trapping by a tree belt: processes and consequences for sediment delivery. Hydrological Processes, 22: 3523-3534. DOI: 10.1002/hyp.6957.

Lewis, S.E., Brodie, J.E., Bainbridge, Z.T., Rohde, K., Davis, A., Masters, B., Maughan, M., Devlin, M., Mueller, J., Schaffelke, B., 2009. Herbicides: A new threat to the Great Barrier Reef. Environ. Pollut. 157, 2470-2484.

Lewis S.E., Bainbridge Z.T., Sherman B.S., Kuhnert P., Henderson B., Cooper M., Brodie J.E. 2013. Calculating sediment trapping efficiencies for reservoirs in tropical settings: a case study from the Burdekin Falls Dam, NE Australia. Water Resources Research, 49, 1017–1029 (doi: 10.1002/wrcr.20117).

Lizotte, R.E., Shields, F.D., Testa, S. 2012. Effects of a simulated agricultural runoff event on sediment toxicity in a managed backwater wetland, Water, Air and Soil Pollution, Vol. 223 (8), pp. 5375-5389.

Lu, H., Moran, C.J., Prosser, I.P., DeRose, R. 2004. Investment prioritization based on broadscale spatial budgeting to meet downstream targets for suspended sediment loads. Water Resources Research, 40: W09501, doi 09510.01029/02003WR002966.

Lukacs, G., 1998. Costal Freshwater wetlands of North Queensland – imperatives for their conservation. Australian Centre for Tropical Freshwater wetlands of North Queensland, James Cook University, Queensland

Maillard, E., Payraudeau, S., Ortiz, F., Imfeld, G. 2012. Removal of dissolved pesticide mixtures by a stormwater wetland receiving runoff from a vineyard catchment: An inter-annual comparison, International Journal of Environmental Analytical Chemistry Volume 92, Issue 8, Pages 979-994

Marecik, R., Bialas, W., Cyplik, P., Lawniczak, L., Chrzanowski, L. 2012. Phytoremediation potential of three wetland plant species toward atrazine in environmentally relevant concentrations. Polish Journal of Environmental Studies, 21, 697-702.

Markley, J., Refalo, B. 2011. Reducing the environmental impact and optimising nutrient requirements for mill mud applications in cane farming operations. Proc Aust Soc Sugar Cane Technol 33, 11 pp (on CD).

Marshall, N.A., Gordon, I.J., Ash, A.J., 2011 The reluctance of resource-users to adopt seasonal climate forecasts to enhance resilience to climate variability on the rangelands. Climate Change, 107: 511-529.

Masters, B., Rohde, K., Gurner, N., Higham, W., Drewry, J., 2008. Sediment, nutrient and herbicide runoff from cane farming practices in the Mackay Whitsunday region: a field-based rainfall simulation study of management practices. Queensland Department of Natural Resources and Water for the Mackay Whitsunday Natural Resource Management Group, Australia.

Masters, B., Rohde, K., Gurner, N., Reid, D. 2013. Reducing the risk of herbicide runoff in sugarcane through controlled traffic and early-banded application. Agric. Ecosys. Environ., in press (doi:10.1016/j.agee.2012.02.001).

MacLeod, N.D., Ash, A.J., McIvor, J.G. 2004. An economic assessment of the impact of grazing land condition on livestock performance in tropical woodlands. Rangeland Journal, 26, 49-71.

MacLeod, N. D. and McIvor, J. G. 2006. Reconciling economic and ecological conflicts for sustained management of grazing lands. *Ecological Economics* 56, 386-401.

McHugh, A.D., Bhattari, S., Lotz, G., Midmore, D.J. 2008. Effects of subsurface drip irrigation rates and furrow irrigation for cotton grown on a vertisol on off-site movement of sediments, nutrients and pesticides. Agron. Sustain. Dev. 28, 507-519.

McIvor, J.G., Ash, A.J., Cook, G.D. 1995a. Land condition in the tropical tallgrass pasture lands. 1. Effects on herbage production. Rangeland Journal, 17, 69-85.

McIvor, J.G., Williams, J., Gardener, C.J. 1995b. Pasture management influences runoff and soil movement in the semi-arid tropics. Aust. J. Exp. Agric. 35, 55-65.

McJannet, D., Wallace, J., Keen, R., Hawdon, A., Kemei, J., 2012. The filtering capacity of a tropical riverine wetland: II. Sediment and nutrient balances. Hydrological Processes, 26, 53-72.

McKergow, L.A, Prosser, I.P., Hughes, A.O., Brodie, J., 2005a. Sources of sediment to the Great Barrier Reef World Heritage Area. Marine Pollut. Bull. 51, 200-211.

McKergow, L.A., Prosser, I.P., Hughes, A.O., Brodie, J., 2005b. Regional scale nutrient modelling-exports to the Great Barrier Reef World Heritage Area. Mar. Pollut. Bull. 51, 186-199.

McKergow, L.A., Prosser, I.P., Grayson, R.B., Heiner, D. 2004. Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland 2. Water Quality. Australian Journal Soil Research 42, 485-498.

Millennium Ecosystems Assessment, 2005. Ecosystems and Human Wellbeing: Wetlands and Water synthesis. http://www.millenniumassessment.org/documents/document.358.aspx.pdf

Mitchell, A., Reghenzani, J., Faithful, J., Furnas, M., Brodie, J. 2009. Relationships between land use and nutrient concentrations in streams draining a 'wet-tropics' catchment in northern Australia. Mar. Freshwat. Res. 60, 1097–1108.

Moody, P.W. 2011. Environmental risk indicators for soil phosphorus status. Soil Research, 49, 247–252.

Moody, P.W., Aitken, R.L., 1996. Fertiliser management in tropical horticulture: Is it sustainable. In: Stephenson, R.A., Winks, C.W. (Eds), Challenges for Horticulture in the Tropics: Proc. 3rd Aust. Soc. Hort. Sci. and 1st Aust. Macadamia Soc. Res. Conf. Australian Society of Horticultural Science, Burnie, pp. 109-115.

Moody, P.W., Blogg, D., Legrand, J., Schroeder, B.L. and Wood, A.W. 2008. 'SCAMP' and 'SAFEGAUGE FOR NUTRIENTS': two new decision support tools for minimising off-site movement of nutrients. Proc Aust Soc Sugar Cane Technol 30: 275-294.

Murphy, T., Dougall, C., Burger, P., Carroll C., 2013. Runoff water quality from dryland cropping on Vertisols in Central Queensland, Australia. Agric. Ecosyst. Environ. 2012, doi:10.1016/j.agee.2011.07.023.

Murphy, T., Burger, P., Ryan. T. 2012. Paddock to Reef Integrated Monitoring Modelling and Reporting Program, Runoff and water quality from dryland grains cropping and grazing landuse at paddock and multiple catchment scales. Interim Report 2010-11 extended wet season. Queensland Department of Environment and Resource Management, Brisbane, Queensland, Australia.

O'Reagain, P., Bushell, J., Holloway, C., Reid, A., 2009. Managing for rainfall variability: effect of grazing strategy on cattle production in a dry tropical savanna, Animal Production Science, 49, 85 – 99.

O'Reagain, P., Bushell, J., Holmes, B. 2011. Managing for rainfall variability: long-term profitability of different grazing strategies in a northern Australian tropical savanna. Animal Production Science 51, 210-224.

Orr, D.M., Burrows W.H., Hendricksen R.E., Clem R.L., Back P.V., Rutherford M.T., Myles D.J., Conway M.J. 2010. Impacts of grazing management options on pasture and animal productivity in a Heteropogon contortus (black speargrass) pasture in central Queensland. 1. Pasture yield and composition. Crop & Pasture Science, 61: 170-181.

Orr, D.M. and O'Reagain, P.J., 2011. Managing for rainfall variability: impacts of grazing strategies on perennial grass dynamics in a dry tropical savanna. Rangeland Journal, 33(2): 209-220.

Orr, H.G., Carling, P.A. 2006. Hydro-climatic and land use changes in the river lune catchment, North West England, implications for catchment management. River Research and Applications 22, 239-255. DOI: 10.1002/rra.908.

Packett, R., Ford, P.W. and Noble, B. 2005. Overview of the delivery of pesticides to the Fitzroy Estuary and persistent organic pollutants in the food chain. In: Fitzroy in Focus: Coastal science for the Fitzroy region. In: B. Noble, A. Bell, P. Verwey and J. Tilden (eds). Coastal CRC for Coastal Zone, Estuary and Waterway Management. Indooroopilly, Brisbane.

Packett, R., Dougall, C., Rhode. K., Noble, R. 2009. Agricultural lands are hot-spots for annual runoff polluting the southern Great Barrier Reef lagoon. Marine Pollution Bulletin 58, 976-986.

Pannell, D. 2006. Flat earth economics: The far-reaching consequences of flat payoff functions in economic decision making. Review of Agricultural Economics 28, 553-566.

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F., Wilkinson, R. 2006. Understanding and promoting adoption of conservation practices by rural landholders. Australian Journal of Experimental Agriculture, 46, 1407 – 1424.

Park, S.E., Webster, A.J., Horan, H.L., James, A.T., Thorburn, P.J., 2010. Legume rotation crops lessen the need for nitrogen fertiliser throughout the sugarcane cropping cycle. Field Crop. Res. 119, 331-341.

Prayaga, P., Rolfe, J.C. and Stoekl, N. 2010. The value of recreational fishing in the Great Barrier Reef, Australia: A pooled revealed preference and contingent behaviour model, Marine Policy 34, 244-251.

Prosser, I.P., Slade, C.J. 1994. Gully formation and the role of valley-floor vegetation, southeastern Australia. Geology, 22: 1127-1130.

Prove, B. G., Doogan, V.J., Truong, P.N.V. 1995. Nature and magnitude of soil erosion in sugarcaneland on the wet tropical coast of north-eastern Queensland Australian Journal of Experimental Agriculture, 35, 64 1-9

Prove, B.G., Moody, P.W., Reghenzani, J.R., 1997. Nutrient balances and transport from agricultural land and rainforest lands: a case study in the Johnstone River catchment, DAQ3S Final Report. Queensland Department of Natural Resources and BSES, Queensland.

Queensland Wetlands Program 2011. Banana farming for healthier wetlands. A joint initiative of the Australian and Queensland Governments (www.wetlandinfo.ehp.qld.gov.au).

Queensland Wetlands Program (n.d.) Healeys Lagoon. . A joint initiative of the Australian and Queensland Governments www.wetlandinfo.ehp.qld.gov.au.

Randall, G., Delgado, J.A., Schepers, J.S., 2008. Nitrogen management to protect water resources. In: Schepers, J.S., Raun, W.R. (Eds.), Nitrogen in Agricultural Systems. Agron. Monograph. 49. Amer. Soc. Agron., Madison, WI, pp. 911-945.

Rasiah, V., Armour, J.D., Yamamoto, T., Mahendrarajah, S., Heiner, D.H., 2003. Nitrate dynamics in shallow groundwater and the potential for transport to off-site water bodies. Water Air Soil Pollut. 147, 183–202.

Rasiah, V., Moody, P.W. and Armour, J.D. 2011. Soluble phosphate in fluctuating groundwater under cropping in the north-eastern wet tropics of Australia. Soil Res. 49, 329–342.

Rasiah, V., Armour, J.D., Nelson, P.N., 2013. Nitrate in shallow fluctuating groundwater under sugarcane: coral reef health hazard and risk analysis. Agric. Ecosys. Environ. In press (http://dx.doi.org/10.1016/j.agee).

Reedy, K.R. and Gale, P.M. 1994. 'Wetland Process and Water quality: A Symposium Overview' in Journal of Environmental Quality, Vol. 23(5), pp. 875-877.

Renard, K.G., Foster, G.A., Weesies, D.K., McCool, D.K., Yoder, D.C. 1997. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation. Agriculture Handbook 703. United States Department of Agriculture.

Rochester, I.J., 2010. Assessing internal crop Nitrogen use efficiency in high-yielding irrigated cotton. Nutr. Cycl. Agroecosys. 90, 147-156.

Roebeling, P.C. 2006. Efficiency in Great Barrier Reef water pollution control: A case study for the Douglas Shire. Natural Resource Modeling 19, 539-566.

Roebeling, P.C., van Grieken, M.E., Webster, A.J., Biggs, J.S. and Thorburn, P.J. 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., Bush, A., Agnew, J. 2011. Paddock to sub-catchment scale water quality monitoring of sugarcane management practices: Interim report 2010/2011 wet season, Mackay Whitsunday region. Queensland Department of Resource Management. ISBN 9 311662 200501.

Rohde, K., Masters, B., Brodie, J., Faithful, J., Noble, R., Carroll, C., Shaw, C., Dekker, A. and Brando, V. 2006a. Fresh and marine water quality in the Mackay Whitsunday region, 2004–2005. Mackay Whitsunday Natural Resource Management Group, Mackay.

Rolfe, J., Gregg, D. 2012 Valuing Beach Recreation Across a Regional Area: The Great Barrier Reef in Australia, *Ocean and Coastal Management,* 69: 282 -290.

Rolfe, J., Windle, J. 2011a. Assessing community values for reducing agricultural emissions to improve water quality and protect coral health in the Great Barrier Reef, *Water Resources Research* 47 (12): doi:10.1029/2010WR010355.

Rolfe, J., Windle, J. 2011b. Using auction mechanisms to reveal costs for water quality improvements in Great Barrier Reef catchments in Australia. *Agricultural Water Management* 98: 493-501.

Rolfe, J., Windle, J. 2012 Distance decay functions for iconic assets: Assessing national values to protect the health of the Great Barrier Reef in Australia, *Environmental and Resource Economics*, 53(3): 347-365.

Rolfe, J., Greiner, R., Windle, J., Hailu, A. 2011. Testing for allocation efficiencies in water quality tenders across catchments, industries and pollutants: A north Queensland case study. *Australian Journal of Agricultural and Resource Economics 55(4): 518-536.*

Rosewell, C.J. 1993. SOILOSS - A program to assist in the selection of management practises to reduce erosion. Technical Handbook No. 11. Soil Conservation Service of NSW, Sydney.

Roth, C.H. 2004. A framework relating soil surface condition to infiltration and sediment and nutrient mobilization in grazed rangelands of northeastern Queensland, Australia. Earth Surface Processes And Landforms, 29: 1093-1104.

Rustomji, P., Zhang, S.P., Hairsine, P.B., Zhang, L. and Zhao, J., 2008. River sediment load and concentration responses to changes in hydrology and catchment management in the Loess Plateau region of China. Water Resources Research, 44: W00A04, doi:10.1029/2007WR006656.

Sallaway, M.M., Yule, D.F., Mayer, D., Burger, P.W., 1990. Effects of surface management on the hydrology of a vertisol in semi-arid Australia. Soil Till. Res. 15, 227–245.

Scanlon, J.C., Pressland, A.J., Myles, D.J., 1996. Runoff and soil movement on midslopes in north-east Queensland grazed woodlands. Rangeland J. 18, 33–46.

Schroeder, B.L., Wood, A.W., Moody, P.W., Bell, M.J., Garside, A.G., 2005. Nitrogen fertiliser guidelines in perspective. Proc. Aust. Soc. Sugar. Cane Technol. 27, 291-304.

Schroeder, B.L., Wood, A.W., Sefton, M., Hurney, A.P., Skocaj, D.M., Stainlay, T., Moody, P.W. 2010. District yield potential: An appropriate basis for nitrogen guidelines for sugarcane production. Proc. Aust. Soc. Sugar. Cane Technol 33, 193–209.

Shaw, M., Silburn, D.M., Thornton, C., Robinson, B., McClymont, D., 2011. Modelling pesticide runoff from paddocks in the Great Barrier Reef with HowLeaky. MODSIM 2011, Perth WA.

Shipitalo, M.J., Malone, R.W., Owens, L.B., 2008. Impact of glyphosate-tolerant soybean and glufosinate-tolerant corn production on herbicide losses in surface runoff. J. Environ. Qual. 37, 401–408.

Sieling, K., Kage, H., 2006. N balance as an indicator of N leaching in an oilseed rape–winter wheat–winter barley rotation. Agric. Ecosys. Environ. 115, 261–269.

Silburn, D.M., Hunter, H.M., 2009. Management practices for control of runoff losses from cotton furrows under storm rainfall. III. Cover and wheel traffic effects on nutrients (N and P) in runoff from a black Vertisol. Aust. J. Soil Res. 47, 221–233.

Silburn, D.M., Carroll, C., Ciesiolka, C.A.A., deVoil, R.C. and Burger, P., 2011. Hillslope runoff and erosion on duplex soils in grazing lands in semi-arid central Queensland. I. Influences of cover, slope, and soil. Soil Research, 49(2): 105-117.

Silburn, D.M., Foley, J.L., deVoil, R.C. 2013. Managing runoff of herbicides under rainfall and furrow irrigation with wheel traffic and banded spraying. Agriculture, Ecosystems and Environment, doi:10.1016/j.agee.2011.08.018.

Silburn, D.M., Foley, J.L., deVoil, R.C., 2013. Managing runoff of herbicides under rainfall and furrow irrigation with wheel traffic and banded spraying. Agric. Ecosys. Environ. doi:10.1016/j.agee.2011.08.018.

Silburn, D.M., Glanville, S.F., 2002. Management practices for control of runoff losses from cotton furrows under storm rainfall. I. Runoff and sediment on black Vertosol. Aust. J. Soil Res. 40, 1–20.

Silburn, D.M., Simpson, B.W., Hargreaves, P.A., 2002. Management practices for control of runoff losses from cotton furrows under storm rainfall. II. Transport of pesticides in runoff. Aust. J. Soil Res. 40, 21–44.

Silburn, D.M., Kennedy, I.R., 2007. Rain simulation to estimate pesticide transport in runoff. In: Kennedy, I.R., Soloman, K.R., Gee, S.J., Crossan, A.N., Wang, S., Sanchez-Bayo, F. (Eds.), Rational Environmental Management of Agrochemicals: Risk Assessment, Monitoring and Remedial Action. ACS Symposium Series No. 966. American Chemical Society, ISBN 978-0-8412-7420-4, pp. 120–135.

Simon, A., Collison, A.J.C. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. Earth Surface Proc. and Landforms 27, 527-546.

Smith, J., Douglas, G.B., Radke, L.C., Palmer, M. and Brooke, B.P., 2008. Fitzroy River Basin, Queensland, Australia. III. Identification of sediment sources in the coastal zone. Environmental Chemistry, 5(3): 231-242.

Star, M., Rolfe, J. and Donaghy, P. 2011. Economically viable land regeneration in Central Queensland and improved water quality outcomes for the Great Barrier Reef. *The Rangeland Journal* 33:267-276.

Star, M., Rolfe, J., Donaghy, P., Beutel, T., Whish, G. and Abbott, B. 2012. Targeting resource investments to achieve sediment reduction and improved Great Barrier Reef health. *Agriculture. Ecosystems and Environment* in press http://dx.doi.org/10.1016/j.agee.2012.03.016.

Stewart, L.K., Charlesworth, P.B., Bristow, K.L., Thorburn, P.J. (2006). Estimating deep drainage and nitrate leaching from the root zone under sugarcane using APSIM-SWIM. Agricultural Water Management, 81: 315-334.

Stoeckl, N., Hicks, C.C., Mills, M., Fabricius, K., Esparon, M., Kroon, F., Kaur, K. and Costanza, R. 2011. The economic value of ecosystem services in the Great Barrier Reef: Our state of knowledge. Annals of the New York Academy of Sciences 1219 (1): 113-133.

Stork, P.R., Jerie, P.H., Callinan, A.P.L., 2003. Subsurface drip irrigation in raised bed tomato production. I Nitrogen and phosphate losses under current commercial practice. Aust. J. Soil Res. 41, 1283-1304.

Stork, P.R., Lyons. D.J., 2012, Phosphorus loss and speciation in overland flow from a plantation horticulture catchment and in an adjoining waterway in coastal Queensland, Australia. Soil Research, 2012, 50, 515–525.

Strahan, R.,Hoffman, A. 2009. Estimating the economic implications for broad acre cropping farms in the Fitzroy Basin catchments of adoption of best management practices. Department of Employment, Economic Development and Innovation, Queensland, Australia.

Thomas, G.A., Standley, J., Hunter, H.M., Blight, G.W., Webb, A.A., 1990. Tillage and crop residue management affect Vertisol properties and grain sorghum growth over seven years in the semiarid sub-tropics. 3. Crop growth, water use and nutrient balance. Soil Tillage Res., 18: 367-388.

Thorburn, P.J., 1992. Structural and hydrological changes in a Vertisol under different fallow management techniques. Soil Till. Res. 23, 341-359.

Thorburn, P.J., Goodson, M. 2005. Overcoming on-farm constraints to productivity and profitability in a wet tropical area. Proc. Aust. Soc. Sugar. Cane Technol. 27, 158-169.

Thorburn, P.J., Wilkinson, S.N. 2013. Conceptual frameworks for estimating the water quality benefits of improved agricultural management practices in large catchments. Agriculture Ecosystems and Environment, in press (doi:10.1016/j.agee.2011.12.021).

Thorburn, P.J., Dart, I.K., Biggs, I.M., Baillie, C.P., Smith M.A., Keating, B.A., 2003b. The fate of nitrogen applied to sugarcane by trickle irrigation. In: Thorburn, P.J., Bristow, K.L., Annandale, J. (Eds.), Micro-irrigation: Advances in system design and management. Irrig. Sci. 22, 201-209.

Thorburn, P.J., Park, S.E., Bloesch, P., Webster, A.J., Horan, H.L., Biggs, I.M., Biggs, J.S., 2008. Potential impacts of mill mud on nitrogen fertiliser requirements for sugarcane production. Proc. Aust. Soc. Sugar Cane Technol. 30, 367–368.

Thorburn, P.J., Biggs, J.S., Collins, K., Probert, M.E. (2010). Using the APSIM model to estimate nitrous oxide emissions from diverse Australian sugarcane production systems. Agriculture Ecosystems and Environment, 136: 343–350.

Thorburn, P.J., Meier, E.A., Probert M.E., 2005. Modelling nitrogen dynamics in sugarcane systems: Recent advances and applications. Field Crops Res. 92, 337-352.

Thorburn, P.J., Biggs, J.S., Attard, S.J., Kemei, J., 2011a. Environmental impacts of fully irrigated sugarcane production: Nitrogen lost through runoff and leaching. Agric. Ecosys. Environ. 144, 1-12.

Thorburn, P.J., Webster, A.J., Biggs, J.S., Biggs, I.M., 2011b. Meeting global environmental challenges of sugarcane production through an innovative nitrogen fertiliser management system. Plant Soil 339, 51-67.

Thorburn, P.J., Jakku, E., Webster, A.J., Everingham, Y.L. 2011c. Agricultural decision support systems facilitating co-learning: a case study on environmental impacts of sugarcane production. International Journal of Agricultural Sustainability, 9, 322-333.

Thorburn, P.J., Wilkinson, S.N., Meier, E.A., 2011d. Prioritising practice changes in Reef Rescue, CSIRO Water for a Healthy Country National Research Flagship Report. CSIRO, Canberra.

Thorburn, P.J. (1992). Structural and hydrological changes in a Vertisol under different fallow management techniques. Soil and Tillage Research 23: 341-359.

Thorburn, P.J., Biggs, J.S., Weier, K.L., Keating B.A., 2003a. Nitrate in groundwaters of intensive agricultural areas in coastal northeastern Australia. Agric. Ecosys. Environ. 94, 49-58.

Thorburn, P.J., Webster, A.J., Biggs, I.M., Biggs, J.S., Park, S.E., Spillman, M.F. 2007. Innovative management of nitrogen fertiliser for a sustainable sugar industry. Proc. Aust. Soc. Sugar. Cane Technol. 29, 85–96.

Thorburn, P.J., Wilkinson, S.N. 2013. Conceptual frameworks for estimating the water quality benefits of improved land management practices in large catchments Agric. Ecosys. Environ. In press doi: 10.1016/j.agee.2011.12.021.

Thorburn, P.J., Park, S.E., Biggs, I.M., 2003c. Nitrogen fertiliser management in the Australian sugar industry: strategic opportunities for improved efficiency. Proc. Aust. Soc. Sugar Cane Technol. 25, 12pp.

Thornton, C.M., Cowie, B.A., Freebairn, D.M., Playford, C.L., 2007. The Brigalow Catchment Study: II. Clearing brigalow (*Acacia harpophylla*) for cropping or pasture increases runoff. Australian Journal of Soil Research, 45, 496-511.

Valdez-Zamudio, D., Guertin, D.P., 2000. Soil erosion studies in buffelgrass pastures, Land Stewardship in the 21st Century: The Contributions of Watershed Management, Conference Proceedings. USDA, Ft Collins, pp. 282-286.

Valentin, C., Poesen. J., Li. Y. 2005. Gully erosion: Impacts, factors and control. Catena, 63: 132-153. DOI: 10.1016/j.catena.2005.06.001.

van Grieken, M., Thomas, C., Roebeling, P., Thorburn, P. 2011a. Integrating economic drivers of social change into agricultural water quality improvement strategies. Agric. Ecosys. Environ. In press doi:10.1016/j.agee.2011.06.013.

van Grieken, M., Lynam, T., Coggan, A., Whitten, S., Kroon, F. 2011b Cost effectiveness of design-based water quality improvement regulations in the Great Barrier Reef. Agric. Ecosys. Environ. In press doi:10.1016/j.agee.2011.08.010.

Waters, D.K., Carroll C., Ellis, R., Hateley L., McCloskey J., Packett R., Dougall C., Fentie B. 2013. Modelling Reductions of pollutant Loads due to improved management practices in the Great Barrier Reef Catchments – Whole of GBR ,Volume 1. Department of Natural Resources and Mines. Technical Report (ISBN: 978-1-7423-0999)

Wauchope, R.D. 1978. The pesticide content of surface water draining from agricultural fields—a review. Journal of Environmental Quality 7, 459–472.

Wilcox, B.P., Huang, Y.U.N., Walker, J.W. 2008. Long-term trends in streamflow from semiarid rangelands: uncovering drivers of change. Global Change Biology, 14, 1676-1689.

Weber, J.B., Wilkerson, G., Reinhardt, C. 2004. Calculating pesticide sorption coefficients (Kd) using selected soil properties. Chemosphere 55, 157-166.

Webster, A.J., Thorburn, P.J., Bartley, R. 2010. The impact of reduced nitrogen fertiliser application rates on nitrogen loads to surface water. In: Hogarth, D.M. (ed.) International Society Sugar Cane Technologists Proceedings XXVIIth Congress, Veracruz, March 2010 (on CD), 8 pp.

Webster, A.J., Bartley, R. Armour, J.D., Brodie, J.E., Thorburn, P.J. (2012). Reducing dissolved inorganic nitrogen in surface runoff water from sugarcane production systems. Marine Pollution Bulletin, 65, 128–135.

WetlandCare Australia 2008. Wetland Rehabilitation Guidelines for the Great Barrier Reef catchment. Compiled for Department of the Environment, Water, Heritage and the Arts.

Wilkinson, S.N., Hancock, G.J., Bartley, B., Hawdon, A.A., Keen, R.J., 2013. Using sediment tracing to assess processes and spatial patterns of erosion in grazed rangelands, Burdekin River basin, Australia. Agric. Ecosys. Environ. In press doi:10.1016/j.agee.2012.02.002.

Wood, A.W., 1991. Management of crop residues following green harvesting of sugarcane in north Queensland. Soil Till. Res. 20, 69-85.

Wood, A.W., Schroeder, B.L., Stewart, R.L. 2003 The development of site-specific nutrient management guidelines for sustainable sugarcane production. Proc. Aust. Soc. Sugar. Cane Technol 25 (on CD), 12 pp.