Australian Marine Conservation Society

Ms Imogen Zethoven, Director of Strategy

Inquiry into the identification of leading practices in ensuring evidence-based regulation of farm practices that impact water quality outcomes in the Great Barrier Reef

Responses to questions on notice, public hearing, 28 July 2020

Thank you again for the opportunity to speak to the Senate Committee this morning.

As promised, I am getting back to you about a number of matters that came up during my session.

1. Regarding the question from Sen Canavan, AMCS's turnover is published online in our Annual Reports.

2. Regarding the question from Sen McDonald: what is the value of the agricultural sector in the GBR catchment and what is the number of jobs in the industry? Regarding the number of jobs, I refer to the RIS by the Queensland Government into the Reef protection regulations, which found that there are approximately 13,000 producers (consisting of approximately 8,500 graziers and 4,500 growers i.e., sugarcane, horticulture, bananas and grains) operating in the GBR catchment. In regards to the value of agriculture, I refer to the GBR 2019 Outlook Report (p.159), which found that in 2016–17, the gross value of agricultural production in Queensland was $14 billion, with approximately half derived from agriculture within the GBR Catchment.

3. I also attach two published papers in regards to a question by Senator Waters that was asked of my colleague from WWF about international examples of regulation.

Please do not hesitate to contact me if you have any questions. Thank you again for the opportunity to contribute to the Committee's deliberations.

Yours sincerely

Imogen Zethoven

Global Change Biology

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RESEARCH REVIEW

Towards protecting the Great Barrier Reef from land-based pollution

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Abstract

The Great Barrier Reef (GBR) is an iconic coral reef system extending over 2000 km along the north-east coast of Australia. Global recognition of its Outstanding Universal Value resulted in the listing of the 348 000 km^2 GBR World Heritage Area (WHA) by UNESCO in 1981. Despite various levels of national and international protection, the condition of GBR ecosystems has deteriorated over the past decades, with land-based pollution from the adjacent catchments being a major and ongoing cause for this decline. To reduce land-based pollution, the Australian and Queensland Governments have implemented a range of policy initiatives since 2003. Here, we evaluate the effectiveness of existing initiatives to reduce discharge of land-based pollutants into the waters of the GBR. We conclude that recent efforts in the GBR catchments to reduce land-based pollution are unlikely to be sufficient to protect the GBR ecosystems from declining water quality within the aspired time frames. To support management decisions for desired ecological outcomes for the GBR WHA, we identify potential improvements to current policies and incentives, as well as potential changes to current agricultural land use, based on overseas experiences and Australia's unique potential. The experience in the GBR may provide useful guidance for the management of other marine ecosystems, as reducing land-based pollution by better managing agricultural sources is a challenge for coastal communities around the world.

Keywords: agriculture, diffuse pollution, land use, management, marine ecosystem, policy, water quality

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Introduction

The Great Barrier Reef (GBR) is the world's largest coral reef system, extending over 2000 km along the north-east coast of Australia (Fig. 1). It consists of a variety of tropical marine habitats including ~20 000 km^2 of coral reefs, ~43 000 km^2 of seagrass meadows and extensive mangrove forests (Great Barrier Reef Marine Park Authority, 2014). Following community protests over the potential granting of mining applications for oil and gas in the 1960–1970s (Mccalman, 2013), the 344 400-km^2 GBR Marine Park was established under the Federal *Great Barrier Reef Marine Park Act* 1975 (Department of the Environment, 2015b). This Act provides 'for the long term protection and conservation of the environment, biodiversity and heritage values of the Great Barrier Reef Region', with other uses and activities allowed, if consistent with the main object of the Act (Table S1) (Department of the Environment, 2015b). Global recognition of its Outstanding Universal Value, that is 'natural significance which is so exceptional as to transcend national boundaries and to be of common importance for present and future generations of all humanity', followed in 1981 with the listing of the 348 000 km^2 GBR World Heritage Area (WHA) by UNESCO (1981). In Australia, World Heritage Properties such as the GBR WHA are protected under the Federal *Environment Protection and Biodiversity Conservation Act* 1999 as a matter of 'national environmental significance' (Table S1; Department of the Environment, 2015a). In addition, the GBR is protected directly and indirectly by many other federal, state and local government laws and policies in Australia's federal system of government that regulate activities affecting the reef (Mcgrath, 2010; Jacobs Group (Australia) Pty Limited, 2014). The GBR currently contributes an estimated AUS $5.6 billion to the Australian economy, comprising tourism (AUS$5.2 billion), commercial fishing (AUS $160 million) and recreational use (AUS$244 million), and supporting employment of approximately 69 000 full-time positions (Deloitte Access Economics, 2013). These estimates are likely too low as they do not include other ecosystem uses and services that have not yet been quantified (Stoeckl *et al*., 2011).

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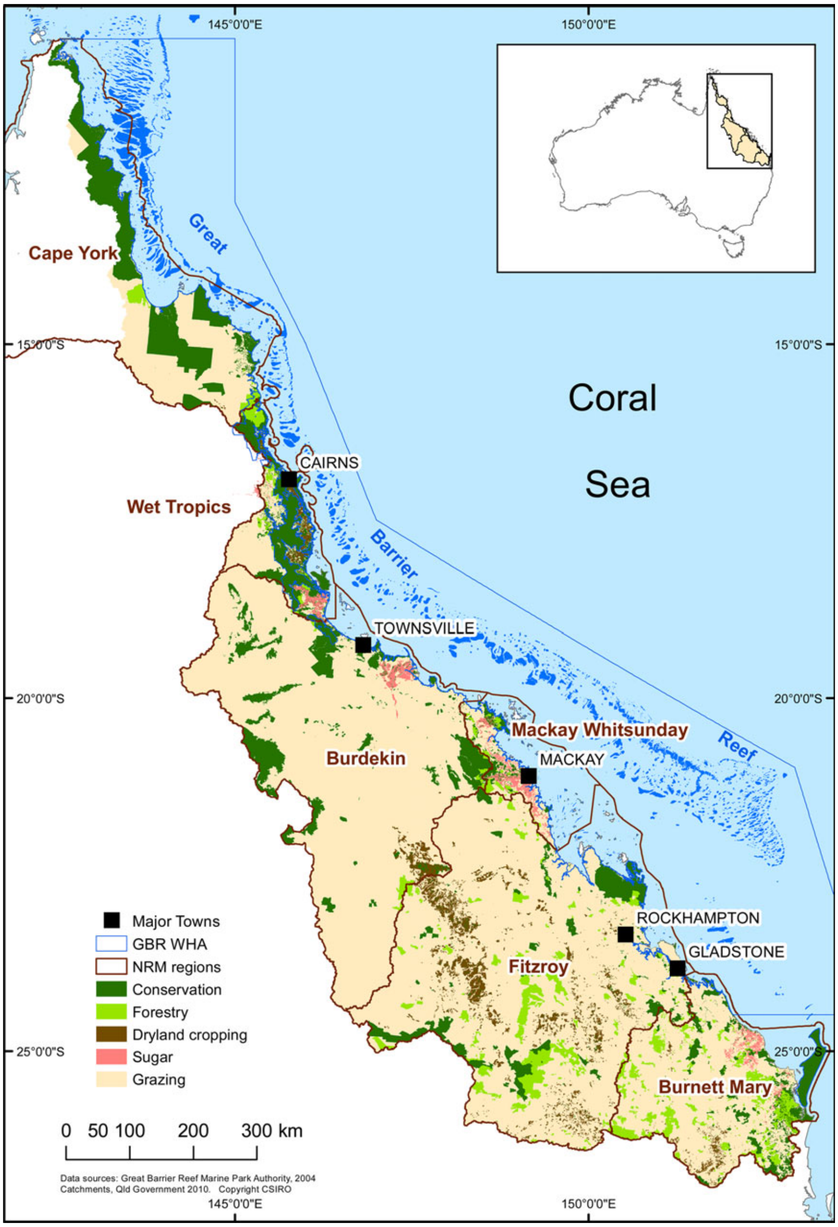

Fig. 1 The extent of the Great Barrier Reef World Heritage Area, and the main land uses on the adjacent catchments, including rangeland cattle grazing (75% of the area), nature conservation (13%), forestry (5%), dryland cropping (~2%) and sugarcane (~1%) (proportions from Waters *et al.* (2014)). The catchment area encompasses six natural resource management (NRM) regions, which are managed by regional NRM bodies responsible for protecting and managing Australia's natural resources.

Despite the high level of national and international protection, the condition of GBR ecosystems has deteriorated over the past decades (Great Barrier Reef Marine Park Authority, 2014). Poor water quality from land-based pollution, exacerbated by extreme weather events, continues to be a major pressure to the GBR (Brodie *et al.*, 2013) and is expected to amplify the impacts of climate change on the future condition of the GBR (Great Barrier Reef Marine Park Authority, 2014). Here, we present evidence on (i) the decline in GBR water quality and ecosystem condition, and (ii) the effectiveness of recent government policy initiatives to reduce discharge of land-based pollutants into GBR waters. Specifically, we examine the effectiveness of the bilateral Reef Water Quality Protection Plan (Reef Plan), the main policy initiative of the Australian and Queensland Governments for protecting the GBR from land-based pollutants since 2003 (Reef Water Quality

Protection Plan Secretariat, 2013b). We conclude that the main pathway of Reef Plan implementation, that is voluntary Best Management Practice (BMP) programmes in existing agricultural land uses, is insufficient to achieve Reef Plan's targets (Thorburn & Wilkinson, 2013; Waters *et al.*, 2014; Department of Environment and Heritage Protection, 2016) and to protect the GBR from declining water quality (Brodie *et al.*, 2009; Kroon, 2012; Wooldridge *et al.*, 2015; Table 1). Furthermore, the threat of land-based pollution is likely to be exacerbated by current policy settings allowing an increase in a variety of pressures (Table S1a, b), including (but not limited to) proposed developments of water resources to support increased agricultural production (Commonwealth of Australia, 2015a). To increase the likelihood of protecting GBR ecosystems from land-based pollution into the future, we identify a combination of improvements to current incentives and future opportunities that may also provide guidance for the management of other marine ecosystems around the world facing similar threats from land-based pollution.

Status and trends of GBR water quality and ecosystem condition

Similar to other coral reef ecosystems around the world (Gardner *et al.*, 2003; Bruno & Selig, 2007), key GBR habitats and species have shown severe declines in abundance and condition over the past decades (Great Barrier Reef Marine Park Authority, 2014). For example, coral cover declined by 50% for the whole GBR, and by 70% along the developed central and southern GBR over the last 27 years (De'ath *et al.*, 2012). Similarly, in the coastal and inshore areas of the GBR, the condition of water quality, corals and seagrasses has declined over the last decade (Fig. 2a, b) after a series of extreme wet seasons (Thompson *et al.*, 2014a,b; Queensland Government, 2015b). Populations of many other species such as sharks and rays, sea snakes, marine turtles, seabirds, dolphins and dugongs have declined significantly, again particularly in central and southern inshore areas, and are expected to further decrease into the future (Great Barrier Reef Marine Park Authority, 2014). The overall decline in GBR ecosystem condition is generally considered to be caused by the cumulative impacts of climate change, poor water quality in inshore areas from land-based run-off, large-scale modification of coastal habitats and fishing pressure (Great Barrier Reef Marine Park Authority, 2014).

Several lines of evidence indicate that water quality in the GBR lagoon along the developed central and southern sections has declined, following large-scale land clearing and associated agricultural development in the adjacent catchments since European settlement in the 1850s (Fig. 3a, b), and the subsequent modification of terrestrial pollutant fluxes (Brodie *et al.*, 2012). River loads to the GBR lagoon are estimated to have increased substantially for suspended sediment (3–6 times), nitrogen (2–6 times), phosphorus (3–9 times) and pesticides (~17 000 kg) (Kroon *et al.*, 2012; Waters *et al.*, 2014). Most of the GBR catchment area is used for agricultural production, namely rangeland cattle grazing (75% of the area), forestry (5%), dryland cropping (~2%) and sugarcane (~1%), with irrigated cropping, horticulture, dairy and bananas each covering less than <1% (Waters *et al.*, 2014; Fig. 1). The remaining area comprises nature conservation (13%) and small (~1%) urban and mining areas (Brodie *et al.*, 2012; Waters *et al.*, 2014). Excess river sediment, nutrient and pesticide loads to the GBR lagoon are derived from (i) surface and subsurface erosion, predominantly in rangeland cattle grazing settings; (ii) fertilizer applications in sugarcane and broad-acre cropping; and (iii) pesticides (particularly photosystem II inhibiting herbicides) primarily applied during sugarcane cultivation (Kroon *et al.*, 2013). Signatures of increased river loads are observed in GBR coral cores for sediment since the 1900s (Mcculloch *et al.*, 2003; Lewis *et al.*, 2007) and for nutrients since the mid-20th century (Jupiter *et al.*, 2008; Mallela *et al.*, 2013) and are associated with increased soil erosion and fertilizer application. Coral and sediment cores from the inshore GBR also show evidence of changes in community composition of corals (Roff *et al.*, 2013) over the last century, indicative of higher water turbidity and increased nutrient availability in the GBR lagoon. Time series of GBR water quality are only available since the early 1990s, but recent analyses show clear correlations between periods of increased river run-off and water turbidity and/or higher nutrient availability (Fabricius *et al.*, 2014; Thompson *et al.*, 2014a,b). Frequent exceedances of water quality guidelines for sediment, nutrient and pesticides are reported for the GBR lagoon (Gallen *et al.*, 2014; Thompson *et al.*, 2014a).

A 2013 scientific consensus statement on *Land use impacts on GBR water quality and ecosystem condition* (Brodie *et al.*, 2013) found that the greatest impacts of degraded water quality are as follows: (i) higher nitrogen availability that increases phytoplankton biomass and promotes outbreaks of the destructive coral-eating crown-of-thorns seastar, and proliferation of macroalgae on inshore reefs that compete with corals for space, and (ii) fine sediments and associated particulate nutrient inputs, leading to reduced light availability for

Table 1 Progress of the Reef Water Quality Protection Plan (Reef Plan) and the Reef 2050 Long-Term Sustainability Plan (Reef 2050 LTSP) is measured against a goal or objective (a), water quality targets (b) and land management and catchment targets (c), based on a 2009 baseline (Reef Water Quality Protection Plan Secretariat, 2009, 2013b; Commonwealth of Australia, 2015b). Reef Plan 2003 did not set targets (The State of Queensland and Commonwealth of Australia, 2003)

(a)

Plan	Goal or objective	Time frame
Reef Plan 2003	Halt and reverse the decline in water quality entering the Reef within 10 years	2003
Reef Plan 2009	Halt and reverse the decline in water quality entering the Reef by 2013	2013
	To ensure that the quality of water entering the Reef from adjacent catchments has no detrimental impact on the health and resilience of the Great Barrier Reef	2020
Reef Plan 2013	The quality of water entering the Reef from adjacent catchments has no detrimental impact on the health and resilience of the Reef	2020
Reef 2050 LTSP	Over successive decades the quality of water entering the Reef from broadscale land use has no detrimental impact on the health and resilience of the Great Barrier Reef	2035

(b)

Plan	2013 target		2018 target				2020 target
Reef Plan 2009	A minimum 50 per cent reduction in nitrogen and phosphorus loads at the end of catchments	A minimum 50 per cent reduction in pesticides at the end of catchments	–	–	–	–	A minimum 20 per cent reduction in sediment load at the end of catchments
Reef Plan 2013	–	–	At least a 50 per cent reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen loads in priority areas	At least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment and particulate nutrients in priority areas	At least a 60 per cent reduction in end-of-catchment pesticide loads in priority areas	–	–
Reef 2050 LTSP	–	–	At least a 50 per cent reduction in anthropogenic end-of-catchment dissolved inorganic nitrogen loads in priority areas, on the way to achieving up to an 80 per cent reduction in nitrogen by 2025	At least a 20 per cent reduction in anthropogenic end-of-catchment loads of sediment in priority areas, on the way to achieving up to a 50 per cent reduction by 2025	At least a 60 per cent reduction in end-of-catchment pesticide loads in priority areas	At least a 20 per cent reduction in anthropogenic end-of-catchment loads of particulate nutrients in priority areas	

Table (continued)

(c)

Plan	2013 target	2018 target				2020 target
Reef Plan 2009	A minimum of 50 per cent late dry season groundcover on dry tropical grazing land	–	–	–	–	–
Reef Plan 2013	–	90 per cent of sugarcane, horticulture, cropping and grazing lands are managed using best management practice systems (soil, nutrient and pesticides) in priority areas	Minimum 70 per cent late dry season groundcover on grazing lands	The extent of riparian vegetation is increased	There is no net loss of the extent, and an improvement in the ecological processes and environmental values, of natural wetlands	–
Reef 2050 LTSP	–	90 per cent of sugarcane, horticulture, cropping and grazing lands are managed using best management practice systems (soil, nutrient and pesticides) in priority areas	Minimum 70 per cent late dry season groundcover on grazing lands	The extent of riparian vegetation is increased	There is no net loss of the extent, and an improvement in the ecological processes and environmental values, of natural wetlands	–

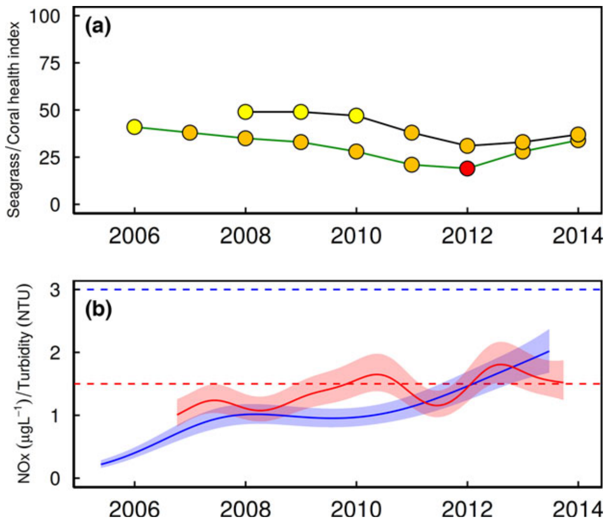

Fig. 2 Decline in coral (black line) and seagrass (green line) health indicators (a), and trend in the concentration of NO_x (nitrate and nitrate, blue line) and turbidity (red line) (b) with shaded areas representing the 95% confidence intervals of those trends, in the inshore area of the Great Barrier Reef World Heritage Area (GBR WHA) from 2005 to 2014. The dashed lines represent the water quality guideline values for these variables (Department of Environment and Heritage Protection, 2009; Great Barrier Reef Marine Park Authority, 2010). Modified from data reported by Thompson *et al.* (2014a) and Queensland Government (2015b). Note: The coral health index aggregates data for the attributes coral cover, macroalgal cover, density of juvenile corals and the rate of coral cover increase. The seagrass index aggregates data for the attributes seagrass abundance, reproductive effort and carbon–nitrogen ratio in seagrass leaves. The indices are for the inshore areas of the central and southern sections (coral) and the whole (seagrass) GBR WHA. Red = very poor, orange = poor, yellow = moderate, light green = good, dark green = very good. The data used for the water quality trend estimates are from water sample collections conducted three times per year at 20 fixed sites (NO_x) and continuous instrument records from 14 fixed sites (turbidity).

photosynthesis of inshore seagrasses and coral reefs. The statement recommended that ongoing effort is required to improve water quality in river run-off from the GBR catchments to enhance the resilience of GBR ecosystems to other disturbances, such as increasing sea temperatures, ocean acidification and extreme weather events (Brodie *et al.*, 2013).

The Reef 2050 Long-Term Sustainability Plan

In response to the request of the World Heritage Committee for a coordinated and comprehensive long-term plan to restore and protect the Outstanding Universal Value of the GBR WHA (UNESCO, 2014), the Australian and Queensland Governments released the

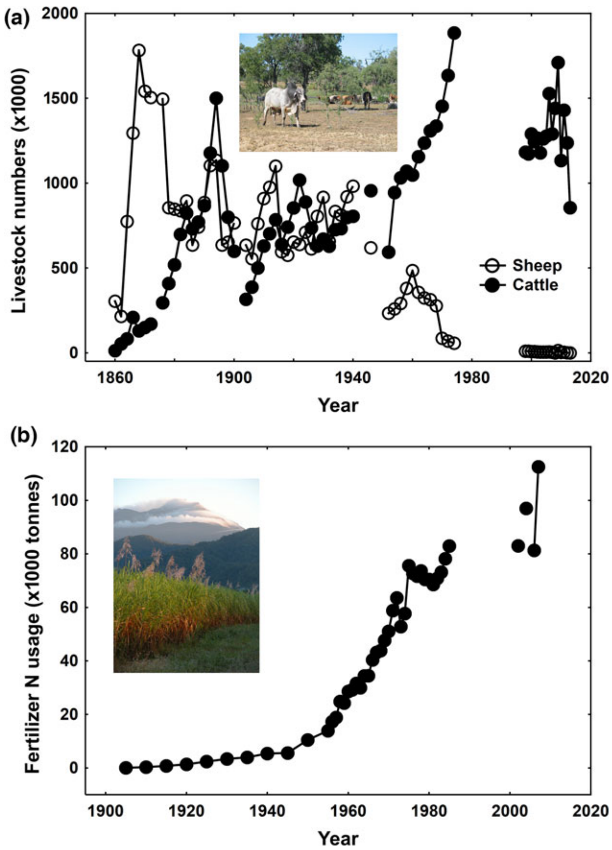

Fig. 3 Examples of changes in agricultural commodities and land management in the Great Barrier Reef (GBR) catchments since European settlement in the late 1850s. Total number of sheep and meat cattle (a) in the Burdekin region from first introduction in the 1860s to 2012–2013, and usage of nitrogen fertilizer (b) in the GBR catchments from 1910 to 2012–2013. Modified from data reported in Lewis *et al*. (2007), Pulsford (1993), and the Australian Bureau of Statistics (2014a,b). Note that there are periods of missing data. Photos © CSIRO.

Reef 2050 Long-Term Sustainability Plan (Reef 2050 LTSP) in March 2015 (Commonwealth of Australia, 2015b). The Reef 2050 LTSP sets out to address the key risks to the GBR, namely climate change, land-based pollution, coastal land-use change and direct use. The 2035 objective for the Reef 2050 LTSP for land-based pollution is 'over successive decades the quality of water entering the Reef from broadscale land use has no detrimental impact on the health and resilience of the Great Barrier Reef', with associated targets for water quality and for land and catchment management (Table 1a, b). The Reef Plan, the primary policy initiative of the Queensland and Australian Governments to protect the GBR from land-based pollutants since 2003 (The State of Queensland and Commonwealth of Australia, 2003; Reef Water Quality Protection Plan Secretariat, 2009, 2013b), is the main foundational programme of the Reef 2050 LTSP to achieve this objective and associated targets. Given its integral and continuing role in managing land-based pollution in the GBR catchments, we next present the development and implementation of the Reef Plan in more detail and discuss the evidence of its effectiveness in reducing discharge of land-based pollutants into GBR waters.

The Reef Water Quality Protection Plan

To protect the GBR from diffuse source pollution from agricultural land uses, the Queensland and Australian Governments jointly released the Reef Plan in 2003 (The State of Queensland and Commonwealth of Australia, 2003). The Reef Plan 2003 was revised and updated in 2009 (Reef Water Quality Protection Plan Secretariat, 2009) and in 2013 (Reef Water Quality Protection Plan Secretariat, 2013b), with new goals and associated water quality targets and land management and catchment targets (The State of Queensland and Commonwealth of Australia, 2003; Reef Water Quality Protection Plan Secretariat, 2009, 2013b; Table 1a–c). The development and implementation of each of the three Reef Plans have been informed by scientific syntheses and consensus statements (Baker, 2003; Brodie *et al*., 2008, 2013). Overall, the Reef Plan aims to provide a coordinated and collaborative approach to improve water quality through industry-led BMP programmes (Queensland Government, 2015c) that describe steps farmers can take to improve water quality, and through programmes such as Reef Trust (Department of the Environment, 2015c) that provide incentives and extension activities to support voluntary adoption of BMPs. Other programmes target research, planning and regulations, as well as 'systems repair' activities supporting wetland and riparian restoration (Reef Water Quality Protection Plan Secretariat, 2014). Both the Queensland and Australian Governments have supported the implementation of the Reef Plan through (i) investing AUS$375 million for Reef Plan 2009 from 2008 to 2013 (Reef Water Quality Protection Plan Secretariat, 2013b; Commonwealth of Australia, 2014), and (ii) expected investments of AUS$575 million in Reef water quality initiatives from 2015 to 2020 (The Great Barrier Reef Water Science Taskforce, 2015). Progress towards meeting the Reef Plan goals and targets continues to be evaluated by a Paddock to Reef Monitoring, Modelling and Reporting Program (Carroll *et al*., 2012) and reported via Reef Report Cards (Queensland Government, 2015a). The latest Reef Report Card for 2014 reported trends from a 2009 baseline, including (i) increased uptake of BMPs, (ii) continued overall loss of wetlands and riparian areas, (iii) modelled (but not measured) reductions in terrestrial pollutant loads entering the

GBR and (iv) continued poor condition of the inshore marine environment (Queensland Government, 2015a, b). While progress was made, the 2013 targets set for water quality and for land and catchment management (Table 1b, c) have not been met (Queensland Government, 2015a). Reef Plan 2013 states that to achieve its goal by 2020 (Table 1a), sustained and greater effort will be needed, including transformational changes in some farming technologies (Reef Water Quality Protection Plan Secretariat, 2013b).

Achieving the Reef Plan goal and targets for reduction in land-based pollution

In GBR grazing lands, management practices that control surface and subsurface erosion are well understood (Thorburn & Wilkinson, 2013). Practices centre on targeted grazing and vegetation management to reduce soil erosion, such as (i) maintaining ground cover and pasture biomass during the dry season and drought years to control hillslope erosion, (ii) increasing the proportion of deep-rooted native perennial grasses to control gully erosion and (iii) retaining trees in riparian areas to stabilize and reduce erosion from stream banks (Thorburn & Wilkinson, 2013; Bartley *et al.*, 2014a,b). In GBR cropping lands, the primary path to reducing nitrogen losses is through reducing excess application of nitrogen fertilizer by better matching fertilizer inputs to yields (Thorburn & Wilkinson, 2013) of intensive crops like sugarcane (Thorburn *et al.*, 2011a), bananas (Armour *et al.*, 2013) and cotton (Rochester, 2011). Technologies such as seasonal climate forecasting (Thorburn *et al.*, 2011b) and precision agriculture (Bramley *et al.*, 2008) may have a role to play in better matching fertilizer inputs to yields. Use of enhanced efficiency fertilizers may also help farmers reduce nitrogen losses, although the agronomic and environmental benefits of these fertilizers have been inconsistent in sugarcane and other crops (Verburg *et al.*, 2014). Practices that reduce pesticide losses from GBR cropping lands involve improved spatial and temporal targeting of applications, as well as minimizing run-off and movement of sediment containing pesticides (Thorburn *et al.*, 2013b).

Achieving the widespread or complete adoption of established or potential BMPs (e.g. as described by Thorburn & Wilkinson (2013)) required to reduce land-based pollution is a substantial socio-economic challenge (Thorburn *et al.*, 2013a; Van Grieken *et al.*, 2013b; Rolfe & Gregg, 2015). A considerable body of research suggests that landholders who adopt BMPs that maintain their pastures in better condition or apply fertilizer and pesticides at reduced rates are likely to generate better financial returns (Ash *et al.*, 1995; O'reagain *et al.*, 2011; Van Grieken *et al.*, 2014; Star *et al.*, 2015). However, several factors may lead to farmers deciding not to consider the adoption of BMPs into their business. In grazing lands, seasonal economic drivers such as low cash flow commonly experienced by graziers during droughts may encourage overgrazing and reduced ground cover during droughts (Thorburn & Wilkinson, 2013; Star *et al.*, 2015). In sugarcane, aiming to achieve maximum yield to increase sugar mill profitability may encourage overapplication of fertilizers and pesticides (Van Grieken *et al.*, 2013a). For all agricultural industries, transitioning from existing management practices to those with reduced pollutant exports can involve substantial upfront capital costs associated with acquiring new infrastructure, and transaction costs in exploring practice changes, accessing support and planning and executing changes (Pannell & Vanclay, 2011; Coggan *et al.*, 2014; Rolfe & Gregg, 2015). In grazing, it can also involve a substantial opportunity cost of lost production during the several years it will take for pasture cover and composition improvements to occur. Overall, these costs mean that many, but not all, farmers face a likely reduction in production and profitability in the short term and, in some cases, the long term when transitioning to BMPs (Thorburn *et al.*, 2013b; Department of Environment and Heritage Protection, 2016). This range in financial outcomes, together with variation and uncertainty in water quality outcomes, leads to a wide variety in pollutant abatement costs associated with different BMPs in both grazing (Fig. 4a) and cropping (Fig. 4b) across the GBR catchments (Thorburn *et al.*, 2013b; Department of Environment and Heritage Protection, 2016). The challenge of BMP uptake is clearly exhibited in the 2014 Reef Plan Report Card (Queensland Government, 2015a), reporting mostly poor to very poor progress towards the 2018 land management targets (Table 1c), despite the significant public investment to date (The Great Barrier Reef Water Science Taskforce, 2015). The variability in cost-effectiveness across industries, BMPs and landscapes indicates, however, that progress towards achieving Reef Plan targets and goals can be accelerated through spatial prioritization of investment in water quality initiatives (Star *et al.*, 2015; Department of Environment and Heritage Protection, 2016).

The potential reductions in sediment, nutrient and pesticide discharges, resulting from the adoption of BMPs in agricultural land uses across the GBR catchments, have been assessed using both mechanistic catchment (Waters *et al.*, 2014) and empirical (Thorburn & Wilkinson, 2013) modelling. The former is the basis for assessing progress towards Reef Plan targets for reductions in river loads of sediment, nutrients and pesticides due to practice uptake (Queensland

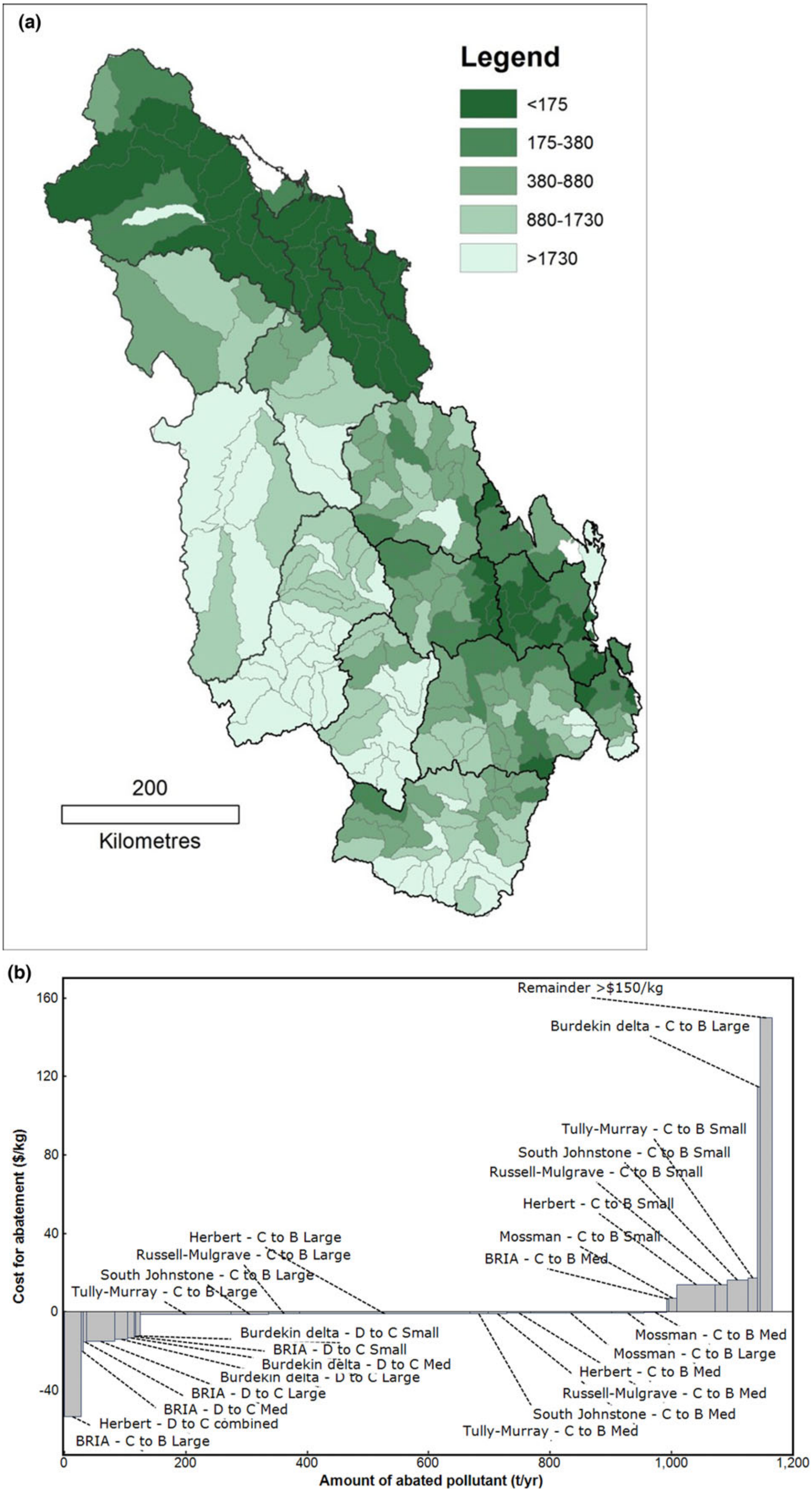
(a)
Legend
<175
175-380
380-880
880-1730
>1730
200
Kilometres
(b)
Cost for abatement ($/kg)
Amount of abated pollutant (t/yr)
Remainder >$150/kg
Burdekin delta - C to B Large
Tully-Murray - C to B Small
South Johnstone - C to B Small
Russell-Mulgrave - C to B Small
Herbert - C to B Small
Mossman - C to B Small
BRIA - C to B Med
Herbert - C to B Large
Russell-Mulgrave - C to B Large
South Johnstone - C to B Large
Tully-Murray - C to B Large
Burdekin delta - D to C Small
BRIA - D to C Small
Burdekin delta - D to C Med
Burdekin delta - D to C Large
BRIA - D to C Large
BRIA - D to C Med
Herbert - D to C combined
BRIA - C to B Large
Mossman - C to B Med
Mossman - C to B Large
Herbert - C to B Med
Russell-Mulgrave - C to B Med
South Johnstone - C to B Med
Tully-Murray - C to B Med
160
120
80
40
0
-40
0
200
400
600
800
1,000
1,200

Fig. 4 Spatial variability in pollutant abatement costs of best management practice (BMP) implementation to reduce sediment export from grazing lands in the Burdekin and Fitzroy catchments (in AUS$ per tonne) (a), and nitrogen export from sugarcane for different farm sizes and catchments in the Wet Tropics and Burdekin NRM regions (AUS$ per kilogram) (b). Costing baseline is 2015 from which adoption of a suite of BMPs are costed. For sediment export in the Fitzroy and Burdekin catchments a reduction in stocking rate shifts land condition from 'C' or current practice to 'B' representing best practice (a), and for cane reducing nitrogen application rates and surface application has different costs across catchments and farm sizes from 'D' or dated practice to 'C' current practice and 'B' best practice (b). Both figures sourced with some adaptation from Department of Environment and Heritage Protection (2016).

Government, 2015a). Empirical modelling for dissolved nitrogen suggests that complete adoption of 'cutting edge' (i.e. beyond BMPs) nitrogen management practices would reach water quality targets (Thorburn & Wilkinson, 2013), although the mechanistic modelling suggests reductions would not be as great (Waters *et al.*, 2014). Assessing true progress towards the targets is more difficult because practices may or may not be fully adopted, they may only be trialled or adopted for a limited period of time, or they may be modified with unintended reduced environmental benefits (Pannell & Vanclay, 2011). Both modelling approaches, however, suggest that even complete adoption of industry-supported BMPs for reducing sediment and nitrogen discharges from the GBR catchments would not achieve water quality targets stipulated in government policy for these two pollutants (Thorburn & Wilkinson, 2013; Waters *et al.*, 2014). The situation is more optimistic for pesticides, with predictions that the full adoption of BMPs will reduce discharge of pesticides from the GBR catchments to the levels specified in targets (Waters *et al.*, 2014).

In summary, the principles of BMPs, Reef Plan's main instrument to reduce diffuse source pollution from existing agricultural land uses, are well understood in the GBR catchments (Thorburn & Wilkinson, 2013). Furthermore, cost abatement analyses indicate progress towards achieving Reef Plan targets and goals can be accelerated through spatial prioritization of investment in water quality initiatives (Star *et al.*, 2015; Department of Environment and Heritage Protection, 2016). However, such change will be costly and complete uptake of BMPs is unlikely to achieve Reef Plan's sediment and nutrient targets in most GBR catchments (Thorburn & Wilkinson, 2013; Waters *et al.*, 2014; Department of Environment and Heritage Protection, 2016). Importantly, recent analyses indicate that the Reef Plan water quality targets, even if they were achieved, will not fully protect GBR ecosystems from exposure to land-based pollutants (Brodie *et al.*, 2009, 2014; Kroon, 2012; Brodie & Lewis, 2014; Wooldridge *et al.*, 2015). The new water quality targets for the Reef 2050 LTSP require larger reductions in nutrients and sediment by 2025 (Table 1b) (Commonwealth of Australia, 2015b); however, it is unclear how these targets will be achieved with the implementation of Reef Plan being the main foundational programme to address land-based pollution. Hence, innovative approaches and practices are needed in addition to BMP implementation to protect the Outstanding Universal Value of the GBR WHA from land-based pollution.

Future directions for improving GBR water quality and ecosystems

The challenge of delivering Reef Plan targets in the GBR catchments is illustrated by slow progress leading to adjustments of previous Reef Plan goals and targets since 2003, and recent development of the more stringent 2035 objectives in the Reef 2050 LTSP (Table 1a–c). The situation in the GBR is not unique, with recent reviews providing recommendations to reduce excessive or inappropriate input to coastal marine ecosystems of nitrogen and phosphorus from diffuse sources such as agriculture (Galloway *et al.*, 2008; Vitousek *et al.*, 2009; Canfield *et al.*, 2010; Elser & Bennett, 2011). Demonstrated effective approaches to reduce agricultural pollution to coastal waters do exist, as highlighted in a recent review on decreased fluxes of sediment and nutrients at the end of river and associated declines in nutrient concentrations and algal biomass in receiving coastal waters (Kroon *et al.*, 2014). These recommendations and approaches comprise two main components: (i) identifying management practices and/or land uses with acceptable pollutant export rates and (ii) having effective incentives for the adoption of these practices and/or land uses. Above we described the current incentives and management practices for existing agricultural land uses in the GBR catchments. In this section, we consider examples of improvements that can be made to current incentives, as well as potential transformational changes to current agricultural land uses to protect the GBR ecosystems from land-based pollution.

Combining different policy instruments to reduce diffuse pollution from agricultural land uses

Voluntary instruments such as grants programmes to farmers to implement BMPs provide the basis for the implementation of the Reef Plan (Mcgrath, 2010; Reef Water Quality Protection Plan Secretariat, 2014). While improved targeting of these programmes to areas generating disproportionally large amounts of pollutants would very likely deliver either improved outcomes, or equivalent outcomes at lower cost (Van Grieken *et al.*, 2013a), recent evaluations show that there are no cheap or easy ways of delivering the scale of change required to protect the GBR from diffuse sources of pollution (Van Grieken *et al.*, 2013a; Department of Environment and Heritage Protection, 2016). Furthermore, voluntary programmes alone are unlikely to deliver the scale of change required to protect the GBR (Van Grieken *et al.*, 2013a; The Great Barrier Reef Water Science Taskforce, 2015), as has been concluded elsewhere in Australia and overseas (Gunningham & Sinclair, 2004; Cary & Roberts, 2011; Roberts & Craig, 2014). These programmes can nevertheless provide a powerful stimulus to change, especially if part of a wider policy mix.

Globally, the only significant reductions in agricultural pollution to coastal ecosystems have been achieved through legislation and regulation supported by long-term political commitment (e.g. China, Denmark) or by the combined effects of declining economic subsidies, fertilizer use and livestock numbers following the collapse of the Soviet Union (e.g. several rivers in eastern Europe) (Kroon *et al.*, 2014). In Denmark, for example, five national action plans were implemented and enforced to regulate nitrogen fertilizer use over two decades (Kronvang *et al.*, 2008; Windolf *et al.*, 2012). Each action plan was underpinned by a variety of descriptive, and sometimes quantitative, policy measures, such as (i) nitrogen fertilizer application to crops at 90% of economic optimum and (ii) retirement and decrease in arable land (Windolf *et al.*, 2012). More generally across Europe, the European Union Nitrates Directive is the main regulation to reduce the environmental impacts of diffuse source pollution from agricultural land uses (Van Grinsven *et al.*, 2012). Implementation of the Directive since 1995 has contributed to reductions in (i) nitrate concentrations in fresh surface waters in north-west Europe (Van Grinsven *et al.*, 2012), and (ii) nitrogen and phosphorus river loads delivered to the North Sea (Grizzetti *et al.*, 2012). While specific management actions implemented in these examples may not be directly applicable to the GBR catchments, they indicate that targeted regulatory policy approaches can greatly enhance the protection of downstream aquatic ecosystems from land-based pollution.

There has been some consideration of regulation to address the continued discharge of poor water quality from GBR catchments into the lagoon (Brodie *et al.*, 2008; Harvey *et al.*, 2014). In 2009, the Queensland Government shifted to a risk-based regulatory framework through the *Great Barrier Reef Protection Amendment Act* 2009 under the *Environmental Protection Act* 1994 to 'reduce the impact of agricultural activities on the quality of water entering the reef' (Department of Environment and Heritage Protection, 2015) (Table S1). However, after 3 years the Queensland Government stopped enforcing this framework, apparently without any assessment of its efficacy (Harvey *et al.*, 2014; Queensland Audit Office, 2015). Risk-based regulatory approaches have the potential to provide the 'stick' that will support other voluntary and incentive based approaches, albeit they are not intended to prohibit or otherwise preclude existing agricultural activities. According to Wulf (2004), the Australian Government also has the power under the *Great Barrier Reef Marine Park Act* 1975 (Department of the Environment, 2015b) and the *Environment Protection and Biodiversity Conservation Act* 1999 (Department of the Environment, 2015a) to control land-based pollution into the GBR Marine Park and WHA, but has to date not applied this provision. Several studies have emphasized the lack of effective legislative and regulatory instruments governing agricultural land uses and management in catchments discharging into the GBR WHA (Wulf, 2004; Mcgrath, 2010; Jacobs Group (Australia) Pty Limited, 2014), highlighting the opportunity to address land-based pollution using such instruments.

Harmonization of multisectoral policies to protect GBR water quality

The environmental legal system that regulates land-use planning and development in the GBR catchments is complex, with 26 Federal and Queensland Government acts and regulations being directly relevant to the management of the GBR WHA, including the protection and management of its water quality (Jacobs Group (Australia) Pty Limited, 2014; Commonwealth of Australia, 2015b). A number of the current government acts, regulations and policies that affect land-based pollution and GBR water quality are inconsistent and do not align with the objective and targets of the Reef 2050 LTSP (Table S1). Three examples of this misalignment are as follows. First, the Federal government is proposing the development of water resources to support increased agricultural production in northern Queensland (Commonwealth of Australia, 2015a), including one of the catchments discharging into the GBR lagoon (the Burdekin). Such increases would likely work against the recently reported reductions in land-based pollution to the GBR lagoon (Queensland Government, 2015a) and are counter to recent marginal abatement analyses that imply some reduction in the production

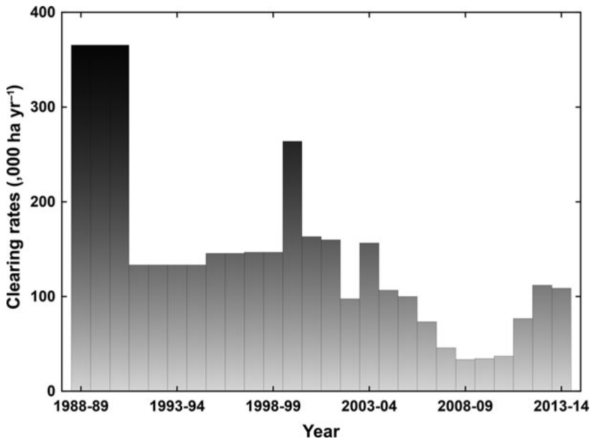

Fig. 5 Annual average clearing rates of remnant and non-remnant woody vegetation in the Great Barrier Reef (GBR) catchments, from 1989/1999 to 2013/2014 (Department of Science Information Technology Innovation and the Arts, 2014; Department of Science Information Technology and Innovation, 2015; Queensland Audit Office, 2015). Note: The Queensland Vegetation Management Act was passed in 1999 and commenced in 2000 to 'regulate the clearing of vegetation', including improvement of land management in the GBR catchments (Reef Water Quality Protection Plan Secretariat, 2005; Department of Natural Resources and Mines, 2015a). Following a spike in clearing in 1999/2000 due to policy changes, clearing rates in the GBR catchments decreased substantially to less than 50 000 ha yr^{-1} in 2007/2011 (Department of Science Information Technology Innovation and the Arts, 2014). Clearing rates have since increased almost fourfold, from 31 000 ha yr^{-1} in 2008/2009 to 109 235 ha yr^{-1} in 2012/2013 (Department of Science Information Technology and Innovation, 2015; Queensland Audit Office, 2015). This rise coincided with the reform of the vegetation management framework by the Queensland Government in 2013 (Department of Natural Resources and Mines, 2015b), to 'support the development of high value agriculture in areas with appropriate land and available water' (Queensland Government Cabinet and Ministerial Directory, 2013). Statistics for Cape York NRM region include both west and eastern drainages, but comprise only a small proportion (<7% maximum) of annual average clearing rates in the GBR catchments.

of current agricultural commodities is required to achieve larger reductions in pollutant export (Department of Environment and Heritage Protection, 2016). Second, recent amendments to Queensland's *Vegetation Management Act* 1999 (Department of Natural Resources and Mines, 2015a,b) to support agricultural development coincided with an almost fourfold increase in woody vegetation clearing rates in the GBR catchments (Queensland Audit Office, 2015; Fig. 5), likely promoting soil erosion and sediment run-off to the GBR lagoon (Department of Science Information Technology Innovation and the Arts, 2014). Finally, the provision of some forms of drought assistance to graziers by the Federal and Queensland Governments can generate incentives to manage properties in ways that may result in overgrazing and consequently increase the likelihood of sediment erosion (Productivity Commission, 2009; Mccoll & Young, 2010). For example, fodder subsidies may increase stock retention, and other financial subsidies may reduce the incentives for enterprise restructuring to better manage drought impacts on pasture and natural resources generally (Productivity Commission, 2009). Addressing these and other inconsistencies amongst Federal and Queensland acts, regulations and policies provide a considerable opportunity to improve the protection of GBR ecosystems from land-based pollution by assessing their effectiveness in protecting the Outstanding Universal Value of the GBR WHA.

Potential new agricultural products and land uses

Experience both globally (Kroon *et al*., 2014) and in the GBR (Thorburn & Wilkinson, 2013; Waters *et al*., 2014; Department of Environment and Heritage Protection, 2016) suggests that a move beyond traditional agricultural systems is needed to achieve sufficient improvements in water quality to protect the condition of coastal and marine ecosystems. However, not all new land uses which may deliver reduced pollutant loads are desirable, or necessarily permanent. For example, nutrient fluxes in several eastern European rivers decreased following the collapse of the Soviet Union and associated declines in economic subsidies to agriculture, fertilizer use and livestock numbers (Kroon *et al*., 2014). While this was a positive for water quality in the region, the local socio-economic impacts were likely undesirable. Further, fertilizer use and livestock numbers have increased again following the recovery and consolidation of agricultural production (Kraemer *et al*., 2011). Disruptions to agricultural systems have also occurred in the GBR catchments, albeit not in the context of water quality improvement. For example, sheep grazing was introduced to the Burdekin in the 1860s with sheep numbers well over 0.5 million up to the 1940s (Lewis *et al*., 2007; Fig. 3a). Sheep grazing has now all but disappeared (Fig. 3a) due to a combination of environmental (e.g. drought, disappearance of suitable feeding grasses) and market (e.g. drop in wool demand) forces (Lewis *et al*., 2007). Similarly, the Queensland tobacco-growing industry completely disappeared and transformed into sugarcane and mixed horticulture following deregulation of tobacco growing by the Australian Government in the 1990s (Griggs, 2002). Finally, the extent of sugarcane production in the GBR catchments has been controlled by Governments, with policy changes allowing the doubling of sugarcane producing area in the second half of the 20th century, particularly in the 1990s (Griggs, 2011). Sugarcane is now the dominant land use on coastal floodplains in the Wet Tropics, Burdekin and Mackay–Whitsunday regions (Fig. 1), with the expansion contributing to the substantial increase in nitrogen fertilizer application (Fig. 3b). These examples demonstrate that policy, economic and environmental conditions can and do result in substantial changes in land use over time. The challenge therefore is to identify new agricultural products or land uses that are compatible with societal expectations and will be environmentally sustainable.

Products or land uses that are able to effectively harness the built, natural, social, financial and human capitals in the GBR catchments are more likely to succeed. Cattle grazing now dominates land use with ~5.2 million cattle in the GBR catchment area (Australian Bureau of Statistics, 2014b; Fig. 1). The introduction of hoofed animals in the 1860s (Fig. 3a), in combination with grazing patterns different from native animals, is likely to have initiated excess erosion (Lewis *et al.*, 2007). Since then, a reduction in native vegetation cover, conversion to pasture and poor soil condition, and associated changes in freshwater flow regimes, have all contributed to increase soil erosion in the GBR catchments (Thorburn *et al.*, 2013b; Bartley *et al.*, 2014a). Although the principles of reducing erosion in GBR grazing lands are well understood (Thorburn & Wilkinson, 2013; Thorburn *et al.*, 2013b), there are short-term socio-economic impediments to adoption of more conservative management of grazing (as described above) to achieve longer term benefits of increased native ground cover (that reduces erosion) and productivity (Thorburn *et al.*, 2013b). Conservative grazing management could be supported by valuing other products derived from these lands. For example, management to increase pasture biomass may increase carbon stocks in soils (Allen *et al.*, 2013) which could be eligible for financial support under the Australian Government's Greenhouse Gas Emissions Reduction Fund (Department of the Environment, 2014). Likewise, reafforestation of riparian areas to reduce stream bank erosion could be eligible. Thus, current carbon sequestration policy could support the maintenance and rehabilitation of ground cover in grazing lands and be of financial benefit to graziers. Reducing soil erosion in grazing lands could also be supported through a change in grazing animals (away from hoofed animals) and grazing patterns, for example by promoting the commercial harvesting of native grazing fauna such as kangaroos (Grigg, 2002; Ampt & Baumber, 2006; Department of the Environment, 2013; Queensland Government, 2014).

Intensive cropping such as sugarcane, bananas and cotton only comprises a relatively small area of the GBR catchment area (Fig. 1), but contributes a disproportionally large amount to the current inorganic nitrogen load to the GBR lagoon (Kroon *et al.*, 2012, 2013; Waters *et al.*, 2014). Similar to reducing erosion in grazing lands, the principles of managing nitrogen fertilizer to reduce nitrogen export in GBR cropping lands are well understood (Thorburn & Wilkinson, 2013; Thorburn *et al.*, 2013b). However, farmers are reluctant to reduce fertilizer applications because of the perceived risk of having crop yields limited by nitrogen stress and the relatively small financial costs of overapplying nitrogen (Thorburn *et al.*, 2013b). New markets that may transform the products derived from cropping lands may deliver incentives to reduce fertilizer application. For example, sugarcane is a globally important feedstock for biofuel because of its net reduction in greenhouse gas emissions when ethanol produced from sugarcane is used instead of fossil fuels (De Vries *et al.*, 2010). Nitrous oxide emissions from soils are the largest source of greenhouse gas emissions in sugarcane production (Macedo *et al.*, 2008; Thorburn *et al.*, 2009; De Vries *et al.*, 2010; Renouf *et al.*, 2010). Nitrogen fertilizers stimulate nitrous oxide emissions so avoiding over-application of nitrogen fertilizer is important to increase the value of biofuel production (Thorburn *et al.*, 2009; Renouf *et al.*, 2010). In the GBR catchments, government policies that supported biofuel as the dominant product of sugarcane would thus facilitate lower nitrogen fertilizer inputs to (and hence losses from)

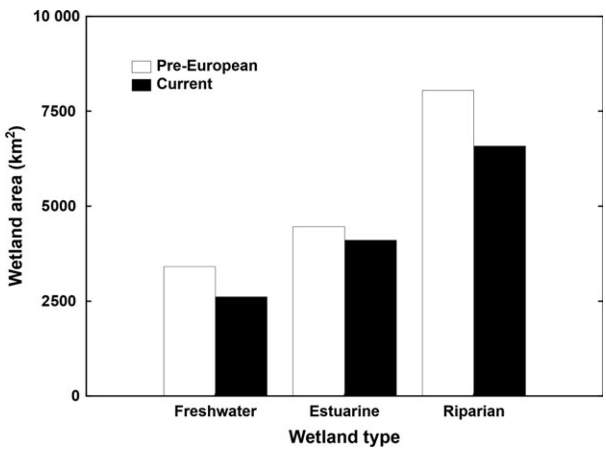

Fig. 6 Areas of three different types of wetlands in the GBR catchments, prior to European settlement in the late 1850s and current (Reef Water Quality Protection Plan Secretariat, 2013a; Department of Environment and Heritage Protection, 2014); for wetland definitions, see Department of Environment and Resource Management (2011).

sugarcane farming (Thorburn & Wilkinson, 2013; Thorburn *et al.*, 2013b). The long-term reduction of nitrogen export to the GBR lagoon could also be supported by replacing current high-input crops with other crops suited to the climate and soils of the GBR catchments but requiring lower nitrogen fertilizer input. A range of broad-acre and horticultural crops that meet these requirements are currently grown in the GBR catchments, such as grains, cereals and low input tree crops (e.g. macademias) (Thorburn & Wilkinson, 2013), and promotion of these through policy intervention would reduce nitrogen exports to the GBR lagoon.

Other options to deliver reductions in land-based pollution involve the hydrological restoration of landscapes. In the GBR catchments, coastal development has substantially altered natural river flow regimes through large-scale land clearing, wetland drainage, construction of barriers to flow and surface water diversion (Great Barrier Reef Marine Park Authority, 2014; Lough *et al.*, 2015). Across the GBR catchment area, 23% of vegetated freshwater wetlands, 8% of estuarine wetlands and 18% of riparian wetlands have been lost from pre-European extent (Fig. 6; Reef Water Quality Protection Plan Secretariat, 2013a; Department of Environment and Heritage Protection, 2014). These reductions in wetland extent are spatially variable; for example, loss of vegetated freshwater wetlands in the northern Cape York region is <1%, but it is >50% in the Wet Tropics region (Department of Environment and Heritage Protection, 2014). Despite the implementation of the Reef Plan, overall loss of wetlands and riparian areas in the GBR catchments continued between 2009 and 2013 (Queensland Government, 2015a). These modifications of the ecological filtering and buffering capacity of landscapes, and associated altered flow regimes, contribute to more sediment and nutrients entering the GBR lagoon (Kroon *et al.*, 2013; Great Barrier Reef Marine Park Authority, 2014; Lough *et al.*, 2015).

Comprehensive programmes of hydrological restoration of landscapes can deliver large reductions in land-based pollution (Walling & Fang, 2003; Chu *et al.*, 2009; Mclellan *et al.*, 2015). Globally, substantial effort is going into re-establishing environmental flows (Postel & Richter, 2003) and improving the ecological filtering and buffering capacity of landscapes (Bernhardt *et al.*, 2005). Removal of small dams and weirs in headwater catchments has resulted in the formation of new river channels, restored riparian vegetation and improved fish passage and spawning habitat within a year (Stanley & Doyle, 2003; Rood *et al.*, 2005; O'connor *et al.*, 2015). The removal of larger dams has also demonstrably improved flow and fish passage, sometimes within weeks (Service, 2011; Lovett, 2014; O'connor *et al.*, 2015). Establishing more natural drainage and vegetation patterns is expected to further increase hydraulic, sediment and nutrient residence times and enhance the opportunity for landscape mitigation of terrestrial pollutant fluxes (Walling & Fang, 2003; Burt & Pinay, 2005; Mclellan *et al.*, 2015). For example, restoration of the Kissimmee River in Florida, USA, resulted in hydrological processes recovering towards prechannelization conditions within 10 years (Anderson, 2014). The restoration or creation of wetlands (Verhoeven *et al.*, 2006) and riparian zones (Tomer & Locke, 2011) can result in full recovery of nitrogen storage and cycling processes within 25–30 years (Moreno-Mateos *et al.*, 2012), while restoration of native seagrass (Mcglathery *et al.*, 2012) or oyster beds (Schulte *et al.*, 2009) is likely to contribute to deposition and retention of suspended sediment, nutrient cycling and water filtration (Cloern, 2001; Mcglathery *et al.*, 2012) and may significantly reduce concentrations in receiving waters (Cerco & Noel, 2010). Consistent with international experience, GBR catchment modelling shows that improved riparian protection, including riparian fencing, can contribute significantly to sediment load reductions (Waters *et al.*, 2014). The capacity of GBR wetlands to improve water quality is less well understood, and not yet quantified (Thorburn *et al.*, 2013a), but is likely to play a role in improving ecosystem buffering capacity. In comparison with international efforts, however, the spatial scale of current restoration efforts in the GBR catchments (Commonwealth of Australia, 2014) is not commensurate with the scale of catchment degradation affecting GBR water quality and ecosystem condition (Great Barrier Reef Marine Park Authority, 2014).

Reducing land-based pollution through the restoration of hydrological function is unlikely to be achieved without the retirement of at least some agricultural land (Mclellan *et al.*, 2015). Indeed, retirement of agricultural land has been used as a tool for reducing diffuse pollution, for example by discontinuing production on land areas with highly erodible soils or requiring high fertilizer input in environmentally sensitive (i.e. high risk) areas (Frisvold, 2004; Bennett, 2008; Stokstad, 2008; Windolf *et al.*, 2012). The spatial scales at which agricultural land retirement is implemented can range from retiring land along water courses to establish riparian buffer zones, to converting specific areas of agricultural land to wetlands, to complete reforestation or afforestation of former grazing and cropping lands. In the USA, the establishment of the Conservation Reserve Program (CRP) in 1985 resulted in over 13 million ha (10%) of croplands taken out of production by 2004 (Frisvold, 2004). While not a primary objective, land retirement under the CRP was found to significantly reduce surface water pollution (Frisvold, 2004; Tomer & Locke, 2011). Subsequent programmes were established to, for example, restore cropland to wetlands and retire agricultural land for conservation purposes (Frisvold, 2004). Combined, this resulted in 90% of all US conservation spending going to land retirement by 2000. In China, large-scale soil conservation programmes implemented in the Yellow River (~17 000 km^2) and the Yangtze River (≥84 000 km^2) since the late 1950s have become effective in reducing river sediment loads since the late 1970s (Chu *et al*., 2009). These programmes included land terracing, tree and grass planting and construction of sediment trapping dams (Chu *et al*., 2009), with afforestation contributing almost 20% to the total reduction in sediment inputs into the Yellow River (Walling & Fang, 2003). The more recent Sloping Land Conversion Program, with an investment of over US$40 billion, aimed to convert 14.67 million ha of cropland to forests by 2010 (Bennett, 2008). Overall, retirement of agricultural land can be a cost-effective measure to reduce diffuse pollution, with appropriately targeted efforts generating sufficient water quality benefits to outweigh the costs (Ribaudo *et al*., 1994; Yang *et al*., 2003; Luo *et al*., 2006; Liu *et al*., 2013). As such, it warrants serious consideration as an additional tool to reduce land-based pollution into the GBR WHA.

Conclusions: protecting the GBR into the future

Coastal marine ecosystems around the world have been transformed due to changes in terrestrial fluxes of freshwater, sediment, nutrients and other contaminants resulting from anthropogenic disturbances (Carpenter *et al*., 1998; Lotze *et al*., 2006). The GBR WHA is no exception, with degradation of water quality and ecosystem condition linked with increases in land-based run-off of suspended sediment, nutrients and pesticides since the 1850s (Brodie *et al*., 2013; Great Barrier Reef Marine Park Authority, 2014). To halt and reverse the decline in reef water quality, the Queensland and Australian Governments have developed and supported the implementation of the Reef Plan since 2003 (The State of Queensland and Commonwealth of Australia, 2003; Reef Water Quality Protection Plan Secretariat, 2009, 2013b). While progress was reported in the uptake of agricultural BMPs and consequent reductions in (modelled) river pollutant loads (Queensland Government, 2015a), Reef Plan's 2013 targets for water quality and land and catchment management (Table 1) have not yet been met. Moreover, the inshore marine environment continues to be in poor condition (Queensland Government, 2015a,b; Fig. 2a, b), making it highly unlikely that Reef Plan's 2020 goal of 'no detrimental impact' will be met. This is of concern given that the Reef Plan is the main foundational programme to achieve the objective and targets associated with land pollution in the Reef 2050 LTSP (Table 1).

Reducing the siltation and eutrophication of coastal marine ecosystems by better managing agricultural sources at local and regional scales is a challenge for coastal communities around the world (Cloern, 2001; Boesch, 2002). Globally, substantial effort and investment is going into re-establishing environmental flows (Postel & Richter, 2003), controlling sediment erosion and transport (Walling, 2006) and reducing nutrient fluxes to coastal waters (Cloern, 2001; Boesch, 2002). Only relatively few studies, however, have measured reduced fluxes of sediment and nutrients at the end of river and associated declines in nutrient concentrations and algal biomass in receiving coastal waters following deliberate management of agricultural diffuse sources (Kroon *et al*., 2014). The scarcity of successful outcomes for coastal marine ecosystems is at least partly due to the fact that solutions are being sought within the context of 'business as usual' approaches to agricultural land uses and management. Indeed, studies from Eastern Europe demonstrate the magnitude of change required in agricultural systems to measurably reduce nutrient fluxes at the end of river within time frames of 10–20 years (Kroon *et al*., 2014).

In the case of the GBR WHA, protecting GBR water quality and ecosystems from land-based pollution remains a major challenge despite a generally positive policy environment. Substantial scientific effort has gone into understanding the functioning of, and threats to the GBR ecosystems (Great Barrier Reef Marine Park Authority, 2014), including the impacts (Schaffelke *et al*., 2013), sources (Kroon *et al*., 2013) and management (Thorburn *et al*., 2013a) of diffuse sources of pollution. Moreover, scientific syntheses and consensus statements have informed the development and implementation of the three consecutive Reef Plans (Baker, 2003; Brodie *et al*., 2008, 2013). Evidence from both biophysical and socio-economic sciences now increasingly suggests, however, that current efforts are insufficient to achieve the Reef Plan 2013 and Reef 2050 LTSP targets within the time frames stipulated in government policy.

To increase the likelihood of protecting GBR water quality and ecosystems from land-based pollution into the future, we identify improvements that can be made to current approaches, as well as potential opportunities for new agricultural products or land uses that reduce pollutant exports. The intention of our suggestions is to support the ongoing discussion on GBR catchment management for desired ecological outcomes for the GBR WHA and to explore new approaches for improved GBR protection based on overseas experiences and Australia's unique potential. We argue that it is only through a good understanding of the mixture of potential approaches required to achieve the Reef 2050 LTSP objective of 'no detrimental impact' that informed decisions can be made about effectively addressing poor water quality from land-based run-off. Given its iconic status and its location in one of the world's highest ranking countries for education and standard of living (Malik, 2014), it would be a dire outlook indeed for coastal marine ecosystems around the world were Australia not able to restore and protect the Outstanding Universal Value of the GBR WHA now and into the future.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Table S1. Purpose, impacts and status of selected acts and regulations (a), and Policies critical to managing the threat of land-based pollution (b) to the Great Barrier Reef World Heritage Area.

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Viewpoint

Informing policy to protect coastal coral reefs: Insight from a global review of reducing agricultural pollution to coastal ecosystems

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ABSTRACT

The continuing degradation of coral reefs has serious consequences for the provision of ecosystem goods and services to local and regional communities. While climate change is considered the most serious risk to coral reefs, agricultural pollution threatens approximately 25% of the total global reef area with further increases in sediment and nutrient fluxes projected over the next 50 years. Here, we aim to inform coral reef management using insights learned from management examples that were successful in reducing agricultural pollution to coastal ecosystems. We identify multiple examples reporting reduced fluxes of sediment and nutrients at end-of-river, and associated declines in nutrient concentrations and algal biomass in receiving coastal waters. Based on the insights obtained, we recommend that future protection of coral reef ecosystems demands policy focused on desired ecosystem outcomes, targeted regulatory approaches, up-scaling of watershed management, and long-term maintenance of scientifically robust monitoring programs linked with adaptive management.

1. Introduction

The continuing degradation of coral reefs around the world (Bruno and Selig, 2007; De'ath et al., 2012; Gardner et al., 2003) has serious consequences for the provision of ecosystem goods and services to local and regional communities. While climate change is considered the most serious risk to coral reefs around the world, agricultural pollution threatens approximately 25% of the total global reef area (Burke, 2011) (Fig. 1). To ensure the future of coral reefs, the 2012 Consensus Statement on Climate Change and Coral Reefs has called for the immediate management of local anthropogenic pressures including reducing land-based pollution (12th International Coral Reef Symposium, 9–13 July 2012). Attempts are being made to reduce land-based pollution to coral reefs (Brodie et al., 2012; Richmond et al., 2007), however, these efforts are impeded by a current paucity of studies demonstrating whether improvements to coral reef health are realized following watershed management. For the next 50 years, riverine fluxes of sediment, nitrogen (N) and phosphorus (P) to tropical coastal areas are projected to increase (Mackenzie et al., 2002). It is therefore timely to inform coral reef policy using insights gained from global cases that were successful in reducing agricultural pollution to coastal ecosystems.

Here, we synthesize successful examples of reduced agricultural pollution that could be used as a model to improve coral reef water quality, with the assumption that improved water quality will result in a concomitant improvement in ecological health of coral reefs. Previous reviews of the problem of coastal eutrophication (Boesch, 2002; Cloern, 2001) do not include recent reports on reduced fluxes of sediment and nutrients at end-of-river (Chu et al., 2009; Duarte et al., 2009; GEF-UNDP, 2006; Pastuszak et al., 2012; Stålnacke et al., 2003; Windolf et al., 2012), and associated declines in nutrient concentrations and algal biomass in receiving coastal waters (Carstensen et al., 2006; Duarte et al., 2009; Jurgensone et al., 2011; Oguz and Velikova, 2010). Our review focuses on restoration of diffuse fluxes of freshwater, suspended sediment, and nutrients, while acknowledging the presence of other pollutants (e.g. pesticides, herbicides and heavy metals) and their potential impact on coral reef resilience (Van Dam et al., 2011). We first summarize the global evidence for changes in freshwater flow regimes and terrestrial pollutant fluxes to coastal and coral reef environments. Next, we outline how coral reefs are affected by resultant changes in water quality. We then examine the effectiveness of land-based efforts aimed at restoring

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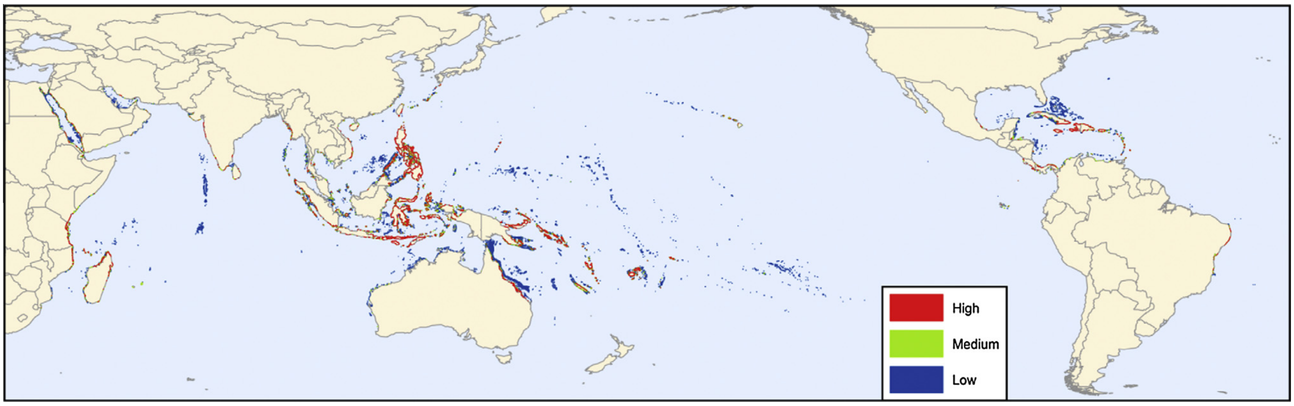

Fig. 1. Global classification of the threat from watershed-based pollution to coral reefs, using an index based on estimated erosion and nutrient fertilizer runoff from agriculture delivered by rivers to coastal waters (data from Burke, 2011).

more natural fluxes to coastal and coral reef environments and reversing ecosystem degradation. We conclude with the insights gained into effective management of agricultural pollution from multiple global examples where reductions of land-based pollution to coastal ecosystems have been achieved. Because patterns in coastal water quality data following land use change display similar trends globally (Boesch, 2002; Cloern, 2001; Mackenzie et al., 2002; Syvitski et al., 2005), we envisage that the insights from effective management examples in non-tropical systems can be successfully transferred to coral reefs.

2. Alteration of terrestrial freshwater, sediment and nutrient fluxes to coastal waters

Globally, humans have altered terrestrial fluxes of freshwater (Vörösmarty and Sahagian, 2000), sediment (Syvitski et al., 2005), and nutrients (Mackenzie et al., 2002) to coastal marine waters, including to coral reef environments (Hendy et al., 2002; Hungspreugs et al., 2002; McCulloch et al., 2003; Prouty et al., 2009; Yamazaki et al., 2011). Natural river flow regimes, including magnitude, frequency, duration, timing, and rate of change, have been modified through surface water diversion, dam construction, aquifer mining, and wetland drainage and deforestation (Vörösmarty and Sahagian, 2000). This includes modification of flow regimes in tropical coastal catchments upstream from coral reefs in both the Atlantic (Porter et al., 1999) and Indo-Pacific (Pena-Arancibia et al., 2012). Impoundments and diversion of surface water enhance evaporation and reduce run-off, altering the magnitude and timing of freshwater flows (Vörösmarty and Sahagian, 2000). In contrast, the loss of water storage capacity associated with wetland drainage and deforestation results in lower evaporation, increased runoff, and more variable hydrographs (Vörösmarty and Sahagian, 2000). The resulting changes in long-term net runoff have modified coastal salinity, nutrient stoichiometry and biogeochemistry (Cloern, 2001), including on coral reefs (Porter et al., 1999).

Fluxes of terrestrial sediment to coastal marine waters have been modified by humans around the world (Syvitski et al., 2005). Increases in these fluxes are due to soil erosion, associated with changes in surface runoff, deforestation, coastal development, urbanization, agricultural practices, and mining. In tropical coastal regions, annual fluxes of suspended sediment have increased by approximately 1.3 times, with 16% of the current flux retained in impoundments. This is exemplified in the Great Barrier Reef region, where a large proportion of terrestrial sediment is trapped by multiple reservoirs (e.g. 10–90% depending on flow in the Burdekin Falls Dam, (Lewis et al., 2009)). Notwithstanding, annual fluxes of terrestrial sediment to the lagoon are still estimated to have increased more than fivefold since European settlement (Kroon et al., 2012) (Fig. 2). On the other hand, reductions in sediment fluxes to coastal areas are primarily due to retention within impoundments (Syvitski et al., 2005). Reservoirs now retain 26% of the global sediment flux, resulting in an overall 10% decrease compared to the prehuman sediment load (Syvitski et al., 2005). Overall, these changes in terrestrial sediment fluxes to coastal ecosystems directly affect habitat formation of benthic environments through enhanced sedimentation or coastal erosion.

Global fluxes of nitrogen (N) and phosphorus (P) to coastal areas have increased due to human activities (Cloern, 2001; Galloway et al., 2008), with a doubling of riverine, reactive N and P fluxes in the preceding 150 years (Galloway et al., 2004; Mackenzie et al., 2002), and a rise in atmospheric deposition of N from land to coastal areas (Galloway et al., 2004). Increases in these fluxes to the coastal zone are due to agricultural crop and livestock production, fertilizer application, discharge of urban and industrial sewage, and fossil fuel burning (Galloway et al., 2008), as well as removal of the ecosystems' filtering and buffering capacity (e.g. riparian zones and floodplain wetlands, (Verhoeven et al., 2006). Further substantial increases in riverine fluxes of N and P to coastal areas are projected (Galloway et al., 2004), particularly in tropical regions (Mackenzie et al., 2002). Nutrient loadings to the Great Barrier Reef lagoon, for example, have increased 6-fold for N and 9-fold for P since European settlement in the 19th century (Kroon et al., 2012) (Fig. 2). Excess nutrient inputs to coastal areas increase net primary production and lead to eutrophication (Cloern, 2001), which in extreme cases causes widespread hypoxia (Diaz and Rosenberg, 2008), and contribute to loss of ecosystem diversity, structure and functioning (Lotze et al., 2006).

3. Impacts of land based pollution on coral reefs

Modification of terrestrial pollutant fluxes, and consequent declines in reef water quality have resulted in detrimental impacts on physical, ecological and physiological processes of reef-building corals (Coles and Jokiel, 1992; Fabricius, 2011). Compared to other terrestrial pollutants, the effects of changes in freshwater fluxes on coral reefs have received relatively little attention. Proxy records from coral cores indicate both enhanced (Hendy et al., 2002) and reduced (Prouty et al., 2009) freshwater fluxes into tropical waters since the late 19th century. Cases of coral mortality, bleaching and disease, associated with reduced salinity due to extreme rainfall, land runoff, and groundwater discharge, have been documented on coral reefs around the world (Coles and Jokiel, 1992). Conversely, reduced freshwater fluxes may result in increased

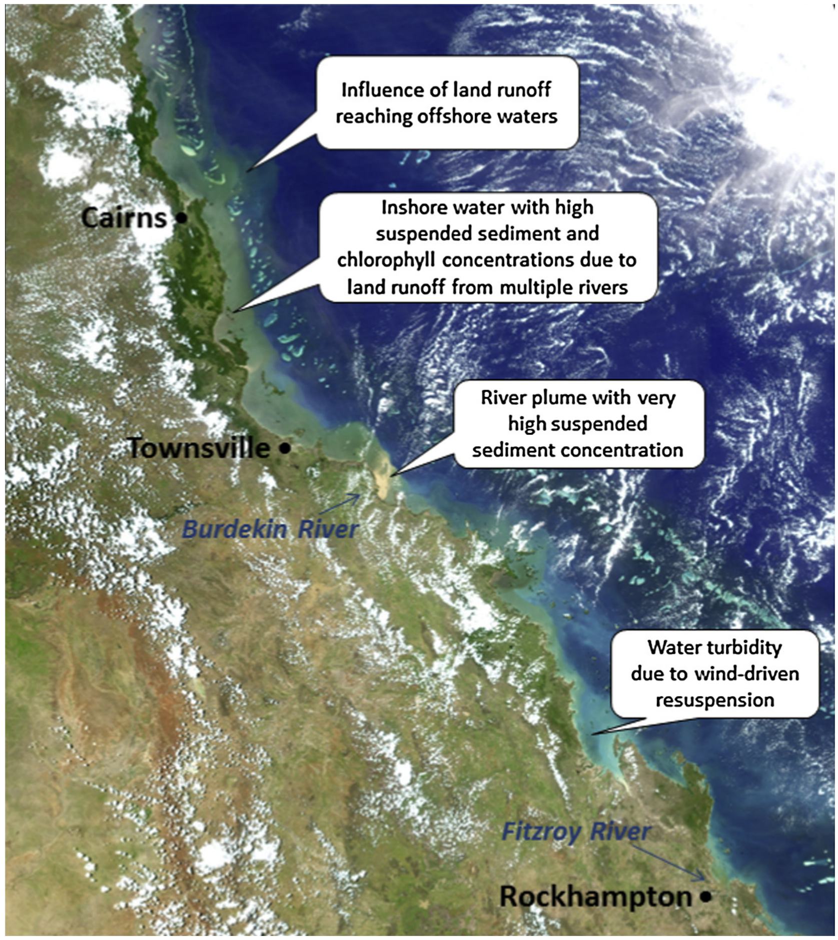

Fig. 2. River flood plumes carrying watershed-based pollutants reaching the Great Barrier Reef, Australia, captured by MODIS Aqua on 10 February 2007.

salinity in coastal embayments, detrimentally affecting downstream coral communities (Porter et al., 1999). While corals can generally tolerate short changes in salinity, longer exposure to salinities outside their normal range leads to reduced growth and production, with mortality occurring at both low and high salinities (Coles and Jokiel, 1992). The exact responses (including acclimation) depend on the coral species, the magnitude of salinity change compared to background levels, and the exposure time (Berkelmans et al., 2012). However, it is currently unknown whether adverse effects of salinity on coral reefs have become more frequent or extensive with alteration of freshwater flow regimes to tropical coastal waters.

Cores of reef sediment and corals have indicated both increases (McCulloch et al., 2003) and decreases (Hungspreugs et al., 2002) in terrestrial sediment fluxes to coral reefs since the 1900s. Increases in sediment fluxes can result in smothering of coral reef organisms due to the settling of suspended sediment (sedimentation), as well as in reduced light availability for photosynthesis due to turbidity caused by suspended sediment in the water column (Fabricius, 2011). Sedimentation can lead to profound changes in coral populations affecting all life history stages. High sedimentation rates may reduce larval recruitment by making the settlement substratum unsuitable (Dikou and van Woesik, 2006). After settlement, sediment composition and short-term sedimentation affect the survival of coral recruits, and inhibits growth of adult corals through reduced photosynthesis and production (Fabricius, 2011). Extensive or excessive sediment exposure can also result in coral disease (Sutherland et al., 2004) and mortality (Victor et al., 2006), and concomitant phase shifts to macro-algal dominance have been observed (De'ath and Fabricius, 2010; Dikou and van Woesik, 2006). Recovery is possible from short-term or low levels of sedimentation (Fabricius, 2011) as the polyps of many coral species exhibit sediment rejection behavior comprising of ciliary currents, tissue expansion, and mucus production (Stafford-Smith and Ormond, 1992). The exact responses to sedimentation depend on the coral species, duration and amount of sedimentation, and sediment types (Fabricius, 2011).

Enriched signatures of N isotopes in coral cores and tissues indicate increased fluxes of terrestrial N to coral reefs from agricultural and sewage run-off since at least the 1970s (Jupiter et al., 2008; Marion et al., 2005; Yamazaki et al., 2011). Likewise, cores of reef sediment and corals have indicated an increase in terrestrial phosphorus fluxes to coral reefs in the 20th century, associated with soil erosion, sewage, aquaculture and mining operations and harbor development (Chen and Yu, 2011; Dodge et al., 1984; Harris et al., 2001; Mallela et al., 2013). Corals are mostly adapted to low-nutrient environments and increases in primary production and eutrophication due to enhanced nutrient loads can detrimentally affect corals (Fabricius, 2011). Direct effects of increased nutrients are generally at the physiological level but so far there is little evidence that this leads to coral mortality. However, indirect effects of nutrient pollution are profound. For example, phototrophic hard corals can be out-competed by other benthic primary producers in high nutrient environments, leading to the establishment of macro-algae. High nutrient availability generally leads to increases in phytoplankton populations which in extreme cases reduce benthic light availability and cause seasonal hypoxia (Diaz and Rosenberg, 2008). Resultant organic enrichment can cause a shift to heterotrophic and/or filter feeding communities, and plays a role in driving population outbreaks of the coral-eating crown-of-thorns starfish (Fabricius, 2011), one of the main causes of coral cover declines on the Great Barrier Reef (De'ath et al., 2012). Overall, eutrophication can result in increased coral disease (Sutherland et al., 2004; Vega Thurber et al., 2013) and mortality, and contribute to loss of coral diversity, structure and function, including phase shifts to macroalgae (Fabricius, 2011).

4. Restoration of terrestrial freshwater, sediment and nutrient fluxes to coastal waters

The reduction of siltation and eutrophication of coastal marine ecosystems by better managing agricultural sources at local and regional scales is a challenge for coastal communities around the world (Boesch, 2002; Cloern, 2001), including those bordering coral reefs (Brodie et al., 2012). Globally, substantial effort is going into re-establishing environmental flows (Postel and Richter, 2003). In headwater catchments, more natural flow regimes are being reinstated through, for example, including high flows in dam releases (Rood et al., 2005) and removing small dams and weirs (Stanley and Doyle, 2003). Ecological outcomes in downstream reaches have been documented within a year, and include formation of new river channels, restored riparian vegetation, and improved fish passage and spawning habitat (Rood et al., 2005; Stanley and Doyle, 2003). Restoration of more natural flow regimes to coastal marine waters is being attempted through, for example, removal of large dams (Service, 2011), buying back irrigation water (Pincock, 2010) or agricultural land (Stokstad, 2008), and restoration of coastal floodplains (Buijse et al., 2002). Such larger-scale interventions have only commenced in recent years, and consequently, we were unable to find any documented examples of restored freshwater flow regimes into coastal waters (Table 1a). Nevertheless, while it is expected that freshwater flows should return to more natural regimes almost immediately, recovery of associated physical and biological processes may take years to decades (Hart et al., 2002).

Despite significant investment in sediment erosion and transport control measures (Bernhardt et al., 2005), we found only one documented example of reductions in net fluxes of sediment reaching coastal marine waters following land-based restoration efforts (Tables 1b and 2). In China, targeted management of pollutant sources and pathways to conserve soil and water has contributed to reducing sediment fluxes to coastal waters (Chu et al., 2009). Soil and water conservation programs in China were first legislated in the 1950s following concern about local agricultural and industrial productivity and flooding downstream (Shi and Shao, 2000). Implementation at large spatial scales (e.g. 0.92 M km^2 of land terracing, tree and grass planting, and construction of sediment trapping dams), mostly in the Yellow and Yangtze basins, has reduced sediment fluxes to coastal waters by an estimated 11.5 Gt during 1959–2007 (Chu et al., 2009).

Terrestrial fluxes of N and P to coastal waters have been reduced following management of point sources, such as waste water treatment plants, phosphate mines and P-detergents (Boesch, 2002; Cloern, 2001) (Tables 1c and 2). For example, regulation has reduced the contributions from waste water treatment plants and industrial discharges to total annual average N and P loads to the Danish coast from ~50% to <10%, and from 59% to ~20%, respectively, over 14 years (Carstensen et al., 2006). The nutrient regulation in Denmark followed lobster mortality in coastal waters in the 1980s which was attributed to algal blooms and hypoxia induced by agricultural nutrient run-off (Windolf et al., 2012). Similar declines in nutrient loads from point sources have resulted in reductions in coastal nutrient and chlorophyll *a* (chl *a*) concentrations (Greening and Janicki, 2006), enhanced benthic irradiance (Greening and Janicki, 2006), seagrass recovery (Tomasko et al., 2005), and concomitant decline in macroalgae (Cardoso et al., 2010; Vaudrey et al., 2010), including on coastal coral reefs (Laws and Allen, 1996; Smith et al., 1981). Further recovery, including to a coral-reef dominated state, may be partly constrained by nutrient sources other than point sources (Hunter

Table 1
Published evidence on the effectiveness of land-based management to restore more natural fluxes of (a) freshwater, (b) suspended sediment, and (c) nutrients, to coastal marine environments, resulting in improved coastal water quality and ecosystem condition.

Land-based management regime	Evidence for change			Source
	Riverine fluxes	Coastal water quality	Coastal ecosystem condition	
(a)				
Large dam removal	None reported effects likely to be immediate	None reported effects likely to take yrs/decades		Hart et al. (2002), Stokstad (2008), Pincock (2010), Buijse et al. (2002), Service (2011)
Coastal floodplain restoration Buy back irrigation water Buy back agricultural land	None reported effects likely to take yrs/decades			
(b)				
Source control, e.g. terraces, revegetation	Yellow and Yangtze rivers	None reported effects likely to take decades/centuries		Shi and Shao (2000), Chu et al. (2009)
Transport control, e.g. sediment trapping dams	Yellow and Yangtze rivers			
(c)				
Point sources				
STPs, P mines, P-free detergent	Various examples from USA, Europe	Decline in nutrients, turbidity, Chl *a*	Partial recovery of aquatic communities, e.g. seagrass and coral	Boesch, (2002), Smith et al. (1981), Hunter and Evans (1995), Laws and Allen (1996), Tomasko et al. (2005), Greening and Janicki (2006)
Diffuse sources				
Application control measures, e.g. reduction in N and P fertiliser use, decrease in livestock numbers and manure use, changes in land use	Danube, Daugava, Elbe, Leilupe, oder and Vistula rivers, Denmark, The Netherlands	Decline in nutrients and phytoplankton biomass	Concomitant changes in flora and fauna, but no complete recovery	Mee (2001), Stålnacke et al. (2003), Carstensen et al. (2006), GEF-UNDP (2006), Duarte et al. (2009), Oguz and Velikova (2010), Hansen and Petersen (2011), Jurgensone et al. (2011), Pastuszak et al. (2012), Windolf et al. (2012)
Transport control measures, e.g. wetland restoration	Denmark	None reported effects likely to take decades/centuries		Windolf et al. (2012)

and Evans, 1995), as well as obscured by increases in human population, changes in diffuse sources and variation in freshwater discharge (Williams et al., 2010).

Reducing diffuse source loads becomes increasingly important where point source discharges comprise only a small percentage of the total N and P loads, such as in the Great Barrier Reef (GBRMPA, 2009). Major recent reviews provide recommendations to reduce excessive or inappropriate input of N and P from diffuse sources such as agriculture, fossil-fuel and animal husbandry (Canfield et al., 2010; Elser and Bennett, 2011; Galloway et al., 2008; Vitousek et al., 2009). Deliberate management of agricultural diffuse pollution has contributed to reducing nutrient fluxes to coastal waters in Denmark (Windolf et al., 2012) and The Netherlands (Duarte et al., 2009) within decades (Tables 1c and 2). Moreover, decreasing nutrient fluxes have been measured in several Eastern European rivers, namely the Danube, Daugava, Elbe, Leilupe, Oder and Vistula rivers, in the years following economic decline and associated drop in agricultural subsidies in the early 1990s (Duarte et al., 2009; GEF-UNDP, 2006; Mee, 2001; Pastuszak et al., 2012; Stålnacke et al., 2003). While not the result of a dedicated management strategy, these Eastern European examples demonstrate the magnitude of change required in agricultural management to reduce nutrient fluxes at end of river within timeframes of ten to twenty years. Subsequent declines in nutrient concentrations and phytoplankton biomass have been reported in the Western Dutch Wadden Sea and South East North Sea (Duarte et al., 2009), the Danish straits (Carstensen et al., 2006; Duarte et al., 2009), the Gulf of Riga (Jurgensone et al., 2011), and the Black Sea (Oguz and Velikova, 2010), respectively. Whilst the Danish straits and the Black Sea also show some concomitant changes in flora and fauna (Hansen and Petersen, 2011; Oguz and Velikova, 2010), complete recovery to pre-impact conditions has not been reported.

Finally, restoration of coastal ecosystems' filtering and buffering capacity is expected to enhance sediment and nutrient retention and assimilation during catchment transport processes. Improving an ecosystem's buffering capacity, for example through restoration or creation of wetlands (Verhoeven et al., 2006) and riparian zones (Tomer and Locke, 2011), can result in full recovery of N storage and cycling processes within 25–30 years (Moreno-Mateos et al., 2012). If critical nutrient loads are surpassed, however, undesirable phase-shifts can occur in these wetland and riparian ecosystems (Verhoeven et al., 2006), potentially reducing the systems' capacity for nutrient cycling (Cardinale, 2011). Establishing more natural drainage and vegetation patterns is expected to further increase hydraulic, sediment, and nutrient residence times and enhance the opportunity for landscape mitigation of terrestrial fluxes (Burt and Pinay, 2005; Whalen et al., 2002). Enhancing an ecosystem's filtering capacity, for example through restoration of native seagrass (McGlathery et al., 2012) or oyster beds (Schulte et al., 2009), will contribute to deposition of suspended sediment, nutrient cycling and water filtration (Cloern, 2001; McGlathery et al., 2012) and may significantly reduce total sediment and nutrient loads to receiving waters (Cerco and Noel, 2010). However, despite significant investments in improving ecological filtering and buffering capacity (Bernhardt et al., 2005; Moreno-Mateos et al., 2012; Whalen et al., 2002), concomitant reductions in total pollutant loads to coastal marine waters have not been documented and may take decades to centuries.

5. Reversal of coral reef degradation

Commensurate with the lack of evidence of restored flow regimes and sediment fluxes to tropical coastal marine waters, the resultant ecological outcomes for coastal coral reefs remain unknown. Corals have the capacity to recover from short-term exposure to both low and high salinity (Coles and Jokiel, 1992), as exemplified by partial (Goreau, 1964) and complete (Egana and Disalvo, 1982) recovery of corals within days to months following mass expulsion of zooxanthellae after heavy rainfall. Similarly, following short-term or low levels of sedimentation, structural (i.e. polyp re-colonization) (Wesseling et al., 1999) and functional (i.e. photosynthetic activity) (Philipp and Fabricius, 2003) recovery within days to weeks has been demonstrated for some, but not all, coral species. Coral growth recovered within weeks following short-term enrichment of N, and of N and P combined, but not of P (Ferrier-Pages et al., 2000). It is unlikely for such swift recovery to occur following restoration of more natural freshwater, sediment and nutrient fluxes, given that coral ecosystem processes would have been chronically impacted for years to decades, if not centuries.

The well-known case of Kane'ohe Bay, Hawaii, is the only example demonstrating partial reversal of coral reef degradation

Table 2
Global examples of reductions in terrestrial fluxes of sediment and nutrients to coastal marine waters, following quantified changes in watershed management with timeframes of detection.

Pollutant	Location	Scale of change	Flux reduction	Detection timeframe (yrs)	Source
Reduction in river pollutant fluxes					
Sediment	Yellow river	Soil and water conservation ~17,000 km^2	23% (0.3 Gt yr-l)	28	Chu et al. (2009)
	Yangtze river	Soil and water conservation ≥84,000 km^2	12% (60 Mt yr-l)	17	
Nutrients	Danube river	Reduction in fertiliser use (N 50%, P 70%), and livestock numbers (>50%)	60% (N), 50% (P)	8	Mee (2001), GEF-UNDP (2006)
	Elbe river	Reduction in agricultural N surplus (>50%), and industrial/STP emissions (N 50%, P 32%)	30% (N, P)	15	Hussian et al. (2004)
	Lielupe river	Reduction in fertilizer use (N 90%, P 95%), and livestock numbers (70%)	Significant declines in nutrient levels	11	Stålnacke et al. (2003)
	Oder river Vistula river	Reduction in fertiliser use (NPK 43.1%)*, livestock numbers (20.6%)*, and STP emmissions (N 27-60%, P 61-73%)	25% (N), 65% (P) 20% (N), 15% (P)	20	Jankowiak et al. (2003), Pastuszak et al. (2012)
	Denmark	Reduction in fertiliser use (N 50%), and industrial/STP emissions (N 74%, P 88%)	51% (N), 64% (P)	20	Kronvang et al. (2008), Hansen and Petersen (2011), Windolf et al. (2012))

* For whole of Poland for period 1988-1997.

following a reduction in terrestrial nutrient fluxes. Following sewage diversion in 1978, turbidity, nutrients and chlorophyll *a* concentrations, as well as macroalgae biomass, declined within months (Laws and Allen, 1996; Smith et al., 1981). In the next few decades, coral cover more than doubled and subsequently stabilized, however, further recovery may at least be partly constrained by nutrient sources other than sewage outfalls, by modified freshwater and sediment fluxes resulting from historical and recent changes in the Bay and its catchments (Hunter and Evans, 1995), and by additional impacts of introduced macroalgae (Conklin and Smith, 2005).

To reverse coral reef degradation, it is critical to define the different ecosystem states of a coral reef system, and understand the ecological processes that drive the change from one state to another. This relates to the concept of resilience, i.e. the capacity of an ecosystem to absorb perturbations before it shifts to an alternative state with different species composition, structure, processes and functions (Folke et al., 2004). For coral reefs, multiple alternative states can exist and have been documented for coral reefs, generally dominated by organisms other than reef-building coral (Gardner et al., 2003; Hughes et al., 2010; Mumby et al., 2007). Chronic environmental pressures such as changes in terrestrial fluxes of freshwater, sediment, and nutrients (De'ath and Fabricius, 2010; Dubinsky and Stambler, 1996; Fabricius, 2011) reduce resilience by decreasing the threshold at which the coral-dominated state shifts into a different state. A return to the more desirable coral-reef dominated state by reducing chronic drivers of change such as land-based pollution may be difficult to achieve due to the inherent stability of the degraded state, known as hysteresis (Mumby and Steneck, 2011).

6. Reducing agricultural pollution

We identified multiple examples in the global literature where reductions of land-based pollution to coastal ecosystems have been achieved (Table 2). Most examples comprise reduced nutrient fluxes from point sources, such as waste water treatment plants, through legislative mandates and regulatory enforcement (Boesch, 2002; Cloern, 2001). More recent examples also include studies demonstrating reduced sediment and nutrient fluxes from agricultural land use (Chu et al., 2009; Duarte et al., 2009; GEF-UNDP, 2006; Pastuszak et al., 2012; Stålnacke et al., 2003; Windolf et al., 2012). These examples provide us with the following insights into effective management of agricultural pollution.

First, the desired outcomes of agricultural management for coral reef ecosystems need to be clearly defined, and underpinned by knowledge of the processes that determine the trajectories of ecosystem recovery. The substantial large-scale and long-term decline in coral reef condition over recent decades (Bruno and Selig, 2007; De'ath et al., 2012; Gardner et al., 2003) has, in part, been linked to agricultural pollution. Attempts to reverse this decline, however, are generally constrained to improving agricultural and land-based pollution *per se* (Brodie et al., 2012; Richmond et al., 2007) without due consideration of the effort required to achieve desired outcomes for coral reefs. Consequently, many management efforts are not targeting the critical sources and ecological processes that underpin the pollution problem being remedied (Palmer, 2009). Similar to temperate systems, a return to a particular past state may be unlikely, and other perturbations such as climate change, overfishing, and invasion by non-native species may prevent a simple reversal of coastal ecosystem degradation following improvements to upstream water quality (Duarte et al., 2009; Jurgensone et al., 2011; Oguz and Velikova, 2010). Hence, when linking the implementation of agricultural management targets to ecosystem condition in reef waters, a range of possible outcomes with associated trajectories should be considered (Palmer, 2009; Perry and Smithers, 2011).

Second, management approaches that have resulted in reduced agricultural pollution to coastal ecosystems have all been non-voluntary (Boesch, 2002; Chu et al., 2009; Cloern, 2001; GEF-UNDP, 2006; Pastuszak et al., 2012; Stålnacke et al., 2003; Windolf et al., 2012), indicating that voluntary approaches alone may not be sufficient to achieve improvements. These reductions were achieved through legislation and regulation supported by long-term political commitment (e.g. China, Denmark) (Shi and Shao, 2000; Windolf et al., 2012) or declining economic subsidies, fertilizer use and livestock numbers following the collapse of the Soviet Union (eastern Europe) (GEF-UNDP, 2006; Jankowiak et al., 2003; Pastuszak et al., 2012; Stålnacke et al., 2003). In Denmark, for example, five national action plans were implemented and enforced to improve waste water treatment, and regulate N fertilizer and manure use over two decades (Kronvang et al., 2008; Windolf et al., 2012). Each action plan in turn has been underpinned by a variety of descriptive, and sometimes quantitative, policy measures, such as strict timeframes for manure storage, a 10% cut in the optimal N quota to crops, and a decrease in arable land (Windolf et al., 2012). While specific details may differ in tropical countries, the examples from China and Europe indicate that targeted regulatory policy approaches can greatly enhance the protection of downstream coral reef ecosystems from land-based pollution.

Third, management efforts to control agricultural pollution need to be at relevant spatio-temporal scales to achieve desired ecological outcomes on downstream coral reefs. The magnitude of effort required to obtain significant pollution reductions is exemplified in non-tropical systems, including (i) (unintended) large cuts in pollutant sources (e.g. ~95% cut in fertilizer use and ~70% drop in livestock numbers in Latvian rivers (Stålnacke et al., 2003)), (ii) application at large spatial scales (e.g. 84,000 km^2 of land terracing, tree and grass planting, and construction of sediment trapping dams in China (Chu et al., 2009)), and (iii) adaptive implementation over decadal time frames (e.g. >25 years in Denmark (Windolf et al., 2012)) (Table 2). Across all European rivers, substantial decreases in the nutrient input from agriculture contributed to nutrient load reductions at end-of-river. The Chinese and Danish cases further demonstrate that targeted and simultaneous implementation of a combination of measures will augment reductions of pollutant fluxes at watershed outlets. Enhanced targeting and upscaling of management efforts in agricultural systems will improve the condition of coral reef ecosystems, whilst also preventing further detrimental impacts from predicted increases in sediment and nutrient fluxes in the next 50 years.

Finally, sustained monitoring at appropriate spatio-temporal scales is required to ascertain whether agricultural management results in desired improvements of downstream coral reef ecosystems. Importantly, these monitoring programs should be driven by the development of critical questions and objectives, a conceptual understanding of linkages between desired outcomes and land-based pollution (Bartley et al., 2014), robust statistical design, and adaptive review cycles (Lindenmayer and Likens, 2009). In complex systems such as coral reefs, this would maximize the probability of detecting trends following management intervention, which could take years to decades even in comprehensively monitored systems (Darnell et al., 2012; Meals et al., 2010). Importantly, consideration of desired outcomes for coral reefs in monitoring programs will focus efforts towards detecting change in relevant metrics. For example, specific biological indicators have been identified that link changes in marine water quality to changes in the condition of coral reef ecosystems (Cooper et al., 2009). Similar metrics in upstream watersheds will enable the assessment of progress early in the management phase and alert managers to potential unintended consequences, e.g. multiple sediment trapping dams (Chu et al., 2009) may act as flow and fish migration barriers. This emphasizes the need to maintain long-term monitoring programs to provide feedback on ecosystem condition, linked with adaptive management programs (Lindenmayer and Likens, 2009; Meals et al., 2010).

7. Conclusion

The protection of coral reefs from human pressures on regional and local scales, such as increased fluxes of freshwater, sediments and nutrients, is particularly pertinent in the context of global environmental changes, such as rising sea temperatures, ocean acidification, increase in severity of tropical storms and sea level rise (Anthony et al., 2011; Carpenter et al., 2008; Pandolfi et al., 2011). Recent research has confirmed the ongoing degradation of coral reef ecosystems around the world (Bruno and Selig, 2007; De'ath et al., 2012; Gardner et al., 2003), but global examples of watershed management demonstrating the halting or reversing of coral reef decline are not readily available. Our global review demonstrates that transformative change in agricultural management for coastal ecosystem outcomes is achievable. For coral reef ecosystems, future protection demands policy focused on desired ecosystem outcomes, targeted regulatory approaches, upscaling of watershed management, and long-term maintenance of scientifically robust monitoring programs linked with adaptive management. Implementing these recommendations will increase the resilience of desired, coral-dominated states within a timeframe (years to decades) where more extreme perturbations associated with climate change are expected.

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