Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses

J.E. Brodie a,⁎, F.J. Kroon b, B. Schaffelke c, E.C. Wolanski a, S.E. Lewis a, M.J. Devlin a, I.C. Bohnet d, Z.T. Bainbridge a, J. Waterhouse a, A.M. Davis a

a Catchment to Reef Research Group, Australian Centre for Tropical Freshwater Research, James Cook University, Townsville, Qld 4811, Australia
b CSIRO Ecosystem Sciences, P.O. Box 780, Atherton, Qld 4883, Australia
c Australian Institute of Marine Science, PMB 3 Townsville MC, Qld 4810, Australia
d CSIRO Ecosystem Sciences, P.O. Box 12139, Earlville BC, Cairns, Qld 4870, Australia

⁎ Corresponding author. Tel.: +61 7 4781 6435; fax: +61 7 4781 5589.
E-mail addresses: jon.brodie@jcu.edu.au (J.E. Brodie), frederieke.kroon@csiro.au (F.J. Kroon), b.schaffelke@aims.gov.au (B. Schaffelke), eric.wolanski@jcu.edu.au (E.C. Wolanski), stephen.lewis@jcu.edu.au (S.E. Lewis), michelle.devlin@jcu.edu.au (M.J. Devlin), iris.bohnet@csiro.au (I.C. Bohnet), zoe.bainbridge@jcu.edu.au (Z.T. Bainbridge), jane.waterhouse@jcu.edu.au (J. Waterhouse), aaron.davis@jcu.edu.au (A.M. Davis).

1. Introduction

Degradation of coastal and marine ecosystems due to the effects of terrestrially derived pollution is a universal issue and the subject of intense management activity (Doney, 2010). The majority of coral reefs around the world are threatened by human activities (Burke et al., 2011) and many show signs of degradation (e.g. Pandolfi et al., 2003). In many areas reefs are exposed to a combination of stresses including destructive fishing practices, overfishing or loss of herbivoruous fish and other grazing organisms, increased discharge from the land of sediment, nutrients and pesticides, coral predator outbreaks linked to trophic changes in the system, increased bleaching associated with global climate change, and increased incidence of and severity of coral diseases (Burke et al., 2011; Maina et al., 2011). These pressures have led to precipitous declines in coral cover from values near 60% more than 50 years ago to 20% recently, and led to persistent shifts from coral dominance to non-coral and algal dominance (Hughes et al., 2010a; Mumby et al., 2007; Norström et al., 2009). The Great Barrier Reef (GBR) is situated on the north-east coast of Australia (Fig. 1). It has the status of a Marine Park under joint Australian (Federal) and Queensland State Government arrangements and has been a declared World Heritage Area in 1981. Despite this protected status, the coral cover on the GBR has also declined, albeit the timing and trajectory of the decline is in some dispute (Bellwood et al., 2004; Bruno and Selig, 2007; Hughes et al., 2011; Sweatman and Syms, 2011; Sweatman et al., 2011; Osborne et al., 2011). The causes of this decline are manifold and are to some degree reef-specific: terrestrial runoff of sediment and nutrients with the associated crown-of-thorns starfish outbreaks (Brodie et al., 2005, 2008a,b, 2011; De’ath and Fabricius, 2010; DeVantier et al., 2006; Fabricius, 2005; Fabricius et al., 2005, 2010); coral bleaching and mortality associated with climate change (Berkelmans et al., 2004; Hoegh-Guldberg et al., 2007; Hughes et al., 2007) and coral diseases (Haaplaylä et al., 2011) have been implicated. Ocean acidification and its effects on coral calcification is a newer threat just becoming apparent on the GBR (Coope et al., 2008; De’ath et al., 2009). In addition, these stressors do not act in isolation and interactions between these stressors are very likely but not yet well studied (Borges and Gypens, 2010). Other ecosystems such as coastal seagrass meadows are also believed to be under pressure in parts of the GBR World Heritage.
Area due to chronic effects of declining water quality as a result of river discharge (McKenzie et al., 2010) although overall seagrass health in the GBR region is seen as being in better condition than many other global regions (Waycott et al., 2009). Mangrove forests are generally considered to be in good condition subject only to small localised losses from coastal development (Schaffelke et al.,...
2005) but well protected from physical removal and disturbance through Queensland Fisheries legislation.

In 2008 the current state of knowledge regarding the degradation of GBR ecosystems due to terrestrial pollutant runoff was reviewed by a group of scientists and a ‘Scientific Consensus Statement’ was prepared for the Queensland Government (Brodie et al., 2008a,b). The conclusions of the Consensus Statement were:

1. Water discharged from rivers to the GBR continues to be of poor quality in many locations.
2. Land derived contaminants, including suspended sediments, nutrients and pesticides are present in the GBR at concentrations likely to cause environmental harm.
3. There is strengthened evidence of the causal relationship between water quality and coastal and marine ecosystem health.
4. The health of freshwater ecosystems is impaired by agricultural land use, hydrological change, riparian degradation and weed infestation.
5. Current management interventions are not effectively solving the problem.
6. Climate change and major land use change will have confounding influences on GBR health.
7. Effective science coordination to collate, synthesise and integrate disparate knowledge across disciplines is urgently needed.

One of the major challenges facing us in management of terrestrial pollution of the GBR is to be able to set scientifically based catchment targets which we are confident that, if achieved, will lead to the recovery of GBR ecosystems to a desired level. The only way of doing this is to have a robust modelling framework from the paddock and management practice scale to the reef scale. Such a model would allow us to robustly predict the effect of management interventions for reef health. Almost all of the current research effort in the GBR water quality field, much of it discussed in this paper, is in some way adding to our ability to build this model.

In this paper, we will review the sources, fate and effects of increased terrestrial runoff to the coastal and inshore GBR as well as the current management response, focusing on information since 2008, and re-evaluate the statements in the Consensus Statement against this new information.

2. The GBR catchment area (GBRCA)

The catchment draining into the GBR contains 35 defined river basins covering an area of over 424,000 km² (Furnas, 2003). Important land uses in the catchment (Fig. 1) include rangeland beef grazing (314,000 km²), sugarcane cultivation (5700 km²), horticulture (630 km²), other cropping including grain and cotton cultivation (11,600 km²), urban areas (2600 km²) and native forest (55,900 km²) (Waterhouse et al., 2009). Discharges of suspended solids, nutrients and pesticides from the catchment to the lagoon has increased greatly over the last 200 years (McKergow et al., 2005a,b; Kroon et al., 2012) due to wide-scale agricultural, urban and mining development (Furnas, 2003; Waterhouse et al., 2012).

3. River loads of suspended solids, nutrient and pesticides to the GBRWHA

The estimates of current and pre-European river loads of total suspended solids (TSS), nitrogen and phosphorus discharged from the GBRCA into the GBRWHA have been refined over the last three decades. Water quality monitoring programs have been conducted at the end of catchments in the GBRCA since the early 1970s. The objectives of these programs vary and have ranged from legislative requirements under the Queensland Water Act (2000), to providing information on sediment dynamics in river cross-sections during high-flow, to supporting the Reef Water Quality Protection Plan (Reef Plan) 2003 and 2009 (Queensland Department of the Premier and Cabinet, 2003, 2009). The first river load estimates of TSS, nitrogen and phosphorus to the GBRWHA were derived from time-integration of monitored discharge and concentration data (Belperio, 1983; Furnas, 2003). Subsequent studies estimating river loads to the GBRWHA have modelled diffuse sources and transport processes through a stream network, using SedNet (Sediment Budget River Network) and ANNEX (ANnual Nutrient EXport) (McKergow et al., 2005a,b; see also Cogle et al., 2006). Specifically, the SedNet model constructs budgets of the primary sources, stores and fluxes of fine, suspended sediment for each link in a river network (Wilkinson et al., 2009). The associated ANNEX module constructs budgets of nitrogen and phosphorus, representing particulate dissolved inorganic and dissolved organic forms (Young et al., 2001). In parallel, sediment, nutrient and herbicide loads based on monitoring data have been estimated using a variety of methods, with different load estimation techniques applied to different basins, or to different sets of monitoring information for individual basin (e.g., Brodie et al., 2009a; Joo et al., 2012; Kroon et al., 2012; Neil et al., 2002). Kroon et al. (2012) provides a comprehensive compilation of available catchment water quality and flow monitoring data across all basins draining to the GBR, as well as a synthesis of all pre-European and current load estimates extending to all GBR basins.

3.1. Suspended sediment

The TSS supply to the GBR lagoon originates from the three forms of erosion – hillslope, gully and streambank (McKergow et al., 2005a). Recent estimates indicate that since European settlement in the GBRCA (c. 1850), the mean annual TSS load to the GBR has increased by 5.5 times to 17,000 ktonnes/yr (Kroon et al., 2012). The large beef grazing dominated catchments of the Fitzroy and Burdekin contribute over 50% (7400 ktonnes/yr) to the mean annual anthropogenic (human caused) TSS load of 14,000 ktonnes/yr to the GBR lagoon (Kroon et al., 2012). Hillslope erosion is generally considered to be the dominant source due to low pasture cover (McKergow et al., 2005a), however, the importance of gully erosion has been highlighted in recent research especially in some catchment locations (Bartley et al., 2010a,b; Tims et al., 2010; Wilkinson et al., in press; Hughes et al., 2011; Andrew Brooks, pers. com.). Erosion may be severe in areas of cropping and urban development on high slope lands but such areas are of small extent and present generally a local issue. Mining may also contribute (Lucas et al., 2010) but this is an under-researched area.

3.2. Nutrients

Riverine discharge is the single largest source of nutrients to inshore areas of the GBR lagoon (Furnas et al., 1997, in press). Other sources include atmospheric inputs following rainfall events (Furnas et al., 1995), planktonic and microphytobenthic nitrogen fixation (Furnas et al., 2011), Coral Sea upwelling (Furnas and Mitchell, 1986), deposition of dust from storms generated in the interior of Australia (Shaw et al., 2008), transport of nitrogen oxides from sugararcne fertiliser through the air (Paton-Walsh et al., 2011) and wind resuspension of nearshore sediments and their associated nutrients (Gagan et al., 1987; Walker and O’Donnell, 1981).

Recent estimates indicate that since European settlement in the GBRCA, the mean annual total nitrogen (TN) load to the GBR lagoon has increased by 5.7 times to 80,000 tonnes/yr (Kroon et al., 2012);
The anthropogenic nitrogen load comprises 11,000 tonnes/yr dissolved inorganic nitrogen (DIN), 6900 tonnes/yr of dissolved organic nitrogen (DON), and 52,000 tonnes/yr of particulate nitrogen (PN). Similarly, the mean annual total phosphorus (TP) load has increased by 8.9 times to 16,000 tonnes/yr, with the anthropogenic phosphorus loads comprising 800 tonnes/yr of dissolved inorganic phosphorus (DIP), 470 tonnes/yr of dissolved organic phosphorus (DOP), and 13,000 tonnes/yr of particulate phosphorus (PP). These nutrient increases are driven by the application of fertiliser on sugar cane, horticulture, and other cropping areas in the GBRCA (Rayment, 2003; Waterhouse et al., 2012), and losses of particulate bound nutrients from agricultural and urban lands due to soil erosion (Brodie and Mitchell, 2005; McKergow et al., 2005b; Waterhouse et al., 2012).

Since European settlement, the predominant form of nitrogen that is delivered to the GBR lagoon has changed. Discharge of nitrogen from natural landscapes is predominantly in the form of DON (Brodie and Mitchell, 2005; Harris, 2001), including in runoff from undisturbed forests in the GBR catchment (Brodie and Mitchell, 2006). In contrast, nitrogen discharge from agricultural and urban lands is dominated by DIN derived from fertiliser and sewage wastes (Mitchell et al., 2009; Thorburn et al., 2011b), and PN derived from soil erosion (Brodie and Mitchell, 2005). This shift from a predominantly DON discharge to a predominantly PN and DIN discharge may cause noticeable changes in coral and macroalgal communities (Fabricius, 2005).

3.3. Pesticides

Pesticides would have been absent in runoff from the GBRCA prior to European settlement. Since then, recent estimates suggest that at least 30,000 kg/yr of herbicides are exported to the GBRWHA (Kroon et al., 2012). This estimate comprises photosystem-II (PSII) inhibiting herbicides only (atrazine, ametryn, hexazine, diuron, simazine and tebuathion), for which monitoring information has become available in recent years. It is likely an underestimate of the total pesticide load to the GBRWHA, as not all pesticides known to be used in the GBR catchment and leaking into its waterways are currently being monitored. Better methods of estimating loads of pesticides from unequally distributed land uses across a catchment are also being developed (Cook and Knight, 2011). Atrazine, ametryn, hexazine, and diuron originate predominantly from the sugarcane industry (Bainbridge et al., 2009a; Davis et al., in press, 2012), with atrazine also being used in grains cropping, and tebuathion and simazine originating from the beef grazing industry and forestry plantations, respectively (Lewis et al., 2009a; Shaw et al., 2010; Waterhouse et al., 2012). As a result of the initiation of the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Paddock to Reef Program; see Carroll et al., 2012) more systematic monitoring of pesticide residues across the GBR and its catchments have shown very widespread contamination of rivers and streams draining to the GBR and estuarine waterways with a range of pesticides. These measurements show frequent exceedances of Australian water quality guidelines often to the extent of 10–50 times the guideline (trigger value) for pesticides such as atrazine, diuron and metolachlor (Smith et al., 2012). More effective ways of detecting and measuring organism pesticide exposure have been developed including the use of biomarkers in crabs (van Oosterom et al., 2010) and phytotoxicity bioassays (Shaw et al., 2012).

3.4. Improving load estimation and source attribution

Since 2008 a number of research studies have and will allow us to improve the accuracy and estimate of uncertainty in our end-of-river load estimates, and more accurately attribute sources of pollutants to particular sub-catchments, landuses and landscape types. These improvements are summarised below.

(1) Improved load estimation through monitoring. For example, Wallace et al. (2008, 2009) showed that in overbank flow conditions in the Tully River a large proportion of the load bypassed the river gauge sampling location, thus contributing an additional 30–50% to the loads, including 47% and 32% to the loads of N and P, respectively. Wallace et al. (2012) have now extended this analysis of underestimation of discharge loads to other rivers in the GBRCA with variable increased loads above those at the gauge depending on river type and geomorphology.

(2) Improved modelling. For example, Bartley et al. (2012) reviewed existing research to provide greatly improved land-runoff concentration values from specific landuse types for model parametrisation. Lynam et al. (2010) showed that Bayesian Belief Networks (BBNs) could be used to assess the probabilities of changed management practice scenarios improving sediment and nutrient loads at end of catchment while Shenton et al. (2010) showed how BBNs could link improved management to marine ecosystem outcomes.

(3) Improved load estimation using more sophisticated algorithms that better account for the variable flow and concentration profiles found in northern Australian rivers and enabling better estimates of load estimation uncertainty (for example, Joo et al., 2012; Kuhnert et al., in review).

(4) Improved identification of specific catchment areas, sub-catchments, landuses and erosion processes that are the sources of contaminants reaching the river mouth. For example, the sources and transport of suspended sediment in the Fitzroy River catchment delivered to downstream environments including the GBR has now been well documented (Amos et al., 2009; Douglas et al., 2006a,b, 2008, 2010; Hughes et al., 2009; Smith et al., 2008). One important finding was that most of the fine sediment (<10 μm) transported from the catchment to the inner-shelf region of the GBR during flood events was sourced to Tertiary basaltic soil types in the north-western region of this river catchment (Douglas et al., 2008; Hughes et al., 2009; Packett et al., 2009; Smith et al., 2008). Comparable studies in the Burdekin catchment (Bainbridge et al., 2010; Maher et al., 2008) showed that the bulk of the suspended sediment reaching the mouth of the Burdekin River came from only two out of the five major sub-catchments. Studies in cropping lands have clearly shown that the overwhelming source of DIN discharge is from fertiliser use (Mitchell et al., 2009; Waterhouse et al., 2012; Thorburn et al., 2011b; Thorburn and Wilkinson, in press).

4. Transport and fate of pollutants in the GBRWHA

4.1. The dispersal of land-derived pollutants

Flooding river discharges into the GBR lagoon typically form extensive plumes (Brodie et al., 2010; Devlin and Brodie, 2005; Devlin and Schaffelke, 2009). The plumes of the 2010/2011 wet season were extraordinarily prolonged and covered a large area (Fig. 2). Short-term concentrations of both particulate and dissolved nutrients in flood volumes are extreme compared to non-flood conditions (in the range 2–30 μM nitrogen and 0.1–1 μM phosphorus) (Devlin and Brodie, 2005). However, the distribution and movement of the individual constituents in a flood plume varies considerably between the wet and dry tropic rivers. Wet Tropic Rivers (e.g. Tully River) have some flow throughout the dry season and regular, high flow in the wet season with rapid flushing times.
The Wet Tropic Rivers are generally fresh to the mouth of the river, where they discharge into the adjacent coastal seawater. Dry Tropics Rivers (e.g. Burdekin River) have negligible or no flow during the dry season and can have substantial upstream tidal seawater intrusions. The wet season discharges of these rivers vary enormously between years, and can form very large flood plumes extending far into the GBR lagoon. Flood plumes move in response to prevailing weather conditions along and across the coastal shelf with the plume often forming an estuarine mixing zone far away from the coast.

River runoff also transports agricultural pollutants such as pesticides (predominantly herbicides) into the GBR lagoon. High concentrations of herbicides have been detected in flood plume waters, which for periods of weeks exceed water quality guidelines for the protection of aquatic life (Bainbridge et al., 2009a; Brodie et al., 2012; Lewis et al., 2009a, 2012a; Shaw et al., 2010; Kennedy et al., 2012; Shaw et al., 2012). The composition of compounds indicated their land use source and most herbicides were associated with sugar cane cultivation (Lewis et al., 2009a).

4.2. Fate of pollutants in the marine environment

The fate of the discharged pollutants in the marine environment has been elucidated through monitoring and modelling activities as well as the analyses of remote sensing imagery (e.g. Devlin et al., 2012; Brodie et al., 2010; Devlin and Brodie, 2005; Devlin and Schaffelke, 2009; Lewis et al., 2009a). Material dynamics can be partially understood from studies of the concentration changes occurring in the plume as mixing with seawater progresses (Dagg et al., 2004). Most suspended solids and particulate nutrients settle...
from the plume quickly and are deposited within a few kilometres of the river mouth (e.g. for the Fitzroy River see Webster and Ford, 2010). In plumes from the Burdekin River, suspended solids concentrations can drop from more than 500 mg L\(^{-1}\) in the river at zero salinity close to the river mouth to less than 10 mg L\(^{-1}\) at salinities around 5–10 psu (Bainbridge et al., 2012). Fine benthic sediment is continuously resuspended in shallow waters (<10 m) by the prevailing south east wind regime and tidal currents and transported north along the coast (Larcombe et al., 1995; Radke et al., 2010). Coarser sediments are mostly retained near the coast in sand bars, beach ridges and subaqueous dunes (Ryan et al., 2007).

Dissolved inorganic nutrient concentrations are relatively high in peak flow conditions in the GBR rivers, particularly those rivers that drain fertilised agriculture (e.g. Mitchell et al., 2005, 2009). Dissolved nutrients move conservatively through the low salinity areas of the estuarine plume, indicating very little biological uptake. However, in areas of higher salinity (25–36 psu) towards the offshore boundary of the plume, there is higher biological uptake of nutrients (Brodie et al., 2010; Devlin and Brodie, 2005; Devlin and Schaffelke, 2009). Dissolved inorganic nutrients are not taken up biologically close to the river mouth because phytoplankton growth is light limited in the highly turbid plume waters (suspended solids >10 mg L\(^{-1}\)) (Turner et al., 1990). Thus, inorganic nutrients are transported over long distances, often greater than 50 km, exposing inshore reefs to high inorganic nutrient concentrations for short periods of time (Brodie et al., 2006; Devlin and Brodie, 2005; Devlin et al., 2010; Rohde et al., 2006; Schaffelke and Slivkoff, 2007; Schroeder et al., 2009). However, the particulate nutrients are initially transported only short distances from the river mouth before `settling out' from the plume through gravity controlled processes and flocculation (Bainbridge et al., 2012). Satellite images and plume sampling reveal that algal blooms develop when turbidity declines to values of less than 10 mg L\(^{-1}\) and clear skies allow increased phytoplankton photosynthesis (usually 2–5 days after the peak discharge occurs) (Bainbridge et al., 2012; Brodie et al., 2010).

Recently, coral proxies (Ba, Y, Mn) have provided evidence of increased sediment export to the GBR lagoon from the Burdekin River catchment (Lewis et al., 2007; McCulloch et al., 2003) while nitrogen isotopes in the (insoluble) organic component of the coral skeleton have been used to quantify increases in nutrient loads to ecosystem response. For example the SLIM model, coupled with long-term water turbidity data from Cleveland Bay in the central GBR, revealed that tradewinds only slowly transport terrigenous fine inshore sediments during the dry season out of the bay, leading to net accumulation (Lambrects et al., 2010). Before European settlement, Cleveland Bay corals grew in waters with clarity sufficient for benthic production for at least 200 days a year as opposed to only 60–90 days at present (Brodie et al., 2012; Lambrects et al., 2010), demonstrating the influence of human-induced increased riverine fine sediment inflow to the GBR lagoon.

4.4. Pollutant residence times

Residence time, i.e. the amount of time that a parcel of water spends in a semi-enclosed waterbody before it is transported to the open ocean, is a widely used metric to understand and predict the effects of contaminants in estuarine and coastal waters (Jickells, 1998). Longer water residence times promote an increasing build-up of land-derived pollutants. Perhaps the most important factor determining susceptibility of coastal ecosystems to adverse effects from land-derived pollutants is the amount of exchange between the water body and the open ocean. Thus water bodies with low exchange rates with the ocean seem to be particularly vulnerable to the effects of pollution.

There have been a number of previous studies of residence times or flushing times of water (and implicitly of pollutants) in the GBR lagoon. Hancock et al. (2006) used radium isotopes as tracers to estimate that inner lagoon (water) flushing times in the southern GBR were 18 days and in the central GBR 45 days. Luick et al. (2007) used hydrodynamic models with simulated neutrally buoyant tracer particles to estimate that residence times can vary from ca. 1 month to 1 year. They note that these times are applicable to contaminants only if the substances are in solution in the
water column. Wang et al. (2007) used salinity (from archival records) as a tracer to assess flushing times of solutes and pollutants in the central GBR lagoon. Choukroun et al. (2010) used drifters deployed in the western Coral Sea to assess surface circulation in this area and residence times in the GBR. The consensus of these studies is that water residence times for the GBR are relatively short and rapid flushing with residence times of the order of weeks up to possibly one year. However, Andutta et al. (2011) have shown that in bays along the coast hypersaline waters escape sideways by the residual longshore southward currents, and are transferred from bay to bay until steady-state conditions are reached after about 100 days, indicating a longer residence time of inshore waters.

The residence times and flushing rates of pollutants, especially those that behave in a non-conservative mixing regime, are not necessarily linked to the movement of the water (Brodie et al., 2012). The key pollutants discharged to the GBR (suspended solids, reactive nitrogen, phosphorus and herbicide residues) behave in a non-conservative way during transport and mixing (Brodie et al., 2012). The bulk of these pollutants are exported into the GBR during a period of a few days per year during river floods and remain in the GBR lagoon for long periods (measured in years to decades) and are biologically active and capable of producing adverse effects on GBR ecosystems for long periods (years to decades; Brodie et al., 2012).

5. Spatial and temporal distribution of pollutants in the GBR

The water quality in the GBR lagoon is characterised by large temporal and spatial variations. On a spatial scale, distinct cross-shelf gradients from the coast to the shelf-break are the most noteworthy feature. Concentrations of particulate water quality variables such as TSS, PN and PP, Chl a and Secchi depth (a measure of water clarity) are on average 0.6- to 5-fold higher in the coastal and inshore areas compared to the offshore lagoon, especially adjacent to the Wet Tropics and Burdekin regions in the central GBR (Brodie et al., 2007; De’ath and Fabricius, 2008, 2010). These gradients are less pronounced in the far Northern region of the GBR (Brodie et al., 2007; De’ath and Fabricius, 2008, 2010) and also for dissolved nutrients (Furnas et al., 1995, 1997). Repeated local-scale water quality measurements in the inshore lagoon also show distinct water quality gradients away from the coast (Cooper et al., 2007; Schaffelke et al., 2003, 2012). Modelling of the catchment to lagoon connection showed a strong correlation between the North to South enrichment gradient of Chl a and the concentration of dissolved inorganic nitrogen in riverine flood waters (Wooldridge et al., 2006). Mid and outer shelf concentrations of Chl a also vary from south to north, but not in a consistent latitudinal gradient and are likely to be driven by external factors such as upwelling, currents and tidal mixing (Brodie et al., 2007).

On a temporal scale, the water quality in the coastal and inshore GBR lagoon can change dramatically for short periods of time. Elevated concentrations of nutrients (both dissolved and particulate) and of suspended solids are observed in waters affected by riverine flood plumes during the summer wet season (Devlin and Brodie, 2005; Devlin and Schaffelke, 2009; Devlin et al., 2012) (see also discussion in Sections 4.2 and 4.3) and after resuspension of bottom sediments by strong winds or tidal currents (Furnas, 1989). The nutrients introduced or released during these events are rapidly taken up by pelagic and benthic algae and microbial communities (Alongi and McKinney, 2005), sometimes fuelling short-lived phytoplankton blooms and high levels of organic production (Furnas, 1989; Furnas et al., 2005, 2011). This organic matter is cycled through the marine food web and transformed e.g. into marine snow particles that may be deposited on benthic communities, such as coral reefs, and influence their structure, productivity, and health for long periods; this ultimately decouples event-driven inputs of nutrients from their long-term ecosystem effects (see for example, Anthony and Fabricius, 2000; Fabricius and Wolanski, 2000; Fabricius et al., 2003).

The variability of water quality parameters in the inshore GBR lagoon is influenced by a complex interplay of these temporal and spatial factors (Schaffelke et al., 2012). Concentrations of nutrients, suspended sediments and Chl a are influenced by season with higher concentrations during the summer months, especially during years with high rainfall and at sites exposed to river runoff. During winter, water quality variables, especially nutrients, show very low concentrations (i.e. close to or below detection limits), except for periods of strong winds leading to resuspension of settled fine sediments which intermittently increases water turbidity (Schaffelke et al., 2012). Compared to other coral reef areas (mainly in the Caribbean and Florida), water column concentrations of suspended solids, Chl a and soluble reactive phosphorus are higher in the inshore GBR, while concentrations of dissolved nitrogen are lower (see references in Schaffelke et al., 2012).

Knowledge of the spatial and temporal extent of pesticide distribution in GBR waters has expanded considerably in recent years. Monitoring using the time-integrating passive sampler devices (Shaw and Müller, 2005) detected low levels of herbicides at all sampling sites in the inshore GBR lagoon throughout the year, generally with higher concentrations in the wet season (Kennedy et al., 2012; Shaw et al., 2010). Herbicides generally also showed distinct cross-shelf gradients with higher concentrations close to the coast, as did other water quality constituents (see above). While herbicide concentrations do not exceed water quality guidelines during most of the year, they are still detectable, and the consequences for marine life of low level chronic exposure to a mixture of herbicides are unknown (see risk assessment in Lewis et al., 2012a). The detection of herbicides during the dry season is surprising. Only in the Wet Tropics region, where rivers are flowing throughout the year albeit with much reduced flow during the dry season, can some export of herbicides be expected. However, the persistence of pesticides in tropical marine water is unfortunately very poorly understood. It is possible that compounds adsorb to settling particulate matter, which would reduce light-dependent degradation, and are intermittently resuspended back into the water column by wind. Longer than expected persistence of herbicides is also indicated by finding detectable concentrations of herbicides in the far Northern GBR (Kennedy et al., 2012), far distant from their likely agricultural sources. These herbicides are likely transported northward by longshore water currents (e.g. Luick et al., 2007) from agricultural catchment sources adjacent to the central GBR.

6. Ecological effects of pollutants in the GBRWHA

Riverine loads of excess nutrients, sediments and other pollutants clearly influence the water quality in the coastal inshore GBR. This has changed the environmental conditions for GBR species and ecosystems in the long-term, i.e. chronic changes in marine areas exposed to land runoff from developed catchments since European settlement, and intermittently, i.e. acute changes after major flood events. The understanding of the effects of water quality changes on GBR species and ecosystems has improved enormously over the last decade. Recent authoritative reviews are available of the evidence for causal relationships between water quality change and ecosystem health for corals, seagrasses and mangroves (e.g. Brodie et al., 2008b) and of the effects of eutrophication and increased sedimentation on coral reefs and factors influencing the susceptibility and resilience of reefs to these
and inshore GBR is the reduction of light availability (Collier et al., 2005). The symptoms of this decline are many sites with reduced seagrass abundance, shrinking meadow area, reduced seed production and increased epiphyte loads. The environmental conditions at many sites point towards light limitation of seagrasses and nutrient enrichment, which is also reflected in long term increases of seagrass tissue nutrient content, particularly in the Wet Tropics and Burdekin regions (McKenzie et al., 2010; Mellors et al., 2005). The most common cause of seagrass loss in the coastal and inshore GBR is the reduction of light availability (Collier et al., 2012), principally caused by chronic and intermittent increases in suspended sediments and organic particles leading to increased turbidity (Schaffelke et al., 2005). In contrast, the direct effects of increased dissolved nutrients on GBR seagrass health are less understood but current nutrient loadings in the GBR seem to have not reached critical levels for seagrasses (Waycott et al., 2005). The herbicide diuron, which is commonly found in GBR coastal and inshore waters (see above) adversely affects seagrass productivity at concentrations such as those measured in flood plumes (McMahon et al., 2005; Haynes et al., 2000) but the role of herbicides in the currently recognised seagrass decline in the GBR is unknown. The capacity of seagrass meadows to recover from the decline cannot be predicted but will be controlled by the interactions between light availability, nutrient loads and the availability of seeds to form the foundation of new populations (McKenzie et al., 2010).

6.2. Mangrove and benthic microalgal ecosystems

Only few studies have focused on the effects of changing water quality on mangroves in the GBR region. Significant increases in mangrove areas along the GBR coast have been reported in recent decades, correlated with catchment development such as vegetation clearing causing higher sediment loads in runoff and the construction of a major river barrage (on the Fitzroy River) reducing river flows (see review in Schaffelke et al., 2005). The unusual dieback of a mangrove species in the Mackay region from the mid-1990s to the early 2000s was initially hypothesised to be associated with the high levels of diuron and other herbicides present in mangrove sediments and porewater, and in stream/drain waters flowing into mangrove areas (Duke et al., 2005; Duke, 2008). However, recent surveys of the affected mangrove communities have shown an overall improvement in their health despite herbicide levels remaining high at some sites (Wake, pers. comm.). These conflicting findings suggest that cause and effect relationships for these diebacks are yet to be fully understood.

The effect of herbicides on benthic microalgae has been studied by Magnusson et al. (2008, 2010, 2012). They found that growth rates and photosynthesis in two tropical benthic microalgae; Navicula sp. (Heterokontophyta) and Nephroselmis pyriformis (Chlorophyta) were affected at diuron concentrations that have been detected in coastal areas of the GBR. For the three PSII-inhibiting herbicides – diuron, hexazinone and atrazine – the order of toxicity (EC50 range) was diuron (16–33 nM) > hexazinone (25–110 nM) > atrazine (130–620 nM) for both algal species (Magnusson et al., 2008). Binary mixtures of herbicides showed additive toxicity (Magnusson et al., 2010) and longer exposure studies (4 weeks) to diuron in these microalgal communities showed the development of tolerance to diuron during this period, accompanied by a shift in species composition towards communities dominated by diatoms (Magnusson et al., 2012). This showed that chronic pesticide pollution can induce shifts in community structure.

6.3. Coral reefs

Of the three ecosystem types discussed here, the responses of coral reefs to changed water quality are the best understood due to a large, ongoing research effort, especially in the GBR (e.g. Cooper et al., 2009; Fabricius, 2011; Fabricius et al., 2012). Many coastal and inshore coral reefs around the world are affected by eutrophication and increased sedimentation (Fabricius, 2005). Nutrient enrichment, increased turbidity and sedimentation, especially of organic-rich suspended matter lead to trophic changes in the ecosystem, reduced coral recruitment and diversity, the replacement of corals by organisms that compete with corals for space and resources such as macroalgae and filter feeders and more frequent outbreaks of coral-eating crown-of-thorns starfish (Brodie et al., 2005; Fabricius, 2005, 2011 and references therein; Fabricius et al., 2010; Wismer et al., 2009). Recent results also suggest that increasing suspended sediment in coral reef environments may reduce settlement success or survival of coral reef fishes. This disruption of suitable habitat selection for juvenile fish may compound the effects of habitat loss on coral reefs (Wenger et al., 2011).

The effects of chemical pollutants on reef-building corals were recently reviewed by van Dam et al. (2010) and recent research has also looked at the potential effects of pesticides on foraminifera (van Dam et al., 2012) and reef fish (Botté et al., 2012). While insecticides can affect survival, reproduction, and early life stages of corals (Markey et al., 2007), these are rarely detected in GBR waters (Kennedy et al., 2012). In contrast, herbicides are widely found in the coastal and inshore GBR waters (see above: Kennedy et al., 2012; Lewis et al., 2012a). Corals can recover from short-term low level exposure to herbicides, however chronic exposure can lead to decreased photosynthetic rates, bleaching, partial colony mortality, reduced tissue lipid content and reduced fecundity. In addition, concentrations during flood events are sufficiently high for effects to occur (Lewis et al., 2009a, 2012a).

The effects of multiple stressors on corals are also emerging. Flood plumes generally have simultaneously low salinities, with elevated concentrations of nutrients, suspended sediments and several pesticides, and additive or even synergistic effects between pollutants but also with other environmental factors such as elevated temperatures may occur and affect the health of coral reefs (Lewis et al., 2012a; Negri et al., 2011; Wooldridge, 2009; Wooldridge and Done 2009; Wooldridge et al. 2011). However, not all reefs are equally vulnerable to degradation from exposure to land run-off and pollution; both the exposure to nutrients, sediments and pollutants and site-specific environmental conditions determine their susceptibility. The most vulnerable habitats are reefs close to river mouths (especially at larger depths); poorly flushed reefs; reefs on shallow continental shelf areas (especially if surrounded by fine, soft sediment areas prone to resuspension); reefs disturbed in the past (e.g. by severe storms and coral bleaching) and reefs with low abundances of herbivorous fish (Fabricius, 2011).

A number of linked studies have examined the growth dynamics of close coastal turbid reefs, either using the history of the reefs through the Holocene (from reef cores) or short term (a few years) studies on the reefs themselves, to assess their response to current turbid conditions. The reefs considered include Dunk Island near Tully (Perry and Smithers, 2010; Perry et al., 2011); Middle Reef near Townsville (Browne et al., 2010); Paluma Shoals north of Townsville (Perry and Smithers 2006; Perry et al., 2008; Palmer et al., 2010); King Reef near Tully (Roche et al., 2011); Lugger Shoals near Tully (Perry and Smithers 2006; Perry et al., 2009) and all these
reefs plus others (22 in total) in a combined analysis (Perry and Smithers, 2011). Overall it appears most of these close coastal reefs are resilient to the turbid conditions in which they exist, but that over the Holocene they have experienced periods of growth (‘turn on’) and growth cessation (‘turn off’) driven by environmental conditions. Their current status may not be affected by the increased loads of terrestrial sediment from anthropogenic catchment erosion as a result of this resilience that has developed over time.

The now advanced understanding of the responses of coral reef organisms to water quality in the GBR has recently been applied in the development of local water quality guidelines (GBRMPA, 2009). The guideline trigger values for nutrients, suspended solids, Chl a and Secchi depth are based on the analysis of relationships between large-scale data on water quality and coral reef biodiversity and if exceeded, changes to ecosystem properties can be expected, e.g. increasing cover of macroalgae or decreasing taxonomic richness of corals (Brodie et al., 2011; De’ath and Fabricius, 2008, 2010). Current GBR water quality monitoring data are reported in the context of compliance with the GBR water quality guidelines (e.g. Schaffelke et al., 2012).

7. Current management response to land runoff from the GBRCA

During the 1980s and 1990s, research and monitoring in the GBRCA and GBRWHA identified land runoff of pollutants as a pressure to the health of the GBR and formed the baseline of our understanding of (i) the nature and sources of pollutant generation in the GBRCA, (ii) the transport of pollutants from the GBRCA into the GBR, (iii) the effects of pollutants on specific GBR organisms and ecosystems, (iv) management options to reduce pollution, and (v) the socio-economic and political realities of how improved management could be achieved. By the early 2000s, a reasonable consensus existed on these issues and how they could be addressed by improved management (e.g. Brodie et al., 2001a,b; Furnas, 2003; Great Barrier Reef Protection Interdepartmental Committee Science Panel, 2003; Productivity Commission, 2003; Williams et al., 2002). In particular, the ‘Great Barrier Reef Water Quality Action Plan’ (Brodie et al., 2001a), commissioned by the Federal Minister for the Environment, set some initial targets for reducing pollution loads from the GBRCA to the GBRWHA and spurred further action, including the development of Reef Plan (see below).

All of these initiatives focus on diffuse pollution, based on the assumption that point source pollution such as sewage are already well managed and in most cases, regulated. However, this was generally not the case until after 1991 when most island sewage treatment plans that discharge into the GBRWHA were required to upgrade to a tertiary treatment standard (Brodie, 1994; Waterhouse and Johnson, 2002) and ongoing upgrades to coastal sewage plants were required under Queensland Government policy (Queensland State of the Environment Report, 2007). Regulations introduced in 2000 targeted aquaculture discharge into the GBRWHA which also improved the management of point source discharges into the GBR.

7.1. Overarching policy – Reef Plan

In 2003, the Australian and Queensland Governments released the Reef Plan (Queensland Department of the Premier and Cabinet, 2003). This plan aims to halt and reverse the decline in water quality entering the Reef within 10 years, and strictly focuses on diffuse pollution from agriculture. To achieve its aim, the Plan states the following two objectives: (i) Reduce the load of pollutants from diffuse sources in the water entering the Reef, and (ii) Rehabilitate and conserve areas of the Reef catchment that have a role in removing water borne pollutants. The Plan outlines a set of activities to be carried out by multiple designated participants that would lead to an actual plan of on-ground activities.

In 2009, Reef Plan 2003 was revised and updated (Queensland Department of the Premier and Cabinet, 2009) with better defined targets and actions. In addition to Reef Plan’s 2003 aim, Reef Plan 2009 also aims to ensure that by 2020 the quality of water entering the GBR from adjacent catchments has no detrimental impact on the health and resilience of the GBR. The Reef Plan 2009 includes targets and goals for water quality improvement and management practice change by 2013 and 2020.

Although load reduction targets are not formally defined in Reef Plan, for Reef Plan reporting purposes they are considered to have been set for anthropogenic loads (i.e. current loads minus pre-European loads). The load reduction targets are (i) a minimum 50 per cent reduction in nitrogen and phosphorus loads at the end of catchments by 2013, (ii) a minimum 50 per cent reduction in pesticides at the end of catchments by 2013, (iii) a minimum of 50 per cent late dry season groundcover on dry tropical grazing land by 2013, and (iv) a minimum 20 per cent reduction in sediment load at the end of catchments by 2020. As the current end date for Reef Plan draws near (2013) a new version (Reef Plan 2014) is in the initial stages of investigation.

7.2. Incentives for improved management – Reef Rescue

In 2007, the Federal Government implemented Reef Rescue, an AU $200 million investment for on-ground works, monitoring, research and partnerships over 5 years (Australian Government, 2007). This voluntary program’s objective is to improve the water quality of the GBR lagoon by increasing the adoption of land management practices that reduce the run-off of nutrients, pesticides and sediments from agricultural land. Reef Rescue was developed following an initiative by the Reef Water Quality Partnership, including agricultural industries, regional NRM bodies and natural resource and environment managers. It built on key activities conducted under Reef Plan, including the development of local and regional Water Quality Improvement Plans (e.g. Dight, 2009; Drewry et al., 2008; Kroon, 2009 and references therein), and the Nutrient Management Zones process (Brodie, 2007).

Whilst forming an integral component of Reef Plan 2009, Reef Rescue has its own five-year outcome targets (i.e. by 2013). Both initiatives specify management action, catchment condition and end-of-catchment pollutant load targets for 2013 reported by catchment, regional and GBR-wide scales (see Table 1).

In 2008, Reef Rescue funded many on-ground land management projects across the GBRCA, mainly in the sugarcane and grazing industries but also in dairy farming and horticulture. Projects include the introductions of new farming practices; fencing along streams for cattle management with off-stream watering points; machinery modifications including harvesters, fertiliser and pesticide application gear; and cultivation and tillage equipment and practices.

7.3. Agricultural regulation – Great Barrier Reef Protection Amendment Act 2009

In 2009, the Queensland Government introduced the Great Barrier Reef Protection Amendment Act 2009 (Reef Protection Package). The Act introduces regulations to improve the quality of water entering the GBR, and applies to sugarcane growing and cattle grazing properties in the Burdekin Dry Tropics, Wet Tropics and Mackay Whitsunday catchments in North Queensland. In these areas, the Act provides for the implementation of (i) Farm Environmental Risk Management Plans in sugarcane cultivation and beef grazing, (ii) Fertiliser management in sugarcane through a calcula-
tor' for sustainable fertiliser rates, (iii) Erosion management in grazing through managing pasture cover, and (iv) Pesticide management through application management and buffer strips. The Act and its regulations were implemented during 2010 and 2011. The pesticide regulations are of particular interest in that the Queensland Government has largely moved ahead of the Australian Government pesticide regulatory regime by introducing strong management measures for the GBR which have not been enforced in other parts of Australia (King et al., in press).

7.4. Monitoring and reporting

The success (or otherwise) of Reef Plan 2009, Reef Rescue and the Reef Protection Package is being assessed using an integrated monitoring, assessment and reporting program referred to earlier – the Paddock to Reef Program (Carroll et al., 2012; Paddock to Reef Program, 2009). The program commenced in 2009 and the first report card was released in September 2011 (The State of Queensland, 2011). The program is built around a number of components including (a) management practice adoption monitoring and auditing; (b) paddock monitoring and modelling involving collecting runoff during actual rainfall events and rainfall simulation. Modelling is used to extend results from one situation to another not part of the monitoring scheme; (c) catchment monitoring and modelling to assess the water quality entering the GBR lagoon and to determine trends in water quality over time; identify potential source areas of contaminants; link plot to paddock to river scales; and validate and calibrate the existing catchment models; (d) marine monitoring including inshore biological monitoring of inshore coral reefs and intertidal seagrass meadows; and inshore water quality and flood plume monitoring focussing on TSS, nutrients, Chl a, salinity, pesticides, temperature, turbidity and light conditions; and (e) reporting on progress through an annual ‘Report Card’ supported by detailed technical reports.

Water quality guidelines for the GBR (GBRMPA, 2009) were developed from previous analysis of likely candidate measures (De'ath and Fabricius 2008; Moss et al. 2005). The development of new and improved water quality indicators has progressed including the use of coral pigmentation (Cooper and Fabricius, 2012), biofilms (Kriwy and Uthicke, 2011) and a bioindicator index system (Fabricius et al., 2012). Bioindicator and index systems for multiple pesticides have also been developed (Lewis et al. 2012a; Shaw et al. 2012) and some are in use (see Smith et al., 2012; Kennedy et al., 2012). Criteria for assessing the eutrophication status of the GBR are developed to a draft state (Brodie et al., 2011).

8. Social and economic research for improved management of the GBRCA and lagoon

There has been a proliferation of social and economic research over the past few years aiming to support the Reef Plan’s goal to halt and reverse the decline in water quality entering the GBR. Guided by the results of biophysical research, social and economic research at the GBR-scale supported, as stated in section 9.1, prioritisation of management responses under a constrained budget (e.g. Cotsell et al., 2009).

Detailed socio-economic research has focused on specific land uses, industries and geographical regions identified as ‘water quality hotspots’. For example, to support development and delivery of local and regional Water Quality Improvement Plans (WQIPs) NRM organisations have worked closely with multidisciplinary teams of researchers who could provide the biophysical and socio-economic science underpinning these plans (e.g. Kroon, 2009 and references therein).
To date, a major research effort has focused on selecting and prioritising land use and management practices that have high potential to improve water quality and provide benefits to growers (e.g. improved crop yields and higher gross margins) and the GBR (e.g. reduced impact from fertiliser run-off) (Bohnet et al., 2008a; Bohnet et al., 2011a; Roebeling et al., 2009a,b). Other research has focused on mechanisms (e.g. regulations, incentives) and policy options (e.g. spatially explicit, industry based) for implementing land use and management practice change (Coggan and Whitten, 2008; Reeson et al., 2011; Rolfe and Whitten, 2011a,b; Rolfe et al., 2011; Smajgl et al., 2009, 2010; Star et al., 2011; van Grieken et al., 2011a, b; Windle and Rolfe, 2011). To maximise the likelihood of adoption of management practices and policies that improve water quality, social research has provided insights into land managers motivations, risk perceptions and other factors that influence their land management decision-making (Bohnet et al., 2011b; Flick et al., 2010; Greiner and Miller, 2008; Greiner et al., 2009). In addition, landholder profiles and typologies have been developed to better tailor policies and improve agricultural extension program design (Bohnet, 2008; Bohnet et al., 2011b; Entage, 2009).

To deal with and reduce uncertainty of the numerous factors that influence water quality in the GBRCA and lagoon and the effects that changes in water quality may have on reef ecosystems, a number of different assessment approaches have been applied in the GBR context. Based on literature reviews, Sherman and Henderson (2008) developed a guide for addressing the margins of safety and reasonable assurance requirements for marine and estuarine water quality protection. Whitten et al. (2008) developed reasonable assurance points of reference for water quality management based on targets for practice change adoption. Using a Bayesian Belief Network model, Lynam et al. (2010) illustrate how adaptive modelling with managers and scientists has been used as a tool to reduce uncertainty and guide where, when and what interventions are most likely to achieve desired water quality outcomes. Finally, scenarios or visions have been developed for the future of the region at different scales to provide insights into long-term risks and opportunities that are likely to affect a complex region such as the GBRCA (Bohensky et al., 2011; Bohnet, 2010; Bohnet et al., 2008b, 2010). Importantly, these studies have brought a wide range of stakeholders and local communities together and thereby contributed to social and ecological knowledge integration and social and collective learning.

The importance of institutional arrangements that allow the numerous stakeholder groups to contribute and commit to successful implementation of strategies for water quality improvement has been recognised through a range of research efforts. Collaborative partnerships for effective governance have been studied at a range of scales (Lane and Robinson, 2009; Robinson et al., 2011a) and evaluated using a SMART (Specific, Measurable, Achievable, Relevant and Timed) assessment (Robinson et al., 2009). A review by Kroon et al. (2009) highlight the challenges of integrating scientific and local knowledge into a local water quality improvement plan. However, by incorporating the range of local needs, values, aspirations and priorities, water quality improvement plans are more likely to be supported by local communities (Bohnet and Kinjun, 2009; Larson, 2009).

Efforts to monitor, report and adapt management strategies for water quality improvement have included development of a water use benefit index that the community can use as a tool to monitor water related trends in the GBRCA (Smajgl et al., 2009). To adapt management strategies if, when and where required, Eberhard et al. (2009) developed a protocol to guide practical steps towards an adaptive approach to water quality management. Also advocating an adaptive management approach to water quality management, Broderick (2008) suggests that success should be judged on its environmental outcomes and on the quality of learning.

More recent research has applied an ecosystem services approach to map the value of riparian vegetation (Pert et al., 2010), analyse trade-offs between multiple ecosystem services and stakeholders (Butler et al., in press), and assess the economic value of the GBR (Stoeckl et al., 2011). Other recent socio-economic work has focused on a range of issues related to tourism and the GBR (Coughlan, in press; Farr et al., 2011; Gregg and Greiner, 2008; Kragt et al., 2009; Pabel and Coghlan, 2011).

While some of the social and economic research reported here has directly supported local water quality improvement plans by providing the underpinning science (see for example Kroon, 2009 and references therein), other studies have contributed to the social and economic knowledge base in the GBRCA and lagoon. Besides providing critical baseline information for future work, research in the social and economic area has contributed to building partnerships, knowledge integration (social, economic and ecological as well as local and scientific) and learning about the social-ecological catchment to reef system. Research to date will form the basis for future work so that through adaptive management (e.g. Eberhard et al., 2009) we, i.e. scientists, government, NRM managers, local communities, and other stakeholders, can collectively assess the efficacy of recommended management practices and explore new cost-effective options to improve water quality if and where required (Kroon and Brodie, 2009). However, in order to achieve this important goal, future research will have to be better coordinated, synthesised and also focused on social and collective learning.

9. Evaluation of the current management response and considerations for improved management in the GBR catchment

9.1. Prioritisation of water quality pollutants

Currently the prioritisation of management response between different pollutants, different land uses/industries and different regions has used relatively crude methods such as Multiple Criteria Analyses and relatively simple non-weighted assessments of several parameters (Brodie and Waterhouse, 2009; Brodie et al., 2009b; Cotsell et al., 2009; Greiner et al., 2005; Waterhouse et al., 2012). While these have proved useful for the initial prioritisation of investment under Reef Rescue and selection of priority management areas under the Reef Protection Package, more sophisticated analyses are needed to confidently prioritise between pollutants. For example, a robust discussion is currently occurring globally as to the necessity of reducing nitrogen loads or phosphorus loads or both to prevent eutrophication in estuarine, coastal and marine environments (Conley et al., 2009; Howarth et al., 2011; Paerl, 2009; Schindler et al., 2008). Such considerations are also relevant to the GBR but a comprehensive risk assessment that would allow us to prioritise between different pollutants has not been carried out. The GBR lagoon is believed to be generally N limited (Furnas et al., 2005) but it is unclear as to whether N or P is the first priority for management in the GBRCA or whether both should be a priority. Current targets for N and P under Reef Plan (2009) and Reef Rescue treat N and P as equally important but this is not based on robust scientific analyses of relative importance, but does draw on best available knowledge at the time. Currently a more sophisticated risk analysis has commenced, using improved methodology and data inputs, to be completed by 2014 and able to be used for the development of Reef Plan 2014.
9.2. New management practices and their effectiveness

A large amount of recent research has focussed on losses of chemicals from cropping systems and how new and improved agricultural management practices in the GBRCA can reduce these losses. The effectiveness of currently recommended practices as well as newer Best Management Practices has also been assessed. For example, in sugarcane cropping, Robinson et al. (2011b) showed that sugarcane as a crop is particularly inefficient at using applied nitrogen fertiliser that is present as nitrate in the soil rather than ammonium and thus prone to lose large amounts of nitrate to off-farm environments. Scott et al. (2010) showed the potential for enzymes to destroy atrazine in irrigated sugarcane tailwater capture pits. Thorburn and colleagues have long-studied the possibility of using the ‘nitrogen replacement’ technique to better manage fertiliser use in sugarcane growing and reduce the current losses of nitrogen from the sugarcane growing system (Biggs et al., in press; Park et al., 2008; Thorburn et al., 2007, 2008a,b, 2009, 2011a–c; Webster et al., 2008, 2012) and the current recommended practice, ‘Six Easy Steps’ has also been evaluated (Schroeder et al., 2010). Armour et al. (in press) showed large losses of nitrate in deep drainage under banana crops in the Wet Tropics with far lesser amounts under sugarcane in similar climatic conditions. However as a result of the research, fertiliser usage in banana crops has declined by as much as 40% since 1995. Masters et al. (in press) showed that Best Management Practices in sugarcane including controlled traffic resulted in load reductions of 60%, 55%, 47%, and 48% for ametryn, atrazine, diuron and hexazinone respectively. Herbicide losses in runoff were also reduced by 32–42% when applications were banded rather than broadcast. Overall, it was the combination of early application, banding and controlled traffic that was most effective in reducing herbicide losses in runoff. The use of a controlled traffic system and banded application of pesticides also gave useful reductions in pesticide runoff under simulated storm rainfall in a cotton cropping system in the Fitzroy catchment (Silburn et al., 2011a,b) and banding was shown to be most effective soon after application. The much greater application rates for two of the herbicides studied than for another resulted in 50–120 times greater runoff losses soon after spraying.

In rangeland beef grazing lands, research into the effectiveness of pasture cover as an erosion prevention management technique have shown that grazing in semi-arid pastures should be managed to maintain >50% ground cover to avoid excessive runoff and soil erosion, degradation of soil productivity and to maintain good off-site water quality (Silburn, 2011a,b; Silburn et al., 2011a). Erosion studies in grain cropping lands (Freebairn et al., 2009) have demonstrated the importance of soil type and conditions of erosion rates. Murphy et al. (in press) showed that considerable quantities of sediment, nutrient (from fertiliser) and herbicides (in this case metolachlor) can be lost under conventional management in dryland grain cropping. However improved practices such as reduced or zero tillage can reduce losses substantially.

Other research has investigated the role and effectiveness of riparian forests and wetlands (constructed and natural) on trapping catchment pollutants before they reach the GBR (e.g. McJannet et al. 2011a,b). They found in a case study in the Tully/Murray River system that a small natural wetland trapped little sediment, some phosphorus for at least a time, but no nitrogen. Thus the benefits of using wetlands to improve water quality may not be evident in unmanaged natural wetland systems when there is strong seasonality in flows and short residence time during the periods of maximum sediment and nutrient loads.

9.3. Time lags in responses

It is well understood that there are long and complex time lags in the response of GBR ecosystems to pollutant loading and thus also time lags in the response of ecosystems to reductions in pollutant loading (e.g. Bainbridge et al., 2009b; Brodie et al., 2009c). Similar problems exist across the globe (e.g. Hart, 2003; Lotze et al. 2011). Clear examples in the GBR context are:

1. Improved pasture cover, riparian vegetation restoration, constructed wetlands and other forms of vegetation management all require long periods to become effective in reducing sediment loads at the end of catchment (e.g. Bartley et al., 2010a,b).
2. Groundwater movement and pollutant transport is often a very slow process with potentially decadal time scales between chemical entry to the groundwater and eventual discharge to river or coastal waterbodies (Lenahan and Bristow, 2010; Rasi-ah et al., 2003, 2010, 2011a,b; Thayalakumaran et al., 2008).
3. Sediment transport through catchments can be a slow process with large-scale storage for long periods e.g. in the Fitzroy catchment (Hughes et al., 2010b) followed by remobilisation in large flood events.

It is assumed that water quality conditions in the inshore GBR lagoon have deteriorated over a long period in response to slowly increasing levels of terrestrial pollution. It is likely that time scales of decades will be required for marine water quality conditions to improve following reductions in catchment pollutant loading. Pesticides are likely to be an exception to this. The pesticides of concern, the PS-II herbicides have half lives of less than one year. If usage is greatly reduced on the GBRCA, concentrations in the GBR lagoon should decline rapidly (for example within 3 years) as the residual herbicides decay.

These time lags mean that even if all the management action proposed to occur under Reef Plan (2009) is successfully implemented, it is highly unlikely that the targets set in Reef Plan (2009) and Reef Rescue will be achieved by 2013 (Kroon, 2012), or even significant change in water quality indicators detected over time periods of several decades (Darnell et al., 2012).

The management of terrestrial pollutant discharge to the GBR implicitly assumes that the impacts of increased loads of nutrients, sediments and pesticides would be reversed if the loads were reduced. However, it is now well known that many ecological systems occur in stable states, shifts between states are nonlinear and the magnitude of change required to return a system to its original state is likely to be larger than the change that caused the shift towards the undesired state (Folke et al., 2004; Scheffer et al., 2001). In addition, there are well documented cases of eutrophied marine systems where reductions in nutrient loading have not returned the systems to their original ecological status (Duarte et al., 2009; Lotze et al. 2011). This is attributed to the range of other factors in the system that have dramatically changed also during the period of increased nutrient loading, such as human population increases, increased carbon dioxide in the atmosphere, freshwater runoff changes, global temperature increases and fish stock losses. These factors are also present in the GBR and may well interfere with attempts to return the GBR to a more desirable condition through pollutant load reductions alone. In coral reef systems the issues of reversibility, time lags and phase change has been the subject of much recent research (Bruno et al., 2009; Elmhirst et al., 2009; Hughes et al., 2010a; Mummy et al., 2007; Nors-tröm et al., 2009). However, further research is required on ecosystem responses to changing water quality responses, particularly in combination with other stressors such as climate change, to quantify the likely time lags of the response of the GBR ecosystems and the nature and trajectory of the response.
9.4. GBRCA freshwater ecosystem management versus GBR marine ecosystem management

A long-standing issue is that catchment management actions in the GBRCA have been directed at protecting GBR ecosystems with little attention given to the health of freshwater ecosystems. This is very evident in the objectives of programs like Reef Plan (2009) where freshwater wetlands and riparian areas are seen only in the context of trapping pollutants and hence reducing pollutant loads to the GBR (e.g. Flick et al., 2010). There is also the view that all management measures taken on catchments which are known to benefit GBR ecosystems will equally benefit freshwater ecosystems. There is little evidence to support this and the proposition has never been adequately analysed. Many stakeholders interested in terrestrial and freshwater ecosystem protection see the total emphasis on the GBR to be misplaced and the need for a more balanced management approach. If we consider riparian vegetation restoration, the varying benefits of this expensive activity need to be assessed for both the GBR and the catchment (Bohnet et al., 2011b; Pert et al., 2010). A good example of the conflicting (and sometimes complimentary) needs of management of agricultural pollution in different ecosystem compartments of the GBR region is the lower Burdekin region where agricultural pollutants from cropping (mainly sugarcane) and hydrological modifications impinge firstly on freshwater wetlands listed under the Australian national wetland register, then on a mainly estuarine Ramsar-listed wetland site and then on the GBRWHA (Davis et al., 2008, in press, 2012).

A number of methods are available that allow for joint consideration of terrestrial and marine biodiversity conservation in protected area management (Stoms et al., 2005; Watts et al., 2009) and the use of such an integrated approach to biodiversity conservation in the GBR region is desirable.

9.5. Achievement of targets

Ambitious targets for pollutant load reduction at end of catchment, and goals for GBR ecosystem health and resilience have been set by Reef Plan (2009) for 2013 and 2020 (Table 1). However, it is unclear whether achieving these targets and goals will result in desirable coastal ecosystem condition in the GBR lagoon. Thorburn and Wilkinson (in press) address the question whether the management practices currently being funded under Reef Rescue and imposed by the Reef Protection Package are likely to be effective in reducing sediment and nutrient loads at catchment scales. They find that management practices to improve cover levels in grazing lands, and reduce fertiliser application rates in sugarcane, horticulture and grain cropping lands are effective in reducing sediment and nutrient loads. A second question is whether the investments in practice changes are sufficient to reach catchment load targets. Thorburn and Wilkinson (in press) find that the magnitudes of practice change required to achieve the catchment load reductions targeted by Reef Plan across the GBR catchments are larger than are currently planned, and they conclude that it remains unclear whether such changes are biophysically, socially and economically realistic.

In current water quality policy in the GBR region, no clear ecological outcomes have been defined for desired status of, and trends in coastal ecosystems. The Reef Plan (2009) goal refers to health and resilience of the Reef, but does not define specific ecological conditions that would support healthy and resilient ecosystems. Marine water quality guidelines have currently been developed primarily for responses of coral reefs (GBRMPA, 2009), but using water quality guidelines as complete surrogates for ecological outcomes may be inappropriate. The GBR water quality guidelines have been developed to “define trigger values to support target setting for water quality leaving catchments” (GBRMPA, 2009). However, the guidelines have not been quantitatively linked with, or informed the (i) end-of-catchment load reduction targets in Reef Plan, nor (ii) Reef Plan’s long-term goal. Hence, it is not certain whether Reef Plan targets will be sufficient to achieve GBR water quality guidelines, and consequently, GBR ecosystem health and resilience (Brodie et al., 2012; Kroon and Brodie, 2009; Kroon, 2012).

If Reef Plan targets and goals were not met in the identified time frame (mostly 2013; see Section 9.3), the conditions of inshore GBR ecosystems are unlikely to improve in the medium-term future. However, more concerning would be the loss of future political support and funding for sustained long term management of land use in the GBRCA, which is required to achieve the desired change. The definition of specific ecological conditions to support a healthy and resilient ecosystem would enable an informed debate on the management actions and policy instruments required to achieve these ecological conditions. Importantly, the social and economic costs of meeting the targets are not well understood, nor are the trade-offs required to meet long-term Reef Plan goals.

In summary, neither the feasibility of achieving these targets, nor their adequacy for GBR ecosystem health and resilience has been fully assessed (but see Kroon, 2012; Thorburn and Wilkinson, in press). Whilst the targets are ambitious and visionary, it appears unrealistic that they could be met by 2013 or 2020, and Kroon’s (2012) first-order assessment suggests they may not be large enough. In our opinion, reaching the Reef Plan and Reef Rescue pollutant load targets in timeframes such as 2013 looks visionary but unrealistic.

9.6. Voluntary incentive-based versus regulatory-based management

Similarly to Reef Rescue, the Reef Protection Package is an integral component of Reef Plan (2009), and is expected to contribute half to the overall required target reduction in river loads. The relative success of the two approaches – voluntary incentive-based and regulatory-based – will be fascinating to assess in the future. However, it is not clear whether appropriate evaluation frameworks are in place to assess the outcomes of each approach separately. To date, measures of the adoption of improved management practices do not distinguish the primary drivers of change (e.g. The State of Queensland, 2011); however future reporting will attempt to identify Reef-Rescue funded improvements as distinct from the broader outcomes of other Reef Plan initiatives including the Regulations (K. Gale, pers. comm.).

9.7. Incorporating new knowledge into reprioritisation of management actions

As our knowledge advances, management interventions may need to be modified. Currently there is no formal mechanism for this apart from the complete revision of initiatives like Reef Plan (as done in 2008 after the first 5 years). A good example is our improved understanding of the comparative role of hillslope erosion versus gully erosion as major sediment sources in dry tropical catchments such as the Burdekin, Fitzroy, upper Herbert and Normanby, initial assessments suggested the total load was dominated by hillslope erosion (McKergow et al., 2005a), whereas recent work has shown that gully erosion is a more important source in some locations within catchments, for example, parts of the Fitzroy (Hughes et al., 2009, 2010b), although care is required in extrapolating to the whole of the catchment (Chris Carroll, pers. com.). The importance of gully erosion has also been shown at the larger catchment scale, for example in the upper Herbert (Tims et al., 2010), Burdekin (Bartley et al., 2010a,b; Wilkinson et al., in press) and Normanby (Brooks et al., unpublished data). Hence, gully
remediation may need to be given higher priority in programs such as Reef Rescue in certain locations than what currently occurs. Another example is the importance of particle size and the physico-chemical nature of particulate matter (e.g. organic content and clay mineralogy) for transport behaviour and risk to marine ecosystems. Only very fine sediments are transported through the Burdekin Falls Dam (Lewis et al., 2009b, in review) to the river mouth and into the GBR lagoon (Bainbridge et al., 2012) and these fine sediments (<5 μm) are sourced from parts of the Burdekin catchment above the dam. Thus it may be recommended that erosion management concentrate on these areas rather than just, as currently done, on the areas of highest gross erosion.

A more coordinated and formal approach is obviously required to synthesise the growing body of work, so management practices can be evaluated holistically and applied effectively. The commencement of a number of research programs to support Reef Rescue and Reef Plan initiatives within a similar timeframe provides an excellent opportunity for a coordinated approach to reporting research outcomes.

10. Conclusions

Water quality associated with pollutant discharge from the GBRCA is still a major issue for GBR ecosystems. Research published since the Scientific Consensus Statement of 2008 (Brodie et al., 2008a) confirms its conclusions that water discharge to the GBR is of poor quality in many rivers and in the associated floodplumes (Brodie et al., 2010; Devlin and Schafferke, 2009; Devlin et al., 2012; Kroon et al., 2012; Packett et al., 2009); contaminants are present in the GBR lagoon at concentrations likely to cause environmental harm (De’ath and Fabricius, 2010; Devlin et al., 2012; Lewis et al., 2009a; Schafferke et al., 2012; Shaw et al., 2010); evidence of the causal relationship between water quality and GBR ecosystem health is more robust (Brodie et al., 2011; Fabricius et al., 2010; Fabricius, 2011; Jupiter et al., 2008; Lewis et al., 2012b; McKenzie et al., 2010; Negri et al., 2011); and that climate change will confound the attribution of ecosystem degradation to single causes such as poor water quality (Borges and Gypens, 2010; Cooper et al., 2008; De’ath et al., 2009; Hughes et al., 2010a).

We still have a number of important knowledge gaps that hinder our ability to better prioritise management actions and assess whether our management interventions are successful. In particular, we need to better understand pollutant transport processes (e.g. overbank flow, groundwater transport and residence times); processes taking place during transport (e.g. denitrification, flood-plain deposition, plume flocculation, biological uptake, pesticide half-lives); the effects of contaminants in the short-term (e.g. during flood plumes) versus the long-term (e.g. chronic effects over years of low level pollution); the long-term effectiveness of many of our recommended management practices in reducing pollution (e.g. constructed wetlands to trap contaminants from cropping or better pasture cover to reduce erosion in rangelands); the effects of removal of vegetation from catchments and reduced infiltration and increased water runoff (but see Thornton et al., 2007); and the effects of pollutants at ecosystem scale versus single species response.

Our evaluation of the current management response to GBRCA runoff has shown that the conclusion from the Consensus Statement that ‘current management interventions are not effectively solving the problem’ has now decisively changed with the introduction of Reef Rescue and the Reef Protection Package. Currently useful management action is being taken, although whether it is enough to achieve the Reef Plan targets or is the most appropriate form of management is uncertain (Thorburn and Wilkinson, in press; Kroon, 2012). However, it is expected that measurable improvements in river and coastal marine water quality, or ecosystem health, may not be detected for up to several decades (Darnell et al., 2012). Whether these programs will be enough to ‘save the reef’ with respect to water quality impacts, however, is unknown and an analysis on ecologically relevant load reduction targets by Kroon (2012) suggests it is not.

Acknowledgements

The authors of this paper would like to acknowledge support for their research activities over the last decade from a number of Australian and Queensland Government programs, including the National Action Plan for Salinity and Water Quality, the Natural Heritage Trust, the Coastal Catchments Initiative, Cooperative Research Centre programs for Reef, Rainforest and Catchment to Reef, the Marine and Tropical Science Research Facility, the Reef Rescue initiative, the National Environmental Research Program and the Reef Plan Paddock to Reef Integrated Monitoring and Modeling Program, as well as CSIRO’s Water for a Healthy Country Flagship Program. Support from Queensland Government agencies including the Department of the Premier and Cabinet and the Department of Environment and Resource Management is also recognised.

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