

## Climate change mitigation for agriculture: water quality benefits and costs

Robert Wilcock, Sandy Elliott, Neale Hudson, Stephanie Parkyn and John Quinn

### ABSTRACT

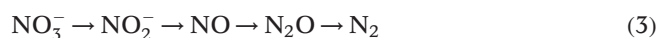
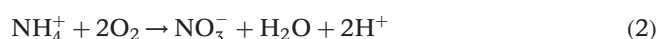
New Zealand is unique in that half of its national greenhouse gas (GHG) inventory derives from agriculture - predominantly as methane ( $\text{CH}_4$ ) and nitrous oxide ( $\text{N}_2\text{O}$ ), in a 2:1 ratio. The remaining GHG emissions predominantly comprise carbon dioxide ( $\text{CO}_2$ ) deriving from energy and industry sources. Proposed strategies to mitigate emissions of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  from pastoral agriculture in New Zealand are: (1) utilising extensive and riparian afforestation of pasture to achieve  $\text{CO}_2$  uptake (carbon sequestration); (2) management of nitrogen through budgeting and/or the use of nitrification inhibitors, and minimizing soil anoxia to reduce  $\text{N}_2\text{O}$  emissions; and (3) utilisation of alternative waste treatment technologies to minimise emissions of  $\text{CH}_4$ . These mitigation measures have associated co-benefits and co-costs (disadvantages) for rivers, streams and lakes because they affect land use, runoff loads, and receiving water and habitat quality. Extensive afforestation results in lower specific yields (exports) of nitrogen (N), phosphorus (P), suspended sediment (SS) and faecal matter and also has benefits for stream habitat quality by improving stream temperature, dissolved oxygen and pH regimes through greater shading, and the supply of woody debris and terrestrial food resources. Riparian afforestation does not achieve the same reductions in exports as extensive afforestation but can achieve reductions in concentrations of N, P, SS and faecal organisms. Extensive afforestation of pasture leads to reduced water yields and stream flows. Both afforestation measures produce intermittent disturbances to waterways during forestry operations (logging and thinning), resulting in sediment release from channel re-stabilisation and localised flooding, including formation of debris dams at culverts. Soil and fertiliser management benefits aquatic ecosystems by reducing N exports but the use of nitrification inhibitors, viz. dicyandiamide (DCD), to achieve this may under some circumstances impair wetland function to intercept and remove nitrate from drainage water, or even add to the overall N loading to waterways. DCD is water soluble and degrades rapidly in warm soil conditions. The recommended application rate of 10 kg DCD/ha corresponds to 6 kg N/ha and may be exceeded in warm climates. Of the  $\text{N}_2\text{O}$  produced by agricultural systems, approximately 30% is emitted from indirect sources, which are waterways draining agriculture. It is important therefore to focus strategies for managing N inputs to agricultural systems generally to reduce inputs to wetlands and streams where these might be reduced to  $\text{N}_2\text{O}$ . Waste management options include utilizing the  $\text{CH}_4$  resource produced in farm waste treatment ponds as a source of energy, with conversion to  $\text{CO}_2$  via combustion achieving a 21-fold reduction in GHG emissions. Both of these have co-benefits for waterways as a result of reduced loadings. A conceptual model derived showing the linkages between key land management practices for greenhouse gas mitigation and key waterway values and ecosystem attributes is derived to aid resource managers making decisions affecting waterways and atmospheric GHG emissions.

**Key words** | agriculture,  $\text{CH}_4$ ,  $\text{CO}_2$ , greenhouse gas,  $\text{N}_2\text{O}$ , water quality

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## INTRODUCTION

Emissions of the greenhouse gases (GHG) carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) increase radiative forcing of the troposphere and affect global climate change (Jain *et al.* 2000; Houghton *et al.* 2001). Atmospheric CO<sub>2</sub> contributes about 60% to the enhanced global greenhouse effect and mainly derives from combustion of fossil fuels. CH<sub>4</sub> contributes about 20% of the greenhouse effect with concentrations increasing at a rate of about 0.4% a year, and N<sub>2</sub>O is responsible for approximately 5% of the global enhanced greenhouse effect with atmospheric concentrations increasing at a rate of 0.2–0.3% a year (Houghton *et al.* 2001). Approximately half of all GHG emissions from New Zealand are from agriculture and predominantly comprise CH<sub>4</sub> and N<sub>2</sub>O in an approximately 2:1 split (CO<sub>2</sub> equivalents). The dominant source of CH<sub>4</sub> is enteric fermentation from ruminant livestock (cattle and sheep) (Equation 1), whereas N<sub>2</sub>O emissions originate from nitrogen (N) applied to soil in fertilizer and from animal dung and urine, via coupled nitrification-denitrification (Equations 2 and 3).



Non-agricultural emissions from New Zealand are predominantly CO<sub>2</sub> from the energy and industrial sectors of the economy.

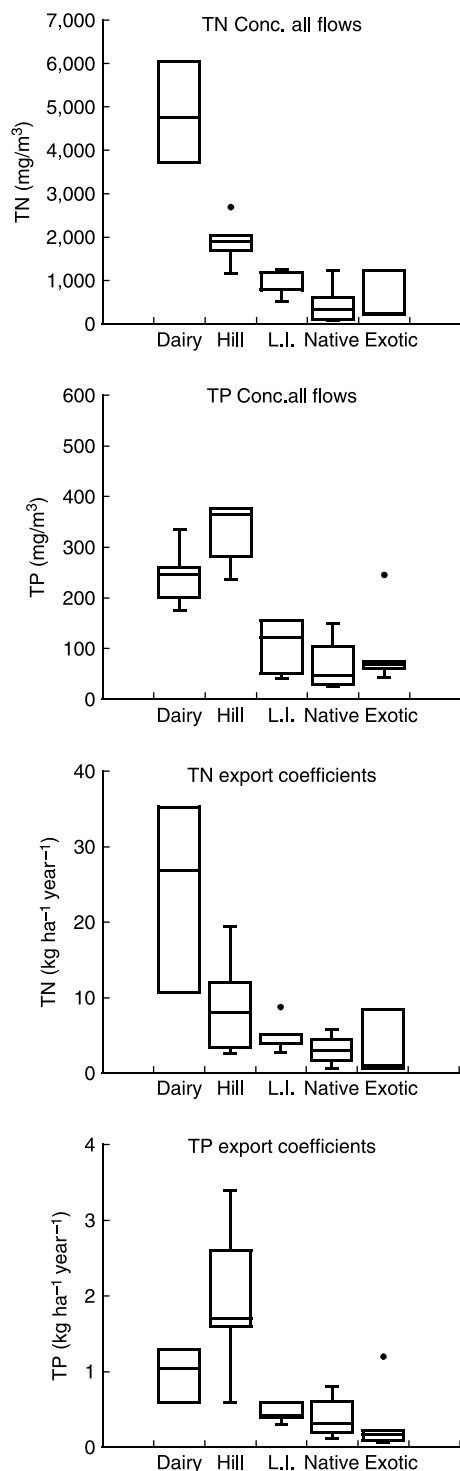
The New Zealand Government has identified key ways to offset agricultural GHG emissions via sequestration of CO<sub>2</sub> by either extensive or riparian afforestation of pasture; nitrogen management through nutrient budgeting and use of nitrification inhibitors, and minimising soil anoxia for mitigating N<sub>2</sub>O emissions (Di & Cameron 2002); and utilizing alternative waste treatment techniques to minimise CH<sub>4</sub> emissions.

Each of these actions also has co-benefits and co-costs (disadvantages) for the aquatic environment because they impinge on important land-water interactions and riparian functions for waterways. In this paper we review the consequences for the aquatic environment of pastoral GHG mitigation options in a temperate climate and derive a conceptual model showing the linkages between land management practices and key waterway values.

## MITIGATION MEASURES

### Extensive afforestation

Establishing new areas of forest, or converting marginal cropping or grazing land to forest is widely recognised as a way of sequestering CO<sub>2</sub> with co-benefits and co-costs for aquatic ecosystems (Farley *et al.* 2005). In New Zealand, radiata pine (*Pinus radiata*) has been the dominant species used in plantation forest and in agricultural afforestation (Fahey *et al.* 2004). Conversion of pasture into pine plantation improves stream water quality and biodiversity by reducing inputs of sediment, nutrients, pathogens and agrichemicals. Stream habitat conditions are similar to those in native forest catchments (Fahey *et al.* 2004; Parkyn *et al.* 2006; Quinn *et al.* 2007) with generally lower concentrations of total N (TN) and total P (TP) in afforested streams than in pasture streams (Figure 1) (Elliott & Sorrell 2002). Export coefficients for TN and TP are also much lower for native and exotic forest than for different kinds of pastoral agriculture (Figure 1). It should be noted, however, that these reductions in concentration and export coefficients occur under long-term steady-state conditions. When pasture is newly afforested there is a large pool of sequestered N that may be released over time, so that the benefits of afforestation in the short term may not be as great as indicated in the box plots (Parfitt *et al.* 2003). Box plots in Figure 1 enclose 50% of the data, with the median values displayed as solid lines. The top and bottom of each box mark the limits of ±25% of the variable population. The lines extending from the top and bottom of each box mark the maximum and minimum values within the data that fall within an acceptable range. Outliers are displayed as individual points.



**Figure 1** | Summary of nutrient concentrations and exports from dairy, hill-country and low intensity (L.I.) pastoral farming, and from plantation (exotic) and native forest in New Zealand (Elliott & Sorrell 2002).

Pine afforestation of highly erodible hill-land, in combination with changes in stock type, and riparian fencing and planting (poplars or native vegetation) was associated with reductions in the export of suspended solids (SS) and a decline by 5°C in downstream stream temperature four years after the changes were made (Quinn *et al.* 2007).

Forest harvesting generally increases sediment yield for a short time, but averaged over the growth cycle (25–30 yrs) the losses from forestry are less than for pasture (Fahey *et al.* 2004). The harvesting phase in production cycles may adversely impact on water and habitat quality.

Other benefits from afforestation include addition of woody debris and leaf litter for food webs and cover habitat for aquatic fauna. Streams are cooler and are less affected by photosynthetic production so that diel excursions of pH and dissolved oxygen are less marked where there is shading by trees (Davies-Colley & Quinn 1998). Contamination of water with faecal microbes is lower in forest than in pasture streams, although the presence of animals results in non-zero background levels in forest. Median faecal coliform and enterococci concentrations in a pine catchment stream were 55% and 17%, respectively, of corresponding values in an adjacent pasture stream (Quinn & Kemp 2002).

Conversion of pasture to forest typically causes a reduction in water yield (200–300 mm/yr) (Fahey *et al.* 2004; Farley *et al.* 2005), with reduced stream flows and groundwater recharge, but this effect diminishes with decreasing annual rainfall. Widespread afforestation can also decrease flood peak flows, as observed from paired small catchments in New Zealand (Duncan 1995).

### Riparian afforestation

Riparian afforestation may reduce nutrient inputs through interception of N and P along hydrological flow paths. Reductions in concentrations of groundwater nitrate of 76–98% have been recorded with the greatest removal (>95%) being by older trees (Ryszkowski & Kedziora 2007). In addition, the ratio of  $\text{NH}_4^+-\text{N}$  to  $\text{NO}_3^--\text{N}$  increases within the soil profile of forested areas. In some instances (notably, where there are wetlands) the end product of riparian denitrification is a high  $\text{N}_2\text{O}:\text{N}_2$  ratio,

but suitable management may be able to reduce net  $\text{N}_2\text{O}$  formation while still achieving substantial denitrification (Groffman *et al.* 2000; Wilcock & Sorrell 2008).  $\text{N}_2\text{O}$  production was 40–60% lower from recent (7–8 year old plantings) than from old (160+ year old) forested areas (Ryszkowski & Kedziora 2007). A number of issues appear repeatedly in the scientific literature (Hefting *et al.* 2006; Quinn *et al.* 2007), raising the following questions:

“Does utilisation of greenhouse gas mitigation measures such as establishment of riparian forests or extensive plantations in upland catchments represent a long-term solution to the issue of nitrogen management?” Nitrogen saturation was identified as a potential issue, causing leakage of untreated nitrate through riparian treatment systems (Hefting *et al.* 2006). Most publications highlight the requirement for more accurate nitrogen budgeting at both farm and catchment scale, with a view to eliminating excess nitrogen inputs.

“Does removal of nitrate from groundwater by denitrification in riparian forests merely transfer the problem from one environmental compartment to another?” Despite considerable research, large uncertainty exists regarding the fate of the nitrate removed from the groundwater. A number of publications indicate that much of this nitrogen may be emitted as  $\text{N}_2\text{O}$ , particularly when the nitrogen removal capacity of the riparian buffer is approached (Hefting *et al.* 2003; Quinn *et al.* 2007; Mayer *et al.* 2007). Elimination of excess nitrogen application is likely to be a vital prerequisite to achieving both an improvement in water quality and a reduction in greenhouse gas emission goals.

Storage and retention of P in riparian zones is controlled by adsorption to soil, uptake and removal of soluble P by plants, immobilisation within microbial biomass and incorporation of organic P into peat (Mayer *et al.* 2007). The bulk of P removal occurs at the upslope edge of the riparian buffer zone as a physical process, largely influenced by the reduced velocity of overland flow and increased surface roughness within the buffer. Efficiency of removal and retention is greatest for short-duration rainfall events (Mander *et al.* 2005). Preferential flow of polluted water into streams via drains has been identified as a cause of reduced efficiency of riparian plantings in the management of nitrogen inflows to streams. In one instance (Monaghan *et al.* 2007), up to 33% of the phosphorus

emitted from a dairy farm was in sub-surface drainage water that bypasses riparian processing. While riparian plantings have been identified as a tool to mitigate inputs of phosphorus associated with particulate material (Gentry *et al.* 2007), no equivalent tool existed for managing inputs of soluble phosphorus from shallow groundwater. Other co-benefits of riparian afforestation for mitigation of P inputs are as for extensive afforestation. Disadvantages include intermittent water quality degradation associated with logging and forest maintenance, and some sediment release due to channel re-stabilisation and localised flooding, including formation of debris dams at culverts (Collier *et al.* 2001).

### Fertiliser management

$\text{N}_2\text{O}$  emissions from pasture may be reduced by reducing the amount of N fertiliser applied to land, avoidance of anoxic conditions leading to formation of  $\text{N}_2\text{O}$  via denitrification, and the use of inhibitors that block nitrification and hence, denitrification.

Commonly used N fertilisers, such as urea and  $\text{NH}_4\text{SO}_4$ , provide a source of plant-available N that stimulates pasture growth and promotes more intensive grazing. The N returned to pasture in dung and urine is often too concentrated for plant uptake and is eventually oxidized to  $\text{NO}_3^-$ , which may subsequently undergo reduction in anaerobic soils zones producing  $\text{N}_2\text{O}$  as an intermediate product of denitrification (Di & Cameron 2002; de Klein & Ledgard 2005). Thus, reducing N loadings (from fertiliser and animal excreta) and minimizing conditions producing soil anoxia (heavy grazing leading to reduced soil infiltration rates, and excessive irrigation rates) are effective ways of limiting  $\text{N}_2\text{O}$  emissions, and of achieving reductions in  $\text{NO}_3^-$  inputs to waterways (Monaghan *et al.* 2007).

Dicyandiamide (DCD) is a commonly used nitrification inhibitor that is water-soluble (23 g/L at 13°C) (de Klein & Ledgard 2005; Suter *et al.* 2006). DCD has a soil half-life of 111–116 d at 8°C, and 18–25 d at 20°C, so that it is best suited to cool climates and is increasingly less effective in warmer and wetter climates (Di & Cameron 2002; Suter *et al.* 2006). In New Zealand it has been estimated that if a nitrification inhibitor was applied after all grazings throughout the year on all dairy farms, it could potentially reduce

N<sub>2</sub>O emissions by 20%, but this estimate is so far untested (de Klein & Ledgard 2005). Furthermore, DCD may pose additional problems in warm-wet climates, either by blocking nitrification-denitrification coupling in wetlands and thereby inhibiting their capacity for N removal, or by exerting an additional N load on waterways. Rates of 10 kg DCD/ha are recommended in New Zealand where it is normally applied twice a year in cooler regions. This corresponds to an additional land loading of 6–13 kg N/ha/yr and could present a supplementary risk of N loss to waterways if some of the DCD was lost in runoff. Countering this, DCD decreases nitrate percolation (Di & Cameron 2002) (for a given stocking rate).

The major environmental benefits from these measures are lower N loads to waterways, with reduced eutrophication, and lower GHG emissions. Recent research (Wilcock & Sorrell 2008) showed N<sub>2</sub>O fluxes up to 100 µmol/m<sup>2</sup>/hr<sup>1</sup> occurred in pastoral streams that had NO<sub>3</sub><sup>−</sup>–N concentrations of 5–18 mg/L.

### Waste management

The use of natural and constructed wetlands for reducing NO<sub>3</sub><sup>−</sup>–N inputs to waterways via denitrification may produce excessive emissions of N<sub>2</sub>O, depending on ambient redox conditions (Ryszkowski & Kedziora 2007; Wilcock & Sorrell 2008). Just how this affects regional N<sub>2</sub>O budgets depends on whether wetlands have significantly different N<sub>2</sub>O:N<sub>2</sub> ratios from other key sources and is not yet well understood (Groffman *et al.* 2000). Wastewater irrigation generates N<sub>2</sub>O when soils are saturated or overloaded with N and is exacerbated if wastewater enters drainage systems and enters waterways by-passing riparian zones (de Klein & Ledgard 2005; Monaghan *et al.* 2007). This can be mitigated by maintaining wastewater application rates at or below soil infiltration rates and avoiding irrigation where rapid transport via drains is likely (Monaghan *et al.* 2007).

Anaerobic ponds used for treating farm wastes may be capped to retain and utilize CH<sub>4</sub> for fuel (with a 21-fold reduction in GHG load following combustion to CO<sub>2</sub>). When coupled with other facultative treatment ponds they can be managed to reduce outputs of N, P and micro-organisms to the aquatic environment (Craggs *et al.* 2008).

## DISCUSSION

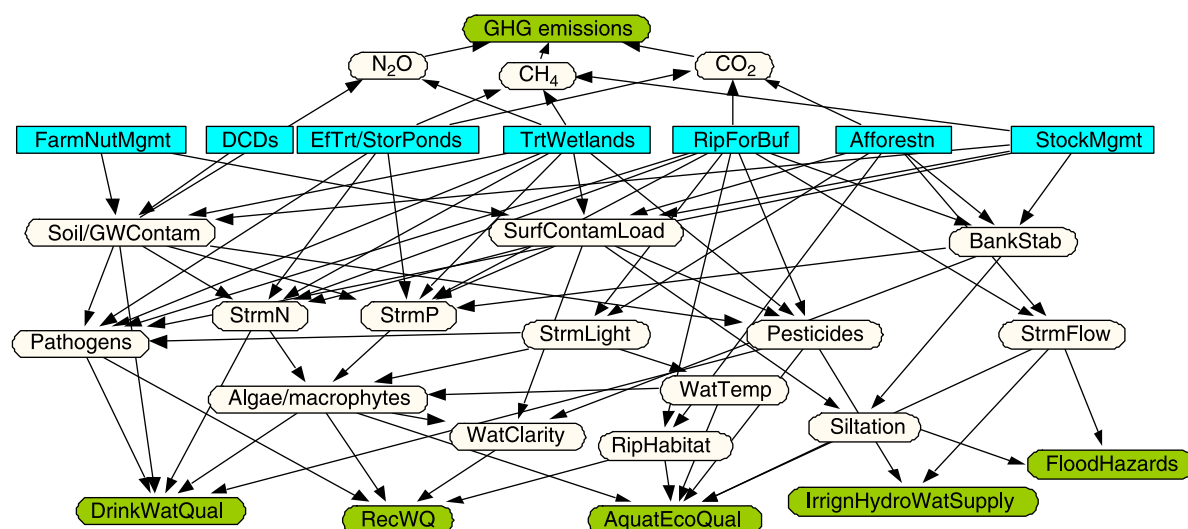
The various strategies for mitigating GHG emissions from pastoral agriculture are for the most part beneficial for waterways. Water quality would be improved by having lower concentrations of N, P, sediment and faecal organisms. Concentrations of pesticide residues (*viz.* herbicides and antihelminthic veterinary remedies like levamisole) would be much lower following conversion of pasture to forest, but not from the fertiliser and waste treatment options (Wilcock 2008). Stream habitat would benefit from increased shading and woody debris (from extensive and riparian afforestation) but may be offset by sediment inputs from logging and bank re-stabilisation (Parkyn *et al.* 2006), and by hydrological changes associated with reduced water yields (Duncan 1995; Farley *et al.* 2005).

Assessing these changes may not be easily achieved experimentally and modelling may be the best option to undertake cost-benefit analyses for GHG mitigation measures involving land use changes. A range of models presently exists or are being developed that can be used to assess land use change effects on water quality for a range of spatial and temporal scales. They generally apply to large scale changes, such as extensive afforestation of pasture and widespread adoption of nitrification inhibitors (*e.g.* The DeNitrification-Decomposition model, DNDC) (Beheydt *et al.* 2007). The OVERSEER nutrient budget model can be used for the assessment of mean annual nitrogen and phosphorus losses from pastoral, cropping and horticultural systems at the farm scale, takes climate and soils into account, and incorporates a component for estimating emissions of N<sub>2</sub>O (Wheeler *et al.* 2008).

### Conceptual model

This review has highlighted the complex interactions that exist between mitigations to control GHG emissions, and aquatic values. These are summarised in a broad conceptual model of interactions (Figure 2). Modelling of co-benefits and potential confounding effects could be carried out within this context. The model is based on the Bayesian Belief Network concept for decision making (Oliver & Smith 1990) and highlights the key linkages and





**Figure 2** | Conceptual model of links between agricultural land management practices for greenhouse gas (GHG) control and key waterway values darker grey (green) via mitigation and/or management options (mid-grey (blue) rectangles) and ecosystem attributes. The top level lists the key GHG. The next level (mid-grey (blue) boxes) contains key land management techniques that mitigate GHG emissions and aquatic impacts, viz. farm nutrient management, nitrification inhibitors (DCDs), effluent treatment ponds and storage ponds for deferred effluent irrigation, effluent treatment by wetlands, riparian forest buffer zones, extensive afforestation, and stock management. The upper level of pale boxes are drivers of key water quality properties that are affected by the management actions in the boxes above them (viz. soil and ground water contaminant loads, surface water contaminant loads, destabilised stream bank sediment). The lower level of pale boxes identifies aspects of the aquatic environment affected by these drivers that more directly influence aquatic values. These include pesticides, pathogens, stream nitrogen and phosphorus, stream light (for plant growth and temperature control), stream flow, algae and macrophytes (larger plants), water clarity, riparian habitat and stream siltation. Stream restoration and water quality standards, criteria or guidelines typically target these aspects. The bottom level of darker grey (green) boxes are key aquatic ecosystem values, functions and uses, viz. drinking water, recreational, aquatic ecology, irrigation and hydroelectric power generation, and flood control. Subscribers to the online version of *Water Science and Technology* can access the colour version of this figure from <http://www.iwaponline.com/wst>.

pathways between GHG mitigation measures, farm management actions, and waterway values.

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