Climate change mitigation measures: Water quality benefits and costs

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Prepared for

Ministry for the Environment

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The document may be cited as:

Wilcock B, Elliott S, Hudson N, Parkyn S, Quinn J. 2008. *Climate change mitigation measures: Water quality benefits and costs.* Report prepared by NIWA for MfE. Wellington: Ministry for the Environment. 71 p.

Published in August 2008 by the
Ministry for the Environment
Manatū Mō Te Taiao
PO Box 10 362, Wellington, New Zealand

ISBN: 978-0-478-30230-1 (electronic)

This document is available on the Ministry for the Environment's website:
www.mfe.govt.nz

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Executive Summary

Methods to mitigate greenhouse gas emissions, notably of methane (CH_4) and nitrous oxide (N_2O) from pastoral agriculture in New Zealand are focused on:

- carbon sequestration through extensive and riparian afforestation of pasture
- management of nitrogen through sound budgeting and/or the use of nitrification inhibitors
- managing soil conditions to avoid anoxic conditions leading to the formation of nitrous oxide
- utilisation of alternative waste treatment technologies to minimise emissions of methane.

These mitigation measures have associated co-benefits and co-costs (disadvantages) for the aquatic environment.

Extensive afforestation co-benefits:

- Reduced inputs of nutrients and contaminants and stabilised stream banks, directly and indirectly by exclusion of livestock.

- Provision of inputs of woody material and leaf litter for food webs, and cover habitat for aquatic fauna.

- Shading of water, reducing summer temperature extremes and primary production from photosynthesis, resulting in less aquatic weed and periphyton, and less variable acidity (pH) and dissolved oxygen (DO) regimes.

- Less extreme rainfall-runoff events. Afforestation of pasture can produce substantial reductions in annual peak stream-flows due to greater interception losses and lower soil moisture under forest than pasture. This has benefits of reducing stream bank erosion and other flooding impacts on habitat that result from small to moderate storm events.

- Improved terrestrial and aquatic biodiversity.

- Improved visual amenity (during tree-growth periods, but not during or after harvesting) and opportunities for recreation (tramping etc).

Extensive afforestation co-costs:

- Conversion of pasture to pine forest typically reduces water yield by about 200-300 mm/yr (6-10 L/s/km^2). The impacts this has on stream flow, the duration of flow in small streams, and groundwater recharge depends on the total rainfall and the hydrogeology of the catchment.

- Pine afforestation may cause intermittent water quality issues related to the long-term production cycle (low/no cover, thinning/maintenance, harvest/clear-felling).

Riparian forestry co-benefits:

- Reduced inputs of nutrients and contaminants via infiltration and uptake, and denitrification as water passes through riparian zones. Stabilised stream banks directly and indirectly by exclusion of livestock.

- Inputs of woody material and leaf litter for food webs and cover habitat for aquatic fauna.

- Shading of water, reduces summer temperature extremes and primary production from photosynthesis, resulting in less aquatic weed and periphyton, and less variable pH and DO regimes.

- Enhanced habitat for fish communities.

- Improved terrestrial and aquatic biodiversity.

- Increased opportunity for walkways along the sides and banks of waterways.

- Improves visual amenity (except if/when harvesting occurs).

- Lower water yield than is generated from pasture, but maybe not as much of a reduction as would come from extensive afforestation of the catchment.

- If grass strips are placed upslope from the woody riparian areas they can achieve some reduction in phosphorus and faecal runoff through settling and filtration processes that occur. Thus, a combination of riparian tree planting for carbon sequestration and grass strips can achieve multiple benefits for stream water quality.

- Reduced impact of downstream flooding by slowing flood flows when water travels over rough vegetation on banks.

- Provision of shade and shelter for stock (might be increasingly necessary due to climate change impacts).

Riparian forestry co-costs:

- Possibly intermittent water quality issues related to the long-term production cycle if production planting is alongside streams, including loss of ground cover as a result of thinning and other maintenance operations, and harvesting – especially, clear-felling.

- Streams with dense stands of aquatic plants are able to take up nutrients from the water column and thereby attenuate loadings to downstream reaches. Heavy shading (as may develop in small, riparian forested, streams) will inhibit such growth and hence reduce attenuation so that more of the nutrient inputs into headwater streams will be transported downstream. Inputs of nutrients to the stream will however be reduced by the riparian margin itself.

- Pulses of sediments may be released from stream banks once shading (from trees) is established along small streams. Stream banks will return to their widths when previously forested. Stream banks will ultimately be more stable, but there may be significant sediment losses for a period when the stream channel is responding to the changed riparian conditions.

- Riparian forest may also increase risks associated with localised flooding, including formation of debris dams at culverts.

- Provision of habitat for pest species (e.g. possums).

Improved nitrogen fertiliser practices – co-benefits:

- Reduced inputs of nitrogen to waterways and lower concentrations of nitrate. This will be especially beneficial to waterways that are sensitive to inputs of nitrogen.

- Improved water quality through reductions in plant and algal growth in waterways where nitrogen is the limiting nutrient. Photosynthetic production by plants can cause extreme daily variations in dissolved oxygen and acidity during summer and have a significantly adverse impact on stream habitat quality.

- Better public health outcomes where nitrate levels may approach potable water limits.

Improved nitrogen fertiliser practices – co-costs:

- The use of nitrification inhibitors may prevent wetlands from denitrifying nitrate (from pasture runoff) reducing their capacity to intercept and remove nitrogen from runoff entering waterways.

- Improved pasture response from nitrification inhibitors may lead to more intensive grazing with increased methane emissions as a result. This may also result in increased faecal pollution of waterways.

Soil management – co-benefits:

- The production of nitrous oxide by denitrification is promoted by anoxic soil conditions. This can be avoided by sound grazing practices that avoid pugging of soils, and by draining wet soils and avoiding over-watering (irrigation).

- Wetlands can be utilised to intercept and attenuate pollutants in pastoral runoff.

Soil management – co-costs:

- The use of drainage networks to promote drainage in wet soils enhances direct entry of pasture runoff to surface waters and short-circuits natural interception and attenuation processes that occur in riparian and other wetlands areas.

- Under certain conditions, wetlands are potent sources of methane and nitrous oxide.

Alternative waste treatment – co-benefits:

- Managed irrigation of wastewater can maximise treatment efficiency for land disposal schemes and at the same time reduce drainage losses to waterways.

Alternative waste treatment – co-costs:

- Land application of wastewater may cause emissions of greenhouse gases where loading rates are too high and soils become saturated.

- Anaerobic wastewater treatment ponds and storage ponds that enable effluent irrigation to be deferred in times of wet weather generate considerable methane emissions (although these can be captured and used as an energy resource).

Estimating water quality co-benefits and co-costs in physical terms

A number of modelling approaches are available or are being developed to enable co-benefits and co-costs to be estimated by means of:

- Capturing the effects of environmental factors such as climate, soils, and slopes on water quality and greenhouse gas emissions.

- Consideration of land-use or land-management factors on water quality.

- Allowing the cumulative effects of mitigation measures at the catchment or national scale, including the case where measures are implemented in a partial or patchy fashion.

- Improving communication of results by predicting effects in hypothetical conditions.

- Improving understanding of processes and longer-term responses occurring on a time-scale longer than those of field trials (for example, predicting the effects of using dicyandiamide, DCD) on long-term nitrogen budgets, or predicting the changes in soil nitrogen stores following afforestation).

The OVERSEER Nutrient Budgets Model (www.agresearch.co.nz/overseerweb) can be used for assessment of mean annual nitrogen and phosphorus losses from pastoral and cropping and horticultural systems, and takes climate and soils into account. The model currently incorporates a component for estimating emissions of nitrous oxide, and this component of the model has been used to assess the effects of farm management on nitrous oxide emissions.

Other useful models include the:

- DeNitrification-DeComposition model (DNDC) (www.dndc.sr.unh.edu)

- EcoMod model being used by AgResearch as a research-level simulation model to investigate nutrient cycling and losses on farms

- SPASMO (HortResearch) nitrogen loss model developed primarily for horticultural systems, although it has been applied to pastoral systems.

- Whole Farm Model (DairyNZ Ltd) for identifying more efficient dairy production systems as well as new dairy systems that meet a wide range of financial, environmental and lifestyle goals.

- LUCI (Crop and Food Research) paddock-scale model for assessing nitrogen leaching from flat fields with potato, maize and wheat, and a ryegrass/pasture component.

- NPLAS (Environment Bau of Plenty): a simple web-based decision support tool for assessment of nutrient losses in the Rotorua Lakes area.

New approaches to estimating co-benefits and co-costs in physical terms

There is a need for models that take into account receiving water sensitivity. This might be achieved through a simple spatial overlay of a given catchment, of sensitive waterbodies with maps of the locations of the most effective mitigation options, to identify the locations where mitigation options will have the greatest impact on water quality. More complex integrated approaches are generally at fairly early stages of development or application in New Zealand: therefore, in the short term, simple methods such as overlays with catchments of sensitive water bodies would be most appropriate.

Another approach would be to extend the scope of the present contract to review land use change and its effects on water quality (good and bad) as it applies to greenhouse mitigation measures proposed for New Zealand agriculture. A critical review of the literature would include all kinds of riparian vegetation experiments and land use conversions and would examine closely a wide range of plant species with a view to optimising the two aspects of greenhouse gas abatement and improved water quality.

A general conceptual model of key interactions between effects on greenhouse gases (GHG) and waterway values of current GHG/land/water management practices is provided, based on the material in this review as a framework for future predictive model development.

1. Introduction & background

The New Zealand government has identified a number of strategies whereby the effects of climate change may be mitigated. The document *New Zealand's climate change solutions: Sustainable land management and climate change, Plan of action* (2007) identifies three policy pillars whereby sustainable land management could address the challenges posed by climate change. The second pillar (reducing emissions and enhancing sinks) has particular relevance for New Zealand agriculture. As the largest contributor to greenhouse gases (GHG) emissions, agriculture will be under greatest pressure to reduce emissions and/or mitigate the impact of emissions.

A number of mechanisms exist to reduce greenhouse gases emissions, or to mitigate their effects. Some of them offer benefits in addition to meeting climate change objectives. Identifying these additional environmental benefits and the circumstances under which they will be achieved, as well as describing the benefits fully is important in establishing an environment in which these measures are likely to be adopted and implemented. Such co-benefits may make adoption of these measures more palatable to producers and the general community, making successful implementation of greenhouse gases mitigation measures more likely.

This report identifies the major co-benefits that climate change mitigation measures will offer to water quality and aquatic habitats. The report explicitly addresses environmental costs and benefits and does not address monetary aspects. Additional benefits are also identified. Some of these arise as a consequence of likely improvement in water quality. Other co-benefits are related to amenity and cultural values. It is important to consider these as well because they collectively indicate that environmental benefits may be achieved at catchment scales.

Three specific objectives were identified for this report:

1. Define the circumstances under which significant water quality benefits or costs are arrived at through climate change mitigation measures that:

    - Sequester carbon dioxide (CO_2); or

    - Reduce emissions of fossil-sourced CO_2; or

    - Reduce emissions of methane (CH_4) or nitrous oxide (N_2O).

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2. Identify work done that credibly quantifies in physical terms the magnitude of benefits (beneficial impacts) and costs (adverse impacts) for water quality associated with particular measures taken to increase mitigation of greenhouse gases.

3. Where a sound basis exists, provide estimates of the water quality benefits or costs, or where such information does not exist, identify how such benefits or costs may be quantified.

These objectives clearly focus this assessment on mitigation measures relevant to the agricultural sector. The *New Zealand greenhouse gas inventory, 1990–2005* (MfE 2007) indicated that agricultural sources are responsible for almost half of the greenhouse gas emitted in New Zealand. Two-thirds of the total greenhouse gas emitted from agriculture in New Zealand is associated with enteric fermentation. There are limited options for mitigation of these emissions. However, about 34 per cent of the total greenhouse gas emitted by agriculture is associated with soil processes (primarily as nitrous oxide) and the sector is well positioned to contribute to mitigations in this area. Furthermore, because nitrogen in soils contributes very significantly to the nitrogen load present in surface and groundwater, direct mitigation of nitrous oxide emissions from excess nitrogen in soils provides opportunities to improve surface and groundwater quality. Indirect mitigation of climate change through increased sequestration of atmospheric carbon dioxide (CO_2), by establishing permanent or long-rotation forest, also creates the potential for water quality improvement.

Although climate change mitigation measures may create the potential for other benefits, it is important to recognise that a range of factors or conditions may have to co-occur to realise these benefits. Identifying potential benefits, as well as the conditions under which they may provide water quality improvements, may prove critical in having these measures viewed favourably by the community, and more importantly, by those having to implement them.

2. Measures to mitigate climate change

Recent and future climate change has been related to changes in the atmospheric concentrations of a few critical gases, most notably CO_2, CH_4 and N_2O. The increases in atmospheric concentrations of these gases over the preceding decades have been linked to human activities, specifically combustion of fossil fuels (CO_2) and increased application of nitrogen-containing fertilisers to soils (N_2O). Increased methane emissions have been associated with human activities, including modifications to wetlands and emissions from increasing numbers of ruminants and anaerobic decomposition of their wastes.

Mitigation of the climate change impacts due to increasing concentrations of greenhouse gases in the atmosphere will centre on two strategies. Amounts of greenhouse gas emitted to the atmosphere will be reduced, while greenhouse gas currently in the atmosphere will be removed. The ideal outcome of these two strategies would be a significant decrease in concentrations of greenhouse gases in the atmosphere. Implementing these strategies will impose direct costs on individuals, resource users and society. It will be increasingly important to identify other benefits that may be derived from measures implemented to mitigate climate change. It is essential that the individuals and resource users who ultimately will have to meet the costs of these mitigation measures have a clear understanding of the environmental and monetary costs and benefits that are associated with them.

2.1 Sources of greenhouse gases in New Zealand

Implementing these strategies provides specific challenges to New Zealand agriculture as a consequence of the sources of greenhouse gases:

1. Approximately 50 per cent of greenhouse gas emissions New Zealand arise from agriculture, primarily from the pastoral sector (MfE 2007).

2. The two dominant gases are:

   a. Methane (CH_4), predominantly from enteric fermentation by ruminant livestock; and

   b. Nitrous oxide (N_2O), derived from nitrogen applied to soil in fertiliser, contained in animal dung and urine, and fixed by plant uptake of atmospheric nitrogen.

   The relative importance of these two gases is 2 : 1 for $CH_4 : N_2O$.

3. Of the nitrous oxide produced by agricultural systems, approximately 30 per cent is emitted from indirect sources, which are waterways draining agriculture. It will be important therefore to focus strategies for managing nitrogen inputs to agricultural systems generally to reduce inputs to wetlands and streams where these might be reduced to N_2O.

2.2 Potential strategies to reduce greenhouse gas emissions

Many strategies for mitigating climate change also have potential to significantly impact on water quality, including the following:

- Carbon sequestration. This is the process whereby carbon present in the atmosphere as carbon dioxide is 'locked-up' or transformed to other materials through the process of photosynthesis. Examples of sequestration include creating substantial areas of permanent forest, or increasing the mass of carbon stored in soils.

- Managing application of nitrogen to soils to meet agronomic requirements (i.e. avoid applying amounts of nitrogen in excess of plant requirements). This may require improved nutrient budgeting and careful attention to fertiliser practice, including nutrients of non-commercial origin.

- Managing pasture soil conditions (in association with nutrient application) to avoid the anoxic conditions that favour production of nitrous oxide by soil micro-organisms.

- Utilising alternate waste treatment techniques to minimise uncontrolled release of methane. Waste management may have to include advanced treatment and capture techniques, followed by use of the recovered methane. The waste remaining after advanced treatment may pose less risk to water resources following disposal to land, thereby offering indirect water quality benefits.

These mitigation measures and the likely co-benefits are discussed in the material that follows, with reference to recent literature. Considerable overlap exists between the various mitigation methods and their likely outcomes for climate change mitigation, and water quality.

2.2.1 Recent information from the literature

Smith et al. (2007; 2008) identified three basic mechanisms whereby emissions of greenhouse gas from agriculture may be mitigated:

1. Reduction in emission of greenhouse gas directly, through improved management of flows of carbon and nitrogen within agricultural landscapes. In practice, this would require actions such as more efficient delivery of nitrogen to crops (to reduce emission of nitrous oxide) and utilising improved feed to reduce methane emissions from livestock.

2. Enhancing removal of greenhouse gas directly from the atmosphere, utilising photosynthetic processes to sequester or 'lock up' carbon dioxide derived from the atmosphere. These sinks may include timber, below-ground root biomass and soil organic carbon.

3. Utilising biofuels to replace or displace other fuel types. This may include use of crop residues in place of fossil fuels with or without conversion to biofuels or diesel.

Table 1 in Smith et al. (2008) associates the three basic mitigation strategies with the various agricultural ecosystems, together with an assessment of the likelihood that these measures would achieve the desired objective. This table is reproduced here as Table 1.

In an associated paper, Smith et al. (2007) related potential climate change mitigation measures to environmental impacts (benefits and drawbacks), including impacts on water quality and water quantity. Their summary is included in this report as Table 2. It is important to note that many of the impacts of the mitigation measures may be either positive ('co-benefits') or negative ('co-costs'). There is likely to be a significant geographic variability in the impact of both climate change mitigation measures and impacts on water quality and quantity. The impacts of many of the mitigation measures may also be inter-related. In addition, both the efficacy of the mitigation measures and the impacts on the environment will alter as the climate changes. This makes prediction of the overall mitigation and impact on environmental variables such as water quality and quantity difficult. Cowie et al. (2007) highlight the potential for synergy between greenhouse gas mitigation measures and other environmental benefits, but confirm that there may be tradeoffs. They identify the critical role that policies would have in promoting and achieving optimal environmental outcomes.

Table 1: Greenhouse gas mitigation measures for agricultural ecosystems. Reprinted from *The Philosophical Transactions of the Royal Society B, Biological Sciences 363 (1492)*: Smith et al. "Greenhouse gas mitigation in agriculture". Copyright (2008), with permission from The Royal Society.

Table 1. A list of proposed measures for mitigating GHG emissions from agricultural ecosystems, their apparent effects on reducing emissions of individual gases (mitigative effect) and an estimate of scientific confidence that the proposed practice can reduce overall net emissions.

		mitigative effects[a]			net mitigation[b] (confidence)	
measure	examples	CO_2	CH_4	N_2O	agreement	evidence
cropland management	agronomy	+		±	***	**
	nutrient management	+		+	***	**
	tillage/residue management	+		±	**	**
	water management (irrigation, drainage)	±		+	*	*
	rice management		+	±	**	**
	agroforestry	+		±	***	*
	set-aside, land-use change (LUC)	+	+	+	***	***
grazing land management/ pasture improvement	grazing intensity	±		±	*	*
	increased productivity (e.g. fertilization)	+		±	**	*
	nutrient management	+		±	**	**
	fire management	+		±	*	*
	species introduction (including legumes)	+		±	*	**
management of organic soils	avoid drainage of wetlands	+	−	±	**	**
restoration of degraded lands	erosion control, organic amendments, nutrient amendments	+		±	***	**
livestock management	improved feeding practices		+		***	***
	specific agents and dietary additives		+		**	***
	longer term structural and management changes and animal breeding		+		**	*
manure/biosolid management	improved storage and handling		+	±	***	**
	anaerobic digestion		+	±	***	*
	more efficient use as nutrient source	+		+	***	**
bioenergy	energy crops, solid, liquid, biogas, residues	+		±	***	**

[a] '+' denotes reduced emissions or enhanced removal (positive mitigative effect); '−' denotes increased emissions or suppressed removal (negative mitigative effect); '±' denotes uncertain or variable response.
[b] A qualitative estimate of the confidence in describing the proposed practice as a measure for reducing *net* emissions of GHGs, expressed as CO_2 equivalence. 'Agreement' refers to the relative degree of agreement or consensus in the literature (the more asterisks, the higher the agreement); 'Evidence' refers to the relative amount of data in support of the proposed effect (the more asterisks, the greater the amount of evidence).

Table 2: **Relationship between potential climate change mitigation measures and the environment.**

Reprinted from *Agriculture, Ecosystems and Environment 118 (1–4)*: Smith et al. "Policy and technological constraints to implementation of greenhouse gas mitigation options in agriculture", pp. 6–28. Copyright (2007), with permission from Elsevier.

Table 1
Summary of possible co-benefits and trade-offs of mitigation options in agriculture

Measure	Examples	Food security (productivity)	Water quality	Water conservation	Soil quality	Air quality	Bio-diversity, wildlife habitat	Energy conservation	Conservation of other biomes	Aesthetic/amenity value
Cropland management	Agronomy	+	+/–	+/–	+	+/–	+/–	–	+	+/–
	Nutrient management	–/+	+		+	+		+		
	Tillage/residue management	+	+/–	+	+		+	+		
	Water management (irrigation, drainage)	+	+/–	+/–	+/–			–	+	
	Rice management	+	+	+/–		+/–			+	
	Agro-forestry	+/–	+/–	–			+	+		
	Set-aside, land-use change	–	+	+	+	+	+	+	–	+
Grazing land management/ pasture improvement	Grazing intensity	+/–			+		+			+
	Increased productivity (e.g. fertilization)	+	+/–							
	Nutrient management	+	+/–	+	+		+	–	+	+/–
	Fire management	+	+			–	+/–			+/–
	Species introduction (including legumes)	+			+			+		
Management of organic soils	Avoid drainage of/restore wetlands	–			+		+	+	–	+
Restoration of degraded lands	Erosion control, organic amendments, nutrient amendments	+	+		+		+		+	+
Livestock management	Improved feeding practices	+			+/–				+	
	Specific agents and dietary additives	+								
	Longer term structural and management changes and animal breeding	+								
Manure/biosolid management	Improved storage and handling	+	+/–		+	+/–				
	Anaerobic digestion					+		+		
	More efficient use as nutrient source	+	+		+	+		+		
Bio-energy	Energy crops, solid, liquid, biogas, residues	–					–	+	–	
	Pertinent references (footnotes)	a	b	c	d	e	f	g	h	i

'+', a positive effect (benefit); '–', a negative effect (trade-off). The co-benefits and trade-offs may vary among regions. Economic costs and benefits are also often key driving variables.

[a] Foley et al. (2005) and Lal (2001, 2004a).
[b] Mosier (2002), Freibauer et al. (2004), Paustian et al. (2004) and Cerri et al. (2004).
[c] Lal (2004b), de Oliveira et al. (2005) and Rockström (2003).
[d] Lal (2001), Janzen (2005), Cassman et al. (2003), Cerri et al. (2004) and Wander and Nissen (2004).
[e] Mosier (2001, 2002) and Paustian et al. (2004).
[f] Foley et al. (2005), de Oliveira et al. (2005), Freibauer et al. (2004), Falloon et al. (2004), Huston and Marland (2003) and Totten et al. (2003).
[g] Lal et al. (2003) and West and Marland (2003).
[h] Balmford et al. (2005), Trewavas (2002), Green et al. (2005) and West and Marland (2003).
[i] Freibauer et al. (2004).

2.2.2 Sequestration of carbon

Johnson et al. (in press) defined sequestration of carbon as: "the transfer of atmospheric carbon dioxide into long-lived carbon pools, such as biomass (trees), biomass products (timber), biomass in soils (roots of perennial species, microorganisms) and recalcitrant organic carbon or inorganic carbon in deeper soil horizons". This distinguishes long-term capture of carbon from general capture of carbon during photosynthesis, where much of the carbon is recycled over short time frames (month to years) in processes such as respiration and decomposition. Developing long-term stores of carbon in soils requires protection from microbiological degradation, adsorption to clays and formation of organo-mineral complexes (Lal 2004).

In general, sequestration of carbon requires changes to land use practice. Lal (2004) identified a series of changes to traditional land use practices that would be required to increase carbon sequestration. These are reproduced in Table 3. Some of these changes are more relevant to New Zealand agricultural practices. Specific carbon sequestration measures or factors that will enhance sequestration and are relevant to New Zealand are discussed below.

Conservation tillage

Changing from plough to low- and no-till systems is likely to increase sequestration of carbon into soils. Reducing soil disturbance and incorporating cover crops in the rotation cycle is likely to increase soil carbon. Retaining crop residues on the soil surface makes the associated carbon available for incorporation, while offering other agronomic benefits. The importance of considering the site-specific conditions is highlighted by Lal (2004) and Johnson et al. (in press), who mention that under some conditions soils carbon might be lost, while nitrous oxide emissions might in fact increase. Adequate time is also required for increases in soil carbon to become evident.

Restoring degraded soils

Restoring degraded soils through establishment of conservation reserves offers potential for sequestration of carbon. This conservation practice might require inputs of limiting nutrients, or addition of organic-rich nutrient sources such as a livestock manure to achieve optimal sequestration. Conversion of marginal cropland to pasture is an example of a restoration practice that may deliver benefits in terms of soil organic carbon content (Lal 2004).

Table 3: Modifications to land use practices necessary to optimise carbon sequestration. Reprinted from *Geoderma 1–2:* R Lal "Soil carbon sequestration to mitigate climate change", p. 22. Copyright (2004), with permission from Elsevier.

Table 7
Comparison between traditional and recommended management practices in relation to soil organic carbon sequestration

Traditional methods	Recommended management practices
1. Biomass burning and residue removal	Residue returned as surface mulch
2. Conventional tillage and clean cultivation	Conservation tillage, no till and mulch farming
3. Bare/idle fallow	Growing cover crops during the off-season
4. Continuous monoculture	Crop rotations with high diversity
5. Low input subsistence farming and soil fertility mining	Judicious use of off-farm input
6. Intensive use of chemical fertilizers	Integrated nutrient management with compost, biosolids and nutrient cycling, precision farming
7. Intensive cropping	Integrating tress and livestock with crop production
8. Surface flood irrigation	Drip, furrow or sub-irrigation
9. Indiscriminate use of pesticides	Integrated pest management
10. Cultivating marginal soils	Conservation reserve program, restoration of degraded soils through land use change

Afforestation

Establishing new areas of forest, or converting marginal cropping or grazing land to forest offers considerable potential for carbon sequestration (Table 3).

A linear relationship exists between biomass carbon inputs to soil and soil organic carbon content. Viewed over a sufficiently long period of time, however, equilibrium may be established between sequestration rate and loss of carbon from the soil. Lal (2004) again highlights the importance of factors such as climate, soil type, tree species and management to the net sequestration of carbon.

Sequestering carbon through afforestation takes place in two basic ways:

1. Carbon may be sequestered to soils; and

2. Carbon may be sequestered in long-lived land cover (trees) or removed as timber products.

Net sequestration of carbon to soils through forestry is not accepted by all researchers. Following measurement of soil carbon beneath a 17–19-year-old pine forest in Northland, Groenendijk et al. (2002) concluded that carbon and nitrogen concentrations were 25–43 per cent lower in forest soils at depths of 120–180 mm than in the adjacent pasture. Following a modelling exercise, Halliday et al. (2003) provided an hypothesis to explain the observed decline in soil carbon concentrations. They were able to relate the observed decline primarily to the soil nitrogen status. The almost immediate decline in soil carbon concentration was attributed to an imbalance in soil C : N ratio, caused by rapid plant nitrogen uptake. Soil carbon declined to maintain a suitable C : N ratio. This hypothesis was supported by Kirschbaum et al. (in press), who also attributed this behaviour to the nutrient status of the soil prior to conversion to forestry. Following an earlier modelling exercise by Kirschbaum et al. (2003) it was concluded that "the slow build up of carbon matches the gain in nitrogen, phosphorus and sulphur."

These studies confirm that determining the net sequestration following afforestation may be complex – it is important to recognise the complex interactions between soil nutrient status, soil properties, rainfall and ultimately, the influence of climate change itself.

2.2.3 Reduction of methane emissions from animal waste management systems

Methane emissions from agriculture are dominated by enteric fermentation, yet it is useful to consider emissions from animal wastes. While improved management of animal wastes in New Zealand will probably not reduce atmospheric methane emissions significantly, the water quality co-benefits may be important.

Improved management of animal wastes (waste feed, manure and urine) are probably only feasible for intensive production systems such as piggeries and feedlots, and they may prove useful for some producers in New Zealand. Recovery of the methane from animal wastes would require modification to herd and waste management systems. Waste materials would need to be recovered and transferred to an appropriate storage/treatment facility. Anaerobic treatment would be optimised to favour methane production. The resulting methane would be recovered and used as an energy source (locally or off-site as heating or electrical power). Liquid and solid material remaining after anaerobic treatment would be used as a soil conditioner and nutrient source. The nitrogen would be present primarily as NH_4^+-N, providing two specific benefits: in this form the nitrogen is readily available for plant uptake, while remaining relatively immobile within the soil. Lower leakage of nitrogen applied in this form may be anticipated, provided the nitrogen application rate matches crop requirements.

In addition to water quality co-benefits, additional, direct greenhouse gas mitigation benefits would be achieved through replacement of fossil-derived energy with energy derived from the recovered biogas.

2.2.4 Reduction of methane emissions from surface waters

Wilcock and Sorrell (2008) recently reported that lowland streams in pasture catchments with dense stands of macrophytes, were net sources of methane (as a result of plant decomposition). Annual emissions were compared on a catchment area basis to other non-animal sources within agricultural systems (Table 4). It is clear that stream sources are relatively minor when compared to emissions from intensively grazed sheep and dairy cattle pastures.

Table 4: Annual emissions of methane per ha of catchment from agricultural streams compared with emissions from farmed land sources in New Zealand.

Source of emission	Emission ±SD (mol/ha/yr)	Reference
Toenepi Stream	20±4	Wilcock & Sorrell 2008
Whakapipi Stream (summer)	54±11	Wilcock & Sorrell 2008
Sheep pasture	2,100±90	Lassey et al. 1997
Dairy pasture	18,000±750	Lassey et al. 1997

Clearly, enteric fermentation from ruminant livestock is the single most important source of methane emissions in New Zealand and mitigation measures would probably be best directed toward the improved management of animal wastes (through the creation of point source methane sources, from which the biogas could be captured and used), rather than management of emissions from extensive, non-point sources of methane, such as pasture. Water quality co-benefits may, however, be derived from mitigation strategies directed at either source of methane.

2.2.5 Reduction of N_2O emissions from soils

Globally, 65 per cent of all emissions may be attributed to the soil microbial processes of nitrification and denitrification (Johnson et al. in press). In New Zealand, nitrous oxide represents about 17 per cent of total greenhouse gas emissions (as carbon dioxide mass equivalents). According to the latest greenhouse gas inventory for New Zealand (MfE 2007), 95 per cent of animal wastes are returned to "pasture, range and paddock" (i.e. soils), with the remaining material treated in anaerobic ponds.

Emissions of nitrous oxide from agriculture may be attributed primarily to application of nitrogen fertilisers and animal wastes to soils. Seventy to 90 per cent of ingested nitrogen is excreted by animals via urine and dung. Up to 30 per cent of the nitrogen applied to land as fertiliser or in animal wastes 'leaks' from the soil surface through leaching (Reay et al. 2003). Pasture-based animal production systems create the potential for partial anaerobiosis, which may favour coupled nitrification-denitrification processes. Emission of nitrous oxide from soils and pasture is a consequence of these processes (Oenema et al. 2005). Nitrous oxide may also be lost directly from drains and streams that contain material leaking from pasture systems (Reay et al. 2003; Wilcock & Sorrell 2008).

Emission of nitrous oxide from soils is a function of the amount of nitrogen applied, soil organic content and moisture status, and temperature (Oenema et al. 2005). High moisture levels within the soil may cause anoxic conditions, which will favour nitrous oxide production.

Reducing nitrous oxide emissions will therefore require more efficient use of the nitrogen applied to soils. Management of the nitrogen applied to pasture will need to recognise the significant cycling that takes place through the livestock, where the bulk of nitrogen consumed is returned to the pasture in faeces and urine. Application of nitrogen in excess of crop requirements will create a pool of nitrogen-forms that may be converted to N_2O, while also favouring leakage of nitrogen through the soil profile to shallow groundwater and surface waters (primarily as nitrate). Additional nitrous oxide can be produced from the chemical reduction of nitrate in drainage waters, under anaerobic conditions.

Nitrification inhibitors offer one mechanism for reduced nitrogen leakage from agricultural systems. Nitrification inhibitors block oxidation of ammonium to nitrite/nitrate in soil and thereby permit greater opportunity for uptake of nitrogen (as ammonium) by plants (and animals grazing the pasture). Dicyandiamide (DCD) is marketed by Ravensdown Fertiliser Cooperative Limited as Eco-N™ and by Ballance Agri-Nutrients Limited as DCn. It is currently the only nitrification inhibitor available in New Zealand in a formulation suitable for pastoral applications (Suter et al. 2006). It degrades in soils to CO_2, NH_3 and water and has a soil half-life of 111–116 days at

8°C and 18–25 days at 20°C (Bronson et al. 1989). In addition to nitrification inhibitors, Summit Quinphos market SustaiN, a granulated urea formulation that contains Agrotain® – a urease inhibitor that blocks the conversion of urea to ammonium.

Research conducted in New Zealand using DCD applied to urine patches has shown annual nitrate leaching reductions of 30–79 per cent, reductions of nitrous oxide emission of 61–91 per cent , and increases in annual pasture yield of 0–36%. However, many of these experiments have been conducted under relatively ideal conditions for nitrification inhibitor efficacy (Suter et al. 2006).

Loss of nitrogen applied to soils may also be reduced if the period between application of nitrogen and its subsequent uptake by plants is reduced to a minimum. Introduction of nitrogen into the soil horizon where the crop may utilise the nutrient immediately also favours retention of applied nitrogen within the agrosystem (Amon et al. 2006). Precision farming is an extension to these approaches, where the application of nitrogen is matched to agronomic requirement at paddock scale – application of nitrogen in excess of crop requirements is thereby avoided (Velthof et al. 1996).

Amon et al. (2006) indicate that a large soil C : N ratio makes the nitrogen relatively unavailable for plant uptake. This nitrogen becomes available later in the crop cycle when not actually required by the plant. Animal wastes processed using anaerobic digestion reduce the potential for this situation – the carbon content in the waste is reduced (through formation of methane), while the nitrogen fraction is readily available to the crop, as well as being in a less mobile form. The latter characteristic offers water quality benefits but results in emission of a greenhouse gas, although under controlled anaerobic waste digestion the potential for methane capture exists.

2.2.6 Reduction of nitrous oxide emissions from surface waters

Wilcock and Sorrell (2008) recently reported that lowland streams draining catchments with a significant amount of pasture, or with significant inputs of groundwater containing nitrate from market gardening, were also net sources of N_2O. Annual emissions were compared on a catchment basis (Table 5) and, although small by comparison with emissions from intensively grazed pastures, they may be comparable with low-intensity pastures and other agricultural land uses.

Table 5: Annual emissions of nitrous oxide per ha of catchment from agricultural streams compared with emissions from farmed land sources in New Zealand.

Source of emission	Emission (±SD) (mol/ha/yr)	Reference
Toenepi Stream	6±1	(Wilcock & Sorrell 2008)
Whakapipi Stream	4±1	(Wilcock & Sorrell 2008)
Whangamaire Stream	6±1	(Wilcock & Sorrell 2008)
Dairy pasture	150	(MfE 2007)
Beef cattle pasture	50	(MfE 2007)
Sheep pasture	30	(MfE 2007)
Emissions from nitrogen Fertiliser Irrigated pasture	100, 143	(de Klein et al. 2001)
Emissions from animal waste irrigated pasture	43, 14	(de Klein et al. 2001)
Soils with crop residue incorporation	2.5–286	(de Klein et al. 2001)

3. Climate change mitigation measures and water quality co-benefits and co-costs

Consideration of water quality benefits should also take into account stream habitat quality and health, or life-supporting capacity. Streams are products of their catchments, and changes to land use are closely linked to changes in stream quality. Natural forested riparian systems provide a range of functions ('ecosystem services') that maintain bank and channel stability, and excellent water quality and in-stream habitat. Land clearance for pastoral agriculture, and the subsequent ingress of livestock to riparian areas and streams, has caused the following types of damage (Parkyn & Wilcock 2004):

- **Degraded remnant native vegetation in the riparian zone** – from original land clearance and stock damage from trampling or eating.

- **Reduced shade and shelter** – resulting in drying of soils and microclimate exposure in riparian zones, and heating of the stream water and growth of nuisance algae and macrophytes.

- **Compacted and damaged riparian soils –** with reduced infiltration capacity and reduced trapping capacity for land contaminants.

- **Destabilised stream banks and channels** – resulting in erosion, streambed siltation and water turbidity.

- **Reduced water quality** – owing to mobilisation of sediment, and direct input and overland flow of nutrients and faecal microbes from animal wastes.

- **Degraded stream habitat and reduced stream health** – resulting from the above damages – as indicated by changed composition of aquatic invertebrate animals, and reduced abundance of certain native fish.

Measures to combat climate change that involve alteration to land management or riparian vegetation will have consequences for stream health. The mechanisms for this are outlined below.

3.1 Extensive afforestation

3.1.1 Floods, sediment yield and erosion

Blaschke et al. (2008) have reviewed the literature determining the likely impact that reversion of pasture and afforestation of pasture lands may have on stream flows, floods, erosion and landslip potential in New Zealand. These features are very closely related to water quality issues generally, and relevant conclusions of the Blaschke et al. (2008) report are summarised in the following sections.

Relatively few sediment yield studies comparing different land uses in New Zealand have been undertaken. These may either exclude large events, or be skewed by inclusion of a few extreme events. In general, however, sediment yield of forested and scrub covered catchments is lower than that of pasture-covered catchments. Retention of scrub or bush causes lower sediment yield than pasture or clear-felling.

Sediment yields in large catchments are very strongly associated with unstable geology and high rainfall. Where these two characteristics coincide, very large sediment yields may be anticipated. In addition, these physical characteristics override the impact of vegetation cover.

Table 7 of Blaschke et al. (2008) indicates that large reductions in sediment yield may be anticipated should tree-planting be targeted at erosion-vulnerable terrain. Data from the Waipaoa River catchment indicates that the density of landslides was 0 to 0.2per km^2 for forested catchments, whereas it was 0.4 to 3.2 per km^2 for pasture. In addition, sediment accretion in the channel decreased from 89 mm/yr to 26 mm/yr following afforestation.

Reid and Page (2003) reviewed data for this catchment as well. They identified that a relatively small portion of the landscape produces the majority of landslide material. They estimated that a 40 per cent reduction of the sediment yield may be possible if the most vulnerable 8 per cent of the catchment were forested. The corollary of this was that a reduction of landslides by 40 per cent would reduce sediment yield by only 6 per cent . Results for the Waipaoa River indicated that where terrain is highly erodible because of geological characteristics, elevated sediment yields persist following afforestation. High yields also persist because of the reservoir of easily eroded material within the channel that may be mobilised by future moderate to large flood events.

For small catchments, afforestation or reversion to scrub of the entire catchment may be expected to reduce sediment yields by between 50 and 80%. In larger catchments, high sediment yields are associated with relatively small areas of unstable, easily eroded material and relatively infrequent high rainfall events. Reducing sediment yields by more than 50 per cent requires complete afforestation of areas of high sediment yield.

Blaschke et al. (2008) identified three scenarios:

1. Diffuse (non-targeted) reversion or afforestation of a catchment would reduce sediment yield from areas of geological instability the least. Little impact on overall catchment sediment yield may be anticipated because impacts are highly localised.

2. Widespread reversion to scrub or forest will reduce catchment sediment yields if areas of instability are targeted. Sediment yields will still, however, remain dependent on rainfall.

3. Whole-catchment afforestation will reduce terrestrial erosion because erosion-prone areas will be covered. The benefits will be proportional to the extent of the catchment that is erodible. Low or no benefits may be anticipated from areas of the catchment not at risk of erosion.

Erosion benefits

Reduced erosion offers localised and downstream benefits. Retention of soil on catchment slopes retains nutrients, reducing fertiliser requirements, while also retaining soil-bound pesticide residues.

Widely spaced trees planted within pasture offers the potential to reduce erosion by about 20 to 100%. Close-canopy plantings offer the potential to reduce erosion by between 10 and 100%, while reversion to scrub offers the potential to reduce erosion by about 20 to 100%. The reduction in erosion achieved through these planting scenarios depends on the quality of planting (i.e. the extent of cover of most unstable areas of the catchment).

Floods and sediment yields

Blaschke et al. (2008) concluded:

- Increasing forestry would probably reduce the flood **peak** significantly in small catchments where a significant proportion of the catchment is afforested. However, there would be a smaller impact on the magnitude of flood flows in larger catchments where the proportion afforested is typically smaller.

- The magnitude of flood **volume** in large and small catchments is essentially unchanged following afforestation – the flood flow persists over a longer period.

They also highlighted that the size of the flood flow is dominated by physical factors – topography, the depth and intensity of the precipitation, the soil structure and the underlying geology.

Predicting the impact of afforestation on flood flows was shown to be complex and difficult. Forest blocks tend to be established in small units within a catchment, surrounded by other land uses. Simple calculations demonstrated that impacts on flow (and thereby sediment yield and nutrient export) were difficult to predict, and that overall impacts might be smaller than anticipated.

3.1.2 Carbon sequestration via extensive afforestation

Tables of carbon sequestration rates demonstrate that a relationship exists between the extent of afforestation and the mass of carbon that will be sequestered (e.g. MAF 2007). Meaningful sequestration of carbon dioxide therefore requires establishment of large areas of forest.

General water quality changes

Opportunities exist to improve water quality by targeting areas with existing water quality issues due to current land use practices, or areas that are vulnerable due to grazing. These are likely to be in areas of steep slopes that are particularly prone to soil loss, areas of low productivity farming, dangerous areas where stock losses may be high, or where there are numerous small streams and wetlands and it would be impractical to fence each one.

Options for management of these areas could include passive reversion of pasture to scrub and native forest, active planting of native forest or plantations of pines or other exotic or native trees, or agroforestry systems where appropriate.

Conversion of pastureland into pine plantation generally improves stream water quality and biodiversity by reducing inputs of contaminants such as sediment, nutrients, pathogens and agri-chemicals. Conversion to plantation also provides stream habitat conditions more similar to those of the native forests that covered most of New Zealand before humans arrived (Fahey et al. 2004; Parkyn et al. 2006; Quinn et al.

2007). Removal of grazing animals and planting trees instead reduces erosion of steep land; this typically increases water clarity and reduces siltation that can clog streambeds. Exceptions to this may occur where stream channels have narrowed during the pasture phase after the original conversion from native forest to the pasture phase. In the Waikato basin, channels of many small streams have narrowed by 50 per cent in pasture due to high sediment inputs and grasses invading the stream channel (Davies-Colley 1997). In this situation, re-establishing shade during afforestation appears to lead to a period of increased bank erosion in small streams after the pasture grasses are shaded out and channels widen to the size of a forest stream (Quinn et al. 1997a). This releases the sediment stored in the banks and, this has been predicted to cancel out the benefit of less sediment coming off the hills over the 25–30 year period of the first pine crop rotation at a Waikato hill catchment (Collier et al. 2001). However, this is only expected to occur while the channels widen during the first rotation and streams were predicted to have improved water clarity and export less sediment subsequently. Channel narrowing in pasture decreases as catchment area increases (Davies-Colley 1997) and does not occur in all parts on New Zealand (e.g. not observed in the Nelson area: Baillie & Davies 2002).

Shade from riparian afforestation generally improves stream habitat by controlling blooms of algae and high water temperatures that are associated with absence of sensitive invertebrate species from many unshaded pasture streams (Davies-Colley & Quinn 1998; Quinn 2000). Leaf litter input to pine streams is also similar in quantity and seasonal pattern to native forest streams (Scarsbrook et al. 2001) and this helps restore natural forest food webs (Hicks 1997). Afforestation leads to inputs of large wood that provides a variety of habitat functions in natural streams (Baillie & Davies 2002; Meleason et al. 2002; Collier & Bowman 2003), although modelling studies indicate it takes hundreds of years for natural levels of large wood volume to be restored (Meleason & Hall 2005). These changes result in in-stream plant biomass and fish and stream invertebrate communities in late rotation pine forest streams being similar to those in native forest (Hicks 1997; Rowe et al. 1999; Harding et al. 2000). However, total in-stream biomass and productivity of invertebrates is usually lower in small pine and native forest streams than in pasture, due to lower in-stream plant growth and temperatures (Hicks 1997; Quinn et al. 1997b).

Reduction of stream temperatures following afforestation is greatest and occurs most rapidly in small to medium streams where riparian vegetation can fully shade the stream channel (Davies-Colley & Quinn 1998). A study of reduction in stream temperature as forest recovered following clear-felling, found that annual cooling rates for summer daily mean and maximum temperatures ranged from 0.18 and 0.47 ˚C yr^{-1} for the largest stream (12 m channel) to 1.4 and 1.9 ˚C yr^{-1}, respectively in the smallest stream (2 m wide channel) (J. Quinn, NIWA, pers. comm.).

The influences on water quality and stream habitat of catchment afforestation in pines, that account for 90 per cent of the planted forests in New Zealand, vary during the typical 27-year forest 'rotation' (i.e. period from planting to harvesting and replanting) (Fahey et al. 2004). Nutrient retention by the forest (and hence reduction in ground and surface waters) is expected to be greater in the early-to-middle part of the rotation, when trees are growing most actively, than when trees are reaching maturity (Quinn and Ritter 2003). Harvesting reduces nutrient retention and increases the likelihood of nutrient and sediment loss through soil erosion (e.g. Fahey et al. 2004), but high nutrient retention is re-established with re-growth of the new tree crop (Quinn and Ritter 2003). Nutrient losses are often greatly reduced following clear-felling because of the development of a groundcover by weeds and a soil microbial biomass that help retain soil nutrients on site (Parfitt et al. 2002). The magnitude and duration of logging impacts on stream habitat and biota vary with stream size (i.e. small streams tend to be impacted more by clear-felling but also recover more quickly than large streams), and management practices such as slash management (Collier & Bowman 2003) and retention of forested riparian buffers (Boothroyd et al. 2004; Quinn et al. 2004).

3.1.3 Water quality benefits from extensive afforestation

Review of available information

Nutrients

There is considerable information from monitoring studies in New Zealand that forests improve water quality. A summary of measured nutrient exports from catchments in New Zealand (Elliott and Sorrell 2002) demonstrates that the yield of nitrogen and phosphorus from intensive pasture catchments is greater than from native bush or exotic plantation (predominantly pine) catchments, with intermediate yields from low-intensity pasture (Figure 1). A similar finding holds for concentrations of these nutrients in stream water (Figure 2).

Figures 1–3 are box plots enclosing 50 per cent of the data, with the median value of the variable displayed as a line. The top and bottom of the box mark the limits of ± 25 per cent of the variable population. The lines extending from the top and bottom of each box mark the minimum and maximum values within the data set that fall within an acceptable range. Any value outside of this range, called an outlier, is displayed as an individual point. The dotted line in each box is the mean value.

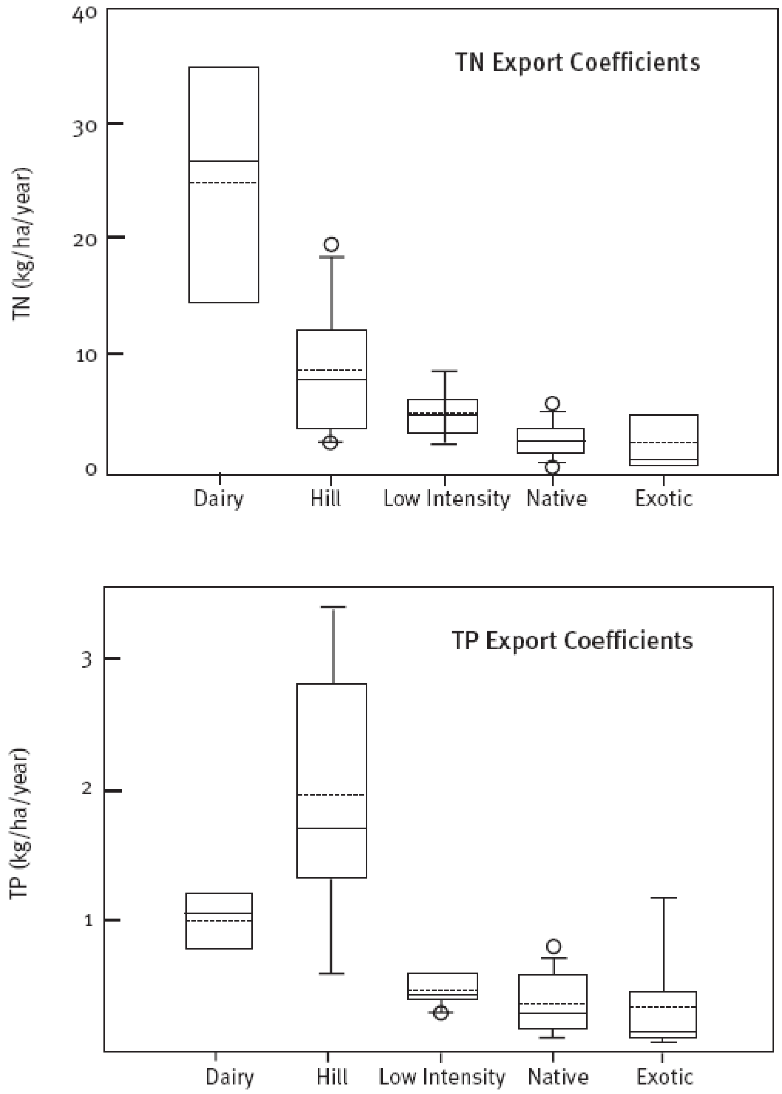

Figure 1: Summary of measured nutrient exports, from Elliott and Sorrell (2002). Exotic = primarily pine plantations. TN and TP are total nitrogen and total phosphorus, respectively.

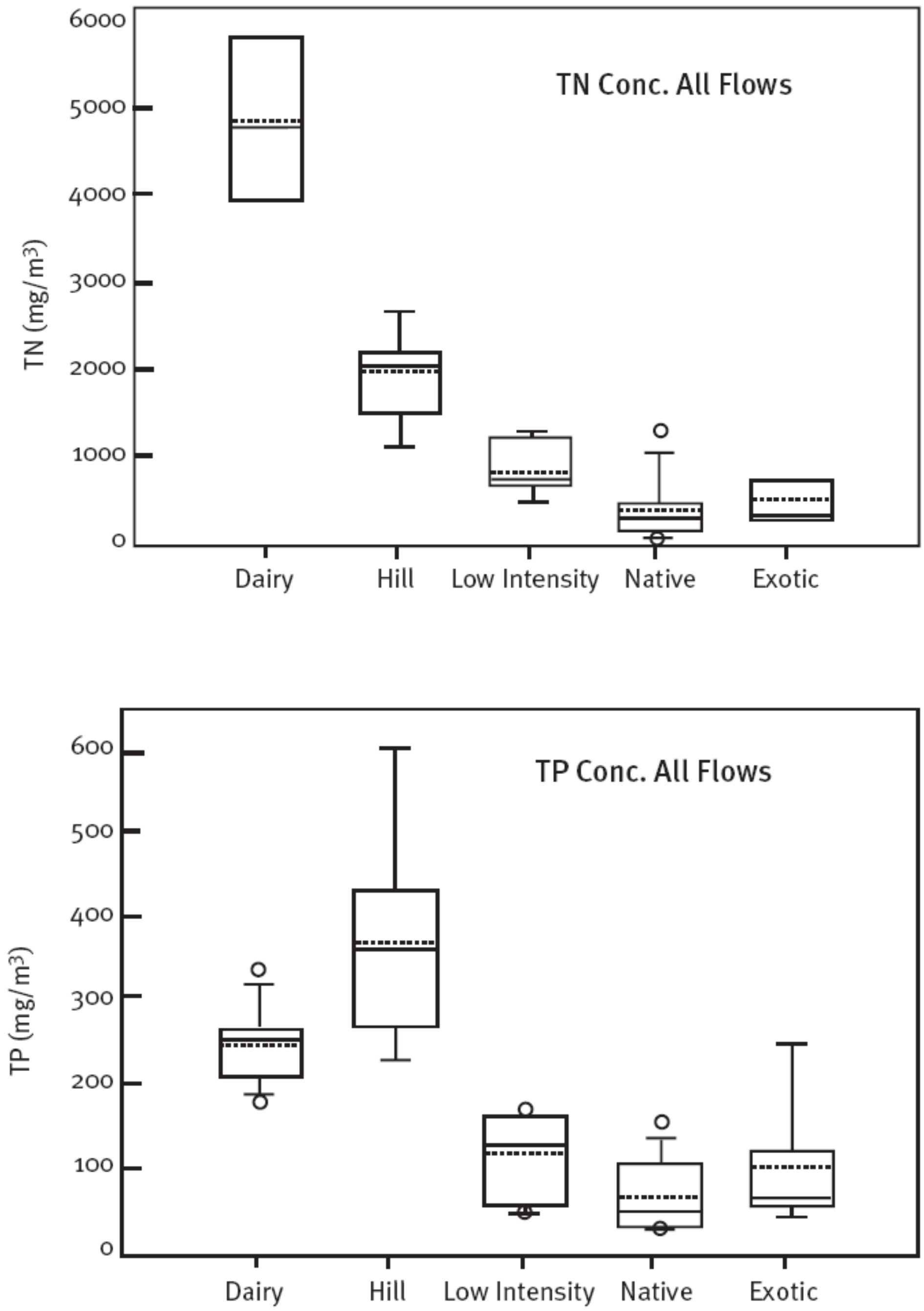

Figure 2: Summary of measured nutrient concentrations in streams, from Elliott and Sorrell (2002). Exotic = primarily pine plantations.

Scarsbrook (2006) found a highly significant correlation between median concentration for three nitrogen species and total phosphorus, and percentage pastoral land cover, in the National River Water Quality Network. It is possible, however, that this reflects (in part) site selection (for example, non-pasture sites being located in naturally less fertile soils). Moreover, the non-pasture sites did not necessarily have tree cover.

Modelling of nutrient yields from the National Rivers Water Quality Monitoring network using the model SPARROW also demonstrates that yields of Total nitrogen and Total phosphorus from pasture areas are higher than for pine and native forest catchments (Elliott et al. 2005). In a more recent application of SPARROW (Woods et al. 2006), the source terms for Total nitrogen were 53 kg/ha/yr, 8.4 kg/ha/yr, and 4 kg/ha/yr for diary pasture, other pasture, and trees respectively. There were low yields for other land-uses such as tussock lands. These source terms are modified by rainfall and soil factors (higher losses for higher rainfall and for poorer drainage), and decay within the streams. For TP, source terms were 4.36, 0.51, and 0.26 kg/ha/yr for dairy, other pasture, and non-pasture land-uses (including trees), respectively. However, for Total phosphorus there was an additional source term relating to erosion, which was added to these yields. Most recently, leaching predictions from the Overseer model have been combined with SPARROW within the CLUES modelling framework (Woods et al. 2006). This takes more explicit account of the variation in land-use intensity for pastoral land-uses.

In paired catchment studies, where there are catchments of contrasting land-use with similar soils and climate, the concentrations and loads in streams with pasture catchments are generally higher than from forest catchments. Examples are:

- Total nitrogen (TN) and total phosphorus (TP) yields from a pine catchment at Purukohukohu, in the central North Island, were 11 and 6%, respectively, of yields from an adjacent pasture site 10 years after the pine catchment was itself converted from pasture (Cooper & Thomsen 1988). The late-cycle concentrations were high compared with earlier stages of the cycle, and reached values expected for pasture catchments (Parfitt et al. 2002), although concentrations dropped considerably post-harvest. In that case, the pine was planted on pasture, and this raises questions about whether the residual elevated soil fertility from the previous pasture was resulting in additional leaching (Parfitt et al. 2003).

- At Mahurangi, the Total nitrogen yield from a catchment with mature Redwoods was 8.9 kg/ha/yr compared with 19.51 kg/ha/yr for an adjacent pasture catchment, based on 1 year of sampling (Stroud & Cooper, 1997).

- Average (geometric mean) concentrations of Total nitrogen and Total phosphorus in a stream with a pine/regenerating scrub catchment at Whatawhata, Waikato, were 52and 71 per cent of pasture values (Quinn & Stroud 2002).

- At Taita, McColl et al. (1977) found the nitrogen yield from pine was 24 per cent of the yield from pasture. Pine was converted from native forest and was planted in western red cedar, Douglas fir, and Corsican pine 12 years before monitoring.

- Macaskill et al. (1997) found that for streams in the Rotorua Lakes Region, median concentrations of Total nitrogen and nitrate were greater in pasture streams than in pine or native bush streams by a factor of 4 to 5, but the concentrations of Total phosphorus were largely independent of land-use.

- In a stream draining late rotation pine at Pakuratahi, coastal Hawkes Bay, total phosphorus was 67 per cent lower than in a pasture stream, but nitrate concentrations were 39 per cent higher (Quinn & Kemp 2001).

- Pine afforestation of the most erodible half of a hill-land catchment at Whatawhata, Waikato (in combination with changes in stock type, riparian fencing and planting, and poplar planting in areas that remained in pasture) was associated with reductions in the export of suspended sediment (76%), Total phosphorus (62%), and Total nitrogen (33%) in the first four years after changes were implemented (Quinn et al. 2007). Annual average stream temperature also declined by 5°C at a site downstream of riparian planting of native shrubs four years after changes at this site.

- Dyck et al. (1983) measured higher nitrate concentrations in soil water under Douglas fir compared with *Pinus radiata*. This suggests that different species of pine will reduce nitrate leaching by differing degrees.

Afforestation also influences the timing and forms of nitrogen and phosphorus loss. In a comparative land use study, losses of nitrogen and phosphorus from pasture were unevenly distributed through the year, with most export occurring as particulate forms during storm events. By comparison, exports from the two forested catchments were more evenly distributed and base-flow dissolved forms of nitrogen and phosphorus formed a higher proportion of total export (Cooper & Thomsen 1988).

Typically pasture soils build up nitrogen over time, and once trees are planted, some nutrients could be removed via leaching. Hence pine forest planted in pasture has the potential to lose more nitrogen than pine planted in bush. As an example, if the soil in pasture contains an additional 1 g/kg nitrogen (0.1 per cent by mass), then there would be 1000 kg of nitrogen per hectare which could potentially be released. Whether this is released depends on the fairly complex cycling of nutrients and carbon in the soil-plant system. Ultimately, when pine is planted on pasture we would expect the losses to revert to levels comparable to those generally found for pines planted on less fertile soil. The time for this adjustment is unknown. There is also limited data to assess whether the losses from pines planted on pasture are greater than pines planted on bush or scrub, and how the losses compare with pasture. The studies at Purukohukohu and Whatawhata are the only documented examples in New Zealand.

The legacy effects of pasture add uncertainty to predictions about the effect of forestry on the losses of nutrients. Further data comparing nutrients in pasture to catchments with pine planted on pasture would be valuable in providing additional information to address these questions. It would also be appropriate to complement such monitoring studies with long-term (decadal) simulation of pasture planted on pine.

Nitrogen-fixing tree/shrub species, such as broom and gorse, can input 100–200 kg N/ha/yr to soil during the phase of high biomass accumulation (Egunjobi 1969; Watt et al. 2003) and gorse leached more nitrate than pine in a comparative study on pumice soils (Dyck et al. 1983). Thus, if reversion from pasture to native forest occurs via an extended phase of gorse or broom dominance, the reduction in nitrogen leaching to waterways may be delayed considerably.

Microbial contamination

Contamination of stream water with faecal microbes is lower in forest than pasture streams, although the presence of feral mammals (e.g. rats, deer, goats, possums) and birds results in non-zero background levels in forest, based on a national survey of freshwater microbial organisms in streams (FMRP programme, McBride et al. 2002). Median concentrations of *E. coli* in that study were approximately tow times larger for pasture catchments than for forest/bush catchments, although there was considerable variability within land-use. Campylobacter concentrations were similar for forest and pasture sites.

Median faecal coliform and enterococci concentrations in Pakuratahi Stream, draining a Hawkes Bay pine catchment, were 55and 17 per cent of corresponding values in an adjacent pasture stream (Quinn & Kemp 2001). At Whatawhata, median *E. coli* levels were an order of magnitude lower than in pasture streams in catchments where pines had recently been planted and pest animals controlled (annual medians 398 cf. 41 *E. coli* /100 ml) (Donnison & Ross 2004). Streams in more established pine plantations and native forest had intermediate *E. coli* levels (83 and 100 *E. coli*/100 ml), probably reflecting input from feral animals.

Within the Waikato Region, Collins (2003) found that concentrations of *E. coli* were approximately four times higher in pastoral streams than in non-pastoral streams, most of which have tree or scrub land cover (Figure 3).

Scarsbrook (2006) found a highly significant correlation between *E. coli* concentrations and percentage pastoral land cover for the National Rivers Water Quality stream monitoring network.

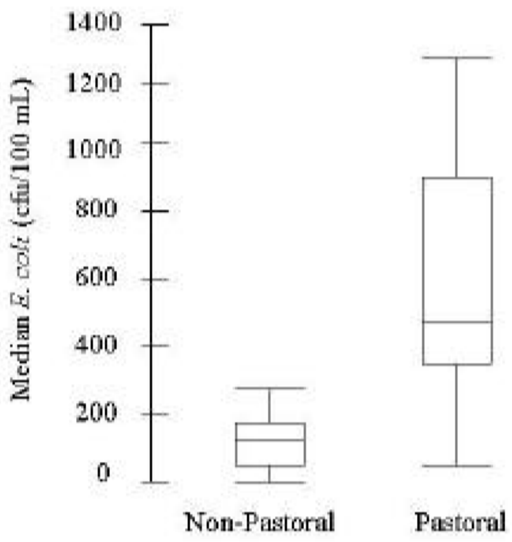

Figure 3: Concentrations of *E. coli* in pastoral and non-pastoral streams in the Waikato. From Collins (2003). Pastoral and non-pastoral are defined as catchment having greater than 90 per cent of the land in pasture or non-pasture land-cover, respectively.

Sediment

Blaschke et al. (2008) reviewed paired small-catchment studies of sediment yield, and concluded that sediment yields for native forest are lower than those from pasture by "substantially greater than 50%, and generally more than 80%", while the reductions for pine forestry in relation to pasture were "substantial, though variable". Also, yields from scrub were "substantially lower than from established pasture, by amounts ranging from 39to 100 per cent". They also refer to studies of large-scale afforestation at Waipaoa, and note that the reductions in erosion following afforestation may take decades to be expressed at the catchment outlet for large catchments.

Modelling sediment yields at a national scale (Hicks et al. 1996), showed the effects of rainfall and geology over-rode effects of land-use. The variability in sediment yields around the country is so great that the relatively small influences of land-use are difficult to detect.

Pine afforestation of the most erodible half of a hill-land catchment at Whatawhata, Waikato, in combination with changes in stock type, riparian fencing and planting, and poplar planting in areas that remained in pasture, was associated with 76 per cent reduction in the export of suspended sediment (Quinn et al. 2007).

Forest harvesting generally (but not always) increases sediment yield for a short time, but averaged over the growth cycle the losses from forestry are less than for pasture (Fahey et al. 2003).

Summary

The above information is summarised in Table 6.

Table 6: Summary comparing water quality effects on receiving water, of pine in relation to pasture.

Water Quality Parameter	Typical reduction due to pine forestry (% of pasture value)	Degree of certainty	Comment
Nitrogen yield	10–50%	Moderate to high, but uncertainty about residual effects of pasture soil fertility	Effect expected to be less/delayed if pine planted on pasture
Nitrogen concentration	15–50%	Moderate to high	Lower than for yield, as forest also reduces flow. Effect on nitrate probably less than for TN
Phosphorus yield	10–50%	Moderate to high	Depends on erosion
Phosphorus concentration	20–100%	Moderate to high	
Sediment yield	10–100%	High	Considerable variability
Turbidity	Significant	Moderate	
E. coli concentration	25–50%	Moderate	Depends on degree of pest control
Temperature	Significant	High	Variability based on stream size

3.1.4 When afforestation would have significant effects on water quality

The results above suggest that tree establishment would have significant effects on water quality. Some particular points to note in this regard are:

- Phosphorus losses are influenced by erosion. Hence tree establishment would have greater effects on phosphorus yields in areas with higher erosion, such as areas with high rainfall and high erodibility (Blaschke et al. 2008). Clearly, planting erosion-prone pasture areas would have a larger benefit.

- Benefits of afforestation would be greatest in areas of intensive farming. For example, establishing trees on intensive sheep and beef pasture on hilly country in the Waikato would have more effect than establishment on extensively-grazed tussock areas.

- There is some uncertainty about the residual effects of previous pasture fertility on the nitrogen losses from afforested areas. If any such effects occur, they are more likely on areas where pasture soil fertility has built up most. However, such areas are also the areas where long-term benefits from trees are likely to be larger due to the greater losses from the more intensive pasture land-use.

- Trees are likely to have most effect on improving water temperatures when planted adjacent to small headwater streams, and they may also have some additional (although uncertain) effect on nutrient removal compared with trees planted elsewhere. However, planting next to streams is likely to have more of an effect on flow reduction, and any harvesting in the riparian margins is likely to have a greater effect on water quality than harvesting elsewhere – depending on stream size. Effects will be greater in smaller streams.

- The benefits to be gained from improvements in water quality depend on the sensitivity of the receiving environment. Reducing the nitrogen load by a tonne would have more water quality benefits for Lake Taupo than for a short West Coast stream. The water quality benefits would be greatest if afforestation were targeted on catchments with sensitive receiving waters or where the receiving water bodies are being stressed.

3.1.5 Summary of water quality benefits from extensive afforestation

Afforestation of stream catchments provides the following water quality benefits:

- Reduces inputs of nutrients and contaminants and stabilises stream banks directly and indirectly by excluding livestock.

- Provides inputs of woody material and leaf litter for food webs and cover habitat for aquatic fauna.

- Provides shading of water, reducing summer temperature extremes and primary production from photosynthesis, resulting in less aquatic weed and periphyton, and less variable acidity (pH) and dissolved oxygen regimes.

- Reduces extreme rainfall-runoff events; afforestation of pasture can produce substantial reductions in annual peak stream-flows due to greater interception losses and lower soil moisture under forest than pasture (Fahey et al. 2004). This has benefits of reducing stream bank erosion and other flooding impacts on habitat that result from small to moderate storm events.

Other benefits

- Improves terrestrial and aquatic biodiversity.

- Improves visual amenity (during tree growth period, but not during or after harvesting).

- Provides other opportunities – recreation (e.g. tramping), generally improves catchment water quality, and is more likely to satisfy cultural requirements.

3.1.6 Disadvantages for water quality from extensive afforestation

Conversion of pasture to pine forest typically reduces water yield[1] by about 200–300 mm/yr over a period of about 10–20 years (Fahey et al. 2004). The impacts this has on stream flow, the duration of flow in small streams, and groundwater recharge depend on the total rainfall and the hydrogeology of the catchment.

Pine afforestation may cause intermittent water quality issues related to the long-term production cycle (i.e. low/no cover, thinning/maintenance, harvest/clear-felling).

3.2 Riparian afforestation

Intensively farmed lowland pasture is highly productive and valuable, which may provide a disincentive to fencing of riparian margins (i.e. loss of productive land). However, riparian protection and planting offer advantages for managing stock (i.e. not having to get them out of waterways or losing stock through drowning or falling into gullies, providing some shade and shelter for them), as well as providing water quality benefits, including direct benefits to livestock health following access to water of better quality.

[1] Equivalent to a reduction of 6-10 L/s/km^2

Riparian planting to protect stream values is recognised as a positive activity and is actively promoted by regional councils (e.g. Environment Waikato's Clean Streams Project) and industry sector groups (e.g. the Dairying and Clean Streams Accord).

Establishment of riparian plantings in a manner that will satisfy emissions trading schemes (ETS), 15 m either side of streams, will create significant buffer zones along stream margins and undoubtedly improve water quality (Quinn et al. 2004).

Riparian management can take several forms, from simple fencing and exclusion of stock to a multi-tier system involving grass buffer strips, production forest, and native forest plantings nearest the stream edge. The riparian buffer zones of most benefit to carbon sequestration would be those of forest tree species. However, periodic biomass removal/replanting of components of the buffer are expected to enhance the long-term maintenance of the nutrient removal capacity of buffer systems. Management of the nutrient attenuation and carbon sequestration functions of buffers needs further consideration.

Piecemeal afforestation of pasture is likely to have limited impact on small to medium flood events, and no impact on larger events (Blaschke et al. 2008). Hence riparian afforestation, even of 15-m wide zones either side of streams, is unlikely to have significant effects on flood flows.

3.2.1 Interception of contaminants by riparian forest

Reduction of nitrate inputs to surface waters by processes within the riparian zone

Riparian zones may deliver general water quality benefits by altering the flows of nitrogen in the landscape. Riparian zones have been associated with uptake and/or conversion of nitrogen moving through soils and groundwater as nitrate. As a consequence, nitrogen inflows to surface water may be reduced. Removal of nitrate in these zones occurs primarily through two processes – denitrification, a biologically mediated process, and direct uptake by vegetation.

A number of criteria must be satisfied for biological removal of nitrate:

- Denitrifying bacteria require a source of available organic carbon.
- Anoxic conditions are required, determined in part by:
  - extent of saturation of soils;

- residence times of groundwater within the soil profile.

- Adequate time is required for the nitrifying bacteria to interact with the carbon-rich substrate – this contact time is determined by the velocity of the groundwater flow.

- As soil organic carbon generally decreases with depth (unless specific deposits exist, e.g. peat), there also needs to be some barrier or confining structure that causes the nitrate-rich water to pass through and interact with the soil horizon(s) where adequate carbon-reserves occur.

Maximum removal of nitrate occurs when groundwater is constrained and flows through or is directed toward soil layers where adequate amounts of carbon exist, oxygen concentrations are reduced, and denitrifying bacteria are abundant. Nitrate removal is less effective where groundwater enters streams through upward flow. Under these conditions, the interaction between the dissolved nutrient, vegetation and riparian soils is not optimal, and in fact may not occur to a measurable extent.

Direct uptake of nitrate by riparian vegetation is also an effective removal mechanism. In this circumstance, criteria quite similar to those for denitrification must be satisfied:

- The tree crop must have a requirement for nitrogen to satisfy growth and maintenance.

- The nitrate-rich groundwater must interact with the root zone (this will also enable the tree roots to contribute carbon to the soil to facilitate denitrification).

- The width of the riparian planting must be adequate to allow sufficient contact time between the contaminated groundwater and the root zone.

A range of efficiencies of nitrate removal by riparian zones were identified from a review of data derived from the National Water Quality Assessment Programme in the USA. Puckett (2004) summarised the factors that explained these results, which included:

- Denitrification in the up-gradient aquifer due to the presence of organic carbon and other electron-donors (removal prior to entry into the riparian zone).

- Long residence time (>50 years) along the groundwater flow paths, allowing even very slow reactions to proceed to completion (removal prior to entry into the riparian zone).

- Dilution of shallow, nitrate-rich groundwater with older, nitrate-depleted groundwater (dilution enhancing apparent removal).

- Nitrate-rich groundwater bypassing riparian zones due to the extensive use of mole and tile drainage systems (non-treatment).

- Nitrate-rich groundwater passing beneath the root zone of riparian margins in deep flow paths, discharging into the stream water without effective uptake or interaction (non-treatment).

An important point made by Puckett (2004) was that "the long residence times of groundwater may pose an issue for future water quality in that in many areas where fertilizers have been applied for over 50 years, it may take that long for groundwater to move completely through the aquifer. Consequently, nitrate applied in the past may continue to be a problem for many decades to come, even if large-scale fertilizer reductions are implemented". Puckett also emphasised the importance of understanding the hydrology and biogeochemistry of groundwater; failure to understand these adequately would probably compromise the efficacy of riparian treatment systems.

Maitre et al. (2003) stressed the importance of understanding the hydrology of the system of interest. They measured nitrogen removal rates ranging between 2.2 and 7.6 mg N/m^2/day (up to 93 per cent removal) during the active growing period in summer. In winter, removal rates decreased to between 27 and 38 per cent of these values. The latter values reflected the higher nitrogen inputs during the winter period, as well as the reduced uptake by vegetation. Rainfall increased groundwater flows and diluted nitrate concentrations, particularly during spring. Reduced nitrate concentrations did not therefore indicate removal from groundwater. Maitre et al. (2003) also identified that denitrification rates were fairly stable. As the load of nitrate flowing through the system increased, the amount of nitrate passing through the zone where denitrification occurred also increased. Removal of dissolved inorganic nitrogen was strongly influenced by groundwater flow rates and therefore, residence times within the treatment zone. These factors were also observed by Poor and McDonnell (2007) in catchments in Oregon USA which had very seasonal rainfall. Pronounced dilution of nitrate-rich groundwater was observed in the winter.

Maitre et al. (2003) noted that efficient removal of nitrate from groundwater by riparian zones requires high-volume groundwater flows through biologically active zones.

Ryszkowski & Kedziora (2007) identified that nitrate concentrations in groundwater may reduce by 76 to 98%, with greatest removal from older trees (>95%). In addition, the ratio of NH_4^+-N to NO_3^--N increases within the soil profile of forested areas. Within forested areas and shelterbelts, nitrous oxide production is also reduced. Nitrous oxide production was 40– 60 per cent lower from recent (7–8 year old plantings) than from old (> 160-year-old) forested areas.

Yamada et al. (2007) estimated the nitrate removal that may be anticipated following establishment of a vegetated riparian buffer. The riparian zone consisted of a series of zonal plantings of a perennial grass (*Pannicum* spp.), lucerne, poplar and walnut trees. The tree species were inter-planted so that the rapidly growing poplars could protect the slower-growing, more valuable walnut trees. Each zone was a 5 m wide strip along a first-order stream. The opposite bank of the stream was retained in corn/soybean rotation, and acted as a control. Groundwater concentrations of nitrate and dissolved oxygen were measured over a six-year period – three years prior to and following canopy closure by the trees. Non-parametric statistics indicated that more substantial decreases in nitrate concentrations were observed in groundwater traversing the riparian plantings than the groundwater subject to the corn/soybean cultivation. The removal was attributed to direct uptake by the vegetation, because the dissolved oxygen concentration in the shallow groundwater was consistently above 5 mg/L, indicating conditions not conducive to denitrification. From this study it was concluded that surface water quality benefits may be anticipated within a few years of establishing a riparian margin.

Hefting et al. (2006) concluded that while direct uptake of nitrate by riparian plantings did occur, in many cases the removal performance was over-estimated because dilution by low-nitrate-containing groundwater was not considered. They also identified factors such as nitrate saturation and pH inhibition of nitrifying bacteria. Their research in the Netherlands indicated that dilution, probable inhibition of nitrification by high nitrate concentrations, and low pH were real phenomena. While they measured increased nitrate removal from a forested riparian zone relative to grassland, the effective removal rate was very similar for both buffer types. Overall nitrate removal rates of 38 to 63 per cent were observed. They felt that the vegetation type was not really significant in terms of nitrate removal under European conditions. The importance of high spatial variability was thought to be very significant, with localised differences in groundwater flow an important contributor to this variability.

While the consensus of recent research has been that riparian plantings, including forests, do reduce nitrate inputs to surface waters, the issue of the fate of the retained nitrogen is under some debate. Earlier work identifies incorporation in vegetation and conversion into unspecified gaseous products as the fate of nitrate in groundwater. More recent research encourages a more critical consideration of the products of denitrification. For example, Groffman et al. (2000) proposed that if nitrate removal rates were high and nitrate concentrations were low, nitrous oxide formation would be increased relative to nitrogen gas (N_2). Reduction of nitrous oxide is strongly influenced by low pH and oxygen concentrations – these inhibit the nitrous oxide reductase enzyme. As a consequence of these observations, Groffman et al. (2000) proposed that nitrous oxide formation in riparian zones was significant and should be included in IPCC calculations. They also made the point that riparian zones may not increase global nitrous oxide production, because the nitrous oxide formed may reduce the nitrous oxide that might form in other environmental compartments, such as estuaries or oceans. Riparian forests (along with wetlands and other water quality management tools) may, however, be amenable to management whereby the ratio of N_2O : N_2 may be optimised to reduce net nitrous oxide formation to a minimum.

Haag and Kaupenjohann (2001) related the nature of landscapes as heterogeneous patchworks, where nutrients are generated, stored or transformed, to mitigation measures such as riparian plantings. Their conclusions were that:

- Most landscape compartments were essentially ‘leaky’ with regard to nitrate, with ‘limited or uncertain’ retention potential.

- Storage of nitrogen within vegetation, soils or groundwater may disguise and ultimately only postpone the eventual emission of excess nitrogen.

- Denitrification, a significant mechanism whereby nitrate may be removed from groundwater in riparian zones, may ultimately only translocate the problem of nitrogen overload from soils and groundwater to the atmosphere.

Haag and Kaupenjohann (2001) concluded their review by stating that reliance on ‘end-of-pipe’ solutions (as they regarded buffer zones), was probably unsustainable in view of the uncertainties inherent with use of natural systems to achieve specific environmental outcomes. Their view was that **greater emphasis must be given to managing inputs of nitrogen to agricultural systems**. They recommended optimization of nitrogen fluxes and budgets at site, farm and regional levels.

A number of issues appear repeatedly in the scientific literature, suggesting that caution is necessary (e.g. Groffman et al. 2000; Hefting et al. 2006; Lovell & Sullivan 2006). These are posed here as a series of questions and answers:

“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' (e.g. 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, considerable 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 N_2O, particularly when the nitrogen removal capacity of the riparian buffer is approached (e.g. Groffman et al. 2000; Hefting. et al. 2003; Mayer et al. 2007). As noted in the answer to the previous question, 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.

Reduction of phosphorus inputs to surface waters by processes within the riparian zone

Storage and retention of phosphorus in riparian plantings depends on a number of processes (Mander et al. 2005):

- Adsorption to soil particles.
- Uptake and removal of soluble phosphorus by plants.
- Immobilisation within microbial biomass.
- Incorporation of organic phosphorus into peat.

The bulk of phosphorus removal takes place at the upslope edge of the riparian buffer 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 decreases as the load of pollutant increases, while retention efficiency is greatest for short duration rainfall events (Mander et al. 2005).

In general, the efficiency of a riparian buffer is increased by having sequential plantings of various plant communities, from grasses through to permanent tree plantings. This was confirmed by the research of Yamada et al. (2007) with regard to nitrate removal.

New Zealand research has demonstrated that stream concentrations of dissolved reactive phosphorus and particulate phosphorus follow seasonal patterns (McDowell

& Wilcock 2007). The material present in the Waiokura Stream in autumn was attributed to material derived from the catchment, whereas the material present in the stream in winter was more likely to be derived from stream banks, mobilised during high-flow events. Improving water quality would require improved riparian management – preventing access of livestock to stream margins, accompanied by riparian plantings that would reduce input from overland flows.

Additional management practices were identified by Monaghan et al. (2007) as necessary in order to reduce phosphorus inputs to surface waters. Their investigation in the Bog Burn Creek in Southland indicated that up to 33% of the phosphorus emitted from a dairy farm was in sub-surface drainage water that bypasses riparian processing. Preferential flow of polluted water into streams via drains has also been identified as a cause of reduced efficiency of riparian plantings in the management of nitrogen inflows to streams (Puckett 2004). Results from a long-term study of phosphorus dynamics in tile-drained watersheds in the Midwest USA indicated that a pool of readily mobilised phosphorus existed within the soil profile (Gentry et al. 2007) that contributed to increased phosphorus concentrations after any hydrological event (rainfall or effluent application). While riparian plantings were identified as a tool to mitigate inputs of phosphorus associated with particulate material, no equivalent tool existed for managing inputs of soluble phosphorus from shallow groundwater. In their earlier investigation of phosphorus loss in shallow groundwater delivered to streams via tile drains, Dils and Heathwaite (1999) identified that strategies such as 'in-stream' constructed wetlands or chemical filters would have to be considered to reduce the loads of soluble phosphorus in surface waters. The requirement for an improved understanding of surface and subsurface transport pathways was also identified.

Reduction of pesticide inputs to surface waters by riparian zones

Reichenberger et al. (2007) recently reviewed mitigation strategies whereby pesticide inputs to surface and groundwater may be reduced. In common with other pollutants, these substances are associated with and transported together with particulate material, as well as being transported in solution. In addition to these factors, the proportion of a specific chemical present in either form is influenced by the chemical properties of the chemical (specifically solubility and partitioning coefficient). In the review, results from 180 studies associated with mitigation of pesticide mobilisation were examined. The efficacy of various mitigation measures were then quantified for sediment and specific pesticides where possible (14 publications). Despite limitations associated with a number of studies, grassed strips were shown to reduce both pesticide and sediment runoff, while riparian forest buffer strips were thought to be less effective. Grassed strips were believed to better intercept sheet surface flow than forested riparian strips, where much of the surface-flow material actually entered the stream due to the formation of channels carrying concentrated flows. Pesticide material entering the riparian margin would also probably be closer to groundwater, which offered an alternate mobilisation pathway. In their evaluation of strategies to mitigate ingress of pesticide residues to surface waters, the effectiveness of riparian buffer strips were rated ‘low’ at farm scale, and ‘very low’ at catchment scale (Reichenberger et al. 2007: table 4).

Riparian plantings (particularly taller, forested ones) offer benefits in terms of interception of spray drift, which may reduce direct deposition of pesticide sprays to waterways very significantly. Results cited by Reichenberger et al. (2007) indicate that relatively narrow (3 to 10 m) wind break and riparian plantings reduced spray drift by at least 95% at wind speeds up to 4.5 m/s. It is likely that accidental inputs of aerially applied fertiliser to surface waters would also be reduced by riparian forests – the trees and shrubs would intercept and retain the fertiliser, attenuating inputs to streams and rivers.

In general, therefore, riparian forests offer:

- Limited benefits in terms of reducing inputs of pesticides to waterways associated with overland flow.

- Very significant potential for reducing contamination of surface waters from agricultural sprays.

Criteria for providing these water quality benefits include adequate height, width and density of forest cover. These criteria influence the movement of air (and associated droplets) through the landscape. Adequate density is also required to reduce air velocities, as well as providing sufficient surface area for adsorption of particles and droplets on vegetation. The efficacy of the riparian planting is also enhanced if it is continuous, reducing passage ways for preferential movement of contaminated air.

3.2.2 Buffer strip width

Buffers of 15 metres wide comprising native forest trees on either side of a stream would provide a sufficient width for the trees to form a self-sustaining community without the requirement for ongoing pasture weed maintenance (Parkyn et al. 2000, Reeves et al. 2006). This width would be sufficient to provide many of the water quality benefits of native forest catchments.

A review (Parkyn et al. 2000) of buffer strip width for supporting sustainable riparian vegetation and provide suitable functions for aquatic life found that for buffers 10–20 metres wide:

- Weed control may always be necessary along edges or for shade-tolerant weeds.

- Success depends upon establishing closed canopy cover early.

- Microclimate conditions comparable to those in forest interiors may not be achieved with buffers smaller than 40 m.

Parkyn et al. (2000) concluded that a buffer width of at least 10 m will achieve most aquatic functions.

3.2.3 Buffer function and form

Buffers of production trees (for timber, fruit, nuts, honey, flax or other products) could provide similar water quality benefits, but may have some impacts associated with harvesting. Selective harvesting of high-value timber trees would be preferable to clearfelling of timber trees in riparian areas, unless sufficient width of native forest tree species were retained along stream banks. Integration of trees within the agricultural landscape has long been a feature of agricultural systems in temperate climates (Herzog 2000). More recently, the positive impacts from inclusion of fruit, nut and other trees within existing agricultural systems has been recognised. In Europe, regulations and other measures prohibit removal of hedgerows and riparian tree belts, while incentives exist to encourage their re-introduction. These measures recognise the biodiversity, water quality and visual amenity benefits provided by these plantings (Herzog 2000).

Actively growing trees provide the benefit of greater use of atmospheric carbon dioxide and they also only take up nitrogen and phosphorus from subsurface flows during this stage. Mature forest trees have less demand for nitrogen and phosphorus and act mainly as a store of carbon. Riparian buffer systems that employ an aspect of renewal may provide better water quality and climate change attenuation benefits.

Peterjohn and Correll (1984) investigated the movement of nutrients through a small cropped catchment. Nutrient retention or removal in cropped land (8% N) was essentially related to removal in the crop itself. Riparian forest appeared to retain more nutrients (89% N). Phosphorus retention of the forest was about 80%; about twice as much as for the cropped land (41%). Calculations indicated that incremental growth of the forest accounted for about 15 kg N/ha/yr, approximately one-third of the nitrogen removed from groundwater. The removal of the other two-thirds of nitrogen from groundwater was attributed to denitrification. Similar results were obtained by Lowrance et al. (1984).

3.2.4 Water quality benefits from riparian forestry

It is widely recognised that riparian zones provide ecosystem benefits that include reduction of particulate-bound and soluble nutrient inputs to surface waters (e.g. Blackwell et al. 1999; Yamada et al. 2007). The extent to which this protection is provided depends on the complex interaction of a number of factors, including the size of the catchment, the hydrological regime, vegetation, soil types and underlying geology. The mechanism whereby the pollutant input is controlled also depends on the nature of the pollutant itself – nitrogen may be taken up by plants or lost in gaseous form following microbial action. Phosphorus may be retained by plant uptake or physical processes within the soil, such as adsorption. Pathogenic organisms may be retained within a riparian zone as a consequence of deposition, followed by inactivation due to desiccation or irradiation.

Ryszkowski & Kedziora (2007) recently reviewed the impact of trees (specifically shelterbelts) and water and nutrient movement through Polish landscapes. Trees generally increase evapotranspiration; this effect is magnified with increasing temperature. Evaporation may increase from about 35 to 50 per cent for shallow groundwater, and 15 to 30 per cent for deeper groundwater. Uptake of dissolved nutrients is similarly influenced by increases in temperature and the resulting increase in evapotranspiration from trees. In farmland with pasture and shelterbelts or forestry blocks, heat is transferred from the pasture to the trees through advection. This increases evapotranspiration from the forested areas, which in turn influences the movement of nutrients.

It is necessary to also consider where in the catchment the riparian zone may be established. Most of the water entering rivers enters via first and second order streams (Correll 2005). On a catchment scale, estimates of input vary between 60 and 70 per cent for first- to third-order streams (large rivers), and up to 90 per cent for first- and second-order streams (Haag & Kaupenjohann 2001). Sixty to 75% of stream flow was attributed to headwater streams in Japan (Anbumozhi et al. 2005). Low-order streams therefore may be expected to contribute most to stream nutrient and pesticides loads. Greenhouse gas mitigation measures associated with forestry may therefore be expected to have the greatest impact on water quality when established in headwater regions, specifically first- and second-order streams.

Revegetation of stream riparian zones provides water quality benefits similar to those of afforestation except that it leads to stream channel enlargement problems (see earlier section) without the benefit of afforesting the rest of the catchment.

To summarise, riparian forestry:

- Reduces inputs of nutrients and contaminants by encouraging infiltration, uptake, and denitrification as water passes through riparian zones.

- Stabilises stream banks directly and indirectly by excluding livestock.

- Provides inputs of woody material and leaf litter for food webs[2] and cover habitat for aquatic fauna.

- Provides shading of water, reducing summer temperature extremes and primary production from photosynthesis, resulting in less aquatic weed and periphyton, and less variable acidity (pH) and dissolved oxygen regimes.

- Improves terrestrial and aquatic biodiversity.

- Provides opportunity for walkways.

- Improves visual amenity.

- Will result in a lower water yield than is generated from pasture but maybe not as much of a reduction as would come from complete afforestation of the catchment.

- If grass strips are placed upslope from the woody riparian areas they can achieve some reduction in phosphorus and faecal runoff through settling and filtration processes that occur (Lucy McKergow, NIWA, unpublished results). Thus a combination of riparian tree planting for carbon sequestration and grass strips can achieve multiple benefits for stream water quality.

- Reduces impact of downstream flooding by slowing flood flows when water travels over rough vegetation on banks.

- Provides shelter for stock (might be increasingly necessary due to climate change impacts).

3.2.5 Disadvantages for water quality from riparian forestry

There may be intermittent water quality issues that are related to the long-term production cycle if production planting is alongside streams. These include loss of ground cover as a result of thinning and other maintenance operations, and harvesting – especially, clear-felling.

[2] A complex of interrelated food chains in an ecological community.

Streams with dense stands of aquatic plants are able to take up nutrients from the water column and thereby attenuate loadings from land to waterways. Shading will inhibit such growth, reducing attenuation so that large nutrient inputs will increasingly be transported (advected) downstream (Parkyn et al. 2005). Inputs of nutrients to the stream will, however, be reduced by the riparian margin itself.

For small streams where tree canopy closure is complete, pulses of sediments may be released from stream banks once shading (from trees) is established. Stream channels will return to widths that they had when previously forested. Stream banks will ultimately be more stable, but there may be significant sediment losses for a period when the stream channel is responding to the changed riparian conditions (Collier et al. 2001; Parkyn et al. 2005).

Riparian forest may also increase risks associated with localised flooding, including formation of debris dams at culverts.

3.3 Improved fertiliser application practice

Reducing nitrous oxide production can be achieved through:

- Reduction in nitrogen applied to land.
- Avoidance of anoxic conditions leading to denitrification.
- The use of inhibitors to block oxidation of reduced forms of nitrogen.

3.3.1 Reduction in applied N

A general reduction in nitrogen loading to land can be achieved through improved management of: fertiliser use; livestock waste products; and the hydrological pathways that connect land to waterways. Much of this can be achieved with nutrient budgeting that takes into account animal waste inputs as direct inputs to waterways. The timing of fertiliser application also needs to be improved to avoid excess nitrogen during periods of anoxic conditions in soil profile.

Benefits of lower nitrogen inputs to waterways include a reduced risk of eutrophication occurring in nitrogen -limited waterways, and better public health outcomes where nitrate levels may approach potable water limits.

3.3.2 Anoxic conditions

Anoxic soils are deficient in oxygen. Soils that are wet for long periods of time and are not free-draining often have oxygen levels at or near to zero. Heavy grazing can cause treading damage and pugging of soils so that drainage is impaired and soils remain wetter for longer. Denitrification of nitrate to nitrous oxide and nitrogen gas occurs in anoxic soils, with the proportion of each gas being dependent on soil pH and other factors (Groffman et al. 2000). Irrigation or prolonged rainfall can cause anoxia in poorly-drained soils.

3.3.3 Nitrification inhibitors

The most commonly used nitrification inhibitor dicyandiamide works by blocking the nitrification process (i.e. the conversion of ammonium, NH_4^+, to nitrate, NO_3^-). Nitrate must be present for denitrification to occur, during which nitrous oxide is formed as an intermediate product. Nitrogen is retained in soil as NH_4^+, which is less mobile than NO_3^- and is thereby more available for plant uptake.

Recommended application rates for DCD vary because of its reduced stability in warmer soils. Rates of 12.5 kg DCD/ha are recommended for one product, normally applied twice a year. This corresponds to an additional loading of 8–16 kg N/ha/yr and could present an additional risk of nitrogen loss to waterways if some of the DCD is lost in runoff.

Because of the costs associated with repeat applications of nitrification inhibitors in warmer climates their benefits for surface waters (reduced inputs of N) may be limited to cooler regions of New Zealand and may also be limited by the effects of climate change (Suter et al. 2006).

There is some uncertainty about the effects of mobilised DCD on wetland function, and most notably the capacity of wetlands to denitrify nitrate in runoff and thereby improve water quality in streams and rivers. This is a subject of current research.

A possible disadvantage of using nitrification inhibitors is that the additional pasture production resulting from better retention of nitrogen may lead to higher stocking rates and greater production of methane.

3.4 Use of wetlands to reduce greenhouse gases

3.4.1 Nutrient attenuation

Wetlands offer the potential for low-cost, low-maintenance nutrient attenuation, notably for removal of nitrate. Interception and filtering of surface and subsurface farm runoff by wetlands can complement good grazing and cropping practices to reduce rates of nutrient losses from agricultural lands (Tanner et al. 2005).

3.4.2 Wetland sources of CH_4 and N_2O

Historically, riparian wetlands have been promoted for their capacity to lower nitrate concentrations via denitrification. Although current research is unable to establish the conditions under which nitrogen gas (80% of the atmosphere) or nitrous oxide (a greenhouse gas) are the dominant end-products of denitrification (Groffman et al. 2006), in principle, nitrous oxide emissions from wetlands can be greatly reduced by promoting conditions that lead to complete denitrification to nitrogen gas. This occurs when groundwater nitrate concentrations are less than 2 mg N/L and flow rates are low (Schipper & Vojvodic-Vukovic, 1998).

Current research indicates that wetlands are net greenhouse gas sources (Wilcock & Sorrell 2008). However, methane production is tiny by comparison with ruminant production in adjacent pasture and nitrous oxide production can potentially be managed so that denitrification goes to completion with nitrogen gas as the end product. Removing wetlands to reduce greenhouse gas emissions would have little effect on overall emissions but would have a negative impact on downstream water quality.

The potential to reduce methane from wetlands is presently unknown but, in any case, it is probably not worth worrying about given the much greater emissions from stock.

3.5 Alternate waste management practices

Application of wastewater to pasture provides a measureable nitrogen loading rate. A direct link exists between soil nitrogen application rates and nitrous oxide produced within the soil. There is also a link between soil moisture status and nitrous oxide production. Reducing the amount of nitrous oxide from pasture systems will necessitate a reduction of excess nitrogen application, as well as a reduction in the occurrence of conditions within the soils likely to facilitate nitrous oxide production. This will require producers to apply nitrogen at optimal rates for pasture and soil uptake, and if feasible, reduce soil moisture conditions, particularly for soils receiving wastewaters.

Effective nutrient budgeting will also be an important factor reducing nitrogen outputs from soils to surface waters. Reducing nitrogen application to satisfy or only slightly exceed pasture or crop requirement will reduce potential for nitrogen transport to surface and ground waters.

3.5.1 Water quality benefits from alternate waste management practices

Applying nitrogen at rates closer to plant requirements, coupled with optimal timing of application (minimising anoxic conditions) and with use of inhibitors, could reduce amounts of nitrous oxide produced, and also reduce the mass of mobile nitrogen within soil/shallow groundwater. All of this should reduce inputs of nitrogen to streams, which will lower the eutrophication potential in streams, lakes and estuaries. Reducing nitrogen inputs to surface water will also reduce potential off-farm nitrous oxide production (within streams, wetlands and estuaries).

3.5.2 Disadvantages for water quality from alternate waste management practices

Land application of wastewater from livestock (e.g. dairy shed effluent) may cause greenhouse gas emissions where loading rates are too high and soils become saturated. This can cause water quality problems if surface runoff occurs or if the excess wastewater enters drainage systems and enters waterways without interception by riparian zones (i.e. short-circuiting). Thus, wastewater application should be matched with soil infiltration rates and not be applied where short-circuiting can occur.

3.6 Implementation of new technologies/strategies

It was noted previously that improved nutrient management on pasture would offer considerable potential for reducing nitrous oxide formation, as well as reducing leakage of nitrogen from pasture systems. Implementation of alternate animal management strategies may also offer water quality benefits.

Changes to the management and housing of animals at critical times (winter/spring), might also be considered. Housing stock in barns or herd houses will reduce inputs of nitrogen to soils at times when nitrous oxide formation is likely to be most favourable (cold conditions, waterlogged soils). The wastes from these housing systems may be more conveniently transferred to anaerobic treatment systems, from which the nutrients may be applied to land at a more favourable time.

Wastes partially treated under aerobic conditions may eliminate a significant proportion of nitrogen prior to application to pasture and encourage a higher proportion of oxidised nitrogen. Reduced nitrogen application rates will mean less nitrogen that is available for nitrous oxide formation and for movement to waterways via shallow groundwater. Application of technologies such as Advanced Pond Systems (APS), developed within NIWA, to dairy shed effluents may be particularly useful in this regard.

Anaerobic pond systems that are used commonly for treating piggery and dairy shed wastewaters also produce significant methane emissions to the atmosphere (Craggs et al. 2008). This can be mitigated by capturing the gas and flaring the methane as carbon dioxide (reducing the GHG load 21-fold) but NIWA research has shown that it is also practical to use the methane to generate electricity and heat, reducing both the farm's demand for external energy substantially or for direct use as a vehicle fuel. Similar methane emission issues are associated with dairy effluent storage for deferred irrigation to land to avoid surface runoff during wet periods (as currently advocated by Environment Southland, pers. comm. C Arbuckle).

Greater use of wetlands for the treatment of nutrient-enriched water may also be encouraged. Wetlands facilitate denitrification, whereby some NO_3^- is reduced and returned to the atmosphere as nitrogen gas, while other influent nitrogen is retained by wetland plants as biomass or is converted to organic forms that are less environmentally damaging.

3.7 Barriers to change

Smith et al. (2008) draw attention to the lack of progress that has been made with implementing climate change mitigation measures despite the biophysical potential that exists. In the context of carbon sequestration "very little progress has been made since 1990 and little is expected by 2010" (Smith et al. 2005). In their opinion the barriers to implementation of climate change mitigation measures would not be overcome without policy/economic incentives (Smith et al. 2008).

A theme common to many assessments of the benefits of climate change mitigation measures is that the process is complex, involves a number of players and requires time (Freudenberger & Harvey 2003; Lal 2004; Smith et al. 2005; Smith et al. 2007; Johnson et al. in press; Smith et al. 2008). Realizing the co-benefits for water quality will be subject to the same issues and complexities. While in some cases water quality benefits may be observed over shorter time-scales, they will be dependent on drivers that will facilitate or encourage the implementation of climate change mitigation measures.

The process of change is often slow and complex. In an assessment of biodiversity benefits following afforestation, Freudenberger and Harvey (2003) noted that "these case studies illustrate that even with a commercial driver such as hard wood plantations, reconstructing native vegetation is a slow and expensive process. It is a process that involves extensive planning and involvement of many landholders and agencies. Reconstruction proceeds in bits and pieces – a few shelter belts, riparian strips and portions of remnants are fenced at any one time, by any one landholder. From a policy perspective this is a major challenge; protection and reconstruction of native vegetation in agricultural landscapes will take long-term institutional support”. These comments probably apply equally to the process of climate change mitigation, and therefore to the likely water quality benefits.

4. Estimates of water quality benefits / costs in physical terms

4.1 Modelling approaches for identifying water quality co-benefits

Computer models have a potential role in assessing the water quality co-benefits of afforestation and nitrification inhibitors, by means of:

- Capturing the effects of environmental factors such as climate, soils, and slopes on water quality and greenhouse gas emissions.
- Consideration of land-use or land-management factors on water quality.
- Allowing the cumulative effects of mitigation measures at the catchment or national scale, including the case where measures are implemented in a partial or patchy fashion.
- Improving communication of results by predicting effects in hypothetical conditions.
- Improving understanding of processes and longer-term responses occurring on a time-scale longer than those of field trials (for example, predicting the effects of DCD on long-term nitrogen budgets, or predicting the changes in soil nitrogen stores following afforestation).

In the following sections a brief description is given of models that could be used to quantify specific aspects of water quality change resulting from greenhouse gas mitigation measures (such as extensive and riparian afforestation, nutrient management).

4.1.1 Nutrients

At the farm or paddock scale, several available models are relevant to assessment of the losses of nutrients and greenhouse gases from farms, and some of these can predict the effect of nitrification inhibitors.

The OVERSEER Nutrient Budgets Model (http://www.agresearch.co.nz/overseerweb) can be used for the assessment of mean annual nitrogen and phosphorus losses from pastoral and cropping and horticultural systems, and takes climate and soils into account. This model currently incorporates a component for estimating emissions of N_2O, and this component of the model has been used to assess the effects of farm management on nitrous oxide emissions (Wheeler et al. 2007). The model includes leaching (but not N_2O) losses from farm block with trees, but that component of the model is relatively unsophisticated as there is little available data (David Wheeler, AgResearch, pers. comm.). Also, the greenhouse gas model is more developed for pasture than for cropping. A new version of the model incorporating DCD for the pastoral component is currently nearing completion, and this incorporates available data from New Zealand and overseas (where relevant). This component of the model will assume DCD is applied to the manufacturers' specifications.

There are several more mechanistic daily simulation models for assessment of nutrient and greenhouse gas losses from farms:

- The DeNitrification-DeComposition model (DNDC) (http://www.dndc.sr.unh.edu) is particularly tuned for assessment of gaseous nitrogen losses, but it has not been applied in New Zealand and it does not have an animal component, limiting its applicability to pastoral systems.

- EcoMod. This model is being used by AgResearch as a research-level simulation model to investigate nutrient cycling and losses on farms (Johnson et al. 2003, 2008). It includes an animal component, and is aimed predominantly at dairy systems. The model predicts greenhouse gas emissions, including the split between nitrous oxide and nitrogen gas. It can also take into consideration the effect of nitrification inhibitors (such as DCD) by simulating a depressed rate of nitrification.

- SPASMO (HortResearch). This is a nitrogen loss model developed primarily for horticultural systems, although it has been applied to pastoral systems. The denitrification loss component of the models is currently being modified to improve the predictions of NO_3^- losses. There are no immediate plans to include nitrification inhibitors in the model, although it would be possible to add them (Steven Green, AgResearch, pers. comm.).

- Whole Farm Model (DairyNZ Ltd). This is a tool for testing farm systems and their components prior to field trials, and demonstrating and extrapolating research findings. This model aims to identify more efficient dairy production systems as well as new dairy systems that meet a wide range of financial, environmental and lifestyle goals.

- LUCI (Crop and Food Research) is a paddock-scale model being used to assess nitrogen leaching from flat fields with potato, maize and wheat, and a ryegrass/pasture component is being developed (or may now be available). This model does not include a GHG or DCD component.

- NPLAS (Environment BOP). This is a simple web-based decision support tool for assessment of nutrient losses in the Rotorua Lakes area. It does not include GHG emissions nor DCDs, and it includes a simple component for trees. It has now been linked with a simplified version of Overseer, but the future of this model is uncertain given the recent modifications to Overseer (incorporation of wetland, new hydrology module).

There are relatively few models for prediction of leaching from natural forest or afforested systems at the block or point scale. Miko Kirschbaum, Landcare Research, has used the CenW model (http://www.kirschbaum.id.au/Welcome_page.htm) to simulate evolution of nitrogen stores under *Pinus radiata* planted on pasture near Canberra, but applications to New Zealand have not been published. Such models would be valuable for assessment of the importance of previous pasture land-use on afforested areas, but this should be done with accompanying model testing.

Models for the loss of phosphorus are generally not as advanced as models for nitrogen loss. This is due to several factors, including the difficulty of predicting the loss of phosphorus in overland flow, complex phosphorus chemistry, and fewer field data. Overseer includes a phosphorus component, although it is less well developed than the nitrogen component.

At the regional or national scale, there are relatively few models that can be used directly to predict the effects of afforestation and DCD on nutrient losses. The CLUES model (Woods et al. 2006) is a system to predict nutrient flux, gross measures of farm economics and employment as a function of land use for each stream and its associated catchment in New Zealand. The model is calibrated to measure data in the National Rivers Water Quality Network. It uses a simplified version of Overseer for prediction of nitrogen losses. This model would be suitable for a regional or national-scale assessment of the effect of afforestation on mean annual nutrient losses. The model is being modified to include mitigation measures such as DCD use (FRST-funded), and a component for prediction of greenhouse gas losses has been commissioned recently by MAF. A microbial component is currently being added to CLUES. The current version of CLUES uses sediment loads from the Hicks erosion surface (Hicks & Shankar 2003), but this does not take changes in land use into account. At present, CLUES can only be used to predict the load of nutrients, whereas the concentration is also of biological relevance. Techniques for calculating concentrations from loads are currently being developed in CLUES projects. The model is relatively simple compared with the field-scale models, and does not take as much local information into account (for example, regional average stocking rates for a land-use are used instead of actual stocking rates). CLUES would be best at providing a broad-scale planning perspective of nutrient losses.

There are no models currently tested in New Zealand for assessment of the effect of riparian afforestation on nutrient yields (beyond the general effects of afforestation).

4.1.2 Temperature

NIWA has developed a detailed stream temperature model that was applied to the Whatawhata catchment, but there are no larger-scale applications. The WAIORA model prepared for the ARC can predict the effect of afforestation on a stream reach.

4.1.3 Sediment load

At the small catchment scale, the GLEAMS model has been used to predict the sediment load into estuaries in the Auckland area (e.g. Stroud et al. 1999). This model has been set up for afforested conditions as well as pasture, but it does not include a forest harvesting model (equivalent areas of bare ground are assumed), and it does not include mass erosion processes such as slips. Hicks and Shankar (2003) developed a prediction of mean annual sediment loads on a 100-m grid for New Zealand, based on geology and rainfall and calibrated to measure sediment loads around New Zealand. This has recently been modified to account for reductions in sediment load by afforestation, using assumed land cover reduction factors and satellite land cover information from the NZEEM model being developed by Landcare Research. The Hicks erosion grid has been included in the CLUES model for phosporus, and sediment load predictions are currently being incorporated into CLUES (these will take a form similar to the Hicks model).

There are few models for predicting the effect of riparian vegetation on bank erosion in New Zealand. The model SedNet has been used outside New Zealand to assess the effects of riparian vegetation, with the relatively simple approach that riparian vegetation reduces the bank erosion by a given factor (typically 95%). Before such a model could be applied in New Zealand, an assessment of the relative contribution of bank erosion would be required.

The risk of landslides has recently been modelled for the Manawatu (Dymond et al. 2006). This model applies empirically-based land-slide risks based on a geology-dependent slope threshold, tree cover, and terrain modelling of whether a slip reaches the stream. This approach has been used to identify farms most suited to erosion control in the Manawatu-Rangitikei catchments. The model is not yet available for the rest of New Zealand.

4.2 Models for assessing cumulative co-benefits, and locations for mitigation measures.

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 in Figure 4. Modelling of co-benefits (and potential confounding effects) should be carried out within this broad conceptual framework.

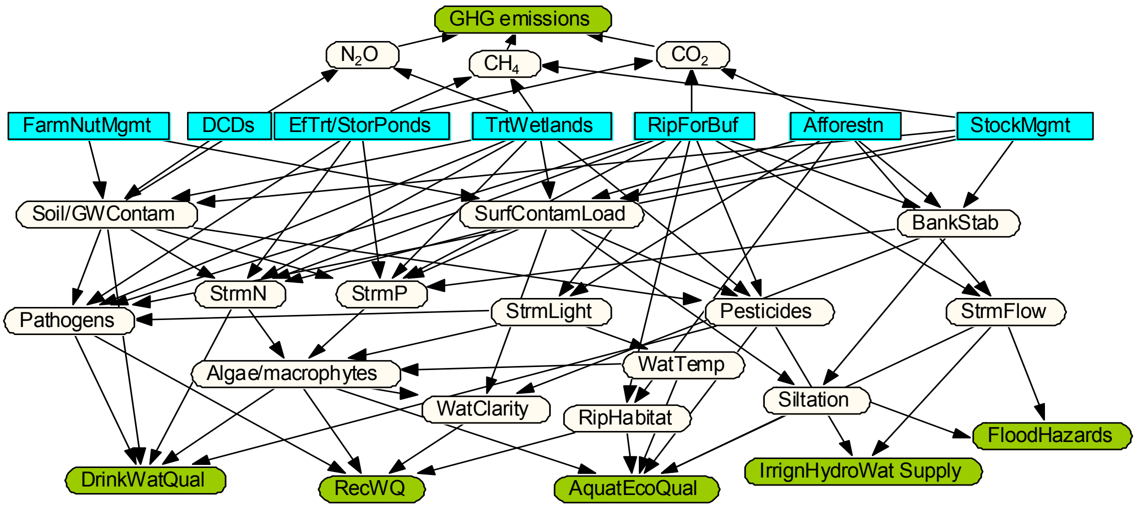

Figure 4: Conceptual model of links between key land management practices for GHG control and key waterway values (green ovals) via mitigation options (blue rectangles) and ecosystem attributes (clear oval) – see Appendix for explanation of terms.

We envisage two approaches in which models can be used to predict the reduction in nutrient loads associated with afforestation, DCD application and reduced nitrogen losses.

1. Run paddock-scale models for a range of stock types, stocking intensities, soil types, and climate, with and without DCD and with or without afforestation. Tabulate the results to give the reduction in nutrient load per unit area to which the mitigation is applied. This can be put alongside information on the reduction in greenhouse gas emissions and cost information to pick the options with the greatest overall benefit. These can then be mapped using databases (Agribase, Fundamental Soils Layer, Climate layers) to show the locations with the best options, and their extent. At present there is enough information to conduct this exercise for the effect of afforestation on nutrient loads (particularly nitrogen), with the limitation of the coarse nature of the assessment of nutrient loss under forestry. Little information is available on the effect of soils, climate, or previous land-use on the loss of nutrients, and

little incorporation of the effect of afforestation on phosphorus loss. With the current developments in Overseer, there should be sufficient information for the assessment of the effects of DCD (under optimal application) in the near future.

2. Run a national-scale model such as CLUES. This is currently feasible for assessing the implications of afforestation on nitrogen and phosphorus loss, again with the limitations of the fairly simple nature of the model, especially in relation to the representation of losses from non-pasture land-uses and the relatively coarse land-management information available in national databases. A benefit of this approach is that in-stream nutrient losses can also be considered, and it is also possible for the user to pose future land-use scenarios. It would be possible to extend this approach to nitrification inhibitors and to reductions in GHG emissions, once these are incorporated into CLUES (as is planned in the next two years).

Similar approaches could be adopted for sediment, although the relevant models are still being developed. Both of these approaches have limitations:

- The assessment would be for contaminant load, not concentrations within streams or lakes. Work is currently underway to include concentration information within CLUES.

- Data limitations occur in two ways: (i) It is often hard to get access to good agricultural databases detailing stock numbers and other important information, and (ii) local variations (e.g. between-paddock and between-farm differences in fertiliser use and grazing practices) are not easily accounted for.

- The assessments do not include information on time-lags between implementation of measures and effects in streams. Time lags can arise from groundwater delays, time for trees to grow (developing carbon storage, stream shade, etc) storage of sediment in the stream network, and adjustment in soil conditions from the previous land-use

- The sensitivity of the receiving environment is not considered. The benefits of a water quality improvement depend on where that improvement occurs. For example, reducing the nitrogen load by certain amount is likely to have more benefit in the catchment of Lake Taupo than in a high-rainfall west coast stream. This relates to the inherent sensitivity of the receiving water body, but also on any cumulative effects of land development already present in the catchment.

4.2.1 Modelling approaches – conclusions

A range of models presently exist or are being developed that can assess land-use change effects on water quality for a range of spatial and temporal scales. Generally these models apply to large-scale changes, such as extensive afforestation of pasture and adoption of nitrification inhibitors. With the exception of predicting water temperature, current models do not adequately predict water quality changes as a consequence of riparian afforestation. Models that can cope with localised changes (e.g. constructed wetlands, riparian afforestation and grass strips) are currently being developed in New Zealand.

5. Other methodologies for measuring costs / benefits in physical terms

The use of models to evaluate co-benefits and costs has been discussed in the previous section. There is undoubtedly a pressing need to develop these models and, where appropriate, link them so that different options for greenhouse gas abatement can be properly evaluated for other environmental consequences. Another means of measuring cost/benefits is through natural, experimental studies on the effects of land use changes on water quality. However, the results of such experiments take many years before meaningful results can be obtained (e.g. Quinn et al. 2007; Wilcock et al. 2007).

Comparative studies at a range of spatial scales may be possible. By substituting space for time the differences in land use practices, GHG emissions mitigation and water quality co-benefits might be established. For example, extensive surveys of existing farming systems and receiving waters might be used to identify key factors in GHG emission mitigation and water quality maintenance in production systems. This would require close integration of disciplines such as environmental science, agricultural science and resource economics.

There is also a need for models that take into account receiving water sensitivity. One can imagine a simple spatial overlay of the catchment of sensitive waterbodies with maps of the locations of the most effective mitigation options, to identify the locations where mitigation options will have the greatest impact on water quality. Some related work is being conducted on a broader view of water quality. For example: assessment of catchments sensitive to dairying is a component of the Pasture21[3] CLUES project; the Environment Waikato Regional Futures project takes a multi-objective approach to land-use development; the Landcare Research EnSUS spatial expert assessment tool considers both land-use pressures and environmental sensitivity. More complex integrated approaches are generally at fairly early stages of development or application in New Zealand: therefore, in the short term, simple methods such as overlays with catchments of sensitive water bodies would be most appropriate.

Another useful approach would be to extend the scope of thie present contract to review land use change and its effects on water quality (good and bad) as it applies to greenhouse mitigation measures proposed for New Zealand agriculture. A critical review of the literature would include all kinds of riparian vegetation experiments and land use conversions and would examine closely a wide range of plant species with a view to optimising both aspects of greenhouse gas abatement and improved water quality. Such a review might also include measures aimed primarily at mitigating water pollution (e.g. wastewater treatment and effluent irrigation, natural and artificial

[3] "Delivering Environmental Solutions for Sustainable Productivity Outcomes for New Zealand's Pastoral Industries" (FRST Contract C10X0603)

wetlands, denitrification walls, riparian planting) (Schipper and Vojvodic-Vukovic 1998; Quinn et al. 2004; Tanner et al. 2005) and look at the greenhouse gas abatement options from those – i.e. the opposite approach to that taken in this review. Such an approach might be useful in addressing areas where water pollution is of particular concern (e.g. the central North Island lakes).

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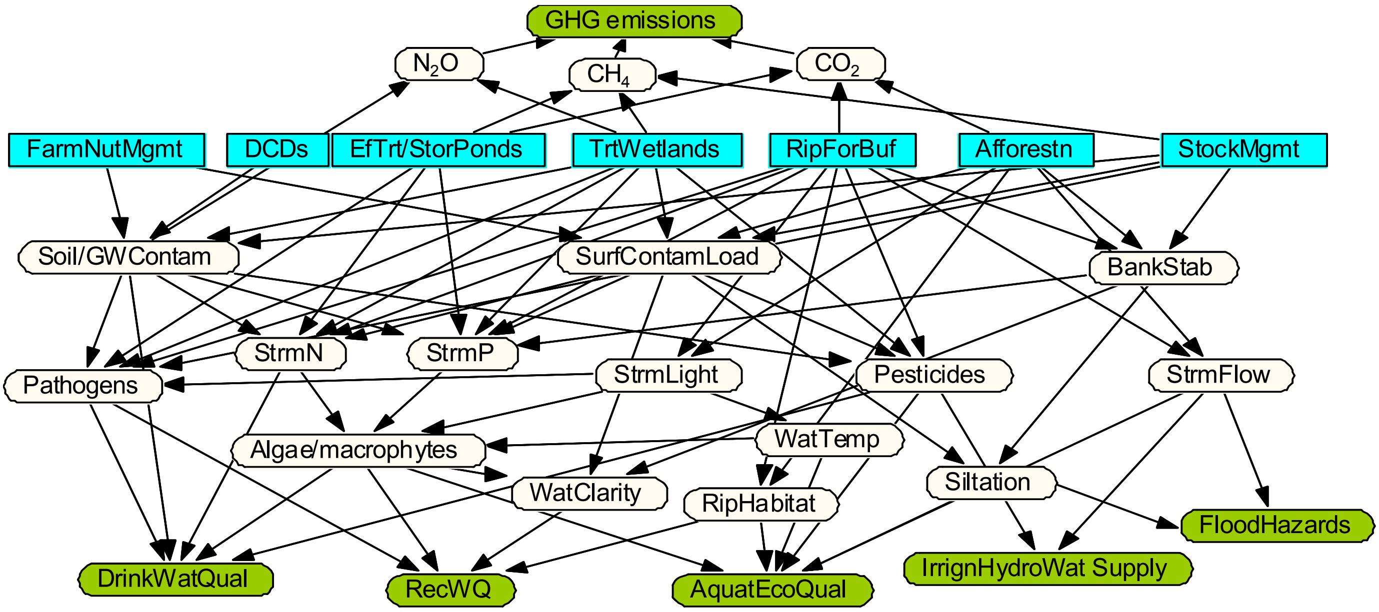
7. Appendix – Conceptual model



Conceptual model of links between key land management practices for GHG control and key waterway values (green ovals) via mitigation/management options (blue rectangles) and ecosystem attributes.

- The top levels lists the key greenhouse gases (GHG), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2). The next level (blue rectangles) contains key land management techniques that mitigate GHG emissions and aquatic impacts: from left to right 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 clear ovals in the middle level are drivers of key water quality properties that are affected by the management actions in the boxes above them: soil and ground water contaminant loads, surface water contaminant loads, destabilised stream bank sediment. The lower-level clear ovals are the aspects of the aquatic environment that are affected by these drivers and have more direct influence on aquatic values. These include pathogens (harmful microbes), stream nitrogen and stream phosphorus, stream light (for plant growth and temperature control), pesticides, stream flow, algae and macrophytes (larger plants), water clarity, riparian habitat and stream siltation. Water quality standards, criteria or guidelines typically target these aspects.

- The bottom level of green ovals are key aquatic ecosystem values, functions and uses: drinking water quality, recreational, aquatic ecological quality, irrigation and hydroelectric water supply, and flood hazard control.

- The conceptual model draws linkage lines between the major pathways to show how the management actions can influence waterway and atmospheric values. Whether these influences are positive or negative depends on the detail of actions in the 'management boxes'.