

CLIMATE REGULATION IN NEW ZEALAND: CONTRIBUTION OF NATURAL AND MANAGED ECOSYSTEMS

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ABSTRACT: This chapter reviews all stocks and fluxes of carbon in New Zealand, and reviews biophysical regulation through surface albedo. The terrestrial environment provides a climate-regulation service by assimilating, transforming, and adjusting to emissions of greenhouse gases that could otherwise lead to undesirable changes in global climate. Quantifying this service requires accounting for both stocks and flows. While greenhouse gas emissions and removals are reported in the national inventory, this inventory accounts only for human-induced changes in greenhouse gases, and omits some natural processes and ecosystems; for example, indigenous forest and scrub are not included but represent the largest biomass carbon pool in New Zealand. Emissions are mainly attributed to the energy and agricultural sectors, while removals come from exotic forestry and natural shrubland regeneration. Erosion plays a role as a carbon sink through natural regeneration of soil carbon on slopes. Biophysical regulation occurs through absorption or reflection of solar radiation (albedo). Forests tend to absorb more radiation than crops or pasture, thus contributing to a lesser extent to global warming. Government currently provides some mechanisms to incentivise sustainable land management in favour of increased forest area on lands unsuitable for agriculture. However, carbon stocks are also at risk of being lost through degradation of natural ecosystems, and this requires active management and mitigation strategies.

Key words: albedo, carbon, greenhouse gas inventory, managed ecosystems, national scale, natural ecosystems, managed ecosystems, trend.

INTRODUCTION

In the last two centuries, the earth has experienced unprecedented concentrations of carbon dioxide, nitrous oxide and methane. The rate of increase in these concentrations in the last 20 000 years is also unprecedented (Millennium Ecosystem Assessment 2005). The increase in temperature in the twentieth century is the largest during any century in the last 11 000 years (Marcott et al. 2013). There is now compelling evidence that this climatic shift is caused by human activities, in particular burning fossil fuels, as well as changes in land cover, increasing fertiliser use, and emissions from industrial processes such as cement manufacturing (Millennium Ecosystem Assessment 2003).

The radiative forcing of the climate system is dominated by long-lived greenhouse gases (GHGs), and in particular by CO₂. Global GHG emissions caused by human activities have grown

substantially since pre-industrial times, with an increase of 70% between 1970 and 2004 (IPCC 2007a). During this period, global annual emissions of carbon dioxide (CO₂) – the most important anthropogenic GHG – grew by about 80%, and represented 77% of total anthropogenic GHG emissions in 2004 (Figure 1).

Changes in climate have significant impacts on human health and well-being. Extreme weather events such as droughts and floods, which are expected to be more common under future climate change, make the environment unsafe by, for example, increasing the prevalence of infectious diseases and disrupting food supplies (Figure 2) (Millennium Ecosystem Assessment 2005). Climate change also affects the biosphere by altering patterns in land productivity, with both positive and negative outcomes (Kirschbaum et al. 2012b), and by shifting ecosystem boundaries, with consequences for biodiversity and pest distribution (Staudinger et al. 2012).

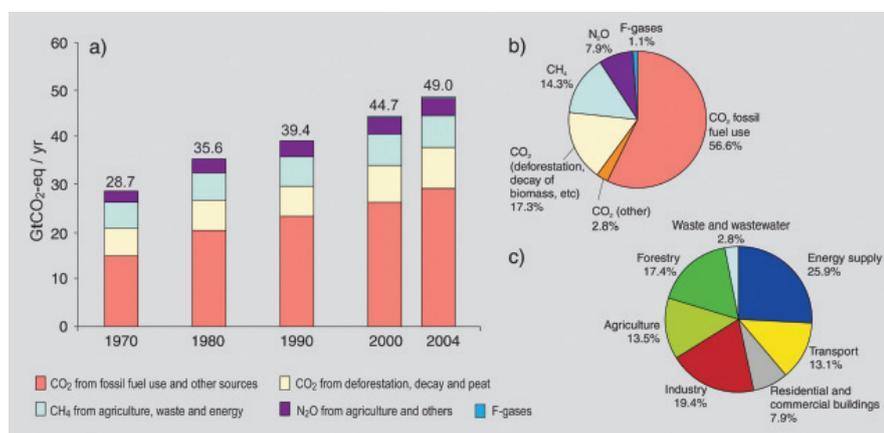


FIGURE 1 (a) Global annual emissions of anthropogenic GHGs from 1970 to 2004⁶ (b) Share of different anthropogenic GHGs in total emissions in 2004 in terms of CO₂-eq. (c) Share of different sectors in total anthropogenic GHG emissions in 2004 in terms of CO₂-eq. (Forestry includes deforestation.) (Figure 2.1 in IPCC 2007b).

Terrestrial ecosystems regulate global climate through two processes (Figure 2):

- *biogeochemical regulation:* ecosystems affect global concentrations of CO₂ and other greenhouse gases (GHGs) by storing them in plant biomass and soil;
- *biophysical regulation:* ecosystems alter radiative forcing by absorbing or reflecting solar radiation (both a function of surface albedo), altering the flux of water vapour to the atmosphere, and changing the energy transfer between the surface and the atmosphere.

Many managed or natural ecosystems affect the concentration of atmospheric carbon dioxide (Table 1). At the global scale, the energy and industry sector

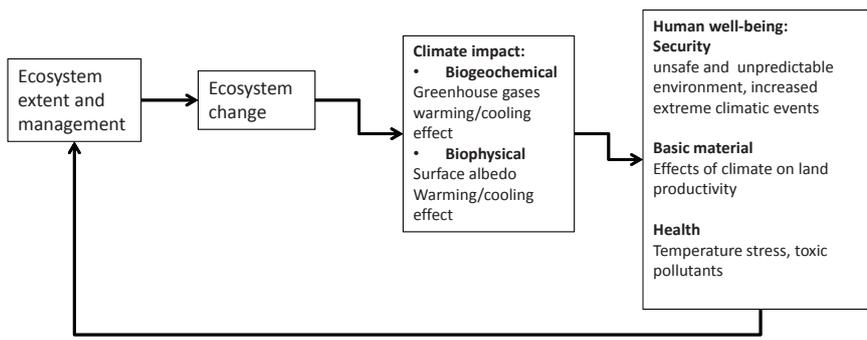


FIGURE 2 Ecosystem effects on climate regulation (adapted from the Millennium Ecosystem Assessment 2003).

TABLE 1 Summary of likely warming and cooling effects of various ecosystems

Driver	Energy /Industry	Livestock farmland	Exotic forestry	Natural forests /shrubland	Fresh-water wetlands
CO ₂	↗	↘	↘	↘	↘
CH ₄		↗			↗
N ₂ O		↗			
Surface albedo		↘	↗	↗	

contributes the greatest amount to carbon emissions. Farmland is usually a net emitter of greenhouse gases, contributing methane from livestock’s enteric fermentation and dung deposition, and nitrous oxide from fertiliser use and urine deposition on the soils. Forests and shrubland have a cooling effect because they sequester carbon in above-ground biomass, although this is partially offset by a warming effect from a lower albedo (Kirschbaum et al. 2011, 2013). Wetlands are the dominant natural source of global CH₄ emissions, but they also act as carbon sinks because their anaerobic conditions prevent decomposition of organic matter. This on-going carbon gain by wetlands tends to counter-balance their emissions of CH₄ (Whiting and Chanton 2001).

In New Zealand, greenhouse gas emissions and removals are reported under the United Nations Framework Convention on Climate Change (UNFCCC). New Zealand’s greenhouse gas inventory report includes the emissions and removals of greenhouse gases from all anthropogenic sources (Ministry for the Environment 2012).

While New Zealand’s emissions of long-lived greenhouse gases represent less than 0.2 per cent of total global emissions, in

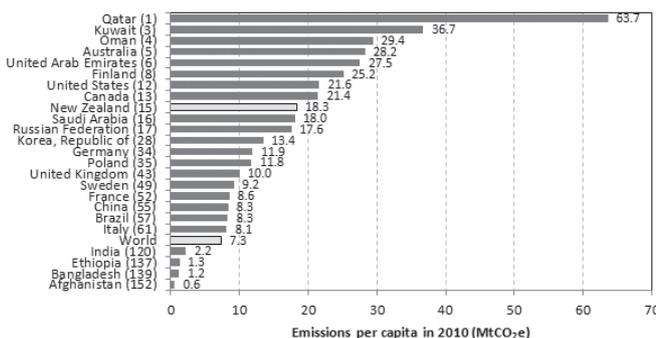


FIGURE 3 Total greenhouse gas emission (tonnes of CO₂ equivalent per year) per capita for selected countries in 2010 (European Commission 2011).

2010 it ranked 15th highest per capita out of 153 countries with populations above 1 million (Figure 3). However, a substantial proportion of the country’s emissions is generated during production of exported goods; when these are excluded and those generated overseas for New Zealand’s imports are included, the country’s ‘consumption’ emissions per capita drop by about 30% (Andrew et al. 2008).

The national greenhouse gas inventory is not exhaustive and some natural processes on managed land are omitted. These include the

capacity of some soils to oxidise methane (Price et al. 2004; Sagar et al. 2008), and the effect of erosion on carbon (Kirschbaum et al. 2009; Dymond 2010). When accounting for the sequestration of carbon from natural reversion of grasslands into shrublands, the inventory only considers non-forest land converted to forest since 1990, and this represents only 5% of the post-1989 forest category (Ministry for the Environment 2012). In reality, native shrublands in the natural forest category are also regenerating, and thus sequestering, carbon to some degree. However, the current inventory assumes that carbon stocks in natural forests do not change until re-measurements of the national plot network.

This chapter compiles information on the major contributors to the greenhouse gas budget in New Zealand, including the state of carbon stocks in various ecosystems and the current fluxes of the major greenhouse gases. It reviews trends in fluxes and conditions of managed and natural ecosystems for climate regulation. It outlines current emissions and sinks from managed and natural ecosystems. Problems that threaten the climate regulation service, options for sustainable management, and knowledge gaps are discussed.

CARBON STORAGE

Soil carbon

Soils represent the largest terrestrial pool of carbon and play an important part in the global carbon cycle. Soil carbon stocks in undisturbed ecosystems are generally in a steady state, with inputs from plants balancing losses through decomposition. The amount of carbon depends on the nature of vegetation, climate and soil type, but this level changes when management changes. For example, soil carbon is generally lower after conversion of pasture into forestry, but this is offset by the subsequent gain in carbon from tree growth (Guo and Gifford 2002; Kirschbaum et al. 2009).

Because soil is a complex system with processes operating at different spatial and temporal scales, numerical modelling is difficult (Vasques et al. 2012). At the broadest spatial scale, climate, the underlying geological substrate and geomorphological processes determine the overall trend in soil carbon; while at the finest spatial scale microbial decomposition of plant and animal residues as well as soil mineralogy and micro-physical structure have a pivotal role in forming and binding soil carbon (Stockmann et al. 2013). Similarly, long-term trends in land cover result in slow changes in soil carbon, while episodic events such as landslides operate at much shorter time scales.

Modelling an environmental parameter and its response to environmental factors over all spatial and temporal scales is a formidable task (Blöschl 1999), so most practical efforts have a narrow, spatial focus. They may, for instance, operate with high spatial resolution at a single location, or may operate over large distances with coarse spatial resolution. Moreover, the temporal

change of soil carbon is difficult to measure, partly because soil carbon changes slowly in response to climate or anthropogenic effects, and partly because national soil carbon sampling efforts are temporally unbalanced. Therefore, most studies either average soil carbon over all time scales or use observational estimates of soil carbon that are assumed to be at a common time. Exogenous information such as a land-cover change map is used to act as a surrogate for temporal change in the past or as part of a future scenario. Each of the above approaches has advantages and disadvantages, depending on the problem being addressed.

Accounting for soil carbon — In New Zealand, a Carbon Monitoring System (CMS) was developed by Landcare Research with funding from the Foundation for Research, Science, and Technology (FRST) and the Ministry for the Environment (MfE). This system uses a statistical model to estimate the total national soil carbon within different land-use and soil types, using a simplified land-use map and soil–climate types (Scott et al. 2002). By incorporating land use, the model can estimate not only current soil C stocks but also potential soil C changes that might accompany future land-use changes.

A refined version of this CMS, described by Tate et al. (2003, 2005) and Baisden et al. (2006), added a first-order estimate of susceptibility to erosion based on slope and annual rainfall (Giltrap et al. 2001). Changes in land cover/use were assumed to be the key drivers of annual and 10-yearly changes in soil C; other drivers (soil/climate/erosivity index) were assumed constant through time. A further refinement to the CMS model incorporated spatial correlation between soil samples, recognising that soils are sampled opportunistically rather than strictly randomly (McNeill et al. 2009, 2010, 2012).

Since the original development of the model, other sources of soil C data have been added by MfE, including random samples within natural forests, existing annual cropland records (McNeill et al. 2010), and records from perennial cropland (McNeill 2012). The inclusion of natural forest samples spatially balances the coverage, albeit in only one land-cover class, while the increase in the number of records reduces uncertainty in the estimated change in soil C caused by changes in land use.

Testing of the national CMS model continues: most recently, Hedley et al. (2012) tested it for stony and non-stony soils. These tests suggested that land management had not measurably affected soil C concentrations during any period after database samples were first collected.

Estimates of soil carbon pools — Scott et al. (2002) estimated soil C for 1990 as 1152±44, 1439±73, and 1602±167 Mt C for the 0–0.1, 0.1–0.3, and 0.3–1.0 m soils layers respectively (mean plus-or-minus the standard deviation). They found that New Zealand soil C values derived from the CMS generally contain higher soil C levels than the default IPCC values, despite the fact that the IPCC values are for undisturbed vegetation.

Tate et al. (2005) refined these estimates of national soil C to 1300±20, 1590±30, and 1750±70 Mt C for the 0–0.1, 0.1–0.3, and 0.3–1.0 m soils layers respectively. These figures are between 9% and 12% higher than corresponding values from Scott et al. (2002), and the standard error is significantly smaller. Tate et al. also found that most soil C is stored in grazing lands (1480±60 Mt to 0.3 m depth), appearing to be at or near steady state. The conversion of these grazing lands to exotic forests and shrubland contributed most to the predicted national soil C loss of 0.6±0.2 Mt C yr⁻¹ over the period 1990–2000. This represented a refinement of their earlier (Tate et al. 2003) estimate of national soil C losses of 0.9±0.4 Mt C yr⁻¹ for all land-use changes over the

1990–2000 period; in that study they identified uncertainties as arising mainly from estimates of area changes and coefficients associated with land-use classes with limited soil C data. The latest greenhouse gas inventory extends by a further 10 years the period over which these stocks and losses are estimated; using an IPCC Tier 1 method, it reports an average loss from conversion of grazing lands over 1990–2010 of 0.41 Mt C yr⁻¹ (Ministry for the Environment 2012).

Table 2 describes some soil carbon density values from Scott et al. 2002 and the values used in the last two National greenhouse gas Inventory Reports (NIR) (Ministry for the Environment 2011, 2012). Note that the NIR 2011 used the CMS model (Tier 2 model) described in this section, while the NIR 2012 returned to a Tier 1 methodology because an in-country Expert Review Team organised by the UNFCCC recommended increased sampling in under-represented land-use classes (especially wetland, croplands and post-1989 forests) to reduce uncertainty and enable any statistically significant changes to be detected (UNFCCC 2011).

TABLE 2 Estimated soil carbon density in tC ha⁻¹ in New Zealand

	Ecosystem type	Scott et al (1997)	(MfE, 2011)	(MfE, 2012) (IPCC defaults)
Natural ecosystems	Natural forest	144-176	92.04 ± 3.66	92.59
	Natural scrub	133-166	-	-
	Wetlands	228	97.35 ± 18.22	92.59
	Tussock grasslands	144-177	-	-
Managed ecosystems	Planted forest (pre-1990, post-1989)	163	88.96 ± 5.45	92.59
	Annual cropland	145	90.99 ± 4.38	59.82
	Perennial cropland	137	101.24 ± 11.83	97.76
	High-producing grassland	147	104.99 ± 3.08	117.16
	Low-producing grassland	151	105.8 ± 4.15	105.55
	Grassland with woody biomass	-	98.42 ± 3.59	92.59
	Settlements	-	105.8 ± 4.15	64.81
Other land	-	64.94 ± 20.63	92.59	

Biomass carbon

New Zealand has seen major clearance of native vegetation, particularly since European settlement. The indigenous forest cover has reduced by 70% from its original cover before human settlement. In comparison, half of the world's forest has been lost in the last 5,000 years, with 5.2 million ha lost in the past ten years (FAO 2012). Deforestation has consequences on carbon with total anthropogenic vegetation carbon loss estimated at 3.4Gt C (Scott et al. 2001).

Tate et al. (1997) compiled the first inventory of biomass carbon stocks in New Zealand. Using a national vegetation map (Newsome 1987) and plant biomass from a review of the literature, they found that more than 80% of carbon in vegetation occurred in indigenous forest ecosystems. Subsequently, Carswell et al. (2008) estimated current total carbon stocks in conservation land and potential carbon stocks based on predictions of how much land could potentially be covered by indigenous vegetation (Leathwick 2001); they then refined the study using additional plot data presented in Mason et al. (2012). Non-forest biomass

carbon was estimated using the values of Tate et al. (1997), while shrubland and forest carbon was estimated from 1243 plots comprising a subset of the national Land Use and Carbon Analysis System (LUCAS) dataset (Payton et al. 2004). Carswell et al. (2008) then used Generalised Regression and Spatial Prediction (GRASP), as outlined in Mason et al. (2012), to extrapolate these data over the entire country to give a total current carbon surface for any land described as either “indigenous forest” or “shrubland” within the Land Cover Database 1996–97 (LCDB1, Ministry for the Environment 2009).

We intersected this layer, which excludes soil carbon, with a basic ecosystem layer (Dymond et al. 2012), itself a combination of Land Cover Database 3 and EcoSat Forests (Shepherd et al. 2002). Average carbon stock per hectare and total biomass carbon stocks were then summarised by ecosystem type (both natural and managed) (Table 3). Nearly 1400 MtC is stored in New

Zealand’s above-ground biomass carbon of indigenous forest and scrub (Table 3), representing 80% of the national vegetation C estimates, and within this indigenous forest and scrub, beech forests have a pivotal role as a biomass carbon stock (especially in the South Island) (Figure 4). Most of these beech forests are managed by the Department of Conservation, and are therefore currently protected.

GREENHOUSE GAS FLUXES

Carbon

Energy, industry and waste — One of the largest sources of greenhouse gas emissions from human activities in New Zealand is the burning of fossil fuels for electricity and transportation (43% of the total GHG emission in CO₂-e) (Ministry for the Environment 2012). The largest contribution in the energy sector is from transport (20% of total emissions), which depends

TABLE 3 Estimated biomass carbon stocks for various natural and managed ecosystems in New Zealand

		Area (kha) ⁴	Carbon density (tCha ⁻¹)	Estimated total carbon stocks (Mt C)	% of total biomass
Natural ecosystems	Mānuka/kānuka Shrubland	1,212	51 ⁽²⁾	61	3%
	Subalpine scrub	478	86 ⁽³⁾⁽⁴⁾	41	2%
	Podocarp forest	63	174 ⁽³⁾⁽⁴⁾	11	1%
	Broadleaved forest	349	202 ⁽³⁾⁽⁴⁾	70	4%
	Beech forest	2,134	219 ⁽³⁾⁽⁴⁾	467	27%
	MixedPodocarp-broadleaved Forest	1,336	200 ⁽³⁾⁽⁴⁾	267	15%
	Mixed beech podocarp broadleaved forest	1,826	224 ⁽³⁾⁽⁴⁾	410	23%
	Mixed beech broadleaved	98	218 ⁽³⁾⁽⁴⁾	21	1%
	Unspecified indigenous forest	419	102 ⁽³⁾⁽⁴⁾	43	2%
	Total indigenous forest and scrub			1,392	79%
	Natural freshwater wetlands	193	31 ⁽¹⁾	6	0.3%
	Pakihi	56	20 ⁽¹⁾	1	0.1%
	Tussock grassland	2,583	11-27 ⁽¹⁻²⁾	57	3%
Managed ecosystems	High-producing grassland	8,765	7	61	3%
	Low producing grassland	1,658	3	5	0.3%
	Cropland: annual	334	5 ⁽⁵⁾	2	0.1%
	Cropland: perennial (orchards, vineyards)	102	19 ⁽⁵⁾	2	0.1%
	Exotic forestry	2,036	- Pre-1990: 124 ⁽⁵⁾ - Post-1989: 88 ⁽⁵⁾	231	13%
TOTAL				1,757	100%

(1) Tate et al (1997) ; (2) Carswell et al (2008) ; (3) Mason et al (2012) (4) LCDB3 (Landcare Research Ltd 2012) with Ecosat Forest categories as specified in (Dymond et al. 2012) and wetland categories from Ausseil et al. (2011a) ; (5) Ministry for the Environment (2012)

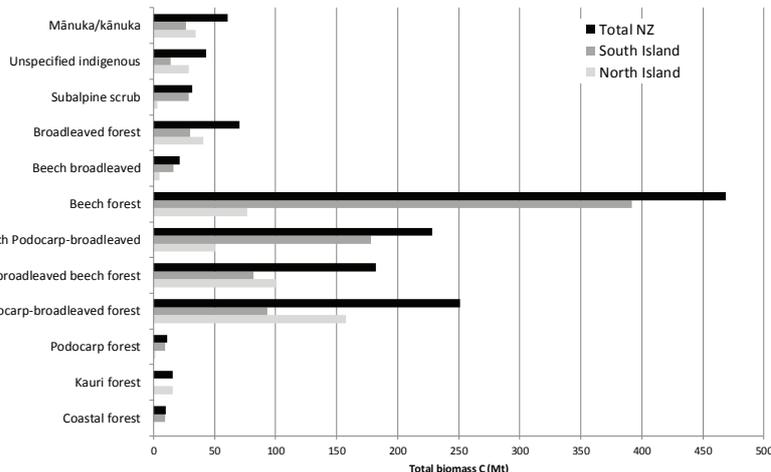


FIGURE 4 Estimated total biomass carbon stocks in indigenous forests (North and South Island).

almost entirely on fossil fuels. Within the OECD, New Zealand has one of the lowest proportions of CO₂-e emissions from power generation (7.5% of total GHG emission, 15% of the energy sector) (IEA 2012), with over 70% of electricity generated from renewable sources in recent years (MBIE 2013). The three remaining IPCC categories contribute only marginally to total emissions: industrial processes (6.7%), solvents (0.04%), and solid waste (2.8%).

Exotic Forestry — Exotic forests in New Zealand sequester carbon through growth of trees for timber and paper production. Exotic forestry occupies about 2 million ha (Table 3) (Landcare Research 2012) with *Pinus radiata*

comprising nearly 90% of the exotic tree production (Ministry for Primary Industries 2012). About one-third of these forests have been established on pasture land since 31 December 1989 and are thus defined as Kyoto forests. Forestry and logging export represent about 1.3% of GDP, with an increase in export revenue of 11% in the year to 2009 (Treasury 2012).

In 2010, the exotic forestry sector sequestered about -23.5 Mt of $\text{CO}_2\text{-e}$ (Ministry for the Environment 2012). However, harvesting trees releases most of the sequestered carbon back to the atmosphere, on a timescale that depends on the end-product of the wood. Therefore, annual sequestration depends on afforestation, deforestation, harvesting, and growth, which are all driven by a complex set of factors including market conditions (Nebel and Drysdale 2010).

Erosion-induced carbon sink — Dymond (2010) estimated the export of soil organic carbon through erosion from New Zealand to the ocean as 4.8 Mt yr^{-1} ($-1.2/+2.4$). Despite this large export, most of this carbon is replaced when regenerating soils sequester CO_2 , although the replacement may take much longer than the loss because regeneration of soils is very slow. In the South Island, all 2.9 Mt C exported to the sea per year is from natural erosion and is expected to be replaced by sequestration, because there have been no major perturbations of the climate or vegetation in the last 5000 years and the landscape will be in approximate equilibrium. However, in the North Island much of the erosion is anthropogenic, and of the 1.9 Mt exported to the sea, only 1.25 Mt is replaced by sequestration of CO_2 ; therefore, carbon is currently lost from North Island soils at a rate of 0.65 Mt C each year, most from the gully and earthflow terrains (Dymond 2010).

Subtracting the soil carbon not buried on the ocean floor ($\approx 20\%$) from the carbon sequestered from the atmosphere by regenerating soils gives a net carbon sink of 3.1 Mt yr^{-1} for New Zealand (due to erosion). Assuming uncertainties of $+50\%$ and -25% for the sequestration, and $+100\%$ and -50% for the release of carbon from the ocean, then the uncertainty of the net sink lies somewhere between -2.0 and $+2.5 \text{ Mt}$. More work is required to confirm these figures because they come from just one study. For example, a recently published paper by Rosser and Ross (2011) suggests that carbon recovery in eroded soils is only 80% of that assumed by Dymond (2012), which would revise the estimated net carbon sink down from 3.1 to 2.7 Mt/yr .

While soil erosion from managed landscapes has considerable negative environmental impacts in New Zealand (Eyles 1983), erosion from unmanaged landscapes, particularly in the South Island, is currently helping to reduce global warming. This natural ‘background erosion’ occurs at a rate at which lost soil can be naturally replaced; it acts like a conveyor belt, taking carbon from the atmosphere and transferring it via soils to the sea floor, where most remains sequestered. In contrast, erosion on managed land can be substantially faster than natural soil regeneration, and in extreme cases whole landscapes can collapse; for example, in some gully terrains, particularly in the North Island, whole hillsides have collapsed. If afforestation for soil conservation purposes was targeted on the gully and earthflow erosion terrains alone, then after the canopy of the trees had closed, the net sink of 0.85 Mt could be increased to approximately 1.35 Mt per year.

Natural forest and shrublands — It is generally assumed that in a ‘mature’ natural forest, carbon uptake via photosynthesis for growth roughly matches losses via respiration within live and dead tissues (Field et al. 1998). However, closer analysis shows global forests sequester more carbon than they lose (Pan

et al. 2011), and this has prompted much speculation as to the causes, such as potential CO_2 and nitrogen fertilisation occurring as by-products of a highly industrialised global economy. New Zealand forests also appear to be net sinks of carbon (Beets et al. 2009)¹. Mason et al. (2012) suggest this is because these forests are still continuing to recover from the widespread disturbance caused by recently ceased logging and mining activities in old-growth forests. Much debate also continues as to whether introduced mammalian herbivores reduce natural forest carbon stocks, and therefore whether controlling these herbivores could increase carbon stocks (Holdaway et al. 2012).

Grasslands began declining in the early to mid-1980s after farming subsidies were removed (MacLeod and Moller 2006). Abandoned agricultural land is usually colonised by shrubland consisting of mānuka (*Leptospermum scoparium*) and or kānuka (*Kunzea ericoides*). These shrubland species have been recognised as an important carbon sink (Trotter et al. 2005). Mānuka/kānuka shrubland is estimated to cover 1.2 Mha (Landcare Research 2012), although this estimate has a large uncertainty because ground-truthing suggests difficulty in the distinction between narrow-leaved shrub types such as gorse, broom and tauhinu. We do not have reliable age distribution information on the full area covered by mānuka/kānuka. The most complete database for mānuka/kānuka stands comprises 40 stands from 2 to 96 years old, with 90% of the distribution below 50 years old (Payton et al. 2010). However, this distribution may not have been randomly sampled and thus may not represent the true age distribution.

Shrubland growth rates depend on environmental factors, so they vary across the country. The growth of mānuka/kānuka throughout New Zealand has been modelled by Kirschbaum et al. (2012a) using a process-based model (CenW, Kirschbaum 1999) calibrated with data from Payton et al. (2010). They predicted growth rates and carbon-sequestration rates of mānuka/kānuka over a 0.05 degree resolution using climate data from the National Institute of Water and Atmosphere (Tait et al. 2006) (Figure 5).

Methane

Methane (CH_4) is a potent greenhouse gas, with a global warming potential (GWP) at least 25 times that of CO_2 over a 100-year period (IPCC 2007a; Shindell et al. 2009)².

Enteric fermentation and manure management — Most methane in New Zealand comes from the farming of ruminant livestock. Ruminants digest cellulose through the action of microbes in the rumen (enteric fermentation), which generates CH_4 . In addition, when manure from livestock decomposes it may emit CH_4 , depending on how the manure is managed. Compared with other countries, New Zealand has a higher proportion of agricultural emissions coming from enteric fermentation than manure management because nearly all animals are grazed outside instead of being housed indoors. The amount of methane released depends on the type, age and weight of the animal; the quality and the quantity of feed consumed; and the energy expenditure of the animal. In New Zealand, methane produced by enteric fermentation is dominated by four animal categories: dairy cattle, beef cattle, sheep, and deer. In 2010, New Zealand had about 6 million dairy cattle, 4 million beef cattle, 32 million sheep, and 1 million deer; enteric fermentation from these represented 32% of New Zealand’s total greenhouse gas emissions and 69% of the agricultural emissions. Although sheep outnumbered cattle, cattle were the dominant contributor to methane from enteric fermentation (64% of methane enteric fermentation).

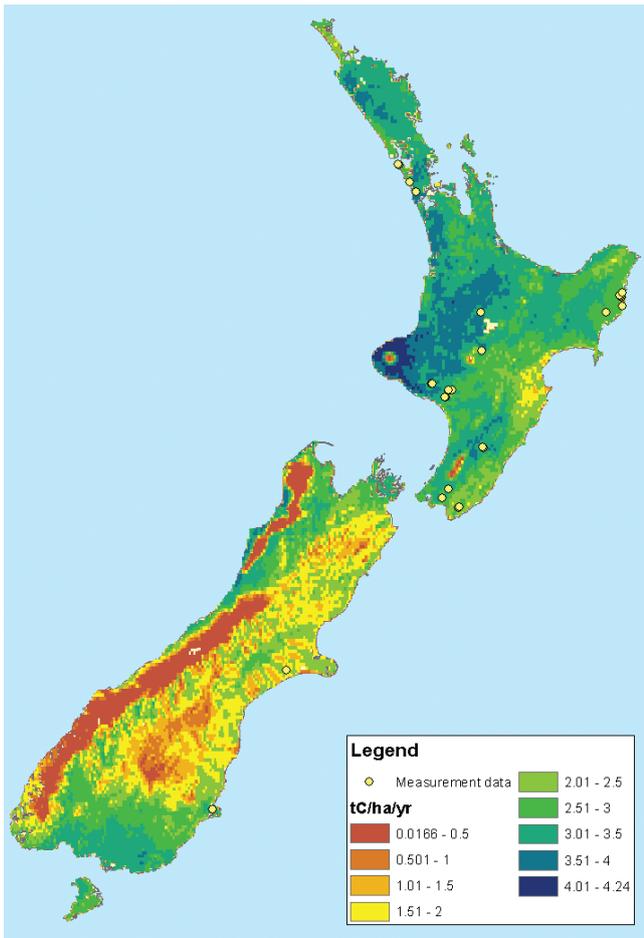


FIGURE 5 Maps of potential carbon sequestration rates for mānuka/kānuka.

Manure management (i.e. systems where manure is managed) also contributes to methane emissions. Dairy farms have effluent storage ponds which produce methane, producing an estimated 0.98 g CH₄ per kg dry weight. At the national level, management of animal waste could contribute between 5 and 15% of total methane emissions (Ministry for the Environment 2012). Recent examination of all sources of waste CH₄ emissions suggests New Zealand's current inventory methodology underestimates CH₄ emission from anaerobic ponds across New Zealand by 264–603 Gg CO₂-e annually (Chung et al. 2013).

Soil methane fluxes — Water-logged soils (anaerobic conditions) emit methane; with organic soils emitting more methane than mineral soils (Levy et al. 2012). However, the area of organic soils under managed ecosystems is very small in New Zealand (Dresser et al. 2011), so emissions are small at national scale³.

Natural freshwater wetlands could also be a source of CH₄ (Roulet 2000), but only about 250 000 ha of natural freshwater wetlands — about 1% of New Zealand's total land area — remain in New Zealand (Ausseil et al. 2011a). In addition, natural wetlands are still being drained, so CH₄ emissions are decreasing. Consequently, the current impact of CH₄ release from natural wetlands is probably small.

Soils can also reduce methane emissions through the action of methanotrophs, a group of soil bacteria that oxidise methane to use it as a source of energy (Saggar et al. 2008). Most soils, including agricultural soils, host methanotrophs. In New Zealand, high rates of methane oxidation occur in soils with intermediate moisture levels, varying with land use (Table 4). Some beech forest soils have some of the highest rates of CH₄ consumption in the world (Price et al. 2004), with the rate mainly influenced by soil water content.

TABLE 4 Soil methane oxidation estimates for various land uses (Saggar et al. 2008)

Ecosystem		Annual consumption (kg CH ₄ ha ⁻¹ yr ⁻¹)
Managed	Dairy pasture	0.50 – 0.6
	Sheep pasture	0.60 – 1
	Unimproved pasture	0.56
Natural	Pine forest	4.20 – 6.4
	Arable crops	1.5
	Native beech	10.5
	Kunzea shrubland	2.3 – 5.1

Nitrous oxide

Nitrous oxide (N₂O) is a potent greenhouse gas because of its strong radiation absorption potential and long atmospheric lifetime. Its global warming potential is estimated at 298 times that of CO₂ over a 100-year period (IPCC 2007a)⁴. It is produced naturally in soils through the microbial processes of denitrification and nitrification (Saggar et al. 2008). In New Zealand, the main source of N₂O emissions is from agricultural soils. New Zealand has adopted a Tier 2 model for estimating N₂O emissions (Figure 6). Various agricultural practices and activities influence the amount of N₂O emitted, including the use of synthetic and organic fertilisers, production of nitrogen-fixing crops, cultivation of high organic content soils, and the application of livestock manure to croplands and pasture. All these practices directly add additional nitrogen to soils, where it can be converted to N₂O. Indirect additions of nitrogen to soils, including atmospheric deposition of volatilised ammonia, can also result in N₂O emissions.

Nitrous oxide emissions in New Zealand arise from three major sources (Figure 6):

- Direct N₂O emission from animal production (pasture, range and paddock animal waste management system; 57% of the nitrous oxide agricultural emissions). This is the result of nitrogen added from animal excreta on pasture soils. The inventory estimates this category using livestock numbers multiplied by nitrogen excretion rates and country-specific emission factors (EF₃PR&P), with urine and dung estimated separately. Nitrogen excretion rates for dairy, beef, sheep, and deer are jointly estimated using the Tier 2 model so that energy requirements, dry matter intake and excreta are all

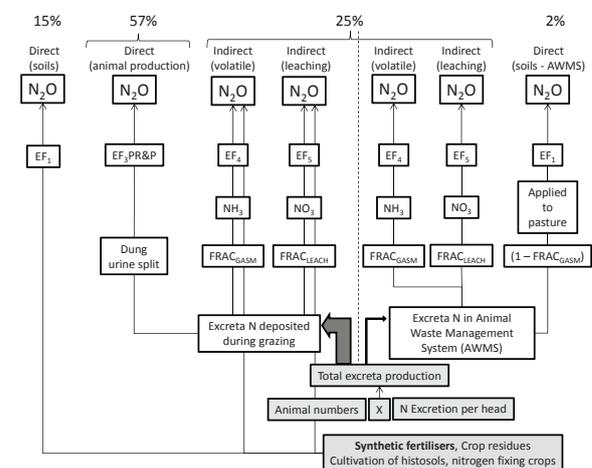


FIGURE 6 Flow chart of the current IPCC national N₂O inventory methodology for pastoral agriculture (adapted from Pickering 2011) with contribution to the total agricultural nitrous oxide emissions.

estimated in a consistent system for each animal type. They vary year to year with livestock productivity for dairy, beef, sheep and deer. Tier 1 animals have fixed international or NZ specific factors. Dairy-grazed pastures produce the highest emissions (10–12 kg N ha⁻¹yr⁻¹) and cow urine is the main source.

- Direct N₂O soil emissions from addition of synthetic fertiliser and spreading animal waste as fertiliser, fixing of nitrogen in soils by crops, and decomposition of crop residues. This direct N₂O soil emission accounts for 17% of the total emissions from the N₂O agricultural emissions, with a dominant contribution from synthetic nitrogen fertilisers.
- Indirect N₂O emissions through leaching and volatilisation of nitrogen from excreta, representing about 25% of the N₂O emissions⁵.

GREENHOUSE GAS EMISSION TRENDS

Historical trends

New Zealand's emissions of anthropogenic greenhouse gases have grown steadily since the mid-19th century (Figure 7). In 1865, when the national population was about 200 000, total emissions from energy and agricultural sources were about 2 Mt CO₂-e yr⁻¹. Agricultural emissions dominated for most of New Zealand's history following European settlement, remaining more than twice the level of energy emissions until the early 1980s. Growth in energy emissions accelerated from the 1960s as more people owned cars (from one car per ten people after WWII to more than one for every two people today) and domestic oil and gas reserves were exploited. From about 1980, the profile for agricultural emissions changed as sheep numbers declined, dairy farming increased, and the use of nitrogen-based fertilisers increased (this fertiliser use is represented in the 'other agricultural' category in Figure 7).

Recent trends 1990–2010

The New Zealand greenhouse gas inventory reports changes from 1990, with annual updates (Figure 8). The most recent report shows an increase of 11.9 Mt CO₂-e yr⁻¹ (20%) for all human-

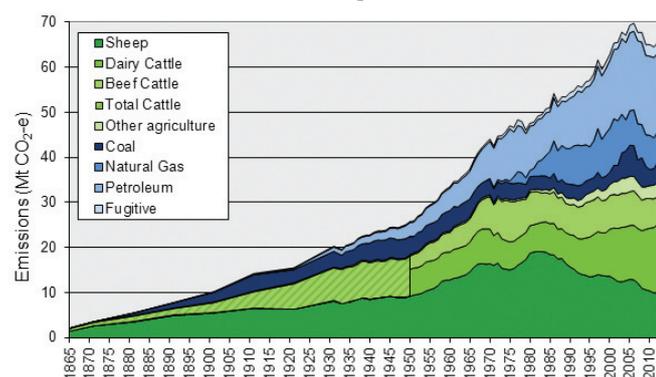


FIGURE 7 New Zealand's agricultural and energy-related anthropogenic greenhouse gas emissions, 1865–2012 (source: own calculations⁷).

induced greenhouse gas emissions since 1990. This increase is mainly due to energy emissions, largely from increased transport. Nevertheless, agricultural emissions remain the largest contributor, and these emissions also rose, mainly due to increases in the number of dairy cattle and use of nitrogen fertiliser (Ministry for the Environment 2012). However, while total emissions rose during 1990–2010, they peaked in 2005 and the downward trend since then is attributed to a weaker economy, which affected both the agricultural and energy sector. Some reasons include

reduction in coal-fired electricity generation, reduction in the numbers of sheep, non-dairy cattle and deer because of droughts (summer 2006 and 2007), and reduction in road transport due to the economic downturn.

The variation in LULUCF emissions is mainly a consequence of harvesting cycles and land-use changes. Many new forests were planted between 1992 and 1998 because of changes in tax regimes, but the rate of new planting declined rapidly after 1998 (just 1900 ha in 2008). Then, between 2008 and 2010, planting again increased in response to the introduction of the NZ ETS, afforestation grants scheme and Permanent Forest Sink Initiative (Ministry for the Environment 2012). The decrease in removal between 2009 and 2010 is mainly due to the increase in harvesting of pre-1990 planted forests and increased new planting (resulting in loss of soil carbon due to conversion from pasture).

Agricultural greenhouse gas emission trends per region

The trend in agricultural greenhouse gases at a regional level can be calculated by multiplying the implied emission factor for each type of animal for each year since 1990 by the animal population in each region (Figure 9). Waikato contributes the most to agricultural greenhouse gases, with its total contribution continuing to increase as dairying continues to intensify. However, the Canterbury region has also steadily increased in greenhouse gas emissions; this is also attributable largely to growth in dairying.

Relative contribution from managed and natural ecosystems

The annual contribution to greenhouse gas fluxes is summarised in Figure 10. Contributions are summarised per sector, to reflect the anthropogenic activities reported in the national greenhouse gas inventory.

Globally, the energy/industry sector emits more greenhouse gases than any other sector; in contrast, New Zealand is distinct from other OECD countries because nearly 50% of its total greenhouse gases come from the agricultural sector. The Land Use, Land-Use Change and Forestry sector (LULUCF) is a sink for carbon, removing 20 Mt CO₂-e yr⁻¹. While the energy/industry, agricultural, and LULUCF sectors are reported in the New Zealand greenhouse gas inventory, other ecosystems and processes are currently not included. These are carbon sequestration from mānuka, methane oxidation from soils, and erosion-induced carbon sequestration; their estimated contributions are shown in Figure 10.

Combining potential carbon sequestration rates (Figure 5) with the area of mānuka/kānuka shrubland from LCDB3 (and removing the post-1989 forest to avoid double accounting with LULUCF) shows that shrubland could sequester about 11 Mt CO₂-e yr⁻¹. This is similar to the erosion-induced carbon sink of around 11 Mt CO₂-e yr⁻¹ (+ 9.1, -7.3); in contrast, the contribution from soil CH₄ oxidation is small, with only 2 Mt CO₂-e yr⁻¹.

Spatial distribution of greenhouse gas fluxes in New Zealand

To map the annual fluxes of greenhouse gases in New Zealand, we adopted a habitat approach and assigned fluxes to major land uses and land covers. Because the agricultural and forest sectors occupy the largest areas in New Zealand, we focused the mapping on these land uses. Shrublands were also included because they are natural ecosystems contributing to carbon sequestration, and soil CH₄ oxidation rates per land use were incorporated using the information in Table 4. Of the categories of ecosystems and processes described above, the energy and industry sector and

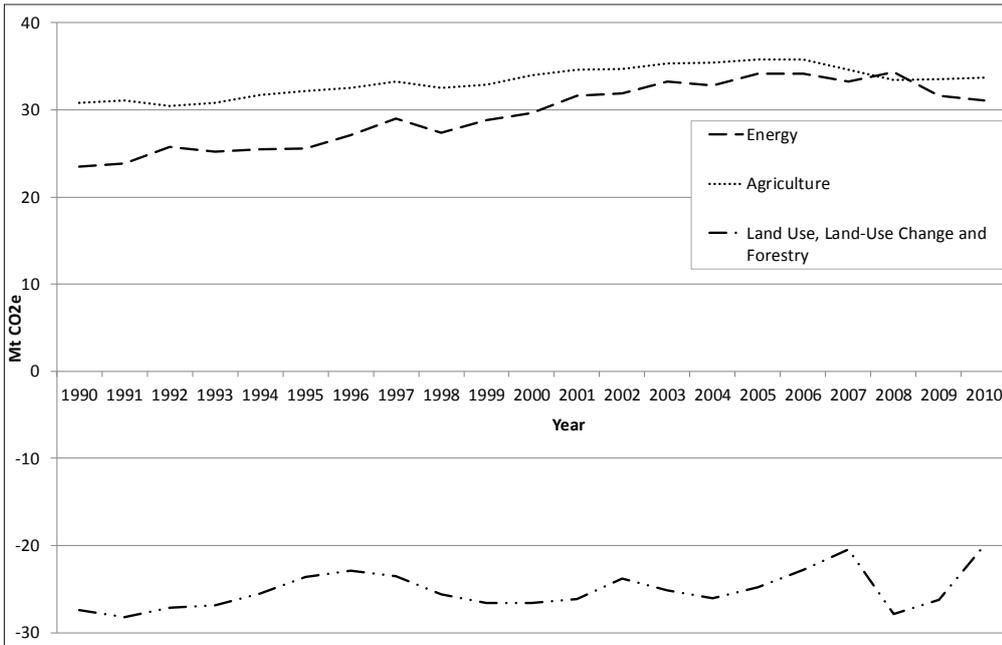


FIGURE 8 Changes in greenhouse gas emissions between 1990 and 2010 (Ministry for the Environment 2012).

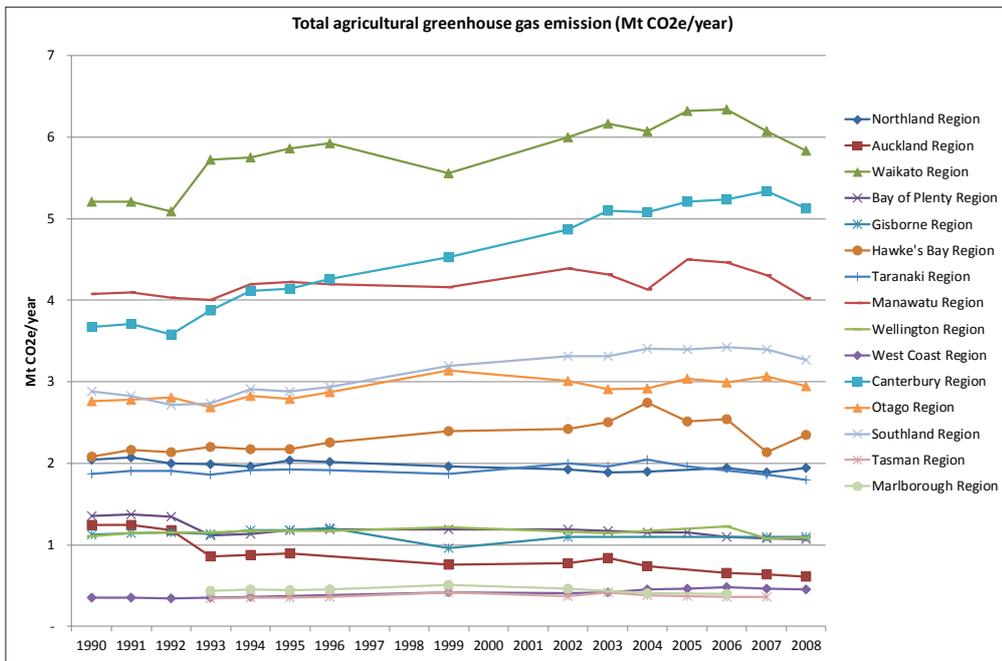


FIGURE 9 Agricultural greenhouse gas emission trends per region (Source: National Greenhouse gas inventories between 1990 and 2010, Ministry for the Environment).

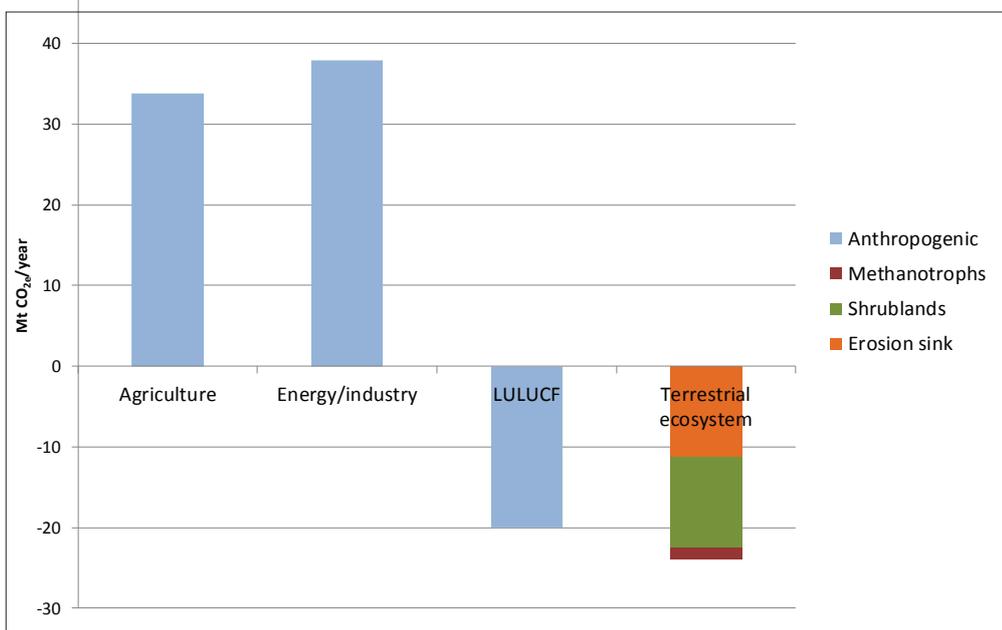


FIGURE 10 Contribution of managed and natural ecosystems and processes to greenhouse gas fluxes (2010).

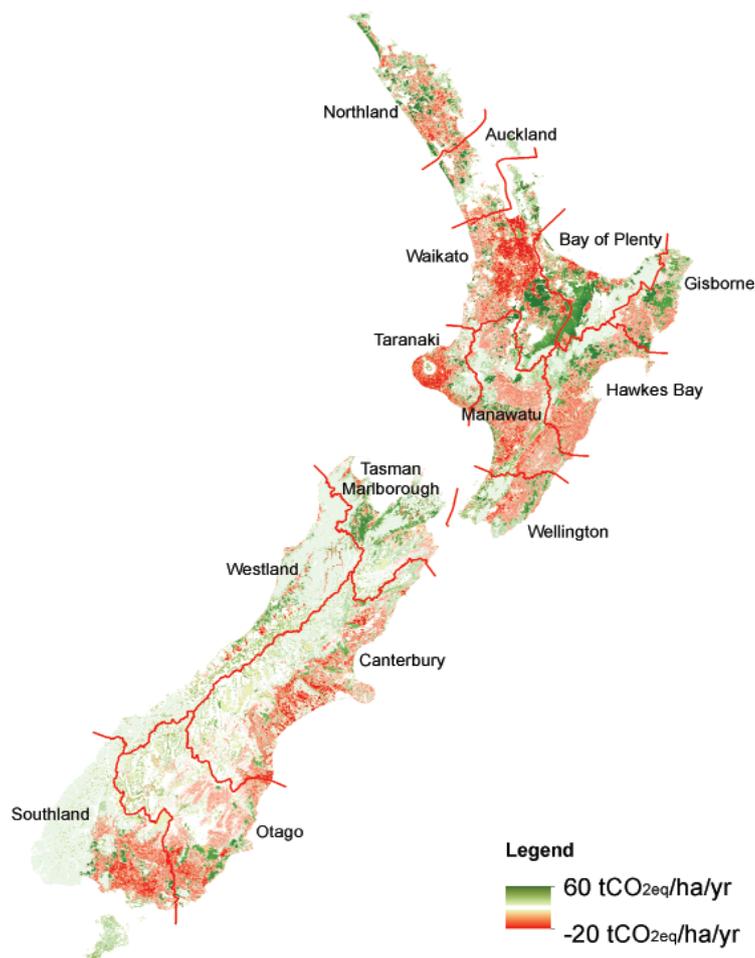


FIGURE 11 Distribution of sources and sinks of greenhouse gases in $\text{tCO}_2\text{eq/ha/yr}$ (2010).

erosion-induced carbon sequestration were not mapped, because these were not spatially explicit.

To generate the spatial distribution of agricultural greenhouse gas emissions, we used a proxy for animal carrying capacity that estimates the stocking rates across the landscape, scaled to animal numbers at the district level (Ausseil et al. 2013). Animal distribution maps were created for the four major species found in New Zealand – dairy cows, beef cattle, sheep and deer – and these were then multiplied by implied emission factors derived from the New Zealand greenhouse gas inventory.

Figure 11 shows the spatial distribution of greenhouse gas fluxes in New Zealand. Sources are spread more widely than sinks, reflecting the larger proportion of land under pastoral systems. The intensity of source is particularly high in the Taranaki and Waikato regions because of their high concentrations of dairy farming. In contrast, sinks of greenhouse gases cover much smaller areas, mostly in the Bay of Plenty and Tasman regions where most of the forestry sector is located. Greenhouse gases are also sequestered, albeit at lower levels (light green colours), in shrubland areas across New Zealand.

BIOPHYSICAL REGULATION: SURFACE ALBEDO

Loosely speaking, radiative forcing is the difference between solar radiation reaching the planet and the amount the planet reflects (albedo), so changes in radiative forcing have important implications for the Earth's overall radiation balance (Brovkin et al. 1999; Betts 2000). Vegetation types that store more carbon, like forests, tend to absorb more radiation than vegetation types

that store less carbon, like crops or pasture. Therefore, carbon storage and direct energy absorption typically change in opposite directions for different vegetation types (Kirschbaum et al. 2013).

Values of albedo (for short-wave radiation) for coniferous forests lie in the range 8–15%; values for pastures, which are more reflective, are usually 5–10% higher (Breuer et al. 2003). For example, Kirschbaum et al. (2011) reported albedos of about 20% for pastures and 12–13% for coniferous forests, with albedo diminishing over the first 10 years of forest growth. Slow-growing boreal forests may take several decades before forest canopies reach representative forest albedo values (Bright et al. 2012).

The importance of radiative changes is also directly proportional to the magnitude of incoming solar radiation, which can vary more than two-fold across the globe. Across the globe, mean daily radiation, received at ground level and averaged over a whole year, ranges from about $10 \text{ MJ m}^{-2} \text{ d}^{-1}$ in polar regions, to about $20 \text{ MJ m}^{-2} \text{ d}^{-1}$ near the equator (Stanhill and Cohen 2001). Near the equator there is little seasonality, but further towards the poles an increasingly distinct seasonal cycle ranges from complete darkness in winter to daily radiation receipt in summer similar to that in equatorial regions.

Forster et al. (2007) summarised worldwide studies that estimated surface albedo changes with agricultural expansion since pre-industrial times. They concluded that land-use change probably caused radiative forcing of $-0.2 \pm 0.2 \text{ W m}^{-2}$, leading to global cooling by about $-0.1 \text{ }^\circ\text{C}$. This broad global pattern probably varies significantly across regions, with radiative forcing ranging between 0 and -5 W m^{-2} , depending on changes in land use (Forster et al. 2007).

Betts (2000), and later Bala et al. (2006; 2007), also showed that the benefit of tree plantings could be much diminished or even become negative, depending on the extent of albedo changes, incident radiation and carbon-storage potential at different locations. This is particularly important for sites with extended snow cover, which can greatly increase albedo differences, and for sites where trees grow poorly, thus reducing the carbon storage benefit (Betts 2000).

In more temperate regions like New Zealand, snow cover is less important, and albedo effects resulting from vegetation shifts are therefore likely to be less important than in boreal regions with extended snow cover. Detailed measurements from New Zealand showed that for young forest stands carbon storage and radiation absorption had effects of comparable magnitude (Kirschbaum et al. 2011), but for stands storing more than 25 t C ha^{-1} , the carbon-storage effect became dominant (Figure 12). Over the whole stand rotation of a conifer forest, albedo changes negated the benefit from carbon storage by about 20% (Kirschbaum et al. 2011).

Albedo change is increasingly being recognised as an important influence on climate change. Consequently, recent studies have explicitly included it in analyses of the net climate change consequences of land-use change (Jackson et al. 2008; Schwaiger and Bird 2010; West et al. 2010; Anderson-Teixeira et al. 2012; Bright et al. 2012; Kirschbaum et al. 2013). This is warranted because albedo changes cause radiative forcing in much the same way as GHGs; moreover, changes in albedo occur more or less instantaneously with changes in land cover, and they can be readily reversed, with similarly rapid consequences.

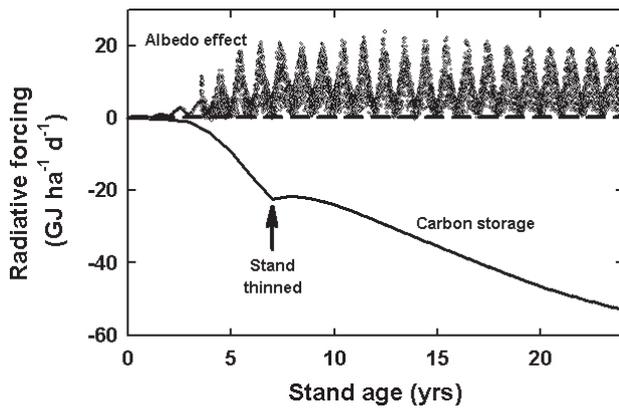


FIGURE 12 Radiative forcing from albedo and carbon storage after planting a forest stand onto former pasture. The stand was thinned in 1980, which accounts for the discontinuity in the slope of the lines depicting carbon-based radiative forcing. Redrawn from Kirschbaum et al. (2011).

However, the discussion above only addresses the direct albedo-based difference in vegetation coverage, but land-cover types also differ in their indirect effects on surface radiation balance. These indirect effects act primarily through changes in evaporation rates. Forests usually have higher evapotranspiration rates than pastures, mainly because forest canopies intercept more rain, resulting in increased total evapotranspiration, while in drier regions, the more extensive and deeper root systems of trees may also access water reserves deeper than those accessible to grasslands, which occupy mainly the upper soil layers. The increased amount of water vapour in the atmosphere absorbs some outgoing long-wave radiation and adds to global warming. Conversely, this extra water transferred to the atmosphere must eventually be returned to the surface as precipitation, which involves cloud formation and because low clouds tend to have high albedo, they reduce the short-wave flux to the surface and thus cool the Earth.

These indirect effects are difficult to compute but some studies have shown that some of these secondary effects can be of comparable magnitude to the direct radiative effects (Boisier et al. 2012). However, most act in the opposite direction to the direct radiative effects, and their importance increases from boreal to tropical regions (Bala et al. 2007).

Albedo changes also continue to act indefinitely, in contrast to the effects of GHGs, which are eventually reduced through the gases' natural breakdown or transfer to the deep oceans. Thus, the radiative forcing attributable to albedo differences persists for as long as the different land covers are maintained. Clearly then, albedo changes offer a mitigation option with two important advantages: the effects are immediate, and they are persistent.

DISCUSSION

Threats and opportunities for natural ecosystems

Natural ecosystems, especially indigenous forests, are a significant net carbon sink in New Zealand. However, their capacity to store carbon, and thus the value of the service they give to human society, depends on their condition and trends (Table 5). Globally, forest management continues to play a pivotal role in net removals of greenhouse gases, with forests continuing to act as net sinks of CO₂ in spite of the continued emission of CO₂ from tropical deforestation (Pan et al. 2011). Within New Zealand, indigenous forests may be at steady state for carbon, but the biggest threats for carbon losses are through natural disturbances. These range from infrequent catastrophic events, such as earthquakes

TABLE 5 Conditions and trends in natural ecosystems

Ecosystem	Condition	Trend
Native forest	Subject to natural disturbances with variable recovery rates (Carswell et al. 2008)	Some loss of extent (Ausseil et al. 2011c) with consequences on carbon stocks loss
Native shrubland	Regenerating shrublands – some succession to tall forest	Marginal grassland reverting to shrubland, but long-term establishment is highly dependent on commodity prices
Freshwater wetlands	More than 50% of remaining wetlands are in poor condition (Ausseil et al. 2011a)	Continued loss due to drainage (Ausseil et al. 2011c)
Tussock grasslands	Exotic conifer and <i>Hieracium</i> invasions	Loss of extent due to conversion to agriculture and forestry (Weeks et al. 2012)

and volcanism, to frequent and less catastrophic events such as windthrows (Carswell et al. 2008). Their impact on carbon depends on the rate of vegetation recovery after disturbance and the rate at which the wood decays. Other climate changes (CO₂ fertilisation, global warming, increased precipitation in some places, nitrogen deposition) are likely to have short-term beneficial effects on carbon storage. Droughts in eastern areas of New Zealand, however, would decrease the productivity and rates of carbon storage in the medium term. Legacies of repeated burning and grazing can also constrain the potential forest composition and consequent carbon storage.

Wetlands can be a source of CH₄ and a sink of CO₂. However, when they are drained the water table drops and they can then oxidise and lose a large amount of carbon. Drainage historically occurred during conversion to agriculture and is still common practice in parts of the country. A study in the Waikato showed that a wetland converted into pasture lost 3.7 t C⁻¹ ha⁻¹ yr⁻¹ in the first 40 years (Schipper and McLeod 2002), slowing to about 1 t C⁻¹ ha⁻¹ yr⁻¹ more recently (Nieveen et al. 2005). More research is needed to better understand how the draining of wetland soils affects methane and carbon emissions.

Tussock grasslands are mainly located in the South Island. Conditions in most tussock areas are degrading because of the effects of burning (which reduces biomass and soil carbon), and invasion by the weed *Hieracium*, which significantly reduces biomass C but adds a small amount (~1%) to soil C (Kirschbaum et al. 2009). During the last 20 years, 50 000 ha of tussock grasslands have been converted for agriculture (Weeks et al. 2012), but the consequences of these conversions on carbon have yet to be assessed.

Carbon storage could be increased through afforestation of non-forest lands on conservation land (Carswell et al. 2008). In a follow-up to work by Carswell et al. (2008), Mason et al. (2012) used potential vegetation cover to estimate the level of potential carbon stocks on conservation land. These stocks could contain 461 Mt more than at present, mainly through an increase in the areas of lowland podocarp–broadleaf forest. The time required for the existing seral vegetation to complete its succession ranges from a few decades to as long as 300 years. However, many questions remain as to whether conditions on these lands will still support forest successions documented by earlier authors (e.g. contrast McKelvey (1973) with Payton et al. (1989)). Other cost-effective gains could include promoting the succession from existing shrubland to tall trees, controlling browsing animals, and preventing fires (Carswell et al. 2008).

Several studies have investigated the benefit of converting marginal pasture land into indigenous forest through afforestation or natural reversion into shrubland (Trotter et al. 2005; Kirschbaum et al. 2009). This conversion would also provide benefits from increased erosion control, enhanced biodiversity and other ecosystem services (Ausseil and Dymond 2010).

Policy options for managed ecosystems

Because New Zealand is highly forested for an OECD country and can promote widespread land-use change, managing existing forests and encouraging new forests are key tools for managing the national greenhouse gas balance. The New Zealand Government currently provides two incentives for sustainable land management that favour increased forest area on lands unsuitable for agriculture. These are the Permanent Forest Sink Initiative (PFSI) and elements of the Emissions Trading Scheme (ETS) that address forestry. Both schemes are administered by the Ministry for Primary Industries; thus, the role of the Government lies in maintaining the Register of participating forests (and associated tradeable carbon units) and in providing and regulating standards to determine the existence/longevity and number of carbon units accruing to each landholder. Once units have been devolved to landholders the Government plays no role in the marketing or sale of such units. For a landholder to qualify for devolved credits, the forest must meet criteria derived from the Kyoto Protocol according to interpretations of the Protocol specific to New Zealand. These interpretations are a legitimate differentiation between countries.

The ETS and the PFSI differ in that a covenant is required for forests entering the PFSI. This covenant restricts the range of potential future land uses, and while this is seen as a benefit by conscientious purchasers of credits, many landowners view it as a liability; however, if units arising from the PFSI can fetch a premium price, this should compensate for potential future restrictions on land use change. Moreover, the ETS and the PFSI both depend entirely on a strong price for carbon if they are to bring about significant levels of land-use change, but the price of carbon has not been sufficiently high in recent years. Further, the recent decision by New Zealand not to enter into the binding commitments associated with the second commitment period of the Kyoto Protocol introduces further uncertainty about future carbon trading from forests within regulated markets.

Other non-regulatory options encourage reduction of emissions from anthropogenic sources. The New Zealand Energy Strategy encourages energy efficiency and use of renewable energy. It has set a target of 90% of electricity being generated from renewable energy resources by 2025 (Ministry of Economic Development 2011).

Future research directions

Biomass carbon research must now address the potential for management to enhance carbon sequestration during natural regeneration or management of existing forests. This requires several things: adequate incentives; modelling that in turn requires data from large-scale forest manipulation experiments and long-term repeated measurements of natural forest successions; and improved estimates of wood density in indigenous species across broad environmental and ecological ranges. Also lacking are adequate data on carbon sequestration from early successional vegetation, and this should be corrected as soon as possible.

The soil CMS discussed earlier uses a statistical model whose primary output is a series of coefficients. The standard error of

these estimated coefficients determines the overall accuracy of the national soil C estimates, and the uncertainty associated with land-use change between two dates. To some extent, increasing the number of samples for statistical analysis will reduce the standard error of the coefficients. If all samples are independent and the samples are all devoted to a specific land use or soil-climate class, the standard error might be expected to diminish by the square root of the number of samples. Unfortunately, as the number of samples increases, the soil C associated with each sample becomes correlated with other points, so the effective number of degrees of freedom associated with the samples is always less than the true number of samples. Thus, there is a law of diminishing returns when attempting to reduce the standard error of coefficients by additional sampling.

In addition, the present New Zealand CMS uses a small number of environmental predictors (soil, climate, rainfall, land use, topography), but soil C probably depends on the complex interaction of many other factors. Indeed, more complex models for soil C, involving many more environmental variables, significantly reduce the standard error of soil C estimates (McNeill et al. 2012) and therefore improve the precision of estimates of national soil C stocks and stock changes. The principal difficulty is that there are only a limited range of covariate layers representing all the possible factors that might have an associative effect on soil C, and this effectively limits the complexity of the models that can be generated. In short, better models for soil C are likely to depend on improvements in climatic data layers, better models of soil attributes, and more comprehensive information on vegetation types.

The accuracy of CH₄ emissions depends on good data on composition and numbers for animal populations through the year, and data on feed quality. On-going research funded by the Ministry of Primary Industry aims to improve nationwide information on activity data (e.g. animal numbers, liveweights), CH₄ production per animal, and pasture quality. Some attempts have been made to use remote sensing as a tool to predict pasture quality over time (Ausseil et al. 2011b), but uncertainties are still large because many variables influence pasture quality; for example, farm type and pasture types that cannot be remotely detected.

The current method for calculating direct N₂O emissions from agricultural soils in the National Inventory uses a constant emission factor multiplied by the nitrogen inputs. On-going research is improving the emissions factors used for each category defined in the national inventory (Tier 2 methodology). However, N₂O emissions are the result of complex soil microbial processes and properties, while climate and management practices also influence emission levels. Consequently, the ability of the National Inventory method to account for regional differences in N₂O emissions resulting from differences in these factors is limited. An alternative approach uses process-based models (tier 3 approach) that can predict emissions under various environmental and management conditions. For example, the USA has started using a Tier 3 method based on the DayCent model (Parton et al. 1996) to estimate direct N₂O emissions from major crops (US Environmental Protection Agency, 2012). In New Zealand, work is ongoing to test and compare various N-dynamics models, such as the NZ-DeNitrification DeComposition model (NZ-DNDC), and the Agricultural Production Systems sIMulator (APSIM) (Vogeler et al. 2012), using frameworks developed to scale local data to regional and national scales (Giltrap et al. 2013).

Farm management strategies and research needs for reducing CH₄ and N₂O emissions were summarised by O'Hara et al. (2003), but options for reducing agricultural greenhouse gas emissions without affecting outputs are still limited. Nevertheless, New Zealand is a lead player in mitigation research, via the establishment of the New Zealand Agricultural Greenhouse Gas Research Centre (www.nzagrc.org.nz). The NZAGRC is a partnership between the leading New Zealand providers of research on agricultural greenhouse gases and the Pastoral Greenhouse Gas Research Consortium (PGGRc). Four areas of research are promoted: mitigation of enteric methane emissions, mitigation of nitrous oxide emissions, soil carbon research, and integrated low GHG-emitting farm systems.

Few mitigation techniques are available to offset CH₄ emissions from effluent ponds, but research to develop a cost-effective biofiltration technology using methanotrophic bacteria is well advanced (Pratt et al. 2012).

In general, methods used to quantify climate regulation involve biogeochemical regulation (e.g. carbon cycling), although recent research also includes biophysical climate regulation (West et al. 2010; Kirschbaum et al. 2011; Anderson-Teixeira et al. 2012). For example, Anderson-Teixeira and DeLuca (2011) quantitatively valued the climate regulation service of ecosystems based on a combination of the carbon stocks, carbon sequestration, and how long greenhouse gases stayed in the atmosphere. Further recent studies combine biogeochemical effects and the biophysical effects of land use into an index of climate regulation (West et al. 2010; Anderson-Teixeira et al. 2012). This quantification of ecosystem climate services can improve the quality of decisions on climate-related issues. Including economic values for these services would also be helpful.

This chapter reviewed carbon stocks and fluxes (biogeochemical regulation) for New Zealand's main ecosystems, and also reviewed the effects of land-use changes on surface albedo (biophysical regulation). Emissions from managed ecosystems are well accounted for in the national greenhouse gas inventory, with clear pathways for inventory improvements. However, contributions from natural processes and ecosystems still contain many uncertainties, and these warrant further investigation. For example, erosion plays a major role in transferring carbon from the land into the ocean but the amount and timing of this transfer is poorly understood. Similarly, natural ecosystems such as native shrubland might also form significant carbon sinks, especially if seral vegetation can be encouraged on eroded land, but more data are needed to estimate the extent and age distribution of these shrublands.

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ENDNOTES

- 1 In the latest greenhouse gas inventory (Ministry for the Environment 2012) natural forests were considered carbon neutral. However, at the time of writing this chapter, re-measurement of the natural forest permanent sample plot network was underway. Results should enable New Zealand to illustrate whether its natural forests are a net source or sink of carbon or whether the carbon neutral assumption still holds.
- 2 Global warming potentials are still being researched, and the current national inventory still uses values from the second assessment report (IPCC, 1996), in which methane has a GWP of 21.
- 3 In the national greenhouse inventory, only nitrous oxide from organic soils is reported.
- 4 Under the Kyoto Protocol first commitment period, New Zealand is required to use the GWP from the second assessment report, which is 310.
- 5 This emission factor is based on rivers and waterways like the Rhine or Mississippi, so emissions are likely to be overestimated.
- 6 Includes only carbon dioxide (CO₂), methane (CH₄), nitrous oxide (N₂O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphurhexafluoride (SF₆), whose emissions are covered by the UNFCCC. These GHGs are weighted by their 100-year Global Warming Potentials (GWPs), using values consistent with reporting under the UNFCCC.
- 7 Historical emissions have been derived from a range of sources. The 2012 inventory submission to the UNFCCC (MfE, 2012) has been used for emissions in the period 1990–2010. Agricultural emissions before 1990 are derived from historical livestock numbers from the 1996 Agricultural Production Survey release (Statistics New Zealand 1997) and fertiliser use data from NZ Fertiliser Manufacturer's Research Association combined with 1990 implied emission factors. Agricultural emissions in 2011 and 2012 are derived from the Provisional June 2012 Animal Production Statistics (Statistics New Zealand 2012) and 2010 implied emission factors. Energy emissions before 1990 are derived from coal production and trade data from Crown Minerals, per-field gas production data from the 2011 Energy Data File (MED 2011), and petroleum consumption data from Statistics NZ year books (various years). Energy emissions in 2011 and 2012 are derived from the September 2012 edition of the Quarterly Electricity and Liquid Fuel Emissions Data Tables (MBIE 2013) with a one-quarter forecast based on previous December quarters.