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Annual carbon emissions associated with natural disturbance in New Zealand's natural and planted forests

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by Robert J Holdaway, Norman W.H. Mason, Tómas Easdale, John Dymond, Harley Betts, Landcare Research; Steve J. Wakelin, John R. Moore, Scion

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PO Box 2526
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1 Executive Summary

This report quantifies the annual carbon emissions associated with current rates of annual disturbance in New Zealand's pre-1990 and post-1989 forests, within the context of the Durban Agreement. The Durban Agreement includes definitions and a suggested approach to deriving the baseline level of natural disturbance, but the relevant best practice guidance is yet to be finalised. This adds a significant element of methodological uncertainty.

The main natural disturbance agents affecting New Zealand forests are extreme weather events (wind, snow, drought and rain), geological events (earthquakes, volcanic eruptions), pest/pathogen outbreaks, and fire. These disturbance agents act at a range of spatial and temporal scales causing partial dieback/crown damage, death of individual trees or stands, and catastrophic damage to land and forests.

We quantified New Zealand's background level CO_2 emissions from natural disturbance in both natural and planted forest by identifying areas where disturbance has occurred, using a combination of plot-level metadata and historical records (planted forests) or remote sensing (natural forests). Net carbon implications were estimated by modelling stand or tree-level mortality and subsequent growth and wood decay processes, providing a dynamic and explicit treatment of carbon stock change resulting from natural disturbance.

Disturbed pre-1990 natural forest plots were identified on the basis of decreased mean stem diameter and total stand basal area. The mean annual probability of a plot being disturbed was 0.0192, giving an estimated area of pre-1990 forests affected by low-intensity natural disturbances of 150,545 ha yr^{-1}. There was a net loss of carbon of −2.43 Mg CO_2-e ha^{-1} yr^{-1}, (1 Mg = 1 tonne) in disturbed plots compared with a net gain of 3.81 Mg CO_2-e ha^{-1} yr^{-1} in non-disturbed plots. The net emission from the disturbed plots was therefore 6.24 Mg CO_2-e ha^{-1} yr^{-1}, or 939 401 Mg CO_2-e yr^{-1}. The total area of pre-1990 natural forest affected by large-scale landslides during the period 1990–2008 was 4529 ha; 252 ha yr^{-1} on average. This resulted in annual carbon emissions of 81 935 Mg CO_2-e yr^{-1}. Adding the two emission sources together gives total annual emissions of 1 021 336 Mg CO_2-e yr^{-1} from natural disturbance in New Zealand's pre-1990 natural forest. Post-1989 natural forest emissions in 2009 were estimated to be 813 Mg CO_2-e.

For planted forests, the mean annual area affected by wildfires during the period 1990–2009 was estimated to be 412 ha yr^{-1}, causing average annual emissions from fuel combustion and subsequent decay of 45 760 Mg CO_2-e yr^{-1}. The mean annual area affected by severe wind damage in planted forests during the period 1990–2009 was estimated to be 524 ha yr^{-1}, causing average annual emissions of 31 136 Mg CO_2-e yr^{-1}. Emissions from geological events were assumed to be zero. Most of the emissions in planted forests during 1990–2009 are associated with pre-1990 forests due to their greater area and higher average carbon stocks compared with post-1989 forests. Combined emissions from wind and fire in pre-1990 forests were estimated to be 69 435 Mg CO_2-e yr^{-1} compared with 7460 Mg CO_2-e yr^{-1} in post-1989 forests.

We discuss the use of these results to develop baseline natural disturbance estimates for Forest Management Reference Level (FMRL) and Afforestation/Reforestation reporting. We recommend that the FMRL and natural disturbance baselines are developed simultaneously using the same data and consistent methodology. This is required to avoid the expectation of net credits or debits. We also recommend that the IPCC good practice guidance document is closely monitored for ongoing developments in accounting for natural disturbance.

Contents

Page

2 Introduction

As part of the Durban Agreement[1] New Zealand has the option to submit, in 2015, a background level of emissions from natural disturbance for pre-1990 and post-1989 forests. A robust estimate of background carbon losses to natural disturbance provides a baseline by which future disturbance events can be measured, and insures New Zealand against carbon liabilities arising from large-scale events such as ex-tropical cyclones, volcanic eruptions and earthquakes. The Durban Agreement includes definitions and conditions and a suggested approach to deriving the baseline level of natural disturbance. Best practice guidance for applying the Durban Agreement rules to greenhouse gas inventory reporting and accounting is currently being prepared by the IPCC. Until that report has been completed and approved, the minimum standards for estimating the background level of natural disturbance are unclear.

In the context of the Durban Agreement, natural disturbance is defined as "non-anthropogenic events or non-anthropogenic circumstances … that cause significant emissions in forests and are beyond the control of, and not materially influenced by, a Party"[2]. The main natural disturbance agents affecting New Zealand forests are storms (wind, snow, and rain) (Harcombe et al. 1998; Sinclair 2002; Martin & Ogden 2006; Moore et al. 2012), earthquakes (Allen et al. 1999; Hancox et al. 2002), volcanic eruptions (Wilmshurst & McGlone 1996; Neild et al. 1998), drought (Hosking & Hutcheson 1988), pest/pathogen outbreaks (Ridley et al. 2000) and fire (Anderson et al. 2008). Natural disturbances to forests occur at various spatial and temporal scales, ranging from individual leaf, branch or whole-tree mortality due to disease or wind damage, to large-scale landscape-wide events such as severe wind storms, earthquakes and volcanic eruptions.

The purpose of this report is to quantify the annual carbon emissions associated with current rates of annual disturbance in New Zealand's pre-1990 and post-1989 forests. We achieve this by firstly summarising the policy background associated with the Durban accounting framework. We then identify the major natural disturbance regimes in New Zealand and discuss their likely effects on forest carbon stocks. Within this context, we quantify the areas disturbed and the associated carbon losses in natural and planted forest from systematic assessments with permanent plot data, remotely sensed imagery and historical records. We discuss the results in terms of reference-level predictions of annual carbon emissions due to natural disturbance in New Zealand's forests according to the Durban Agreement requirements. This research provides the information required to explicitly incorporate the effects of natural disturbance into the Forest Management Reference Level and Afforestation/Reforestation reporting. An enhanced understanding of the effects of natural disturbance on forest carbon in New Zealand increases our ability to manage the economic implications and risks associated with future natural disturbance events.

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[1] United Nations Framework Convention on climate Change (UNFCCC) 2012. FCCC/KP/CMP/2011/10/Add.1.

[2] United Nations Framework Convention on Climate Change (UNFCCC) 2012. FCCC/KP/CMP/2011/10/Add.1, Decision 2/CMP.7, Annex, paragraph 1.

3 Research Aims

1. Identify the key natural disturbance agents for New Zealand's forests and characterise the disturbance regime of each agent and its potential to cause future carbon loss.
2. Quantify carbon emissions associated with natural disturbance in natural forests, using a combination of permanent plots and remote sensing.
3. Quantify carbon emissions associated with natural disturbance in planted forests, using a combination of permanent plots and written records.
4. Estimate background-level annual carbon emissions due to natural disturbance in New Zealand's forests as required by the Durban Agreement.

4 Background

4.1 Interpretation of the Durban accounting framework for natural disturbance

The climate change meeting in November 2011 in Durban (UNFCCC COP/MOP17) agreed on a "road map" towards a legal agreement on limiting greenhouse gas emissions that will operate from 2020. It also agreed on the accounting rules that will operate in the second commitment period of the Kyoto Protocol from 1 January 2013. These rules are specified in the annex to decision 2/CMP.7 contained in the document FCCC/KP/CMP/10/2011/Add.1.

There are two key elements in the Durban Agreement relating to the impact of natural disturbances on carbon stock changes and greenhouse gas emissions:

1. Mandatory accounting for forest management under a reference-level approach;
2. Option to exclude from accounting emissions (and subsequent) removals on lands subject to significant natural disturbances.

4.1.1 Forest management reference level

In the first commitment period of the Kyoto Protocol, only emissions and removals from afforestation, reforestation and deforestation since 31 December 1989 were included in accounting. Accounting for emissions and removals in pre-1990 forests was optional and New Zealand elected not to include these forests, which comprise almost all the country's natural forest area and about two-thirds of its planted forest area. A consequence of the Durban Agreement is that accounting for forest management is now mandatory, meaning that emissions and removals from all forests will now contribute towards New Zealand's net Kyoto balance of units.

Accounting for forest management in the second commitment period will be based on a "forest management reference level" (FMRL) approach that covers both natural and pre-1990 planted forests (Article 3.4 forest under the Kyoto Protocol). The FMRL is a single value representing the expected average annual net emissions over the commitment period. For New Zealand this value is 11 150 Mt CO_2-e yr^{-1} (a net emission). Credits or debits are generated if the actual total net emissions from pre-1990 forests over the commitment period deviate from the FMRL multiplied by the length of the commitment period. Potential credits – such as from lower levels of harvesting emissions than specified in the reference level – are capped at 3.5% of the 1990 levels, but debits (liabilities) are uncapped if emissions are greater.

New Zealand's submitted FMRL was based on "business as usual" projections of planted forest management, so captured the impact of the pre-1990 planted forest age-class distribution on annual harvest rates. At the time the Reference Level was submitted it was assumed that natural forests have a static carbon stock. A technical correction is required to include any new data sourced, such as a potentially non-zero carbon stock change in natural forest as derived from the remeasured LUCAS natural forest plot data, and also the revised treatment of harvested wood products specified in the Durban Agreement.

4.1.2 Provisions for accounting for natural disturbances

A technical correction is also required to take account of the new provisions for accounting for natural disturbances. The Durban Agreement introduced these new provisions to limit the exposure of countries to the risk of large liabilities arising from emissions caused by major natural disturbances. This is essentially a *force majeure* clause, allowing emissions and subsequent removals on lands affected by significant natural disturbances to be excluded from accounting within the commitment period. To take advantage of this provision, parties in 2015 need to provide separate background levels of natural disturbance for lands subject to Afforestation/Reforestation and Forest Management. The background level is a single value corresponding to the average annual emissions from all sources of significant natural disturbances. For Forest Management accounting (pre-1990 forests), this value would be added to the FMRL before the sum is multiplied by the length of the commitment period. For Afforestation/Reforestation accounting (post-1989 forests), the value is compared with the reported emissions/removals due to natural disturbance over the commitment period.

New Zealand's submitted FMRL includes natural disturbance effects only to the extent that they are captured in the yield tables derived from Forest Carbon Predictor (FCP) simulation of LUCAS planted forest plots. Attritional losses due to biotic and abiotic risk factors are captured within the yield tables, but major geological events are not modelled, and have historically had a relatively minor influence on planted forests in New Zealand.

The Durban Agreement includes definitions and conditions and a suggested approach to deriving the baseline level of disturbance. Best practice guidance for applying the Durban Agreement rules to greenhouse gas inventory reporting and accounting is currently being prepared by the IPCC. This guidance will be contained in the *2013 Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol*, planned for completion in September 2013. Until that report has been completed and approved, the minimum standards of information for estimating the background level of natural disturbance are unclear.

The Durban Agreement suggests a default procedure for determining the background level of natural disturbance, in which the background level is the mean level from a time period including 1990–2009 after outliers are removed. An iterative process for removing the outliers is described that is based on excluding values that are outside twice the standard deviation around the mean. This default procedure was based on analysis of wildfire data presented in national inventory reports. There is some recognition that these data are the exception rather than the rule – with other natural disturbances it is harder to quantify and assign emissions to discrete years. Accordingly, a country-specific approach may be taken to derive the background level.

Regardless of how the background level of natural disturbance is derived, any expectation of net credits or debits during the commitment period must be avoided. This means that an

FMRL must be resubmitted to explicitly include the level of emissions from natural disturbance in pre-1990 forests that can be reasonably expected in 2013–2020, based on a time series including at least the years 1990–2009. A separate background level is required for post-1989 forests. If there is a trend in historical emissions this should be taken into account when estimating the background level.

When accounting for the effects of natural disturbances, a Party can exclude from accounting those emissions that in any single year exceed the afforestation and reforestation background level. There must be methodological consistency between the FMRL and reporting for Forest Management (pre-1990 forests) during the second commitment period, and documentation is required to show geo-referenced location, year and types of disturbances, estimation methods, and demonstration that all criteria for the application of the natural disturbance provisions have been met.

4.2 Definition of "natural disturbance"

In the context of the Durban Agreement, natural disturbance is defined as "non-anthropogenic events or non-anthropogenic circumstances … that cause significant emissions in forests and are beyond the control of, and not materially influenced by, a Party" (UNFCCC 2012). Thus events that cause disturbance to the ecology of a forest (Pickett & White 1985) will only be classed as a natural disturbance under the Durban Agreement if they cause significant carbon emissions. Disturbance events that are anthropogenic in origin but are beyond the control of a Party (e.g. fire lit by an arsonist or damage caused by an invasive pest species) are likely to also be classed as natural disturbances for the purposes of Durban Agreement reporting.

Natural disturbances can be characterised by a number of metrics (Pickett & White 1985). The distribution refers to the spatial distribution of disturbance events across the landscape. The intensity of a disturbance refers to the physical force of the disturbance (e.g. maximum wind speed, earthquake magnitude), while the severity refers to its impact on the organism, community or ecosystem of interest. Disturbance regimes can be quantified in terms of disturbance frequency – which reflects the mean number of events per time period, or return interval – which is the average time between disturbance events.

Natural disturbances are an essential component of a healthy natural forest ecosystem, promoting gap dynamics, regeneration, and maintenance of species diversity and population structure (Pickett & White 1985; Sprugel 1991). Natural disturbances to forests occur at various spatial and temporal scales, ranging from individual leaf, branch or whole-tree mortality due to disease or wind damage, to large-scale events such as cyclones, earthquakes and volcanic eruptions (Attiwill 1994; Bellingham et al. 1995; Allen et al. 1999; Moore et al. 2003; Kurz et al. 2008a). Some disturbance events may result in multiple co-occurring forms of disturbance. For example, a single storm can cause wind throw, landslides, branch breakage, and outbreaks of insect pests and pathogens (Glade 1998; Martin & Ogden 2006; Coomes et al. 2012; Moore et al. 2012). There is generally a negative power-law relationship between the intensity of any particular disturbance event and its return frequency, i.e. low-intensity disturbance events are exponentially more frequent than high-intensity events (Malamud et al. 1998; Hancox et al. 2002). Although natural disturbances are relatively discrete events in time, relative to ecosystem timescales, the legacy of large-scale, low-frequency events on forest carbon may be evident for many decades, centuries or even millennia (McKelvey 1963; Wells et al. 2001; Wardle et al. 2004; Coomes et al. 2012; Williams et al. 2012).

4.3 Natural disturbance in New Zealand forests

New Zealand forests are at risk from a distinct set of natural disturbances due to our oceanic climate and geographic location on a tectonic plate boundary. The potential impact of natural disturbances on New Zealand's planted forests was reviewed in the context of carbon stocks as early as 1992. Maclaren and Wakelin (1992) examined the risk posed by various factors including the probability of occurrence and the likely magnitude of the effect on carbon stocks and longer term carbon storage potential. The focus was not on the expected emissions in a specific time period, but rather on the robustness of an approach to mitigation of greenhouse gas emissions based in large part on forestry.

Natural disturbances in planted forests have long been of interest in New Zealand for purely economic reasons. In most parts of New Zealand, the risk of wind damage limits the species, rotation lengths, and silvicultural practices that are commercially viable. Areas subject to heavy snowfalls or prolonged drought present similar limitations, and pests and diseases have seen formerly widely planted species replaced by less susceptible species. Most of New Zealand's planted forests are composed of species that, in their native range, are subject to fire disturbance regimes and the species provide a positive feedback with fire (i.e. they are pyrogenic). There has been considerable investment in fire prevention in planted forests, and analyses of natural disturbance risks are an important component of assessments of site suitability, profitability and forest value in the management of planted forests. Reviews of past disturbance and future risk to planted forests include Pearce & Alexander (1994), Pearce et al. (2000), Moore et al. (2012) and Watt et al. (2008, 2012).

Disturbance events in natural forests have also been widely studied because natural disturbances play a critical role in determining forest structure and function (e.g. Pickett & White 1985; Wilmshurst & McGlone 1996; Allen et al. 1999; Coomes et al. 2012). The main disturbance agents are volcanic eruptions, earthquakes, storm events and outbreaks of pests and pathogens. Drought and fire also occur (Table 1). In contrast, the effects of natural disturbance are smaller for planted forests due to their active management, location on the landscape, and short rotation period relative to the return frequency of major disturbance events. However, wind and fire can cause considerable forest damage and economic losses (Table 1).

We now briefly summarise each of the main disturbance agents affecting New Zealand's forests in terms of their historical occurrence, distribution, intensity and severity of their effects.

4.3.1 Volcanic eruptions

Large volcanic eruptions are very rare events, but they can have major impacts on forests through blast damage, burial and burning (Wilmshurst & McGlone 1996). Historically, the Hātepe Taupō Eruption in AD 180 was one of the largest in the world over the last 5000 years, depositing ash over most of New Zealand and causing devastating pyroclastic flows and outbreaks of fire that destroyed over 30 000 km^2 of forest (Wilmshurst & McGlone 1996). The concentration of planted forests in the Central North Island volcanic zone and their rotation length combine to make them vulnerable to volcanic activity. Eruptions of the Okataina and Taupō calderas would each affect a large proportion of the more than 500 000 ha of planted forests in the region (Neild et al. 1998). Although the damage caused by such an event would be internationally significant, the risk of such an eruption occurring over the next 50 years is relatively low (e.g. a large eruption ejecting > 10 km^3 of material historically occurs for Taupō every 5000–10 000 years, and the last one was in AD 180; Neild et al. 1998).

Table 1 Examples of natural disturbance events affecting New Zealand forests

Disturbance event	Date	Intensity	Recorded damage	Extent of area affected (km^2)	Reference
Hātepe eruption (Taupō)	180	VEI 7 >120 km^3 ejected	Pyroclastic flow Blast damage Ash fall Fire	>30 000	Wilmshurst & McGlone 1996
Wairarapa Earthquake	1855	8.2 magnitude	Landslides	>20 000	Robbins 1958; Hancox et al. 2002
Tarawera eruption	1886	VEI 5 2 km^3 ejected	Pyroclastic flow Blast damage Ash fall Fire	>200	Nicholls 1959; Walker et al. 1984
Ruahine drought	1914–1915		Widespread tree mortality	>100	Grant 1984
Murchison Earthquake	1929	7.8 magnitude	Landslides	5000–7000	Pearce & O'Loughlin 1985; Hancox et al. 2002
Tahorakuri forest fire	1946		Fire, exacerbated by drought	110 (and 200 of scrub)	Pearce & Alexander 1994
Sirex wood wasp in Central North Island	1949–1950	Epidemic in overstocked stands, exacerbated by drought	Effectively a heavy thinning from below – sub-dominant trees killed.	1200 (planted forests, 33% mortality)	Bain et al. 2011
Balmoral forest fire	1955		Fire	31.5	Pearce & Alexander 1994
Harper/Avoca pinhole beetle (*Platypus* spp.) outbreak	1973–1987	Initiated by heavy snow in 1973	Widespread mortality resulting in 14% decline in live stem biomass	>90	Harcombe et al. 1998
Canterbury wind storm[3]	1975	Wind gusts up to 170 km/hr	Wind throw	110 (planted forests)	Wilson 1976; Moore et al. 2012
Hira forest fire	1981		Fire	19.7 (47% pine)	Pearce & Alexander 1994
Cyclone Bernie[4]	1982	Ex-tropical cyclone, winds of > 130 km per hour	Wind throw Landslides Dieback due to pinhole beetle outbreak	60–120 (wind throw in planted forest); Up to 30% of North Island natural forests damaged	Littlejohn 1984; Shaw 1983; Hosking & Hutcheson 1998; Moore et al. 2012
Ohinewairua[5]	1983		Fire	150 (includes beech forest, shrubland and tussock)	Pearce et al. 2000
Cyclone Bola[6]	1988	Ex-tropical cyclone, winds of >100 km per hour, heavy rain	Wind throw Landslides	260 (planted forests)	Somerville et al. 1989; Moore et al. 2012
Hinewai fire	2011		Fire caused by lightning	3	Wilson 2011

[3] http://hwe.niwa.co.nz/event/August_1975_South_Island_High_Winds

[4] http://hwe.niwa.co.nz/event/April_1982_North_Island_Ex-tropical_Cyclone_Bernie

[5] http://www.ruralfirehistory.org.nz/documents/Ohinewarua.htm

[6] http://hwe.niwa.co.nz/event/March_1988_North_Island_Ex-tropical_Cyclone_Bola

The Tarawera eruption of 1886 was the largest since European settlement, and caused widespread forest damage over >200 km^2 (Nicholls 1959; Walker et al. 1984), creating bare surfaces upon which forest recovery is still taking place (Clarkson & Clarkson 1995). Mt Ruapehu, one of New Zealand's most active volcanoes, has significant eruptions every 50 years or so (the last ones being in 1945 and 1995–1996) but there has been little recorded forest damage associated with recent Ruapehu eruptions (Neild et al. 1998). The 1995 and 1996 eruptions resulted in a layer of ash up to 2 mm thick being deposited over a wide area, but it would take ash deposits greater than 100 mm to kill seedlings and over 500 mm to cause breakage of branches in mature trees (Neild et al. 1998). The August 2012 Tongariro eruption has resulted in some forest dieback (Figure 1; L. Young, Canterbury University, pers. obs.), most likely from exposure to toxic sulphur dioxide gas or ash. This dieback is restricted to about 2 km^2 of short-statured forest located downslope of the crater.

Figure 1 Forest dieback on Mt Tongariro following the eruption on 6 August 2012. Dieback is thought to be caused by exposure to toxic sulphur dioxide gas or ash. Photo taken by Laura Young, 19 November 2012.

4.3.2 Earthquakes

New Zealand lies on a very active plate boundary and large, potentially damaging earthquakes of >7.0 magnitude (M) occur on average once every 2.5 years (GeoNet 2012). The damage to forests caused by earthquakes depends greatly on the intensity, location and depth of the earthquake, as well as the geographical terrain of the adjacent area (e.g. slope, soil type, forest composition). Historically significant earthquakes include the 8.2 M Wairarapa Earthquake of 1855 in which landslides occurred over an area of > 20 000 km^2, and the 7.8 M Murchison Earthquake of 1929, during which 5000–7000 km^2 of predominantly natural forest was affected by landslides (Hancox et al. 2002). Earthquakes can also result in extensive, small-scale (< 0.4 ha) tree mortality due to earth movement, tree toppling and falling rock debris (Allen et al. 1999). Earthquake-damaged forests tend to have high volumes of dead wood and are therefore more susceptible to insect and pathogen outbreaks (Rawlings 1953; Harcombe et al. 1998; Allen et al. 1999).

Currently, perhaps the greatest earthquake risk to forests is associated with the Alpine Fault, which has a 30% probability of producing a large earthquake in the next 50 years (Berryman et al. 2012). The steep, mountainous, forested terrain surrounding this fault line is highly susceptible to landslides (Cullen et al. 2003; Hilton et al. 2011). For example, an estimated 49% of the forested land area within Karangarua catchment in Westland was disturbed by erosion or sedimentation following the Alpine Fault earthquake in 1645 (Wells et al. 2001).

4.3.3 Storms

Storm events are associated with a range of types of damage to forests, including branch and tree breakages due to high wind and heavy snow, landslides, flooding, and ongoing pest/pathogen outbreaks. In New Zealand, the ex-tropical storms such as Cyclone Bola (1988), Cyclone Bernie (1982), Cyclone Giselle (the Wahine storm) and the 1936 cyclone occur on average once every year and are generally more frequent and more intense in the upper North Island (Sinclair 2002). Non-tropical storms are more frequent, especially in the South Island, and can be just as severe[7]. Heavy rain during storm events saturates the soil and can result in widespread occurrence of landslides. Heavy snow can cause stem breakage and destructive avalanches (Harcombe et al. 1998; Allen et al. 1999).

Strong winds can severely damage forests across a range of spatial scales, including defoliation, branch breakage, uprooting of individual trees, and complete stand-level wind throws of multiple hectares in size (Peterson 2000; Ulanova 2000; Martin & Ogden 2006; Moore et al. 2012). Wind damage is more severe when associated with heavy rain as waterlogged soils give way more easily. Susceptibility to wind damage is also higher in recently disturbed stands due to the presence of canopy gaps or edges that catch the wind, but the most important factor is tree height, with thresholds for severe wind damage in natural forests typically being trees over 12–18 m in height (Martin & Ogden 2006). The risk of wind damage to planted forests is a significant factor considered in their management, affecting site and species choice, silviculture and harvesting. Damage may be attritional, involving scattered stem breakage and/or toppling, or widespread, requiring the re-establishment of the stand (Moore et al. 2012).

Cyclone Bernie (1982) is a relatively well documented example of a major ex-tropical storm affecting New Zealand forests. Sustained high winds (and heavy rain) associated with Cyclone Bernie resulted in over 120 km^2 of wind damage to planted *Pinus radiata* forests in the North Island (Littlejohn 1984), and an equivalent or greater amount of damage in natural forest, where initial damage resulted in ongoing dieback up to 10 years following the storm event (Shaw 1983; Hosking & Hutcheson 1998). Cyclone Bola (1988) also caused major damage to planted forests and was unusual in that forests were affected from Northland to Nelson. Since records began in the 1940s, at least 60 000 ha of significant wind damage has been recorded in planted forests. Wood from damaged stands can generally be salvaged, and in the most severe events has amounted to approximately 2% of the total annual harvest (Moore et al. 2012).

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[7] http://hwe.niwa.co.nz/

4.3.4 Pests/pathogens

Outbreaks of insect pests and pathogens are a form of natural disturbance that can cause widespread tree mortality (e.g. Kurz et al. 2008a). A prominent example of an insect pest affecting New Zealand's natural forests is the pinhole beetle (*Platypus* spp.; Hosking & Hutcheson 1998; Allen et al. 1999). A recent pathogen example is kauri dieback (*Phytophthora* taxon *Agathis* (PTA); Beever et al. 2009). Pest/pathogen outbreaks are often associated with another disturbance event (e.g. a major storm, earthquake or drought) that subjects trees to stress and/or leads to an increase in dead wood, allowing the insect or pathogen to increase in numbers. These populations then start attacking live trees, creating a positive feedback that results in further forest dieback (Rawlings 1953; Harcombe et al. 1998; Hosking & Hutcheson 1998; Allen et al. 1999). Ridley et al. (2000) reviewed the risks to indigenous forests presented by introduced pests and pathogens. Species-poor forests, such as the southern beech (*Nothofagus* spp.) forests, are generally more susceptible to outbreaks than species-rich forests. Insect and pathogen outbreaks are, therefore, particularly concerning for planted forests as these rely predominantly on a single species (*Pinus radiata*). However, one reason for the reliance on *Pinus radiata* is its relative resistance to the pests and diseases that limit the use of other species, e.g. *Eucalyptus* spp.

Most insect pests cause suboptimal growth or malformation rather than the death of healthy trees. Exceptions include the bark beetles of Northern Hemisphere coniferous forests (which are not present in New Zealand) and the *Sirex* wood wasp. An epidemic of *Sirex* in Central North Island forests in 1949–1950 followed a prolonged drought and resulted in the death of one-third of trees across 120 000 ha (Bain et al. 2011). However these stands were overstocked and it was recognised that the result was essentially a beneficial thinning operation. *Sirex* is not regarded as a significant pest in well-managed planted forests today, in part due to successful biological control.

There are only six diseases currently regarded as significant in planted coniferous forests in New Zealand. Needle blights caused by *Dothistroma pini* and *Cyclaneusma* spp. are estimated to cause $24 million and $51 million worth of annual lost growth, respectively (Ridley et al. 2000). Root and collar rot caused by *Armillaria* spp. costs a further $37 million in lost growth (Ridley et al. 2000). Swiss needle cast of Douglas-fir (*Pseudotsuga menziesii*), caused by *Phaeocryptosus gauemannii*, and cypress canker caused by *Seiridium* spp. suppress growth and the latter can cause mortality. Pests and diseases are a much more serious problem in eucalypts, which is one reason they comprise only 1% of the total area of the planted forest estate (Ministry for Primary Industries 2012). Outbreaks are unlikely to be considered non-anthropogenic given the long history/policy of control. For a pest outbreak to count as a natural disturbance we would have to show (1) that we were still actively managing the risk and (2) that the damage is greater than the baseline level.

4.3.5 Fire

Historical wildfire data reveal that very little indigenous forest has been damaged by fire over the last two decades (Anderson et al. 2008; Mason et al. 2011). Almost all large wildfires are ignited by humans and begin in and are mainly restricted to grassland, shrubland or peatland communities. Fires generally do not burn far beyond the margins of indigenous forest. This is partly due to the vast majority of remaining indigenous forest occurring where wildfire threat is very low (Mason et al. 2011; McWethy et al. 2010), as well as the very low ignitibility of New Zealand's major canopy-dominant tree species. Even species such as mānuka (*Leptospermum scoparium*) and kānuka (*Kunzea ericoides*) that are considered highly flammable in New Zealand have low ignitibility on an international scale (Mason et al. in prep). Some forest types – mānuka and kānuka in particular – do propagate fire well once the fire has become established (usually in surrounding grassland or shrubland), but this generally requires weather conditions that are favourable to fire spread. These more-at-risk forest types account for a small fraction of pre-1990 forest carbon storage (approximately 3%; calculated from data in Beets et al. (2009)), but as they are early-successional species they are a more significant component of post-1989 carbon storage.

Fire has been spread by humans in New Zealand since pre-European times (McGlone 1989; Ogden et al. 1998) and was the main means used to convert indigenous vegetation to pasture by European settlers (Pearce et al. 2008). Much of the burning was accidental as fires spread and could not be contained. Between 1895 and 1920 1.01 million hectares of indigenous forest were destroyed while the small areas of plantation forest that existed at that time escaped relatively unscathed (Beaglehole 2012). Wildfire data have been collected by the National Rural Fire Authority continuously since 1992 and are used to estimate the emissions from wildfires reported in the national greenhouse gas inventory. On an annual basis, there were on average 3270 vegetation wildfires recorded nationally from 1992 to 2012, with an increasing trend during this period. However, the annual area burned has remained static at an average of 5618 ha per year (NRFA, unpubl. data). Grassland and shrubland fires make up 51% and 40% of the area burned, respectively, with the average annual area of forest burnt just 484 ha. For most of the time series no distinction was made between wildfire in natural and planted forests, but most of the area damaged is believed to be in planted forests. The severity of the damage is normally not recorded – in many cases trees may survive and grow on to maturity, or the wood can be salvaged.

In 1946 the most devastating fire in New Zealand's planted forest history burnt about 11 000 ha of forest and twice that amount of shrubland. Notably this fire followed a drought and the large amount of dead wood left after the fire contributed towards the 1949–1950 *Sirex* outbreak, emphasising the role of multiple agents in disturbance.

4.3.6 Drought

Internationally, drought events are known to have severe impacts on forest growth and tree mortality (Nepstad et al. 2007; Kumagai & Porporato 2012). New Zealand is somewhat buffered from drought due to our maritime climate and relatively high rainfall in forested areas. However, drought is still an important natural disturbance. Drought-induced dieback has been observed in the Ruahine Ranges (Grant 1984, 1991), Kaweka Ranges (Hosking & Hutcheson 1988) and Kaimai Ranges (Jane & Green 1983). Drought-stressed trees are more susceptible to pathogenic attack (Hosking & Kershaw 1985), and drought-stressed ecosystems are more susceptible to forest fire due to their high fuel load (Wilmshurst et al. 1997). *Pinus radiata* is a relatively drought tolerant species, which is one of the reasons why it is able to be grown across a wide range of sites. Less-drought-tolerant species are established on sites with higher rainfall. For example, Douglas-fir is generally established on sites where the average annual rainfall exceeds 1000 mm.

4.3.7 Other disturbances

Other types of natural disturbance are less common but also known to occur throughout New Zealand. Coastal forests may be affected by infrequent tsunamis generated by offshore earthquakes. Flooding following rain events has the potential to induce tree mortality (e.g. Mark et al. 1977). However, the forest communities on natural floodplains are typically adapted to frequent floods, and the impact of flood disturbance to the forest is generally low. Temperature extremes such as unseasonal frost events can damage trees, but is generally not sufficient to cause widespread tree mortality (Bannister 2007; Cieraad et al. 2012). Damage to planted forests is limited by careful species and site choice and maintaining high standards of site preparation, planting stock, planting practices and weed control. Invasive browsing species such as possums (*Trichosurus vulpecula*) and red deer (*Cervus elaphus*) could be considered disturbance agents in New Zealand forests, but would not be classified as disturbances under the Durban Agreement, and are also thought to have minimal effect on forest biomass (Holdaway et al. 2012).

4.3.8 Effects of climate change on natural disturbance regimes

Ongoing anthropogenic climate change may cause an increase in climate extremes such as high rainfall events and more frequent and prolonged drought, placing existing forests under increasing levels of stress (Peterson 2000; Carey-Smith et al. 2010; Clark et al. 2012; Seidl & Blennow 2012). For New Zealand this may mean a greater likelihood of storm damage (e.g. landslides and wind throw), as well as a greater risk of drought and fire in eastern areas (MfE 2008; Clark et al. 2012). Insect pests and pathogens are also likely to increase with increasing temperatures in temperate and boreal regions, and there is evidence that this is already occurring in Europe and North America (Kurz et al. 2008a; Seidl & Blennow 2012). Drought is predicted to become more frequent and more severe, especially in eastern areas (Clark et al. 2011) which could reduce forest growth rates and increase forest susceptibility to pathogens and fire. Although these scenarios suggest generally increasing risks to forests in the long term, Watt et al. (2008, 2012) investigated the effect of climate change on New Zealand's current planted forests and concluded that the 2013–2020 period may not be significantly affected.

4.4 Potential impact of natural disturbance on forest carbon

Natural disturbances in New Zealand can have pervasive and long-lasting impacts on forest carbon stocks at the landscape scale (Coomes et al. 2012) (Figure 2). Carbon emissions often occur over a period of a decade or more following a single disturbance event due to ongoing mortality and CWD decay (Hosking & Hutcheson 1998; Coomes et al. 2012; Mason et al. 2013) (Figure 2). Interactions and synergies between multiple disturbance agents frequently occur (e.g. wind damage and pest/pathogen outbreaks, drought and fire), making direct quantification of carbon emissions associated with a single disturbance event difficult.

Disturbance impacts on forest carbon can be grouped into three categories:

1. Partial dieback/crown damage causing reductions in growth (lower carbon sequestration rates)
2. Tree death (individual or whole stand causing an increase in coarse woody debris (CWD) and carbon loss due to decay)
3. Catastrophic damage to land and forests (e.g. landslides)

Partial damage includes branch breakage by snow, wind (etc.), defoliation, and other forms of tree-level damage that do not directly result in tree mortality. This low-intensity damage can be widespread throughout forests after major disturbances (Harcombe et al. 1998; Coomes et al. 2012). The effect of partial damage on forest carbon is twofold. Firstly, the broken material enters the deadwood pools (CWD, fine woody debris and litter), and then emits CO_2 to the atmosphere as it decays over time. Finer material (e.g. < 10 cm in diameter) tends to decay quite rapidly and it can be assumed that all the carbon in these pools is emitted within the year following the disturbance events. However, since most of the carbon is contained in the main stem of a tree, the carbon loss associated with decay of litter and fine woody debris is minimal. The carbon loss may also be partially offset by increases in tree growth following the disturbance event as trees respond to reduced competition and increased light availability (Bellingham et al. 1995; Tanner & Bellingham 2006). However, damaged trees may also be slower growing and susceptible to subsequent pathogen attack in the years following the disturbance event (Hosking & Hutcheson 1998; Seidl & Blennow 2012). The main methods for quantifying the effects of partial damage on forest carbon is at the individual tree level through the use of permanent plots (e.g. Allen et al. 1999). Growth models used in planted forests incorporate this level of damage, meaning that projections made using these models already take this into account.

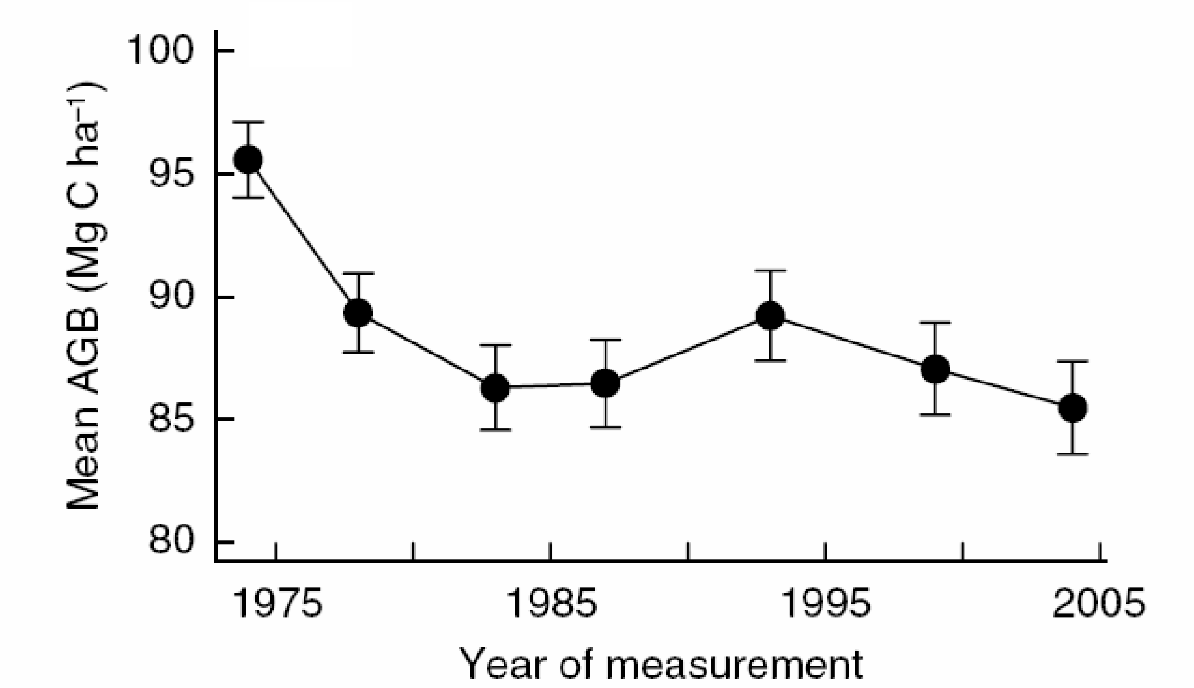

Figure 2 Changes in mean live above-ground biomass carbon stocks (AGB, Mg C ha^{-1}) of 246 stands of mountain beech (*Nothofagus solandri* var. *cliffortioides*) from the Harper/Avoca catchments, Canterbury (±1 SEM) monitored over 30 years. The forests originally consisted largely of mature stands. They were affected by severe windstorms and snowstorms in 1968 and 1973 that caused extensive damage. Populations of pinhole beetles (*Platypus* spp.) then built up on the woody debris before attacking live trees, spreading their associated pathogens and resulting in ongoing mortality through to 1987 (Wardle & Allen 1983; Harcombe et al. 1998). The forest then went through a period of growth until further damage was caused by heavy snow in 1992 and an earthquake in 1994 (Allen et al.1999). Figure adapted from Coomes et al. (2012).

The next level severity of damage is tree mortality. This can range in scale from deaths of individual isolated trees to widespread mortality across whole stands or landscapes. Nearly all natural disturbance agents are capable of killing trees, and thus tree mortality is a key form of forest damage following disturbance. The effects of tree mortality on forest carbon depends on the scale of the dieback (single tree vs whole stand), the scale of recovery (sprouting, recruitment, ingrowth, release of seedlings), and the rate of CWD decay (Mason et al. 2013). The death of large trees transfers a significant portion of the total carbon into the CWD pool. This then decays over subsequent years and decades, gradually releasing its carbon to the atmosphere due to a generally slow decay rate of heartwood. In natural forest, the death of trees creates canopy gaps that initiate the process of forest recovery and secondary succession (Pickett & White 1985), and this growth process may partially offset the carbon lost due to decay of CWD. Tree mortality and CWD dynamics are again best quantified at the individual tree level through the use of permanent plots (Allen et al. 1999; Coomes & Allen 2007; Coomes et al. 2012). Models of planted forest growth take into account some disturbance-induced mortality, but plots in which a significant number of trees are killed are more likely to be abandoned and not contribute towards predictive model development.

The final level of damage is catastrophic damage to land and forests (e.g. due to landslides) that creates a new surface and initiates the process of primary succession (Walker & del Moral 2003). This level of damage transfers all live biomass into dead pools. Soil carbon is typically in a more stable form and is transferred to other locations rather than emitted to the atmosphere. Indeed, there is some evidence to suggest that landslides may act as a net sink of soil carbon (Hilton et al. 2011). Although we know a bit about the rate of re-establishment of forest onto landslide scars (Mark et al. 1989; Smale et al. 1997), relatively little is known about the fate of forest carbon following landslides. A portion of it is likely to be buried in the landslide debris, and all is likely to be emitted to the atmosphere over time through decay processes. Forest recovery on slips is slow (>100 years) as soil takes time to develop and the slope can be unstable for many years. Catastrophic disturbance (i.e. landslides) can be detected using permanent plots (e.g. Allen et al. 1999) and through remote sensing (Chambers et al. 2013; Cheng et al. 2013). Remote sensing has the advantage of being able to operate at a national scale and detect large landslides that may not be captured within the existing plot network. In planted forests catastrophic damage will usually be followed by salvage harvesting (depending on the age of the stand) and stand replacement.

5 Methods

5.1 Overview of approach

Our approach to quantifying New Zealand's background-level emissions from forests due to natural disturbance takes into account the natural disturbance regimes affecting New Zealand's forests, the likely effects of these disturbances on forest carbon, the nature of the available data (both now and in the future), and the Durban requirements outlined above. We estimate carbon losses due to natural disturbances by firstly identifying areas where natural disturbance has occurred, using the best possible combination of plot-level metadata and historical records (planted forests) or remote sensing (natural forests). We then use plot data to estimate net carbon implications by modelling tree mortality and subsequent growth and wood decay processes, providing a dynamic and explicit treatment of carbon stock change resulting from natural disturbance.

We quantified carbon emissions from natural forests and planted forests separately. This was done because natural and planted forests occupy different parts of the landscape and are therefore subject to different disturbance regimes. Natural forests tend to be on steeper, landslide-prone, higher altitude sites than planted forests. Fire is a major threat to planted forests because these are mostly comprised of pyrogenic species and tend to be located closer to human settlement and are bisected by access roads (which are a major predictor of fire incidence in New Zealand). The types of data available also vary between planted forests and natural forests. Forest-owner records for planted forests provide detailed annual records of major disturbance events, whereas extensive multi-year plot data and remote sensing must be used for natural forests. Finally, planted forests typically consist of single-age, single-species stands, which allows for a more standardised modelling approach that includes potential salvage logging following disturbance events. Natural forests are usually multi-species and multi-aged requiring a more detailed plot-based modelling approach.

5.2 Quantifying carbon emissions from natural forest

Both permanent plot data and remote sensing data were used to quantify the area disturbed in natural forests. Remeasurement data from permanent plots provide a quantitative record of tree death and reductions in growth caused by partial dieback and crown damage (i.e. low-intensity disturbance). Since the area affected by catastrophic disturbances was not likely to be adequately sampled using the exiting plot network, we used remote sensing techniques to separately assess catastrophic disturbances (landslides). All carbon emissions resulting from natural disturbances of natural forest were quantified using CWD modelling to capture the process of wood decay over time. Data constraints mean that the estimates of area disturbed and associated carbon losses for natural forest are based on pre-1990 natural forest data and do not strictly relate to the years 1990–2009. As there are only sufficient data for a single estimate of carbon emissions from natural disturbance (i.e. using two points in time), this value was annualised and applied across all years for the 1990–2009 period. There were no data on annual variability in emissions.

5.2.1 Plot survey data

Data from permanently marked inventory plots were sourced from the National Vegetation Survey of New Zealand (NVS; Wiser et al. 2001). Only 20 × 20 m (400 m^2) geo-referenced plots that contained stem diameter data that had been remeasured at least once were selected. Plots were further limited to those with woody species total cover of >30%. Data were checked for errors and plots containing significant data issues (e.g. non-matching tags, stems missed during the initial measurement, and diameter typos) were removed. Plot locations were compared with the LUCAS 1990–2008 land use map, and the ~5% of plots that were not located in the post-1990 natural forest class (class number 71) were removed from the dataset. The final dataset comprised 348 557 stems from 3077 permanent plots sampling New Zealand's pre-1990 natural forest, spanning a rainfall gradient of 630–8700 mm yr^{-1} and a mean annual temperature gradient of 4.5° to 16.0°C (climate data from Leathwick et al. 2003). Altitude ranged from 0 to 1520 m, and the median slope was 25 degrees. Plots were generally located along lines that had random bearings and starting points from streams, and had been established and remeasured using standard methods (Hurst & Allen 2007).

Where multiple measurement periods existed (N = 1049 plots) the measurement period closest to the mean measurement interval (10 years) was used. Census intervals varied among plots and ranged from 2 to 30 years (median = 10), and spanned the period 1969–2012. The median start year was 1979 and median end year was 1993 but there was much variation and 1567 plots had either their start or end date within the 1990–2009 period. A subset of 149 sample plots had both start and end dates within 1990–2009; 1141 had their measurement interval midpoint somewhere in this period. For the purpose of this analysis all 3077 plots were used to ensure the best possible national coverage and sample size.

5.2.2 Coarse woody debris (CWD) decay data

Data on wood density in live trees and CWD logs were obtained from various sites throughout New Zealand (D. Peltzer, unpubl. data); this dataset contains wood density samples for 53 species. Collectively these species contributed more than 95% of the carbon entering the CWD pool in the studied plots. Cores from live trees were taken using an auger. Samples from CWD logs were taken using an auger or by cutting a disk or wedge in the wood. The dimensions of all wood samples were taken while fresh, to the nearest millimetre, to permit calculation of fresh volume. Volume measurements in disks and wedges included hollows, so that our measure of CWD density incorporates carbon loss due to hollowing in logs in advanced stages of decay.

Diameter at breast height was recorded for live trees. The diameter of CWD logs was recorded at both ends and halfway along the length, with the mean of these measurements taken as a measure of log diameter (Richardson et al. 2009). The time since death for all CWD logs was estimated by matching tags on CWD logs to those of live trees previously recorded in surveys of permanent forest plots (survey data were obtained from NVS). The time of death was estimated as halfway between the year in which the stem was last observed alive and the year of the survey in which it was first recorded as dead. The density of samples taken from live trees and CWD logs was determined by dividing the weight of wood samples oven-dried to a constant weight by their fresh volume. The density of CWD logs was expressed as a proportion of the wood density of live trees. The type of CWD was recoded as one of four categories: log, snag (standing dead tree), stump, suspended. Climate data were obtained for the location of each sample from the climate maps underpinning the Land Environments of New Zealand (LENZ) ecosystem classification (Leathwick et al. 2003).

5.2.3 Coarse woody debris (CWD) decay modelling

We used a boosted regression tree (BRT) framework to model CWD wood density (Elith et al. 2008). We included the following variables in the model: time since death, trunk diameter, species identity, type of CWD (snag, branch, log, stump and suspended), mean annual rainfall and mean annual temperature. We chose to use BRTs because interactions between these variables are believed to strongly influence on decay rates, and BRTs model interactions between predictors very well.

In all BRT models we used a tree complexity of 3 and learning rate of 0.001. The maximum number of trees was set at 20 000, though this limit was never reached. Model goodness of fit was assessed using the cross-validated Pearson correlation between observed and fitted values of the response. Cross validation was performed by randomly removing 10% of the plots from the dataset, fitting the BRT model on the remaining plots and then assessing the goodness of fit on the removed plots. We can measure the influence of predictors in BRT models by recording the proportion of regression tree “branches” involving each variable. Regression trees work by making repeated dichotomous divisions of the data, and measuring the influence of predictors this way indicates how frequently each variable was used to divide the data. The other way to measure predictor influence is the range of values spanned in the partial contribution plots.

For the purposes of predicting CWD decay in our plots, we were obliged to omit CWD type, since this is generally unknown for the tress that died between plot measurements.

While we do not have CWD measurements for all the species in our study, we are still able to use a BRT model including species identity as a predictor. We did this by holding the effect of species identity constant when predicting decay rates for species not included in our CWD decay dataset.

5.2.4 Estimation of carbon in live trees

Above-ground biomass of individual stems was calculated using an allometric formula previously designed for use across a range of New Zealand forest species (Beets et al. 2012b). The Beets et al. allometry estimated carbon separately for the trunk, branches and leaves. Wood density values were taken from Beets et al. (2009), or from a unpublished Landcare Research database of live-wood density measurements for species not listed in Beets et al. (2009). Height was estimated from stem diameter, with the models fitted by Mason et al. (2012), using height measurements from the LUCAS natural forest plot network of more than 1200 survey plots (20 × 20 m) located throughout the indigenous forests of New Zealand. We assumed that the carbon content of above-ground biomass was 50% following accepted practice (Coomes et al. 2002).

5.2.5 Modelling carbon lost to decay in disturbed plots

To identify plots in which disturbance had occurred we used criteria based on observed changes in mean stem diameter and total stand basal area between measurements. Specifically, we identified "disturbed" plots as those that had a decrease in mean tree size and a decrease in total stand basal area, as would be observed in the immediate period following a natural disturbance event that results in the death of larger canopy trees and possible recruitment of smaller seedlings. Our criteria are similar to those used by Coomes and Allen (2007) to identify "disturbed" stands in mountain beech forest, except that we use a decline in basal area sub-criterion instead of their increase in number of stems sub-criterion. The Coomes and Allen (2007) criteria were not suitable because an observed decrease in stem size and increase in stem number could be in response to recruitment in the absence of disturbance or a disturbance that occurred before the measurement interval, meaning we would not capture the carbon lost to mortality, because the mortality would not have been observed during the measurement period. Furthermore, while the Coomes and Allen criteria were developed for mono-specific forest, in multi-species forest there could be a considerable lag between basal area loss and increases in stem density, e.g. due to understory competition from crown fern (Coomes et al. 2005).

Carbon retained in CWD from stems that died during the study period was estimated using a BRT model of CWD wood density (expressed as a proportion of live wood density) including species identity, trunk diameter and time since death as predictor variables. The diameter of logs entering the CWD pool following tree mortality was assumed to be the same as the diameter of the tree when last observed alive. The year of death was assumed to be midway between the survey when the tree was last observed alive and the following survey date. We estimated carbon content for each CWD log at the final plot measurement by entering the species identity, trunk diameter and time since death into the BRT model. The modelled final CWD carbon content for each stem was summed for each plot to give the total carbon associated with mortality that was retained as CWD at the final plot measurement.

It was assumed that carbon in branches and leaves was lost instantaneously when trees died, so that only that contained in the trunk entered the CWD pool. Data from our national CWD dataset suggest that all leaves decompose within the first 5 years after stem death, while twigs are generally lost within the first 10 years. In most cases, branches decay in less than 15 years. We were unable to account for retention of carbon from decaying wood in the litter, soil or microbial biomass, which might slightly overestimate the negative impact of canopy tree mortality on carbon storage. We also did not account for carbon losses due to decay of CWD present at the beginning of the measurement period. This will mean our net carbon change estimates have a positive bias. However, legacy CWD decay can be assumed to be the same for both disturbed and non-disturbed plots providing they had on average the same initial CWD stocks and this positive bias has no influence on the difference between net carbon change estimates for disturbed and not-disturbed plots.

For each plot we calculated carbon gain due to growth of stems alive at both measurement periods, carbon gain due to new recruits, carbon lost from live stem pool due to mortality, and the carbon associated with this mortality that was retained in CWD residue. All carbon change estimates were converted into an annual rate by dividing by the number of years between measurements.

5.2.6 Accounting for bias in NVS plot locations

The non-random location of NVS plots across regions could potentially lead to biased estimates of natural disturbance due to oversampling in some Regional Districts (e.g. Canterbury, Southland) and undersampling in others (e.g. Northland). To overcome this potential bias we calculated average carbon fluxes at the regional level, and then calculated the area-weighted average carbon flux across regions:

$$National\ C\ flux = \frac{\sum(Regional\ C\ flux \times Regional\ forest\ area)}{Total\ forest\ area} \quad (1)$$

5.2.7 Remote sensing of large-scale landslides

The 1990 and 2008 satellite images used for the LUCAS land-use mapping were used to estimate large-scale changes in natural forest cover due to natural disturbances. Such disturbances are likely to be missed by the permanent plot network due to low plot numbers in some areas and possible field sampling bias (e.g. avoiding dangerous unstable and steep areas that are prone to slipping).

Imagery and image preparation

LANDSAT imagery covering New Zealand for 1990 at 30-m resolution was used for the baseline coverage and we compared this with 10-m-resolution SPOT imagery acquired for 2008. All images were rectified to NZTM, masked to exclude cloud, standardised for solar elevation and corrected for topographic effects before they were mosaicked to a single national coverage for each date.

Image segmentation

The national layers were then segmented using a segmentation routine in which land cover was classified into homogeneous polygons, of at least 0.5 ha in area, on the basis of reflectance values in bands. By comparing the two segmented coverages, areas that changed from low shortwave reflectance values (i.e. vegetated) to high values (i.e. little or no vegetation) between dates were automatically identified and added to a "checklist" of "change polygons". This list represented possible landslides that occurred between 1990 and 2008.

Manual checking

We developed a “check tool” in ERDAS Imagine that presented each “change polygon” on both dates of imagery simultaneously, together with 2.5-m-resolution true-colour SPOT Maps imagery from 2009 as an additional guide, to enable us to manually confirm each feature as being a landslide or not (see Figure A1.1 in Appendix I for screen shot from the check tool). In this way we manually examined 13 732 “change polygons”, of which 2030 were determined to be actual landslides that occurred sometime between 1990 and 2008.

Carbon emissions from landslides

Confirmed landslide polygons were intersected with a national map of current live carbon storage in natural forest and shrubland (Mason et al. 2012) to gain an estimate of pre-landslide biomass carbon stocks. It was assumed that all live carbon stocks were transferred into dead biomass pools following the landslide event. Annual carbon emissions from landslides were estimated from the CWD decay models. Specifically annual carbon emissions from landslides were therefore estimated as the average annual percentage carbon loss from CWD decay (estimated from disturbed NVS plots) multiplied by the total carbon stock of each landslide polygon. Our analysis assumed zero regrowth of live biomass in the decade immediately following the landslide event. While this is not strictly true as regrowth some regrowth can occur within this period (Mark et al. 1989; Smale et al. 1997), carbon gain are due to regrowth is likely to be small relative to the total carbon loss due to mortality and decay over that period.

5.2.8 Estimation of post-1989 natural forest emissions

Very few data currently describe the area of natural post-1989 forest and its associated carbon stock. Unlike pre-1990 natural forest, which can be assumed to be in a relatively stable state, both the area and the carbon stocks associated with post-1989 natural forest have changed substantially over the 1990–2009 period. Total area of post-1989 forest was estimated by the Ministry for the Environment (MfE) as 44 800 ha in 2008 (Andrea Brandon, pers. comm., 2013). Using this value, we estimated the total forest area for 1990–2009 by assuming a linear increase in forest area over this period such that it resulted in 44 800 ha in 2008 (Table 2).

Carbon stock estimates for post-1989 forest provided by MfE (Andrea Brandon, pers. comm., 2013) were 98.7 Mg CO_2e ha^{-1} based on measured plot data in 2012; 58.3 Mg CO_2e ha^{-1} in 2008, and 10.1 Mg CO_2e ha^{-1} in 1990. The last two numbers are back-cast estimates conducted by MfE from the 2012 data. We fitted an exponential relationship to these estimates and used that relationship to predict annual average carbon stock for 1990–2008 (Table 2).

To estimate emissions from natural disturbance in post-1989 natural forests we then expressed the annual carbon emissions in pre-1990 forests as a percentage of total carbon stock, and then multiplied this number by the estimated post-1989 forest carbon stock. This provided annual carbon emissions from natural disturbance in post-1989 forests for the period 1990–2009.

Table 2 Area estimates for post-1989 natural forest and their associated carbon stock for 1990–2009

Year	Post-1989 natural forest area (ha)	Post-1989 natural forest carbon stock (Gg CO_2-e)	Average carbon stock (Mg CO_2-e ha^{-1})
1990	2358	24	10.1
1991	4716	52	11.1
1992	7074	87	12.3
1993	9432	128	13.6
1994	11 789	178	15.1
1995	14 147	236	16.7
1996	16 505	304	18.4
1997	18 863	385	20.4
1998	21 221	479	22.6
1999	23 579	590	25.0
2000	25 937	718	27.7
2001	28 295	867	30.6
2002	30 653	1039	33.9
2003	33 011	1238	37.5
2004	35 368	1469	41.5
2005	37 726	1734	46.0
2006	40 084	2039	50.9
2007	42 442	2389	56.3
2008	44 800	2791	62.3
2009	47 158	3252	69.0

5.3 Quantifying carbon emissions from planted forest

5.3.1 Types and severity of disturbance

Models for predicting mortality in planted forests are important components of forest growth modelling systems (Weiskittel et al. 2011). Mortality in forest plantations can be categorised into three types (Kimberley 2007):

- *Attritional mortality* – low-level mortality not due to rare catastrophic events in stands that are not at excessively high stockings.
- *Catastrophic mortality* – mortality occurring as a result of rare and catastrophic events such as major storms and fires.
- *Competition-induced mortality* – this occurs in highly-stocked stands when competition becomes intense causing the smaller, less vigorous trees to die.

Catastrophic events are usually modelled outside of stand growth models used for planted forest management. Competition-induced mortality is a key element of stand modelling in New Zealand but the extent to which attritional mortality is included varies. In practice most operational growth-models in New Zealand are developed from empirical data obtained from permanent sample plots (Mason 2005). Distinguishing between competition-induced mortality and attrition within sample plots can be difficult.

New Zealand’s submitted FMRL includes natural disturbance effects only to the extent that they are captured in the yield tables derived from Forest Carbon Predictor (FCP) simulation of LUCAS planted forest plots. The FCP simulates the four biomass pools (above-ground biomass, below-ground biomass, dead wood and litter) and accounts for transfers between them due to factors such as needle fall, mortality, pruning and thinning. It integrates the 300 Index Growth Model (Kimberley et al. 2005), a wood density model (Beets et al. 2007), a stand tending model (Beets & Kimberley 2011) and the C_Change carbon allocation model (Beets et al. 1999), to enable predictions of carbon stocks and changes in New Zealand's planted forests.

Carbon stock changes during the second commitment period will be estimated based on stock changes in a network of permanent sample plots, supplemented by LiDAR (Beets et al. 2011, 2012a). In pre-1989 forests ground plots are located on a 4 × 4 km grid, with each plot representing 1600 ha. In pre-1990 forests plots are located on an 8 × 8 km grid. The plot networks are too sparse to capture the smaller annual areas affected by catastrophic disturbance. The approach taken was to use available datasets on catastrophic fire and wind damage areas in planted forest as the basis to calculate emissions from those sources (Figure 3).

It was assumed that the effects of biotic disturbance agents had been captured by the growth models used to derive the FMRL for pre-1990 forest accounting (Table 3). Climatic factors were also assumed to be captured, but not catastrophic damage (e.g. through fire, wind, earthquakes, volcanism). However, it can be assumed that there was no catastrophic damage to planted forests during the baseline period other than from wind and fire and it is reasonable to assume the same for the commitment period to 2020. The estimated natural disturbance baseline is therefore based on estimates of catastrophic wind and fire damage. A discussion of the background level of disturbance due to existing pests and diseases is included in Appendix II.

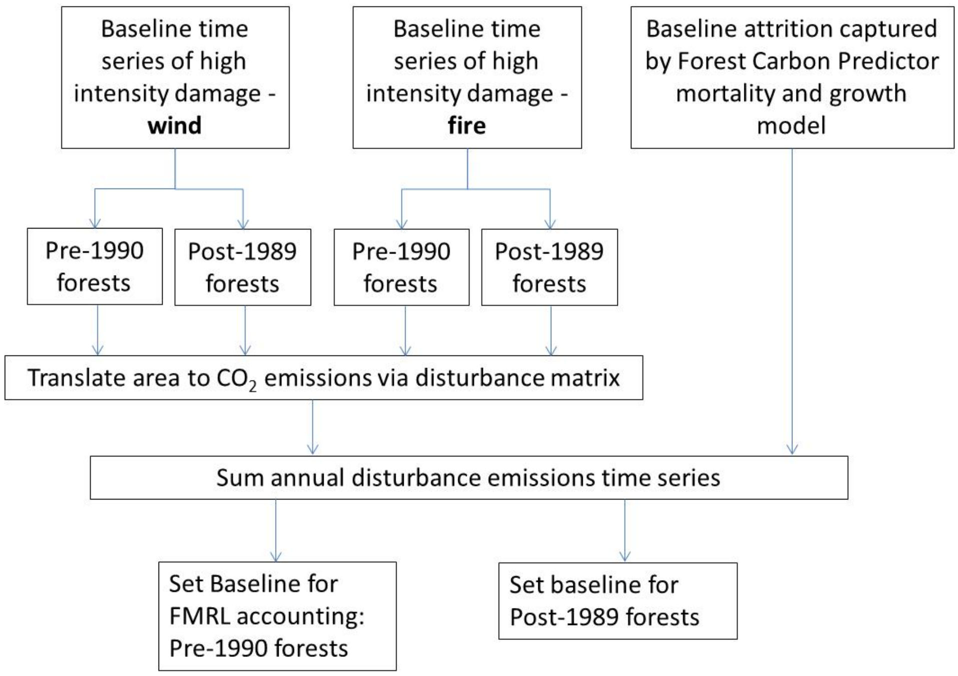

Figure 3 Estimating baseline natural disturbance in planted forests.

Table 3 Treatment of planted forest disturbance agents within Forest Management Reference Level (FMRL) accounting and natural disturbance baseline

Disturbance	Assumed included within FMRL (300 Index GM)	Baseline 1990–2009 additional catastrophic damage
Abiotic factors		
Wind	Attrition only	To be calculated
Fire	No	To be calculated
Other climatic (e.g. drought, snow)	Yes	Assume zero
Physiological needle blight	Yes	Assume zero
Volcanic eruptions	No	Assume zero
Earthquakes	No	Assume zero
Tsunamis	No	Assume zero
Landslides	No	Assume zero
Biotic factors		
Vertebrate pests	Yes	Assume zero
Invertebrate pests	Yes	Assume zero
Diseases	Yes	Assume zero
Invasive weeds	Yes	Assume zero

5.3.2 Catastrophic fire damage

IPCC default approach to estimating wildfire emissions

For greenhouse gas inventory reporting of wildfire emissions, New Zealand follows the IPCC "Good Practice Guidance" approach outlined in IPCC (2003), using a mixture of IPCC default factors and New Zealand country-specific factors. Data from the National Rural Fire Authority are used for area burnt and a New Zealand estimate is used for the mass of biomass available for combustion. This is based on a weighted emission factor calculated from the average biomass density for natural forest and the LUCAS national average yield tables for pre-1990 and post-1989 planted forests (MfE 2012).

Default IPCC values are used for the combustion factor and greenhouse gas emission factors. The IPCC default combustion factor for non-eucalyptus temperate forest is 0.45, referenced to Prasad et al. (2001)[8] and Robinson (1989). This is lower than the estimates for land-clearing burns in the same forest type (0.51) and post-logging slash burns (0.62). The post-logging slash burn estimate of 0.62 was also the average estimated by Robertson (1998) in New Zealand; this was one of four sources for the IPCC value. The IPCC default values for boreal forests are similar: 0.43 for crown fires, 0.15 for surface fires and 0.4 overall. IPCC default factors are used to convert the biomass burned to the amount of each greenhouse gas emitted.

A revised methodology was introduced by the 2006 Guidelines, using the equation in Box 1 and the emission factors in Box 2.

[8] This study is of questionable relevance – Prasad et al. (2001) studied three sites in tropical moist mixed secondary deciduous forests and estimated combustion factors ranging from 0.16 to 0.30.

Box 1. Equation for estimating greenhouse gas emission from fire IPCC (2006)

IPCC (2006) EQUATION 2.27: ESTIMATION OF GREENHOUSE GAS EMISSIONS FROM FIRE

$L_{fire} = A \cdot M_B \cdot C_f \cdot G_{ef} \cdot 10^{-3}$

Where:

L_{fire} = amount of greenhouse gas emissions from fire, tonnes of each GHG, e.g. CH_4, N_2O, etc.

A = area burnt, ha

M_B = mass of fuel available for combustion, tonnes ha^{-1}. This includes biomass, ground litter and dead wood.

C_f = combustion factor, dimensionless (default values in table 2.6, IPCC (2006))

G_{ef} = emission factor, g kg^{-1} dry matter burnt (default values in table 2.5, IPCC (2006))

Box 2. Emission factors for extra-tropical forest burning IPCC (2006)

Extra-tropical forest:

Emission factors (g kg^{-1} dry matter burnt) for various types of burning.
Values are means ± sd and are based on the comprehensive review by Andrea and Merlet (2001)

CO_2	CO	CH_4	N_2O	NO_x
1569 ± 131	107 ± 37	4.7 ± 1.9	0.26 ± 0.07	3.0 ± 1.4

Area damaged by wildfire, A

In general New Zealand does not have the type of fire ecology present in Australia, southern Europe or Western North America – the temperate maritime climate makes prolonged hot, dry spells less likely, dry lightning is uncommon and few species are adapted to require or cope with frequent fires. However, while indigenous forest in New Zealand is not as flammable as some counterparts overseas (Geddes 2005), the introduced conifers that dominate planted forests readily burn and mature stands of gorse (*Ulex europaeus*), which colonises marginal lands (including within and among planted forests), are a relatively high risk vegetation types in terms of flammability, fuel loads and the potential for extreme fire behaviour (M. Alexander, pers. comm.).

In 1919 the State Forest Service (later New Zealand Forest Service, NZFS) was established with the aims of sustained yield management of the natural forests and the establishment of plantations of exotic species. By 1936 planted forest area had increased from 77 000 ha in 1921 to 317 000 ha (MAF 2000). Afforestation rates were low in the 1940s and 1950s but increased again from the 1960s; by 1990 planted forest area had reached 1 261 000 ha, about half of which was managed by the State (MAF 2001). Throughout this period the protection of forests from fires was a major concern of the NZFS. Developments in plantation fire protection over this time have been described by Cameron et al. (2007), Pearce et al. (2008), Guild and Dudfield (2010) and in a comprehensive history of rural fire-fighting in New Zealand (Beaglehole 2012). A turning point was the prolonged drought in the summer of 1945/46 that led to a series of devastating fires across the North Island. The biggest of these was the Tahorakuri fire, which burnt over 30 000 ha. including 11 000 ha of plantation. Guild and Dudfield (2010) report total losses of State and private plantations to fires during that period of over 16 000 ha. One response was the first piece of legislation dedicated to rural fire – the Forest and Rural Fires Act 1947 (Cameron et al. 2007).

During the NZFS era to 1987, there was at least one major forest fire (500+ ha) each decade, with an average annual plantation loss of about 640 ha (or 0.16% of the total planted estate) (Cameron et al. 2007; Table 4). The NZFS was corporatised in 1987, with cutting rights progressively sold to private companies. The disestablishment of the NZFS – the de facto lead agency for forest fire management – coincided with a long dry period in Canterbury and several plantation forest fires including the Dunsandel fire in 1988 (Table 4). This provided the impetus for a review of rural fire fighting and the establishment of the National Rural Fire Authority (NRFA) (Beaglehole 2012).

Table 4 Significant plantation forest fire events 1940–2012 (from Cameron et al. 2007 and the NFRA database).

Year	Fire	Location	Forest type	Burnt area (ha)
1940	Eyrewell	Canterbury	Exotic plantation	469
1946	Tahorakuri	Taupo	Exotic plantation + scrub	30 738
1955	Balmoral	Canterbury	Exotic plantation	3152
1970	Mawhera	West Coast	Exotic plantation	400
1971	Slopedown	Southland	Exotic plantation	295
1972	Allanton	Otago	Exotic plantation	139
1972	Rankleburn	Southland	Exotic plantation	422
1973	Ashley	Canterbury	Exotic plantation	194
1973	Mohaka	Hawkes Bay	Exotic plantation	368
1975	Waimea	West Coast	Exotic plantation	370
1976	Hanmer	Canterbury	Exotic plantation	798
1977	Wairapukao	Bay of Plenty	Exotic plantation	432
1981	Hira	Nelson	Exotic plantation	1972
1988	Dunsandel	Canterbury	Exotic plantation	185
1994	Purakaunui	Otago	Exotic plantation	210
1995	Berwick	Otago	Exotic plantation +scrub	255
1996	Mohaka	Hawkes Bay	Exotic plantation	241
1997	Aupouri	Northland	NA	260
1997	Harakeke	Nelson	NA	532
1998	Bucklands Crossing	Otago	Exotic plantation +scrub	200
2002	Miners Road	Canterbury	Exotic plantation +grass	197
2004	Irvines	Nelson	Exotic plantation	200
2005	Mohaka	Hawkes Bay	Exotic plantation	240
2006	Maringi	Wairarapa	Exotic plantation	193
2007	Waipoua	Northland	Exotic plantation +wetland	224
2008	Para Rd	Marlborough	Exotic plantation	84
2009	Tadmor	Nelson	Exotic plantation	~600
2010 (Feb)	Mt Allen	Otago	Exotic plantation +scrub	710
2010 (Dec)	Mt Allen	Otago	Exotic plantation	95
2010	Glenhope	Nelson	NA	200
2010	Poutu	Northland	Exotic plantation +scrub	115
2011	White Cliffs, Horeke	Northland	Exotic plantation +scrub	345

The area of plantations in New Zealand continued to increase after 1990, with the mid-1990s seeing record annual planting rates. By April 2010 there were 1.7 million hectares, about one-third of which had been established since 1990 (MAF 2010). The average annual area lost to wildfires has been lower over this period, both as a percentage of total plantation area and in absolute terms. The NRFA established a database to record vegetation fire information (Table 4; Appendix III) – this database provides the forest fire data used as the basis for national greenhouse gas inventory reporting under the United Nations Framework Convention on Climate Change (e.g. MfE 2012) and Montreal Process reporting (MAF 2009).

Anderson et al. (2008) present an analysis of vegetation fires data from the NRFA database 1991/92 to 2006/07 (years ending 30 April). Forests (including plantations, natural forests and other vegetation dominated by trees) made up just 6% of the vegetation area burned. The average annual area of forest burned was 386 ha, ranging from 119 ha in 1991/92 to 1399 ha in 1997/98. Although there was a clear increasing trend in the number of fires each year, there was no significant trend in area burnt. However, the area of forest burnt in each of the last 5 years has been greater than the average reported by Anderson et al. (2008); the average area of forest burnt annually since 1991/92 is now 482 ha.

Anderson et al. (2008) caution that the data in the NRFA database are not of a high standard. The system relies on annual returns provided by the individual Rural Fire Authorities, of which there were 71 in 2010. Not all returns are provided by all RFAs each year, and not all returns include all the information required. The reported areas themselves may not be accurate, as "filling in the paperwork" may not be regarded as a high priority compared with the difficult and dangerous work of protecting lives and property.

The database provides little of the information required to accurately calculate emissions from forest sub-categories within the annual greenhouse gas inventory. Plantation forest fires were not distinguished from natural forest fires until 2008/09. For national greenhouse gas inventory reporting purposes it has been assumed that 87.5% of forest fires occurred in plantations, based on the observation from Fogarty and Pearce (1995) that about 350 ha of plantation burned annually while the total area of forest burnt averaged about 400 ha. In the last 3 years (for which plantation fire data are available separately) the plantation share was 37%, 94% and 94% – an average of 75%. The estimated annual wildfire area in planted forests for the period 1990–2010 is shown in Figure 4. The original data were for years ending 30 April and covered the years 1991/92 to 2010/11. Values have been converted to calendar years (values for 1990 and 1991 incorporate averaging). The mean area of planted forest burnt annually was estimated to be 412 ha.

The data cannot be used to distinguish between pre-1990 and post-1989 planted forests[9] because the age and former land use is not recorded. In future this may be available through the Emissions Trading System (ETS), together with an estimate of emissions. Allocation of wildfire area to pre-1990 and post-1989 planted forests was made in proportion to their contribution to total planted forest each year. This was estimated using LUCAS data supplied by MfE (Nigel Searles, MfE, pers. comm.).

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[9] Post-1989 planted forests are those that meet the Kyoto Protocol accounting requirements for Afforestation/Reforestation under Article 3.3; Pre-1990 forests are accounted for as Forest Management under article 3.4.

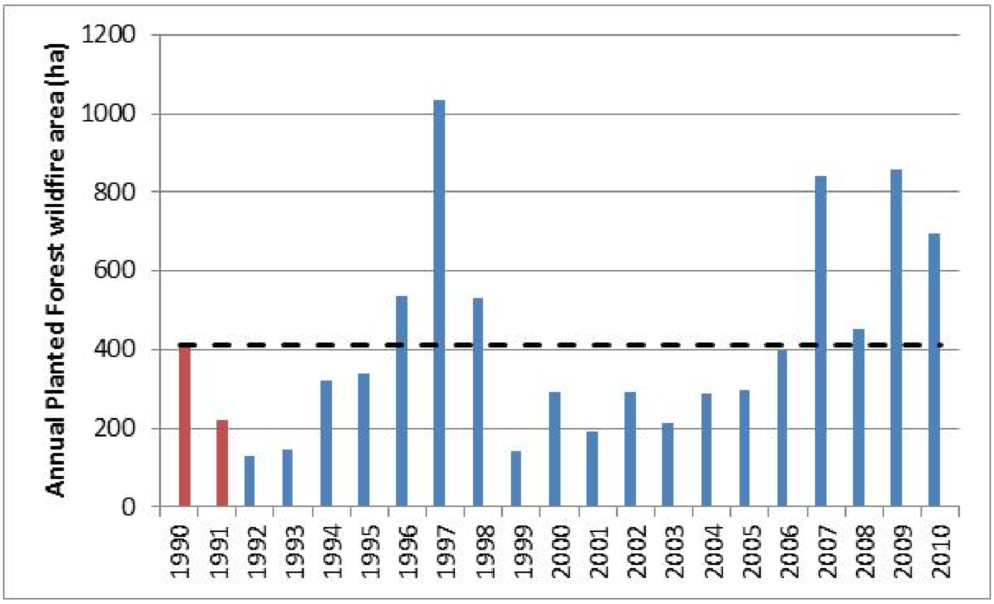

Figure 4 Annual planted forest wildfire area from National Rural Fire Authority data. Dashed line represents mean value of 412 ha per year.

Mass of fuel available for combustion, M_B

The mass of fuel available for combustion (M_B) represents the total amount of potentially flammable biomass (per ha), and includes live biomass, ground litter and dead wood. There is no specific information in the NRFA database to help determine the biomass present before wildfires (see Appendix III for a summary of historical fire records for planted forests including description of salvage logging). Yield tables for national planted forest from LUCAS were therefore used to determine the biomass present in the above-ground biomass, below-ground biomass, dead-wood and litter pools at the average age each of planted forest sub-category. Thus it was assumed each year that wildfires would affect forests at the average age carrying the carbon stock expected for that age. Linear interpolation was used to estimate yield table values between whole-number ages. Average forest carbon per hectare is initially low in post-1989 forests because the forests are so young – high annual planting rates through the mid-1990s keep the average age down but the age and hence carbon stock then increases because the planting rate is not maintained (Figure 5). Pre-1990 forests may include residues from previous rotations, but these have not been included.

For the purpose of determining baseline emissions in pre-1990 and post-1989 forests, it was assumed that the risk from wildfires was not age-dependent. Age-class data for planted forests was obtained from LUCAS (Nigel Searles, MfE, pers. comm.) and used to determine the average forest age each year from 1990 for pre-1990 and post-1989 forests separately (Figure 6).

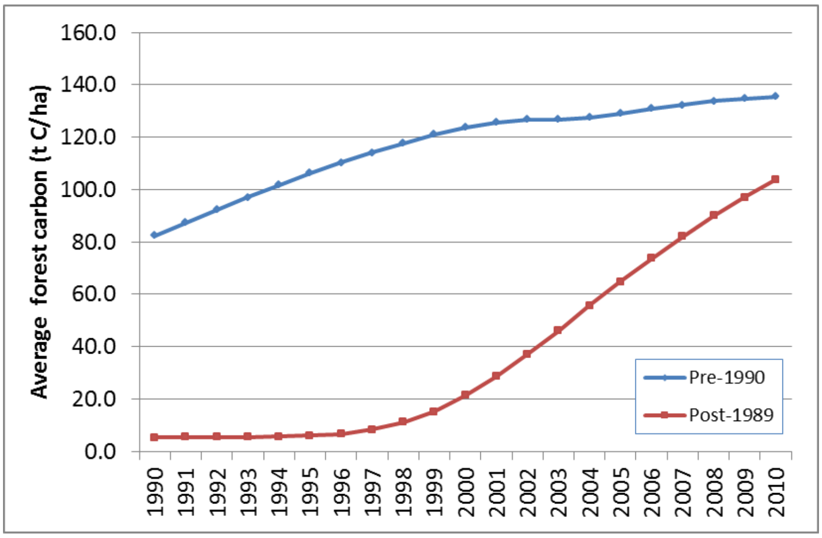

Figure 5 Forest carbon (excluding mineral soil) at the mean forest age for pre-1990 and post-1989 planted forests.

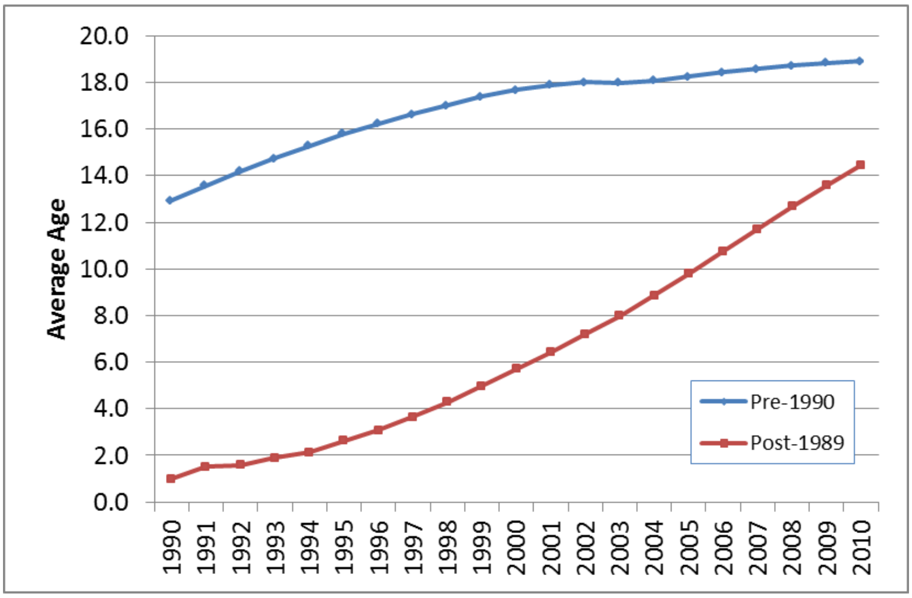

Figure 6 Average age of pre-1990 and post-1989 planted forests.

Combustion factor (Cf)

The impacts of a wildfire on forest biomass can be expressed by a disturbance matrix that reflects transfers from (a) live biomass pools to dead organic matter pools; (b) all pools to the atmosphere through combustion; and (c) the above-ground biomass pool to harvested wood products pool through salvage logging. The combustion factor (*Cf*) represents the proportion of mass of fuel available (M_B) that is combusted during a fire.

Fuel consumption has been shown to be highly variable in experimental burns, with fuel characteristics (size, distribution and total amount) and fire weather affecting fuel moisture and fire behaviour (de Groot et al. 2007). A low-intensity ground fire may burn only litter while a high-intensity fire with crowning will be far more destructive. There is no indication in the NRFA database of the proportion of biomass oxidised or whether the fire was a stand replacement event. The management response may be to clear and replant (with or without salvage logging) or to allow stands to recover and grow on, perhaps at a reduced stocking.

Wildfire reviews summarised in section 4.3.5 suggest that a relatively high proportion of litter and a relatively low proportion of above-ground biomass can be expected to be consumed in most fires. Forest Carbon Predictor simulations show that in first-rotation stands, litter increases from zero at planting to about 18–19% of combined above-ground stand biomass and dead organic matter at mid-rotation (after pruning and thinning) and then declines to about 3% by the end of a 28-year rotation. If stem litter, all needles and dead branches on standing trees are added to needle and branch litter, the proportion is constant at about 35–40% for the first half of the rotation, then declines to about 14% at maturity. A default combustion factor of 0.45 implies that some stem, bark, and live branches will also be consumed by fire. Given the lack of dead wood and litter at the very young average-age for the post-1989 forests, almost half of all above-ground biomass (including non-planted vegetation) would have to be consumed in order to achieve a combustion factor of 45%. This portion then decreases to about 15% as more litter and dead wood is generated through silviculture, before increasing again as the dead organic matter decays. In mature stands about 30% of stem and bark would need to be consumed on top of all needles, branches and dead organic matter. De Groot et al. (2007) report that overstorey fuel consumption in large Canadian fires is typically limited to 0–25% of the total above-ground fuel load unless the trees are of very small diameter.

In summary, the combustion factor in mature stands and in surface fires is likely to be much less than the default of 0.45. In newly established stands the default value could be exceeded, as stem and branch diameters are smaller, non-planted weed species make up a larger proportion of stand biomass, and in second-rotation forests there may be significant amounts of harvest residues available to burn. In the absence of any definitive country-specific value, the default value has been retained and applied to the three above-ground pools: above-ground biomass, dead wood and litter.

Economic analyses of wild fire impacts on planted forests provide another potential source of information on combustion and salvage. Cooper and Ashley-Jones (1987) and Cameron et al. (2007) both report costs of fire administration, prevention, detection and control but not the potential or actual losses due to fires. Wu et al. (2009) attempted to estimate the economic cost of all wildfires in New Zealand. They included pre-suppression costs, suppression costs and also the direct and indirect costs of the fires themselves. The short-term economic loss is affected by the degree to which timber can be salvaged. This also affects the estimation of the background level of emissions since the carbon in logs removed through salvage harvesting is excluded from accounting. Salvage harvesting is more expensive than normal harvesting and there may not be markets for smaller logs. However, if a stand is too badly damaged to recover, the priority is usually to clear and replant the site as soon as it is safe to do so. The proportion salvaged depends on the marketability of logs, which will be driven by factors affecting log quality and dimensions such as age, species, and silviculture.

Manley (2001) modelled the effects of disturbance on forest value and assumed that stands younger than 19 and pulp logs would not be salvaged, but 90% of sawlog volume would be salvaged from older stands. In the 1995 Berwick fire stands older than age 12 were salvaged (Fogarty et al. 1996). In deriving baseline emissions due to natural disturbance it was assumed that there would be no salvage in post-1989 forest fires between 1990 and 2009 (Table 5). In pre-1990 forest wildfires it was assumed that half of the unburned above-ground biomass would be salvaged.

Decay of post-disturbance residues

Biomass not salvaged or consumed by fire is assumed to decay. In practice some biomass will be converted to charcoal which does not decay rapidly compared to dead wood, but a lack of data on rates of formation and turnover precluded development of a default methodology (IPCC 2006) so this has been ignored.

Post-disturbance residues have been modelled using the first-order-decay approach presented by Pingoud and Wagner (2006) and adopted by the IPCC (2006)for estimating changes in the pool of harvested wood products. The formula assumes continuous inflow into the dead organic matter pool and continuous outflow through decomposition (Box 3). Decay constants were obtained from work in New Zealand *Pinus radiata* plantations on stems (Garrett et al. 2010) and roots (Garrett et al. 2012). The stem rate was also applied to branches as rates were not significantly different (Table 6). The litter decay rate was derived from the rate determined by Beets et al. (1999).

Regional mean annual temperatures (MAT) were estimated by overlaying the NIWA GIS layer for MAT with the LUCAS planted forest land use map, within the amalgamated NEFD regions for which wind and fire area data were summarised (Table 7).

Table 5 Disturbance matrices for wildfire impacts on planted forest carbon pools

POST-1989

To → From ↓	Above-ground biomass	Below-ground biomass	Dead wood	Litter	Soil organic matter	Harvested wood products	Atmosphere (burning)	Sum of row (must equal 1)
Above-ground biomass			0.55	0	0	0	0.45	1
Below-ground biomass		0	1	0	0		0	1
Dead wood			0.55	0	0	0	0.45	1
Litter				0.55	0		0.45	1
Soil organic matter					1		0	1

PRE-1990

To → From ↓	Above-ground biomass	Below-ground biomass	Dead wood	Litter	Soil organic matter	Harvested wood products	Atmosphere (burning)	Sum of row (must equal 1)
Above-ground biomass			0.275	0	0	0.275	0.45	1
Below-ground biomass		0	1	0	0		0	1
Dead wood			0.55	0	0	0	0.45	1
Litter				0.55	0		0.45	1
Soil organic matter					1		0	1

Box 3. Decay formula used for unburnt stocks

Formula to estimate stocks of decaying dead organic matter from previous stocks and current year inflow

(a) $C(i+1) = e^{-k} . C(i) + [(1-e^{-k})/k] . Inflow(i)$

(b) $Emissions(i) = C(i+1) - C(i) – Inflow(i)$

Where

C(i)	= carbon stock at the start of year i. C(1990) = 0.
k	= decay constant of first order decay
Inflow(i)	= additions to the carbon stock during year i.

Table 6 Decay constants (k) for decaying post-disturbance residues

Pool	k*	Source
Dead wood	$0.0376 \times e^{(0.093 \times MAT)}$	Garrett et al. 2010 (Stem)
BGB	$0.0684 \times e^{(0.093 \times MAT)}$	Garrett et al. 2012 (Root)
Litter	$0.081 \times e^{(0.093 \times MAT)}$	Beets et al. 1999 (Litter)

* MAT = mean annual temperature

Table 7 Regional MAT for use in decay models

Region	Mean MAT
Northland	15.1
Auckland	14.6
CNI	12
East Coast/Hawke’s Bay	12.5
SNI	12.1
Nelson/Marlborough	11.1
West Coast	11.1
Canterbury	10.6
Otago/Southland	9.2

5.3.3 Catastrophic wind damage

Area damaged by wind

The approach used to estimate emissions due to wind damage is the same as that used for wildfires, based on an estimate of the area disturbed, a disturbance matrix and post-disturbance decay. As a long narrow island nation isolated from large land masses, New Zealand is regularly subject to strong winds from all directions. Damage to forests is usually associated with ex-tropical cyclones (mainly North Island but also further south) or topographically enhanced winds (e.g. on the Canterbury Plains) (Martin & Ogden 2006).

Trees are most likely to uproot or break when they are subjected to sudden changes in wind load to which they are not acclimatised (Gardiner et al. 2000). In New Zealand planted forests this is commonly associated with thinning or the harvest of adjacent stands (Somerville 1995). Stem breakage and wind throw affect wood flows, increasing short-term supply through salvage but reducing sustainable yield. Net revenues from salvage harvesting are lower because costs are higher, yields lower, and quality may be lower due to the harvest of suboptimal age classes and fungal attack (McFarlane et al. 2002). Woody debris from wind throw also creates favourable conditions for insect populations and wildfires. Less visible but of greater economic consequence are the negative impacts of wind on stem form and wood qualities and the opportunity cost from not implementing otherwise economically optimal thinning regimes because of the need to maintain stand wind-firmness. A survey found that plantation forest owners regarded wind as a more serious problem than fire or existing pests and diseases[10] (Cameron et al. 2007).

Moore et al. (2012) compiled available data for wind damage in planted forests from as early as 1945 to 2009. This start date coincides with a storm in Canterbury that damaged 1694 ha, and until the later 1970s wind damage was viewed as a problem confined largely to Canterbury and some other parts of the South Island. However, this view changed after two ex-tropical cyclones in 1982 and 1986 caused severe damage in planted forests in the Central North Island and Nelson (Moore et al. 2012; Table 8). The database contains 78 records of damage from 62 storm events. The median area damaged in an individual event was 90 ha, while in eight events the area damaged was greater than 1000 ha (Table 8). The total reported area damaged was 63 200 ha or about 970 ha per annum.

Table 8 Significant (>1000 ha) planted forest wind damage events 1940–2009

Year	Event	Location	Area affected (ha)	Source
1945		Canterbury	1694	Prior 1959; Moore et al. 2012
1964		Canterbury	2917	Moore et al. 2012
1968	Cyclone Giselle (Wahine storm)	Nelson, Canterbury	1963	Irvine 1970
1975	Canterbury wind storm, gusts to 170 km per hour	Canterbury	11 000	Wilson 1976; Moore et al. 2012
1982	Cyclone Bernie, ex-tropical cyclone, winds > 130 km per hour	Central North Island	12 000	Littlejohn 1984, Moore et al. 2012
1988	Cyclone Bola, Ex-tropical cyclone, winds >100 km per hour, heavy rain	North Island, Nelson	26 382	Somerville et al. 1989; Moore et al. 2012
2004 (Feb)	SW storm	SNI, West Coast	2648	Moore et al. 2012
2004 (Oct)	SW storm	Nelson & Marlborough	1213	Moore et al. 2012
2008	NE cyclone	CNI, SNI, Nelson, Marlborough, west Coast	2592	Moore et al. 2012

[10] Only the introduction of pests and diseases not currently found in New Zealand was ranked ahead of wind.

Records of wind damage were not available in all regions for the same period, so start dates for the regional time series varied from 1945 to 2003. This partly reflects the distribution of forests over time and the occurrence of significant wind damage events.

Nationally an average of 0.21% of net stocked area was damaged by wind each year over the entire 65-year period. The rate since 1990 has been 0.07% of net stocked area. The largest event – Cyclone Bola in 1988 – accounts for 41% of the area damaged, while the four most damaging events account for 73%. The pre- and post-1990 time series also show regional differences. For example, there has been more damage recorded in Nelson since 1990 than in the 40 years prior to 1990, while only 594 ha has been damaged in Canterbury since 1990 compared with 16 650 ha in the 46 years before 1990 (Moore et al. 2012).

The time series of wind damage areas analysed by Moore et al. (2012) can be assumed to have captured the major events over that time, but there are undoubtedly many smaller events that have been recorded in addition to the attritional wind damage that is likely to be captured within the permanent sample plots used to develop growth models. The long return period for major events and the short time series available make analysis difficult. If data for the 1990–2009 period are used to derive the baseline for natural disturbance, the most damaging wind event in New Zealand's plantation history (Cyclone Bola) will miss out on inclusion by just 2 years. This can be justified when setting an expectation for 2013–2020 because an event of the magnitude of Cyclone Bola was estimated to have a return period of 100 years. The return period for an annual total of 5000 ha damaged was estimated to be 20 years (Moore et al. 2012). The average annual area damaged over the 1990–2009 period was 524 ha (Figure 7).

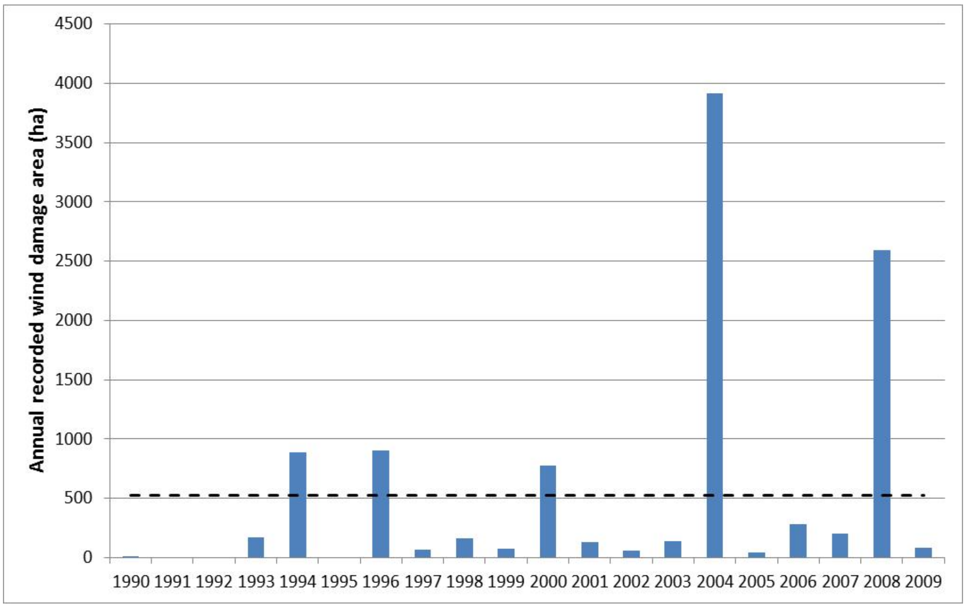

Figure 5 Area of annual recorded wind damage in New Zealand planted forests 1990–2009. Dashed line indicates average damaged area: 524 ha per year.

Pre-disturbance biomass

Capturing the carbon emissions associated with wind-damage events is difficult. Moore et al. (2012) caution that information on volume or biomass lost due to wind damage is difficult to obtain because commercial forest owners and managers are sensitive about releasing this information. Although we intended to gather this information from forest owners, initial discussions suggested that the level of emissions from this source were not significant enough to warrant this degree of attention. The same pre-disturbance carbon stock was assumed as for wildfire, equal to the stock at the average forest age each year.

Disturbance matrix and salvage

Wood from damaged stands can generally be salvaged. Smaller areas can be subsumed within the normal harvesting activity while large damaged areas can be salvaged over several years. In the 1945 wind-throw event in Canterbury, the underdeveloped infrastructure meant that not all timber could be salvaged. Thirty years later, log exports to Japan and China and stockpiling under sprinklers were used to maximise recovery from stands damaged in the 1975 storm (Turner 1989). Salvage harvesting became the normal operation for one Canterbury forest owner, with 90% of their harvest from Canterbury plains forests being from post-wind-throw salvage operations (Studholme 1995). In stochastic wind-throw models run for this resource, Bown and Bilek (1997) assumed that there would be no price or cost penalty in salvage harvesting but that the timber volume recovered would be reduced by 20%. It was assumed that timber could be recovered for up to 5 years.

Everham and Brokaw (1996) criticised the inconsistency in recording wind damage and classified attempts to quantify damage into six types: (1) stem damage, (2) branch damage, (3) canopy damage, (4) mortality, (5) volume or mass changes, and (6) classification categories. Volume loss is most commonly reported for plantation forests. Following a 1982 storm event in Kaingaroa Forest it was determined that there was a 7% reduction in recoverable volume because some trees could not be accessed, with a further 6% reduction due to stem breakage (Somerville 1995). Nieuwenhuis and Fitzpatrick (2002) estimated a volume loss of 27% for the broken stems in a study area of wind-thrown Sitka spruce in Ireland in 1998. Since only 13% of stems were broken, the overall volume loss was just 2.6%.

Salvage is least likely in very young stands and the proportion salvaged is likely to be lowest if very large areas have been damaged in the same event (due to logistics) or if small scattered areas are damaged (due to the lack of economies of scale). It can be assumed that the recorded events in the time series are serious enough to warrant stand-replacement. This means that the site will be cleared to the extent needed to ensure successful crop re-establishment.

In a normal harvesting event, about 70% of above-ground biomass is recovered as harvested wood products (MfE 2012). This was reduced to 60% for wind-throw salvage in pre-1990 forests (Table 9). In post-1989 forests it was assumed that there was no salvage.

Decay of post-disturbance residues

It was assumed that damaged stands would be replaced. Unsalvaged dead organic matter in damaged stands was assumed to decay at the same rates used for post-wildfire decay.

Table 9 Disturbance matrix for impacts of wind damage on planted forest carbon pools

Post-1989

To → From ↓	Above-ground biomass	Below-ground biomass	Dead wood	Litter	Soil organic matter	Harvested wood products	Atmosphere (burning)	Sum of row (must equal 1)
Above-ground biomass			1	0	0	0	0	1
Below-ground biomass		0	1	0	0		0	1
Dead wood			1	0	0	0	0	1
Litter				1	0		0	1
Soil organic matter					1		0	1

PRE-1990

To → From ↓	Above-ground biomass	Below-ground biomass	Dead wood	Litter	Soil organic matter	Harvested wood products	Atmosphere (burning)	Sum of row (must equal 1)
Above-ground biomass			0.4	0	0	0.6	0	1
Below-ground biomass		0	1	0	0		0	1
Dead wood			1	0	0	0	0	1
Litter				1	0		0	1
Soil organic matter					1		0	1

5.3.4 Summary of model used to estimate emissions from wildfires and wind

The model developed to estimate baseline emissions from plantation wildfires follows the IPCC approach from the 2006 Guidelines (IPCC 2006). The wind emissions model was based on the fire model but with biomass burning excluded. Inputs and assumptions were:

- Plantation forest wildfire area by region from the NRFA database. Before 2008/09 it was assumed that 87.5% of burned forest was planted forest. Wind damage areas from a database held by Scion.
- An optional area adjustment can be applied by region if there are grounds to increase or decrease the area recorded. This factor has been retained at 1.
- Age-class-distribution information from LUCAS was used to determine the proportion of the total planted forest in post-1989 and pre-1990 forest for each year from 1990 to 2009. These proportions were used to allocate disturbance area within each region to forest sub-category (i.e. it was assumed that wildfires and wind damage affect pre-1990 and post-1989 forests in proportion to their contribution to the national resource, and the national proportion of post-1989 forest applies within each region[11].
- Pre-disturbance biomass by the four pools (above-ground biomass, below-ground biomass, dead wood and litter) was derived from the LUCAS planted forest yield table values at the average forest age.
- A simple disturbance matrix was used to transfer biomass to the dead organic matter pools and to the harvested wood products pool (salvage) where appropriate. Mineral soil carbon was assumed to be unaffected by disturbance.
- The proportion of above-ground biomass and dead wood oxidised in fires was set at 0.45 (IPCC default).
- Material that was not consumed by fire or salvaged was assumed to decay. Regional decay rates for stems and branches (Garrett et al. 2010) and roots (Garrett et al. 2012) were used (Box 4). No allowance was made for reduced decay of charcoal.
- CO_2, CH_4 and N_2O emissions from wildfires were calculated using IPCC default emission factors (IPCC 2006).
- The model assumes that all fires and catastrophic wind damage result in stand replacement – all biomass is either oxidised, removed from the forest or left to decay. No ongoing mortality or losses due to subsequent insect attack or fires were included.

[11] In fact some regions have a greater proportion of post-1989 forest than others, but the data was not readily available and affects only the decay rate of unburned dead organic matter.

6 Results

6.1 Natural forest carbon emissions

6.1.1 CWD decay modelling

In the BRT model including all predictor variables (the "full" model), species identity had the strongest influence, followed by CWD type, time since death and diameter (Table 10). Figure A1.2 (Appendix) confirms the overriding influence of species identity, with its partial-contribution plot spanning a much larger range than any of the other variables. In the full model the strongest interactions were between species identity and either CWD type (50% of all regression tree branches) or time since death (20% of all regression tree branches). This suggests that species identity strongly moderates the effect of CWD type and time since death on decay, and that climate variables are relatively unimportant.

The best BRT model for CWD wood density (as a proportion of live wood density) included species identity, CWD type, trunk diameter and time since death. However, because the fate of dead trees (e.g. fallen or standing dead) was not recorded in our data, we were unable to use CWD type in predicting wood density of trees dying between plot measurements. The next best model included species identity, trunk diameter and time since death. Removing CWD type caused a significant decrease the cross-validated correlation between fitted and observed values, as did the subsequent removal of species identity. Recording the type of CWD in forest monitoring plots (as done in LUCAS natural forest plots) could considerably increase the accuracy with which we can predict C losses to CWD decay.

Table 5 Contribution of predictor variables in the full boosted regression tree model. Contribution indicates the percentage of regression tree "branches" involving each variable

Variable	Contribution
Species Identity	45.90
CWD type	19.24
Trunk diameter	17.50
Age	14.32
Temperature	1.63
Rainfall	1.41

6.1.2 Carbon losses from disturbed plots

We classified 622 (20.2 %) of the 3077 repeatedly surveyed plots as having been disturbed during their measurement interval, based on a decrease in mean stem diameter and decrease in total stand basal area. The median time interval between measurement periods was 10.1 years, the mean annual probability of disturbance of 0.0192, and the average disturbance recurrence frequency was 62.7 years. There was significant variability in the number of plots situated within each region and in the mean annual probability of disturbance among regions (Table 11). Northland, West Coast, Waikato, Canterbury, Hawke's Bay and Southland had above-average annual probabilities of disturbance.

There was a net loss of carbon (adjusted for regional sampling bias) in disturbed plots of −2.43 Mg CO_2-e ha^{-1} yr^{-1}, compared with a net gain of 3.81 Mg CO_2-e ha^{-1} yr^{-1} in non-disturbed plots (Tables 11 & 12). This net loss was due to disturbed plots having higher carbon loss due to tree mortality and subsequent decay than non-disturbed plots. Carbon gain due to growth of live trees was also lower in the disturbed plots (Table 12).

Multiplying the mean annual probability of disturbance (0.0192) by the total pre-1990 natural forest area (7 846 605 ha) gives an estimated area affected annually by low-intensity natural disturbances of 150 545 ha. If the carbon change estimate for non-disturbed plots is used as a baseline, the average net emission from the disturbed plots is 6.24 Mg CO_2-e ha^{-1} yr^{-1}. Multiplying these estimates together gives a total annual carbon emission from low-intensity plot disturbance of 939 401 Mg CO_2-e yr^{-1}.

Table 6 Regional estimates of annual probability of disturbance carbon stock change in disturbed natural forest plots

Region	Total natural forest area[1] (ha)	*N* plots	*N* disturbed plots	Annual probability of disturbance[2]	Annual area disturbed (ha yr^{-1})	C change from mortality and CWD decay (Mg CO_2-e $ha^{-1} yr^{-1}$)	C change from growth of live stems[3] (Mg CO_2-e $ha^{-1} yr^{-1}$)	Net C flux (Mg CO_2-e $ha^{-1} yr^{-1}$)	Total net C flux (Mg CO_2-e yr^{-1})
Northland	384 507	35	17	0.0598	22 994	−7.77 (4.21)	3.98 (0.36)	−3.79 (4.38)	−87147
Auckland	142 607	46	3	0.00744	1062	−1.67 (0.45)	4.14 (0.44)	2.47 (0.35)	2623
Waikato	610 155	185	52	0.0218	13 271	−7.07 (0.87)	4.60 (0.68)	−2.47 (1.02)	−32779
Bay of Plenty	576 246	114	19	0.0131	7574	−5.58 (2.44)	3.87 (1.92)	−1.71 (2.56)	−12952
Gisborne[5]	229 621	7	0	NA	NA	NA	NA	NA	NA
Hawke's Bay	414 338	101	23	0.0192	7951	−6.00 (1.30)	5.63 (0.67)	−0.37 (1.34)	−2942
Manawatu	571 910	132	18	0.0140	7974	−4.74 (2.18)	5.97 (2.31)	1.23 (0.79)	9808
Taranaki	281 747	15	2	0.00725	2043	−13.74 (12.49)	2.30 (1.23)	−11.44 (13.72)	−23372
Wellington	287 258	199	22	0.00931	2675	−8.30 (1.99)	0.57 (1.46)	−7.73 (2.39)	−20678
Tasman	586 870	135	17	0.0114	6682	−6.24 (1.67)	4.25 (0.67)	−1.99 (1.42)	−13297
Nelson[5]	20 229	1	0	NA	NA	NA	NA	NA	NA
Marlborough	313 131	371	33	0.00762	2385	−5.82 (0.85)	3.40 (0.51)	−2.42 (1.02)	−5772
West Coast	1 463 320	251	88	0.0284	41 629	−7.52 (1.11)	3.77 (0.69)	−3.75 (1.25)	−156109
Canterbury	405 208	798	190	0.0203	8241	−4.22 (0.28)	4.15 (0.16)	−0.07 (0.33)	−577
Otago	249 351	122	14	0.00965	2406	−4.89 (1.49)	4.64 (1.01)	−0.25 (1.80)	−602
Southland	1 310 109	565	124	0.0180	23 658	−5.08 (0.63)	2.72 (0.46)	−2.36 (0.70)	−55833
North Island[4]	3 498 389	834	156	0.0187	65 543	−6.35 (2.71)	3.92 (1.17)	−2.43 (2.68)	−159269
South Island[4]	4 348 217	2243	466	0.0195	85 002	−5.99 (0.96)	3.56 (0.57)	−2.43 (1.03)	−206555
All New Zealand[4]	7 846 605	3077	622	0.0192	150 545	−6.15 (1.74)	3.72 (0.84)	−2.43 (1.77)	−365824

[1] Based on LUCAS 1990–2008 land use map class 71, pre-1990 natural forest.
[2] Estimate accounts for sample interval variability among plots.
[3] Includes carbon gain from recruitment.
[4] Area-weighted average carbon fluxes calculated using equation 1.
[5] Gisborne and Nelson regions did not have sufficient number of plots to gain a reliable estimate of the area disturbed

Table 7 Average carbon stock changes (and standard errors) in disturbed and non-disturbed natural forest plots, and across all plots in the dataset

	Disturbed plots	Non-disturbed plots	All plots
N plots	622	2455	3077
Initial live stem carbon stock[1] (Mg CO_2-e ha^{-1})	502.7 (62.6)	508.6 (29.6)	511.4 (26.6)
Carbon fluxes[1] (Mg ha^{-1} yr^{-1})			
Growth	2.71 (0.73)	4.19 (0.48)	3.90 (0.42)
Recruitment	1.01 (0.40)	1.69 (0.39)	1.54 (0.33)
Mortality	−11.83 (3.49)	−3.81 (0.64)	−5.64 (1.00)
CWD[2]	5.68 (1.75)	1.72 (0.31)	2.64 (0.51)
Net carbon flux	−2.43 (1.77)	3.81 (0.58)	2.43 (0.68)

[1] National carbon stock and carbon fluxes are area-weighted averages calculated using equation 1.

[2] CWD residue from trees that died during the study period; does not include flux from legacy CWD that was present before the start of the study period which is assumed to be constant across all plots.

6.1.3 Carbon losses from landslides 1990–2008

The total area of natural forest affected by landslides during the period 1990–2008 was 4529 ha (0.06% of the total forest area), or 252 ha yr^{-1} on average (Table 13). Landslides were smaller on average in the North Island (0.8 ha) than the South Island (1.8 ha). Most of the landslides occurred in Southland (44% of total area), followed by the West Coast (17%) and Tasman (14%). The high occurrence of landslides in Southland was probably due to the contribution of Fiordland, with its combination of steep geography, large forest area, and the occurrence of significant disturbance events (e.g. 6.8 and 7.2 magnitude earthquakes in Fiordland in1993 and 2003). It was not possible to assign a specific year to each landslide event, so the inter-annual variability is unknown.

The total live carbon stock in areas affected by landslides over the period 1990–2008 was 2 718 891 Mg CO_2-e, or 151 049 Mg CO_2-e $year^{-1}$ on average (Table 13). This provides an upper estimate of carbon loss, assuming that all live stem carbon was emitted to the atmosphere following the landslide occurring. Accounting for carbon stored as CWD following the landslide event, and its subsequent decay, we estimate carbon emissions from large-scale landslides of 1 474 833 Mg CO_2-e over the period 1990–2008, or 81 935 Mg CO_2-e yr^{-1} on average. The average emissions per ha disturbed by landslides was 19.24 Mg CO_2-e ha^{-1} yr^{-1}.

None of the geo-referenced locations of the NVS plots occurred within the mapped landslide areas. This suggests that carbon emissions from landslides estimated through remote sensing can be treated as independent for carbon emissions from plot-level disturbances. Adding the two emission sources together gives a total of 1 021 336 Mg CO_2-e yr^{-1} emissions from all sources of natural disturbance in New Zealand's pre-1990 natural forest. For comparison, this value represents about 4% of the annual net removals by planted forests.

Table 8 Area of natural forest affected by landslides occurring between 1990 and 2008 and their associated carbon (C) emissions

Region	Total natural forest area[1] (ha)	Area checked[2] (%)	Identified number of landslides	Identified landslide area (ha)	Mean area per landslide (ha)	Est. total landslide area (ha)	Total live C affected[3] (Mg CO_2-e)	Total C emissions[4] (Mg CO_2-e)	Annual C emissions[4] (Mg CO_2-e yr^{-1})
Northland	384 507	31	2	1.3	0.6	4.1	1454	789	44
Auckland	142 607	81	1	0.8	0.8	1.0	276	150	8
Waikato	610 155	85	52	46.0	0.9	53.8	25 011	13 567	754
Bay of Plenty	576 246	88	122	110.9	0.9	125.4	64 145	34 795	1933
Gisborne	229 621	97	54	53.3	1.0	55.2	21 623	11 729	652
Hawke's Bay	414 338	98	36	38.0	1.1	38.9	8947	4853	270
Manawatu	571 910	71	246	214.3	0.9	300.5	124 531	67 550	3753
Taranaki	281 747	71	270	261.3	1.0	366.8	175 916	95 424	5301
Wellington	287 258	74	12	11.9	1.0	16.0	5320	2886	160
Tasman	586 870	30	94	193.5	2.1	639.8	415 719	225 502	12 528
Nelson	20 229	67	0	0.0	0.0	0.0	0	0	0
Marlborough	313 131	91	3	7.3	2.4	8.0	1506	817	45
West Coast	1 463 320	40	148	313.7	2.1	788.2	452 312	245 352	13 631
Canterbury	405 208	59	14	14.6	1.0	24.7	11 372	6169	343
Otago	249 351	85	19	23.4	1.2	27.5	9561	5186	288
Southland	1 310 109	80	957	1662.78	1.7	2079.3	1 401 198	760 065	42 226
North Island	3 498 389	78	795	738	0.9	962	427 224	231 743	12 875
South Island	4 348 217	59	1235	2215	1.8	3568	2 291 667	1 243 090	69 061
All NZ	7 846 605	67	2030	2953	1.5	4529	2 718 891	1 474 833	81 935

[1] Based on LCBD2 natural forest and shrubland classes.
[2] Area checked is <100% due to cloud cover on the imagery used.
[3] Obtained by intersecting mapped landslide areas with the surface of live stem carbon stock (Mason et al. 2012); does not include existing CWD stocks.
[4] Assuming no regrowth of live biomass and an average % total carbon lost due to CWD decay of 3.01% per year, calculated from CWD model using NVS plot data.

6.1.4 Post-1989 natural forest carbon emissions

The average annual emissions due to both low intensity disturbance and landslides in pre-1990 natural forest were equivalent to 0.025% of the total pre-1990 natural forest carbon stock per year. Multiplying this value by the estimated carbon stock for post-1989 forests (Table 2) provides an annual estimate of the carbon emissions from post-1989 forest for the period 1990–2009 (Figure 8). These increase over time as both the area and the average carbon stock of post-1989 natural forests increase. Post-1989 forest disturbance emissions in 2009 were estimated to be 813 Mg CO_2-e, which equates to just 0.08% of the total annual disturbance emissions from pre-1990 natural forest in 2009 (1 021 336 Mg CO_2-e).

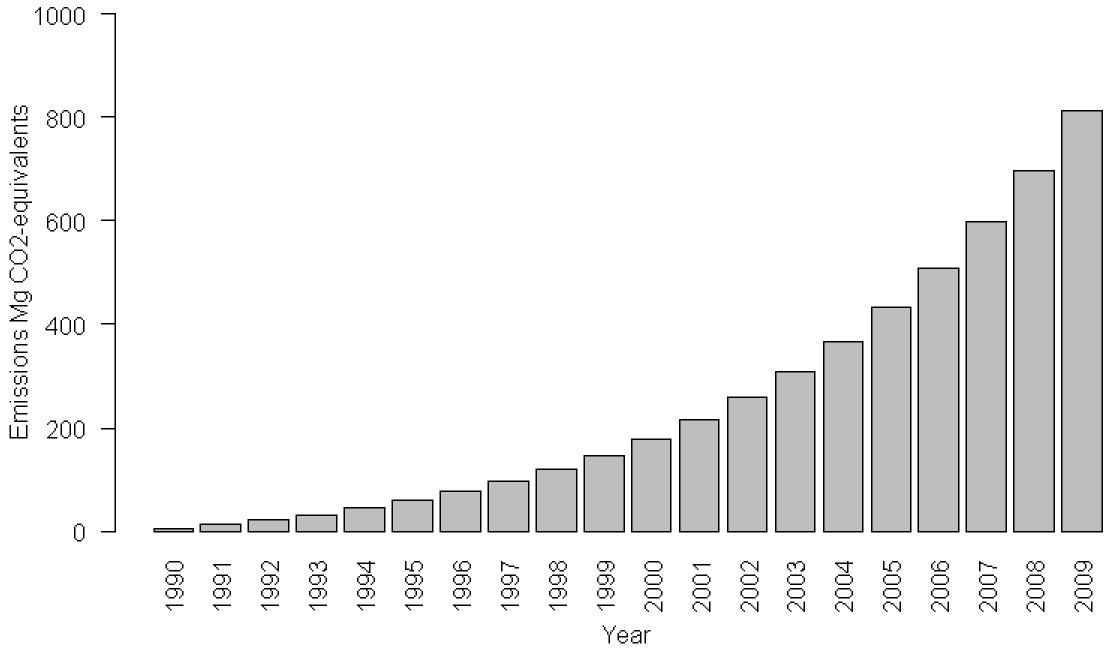

Figure 6 Time series of post-1989 natural forest emissions due to natural disturbance.

6.2 Planted forest carbon emissions

Emissions from wildfires for pre-1990 planted forests and post-1989 planted forests are shown in Figures 9 and 10. Pre-fire biomass is based on the biomass present at the average forest age in both cases. This increases for pre-1990 forests from 1990 to 2000 before reaching a stable level. The post-1989 forest shows the opposite trend – increases in average carbon stock are initially modest, as initial growth rates are low and large areas enter the youngest age class each year through the mid-1990s. Then there is a steady rise in average stock per hectare, which is reflected in a trend of increasing emissions.

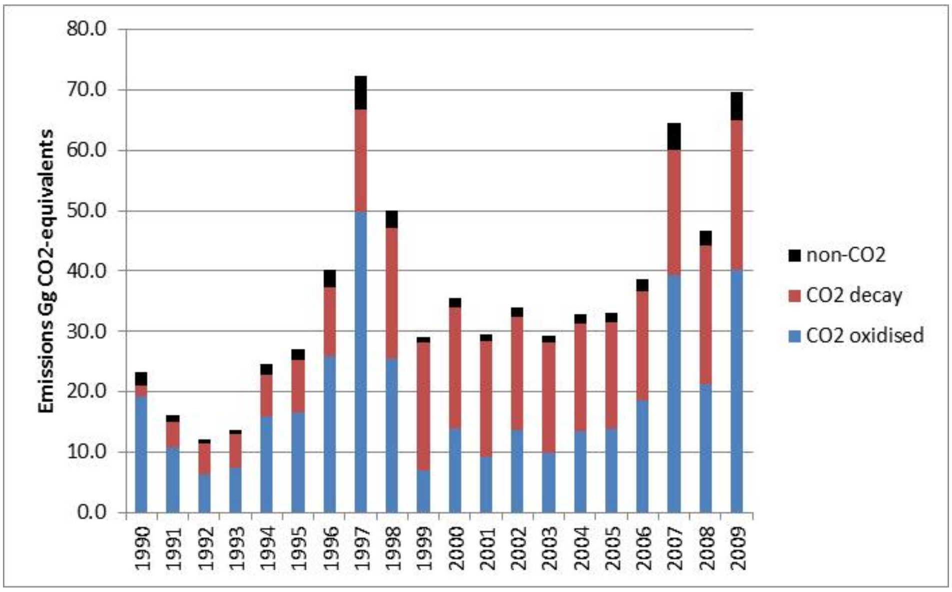

Figure 7 Pre-1990 planted forest wildfire emissions from combustion (CO_2 oxidised and non-CO2) and decay.

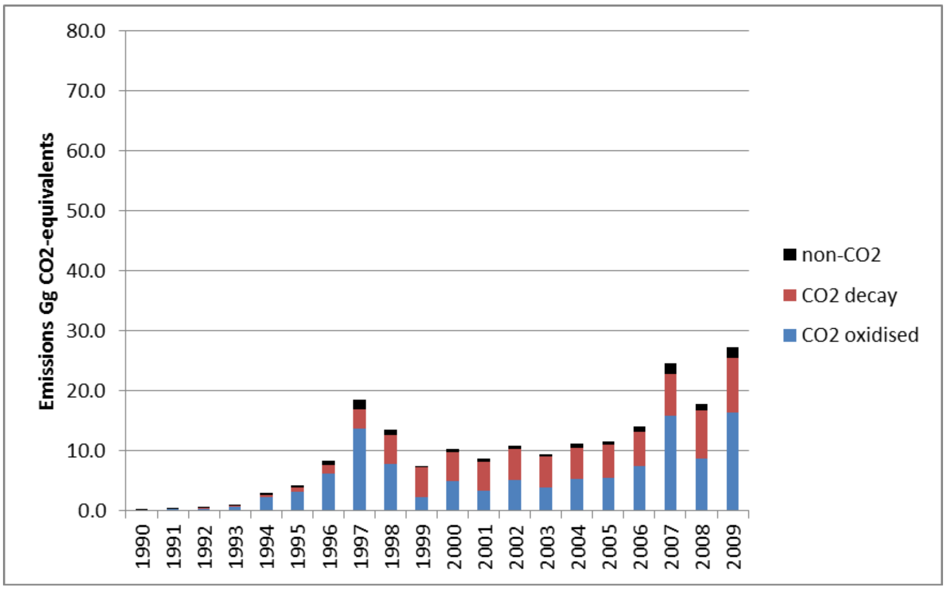

Figure 8 Post-1989 planted forest wildfire emissions from combustion (CO_2 oxidised and non-CO2) and decay.

Emissions from wind damage for pre-1990 planted forests and post-1989 planted forests are shown in Figures 11 and 12. All emissions are from the decay of post-wind-damage residues. The pre-wind-damage biomass is based on the biomass present at the average forest age in both cases. As with the fire emissions, the increasing average age and carbon stock means that the emissions per hectare increase over time.

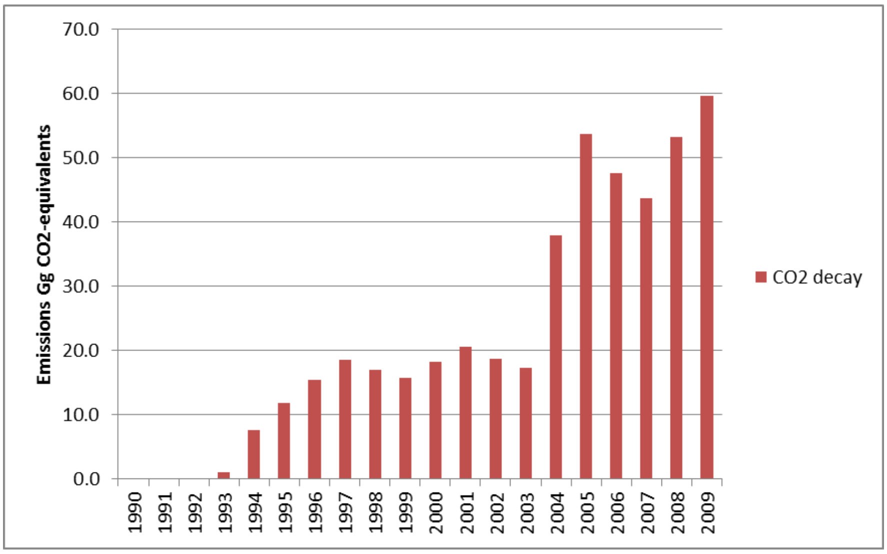

Figure 9 Emissions from wind damage in pre-1990 planted forests.

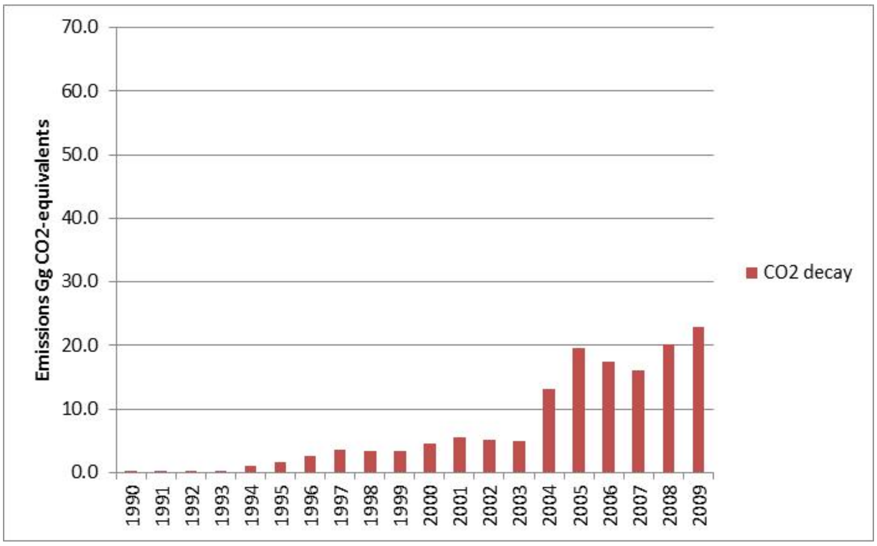

Figure 10 Emissions from wind damage in post-1989 planted forests.

Combined wind and fire emissions from New Zealand's planted forests are shown in Figure 13 and Table 14. The average annual emission was 76.9 Gg CO_2-e for all planted forests, comprising 69.4 Gg CO_2 from pre-1990 forests and 7.5 Gg CO_2 from post-1989 forests. Average net removals by planted forest over this period were about 25 000 Gg per year (MfE 2012).

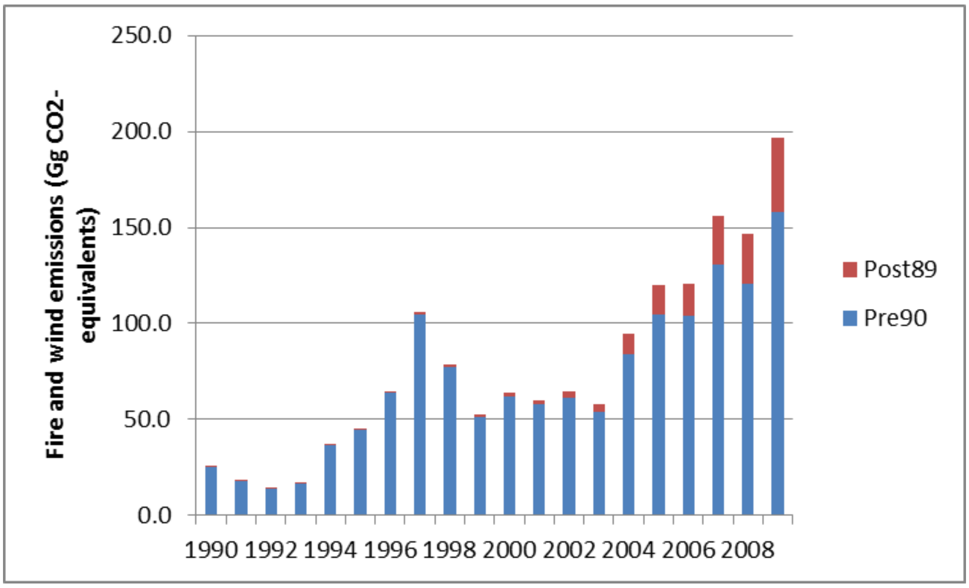

Figure 11 Annual emissions from wind and fire in all planted forests.

6.3 Combined national estimates of annual forest carbon loss for 1990–2008 period

The combined annual emissions from natural disturbance from natural and planted forests for 1990–2008 are provided in Table 14. Pre-1990 natural forest estimates are a constant value as data for individual years were not available. Annual emissions were dominated by pre-1990 natural forests, followed by pre-1990 planted forest, post-1989 planted forest, and finally post-1989 natural forest. The annual emissions from all forests in 2009 (1218.3 Gg CO_2-e) is equivalent to approximately 4.9% of the net removals from planted forests during this period.

The average annual emission over 1990–2009 was 1098 Gg CO_2-e, with a standard deviation of 50.7 Gg CO_2-e. The relatively small standard deviation is not surprising since the estimate for the biggest emission source (pre-1990 natural forest) did not include inter-annual variation. Applying an iterative process to remove outliers that were more than two standard deviations greater than the mean (2009 only) resulted in an average emission from natural disturbance across all forests of 1092 Gg CO_2-e, with a standard deviation of 43.3 Gg CO_2-e.

Table 9 Combined annual carbon emissions for 1990–2009 for natural and planted forests

Year	Pre-1990 natural forests ($Gg\ CO_2$-e)	Post-1989 natural forests ($Gg\ CO_2$-e)	Pre-1990 planted forests ($Gg\ CO_2$-e)	Post-1989 planted forests ($Gg\ CO_2$-e)	All natural forests ($Gg\ CO_2$-e)	All planted forests ($Gg\ CO_2$-e)	All forests ($Gg\ CO_2$-e)
1990	1021	0.0	25.3	0.0	1021.0	25.3	1046.3
1991	1021	0.0	17.9	0.0	1021.0	17.9	1038.9
1992	1021	0.0	13.6	0.0	1021.0	13.6	1034.6
1993	1021	0.0	16.5	0.1	1021.0	16.6	1037.6
1994	1021	0.0	36.5	0.2	1021.0	36.7	1057.7
1995	1021	0.1	44.5	0.3	1021.1	44.9	1066
1996	1021	0.1	63.8	0.7	1021.1	64.5	1085.6
1997	1021	0.1	104.3	1.6	1021.1	105.9	1127
1998	1021	0.1	77.1	1.4	1021.1	78.5	1099.6
1999	1021	0.1	51.5	1.0	1021.1	52.5	1073.6
2000	1021	0.2	62.3	2.0	1021.2	64.2	1085.4
2001	1021	0.2	57.9	2.3	1021.2	60.2	1081.4
2002	1021	0.3	61.3	3.4	1021.3	64.8	1086.1
2003	1021	0.3	54.1	3.6	1021.3	57.8	1079.1
2004	1021	0.4	84.2	10.4	1021.4	94.7	1116.1
2005	1021	0.4	104.5	15.5	1021.4	120.0	1141.4
2006	1021	0.5	104.0	16.8	1021.5	120.8	1142.3
2007	1021	0.6	130.7	25.2	1021.6	155.9	1177.5
2008	1021	0.7	120.8	25.9	1021.7	146.7	1168.4
2009	1021	0.8	157.7	38.8	1021.8	196.5	1218.3

7 Discussion

7.1 Emissions resulting from natural disturbance in New Zealand’s forests

New Zealand forests are influenced by a range of natural disturbance agents that act at different spatial and temporal scales and can have pervasive and long-lasting impacts on forest carbon stocks at the landscape scale (Table 1; Allen et al. 1999; Coomes et al. 2012). Using a combination of permanent plot data, remote sensing, and written records, we estimated the average annual emissions due to natural disturbance across all New Zealand’s forests for 1990–2009 as 1098 Gg CO_2-e. The vast majority (93%) of these emissions was attributable to low intensity disturbance in pre-1990 natural forests. The next biggest sources were emissions due to wind and fire from pre-1990 planted forests (6%) and post-1989 planted forests (1%). Post-1989 natural forests have a very minor contribution (<0.01%) to the total carbon emissions due to their low carbon stock and small total extent. The total emissions from natural disturbance in New Zealand forests over the calibration period (1990–2009) is a relatively small fraction of the country’s total forest carbon budget (e.g. it represents about 5% of the annual net removals by planted forests; MfE 2012) when compared with other countries such as Canada, where natural disturbances are responsible for up to 30% of the country’s total emissions (Kurz et al. 2008b).

7.1.1 Methodological considerations

Our estimates of emissions from low intensity disturbance in pre-1990 natural forests, which account for 93% of total estimated emissions, need to be interpreted cautiously. Using plot data currently available it is impossible to distinguish emissions due to disturbance-related tree mortality and subsequent wood decay from emissions due to natural mortality of large trees in senescing stands, as both these processes have the same plot-level signature (Coomes & Allen 2007). Thus our estimate of emissions in disturbed plots may be an overestimate of actual disturbance-related emissions. However, natural senescence mortality can be expected to be constant thought time in pre-1990 natural forests at the landscape scale. This means that any increases in emissions over and above the baseline rate could be directly attributed to an increase in natural disturbances, providing the same methodology is applied throughout.

Another consideration is that our estimation of net emissions from disturbed plots (6.24 Mg CO_2-e ha^{-1} yr^{-1}) is estimated as the difference between the net change in non-disturbed plots (+3.81 Mg CO_2-e ha^{-1} yr^{-1}) and the net change in disturbed plots (−2.43 Mg CO_2-e ha^{-1} yr^{-1}). Thus it is important for the same data to be used to generate the FMRL and estimate background-level disturbance emissions. The National Vegetation Survey (NVS) plots used in this analysis are national, but are not spatially representative of the entire forest estate. They also do not contain CWD data, which means changes in this significant pool have to be modelled. These methodological differences might mean that our net change estimates differ from those that will be obtained from the specifically designed LUCAS natural forest plot network. To ensure methodological consistency and avoid the expectation of net credits or debits, we therefore recommend that our assessment of emissions from natural disturbance should be recalculated using the LUCAS plot network when the remeasurement data become available.

The calculation of emissions due to wind and fire damage in planted forests relies on a large number of assumptions due to the lack of data. Area data are likely to be underestimated as not all disturbance areas will have been captured. However the relative scale of disturbance compared with the standing area of forest is believed to be adequately reflected. There are few data on which to base pre-disturbance or post-disturbance biomass so it is difficult to verify the suitability of model parameters or the accuracy of estimates of emissions per hectare. Significant improvement of these estimates is likely to require an investment out of proportion to their contribution to New Zealand's greenhouse gas emissions.

Furthermore, the calibration period chosen for planted forests is the minimum permissible. A longer period that captures more major events would make no difference under some of the approaches to calculating a background level and margin, but would affect the calculation of the background level and margin under the default approach. If New Zealand chooses to apply this approach it is possible that reviewers could require the calibration period to be extended.

7.2 Implications for the development of a baseline and margin for Durban reporting

When complete and accepted, the *2013 Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol* will give guidance on the application of the provision for accounting for natural disturbances in the second commitment period of the Kyoto Protocol. The purpose of the provision is to allow Parties to avoid liability arising from emissions caused by natural disturbance that were beyond their ability to materially influence. The intention is that the provision should be otherwise neutral, with no expectations of credits or debits arising from its use.

Five sequential steps for applying the provision have been suggested in the draft guidance:

1. Define the type of disturbances that the Party wishes to be able to exclude from accounting.
2. Establish a consistent and initially complete time series for the calibration period for each disturbance type.
3. Develop the background levels (separately for post-1989 and pre-1990 forests).
4. Develop the margins.
5. Ensure that the method applied does not lead to expectation of net credits or net debits.

From the first two steps, it is possible to calculate the mean level of emissions due to natural disturbance during the calibration period. This is a meaningful value in its own right and is readily understood (e.g. Table 14). However it is important to understand that this is not the same as the “Background Level” defined in the IPCC Guidance. The Background Level is only meaningful in the context of accounting and reporting of emissions and removals during the second Kyoto commitment period. It is an accounting quantity nominated by the Party and is not necessarily based on the mean level of emissions during the calibration period. Since accounting approaches are different for pre-1990 and post-1989 forests, the background levels have different implications in each case and need to be treated separately.

We now interpret of our results within the context of the five steps described above, and present options for the application of a baseline and margin for the FMRL and Afforestation/Reforestation reporting.

7.2.1 Step 1: Types of disturbance

Emissions caused by natural disturbances would normally result in an accounting liability. The special accounting rule for natural disturbance limits that liability and then removes the area affected from the accounting framework during the second commitment period, so any subsequent emissions or removals are not accounted for during the second commitment period. As a general principle, it would be preferable to exclude all emissions from accounting, so all conceivable natural disturbances that may lead to emissions should be considered. Exceptions to this principle include:

- The hypothetical case in which regrowth achieves a higher carbon stock by the end of the commitment period than the carbon lost due to disturbance. In this case it would be better to accept the liability in order to claim the greater credit.
- The possibility that post-2020 accounting rules will “ring-fence” areas for which the natural disturbance accounting rules have been applied previously. This could limit the opportunity to earn credits in future on this land[12].
- Cases where the costs of mapping disturbed areas and quantifying emissions are not offset by the liabilities avoided.

12 Without this limitation in future comitment periods, natural disturbance would be a useful and liability-free way of converting a mature, steady-state carbon reservoir into a young, vigorously-growing sink.

As New Zealand has not made a binding commitment in the second commitment period (CP2), there are no fiscal consequences attached to emissions from natural disturbance. This removes the incentive to apply the provision – it may be preferable to instead account for emissions in full to improve the likelihood that the regenerating forest is available to earn credits in a post-2020 agreement.

If the provision is to be applied, consideration needs to be given to how emissions and removals during CP2 are to be estimated and reported. The intention behind the development of the LUCAS Carbon Monitoring System was that plots would be measured at the start and end of a commitment period. The stock change between measurement dates is then the sum of emissions and removals during the commitment period. With this statistical sampling approach, only disturbances that have a material effect on plots may need to be considered – in effect; "*if it wasn't measured, it didn't happen*". This could lead to issues if inventory reviewers are aware of seemingly major events that were not captured because the plot network was not intensive enough. Conversely, a rare event that happens to affect a plot (e.g. a fire of less than 1 ha that destroys a pre-1990 native forest plot) will be assumed to have affected a much larger area (e.g. over 6000 ha of natural forest). This suggests that <u>all</u> possible disturbance events need to be identified for inclusion within the background, even if they did not occur during the calibration period. In practice the background level can be updated during the commitment period, so if, for example, a meteorite destroys a plot it can be added to the background level as a new disturbance type (with an expected emission of zero assigned)[13]. Similarly, one or two plots could be affected by pathogens to an extent that is much greater than captured in growth models used in the FMRL. This raises the possibility that low-intensity disturbances that reduce growth rather than causing net emissions need to be identified and quantified in the baseline. It may be too late to capture these in an updated Background Level by the time the disturbance has been recognised from the remeasurement and analysis of plots.

It is also possible that LUCAS plots will not be remeasured at the end of the commitment period, so stock change will be derived from projections of the plot data from previous measurements. In this case some other approach to identifying natural disturbance emissions is required (as is currently done for harvesting emissions). The draft guidance offers examples of approaches to identifying lands affected by natural disturbance and their challenges. Since the FMRL for pre-1990 forest accounting must be consistent with the approach used for inventory reporting, and the natural disturbance background level must be consistent with the FMRL, clarity over the intended inventory reporting approach is required before the Background Level can be set.

In summary, all disturbances that have the potential to adversely affect the accounting quantity reported for CP2 should be identified by New Zealand in the 2015 greenhouse gas inventory.

13 The draft text states that while a Party is not eligible to exclude emissions for disturbance types for which it fails to report historical time series of emissions for the calibration period, it can submit such a historical time series later in the commitment period, in which case a technical correction may be needed for the FMRL

7.2.2 Step 2: Establish a consistent and initially complete time series for the calibration period for each disturbance type

For planted forests, calibration-period estimates of annual emissions have been estimated for wind damage and wildfires in both pre-1990 and post-1989 forests. This was achieved through the use of historical records of individual disturbance events. In addition, annual emissions of zero can be reported for certain disturbance types that have not influenced planted forests during the calibration period. These include geological disturbances (volcanic activity, tsunamis, earthquakes) and insect and disease pathogens that are not yet found in New Zealand.

For natural forest the data available were not sufficient to directly estimate annual emissions for each year over 1990–2009. This meant that an average estimate of emissions due to natural disturbance had to be applied to all years (Table 14). Furthermore, it is very difficult to distinguish the individual type of disturbance affecting “disturbed” plots. Our approach therefore used permanent plot data to quantify net emissions from all sources of low-intensity disturbance. This estimate includes tree mortality and reductions in tree growth arising from extreme weather events such as droughts, snow and strong winds, insect and disease infestations and geological disturbances, although the effect of some of these disturbance types (e.g. volcanic eruptions) can be assumed to be zero. The challenges involved in dividing natural forest disturbance emissions into individual disturbance types needs to be taken into consideration when considering how to quantify and report future disturbance emissions (Vanderwell et al. 2013).

Catastrophic disturbance to both land and forest (i.e. landslides) was assessed using a separate method as these disturbances are readily detectable using remote-sensing techniques. This produced a single estimate of the area of pre-1990 natural forest affected by landslides over the period 1990–2008 (which was the date of available imagery). This technique could be used in the future, either on its own or in combination with plot data to assess the area affected by landslides following a severe disturbance event. However, this may not be necessary as the total landslide area detected from 1990 to 2008 (4529 ha) was below the threshold for one pre-1990 natural forest LUCAS plot (6500 ha). Landslides are also able to be captured using permanent plots (Allen et al. 1999), and therefore it can be reasonably assumed that the LUCAS plot network, by design, is capable of capturing nationally significant landslide events in the future.

The calibration time series for post-1989 forest will need to be adjusted to take into account age-class effects when the Background Level is developed. For example, calculation of annual fire emissions in post-1989 planted forests takes into account the average age and hence average stock each year. During CP2, the forest will be older and the stock greater, so a given area burnt will produce more emissions. The same applies to post-1989 natural forest, although the emissions from this forest class are likely to be very minor at a national scale.

7.2.3 Steps 3–5: Develop the background levels and margins such that the expectation of net credits or debits is avoided

Steps 3–5 need to be discussed together. There are, however, distinctly different issues to consider for pre-1990 forests (accounted for as Forest Management lands) compared with post-1989 forests (accounted for as Afforestation/Reforestation lands) because of differences in accounting methodology. Planted and natural forests also need to be considered separately due to differences in their respective FMRLs.

Options for developing the Background Level and margin described in Guidance include: (1) a default approach using the mean calibration period level plus a two standard-deviation margin; (2) the minimum historical level + zero margin; and (3) a zero baseline and zero margin. A standard deviation cannot be calculated for natural forest for use in approach (1) because the time series is not annual – this means that "minimum level" required by approach (2) is also unknown. Careful consideration is required in order to better understand the possible advantages in applying anything other than approach (3). At present, the draft guidance does not provide enough clarity on this issue.

Pre-1990 (Forest Management accounting)

Accounting for pre-1990 forests is against the FMRL, which sets the expectation of future net removals. The intention is that if business-as-usual management is adhered to throughout the commitment period, no credits or debits should accrue. This suggests that an FMRL that assumes no natural disturbance should be acceptable – it just means that all areas affected by "Natural Disturbance" can be excluded from accounting. As long as the areas excluded are randomly distributed with respect to factors that influence emissions and removals (e.g. productivity, likelihood of management interventions such as thinning or harvesting), there would be no expectation of credits or debits from the post-1990 forests remaining within the accounting framework. Nothing would be achieved by incorporating a higher level of natural disturbance into the FMRL through a non-zero Background Level – if actual CP2 emissions turn out to be greater, the Party would be liable for emissions up to the margin. If actual emissions were lower than the calibration-period average, the Party would gain credits. While the "expected value" of credits or debits would be zero, there is still a chance that the average disturbance rate over the commitment period will differ from the average rate so either credits or debits will accrue. This would be completely avoided with a zero baseline and margin.

The exception is if natural disturbance is not randomly distributed. If it was expected that the least productive forest areas would be damaged and removed from accounting, the remaining areas would increase removals on a per hectare basis relative to the FMRL, creating an expectation of credits.

An FMRL for natural forests has not yet been developed. The baseline could be based on a projection of emissions and removals in each plot, summing to give total net removals expected during the commitment period. A background level of emissions from natural disturbance could be subtracted from this, leading to the possibility that New Zealand could gain credits simply because actual emissions from disturbance during the commitment period turned out to be less than the calibration-period rate. Since New Zealand is not intending to gain credits through changes to management of natural forests, a zero baseline and margin seem appropriate.

The FMRL for pre-1990 planted forests is based on a simulation of areas by age class over time, capturing gains due to growth and losses due to harvesting. New Zealand may wish to gain credits from Forest Management land through improved management that increases removals and/or decreases emission relative to the FMRL. Minimising the area excluded from accounting will minimise the risk that post-2020 accounting rules prevent this. Otherwise a zero baseline and margin still appear to be the simplest option.

Under FMRL accounting, the only clear advantage of assuming a non-zero Background Level and margin is that there is a potential upside if the actual rate of natural disturbance during the commitment period is lower than in the past. In this case the pre-1990 forests will deliver higher net removals than the business-as-usual removals reflected by the FMRL and natural disturbance Background Level and margin. The potential upside is of course balanced by a potential downside if the reverse is true.

With a zero Background Level and margin, if the disturbance provision is invoked then for any LUCAS plot disturbed during the commitment period, the disturbance boundary would be delineated and the area and plot removed from stock and stock change calculations (or rather, stock changes would be calculated separately and reported but not entered into accounts).

Post-1989 (Afforestation/Reforestation)

For post-1989 forests, all emissions and removals are accounted for without reference to a baseline. The natural disturbance level effectively provides insurance against unexpected losses, but unlike with FMRL accounting, the expected net removals under business as usual do not need to be reported in advance. *Any* background level will therefore give rise to an expectation of net credits compared with not having a background level, because if there is no background level established there is no opportunity to exclude any emissions that arise from natural disturbances.

Once again, the simplest option appears to be to set a baseline and margin of zero. Any non-zero baseline will require some proportion of natural disturbance emissions to be accounted for.

Further clarification from the IPCC is required before a firm recommendation can be made. One experienced international inventory compiler and reviewer has suggested that New Zealand should adopt approach (2); setting the Background Level to the minimum level of emissions in the calibration period, with a margin of zero. Then in each year of the commitment period where actual emissions were higher, the excess emissions could be removed from accounts provided it could be shown that this increase was not materially affected by actions or inactions within New Zealand. It was also suggested that the same background level be converted to a per-unit-area basis and applied to the post-1989 forest area. The argument against a zero Background Level and margin seemed to centre on the requirement to demonstrate that the emissions were caused “by events and/or circumstances beyond the control of, and not materially influenced by, the Party”. The logic of this is unclear. For example, if a Party follows approach (2) they are not claiming that they had no control over emissions above the minimum calibration-period level, but that they did have control over the minimum-level emissions and will therefore not exclude them. Furthermore, approach (2) ensures that there will never be debits under FMRL accounting while credits will arise whenever actual emissions are less than the calibration-period minimum level. This seems unbalanced, even if the “expectation” is still of no credits or debits.

7.3 Hypothetical case studies

7.3.1 North Island volcanic eruption

Let us suppose a volcanic eruption occurs, in the order of the 1888 Tarawera eruption, somewhere in the Rotorua area. Firstly, we would still have to report all emissions and removals for UNFCCC annual greenhouse gas inventory reporting. Remeasurement of LUCAS plots is the basis for inventory reporting, so if none of the plots were directly affected we would have nothing to report. Since the same plots are used for Kyoto reporting we would also have failed to detect any emissions liability, so there is no need to apply the provision. However, volcanic activity is likely to attract international attention, so we may need to calculate the emissions anyway.

If plots were directly affected, then the accounting quantity will be affected (negatively). In this case we would need to assess whether (a) land use change was likely to take place on the affected areas, and (b) salvage logging was likely to take place. If land use change (deforestation) occurred we would be liable for all emissions so there is no direct need to map the disturbance area and quantify emissions. If salvage logging occurred, then the salvaged carbon must be accounted for as a normal harvesting emission – in some cases that may not leave enough emissions to be worth trying to avoid.

If the provision is to be applied, the first step is to identify and map the affected area, stratifying by forest sub-category (planted and natural pre-1990 forest, post-1989 forest) and preferably also by the severity or nature of damage. One option would be to then recalculate the stock change using the unaffected plots for the new unaffected population. The affected plots may not be sufficient to estimate the disturbance emissions – options are described in the guidance involving remote-sensing with ground-truthing to supplement the plot network. The impact on the affected area would be calculated and reported in the greenhouse gas inventory. If the baseline and margin were zero, the entire emission would be excluded from Kyoto accounting. Finally, because the remaining area to be accounted against the FMRL would then be smaller than the area on which the FMRL was based, a technical correction to the FMRL would be required (as outlined in the guidance).

7.3.2 Alpine Fault earthquake

If a large Alpine Fault earthquake occurred then the majority of the damage is likely to occur in pre-1990 natural forest. Damage is likely to consist of numerous landslides of all sizes (Wells et al. 2001; Allen et al. 1999), branch breakage, tree mortality, and burial of entire forests on the floodplains (e.g. Cullen et al. 2003). An initial survey of the damage is likely to suggest that a large number of LUCAS natural forest plots are affected. It would be useful to identify affected plots and map the damage area (possibly using remote-sensing techniques developed here) as soon as possible after the event occurred, although ongoing tree mortality and additional landslides are likely to continue for some time afterwards.

When the LUCAS natural forest plots are remeasured at the end of CP2, descriptive data on disturbance type could be collected to identify plots affected by this event. This assumes that measuring of the LUCAS plots was both feasible and affordable following Alpine Fault movement and the resulting damage to roads, towns and infrastructure. Carbon stock change at a national scale could then be calculated for all LUCAS plots, and if this does not differ from baseline expectations then the provision would not need to be invoked. However, it is likely that there will be a significant net emission. The area affected by the disturbance would then need to be quantified (either via direct mapping or using the statistical sampling approach based on the number of LUCAS plots affected) and removed from the national carbon stock calculations. Actual losses following the disturbance event could be estimated using data from the disturbed LUCAS plots, and these emissions would be excluded from Kyoto accounting.

8 Recommendations

This study has estimated annual carbon emissions for 1990–2009 for both natural and planted pre-1990 and post-1989 forests, and discussed the use of this information to develop baseline natural disturbance estimates for Forest Management Reference Level (FMRL) and Afforestation/Reforestation reporting. Based on our results we recommend that:

- The IPCC good practice guidance document is closely monitored for ongoing developments in accounting for natural disturbance.
- The FMRL and natural disturbance baselines are developed simultaneously using the same data and consistent methodology. This is required to avoid the expectation of net credits or debits.
- Depending on the methods used to develop the FMRL, a zero baseline and zero margin should be considered as an option for second-commitment-period accounting.
- The natural disturbance provision should only be applied when the disturbance event has significant effects on national carbon stock estimates, and any possible ramifications for post-2020 carbon accounting should be taken into account.
- The LUCAS plot network should be remeasured fully, including CWD, by 2020 to allow reporting on natural disturbance and carbon stock change in both natural and planted forests.

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Appendix I

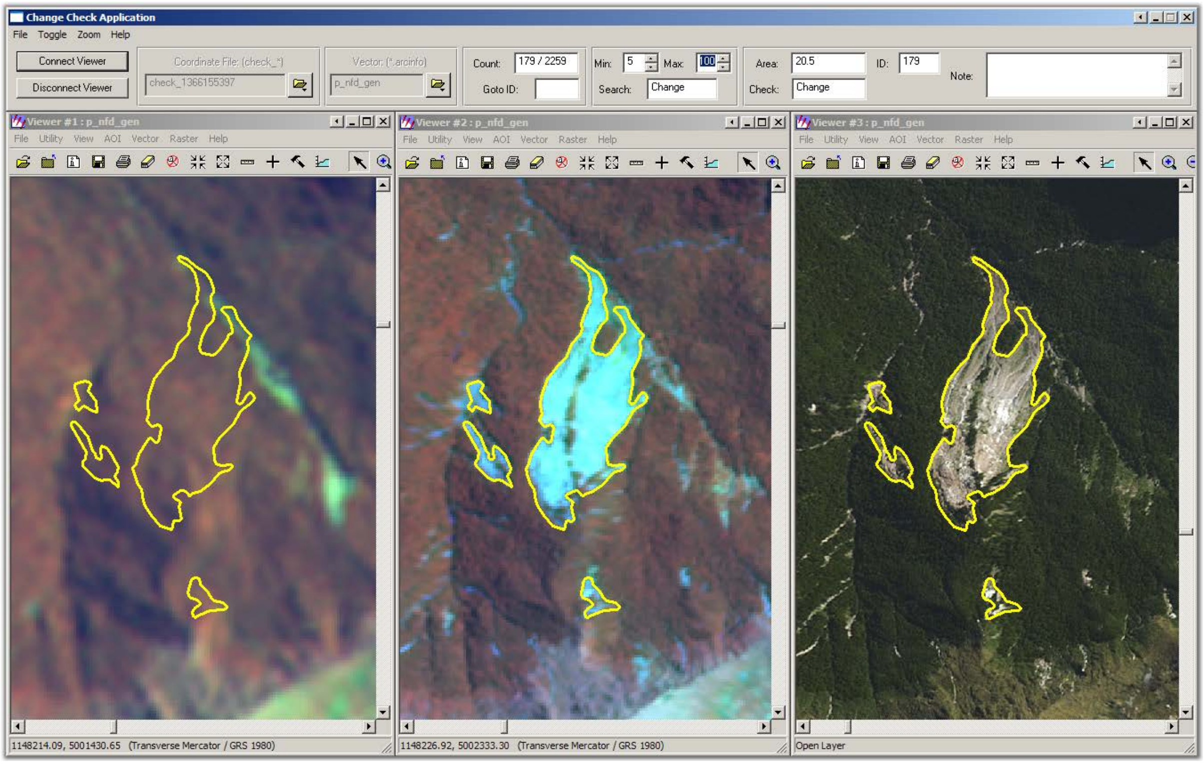

Figure A1.1 Screen shot of check tool in use. Left-hand window contains 1990 LANDSAT imagery, middle window contains 2008 SPOT imagery, and right-hand window contains 2009 SPOT Maps imagery. Yellow outlines are change polygons identified by the segmentation routines as described in the Methods section.

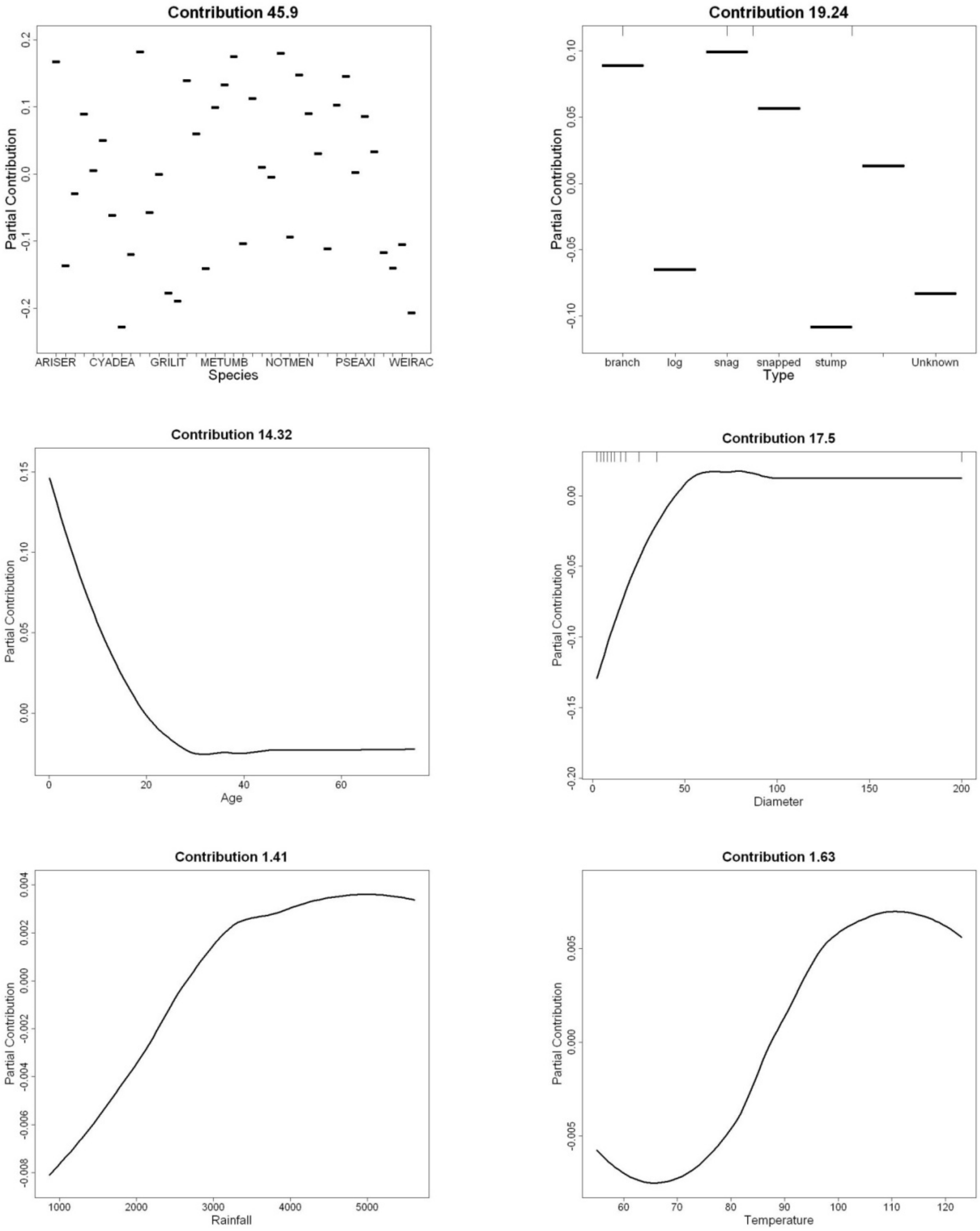

Figure A1.2 Partial contributions of each predictor in the boosted regression tree model of CWD that includes all potential parameters. The species effect is dominant with the highest contribution and greatest range of values; followed by CWD type, then age and diameter. Climate is not an important predictor of CWD decay rates.

Appendix II

Attritional pests and diseases

New Zealand's country report on the Montreal Process criteria and indicators provides an estimate of the trends in the area affected and annual cost of biotic natural disturbances (MAF 2009). The only insect pest of note is *Hylastes ater*, a bark beetle, which affects about 2% of the plantation area and can be associated with seedling death. Introduced mammals are a localised nuisance in some plantation forests. These include rabbits and hares, deer, goats and other feral livestock and the brushtail possum. Possums were recorded as affecting 140 000 ha (8%) of plantation forests, with a decline since reports peaked around 2000 (MAF 2009). Invasive woody weeds need to be controlled with herbicides and/or biological control to allow successful forest establishment. Once established, plantation species can become dominant. There is no national monitoring of the impact of weeds but significant emissions due to invasive weeds are unlikely. None of these disturbance agents are significant enough at a national scale to warrant quantification of emissions.

There are some diseases in planted forests that do cause significant damage. These diseases generally affect tree growth rates and wood quality, rather than causing tree mortality. It was estimated that 18% of the planted forest area was affected by diseases in 2005 (MAF 2009).

Because of their economic significance, there have been several attempts to measure losses in plantation forests due to these diseases (Table A2.1). *Dothistroma septosporum* causes growth losses once infection exceeds 25% of foliage. Sweet (1989) assumed that growth losses were negligible due to an effective spray programme. Pruning and thinning reduce disease incidence and trees become resistant at about 15 years. Bulman (unpubl. data 2007), updating earlier work by New (1989), estimated that the annual growth loss in the susceptible age classes (1–15) was about 3%.

An average growth loss of 2.3% due to *Cyclaneusma* spp. was estimated by van der Pas (1984b) and used by Sweet (1989) to estimate economic losses. Bulman (1988) estimated an average growth loss of 5% for age classes 6–20 based on 15 surveyed forests, with much higher losses on the more-disease-prone sites. Two more recent studies assigned a risk rating to regions based on the average incidence and severity of the disease and used this to determine annual volume loss (Bulman 2001; Bulman, unpubl. data, 2007). Estimates of volume loss varied from 1.6% in Canterbury to 10.9% on the East Coast, with a mean for the 6–20-year age classes of 6.6%. Volume loss outside these age classes is negligible.

Table A2.1. Estimates of national losses attributable to disease in New Zealand plantation forests

Disease	Age classes	Growth loss (%)	Source
Dothistroma needle blight	1–15	0.0	Sweet (1989)
"	"	2.0[1]	New (1989)
"	"	4.0[2]	MAF (2002)
"	"	3.0	Bulman (*unpubl. data* 2007)
Cyclaneusma needle-cast	6–20	2.3	van der Pas (1984a, 1984b); Sweet (1989)
"	"	5.0	Bulman (1988)
"	"	6.6	Bulman (2001)
Armillaria root rot	All	0.3	Sweet (1989)
"		2.0	Hood et al. (2002)
Nectria flute canker	All	0.172	Turner et al. (2007)
Swiss needle-cast	All	26–40[3]	Manley (1985)
"	All	32[4]	Kimberley et al. (2011)
"	All	1.5	This report

1 Not stated in paper, but estimated from data supplied.
2 Double estimate taken from New (1989), due to improved estimate of disease incidence.
3 In one region for Douglas-fir stands only.
4 Nationally for Douglas-fir stands only.

Armillaria spp. cause root rot, which can lead to mortality or reduced growth. Kimberley et al. (2002) found a 25% loss in volume in a first-rotation plantation planted on a cleared native forest site (21% due to mortality and an additional 4% growth reduction). Infection is much less in second rotations and rare in first rotations planted onto grassland. Hood et al. (2002) calculated the economic losses across the plantation estate, assuming volume losses of 20% for first-rotation sites previously under natural forest, 0% for other first-rotation sites and 1.5% for second rotations. At a national level this corresponded to a loss in yield of about 2% of the then current harvest.

Sphaeropsis sapinea (diplodia whorl canker) and *Nectria fuckeliana* (nectria flute canker) cause malformation, growth loss and mortality in some locations. Both are often associated with pruning. Diplodia causes shoot and leader dieback, whorl cankers and crown wilt. Timber degrade due to severe internal blue stain also occurs. Losses in terms of merchantable sawlog recovery can be high on affected sites but the impact on biomass increment at a national level is low. Nectria flute canker was first formally identified in New Zealand in 1996 (Dick & Crane 2009). Whereas diplodia is usually found in warm, dry sites, nectria is currently limited to south of the Banks Peninsula. The impact on growth is uncertain. Turner et al. (2007) assumed a 10% volume loss in infected trees which implies less than 0.2% loss of volume at a national level given the disease incidence in the area affected.

Swiss needle-cast caused by *Phaeocryptopus gaeumannii* has a major impact on the productivity of Douglas-fir in New Zealand. Kimberley et al. (2011) estimated that the cumulative mean reduction in stem volume growth rate since Swiss needle-cast arrived in 1959 was 35% in the North Island and 23% in the South Island, or 32% overall. Mortality rates were the same as in uninfected stands. Douglas-fir makes up 5% of planted forest area, so the growth reduction at a national level is about 1.5%. It is not yet clear whether this growth reduction also applies to newer genetic material.

Several fungi cause dieback and mortality of *Cupressus* spp. and a range of pests and diseases affect other species, but the affected areas are relatively small. Major plantation areas are surveyed each year to monitor existing pests and diseases and identify new problems. Red needle cast caused by an organism newly found in New Zealand, *Phytophthora pluvialis*, is the latest needle disease of *Pinus radiata* to be identified. This had not been detected during the 1990–2009 baseline period.

In summary, damage caused by biotic factors can be assumed to be captured within the growth and mortality models. There have been no catastrophic pest or disease outbreaks causing significant emissions during the baseline 1990–2009 period. Some abiotic risks are also captured in growth models, including drought, snow fall, physiological needle blight, and attritional wind damage.

Based on the average national volume reductions estimated for *Dothistroma* over age classes 1–15 and *Cyclaneusma* over classes 6–20, a total reduction of just under 5% over a full 28-year rotation can be estimated. Turner et al. (2004) assumed a 5% volume reduction resulting from all pathogens combined for their medium scenario in a biosecurity cost–benefit study. Rounding up to 6% would allow for the effects of the other disturbance agents discussed.

Projections for New Zealand's Forest Management Reference Level estimate that pre-1990 planted forests will be a net source during 2013–2020 of 11.15 Mt CO_2-e per year. Without disease there would be a 6% increase in annual growth rate leading to higher gross removals of atmospheric CO_2 but also increased emissions associated with harvesting, as there would be a higher volume at maturity. Emissions from the decay of pruning, thinning and harvest residues would be higher.

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Appendix III

Description of historical fires and salvage logging in planted forest

Pearce and Alexander (1994) selected eight pre-1990 wildfires in planted forests[14] for retrospective analysis, of which only the 1955 Balmoral fire had been documented in detail. Referring to Ward (2005), Guild and Dudfield (2010) note that this fire destroyed 3152 ha of mainly *Pinus radiata* and Corsican pine (*P. nigra* ssp. *laricio*), most of which was 24–32 years old. They suggested that 1.2 million cubic metres of timber was lost; assuming an average wood density of 420 kg m^{-3} and a carbon fraction of 0.5, this implies about 80 t C ha^{-1} in merchantable wood. This may refer to a quantity damaged and unsalvageable rather than oxidised, while the non-merchantable stand component oxidised may have been of a similar magnitude. Prior (1958) reported that this same fire killed about 2428 ha of forest, with another 688 ha scorched but not killed. The *Pinus radiata* was mainly 20–24 m tall and low pruned only, while the stands of other pines were shorter. Three categories of damage were recognised: stands subject to running crown fires had the foliage completely destroyed, while ground-to-crown fires killed the trees and left the foliage scorched but unconsumed. Ground fires scorched tree butts but left the crowns green. Damage was most severe in those stands with heavier than usual slash and/or carrying a high percentage of dead and moribund branches. Reasons for this include:

- 1400 ha had suffered severe wind damage 10 years earlier, with salvage logging leaving behind a higher than usual proportion of material;
- An outbreak of *Psedocoremia suavis* in 1952 partially defoliated and killed a high percentage of 24-year-old *Pinus radiata* on about 300 ha;
- 670 ha of Corsican pine had been thinned to waste between 1949 and 1954;
- There were many unthinned stands carrying over 4000 stems per hectare.

In contrast, a newly planted area on which harvesting slash had been burned before replanting escaped entirely undamaged. In areas that burned during the night, only the upper layers of the needle litter were consumed while in other areas the needle litter and shallow humus layer were completely consumed. Stems felled within the previous 6 years rarely had more than the branches consumed by the fire, while older felled stems sometimes burned completely. Bark on standing trees was rarely burnt unless in contact with slash but cambium damage mainly in *Pinus radiata* had implications for salvage harvesting – 9 months after the fire a large proportion of burnt *P. radiata* stems were badly sapstained, whereas clean Corsican pine logs were still being harvested from some areas 16 months after the fire (Prior 1958).

The other historical planted forest fires assessed by Pearce and Alexander (1994) are included in Table 4 and Table A3.1. A summary was provided of the main tree species involved, the main ages (ranging from 7 to 52 years), silvicultural treatments and predominant type of fire activity (crown or surface). Within the area burnt for each fire there was evidence of both low-intensity surface burning and high-intensity crown fire. However, the authors' focus was on determining the prevailing fire weather index.

[14] An extra 6000 ha grass fire which threatened Oxford forest was also included.

Documentation of fires since 1990 is available at varying levels of detail for fires at Berwick forest (Fogarty et al. 1996); Bucklands Crossing (Pearce et al. 1998), Miners Road (Anderson 2003), Maringi forest (Wishnowsky 2006), Mt Allan (Kerr & Still 2010), and Mosquito Gulley, Poutu (Baker & Smithies 2011). Information on a smaller fire at Para Road, Marlborough, was also extracted (Baker & Dijkstra 2008). Table A3.1 presents information for the significant planted forest fires since 1990 relating to pre- and post-fire biomass.

These fires burned standing forest ranging from age 1 to age 48. Most fires occurred in *Pinus radiata*; the Berwick fire in 1995 was unusual in that it also burnt an old NZFS arboretum. Some fires burned predominantly in cutover sites, while areas of grass and scrub were also burnt. For fire research purposes, the total biomass on site is of less interest than the available fuel load, which is the amount of generally fine combustible material that will support fire spread. Estimates of fuel load were only found for three of the fires, and the amount of fuel actually consumed was not reported. Photographic evidence suggests that even in stands "destroyed" by fire, trees were still standing with branches and often foliage intact.

Table A3.1 Description of significant plantation forest fire events 1990–2012

Year	Fire	Area (ha)	Forest type and fuel load
1994	Purakaunui	210	25 ha of plantation, 210 ha total,
1995	Berwick	255	181 ha of plantation, 74 ha of scrub and recent cutover. *P. radiata* (aged 3), *Thuja plicata* (48), mid-rotation *P. radiata*, larch (13), Douglas-fir (14). Pine and fir included dense gorse understorey 3–5 m high. Crown foliage combustion was greatest in *Thuja*. Calculations used a conservative fuel load of 30 t ha^{-1} . All stands aged 13 and over were salvage logged except the *Thuja.*
1996	Mohaka	241?	> 100 ha of mostly logging slash, but also a small area of mature Douglas-fir. Fttn17
1997	Aupouri	260	
1997	Harakeke	532	
1998	Bucklands Crossing	200	*P. radiata* 20 ha; remainder grazed pasture and mānuka/gorse. Initial ignition in mixed-age pine with a fuel load of 10.5 t ha^{-1}.
2002	Miners Road	197	197.5 ha total; 17.6 ha *P. radiata* aged 4, 8, 14, 24 and 30; remainder mainly grass. Fire within mature pruned stands was largely surface, burning almost all duff material. Crown fires in younger trees.
2004	Irvines	200	20 ha mature, otherwise mainly 1–2 years old or cutover. (Nelson Mail, 6 Dec. 2004)
2005	Mohaka	240	200 ha 15–20 years old, 40 ha newly planted.
2006	Maringi	193	11-, 12- and 13-year-old pines destroyed. Mostly pruned to 6 m and thinned. Available fuel loadings were high – mid-summer, recently thinned, cured grasses following moist spring, ground fuels up to 2 m deep.
2006	Canvastown	215	133 ha of plantation, 82 ha of scrub and grass http://www.ruralfirehistory.org.nz/documents/Canvastown.htm
2007	Waipoua	224	About 100 ha of 10–13-year-old *P. radiata*; also wetlands, scrub. http://www.northernadvocate.co.nz/news/illegal-beach-cooks-ignite-3m-inferno/973478/
2008	Para Rd	84	Overstorey of mixed age class *Pinus radiata* from 5 to 15 years of age; understorey of bracken, gorse, broom, native scrub hardwood species. Older pines pruned to 6 m, thinned to 250 stems ha^{-1}. 83.4 ha damaged or destroyed within the 95.3-ha fire boundary. All required replacement.
2009	Tadmor	600?	541 ha damage to plantations.(Tasman District Council annual report 2010)
2010 Feb	Mt Allan	710	710-ha fire perimeter; 295 ha of standing forest, rest mainly cutover with some scrub.
2010	Glenhope	200	
2010 Dec	Mt Allan	95?	Initially within Feb 2010 fire perimeter: 90 ha cutover; 5 ha standing trees. http://www.odt.co.nz/news/dunedin/142118/mt-allan-blaze-spreads
2010	Poutu	115	83.7 ha of 9–10-year *P. radiata* and 0.1 ha aged 13; also 22.4 ha of roadways, wetland and grass/scrub. Pines had 69 t ha^{-1} of thinnings and prunings of which 22 t ha^{-1} characterised as “available”. Additional available light fine fuel in standing trees. Crown fire stripped needles, fine branches and some bark from standing trees.
2011	White Cliffs, Horeke	345	Mainly *P. radiata* plantation; also scrub.

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