

THE EFFECT OF CLIMATE CHANGE ON NEW ZEALAND'S PLANTED FORESTS

IMPACTS, RISKS AND OPPORTUNITIES

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Executive Summary

Objectives

The objectives of this report were to:

1. Synthesise current knowledge on likely climatic changes in New Zealand over the course of this century (Chapter 1).
2. Summarise current knowledge of the likely direct climate-change impacts on plantation productivity (Chapter 2).
3. Summarise the current knowledge of likely climate-change effects on abiotic (wind and fire risks) and biotic (insects, diseases, weeds) factors, and how changes in these secondary factors are likely to influence plantation productivity (Chapter 3).
4. Develop an interdisciplinary framework which can be used to describe the direct and indirect effects of climate change on plantation productivity. For this framework, define the linkages and feedbacks between direct and indirect effects on plantation productivity. Using this framework, identify key knowledge gaps and uncertainties, and provide a comprehensive research plan to address these uncertainties (Chapter 4).

Major results

Climate change

The mean annual temperature for all of New Zealand is projected to increase by around 2°C from 1990 to 2090. Annual rainfall projections indicate a likely difference in trends, with the western side of both islands trending towards increasing rainfall while the eastern side trends towards decreasing rainfall. The projected increase in temperature is likely to lead to a significant reduction of frost days, an increase in hot days and an increase in the intensity of heavy rainfall events. Drought risk is likely to increase in currently drought-prone areas where reductions in rainfall and increases in evaporation are anticipated. There are indications that the annual mean westerly wind component may increase by about 10% by 2090, and that high winds (wind speeds exceeded 1% of the time) may increase by around on average 2–3%.

Climate change and tree growth

Climate change is likely to have a significant impact on the future growth of trees in planted forests because tree growth responds directly to changes in CO₂ concentration, temperature and nutrient and water availability. There are direct responses to these drivers and indirect interactions and feedback processes.

Increasing CO₂ concentration is generally beneficial for the growth of plants, but the magnitude of the response varies with species and growth stage and is affected by interactions with other environmental factors that can limit (e.g. lack of essential nutrients) or enhance (e.g. warmer and drier conditions) the 'CO₂ fertilisation effect'. CO₂ fertilisation is likely to be most beneficial in the drier parts of the country, such as Otago, Canterbury, Hawke's Bay and East Cape. Nutrient limitations (especially nitrogen) are unlikely to limit the CO₂ response provided that the nutrient levels, which are currently sufficient in most of New Zealand's plantations, can be

maintained into the future. Increasing temperatures can also stimulate decomposition of soil organic matter and mineralise more nitrogen to further boost the nutritional status of trees.

Increases in air temperature expected with climate change are likely to have a predominantly positive effect on plantation growth, as increasing temperature is likely to lengthen the growing season.

While radiata pine is New Zealand's most important plantation species, there has not yet been a comprehensive modelling study to investigate the likely growth response of radiata pine to the combination of likely climatic changes over the short or longer term. Comprehensive models that include these processes exist and are available, but they need to be tested in New Zealand before being used to quantify the forest growth response across the wide edaphic and climatic range over which plantations are grown in New Zealand.

Climate change is also likely to affect many abiotic and biotic factors, which may in turn affect plantation growth and productivity. The main factors which we have considered here include weeds, insects, pathogens and the risks from wind and fire. All of these factors currently cause significant economic losses in planted forests.

Changes in risk from fire and wind

Climate change is likely to increase fire risk and the incidence of damaging winds. Previous research has shown that fire danger is likely to increase significantly in most areas of New Zealand, and that the length of the fire season will probably increase. Increases in fire risk are likely to result in an increase in the incidence of fires and plantation area burned.

It is also predicted that the westerly wind speed component will increase during the winter and spring periods, leading to an increase in the mean and extreme wind speeds for many regions of New Zealand. For the central and upper regions of the North Island, which contain a large proportion of the plantation estate, extreme wind speeds are predicted to decrease. However, the upper and eastern parts of the North Island may be subject to more severe extra-tropical cyclones. In those regions where there is a predicted increase in the severe wind climate, it is likely that the annual frequency of winds sufficient in magnitude to cause widespread damage to forests could increase substantially. Previous research and simulations indicate that the impacts from these projected increases in extreme wind speeds will vary widely, both within and between regions due to differences in the underlying vulnerability of forests in these regions. These impacts are likely to range from little or no change, to a significant increase in the risk of wind damage to planted forests.

Changes in risk from exotic pests

Biotic factors are also likely to be strongly influenced by climate change. As the distribution of weeds, insects and pathogens is strongly determined by climatic conditions, changes in these conditions are likely to result in shifts in the geographic range of many species. Under climate change, global regions from which future invasions might occur are also likely to shift. Simulations suggest that the global areas that pose an invasion risk for New Zealand could expand, and this report highlights the new areas that might currently harbour potential future invaders.

Plant pests - Climate change is likely to result in range expansion within New Zealand of a number of native Australian tree species, such as *Eucalyptus* and *Acacia* spp. and the proliferation of currently ornamental and potentially invasive

weed species such as *Melaleuca quinquenervia* (broad-leaved paperbark) and *Pueraria montana* (kudzu). Climate change is also likely to affect growth rates of weeds through changes in CO₂ concentration, root-zone water storage, temperature and changing length of the growing season. These changes to weed composition and growth rates resulting from climate change are likely to have a detrimental effect on tree growth.

Insect pests - Because of our limited knowledge of climate effects on forest insects in New Zealand, it is difficult to make any generalised assessment about likely climate-change effects on their abundance and distribution. However, it is likely that climate change will increase the risk of establishment of new species from warm-temperate or subtropical regions and generally result in greater abundance of insect pests due to their better survival over winter. Without further research it is difficult to determine how changes in insect abundance and distribution will influence plantation productivity.

Pathogens - Climate change is unlikely to greatly influence the distribution of the main needle-cast diseases of radiata pine in New Zealand as these are already found throughout the country. Climate change is, however, likely to change the regional incidence and severity of some fungal diseases. It is likely that *Dothistroma* needle blight will remain rare in dry east coast regions, which are projected to become even more arid under climate change. The severity of Swiss needle cast, caused by *Phaeocryptopus gaeumannii*, which is the most widespread disease of Douglas-fir is likely to increase with climate change throughout the country as pathogen abundance is strongly correlated with winter air temperature. For the major diseases of radiata pine, changes to pathogen distribution and abundance will need to be quantified before the influence of these on plantation productivity can be determined.

Under climate change, there are also a number of interactions between abiotic and biotic factors, which need to be considered in order to gain a full understanding of likely climate change effects on productivity. These are described fully in the report.

Framework

The numerous interactions between plantation productivity, climatic variables and secondary effects described in this report demonstrate the complexity of the system and the necessity to model the effects of climate change on plantations in an integrated manner. We present such a framework which could be used to determine the effects of climate change on plantation productivity. At the core of the framework is a growth model which can account for the ecophysiological effects of climatic change on plantation productivity. This core model is linked to sub-models accounting for the influence of additional biotic (weeds, pests, diseases) and abiotic (wind, fire) factors on plantation productivity.

The key results and the proposed framework described above have identified a possible approach to quantify the overall effects of climate change on New Zealand's production forests in an integrative way. Key knowledge gaps have also been identified, and a comprehensive research plan to address these uncertainties is presented at the end of section four.

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Introduction

Forests cover about one quarter of the Earth's land surface area. They play a major role in the present and projected future carbon budget as they contain about 80% of all above-ground and 40% of below-ground terrestrial organic carbon. They can become carbon sources when forests are harvested, degraded or damaged or carbon sinks when new forests are planted or growth rates are enhanced.

In New Zealand, the total area of forest cover is approximately 8.2 million hectares, comprising 6.4 million hectares of indigenous forests and 1.8 million hectares of planted forests. Within these planted forests, 89% of the net stocked area is comprised of radiata pine (MAF, 2007). Douglas-fir is the next most common species, making up 6%, with the remainder stocked in hardwoods, cypresses and other softwoods. As new forest planting peaked from 1992 to 1998 (average planted area of 69 000 ha year⁻¹) a large part of the plantation resource is now between 10 and 16 years old (MAF, 2007). Plantations established since 1990 on previously unforested land are potentially eligible to be counted as carbon sinks to counterbalance the emissions during the first Kyoto commitment period (2008 – 2012), and estimates of their total area range from 0.6 to 0.8 million hectares (MAF, 2007).

As of 2006 (MAF, 2007) most of the forest resource within New Zealand was located in the central North Island (550 761 ha), followed by Otago/Southland (215 662 ha), Northland (203 067 ha), Nelson/Marlborough (171 418 ha), southern North Island (168 447 ha), the East Coast of the North Island (157 009 ha), Hawkes Bay (131 768 ha), and Canterbury (113 903 ha). Relatively small areas were also located in Auckland (56 008 ha) and on the West Coast of the South Island (32 009 ha). Because of the current dominance of radiata pine within New Zealand's planted forests, coupled with its wide geographic distribution and its overall economic importance, it is important to understand the response of this species to possible changes in future environmental conditions driven by climate change. A thorough understanding of these changes will enable New Zealand's forestry sector to best adapt itself to the potential impacts of future climate change.

The Fourth Assessment Report of the IPCC (2007c) outlined the scope, vulnerability and the associated uncertainty of climate change on forests globally. Overall the outline drawn highlights a complex picture of positive and negative effects of climate change on forests. The overall expectation is that commercial forest productivity will rise at least modestly in the short and medium term, in response to increases in temperature and CO₂ concentration. This will be counteracted by increases in pest and pathogen outbreaks and increased risk of plantation damage from fire and wind. The IPCC also suggest that the increase in productivity will show a large regional variability and will change from low-latitude regions in the short term to high-latitude regions in the long term.

Although the IPCC provides an overview of how climate change is likely to influence forestry within Australasia, these estimates are necessarily broad, and we do not currently have detailed information describing how climate change is likely to influence plantation productivity in New Zealand at a more local level. Changes in plantation productivity which result from climate change include direct physiological responses to the changes in climate. Climate change will also influence distribution and abundance of many forest pest species (weeds, insects and pathogens) and alter the frequency and intensity of damaging abiotic factors, such as wind and fire.

For this reason, a comprehensive assessment of changes in productivity has to also account for the impacts of changes in these factors as well.

In the first part of this report we will synthesise current knowledge on likely climatic changes in New Zealand over the course of this century. The second part summarises the current knowledge of the likely direct climate-change impacts on plantation productivity. In the third part we examine current knowledge of likely climate-change effects on abiotic (wind and fire risks) and biotic (insects, diseases, weeds) factors, and how changes in these secondary factors are likely to influence plantation productivity. Finally, in the fourth part we develop an interdisciplinary framework which can be used to integrate the direct and indirect climate change effects on plantation productivity. A key aim throughout the first three objectives is to highlight any knowledge gaps or uncertainties, and approaches which can be used to address these issues. The most important issues are summarised in a comprehensive research plan at the end of the report.

1 Our current climate and how it is changing

Summary

Objective

Synthesise current knowledge on likely climatic changes in New Zealand over the course of this century. Indicate any knowledge gaps, the degree of certainty in the projections, and describe what further research could be undertaken to increase our confidence in these projections.

Key results

The mean annual temperature for all of New Zealand is projected to increase by around 2°C from 1990 to 2090. This projection is based on the average of statistically downscaled projections from 12 global climate models, run with the mid-range “A1B” greenhouse gas emissions scenario. Annual rainfall projections (based on the same models and scenario) indicate a likely difference in trends, with the western side of both islands trending towards increasing rainfall while the eastern side trends towards decreasing rainfall. The far north is also projected to receive less rainfall on an annual basis (a 5–10% decrease by 2090 from the 1980–99 average). Seasonally, the largest projected decreases (by as much as 20% by 2090) are for the north and east of the North Island in winter and spring and the largest projected increases are for the southwest of the South Island in winter (up to 40%). Confidence in these projections is higher for temperature than for rainfall, as there is a significant amount of inter-model variability (particularly for the seasonal rainfall projections) between different climate models. This uncertainty has to be considered in any impacts assessment.

The projected increase in temperature is likely to lead to a significant reduction of frost days, an increase in hot days and an increase in the intensity of heavy rainfall events. For every 1 °C increase in temperature there is an approximate 8% increase in the 24-hour 100-year return period rainfall amount. Drought risk is likely to increase in currently drought-prone areas where reductions in rainfall and increases in evaporation are anticipated. There are indications that the annual mean westerly wind component may increase by about 10% by 2090, and that high winds (wind speeds exceeded 1% of the time) may increase by around 2–3% (averaged over all of New Zealand, but as high as 10% in some eastern locations).

1.1 Overview of New Zealand's climate

New Zealand's climate is complex and varies from warm subtropical in the north to cool temperate climates in the south, with severe alpine conditions in the mountainous areas. Mountain chains extending the length of New Zealand provide a barrier for the prevailing westerly winds, dividing the country into dramatically different climate regions. The West Coast of the South Island is the wettest area of New Zealand, whereas the area to the east of the mountains, just over 100 km away, is the driest.

Most areas of New Zealand receive between 500 and 2 000 mm of rainfall (Fig. 1.1) spread throughout the year. Over the northern and central areas of New Zealand rainfall is generally greater in winter than in summer, whereas for much of the southern part of New Zealand, winter is the season of least rainfall.

Median annual temperatures range from 8°C in the south to 18°C in the north of New Zealand (Fig. 1.2). The coldest month is usually July and the warmest month is usually January or February. In New Zealand generally there are relatively small variations between summer and winter temperatures, although inland and to the east of the ranges the variation is greater (up to 14°C). Extended periods (greater than two or three days) with extremely hot (e.g. maxima above 35°C) or cold (e.g. minima below -5°C) days are very rare in low elevation areas of New Zealand, though temperatures drop about 0.7°C for every 100 m of altitude so cold periods are more frequent at high elevations.

Sunshine hours are relatively high in areas that are sheltered from the west and most of New Zealand receives at least 1 800 hours annually (Fig. 1.3). The midday summer solar radiation index (UVI) is often very high in most places and can be extreme in northern New Zealand and in mountainous areas. Autumn and spring UVI values can be high in most areas.

Heavy rainfall events are relatively frequent for New Zealand. At least one 100-year return period event generally occurs somewhere in New Zealand every year, often leading to flooding of rivers. The 100-year average recurrence interval (ARI) 24-hour rainfall total is between 100 and 200 mm for many lower elevation parts of New Zealand. This ARI rises up to around 400 mm in higher elevation areas in the north of both islands and in the east of the North Island. In parts of the Southern Alps, 24-hour rainfall totals exceed 600 mm once in 100 years, on average (Fig. 1.4).

Most snow in New Zealand falls in the mountain areas. Snow rarely falls in the coastal areas of the North Island and west of the South Island, although the east and south of the South Island may experience some snow in winter. Frosts can occur anywhere in New Zealand but are rare in the northern North Island. They usually form on cold nights with clear skies and little wind.

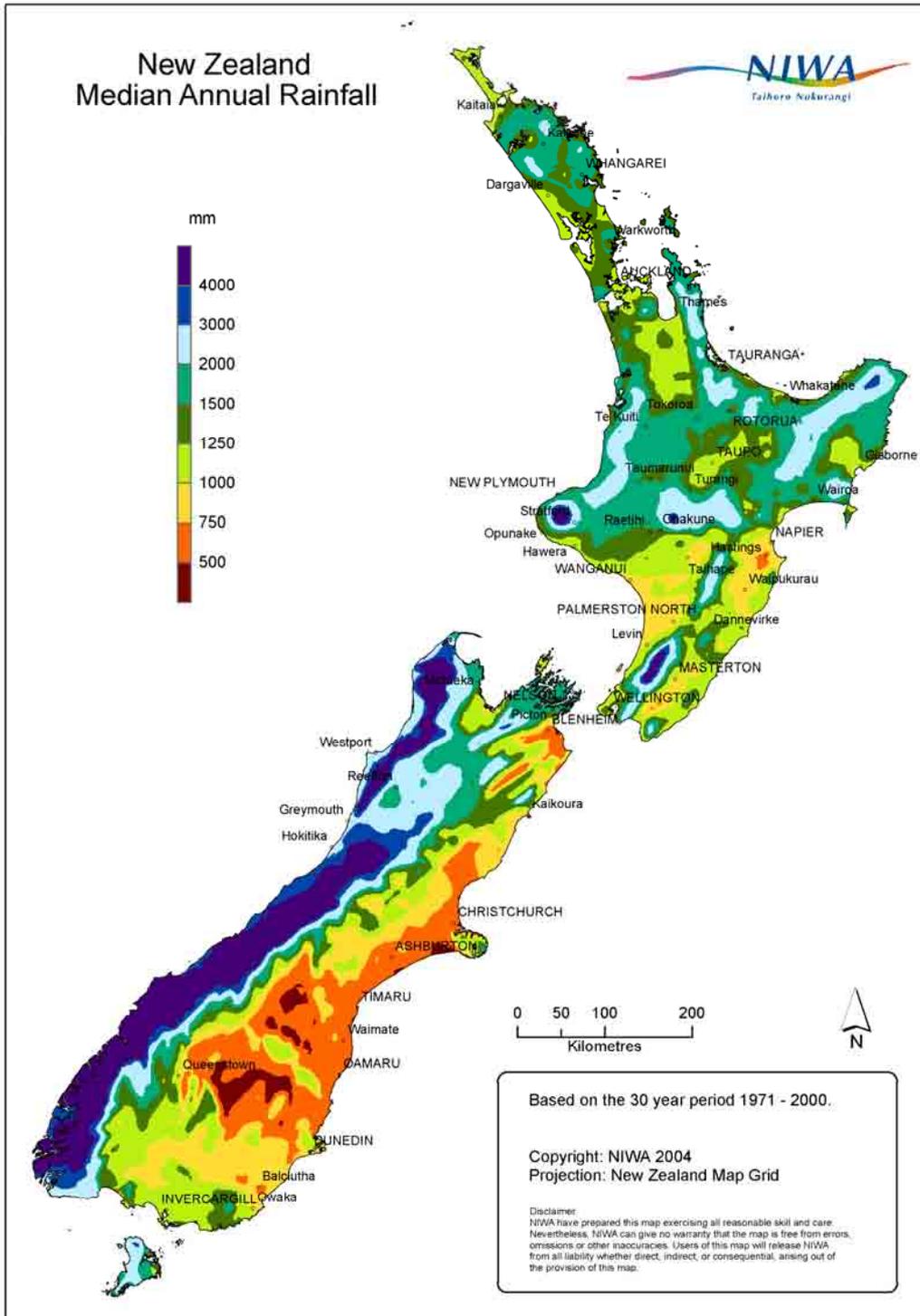


Figure 1.1. Median annual rainfall total (mm), based on the 30-year period 1971–2000.

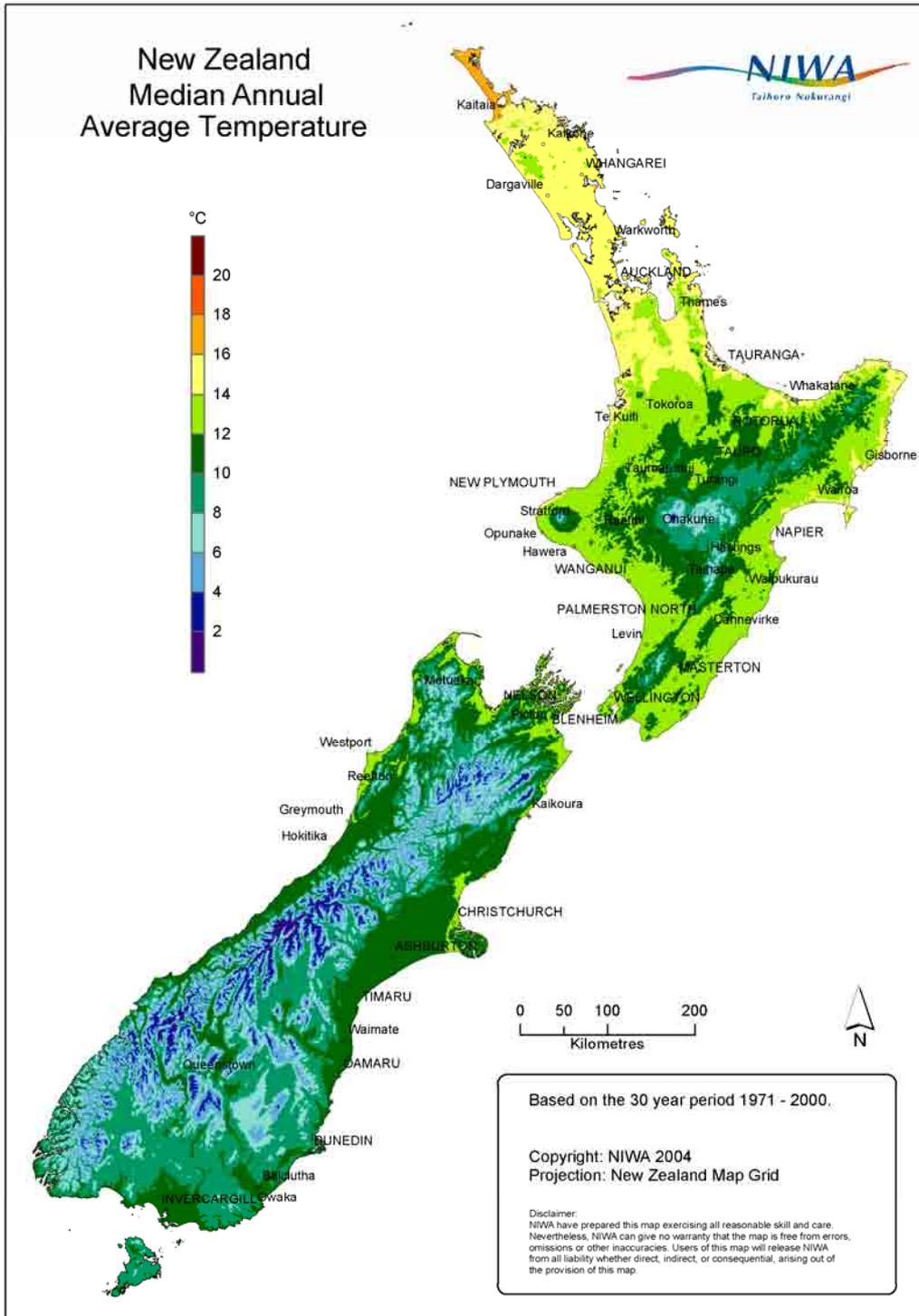


Figure 1.2. Median annual average temperature (°C), based on the 30-year period 1971–2000.

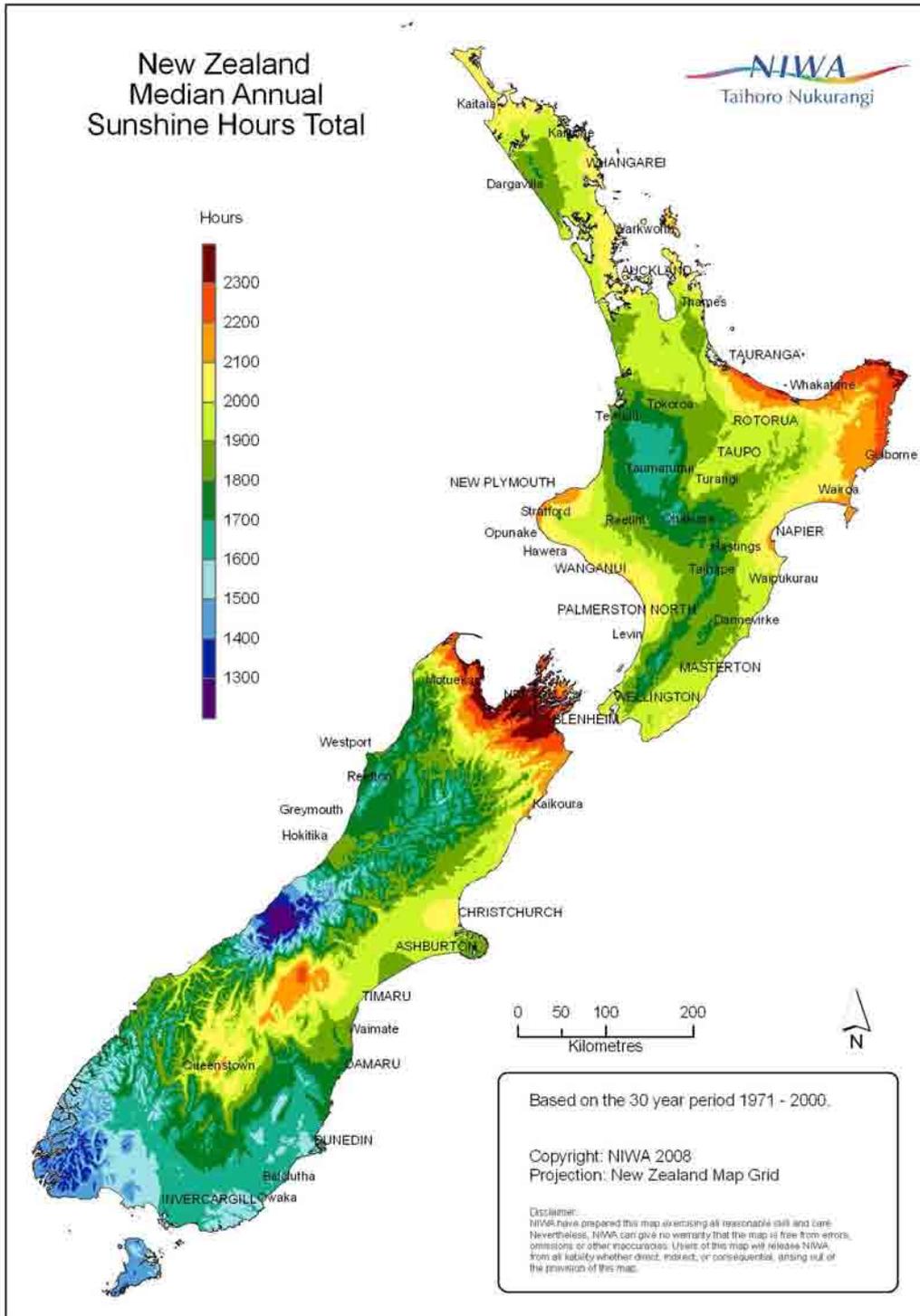


Figure 1.3. Median annual sunshine hours total (hours), based on the 30-year period 1971–2000.

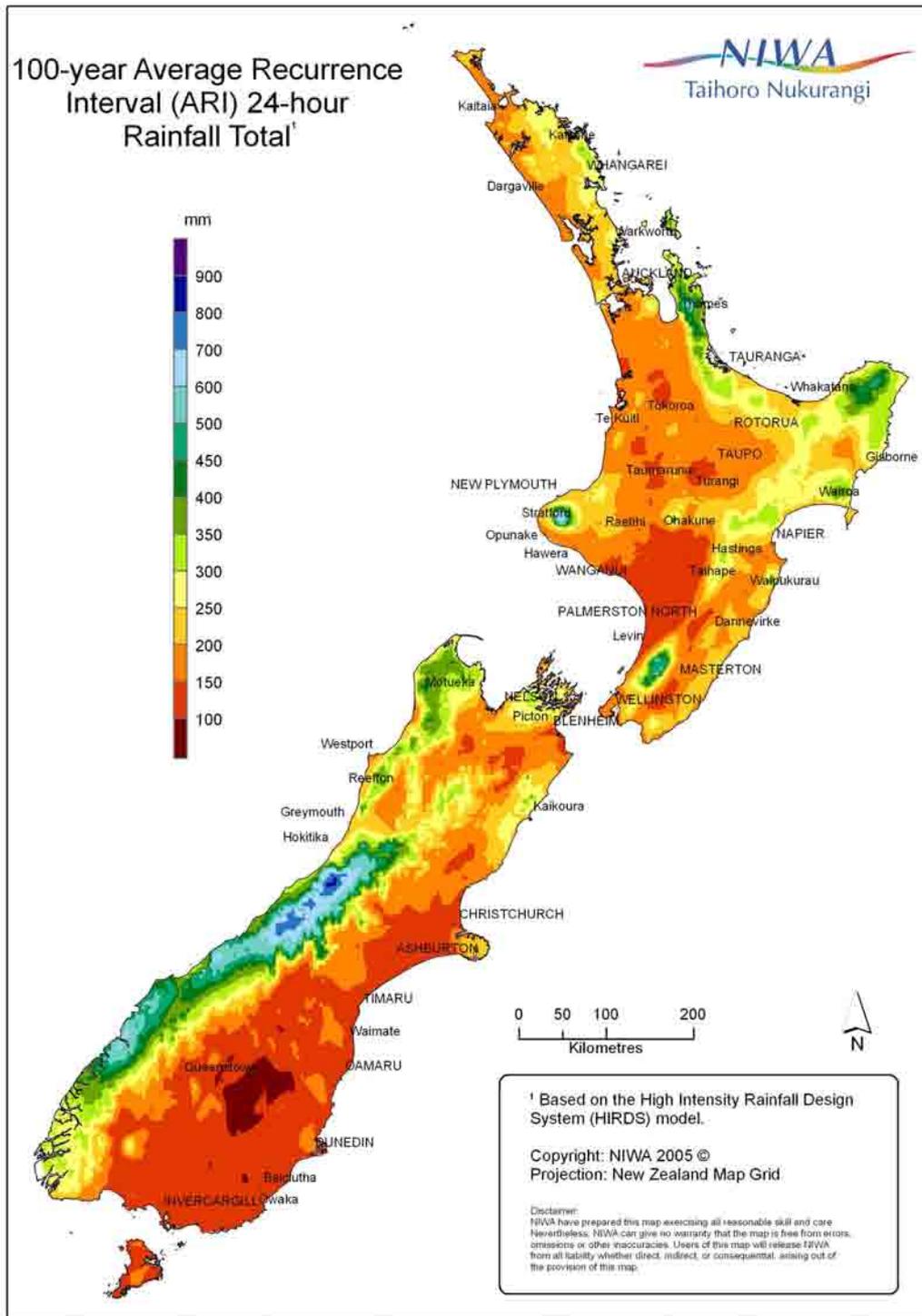


Figure 1.4. 100-year Average Recurrence Interval (ARI) 24-hour rainfall total (mm), based on the High Intensity Rainfall Design System (HIRDS) model.

1.1.1 Recent climate variations in New Zealand

An in-depth study of trends in several climate variables for New Zealand for the period 1951–90 was carried out by Salinger *et al.*, (1995). Mean air temperatures in New Zealand increased by 0.12°C/decade over this period (and have continued to increase at a similar rate up to the present day). Minima increased at almost twice the rate of maxima (0.15 cf. 0.08°C/decade) with a consequent decrease in

temperature range over the whole day (known as the diurnal temperature range, or DTR) and the number of frosts. Sunshine hours also decreased (-0.6%/decade) over the forty year period, as did cloud cover.

In New Zealand, the responses of maximum and minimum air temperature, DTR, sunshine and cloud amount are all dominated by the interaction of regional atmospheric circulation with the hills and mountains. Overall, westerly circulation has increased in the New Zealand region over these four decades. Five of the six regions listed in Table 1.1 exposed to this westerly circulation show DTR decreases, but only one (Southland) shows a significant decrease in sunshine.

Table 1.1. Trends in annual mean, maximum and minimum air temperature, Diurnal Temperature Range (°C/decade), sunshine (%/decade) and 0900 cloud amount (oktas/decade) for New Zealand over the period 1951–1990. Asterisks indicate significance at the 95% confidence level (no asterisk indicates not statistically significant).

Region	Mean temp	Max. temp	Min. temp	DTR	Sunshine total	0900 cloud amount
New Zealand	0.12	0.08	0.15	-0.07	-0.6	-0.07*
Western NI	0.13	0.07	0.18	-0.11	0.2	-0.10*
Eastern NI	0.16	0.14	0.18	-0.04	0.1	-0.04
Western SI	0.18	0.11	0.07	0.03	-1.6	-0.01
Eastern SI	0.10	0.07	0.12	-0.06	0.0	-0.03
Inland Central	0.11	0.04	0.19	-0.15*	0.4	0.04
Southland	0.14*	0.06	0.22*	-0.16*	-1.9*	0.01

Precipitation increased in the south west Pacific to the north east of the South Pacific Convergence Zone (SPCZ) over the 1951–90 period, whilst precipitation decreases occurred near and south west of the SPCZ. For New Zealand, distinct regional changes in precipitation have occurred since 1930 (Salinger and Mullan, 1996). Summer rainfall was higher in North Canterbury, and lower in the north and west of the South Island from 1930–50. Increased east and north east airflow occurred over New Zealand during the period 1951–75. This brought increased rainfall in the north of the North Island, particularly in autumn, with decreased rainfall in the south east of the South Island, especially over the summer. The 1976–94 period is notable for several strong El Niño events, such as 1977/78 and 1982/83. Significant annual rainfall trends have occurred with rainfall decreases in the north of the North Island, and increases in much of the South Island, except the east. Rainfall has increased in winter in all parts of New Zealand except the south east of the South Island.

In New Zealand and the South Pacific, the El Niño Southern Oscillation (ENSO) is a significant source of seasonal and year-to-year climate variability (Nicholls, 1992). The Southern Oscillation Index (SOI) explains up to 40% of year-to-year air temperature variations in the tropical South Pacific. When the Southern Oscillation Index (SOI) is positive, mean annual air temperature anomalies are positive (temperatures are warmer than normal) in the area south west of the SPCZ (includes New Zealand), and negative (temperatures are cooler than normal) to the north east of the SPCZ (Salinger *et al.*, 1996). Opposite air temperature anomalies occur during the El Niño phase (SOI negative).

1.1.1.1 What is the El Niño?

The El Niño is a natural feature of the global climate system. Originally it was the name given to the periodic development of unusually warm ocean waters along the tropical South American coast and out along the Equator to the dateline, but now it is

more generally used to describe the whole "El Niño - Southern Oscillation (ENSO) phenomenon", the major systematic global climate fluctuation that occurs at the time of the ocean warming event.

When an El Niño is not present, trade winds blow westward across the Pacific, piling up warm surface water so that Indonesian sea levels are about 50 cm higher than those in Ecuador. Cool, nutrient-rich water wells up off the South American coast, supporting marine ecosystems and fisheries. Rainfall occurs in rising air over the warm water to the west, and the east Pacific is relatively dry.

During an El Niño, the trade winds weaken, leading to a rise in sea surface temperature in the eastern equatorial Pacific and a reduction of upwelling off South America. Heavy rainfall and flooding occur over Peru, and drought over Indonesia and Australia. The supplies of nutrient-rich water off the South American coast are cut off due to the reduced upwelling, adversely affecting fisheries in that region. In the tropical South Pacific the pattern of occurrence of tropical cyclones shifts eastward, so there are more cyclones than normal in areas such as the Cook Islands and French Polynesia.

The Southern Oscillation Index (SOI) is calculated from the pressure difference between Tahiti and Darwin (Fig. 1.5). Anomalously low values of this index correspond to El Niño conditions, while the opposite conditions with an anomalously high SOI value are called La Niña episodes. El Niño events occur about 3 to 7 years apart, typically becoming established around April or May and persisting for about a year thereafter.

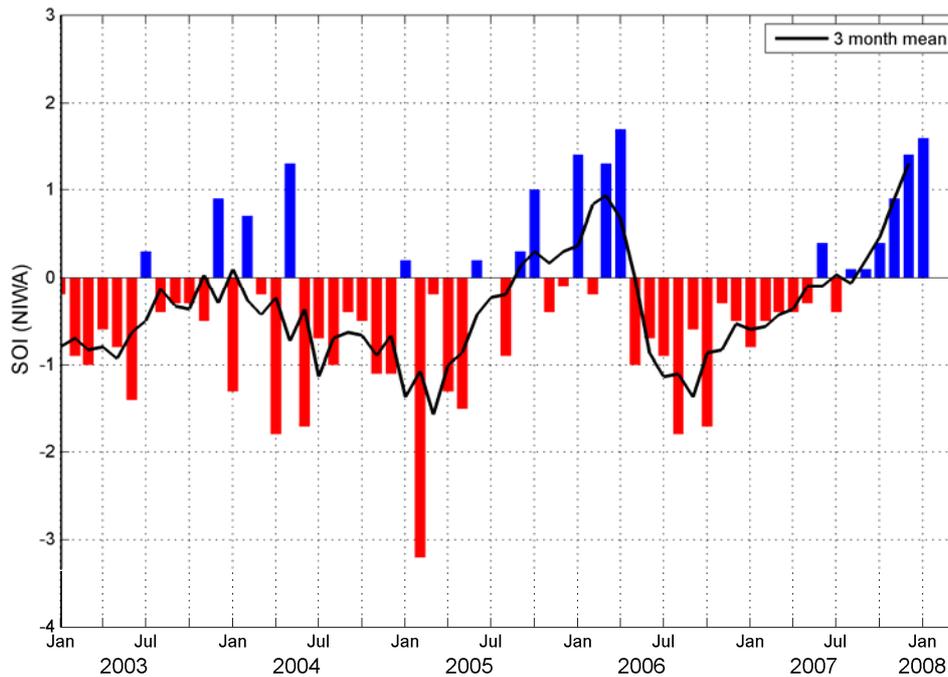


Figure 1.5. A Plot of Southern Oscillation Index (SOI) derived from pressure measurements at Tahiti and Darwin for the period January 2003 – January 2008 inclusive. Values of the 3-month mean below -1 are El Niño events and above +1 are La Niña events.

1.1.1.2 How does an El Niño typically affect New Zealand?

New Zealand is not usually affected as strongly by El Niño conditions as are parts of Australia, but there is nevertheless a significant influence. In El Niño years, New Zealand tends to experience stronger or more frequent winds from the west in summer, leading to drought in east coast areas and more rain in the west. In winter, the winds tend to be more from the south, bringing colder conditions to both the land and the surrounding ocean. In spring and autumn southwesterlies tend to be stronger or more frequent, providing a mix of the summer and winter effects.

The La Niña events which occur at the opposite extreme of the Southern Oscillation Index cycle have weaker impacts on New Zealand’s climate. New Zealand tends to experience more northeasterly winds, which bring more moist, rainy conditions to the northeast parts of the North Island and drier than normal conditions to the southwest of the country (as seen in the summer of 2007/08).

Although the El Niño is an important influence on New Zealand’s climate, it accounts for less than 25% of the year-to-year variance in seasonal rainfall and temperature at most New Zealand measurement sites. East coast droughts may be common during El Niños, but they can also happen in non-El Niño years (for example, the severe 1988-89 drought). Serious east coast droughts do not occur in every El Niño, and the districts where droughts occur can vary from one El Niño to another (although some are more consistently affected than others). However the probabilities of the climate variations discussed above happening in association with El Niño are sufficient to warrant management actions and planning to be taken when an El Niño is expected or in progress.

1.1.1.3 Decadal climate variability

NIWA scientists have recently identified a long-lasting "shift" in New Zealand's climate that occurred around 1977 (Salinger and Mullan, 1999). The shift was characterised by more persistent westerlies on to central New Zealand since 1977, resulting in the west and south of the South Island being about 10% wetter and 5% cloudier with more damaging floods. The north and east of the North Island have been 10% drier and 5% sunnier on average, compared to 1951–76 data. This change-point of 1977 coincided with an eastward movement in the longitude of the South Pacific Convergence Zone, and more frequent El Niño events in the recent record.

This shift is probably due mainly to a Pacific-wide natural fluctuation that is being called the Pacific Decadal Oscillation (PDO), (Mantua *et al.*, 1997), which exhibits phase reversals about once every 20–30 years. The influence of the PDO is well-known in the North Pacific, and has recently also been noted in Australian rainfall (Power *et al.*, 1998). Scientists from Pacific Island countries attending a workshop in Auckland in November 2001 put out a press release suggesting the PDO underwent another phase reversal in 1998.

Long-term warming trends are superimposed on these decadal climate variations. Individual El Niño events bring cooler conditions to New Zealand. Since 1977, however, temperatures have continued to rise, resulting in warmer night time temperatures and fewer frosts nationwide, and an increase in very hot days in eastern areas.

1.2 The natural greenhouse effect

The greenhouse effect is a warming of the earth's surface and lower atmosphere caused by substances such as carbon dioxide and water vapour. These substances let the sun's energy through to the ground but impede the passage of energy from the earth back into space.

Energy emitted from the sun ("solar radiation") is concentrated in a region of short wavelengths including visible light. Much of the short wave solar radiation travels down through the Earth's atmosphere to the surface virtually unimpeded. Some of the solar radiation is reflected straight back into space by clouds and by the earth's surface. Much of the solar radiation is absorbed at the earth's surface, causing the surface and the lower parts of the atmosphere to warm.

The warmed Earth emits radiation upwards, just as a hot stove or bar heater radiates energy. In the absence of any atmosphere, the upward radiation from the Earth would balance the incoming energy absorbed from the Sun at a mean surface temperature of around -18°C , 33° colder than the observed mean surface temperature of the Earth. The presence of "greenhouse" gases in the atmosphere accounts for the temperature difference. Heat radiation (infra-red) emitted by the Earth is concentrated at long wavelengths and is strongly absorbed by greenhouse gases in the atmosphere, such as water vapour, carbon dioxide and methane. Absorption of heat causes the atmosphere to warm and emit its own infra-red radiation. The Earth's surface and lower atmosphere warm until they reach a temperature where the infra-red radiation emitted back into space, plus the directly reflected solar radiation, balance the absorbed energy coming in from the sun. As a result, the surface temperature of the globe is around 15°C on average, 33°C

warmer than it would be if there was no atmosphere. This is called the natural greenhouse effect.

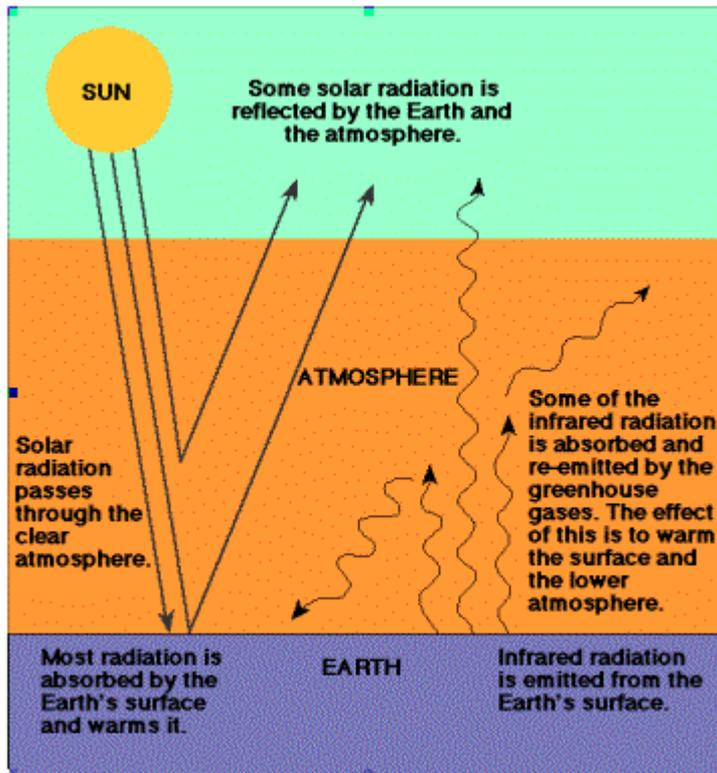


Figure 1.6. A simplified diagram illustrating the greenhouse effect (based on a figure in the 1990 IPCC Science Assessment).

1.3 The effect of increased Greenhouse Gas concentrations

If extra amounts of greenhouse gases are added to the atmosphere, such as from human activities, then they will absorb more of the infra-red radiation. The Earth's surface and the lower atmosphere will warm further until a balance of incoming and outgoing radiation is reached again (the emission of infra-red radiation increases as the temperature of the emitting body rises). This extra warming is called the enhanced greenhouse effect.

The magnitude of the enhanced greenhouse effect is influenced by various complex interactions in the earth-ocean-atmosphere system, which are not included in the discussion above. For example, as the temperature of the earth's surface increases more water vapour is evaporated. Since water vapour is itself a strong greenhouse gas, this is a positive feedback which will tend to amplify the warming effect of (for example) carbon dioxide emissions. Clouds tend both to cool the Earth because they reflect incoming sunlight, and to warm it by trapping outgoing infra-red radiation. The net result over the globe of clouds is a cooling, but it is still uncertain whether this overall cooling will increase or decrease as greenhouse gas concentrations increase. Heat is distributed vertically in the atmosphere by motion, turbulence and evaporation and condensation of moist air, as well as by the radiative processes discussed above.

Thus many processes and feedbacks must be accounted for in order to realistically predict climate changes resulting from particular greenhouse gas emission scenarios. These complications are the source of much of the debate which has occurred about the likely magnitude and timing of climate changes due to enhanced greenhouse gas emissions.

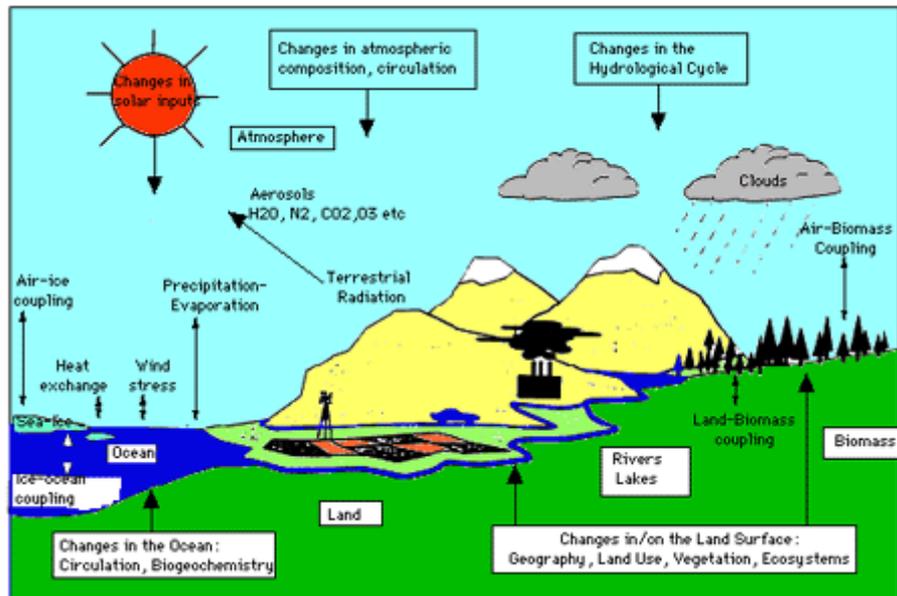


Figure 1.7. A schematic view of many of the processes and interactions in the global climate system (based on Figure 1.1 of the 1995 IPCC Science Assessment).

1.4 Climate change scenarios for New Zealand

Predicting human-induced ("anthropogenic") changes in climate, over the next 100 years, for a particular part of New Zealand requires:

- A prediction of global greenhouse gas and aerosol emissions for the next century.
- A global carbon cycle model to convert these emissions into changes in carbon dioxide concentrations (and similar models for calculating concentrations of other greenhouse gases and aerosols).
- A coupled atmosphere-ocean global circulation model (AOGCM) which uses the greenhouse gas and aerosol concentration information to predict climate variations forward in time.
- Downscaling of the AOGCM results through a procedure which takes account of the influence of New Zealand's topography on local climate. This can be done either statistically or with a high resolution regional climate model.

Given our current knowledge and modelling technology, there are uncertainties in each of these steps. For example, emission predictions depend on the difficult task of predicting human behaviour, including predicting future greenhouse gas emissions. Our understanding of the carbon cycle, and of sources and sinks of non-carbon dioxide greenhouse gases, is still incomplete. Also, there are significant uncertainties in current global climate model predictions, particularly at the regional scale.

The climate change scenario approach recognises these uncertainties. A scenario is a scientifically-based projection of one plausible future climate for a region. For

guidance on regional impacts of climate change, a range of scenarios is desirable. These can span credible estimates of future greenhouse gas emissions, and the uncertainty range in climate model predictions.

The Intergovernmental Panel on Climate Change (IPCC) developed 35 different emissions scenarios in its Special Report on Emission Scenarios (SRES), (Nakicenovic & Swart, 2000). These SRES scenarios cover a range of demographic, societal, economic, and technical-change "storylines" and formed the basis for much of the climate projection work done for the IPCC's assessment reports. The SRES scenarios do not include specific initiatives to control greenhouse gas emissions, such as the Kyoto Protocol, but some of them (e.g. the B1 scenario) include the introduction of clean and resource-efficient technologies. Full AOGCMS (atmosphere-ocean global circulation models) were run for only some of these scenarios. Instead a simpler globally-averaged model was then "tuned" to these AOGCM runs and applied to all 35 SRES scenarios. The IPCC does not contend that any one SRES scenario is more likely than any other – it is as if they have provided a dice for predicting future conditions with 35 equally weighted sides.

Figure 1.8 from the IPCC Third Assessment Report (TAR) illustrates this approach. (a) shows the CO₂ emissions for six illustrative SRES scenarios along with the IS92a "mid-range" scenario used in the Second Assessment Report. (b) shows projected CO₂ concentrations corresponding to these emissions. (c) shows projected anthropogenic sulphur dioxide emissions. The SRES also includes projections for other greenhouse gases and aerosols. Figures (d) and (e) are the projected temperatures and sea levels resulting from the emission scenarios which illustrate that even for a single emissions scenario there is a range of possible temperature projections, reflecting differences between climate models. Note that the real changes in global average temperature over the next century will not be as smooth as in this figure – irregular fluctuations due to natural oscillations in the ocean / atmosphere system, volcanic eruptions, and changes in solar activity will be superimposed.

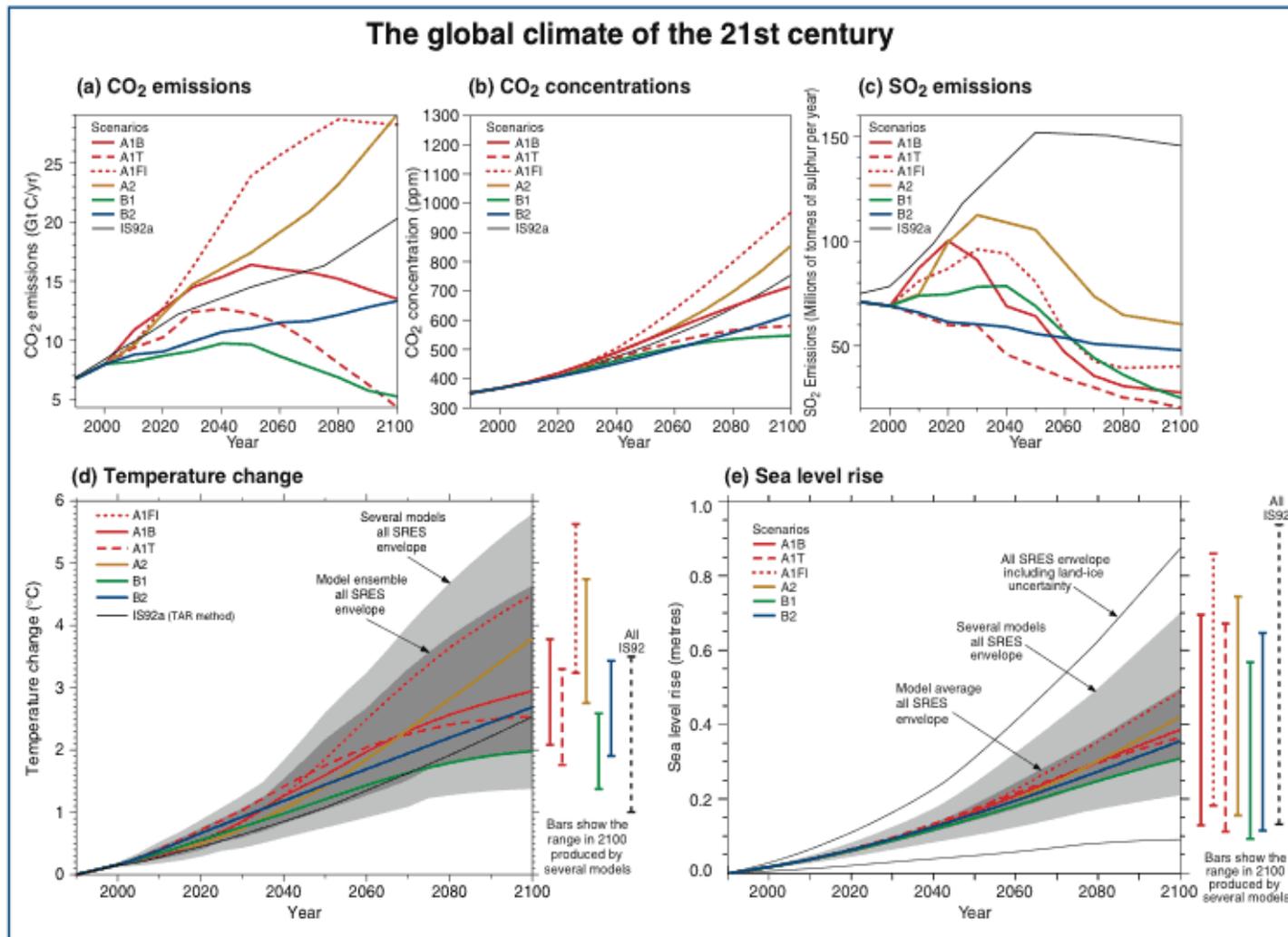


Figure 1.8. Global Climate Scenarios for the 21st Century (from Figure 5 of the WGI Summary for Policymakers of the IPCC's Third Assessment Report, IPCC 2001a).

1.4.1 Downscaling to New Zealand

The global climate models predict trends in broad climate patterns across the Pacific, but do not take account of the effect of New Zealand's topography on the local climate. Approaches which can be taken to infer local changes corresponding to a particular global model simulation include (a) nesting a higher resolution regional climate model covering New Zealand and its surroundings within the global model simulations, and (b) using statistical techniques based on present relationships between regional-scale climate patterns and local climate. Most of the scenarios which have been used so far for assessing the likely impacts of climate change on New Zealand have used statistical downscaling. However scientists are now evaluating initial regional climate modelling results from a system implemented at NIWA using the UK Meteorological Office's unified climate model, with a view to using this to develop new scenarios.

1.4.1.1 The current approach of statistical downscaling

Statistical downscaling starts with historical observations, and calculates "downscaling relationships" between broad regional climate patterns and local climate observations. The downscaling relationships are then applied to the broad future regional climate patterns predicted by the global models, in order to provide more locally-detailed projections for New Zealand (e.g. Mullan *et al.*, 2001b). For the New Zealand scenarios described in the next section we undertook statistical downscaling for two periods: The 20-year period centred around 2040 (2030–2049) and the 20-year period centred around 2090 (2080–2099). We undertook statistical downscaling from 12 different AOGCMs for each of the future periods for a mid-range greenhouse gases emission scenario (A1B – see Fig. 1.8). Scaling factors based on the globally-averaged temperature projections shown in Figure 1.8 can be used to assess the consequences of other emission scenarios.

1.4.2 New Zealand regional climate change scenarios

NIWA scientists followed the downscaling approach used above to prepare New Zealand climate change scenarios for 2040 and 2090, for a "Guidance Manual" for local government organisations. This was published on the web by the Ministry for the Environment in May 2008 (Ministry for the Environment, 2008). Following IPCC practice, NIWA did not indicate a "most likely value" from within these ranges of projections. However, the extreme ends of the range may be slightly less likely than the central values, since they generally result from the climate model which gives the most extreme projection rather than reflecting agreement between a number of models. NIWA also cannot totally rule out changes outside of the scenario range, because of the uncertainties in regional climate modelling. Nevertheless, NIWA believes that these scenarios are a useful tool for central, regional and local government and infrastructure operators, providing a set of plausible futures to use for assessing possible impacts on climate-sensitive activities and functions.

The Guidance Manual contains full details of the climate projections, including tables showing projected ranges for temperature and rainfall changes in various parts of the country and for different seasons. Figures 1.9 and 1.10 below show the 12-model average projected New Zealand temperature and rainfall changes for 2040 and 2090 (based on the A1B emission scenario).

The projected temperature changes for New Zealand are smaller than the globally averaged changes for the corresponding SRES scenario (A1B), because the oceans

surrounding New Zealand are projected to warm more slowly than the globe as a whole for the projection period. For annual mean temperature (Fig. 1.9) by 2090, the mid-IPCC scenario is for approximately 2.0°C warmer over most of the country, with slightly less warming in the south. Seasonally, the strongest warming is projected in winter.

For annual rainfall, the projections (Fig. 1.10) indicate a strong southwest to northeast gradient. This occurs because many of the models project an increase in the frequency or magnitude of airflow across the country from the south-west or west, which tends to bring rain to the West Coast and the main mountain chains and less to the northeast. There is a lot of variability between models, and for some locations even the sign of the change cannot be stated with any confidence. However for 2090 annual mean rainfall, the regions of Taranaki, Manawatu-Wanganui, West Coast, inland Otago, and Southland tend to show increased rainfall for all scenarios, compared to Hawke's Bay and Gisborne which show rainfall decreases. Seasonally, the largest projected percentage decreases are for the north and east of the North Island, in winter and spring.

A note of caution: There is a significant amount of inter-model variability particularly in the projections of seasonal rainfall. This uncertainty, which is masked when merely considering the 12-model average, needs to be fully taken into account in a risk-based assessment framework.

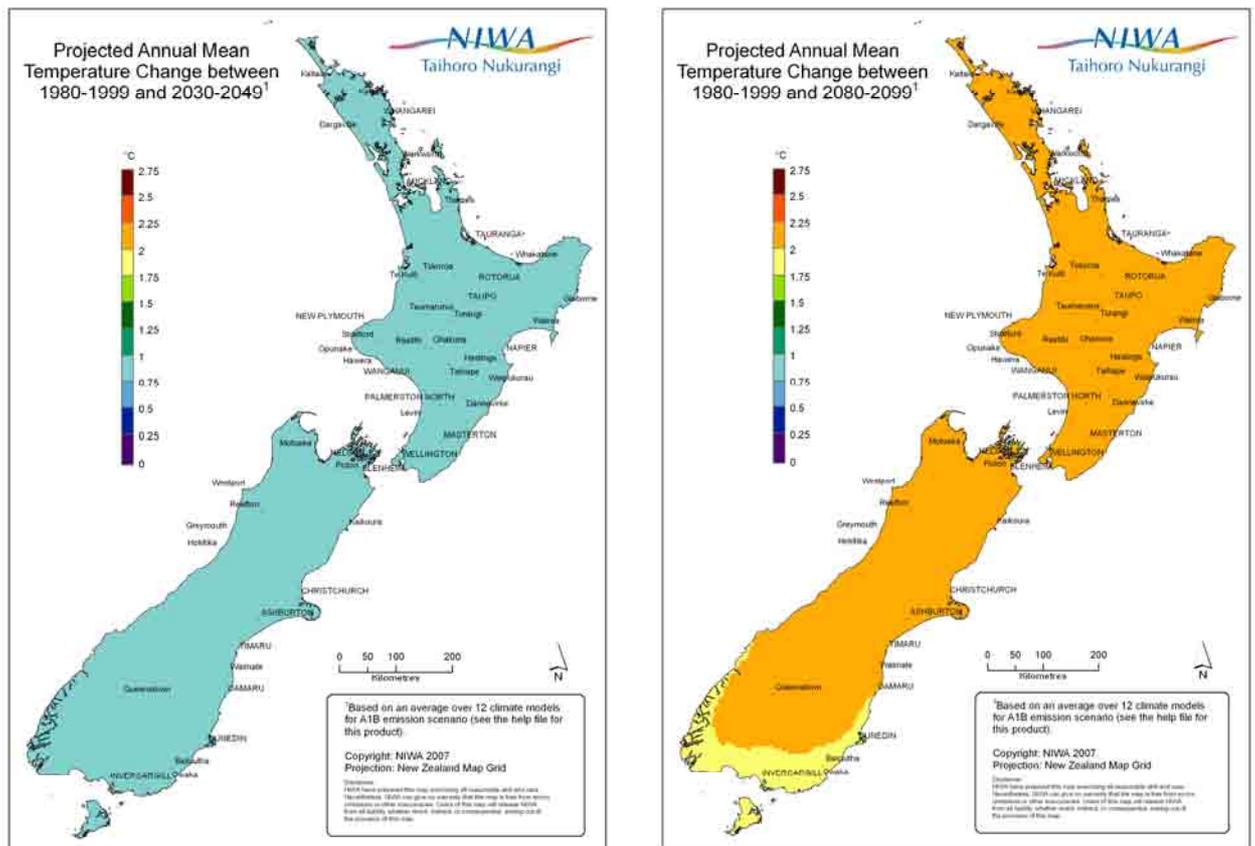


Figure 1.9. New Zealand projected changes in annual temperature (in °C) to 2040 and 2090, relative to 1990, based on a 12-model average of projections and for the A1B emission scenario.

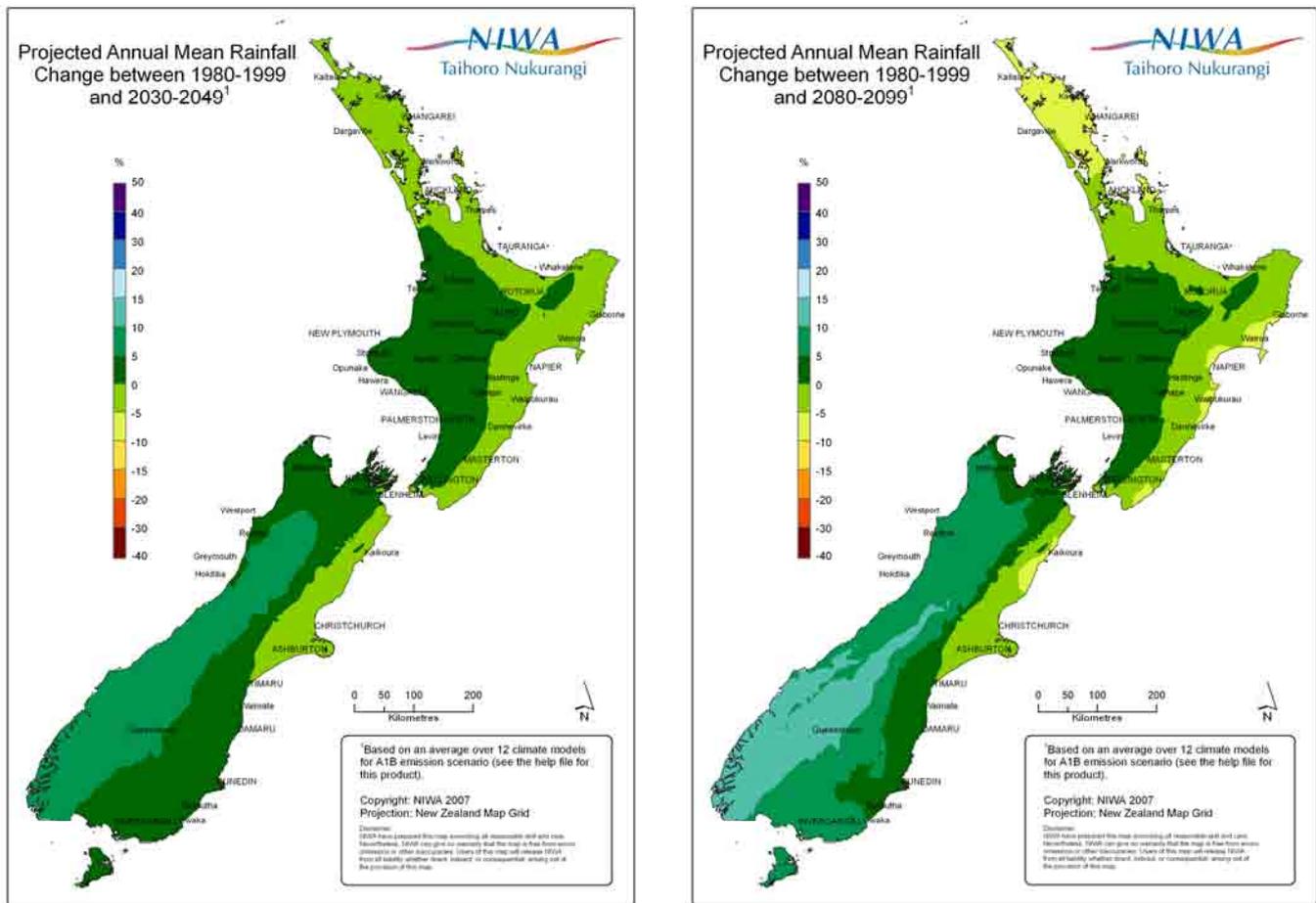


Figure 1.10. New Zealand projected changes in annual rainfall (in %) to 2040 and 2090, relative to 1990, based on a 12-model average of projections and for the A1B emission scenario.

1.4.2.1 Changes in extremes

Some of the most significant environmental, economic and social effects of climate change might be caused by changes in climate extremes (for example floods, droughts, frosts, strong winds, tropical cyclones and storm surges), rather than just changes in mean climate conditions. Unfortunately, estimates of likely changes in regional climate extremes are generally even less certain than projections of likely changes in mean conditions. We have summarised current understanding below: The reader is referred to the Local Government Guidance Manual (Ministry for the Environment, 2008) for more details.

1.4.2.1.1 Daily temperature extremes, frosts

A large decrease in the number of days with screen frosts (temperatures below 0°C at 1.3 m above the ground) is projected for the lower North Island and South Island as the 21st century progresses (Fig. 1.11). Note that because the far north of New Zealand already receives few frosts, the frost frequency there cannot decrease substantially, and also note that this figure is based on a study which analysed output from a single Global Climate Model (GCM).

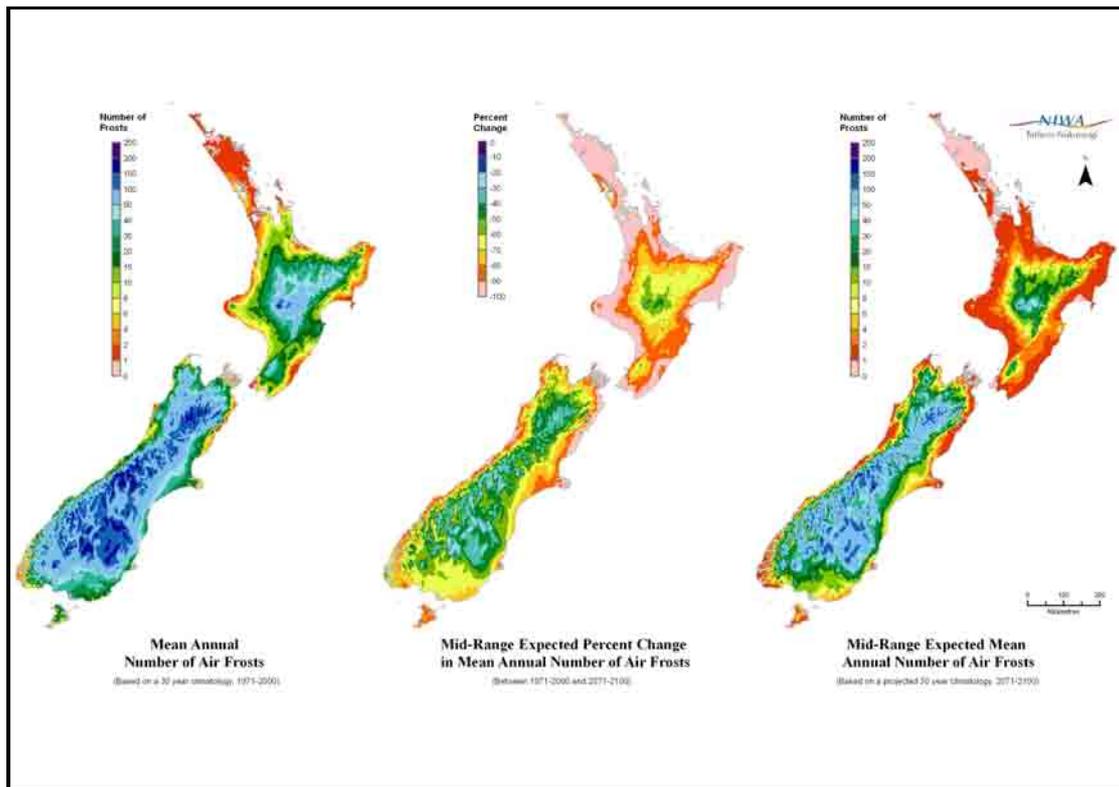


Figure 1.11. New Zealand projected changes in number of screen frosts to 2071–2100, relative to 1971–2000, based on a mid-range climate change projection from the CSIRO-9 GCM.

A change to the mean temperature is also expected to increase the number of ‘hot days’. In the example shown here in Figure 1.12, a ‘hot day’ is defined as a day which exceeded 25°C. The figure shows that for a 2°C increase in the mean temperature (which is consistent with the 12-model average temperature change projection for New Zealand to 2090; see Fig. 1.9) the number of hot days is projected to increase by as much as 50 days with the largest increases occurring in the Bay of Plenty, Waikato, Auckland and Northland regions.

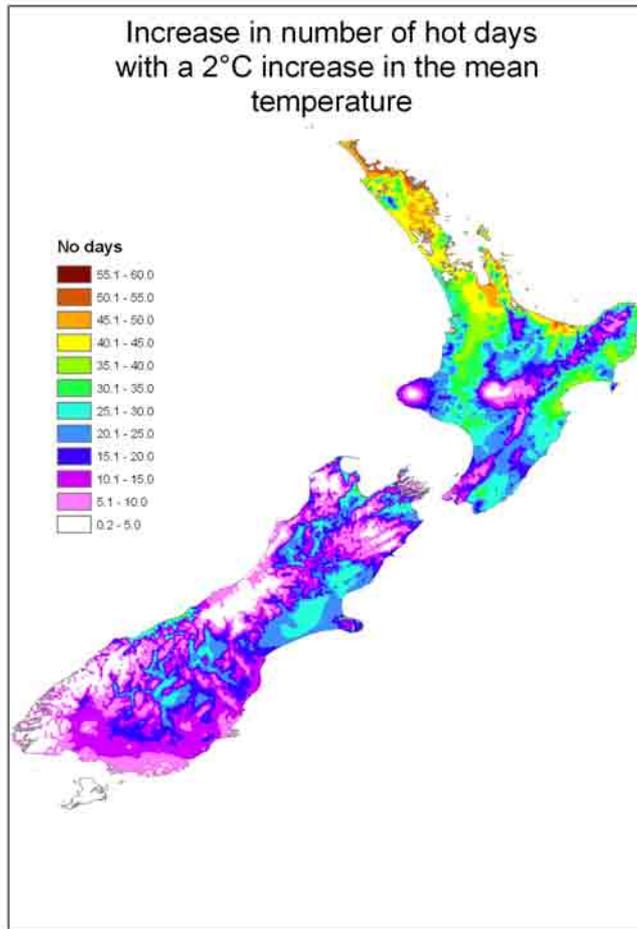


Figure 1.12. Increase in the number of days with a maximum air temperature greater than 25°C (referred to here as ‘hot days’) associated with a 2°C warming in the mean temperature. Such an increase in the mean temperature is consistent with the 12-model average temperature change projection for New Zealand to 2090 (see Figure 1.9).

1.4.2.1.2 Heavy rainfall

A warmer atmosphere can hold more moisture (about 8% more for every 1°C increase in temperature), so the potential for heavier rainfall certainly exists. Figure 1.13 shows the impact of this increased moisture holding capacity on the 24-hour 100-year average recurrence interval (return period) map shown for the current climate as Figure 1.4, given a 2°C warming (consistent with the 12-model average temperature change projection for New Zealand to 2090; see Fig. 1.9). The Figure shows that for broad areas of New Zealand the 24-hour rainfall total expected once in 100 years will increase by 20–30 mm, and that for mountainous locations the total may increase by more than 100 mm.

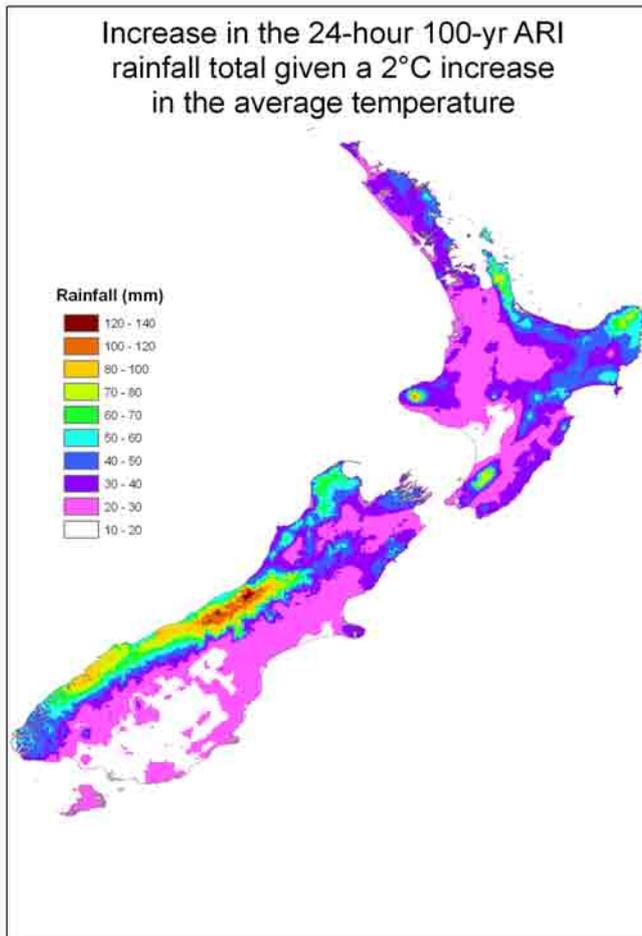


Figure 1.13. Increase in the 24-hour 100-year ARI (Average Recurrence Interval) rainfall total associated with a 2°C warming in the mean temperature. Such an increase in the mean temperature is consistent with the 12-model average temperature change projection for New Zealand to 2090 (see Figure 1.9).

The IPCC has declared that more intense rainfall events are "very likely over many areas". Various modelling studies suggest that heavy rainfall events will occur more frequently in New Zealand over the coming century, but the likely size of this change is uncertain. In broad terms, what is an extreme rainfall in the current climate might occur about twice as often by the end of the 21st century under a mid-range climate change scenario. For high temperature change scenarios, an extreme rainfall in the current climate might occur 3 to 4 times as often by the end of the 21st century. In the Local Government Guidance Manual, NIWA has provided some further guidance on extreme rainfall scenarios which can be used in preliminary "what-if" analyses of, for example, possible implications for storm water drainage systems.

1.4.2.1.3 Drought

NIWA has undertaken some specific research on how the frequency of drought might change over the coming century. A resulting report for the Ministry for the Environment, and the Ministry of Agriculture and Forestry is available on the web (Mullan *et al.*, 2005). The study developed drought risk projections for a range of climate change scenarios based on the IPCC third assessment report, corresponding to approximately the middle 75% of the IPCC global temperature change projection range. Under both the "low-medium" and the "medium-high" scenarios (which

bracketed this 75% range), the drought risk was projected to increase in frequency during the coming century for all areas that are currently drought prone.

Under the "low-medium" scenario, by the 2080s severe droughts (defined for the study as the current one-in-twenty year drought) are projected to occur at least twice as often as currently in the following areas: inland and northern parts of Otago; eastern parts of Canterbury and Marlborough; parts of Hawkes Bay; parts of the Bay of Plenty; and parts of Northland. Under the "medium-high" scenario severe droughts are projected to occur more than four times as often by the 2080s in the following regions: eastern parts of North Otago, Canterbury and Marlborough; much of the Wairarapa, Bay of Plenty and Coromandel; most of Gisborne; much of Northland.

1.4.2.1.4 Snowfall and snowline

It is physically plausible that snow cover will decrease and snowlines rise as the climate warms. However, there are confounding issues. Warmer air holds more moisture, and during winter this moisture could be precipitated as snow at high altitudes. Thus, warming does not rule out increased winter snowfall, although the duration of seasonal snow cover could be shortened.

1.4.2.1.5 Strong winds

Global climate models suggest that for mid-range temperature change projections the mean westerly wind component across New Zealand will increase by approximately 10% of its current value in the next 50 years (Mullan *et al.*, 2001a). The implications of this for strong winds are uncertain but preliminary results from a single run of NIWA's regional climate model suggest only small changes (~2% increase) to extreme winds may occur on average for New Zealand, with slightly higher changes (~10% increase) possible in some locations (e.g. on the Canterbury Plains in the lee of the Southern Alps).

1.4.2.1.6 Ex-Tropical cyclones and mid-latitude storms

This is also an area of considerable uncertainty. Current knowledge suggests the most likely future outcomes over the coming century for New Zealand are that ex-tropical cyclones might be slightly less likely to reach New Zealand, but if they do their impact might be greater. The intensity or frequency of mid-latitude storms might also increase somewhat in our region. However, our level of confidence in these projections is low.

1.4.3 Summary of projected climate changes for New Zealand

Much of the climate change scenario material described here was summarised in a table for the Local Government Guidance Note prepared by NIWA (Ministry for the Environment, 2008), reproduced here as Table 1.2. All estimates in Table 1.2 represent the best current scientific estimate of the direction and magnitude of change. The degree of confidence placed by NIWA scientists on the projections is indicated by the number of stars in brackets:

**** = Very confident, at least 9 out of 10 chance of being correct. Very confident means that it is considered very unlikely that these estimates will be substantially revised as scientific knowledge progresses.

*** = Confident

** = Moderate confidence, which means it is more likely than not to be correct in terms of indicated direction and approximate magnitude of the change.

* = Low confidence, but the best estimate possible at present from the most recent information. Such estimates could be revised considerably in the future.

Table 1.2. Main features of New Zealand climate change projections for 2040 and 2090. (Ministry for the Environment, 2008).

Climate variable	Direction of change	Magnitude of change	Spatial and seasonal variation
Mean temperature	Increase (****)	All-scenario average 0.9°C by 2040, 2.1°C by 2090 (**)	Least warming in spring season (*)
Daily temperature extremes (frosts, hot days)	Fewer cold temperatures and frosts (****), more high temperature episodes (****)	Whole frequency distribution moves right	
Mean rainfall	Varies around country, and with season. Increases in annual mean expected for Tasman, West Coast, Otago, Southland and Chathams; decreases in annual mean in Northland, Auckland, Gisborne and Hawke's Bay (**)	Substantial variation around the country and with season	Tendency to increase in south and west in the winter and spring (**). Tendency to decrease in the western North Island, and increase in Gisborne and Hawke's Bay, in summer and autumn (*)
Extreme rainfall	Heavier and/or more frequent extreme rainfalls (**), especially where mean rainfall increase predicted (***)	No change through to halving of heavy rainfall return period by 2040; no change through to fourfold reduction in return period by 2090 (**)	Increases in heavy rainfall most likely in areas where mean rainfall is projected to increase (***)
Snow	Shortened duration of snow season (***), Rise in snowline (**), Decrease in snowfall events (*)		
Glaciers	Continuing long-term reduction in ice volume and glacier length (***)		Reductions delayed for glaciers exposed to increasing westerlies

Climate variable	Direction of change	Magnitude of change	Spatial and seasonal variation
Wind (average)	Increase in the annual mean westerly component of windflow across New Zealand (**)	About a 10% increase in annual mean westerly component of flow by 2040 and beyond (*)	By 2090, increased mean westerly in winter (>50%) and spring (20%), and decreased westerly in summer and autumn (20%) (*)
Strong winds	Increase in severe wind risk possible (**)	Up to a 10% increase in the strong winds (>10m/s, top 1 percentile) by 2090 (*)	
Storms	More storminess possible, but little information available for New Zealand (*)		
Sea level	Increase (****)	At least 18-59 cm rise (New Zealand average) between 1990 and 2100 (****)	
Waves	Increased frequency of heavy swells in regions exposed to prevailing westerlies (**)		
Storm surge	Assume storm tide elevation will rise at the same rate as <u>mean</u> sea-level rise (**)		
Ocean currents	Various changes plausible, but little research or modelling yet done		
Ocean temperature	Increase (****)	Similar to increases in mean air temperature	Patterns close to the coast will be affected by winds and upwelling and ocean current changes (**)

In considering future climate changes and their impacts it is important to remember that we will continue to experience natural year-to-year variations in climate and in the frequency of extremes. Over the coming century New Zealand will have to deal with the combination of underlying mean climate, changes due to global warming, and natural variations.

1.4.4 Summary pattern of climate change for New Zealand

The broad pattern of change expected involves:

- Increased temperatures, with a greater increase in the winter season and in the north of New Zealand.
- Decreased frost risk but increased risk of very high temperatures.

- Stronger west-east rainfall gradient (wetter in the west and drier in the east).
- Increased frequency of extreme daily rainfalls.
- Increased sea level.
- Increased westerly winds.

and a number of other changes with a lower degree of confidence.

Mid range projections in annual average temperature and precipitation are:

- Temperature increase of 0.75 to 1.0 °C from 1990 to 2040 (50-year change), and 1.75 to 2.25 °C from 1990 to 2090 (100-year change).
- Precipitation change between about -5 to +10% from 1990 to 2040, and about -10 to +15% from 1990 to 2090.

These changes have to be carefully interpreted. The changes will actually cover a wide range of values based on the diverse emission scenarios that could be chosen and also because of climate model uncertainties. The above values of change represent mid range projections.

1.5 Carbon dioxide and changes of other atmospheric gases

Emissions of carbon dioxide (CO₂) due to fossil fuel burning are virtually certain to be the dominant influence on the trends in atmospheric CO₂ during the 21st century, according to the IPCC (2001a).

The atmospheric CO₂ concentration has increased from its pre-industrial concentration of about 280 μmol mol⁻¹ to over 380 μmol mol⁻¹ at present, and is increasing further by about 1.5 μmol mol⁻¹ each year (Keeling, 1993). Future increases of CO₂ concentration will depend on the reduction or non-reduction of CO₂ emitting practices (see top panel in Fig. 1.8).

By 2100, carbon cycle models project atmospheric CO₂ concentrations of 540 to 970 parts per million (ppm) across the SRES scenarios. The IPCC also notes that uncertainties in feedbacks from the terrestrial biosphere could broaden the CO₂ range even further above the values used in the SRES scenarios. The projected concentrations of other non-CO₂ greenhouse gases vary considerably across the SRES scenarios and are summarised in Table 1.3.

Table 1.3. Changes in greenhouse gases by 2100, under the IPCC SRES scenarios.

Greenhouse gas	Current concentration	Change by 2100
Carbon dioxide (CO ₂)	384 ppm	+170 to +600 ppm
Methane (CH ₄)	1745 ppb	-190 to + 1970 ppb
Nitrous oxide (N ₂ O)	314 ppb	-38 to +144 ppb
Tropospheric Ozone (O ₃)	Variable	-12 to +62%
Other (HFCs, SF ₆)	Depends	Wide range

1.6 Climate change modelling – Knowledge gaps and uncertainty

When considering New Zealand's future climate, there are a number of uncertainties that must be taken into account. A major problem is the inability to predict future greenhouse gas emissions, because these depend so strongly on social, economic and political decisions yet to be made. Moreover, even if future emissions were known exactly, different global climate models will vary in the way they parameterise small-scale features of the climate (e.g. clouds) and in the way they simulate climate feedbacks which leads to different globally-averaged temperature changes. Such model differences are amplified at small regional scales, such as over the New Zealand land mass. Figures 1.9 and 1.10 show the projected range of temperature and precipitation changes by 2040 and 2090, relative to the IPCC baseline year of 1990. The range takes account of the difference between climate models, and also the wide variation in future emissions based on what are called the IPCC SRES scenarios (Nakicenovic and Swart, 2000).

Professionals who want to apply adaptation measures often lack sufficiently detailed data on which to modify design or operational plans to account for climate change. This is because professionals require data at the local and regional level, whereas climate change projections are undertaken at the global level. This type of information gap is commonly observed in climate change risk analysis in New Zealand and around the world. While scaling methods (including temporal scaling; i.e. from monthly mean changes to time series of daily values) may suffice (indeed, they may be the only option in some cases), they introduce additional uncertainty. Closing this information gap is an important goal of all climate change research and a key area for innovation.

NIWA's current process of downscaling the global projections to the New Zealand scale has statistical error which may limit its use in studies of extreme events. Typical values of the explained variance of seasonal mean temperature and precipitation at the climate station locations used are around 50 – 70% (temperature) and 20 – 50% (precipitation) (Mullan *et al.*, 2001b). Thus, there is a significant proportion of the variance unexplained by the regression equations used.

The levels of confidence shown in Table 1.2 attempt to sum up all the potential sources of model uncertainties. Typically, the "direction of change" (i.e. increase, stay the same, or decrease) is described with higher confidence than the "magnitude of change". For example, there is very high confidence that the New Zealand mean temperature will increase, but medium confidence that the change for a mid-range emissions scenario will be between 0.75 and 1.0°C by 2040 and between 1.75 and 2.25°C by 2090. Projected changes to extreme events such as the frequency and/or intensity of heavy rainfall events, high winds, storms have the least confidence.

The current global projections describe climate change as a relatively gradual process over the next 50 to 100 years. However it is unlikely that the human component of climate change, which is the driver of the projections, will be experienced as a smooth transition. Natural background variability will amplify change in some years, reflecting in potentially sudden and dramatic changes of the local and regional climate and weather. There is also the prospect of rapid shifts in broad scale systems, as the global system reaches critical thresholds. Understanding of such processes is not well developed, and they are not included in the current suite of AOGCMs (Atmosphere-Ocean Global Climate Models). One of the major limitations in climate change science is that it is not possible to forecast the nature of

the coming decade, yet this information may be of most value to those making critical planning and investment decisions now.

Current research around the globe and in New Zealand focuses on enhancing the degree of confidence with which scenarios can be applied. At NIWA, scenarios of changes in temperature and precipitation generated for the IPCC's 2007 Fourth Assessment Report (AR4) from 12 global climate models have been downloaded and downscaled for New Zealand. Concurrently, NIWA continues to develop the only regional climate model for New Zealand (this is based on and nested within the UK Hadley Centre global climate model, but is run at a higher spatial resolution for the New Zealand area). Analyses of the regional climate model will enable significant improvements to be made in the understanding of climate change on New Zealand climate, and particularly on extreme events.

While there still are, and will continue to be, inter-model differences in climate change projections for New Zealand (and indeed for anywhere in the world), the improvement and increase in the number of models means that the data can now be analysed probabilistically rather than as a low, middle and high projection range. Probabilities of exceeding certain critical thresholds, for example, can be directly incorporated into a climate change risk assessment framework. Concepts from risk management are increasingly being used in climate impact assessment, as they provide an approach that is able to establish appropriate responses that treat risk even when there is considerable uncertainty (IPCC, 2007b).

2 Physiological responses of forests (trees) to climate change

Summary

Objective

Summarise current knowledge of the likely direct climate-change impacts on plantation productivity. Identify modelling approaches which could potentially be used to predict plantation productivity under climate change and indicate any potential limitations they may have. Identify any uncertainties or knowledge gaps in these areas, and identify any issues which currently limit the certainty of projections.

Key results

Climate change is likely to have a significant impact on the future growth of trees in planted forests because tree growth responds directly to changes in CO₂, temperature, and availability of water and nutrients. There are direct responses to these drivers and indirect interactions and feedback processes.

Increasing CO₂ concentration is generally beneficial for the growth of plants, but the magnitude of the response varies with species and growth stage and is affected by interactions with other environmental factors that can limit (e.g. lack of essential nutrients) or enhance (e.g. warmer and drier conditions) the 'CO₂ fertilisation effect'. CO₂ fertilisation is likely to be most beneficial in the drier parts of the country, such as Otago, Canterbury, Hawkes Bay and East Cape. Nutrient limitations (especially nitrogen) are unlikely to limit the CO₂ response provided that the nutrient levels, which are currently sufficient in most of New Zealand's plantations, can be maintained into the future. Increasing temperatures can also stimulate decomposition of soil organic matter and mineralise more nitrogen to further boost the nutritional status of trees.

There are, however, a few key uncertainties at present. It is not certain to what extent the CO₂ response of trees is maintained as trees mature, or whether on-going stimulation of carbon fixation is simply channelled towards root growth and turn-over with little enhancement of above-ground productivity. The possible importance of the phenomenon of photosynthetic down-regulation in a high CO₂ environment has also not yet been resolved.

Increases in air temperature expected with climate change are likely to have a predominantly positive effect on plantation growth. Increasing temperature is likely to lengthen the growing season (and allow some expansion of the plantation estate to higher elevations). Radiata pine has no distinct dormant period and can grow throughout the year, thereby taking advantage of the warmer winter conditions. Based on current correlations with climatic conditions, it is expected that growth of radiata pine is likely to increase in most regions with climate change. Gains in productivity are likely to be most marked in the South Island and high altitude regions within the North Island, with little change due to increased temperatures likely on the east coast of the North Island or in regions north of Rotorua. However, as this response is based on empirical relationships, whereby air temperature may be subject to collinearity with other variables such as solar radiation and site fertility,

further manipulative and process-based research is required to more accurately define the magnitude of this response.

Increasing temperature can potentially have important effects through changing the water balance and water availability for trees. Evapotranspiration rates can be increased through increasing temperature, but that increase may not eventuate if the air becomes more humid due to a reduction in the difference between minimum and maximum temperatures. Evapotranspiration from tree canopies is also likely to change more than that from grass swards. The physiological status of trees is also important, with evapotranspiration of tree stands under stress being more responsive to changes in temperature than that of unstressed stands. Stomatal closure in response to increasing CO₂ concentration may further affect evapotranspiration. These multiple interacting factors can only be assessed through comprehensive modelling approaches that include the relevant physical and plant-physiological processes under climate change.

While radiata pine is New Zealand's most important plantation species, there has not yet been a comprehensive modelling study to investigate the likely growth response of radiata pine to the combination of likely climatic changes over the short or longer term. The understanding of forest growth under a changing climate will require an integrative modelling approach that can account for the various interactions and feedbacks that will occur with changes in temperature, CO₂ and water and nutrient availability. Comprehensive models that include these processes exist and are available, but they need to be tested in New Zealand before being used to quantify the forest growth response across the wide edaphic and climatic range over which plantations are grown in New Zealand.

2.1 Responses of photosynthesis to CO₂ concentration

Of all the aspects of atmospheric and climate change, the increase in atmospheric CO₂ concentration is the most certain, and increasing CO₂ is generally considered to be beneficial for plant growth (the 'CO₂ fertilisation' effect). The effects of elevated CO₂ on plant growth and yield will depend on the photosynthetic pathway, species, growth stage and site condition (e.g. water and nutrient regime, see below).

It has been shown in many experimental studies that C₃ photosynthesis responds strongly to CO₂ concentration, with photosynthesis typically increasing by 25-75% for doubling atmospheric CO₂ concentration (e.g. Kimball, 1983; Cure and Acock, 1986; Drake, 1992; Luxmoore *et al.*, 1993; Kirschbaum, 2000, 2004; Urban, 2003). This generalisation includes observations on *P. radiata* (e.g. Tissue *et al.*, 2001; Turnbull *et al.*, 2004) and persists even after inclusion of the effects of the often-observed photosynthetic down regulation (Gunderson and Wullschleger, 1994).

Recent work has also shown sustained growth increases for plants fumigated in 'free air CO₂ enrichment' (FACE) experiments. This has been observed in wheat fields (Garcia *et al.*, 1998) and in largely undisturbed forests (Herrick and Thomas, 2001; Gunderson *et al.*, 2002). These responses are consistent with theoretical understanding of the effect of CO₂ concentration on photosynthesis at the leaf and stand level (McMurtrie *et al.*, 1992; Long *et al.*, 1996).

A key finding of much past research is that there is not one unique CO₂ response factor, but that the likely response interacts with other limiting factors in the environment (Kirschbaum, 1999a; Körner *et al.*, 2007). In particular, growth responses are likely to be greatest under warm and water-limited conditions and least under cool and nutrient-limited conditions as simulations of NPP for sites with very different climate have shown (Fig. 2.1; Kirschbaum 1999b, 2004).

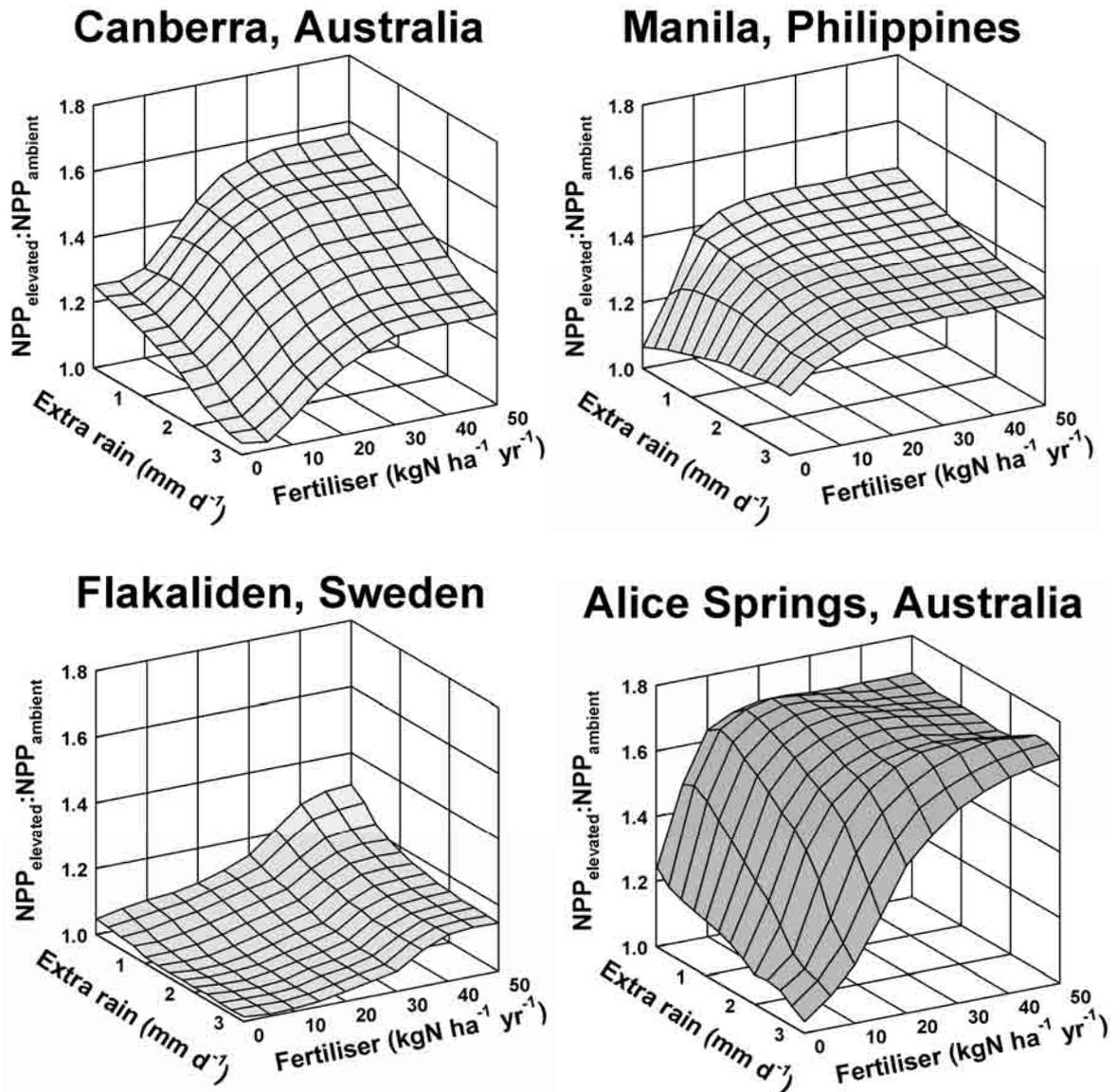


Figure 2.1. Ratio of net primary production (NPP) under doubled and ambient CO₂ concentration under different base conditions. Four climatically very different sites are shown to provide the base climatic conditions, and at each site, simulations were run over a range of water and fertiliser additions. Redrawn from Kirschbaum, (2004).

The reason for the interaction with temperature is that the CO₂ stimulation of photosynthesis increases with increasing temperature (Long, 1991; Kirschbaum, 2004). The reason for the interaction with nutrients is that plant growth requires not only carbon but also nutrients. Any enhancement of carbon supply through higher atmospheric CO₂ simply intensifies the nutrient limitation, with little overall response.

For most of New Zealand's current plantation forests, the CO₂ responsiveness should be fairly unaffected by nutrient limitations. This is because most current plantations have a medium to high foliage nutrient level (especially nitrogen) as a result of active management against nutrient deficiency (Fig. 2.2).

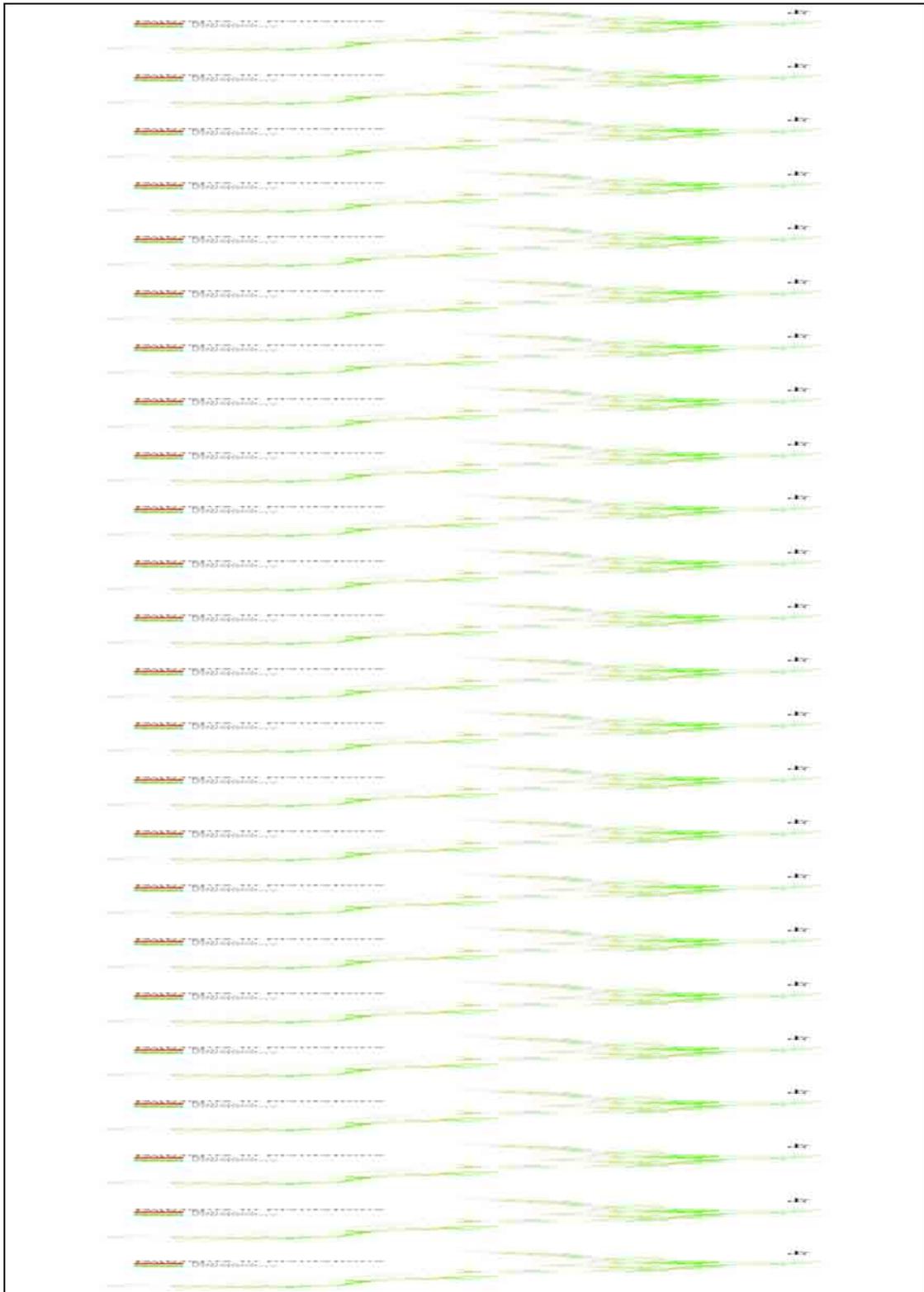


Figure 2.2. Foliar nitrogen levels (satisfactory, marginal, deficient) found in plantations in New Zealand based on foliar nutrient survey (Paul, T. unpublished data).

The reason for the interaction with water availability in Figure 2.1 is more complex. In elevated CO_2 , plants close their stomata, reducing water loss and thus conserving soil moisture. This response is of little consequence in regions with ample water supply, but in regions with limited water supply the conserved moisture can allow on-

going growth over transient dry periods. The strength of that interaction depends on the extent of water limitations, the capacity of the soil to store moisture and other factors. As a result, greater CO₂ responsiveness under drier conditions can break down under specific circumstances.

Applying these considerations to the sample simulations shown in Figure 2.1, indicates that the relative CO₂ responsiveness generally increases with increasing nutrient supply and decreases with the availability of extra water. Responses are low in Sweden because the system tends to be cool and nutrient-limited while water supply is generally adequate. At the other extreme is Alice Springs in Australia, where temperatures are generally high and water availability is generally the limiting resource.

The beneficial effect of CO₂ fertilisation on tree growth is therefore likely to be greater on dryland sites in New Zealand. These regions include eastern Otago, Canterbury, Hawke's Bay, the southern North Island and the East Coast (see Fig. 2.7).

Experimental work has shown that tree growth (Atwell *et al.*, 2003), and especially root growth (Thomas *et al.*, 1999, 2000), and photosynthesis (Tissue *et al.*, 2001) of *P. radiata* grown in New Zealand increases in response to increasing CO₂ in a similar manner to that of other species studied overseas. However, the growth response to increased CO₂ for *P. radiata*, grown across an environmental gradient in New Zealand has not yet been tested or modelled. Such modelling would help to quantify the potential productivity response under New Zealand conditions.

2.1.1 Mature plant responses

For logistic constraints, most research on CO₂ responses of plants has been carried out on young plants, or plants of smaller status. As a result there is still uncertainty as to whether mature stands will respond to increasing CO₂ concentration in the same way as young stands. The answer to that question is complicated by the fact that stand growth normally slows as stands mature but the reasons for that slowing are not well understood (Ryan *et al.*, 1997). It has been shown in a theoretical analysis that depending on the causes of age-related declines in productivity, and the age over which a response to CO₂ is assessed, a range of possible apparent response patterns are likely (Kirschbaum, 2005).

For instance, if maturity-related decreases in productivity are expressed as a function of stand size (rather than stand age), then a stand grown under elevated CO₂ might be affected by these size-related growth reductions at an earlier age. A CO₂ response might therefore be present if growth is expressed as a function of stand size, but not if growth is expressed as a function of stand age.

Experimental evidence on this important question is very limited owing to the logistic constraints of modifying the whole environment of a large tree canopy. Of the small number of observations, some indicate on-going growth enhancements (e.g. Delucia *et al.*, 2005). Some indicate on-going stimulation of carbon gain, but with most of the extra carbon used for greater root growth and below-ground allocation (Norby *et al.*, 2004). Others indicate only a brief growth stimulation that is not sustained over subsequent years (e.g. Asshoff *et al.*, 2006), or a growth stimulation that diminishes gradually over a period of 30 years (Hättenschwiler *et al.*, 1997). Another similar study (Tognetti *et al.*, 2000) found no significant increase in radial stem growth due to CO₂ enrichment over a long period of growth. Interestingly, in the same experiment where Asshoff *et al.* (2006) did not find significant growth differences based on tree

ring and basal area growth, Zotz *et al.* (2005) found CO₂ enhancement of photosynthesis of around 40% in the third year. It is not uncommon for photosynthetic stimulation by CO₂ to be on-going, with the extra carbon allocated below-ground, with little on-going above-ground growth. In cases where growth enhancements are not sustained, it is not clear whether that is due to nutrient feed-backs, size-related feed-backs or due to other factors like allocation shifts that might make mature stands less responsive to increasing CO₂.

The CO₂ response of mature trees is obviously important because the plantation estate consists of stands of different ages. Sustained growth over a long period of time means that trees reach harvestable size sooner than if only early growth is stimulated. This can have a significant effect on the economics of production forestry (shorter rotations). It does not necessarily enhance carbon stocks, however, and might simply increase the rate of turnover. A beneficial effect on global carbon balances might result if more, and longer-lasting, forest products could be produced, or if wood could be used as bio-fuel to replace fossil fuel.

There is no New Zealand-specific study on mature-tree CO₂ responses of plantation tree species, so our current knowledge is based on overseas studies. If only young trees respond to elevated CO₂ it will still give young trees a better start. Nursery stock for planting may improve through such an early response, and such better earlier growth will be beneficial throughout the stand's life. If trees continue to respond to CO₂ throughout their life, the beneficial effect will continue to accumulate. For many plantations, the initial response might be the most important because for rotation lengths of less than 30 years, good early growth is essential.

2.1.2 Photosynthetic down-regulation for plants in a high CO₂ environment

Many workers who have exposed plants to elevated CO₂ concentration have observed a degree of photosynthetic down-regulation over exposure times of weeks or longer (e.g. Gunderson and Wullschleger, 1994; Long *et al.*, 1996; Wolfe *et al.*, 1998). Part of the phenomenon of down-regulations has been explained as an experimental artefact in that downward acclimation tends to be more pronounced if plants are grown in smaller pots (Arp, 1991; Thomas and Strain, 1991).

More generally, plants regulate their various processes for an overall balance. If one of their required constituents, namely carbon, becomes more readily available as a result of increased CO₂, then the question arises whether the plant can utilise that extra resource. If plants are grown with restricted root volumes then carbohydrates, even under normal CO₂, may be in adequate supply. This means that excess carbohydrate cannot be utilised and plants down-regulate their photosynthetic capacity to match supply with a limited demand (Erice *et al.*, 2006). So, it is consistent to find the degree of down-regulation increasing with diminishing root-zone volumes. Little down-regulation is observed when the source-sink balance is manipulated in favour of creating more sinks (Erice *et al.*, 2006), or for plants that are grown in larger pots or in the field (Arp, 1991; Pendall *et al.*, 2004). In some studies on pines, down-regulation was observed not to occur in current-year needles but only in newly grown needles that developed under changed atmospheric conditions (Turnbull *et al.*, 1998). In other studies, down-regulation was observed in neither older nor newly grown needles (Tissue *et al.*, 2001). Tissue *et al.* (2001) attributed the lack of down-regulation in their study to the continuance of an ongoing growth potential that presented sinks for carbohydrates.

Even without limitation on growth potential, a degree of 'downward acclimation' has to be expected if increased photosynthetic carbon gain cannot be matched by similarly increased nutrient supply (in this case through reduced internal nutrient status). This would result in lower foliar nutrient concentrations and consequently reduced inherent photosynthetic rates (e.g. Rastetter *et al.*, 1992, 1997; Kirschbaum *et al.*, 1994, 1998; Wolfe *et al.*, 1998). It is unclear whether there is additional downward acclimation in addition to these readily understood processes. This constitutes an important on-going uncertainty in our understanding of overall plant responses to changing climatic conditions.

The uncertainty about possible down regulation, or the extent of it, raises a question about how the effect of CO₂ fertilisation should be used to model tree growth under climate change. It is desirable to explicitly model nutrient dynamics so that any reduction in photosynthetic capacity through changing nutrient availability can be explicitly accounted for. Beyond that it might be feasible to adopt a compromise solution where model runs could use maximum and minimum realistic growth enhancements by elevated CO₂. It may also be desirable to further analyse the findings of different research studies to resolve the still apparent contradictions between different research findings, and to derive an agreed common understanding of the key processes that operate in natural systems.

2.2 Growth response to temperature

Photosynthesis can be strongly affected by temperature, but many plants also exhibit remarkable tolerance up to very high temperatures. Plants also acclimate strongly to their actual temperature environment. So, plants in general are not expected to experience significant problems from a temperature increase *per se*. Water interactions under warmer conditions constitute a different problem and are dealt with in a subsequent section.

There are also important interactions between the photosynthetic temperature response and other environmental factors such as CO₂ concentration (Rawson, 1992; Morison and Lawlor, 1999). The temperature optimum of photosynthesis shifts to a higher temperature in elevated CO₂, and the response of photosynthesis to increasing CO₂ increases with increasing temperature (Long, 1991; Kirschbaum, 1994). This photosynthetic response drives differential growth responses, with plants grown in higher temperatures usually responding more strongly to increasing CO₂ concentration than plants grown in lower temperature conditions (Rawson, 1992; Morison and Lawlor, 1999). Because of the multitude of interacting factors, these relationships are generally not very tight (Morison and Lawlor, 1999).

Total biological productivity of systems that currently experience low temperatures responds strongly to any increases in temperature (Lieth relationship; Fig. 2.3). For systems at an annual mean temperature of about 0°C, one can expect productivity increases of almost 10% per degree of warming. That stimulation diminishes greatly for systems already experiencing warmer temperatures. The predicted growth response is reduced to only about 3% per degree warming for systems currently experiencing a mean annual temperature of 20°C.

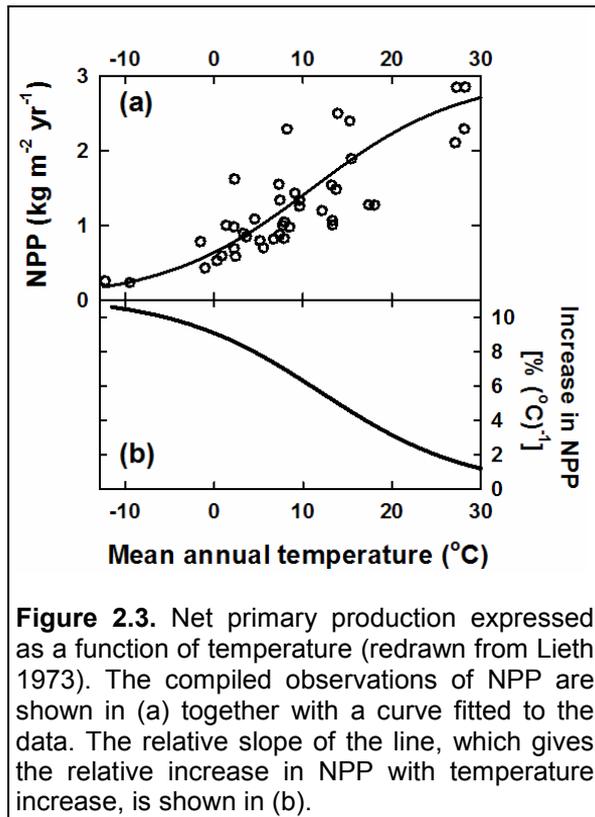


Figure 2.3. Net primary production expressed as a function of temperature (redrawn from Lieth 1973). The compiled observations of NPP are shown in (a) together with a curve fitted to the data. The relative slope of the line, which gives the relative increase in NPP with temperature increase, is shown in (b).

This responsiveness to increasing temperature is principally driven by the lengthening of the growing season (Lieth, 1973). This has been demonstrated recently by Kerkhoff *et al.* (2005), who showed that net primary production (NPP) expressed as NPP per month of growing season (defined as a mean monthly temperature above 0°C) was virtually invariant with temperature (Fig. 2.4). In other words, provided mean monthly temperature is above 0°C, it matters very little what temperatures plants experience.

Under New Zealand conditions, one would therefore expect little generic productivity response to increasing temperature as few sites experience winter temperatures with mean monthly temperatures below 0°C.

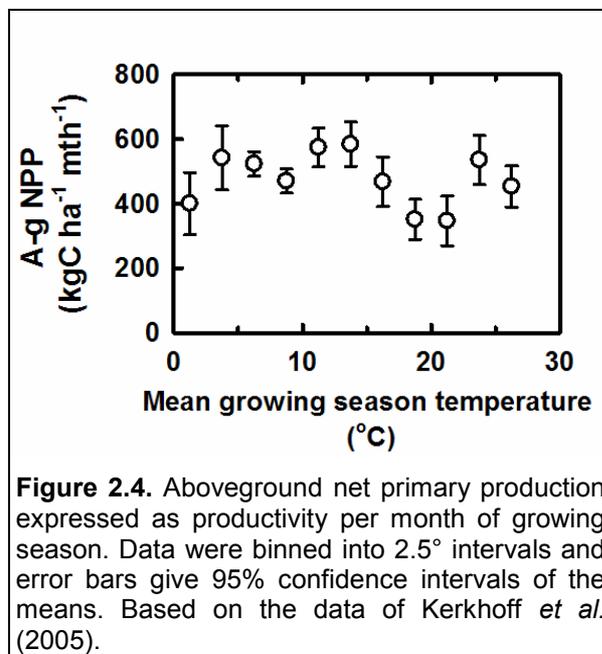


Figure 2.4. Aboveground net primary production expressed as productivity per month of growing season. Data were binned into 2.5° intervals and error bars give 95% confidence intervals of the means. Based on the data of Kerkhoff *et al.* (2005).

Despite these generic observations, there is some evidence to indicate that growth of *P. radiata* will positively respond to increases in air temperature resulting from climate change. A number of studies have used empirical modelling techniques to determine the key climatic drivers of volume and height productivity in *P. radiata* across the national estate. Almost all of these studies conclude that temperature is a key determinant of productivity (Jackson and Gifford, 1974; Hunter and Gibson, 1984; Watt *et al.*, 2008). Although two of these studies indicate a temperature optimum for growth within New Zealand (Jackson and Gifford, 1974; Hunter and Gibson, 1984) the third (Watt *et al.*, 2008), which included fertilised plots, did not. Determining if

there is an optimum temperature for growth is difficult using empirical data from New Zealand as the high temperatures which occur in forests in the far north are also associated with nutrient- deficient soils. It is therefore difficult to ascertain whether reductions in growth in northern regions are attributable to poor soil fertility or high temperatures.

As *P. radiata* has no period of dormancy it is likely that increases in temperature will result in greater growth through extending the length of the growing season. Previous research (Whitehead *et al.*, 1994) has shown that needle elongation in *P. radiata* is related to thermal time above a base mean daily temperature of 6°C. Given the sensitivity of *P. radiata* growth to frosts, it is likely that this base temperature may at least partially reflect a mean daily temperature above which ground frosts do not regularly occur.

These observations are consistent with a recent study which presents the most comprehensive examination to date of how *P. radiata* responds to climate (Palmer *et al.*, in prep). In this study measurements of mean annual volume increment at age 30 (MAI300) from 1 764 permanent sample plots, covering the complete environmental range within New Zealand were regressed against a range of explanatory variables. The two climatic variables of most importance were found to be mean annual number of ground frosts (*G*) during autumn and mean annual fractional available root-zone water storage (see below). The partial relationship between *G* and MAI300 shows relative insensitivity until four ground frosts /month, are reached after which there is a sharp decline in MAI300 (Fig. 2.5). The threshold for growth reductions of four ground frosts per month is roughly equivalent to a mean annual temperature of 12°C.

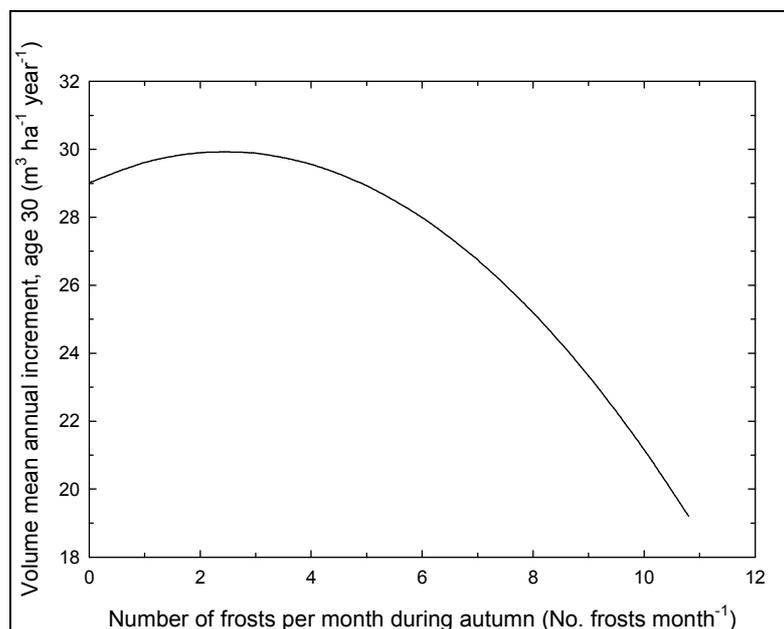


Fig. 2.5. Influence of number of frosts per month during autumn on volume mean annual increment at age 30. For this graph values of mean annual fractional available root-zone water storage were held at average values of 81% while the partial response surface was generated.

These results suggest that growth of *P. radiata* is likely to increase in most regions under climate change. Gains in productivity are likely to be most marked in the South Island and high altitude regions within the North Island, with little change due to increased temperatures likely on the east coast of the North Island or in regions north

of Rotorua, where mean annual temperatures exceed 12°C. However, as this response is based on empirical relationships, whereby the number of ground frosts is subject to collinearity with other variables such as solar radiation and site fertility, further manipulative and process-based research is required to determine if the thresholds for the growth response given here are correct.

2.3 Temperature and CO₂ effects on respiration rate

Plants gain carbon in photosynthesis. Part of that carbon gain is used for maintenance respiration and the remainder can be used for growth. It can be readily observed that respiration rates increase with short-term increases in temperature. It used to be thought that if these higher respiration rates were maintained for plants exposed to higher temperatures for longer periods. It would therefore follow that at higher temperatures a greater fraction of fixed carbon would be lost in respiration, with less available for growth, so growth rates would decrease with increasing temperature (Fitter and Hay, 1981; Woodwell, 1987; Melillo *et al.*, 1990).

Work over a number of years has shown that the control of respiration rate follows a more intricate pattern (e.g. Gifford, 2003), with acclimation to temperature after exposure to altered conditions for more than a few days. This was more fully investigated by Gifford (1995) in an experiment with *Triticum aestivum*. He expressed his observations as the ratio of respiration rate to photosynthesis measured over 24 hours (Fig. 2.6).

When Gifford (1995) grew plants at 15°C, he observed that about 36% of carbon gained in photosynthesis was lost in respiration. When he transferred plants to 25°C, there was an initial increase in the ratio of respiration rate to photosynthesis over the first day, but that ratio decreased over subsequent days. After six days, the ratio was essentially the same as that which had been observed at 15°C. When plants were transferred back to 15°C, there was an initial response to the new temperature. After another four days, however, the ratio was again similar to the ratio before plants had acclimated to the different temperatures (Fig. 2.6a).

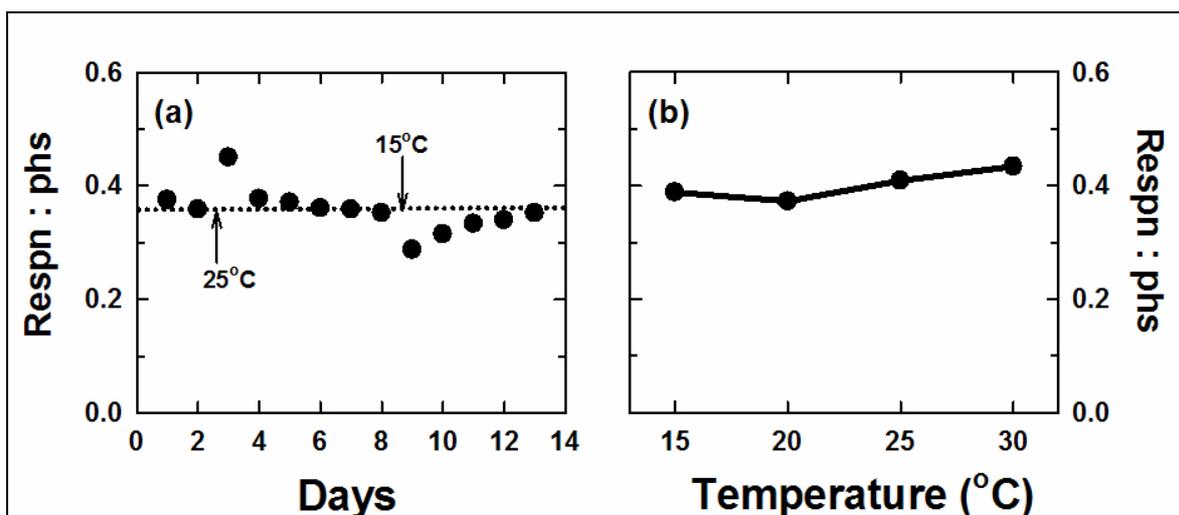


Figure 2.6. Changes in the respiration rate to photosynthesis ratio in response to changes in temperature over several days in (a) and expressed as a function of temperature in fully acclimated plants in (b). Data redrawn from Gifford (1995). Plants in (a) were initially held at 15°C. Arrows indicate the times when plants were transferred first to 25°C and then back to 15°C.

When the ratio of respiration rate to photosynthesis was expressed as a function of growth temperature in acclimated plants (Fig. 2.5b), he observed a slight increase in the ratio with temperature, but the increase was only slight (from 0.39 to 0.43 between 15 and 30°C). Similar results were obtained with other species, including *Pinus radiata* and *Eucalyptus camaldulensis* (Gifford, 1994; R.M. Gifford, pers. comm.) and cottonwood, *Populus deltoides* (Turnbull *et al.*, 2004). Hence, Gifford (1994; 1995) and others (e.g. Körner, 1996; Waring *et al.*, 1998) concluded that the ratio of respiration to photosynthesis does not deviate significantly from constancy over a range of temperatures, and that increases in global temperature are not likely to lead to significantly increased carbon losses in respiration.

Similarly, there is the question of a direct effect of CO₂ concentration on plant respiration rates. There have been a number of well-documented studies showing a short-term reduction in respiration rate in response to moderate increases in CO₂ concentration (see Gonzalez-Meler *et al.*, 1996). However, it is not clear whether these short-term responses lead to similar longer-term responses.

In longer-term experiments, respiration rates are mostly observed to be decreased, but such findings are not universal (Amthor, 1991; Gonzalez-Meler *et al.* 1996; Drake *et al.* 1999). Any analysis is complicated by the fact that respiration rate responds to various plant internal factors, and these are usually altered by growth in different CO₂ concentrations. For instance, plant tissue grown in higher CO₂ concentration almost invariably has a lower nitrogen concentration than tissue grown at lower CO₂. Since respiration rate usually increases with increases in tissue nitrogen concentration, it is not clear whether lowered respiration rate is a direct response to increased CO₂ concentration or an indirect response via lowered tissue nitrogen concentration.

Gifford (1995) conducted a detailed study of respiration responses to changes in CO₂ concentration. When he expressed his findings as the ratio of respiration rate to photosynthesis over a 24 hour period (similar to his findings for the response to temperature), he found no consistent effect of CO₂ concentration on the ratio of respiration rate to photosynthesis. He concluded that for assessing the impacts of changing CO₂ concentration and temperature, it would be best to assume that the ratio of respiration rate to photosynthesis does not change.

2.4 Water relations

Recent research has developed surfaces which describe water balance throughout New Zealand's plantation estate (Palmer *et al.* submitted). The study used spatially interpolated long term climate surfaces, derived by NIWA from historical climate records. The model SwatBal was then used to determine fractional available root-zone water storage (W_f) across New Zealand, assuming *P. radiata* covers the entire country (Figure 2.7). Fractional available root-zone water storage is a useful indicator of plant-available water as it provides an estimate of the average percentage of available water within the root-zone, over a specified time period.

Figure 2.7 shows that the driest parts of the country are located on the east coast of both Islands. In these regions W_f reaches annual averages as low as 20 to 30% in Otago. Large areas within Canterbury, Marlborough, Hawke's Bay, Manawatu and Wairarapa also experience a relatively low annual average W_f of 40-60%.

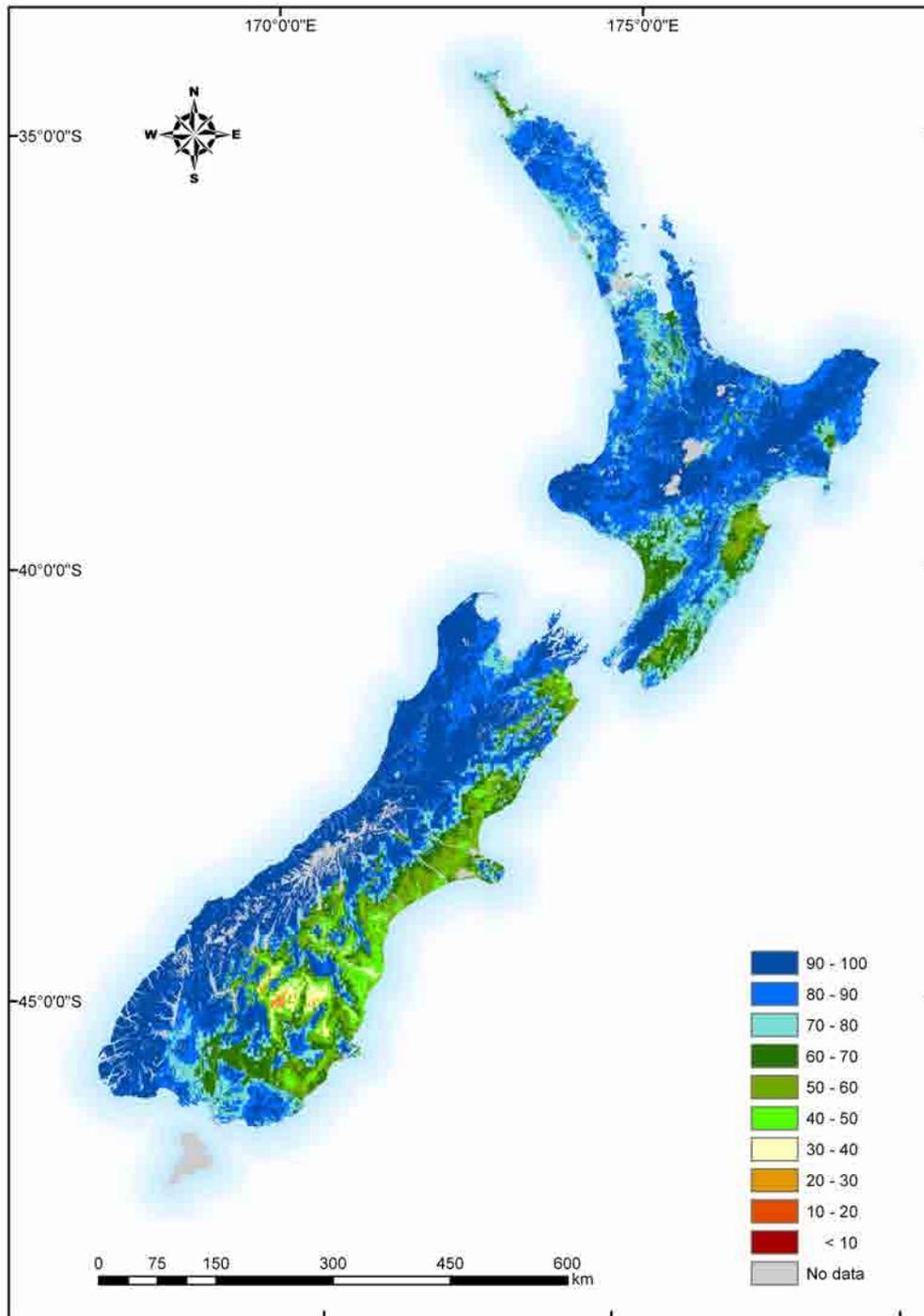


Figure 2.7. Modelled annual mean fractional available root-zone water storage (W_f) for New Zealand.

2.4.1 Quantification of water supply on productivity-current knowledge

Soil water balance has long been recognised as a major determinant of *P. radiata* growth at specific locations exhibiting seasonal water deficits (Arneith *et al.*, 1998a, 1998b; McMurtrie *et al.*, 1990; Richardson *et al.*, 2002; Watt *et al.*, 2003b). These studies have all shown reductions in growth with declining root-zone water storage as estimated by soil water balance models.

Less research has investigated the influence of water balance for forest productivity across the national extent. In a recent study, spatial estimates of soil water balance under the entire *P. radiata* estate shown in Figure 2.7 were used, in conjunction with other environmental variables, to estimate volume mean annual increment at age 30 (MAI300) for *P. radiata* across a large number of permanent sample plots (Palmer *et al.*, in prep). As previously described (section 2.2), this study showed mean annual fractional available root-zone water storage (W_f) to be a significant determinant of MAI300. MAI300 did not decrease as W_f declined until values for W_f of 80% were reached (Fig. 2.8). After this threshold further reductions in W_f induced a decline in MAI300 across the data range to W_f values of 40%. At values of W_f below 60%, MAI300 was quite sensitive to reductions in W_f . For example, a reduction in W_f from 60% to 50% reduced MAI300 by 17% (20.8 vs. 25 $\text{m}^3 \text{ha}^{-1} \text{year}^{-1}$).

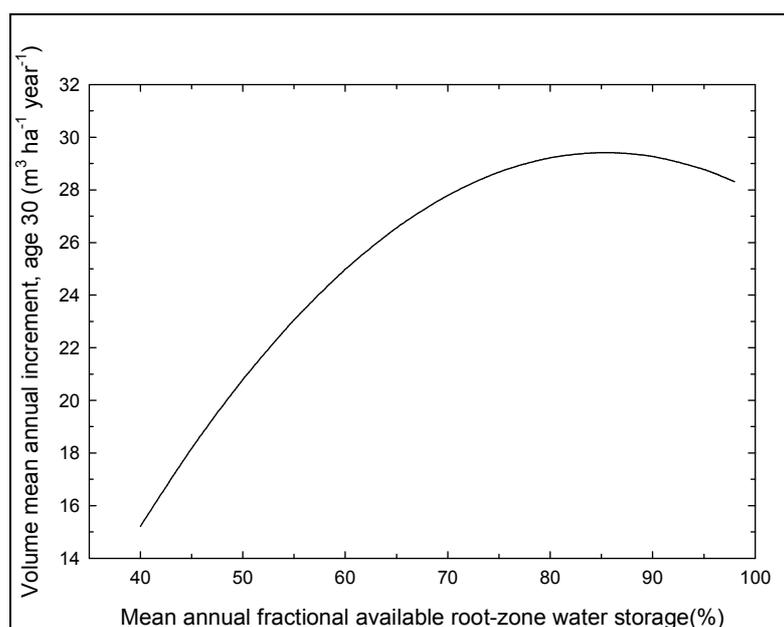


Fig. 2.8. Influence of mean annual fractional available root-zone water storage on volume mean annual increment. For this graph values for the mean number of frosts per month during autumn were held at average values within the national PSP dataset of 4.5 frosts month^{-1} .

While water availability is currently not limiting for most regions in New Zealand (Figure 2.7), it needs to be assessed whether this situation will be the same under different climatic conditions in the future. It may be possible that water limitations may be overcome in areas currently subject to regular water deficits, or, on the other hand, water limitations may emerge in areas currently unaffected by water shortage. Reductions in W_f by 10% are entirely possible, as this magnitude of change has occurred at a dryland forest in Canterbury over the last 60 years (Watt *et al.*, unpub. data). Assessing likely changes in water limitation is thus a key question for the future productivity of pine plantations in New Zealand.

This type of analysis requires an independent assessment of climate change effects on precipitation and water-loss rates. Anticipated changes in precipitation can be obtained from runs of global circulation models. The rate of water loss is affected by the interactions between physical processes (e.g. solar radiation, moisture saturation deficit of the air, temperature and wind speed) and plant processes (e.g. leaf area index, aerodynamic roughness of plant canopies and stomatal conductance). Some of these processes change in a predictable manner with climatic changes so that the net result of anticipated changes can be computed. The water availability for New

Zealand's plantation forests under climate change could therefore be determined by running water balance models such as SwatBal or CenW with input parameters that correspond to changing climatic conditions.

2.4.2 Evapotranspiration rate as a function of temperature

Water can be lost from stands of trees by evaporation from the surface of foliage, trunks, branches or the soil surface. It can also occur by transpiration, which is the loss of water from the inside of leaves or needles, from where it can be replenished from soil-water if that is available. These two processes are often jointly described as evapotranspiration.

Water can also be lost by surface run-off, sub-surface sideways flow or deep drainage. These losses are generally confined to conditions when there is excess water whereas conditions of water shortage are of more serious concern. The replenishment of soil water resources during wet periods, and the extent to which the soil can store water, are also of critical importance in determining the ability of forest stands to sustain productivity, or at least avoid tree mortality, over extended dry periods.

Evapotranspiration is strongly affected by temperature because warmer air can hold more moisture. This is diagrammatically shown in Figure 2.9 which shows saturated vapour pressure as a function of temperature. Saturated vapour pressure is the vapour pressure that air can hold at a given temperature without water condensing out.

As a generalised approximation, the actual humidity of the air can be taken as the vapour pressure at equilibrium with the previous night's minimum temperature (T_{min}). The vapour pressure at equilibrium with daytime temperatures (T_{day}) determines the humidity of the air inside the leaves of plants. The difference between those two vapour pressures gives the vapour pressure deficit (VPD) of the air, which is the principle driving force for evapotranspiration from plant canopies.

As a first approximation, more water can be lost by evapotranspiration as it gets warmer. If the diurnal temperature range (the difference between daytime maximum and overnight minimum temperatures) does not change (see below) it will lead to an increase in the vapour pressure deficit because the saturated vapour pressure curve is steeper at higher than lower temperatures (Fig. 2.9).

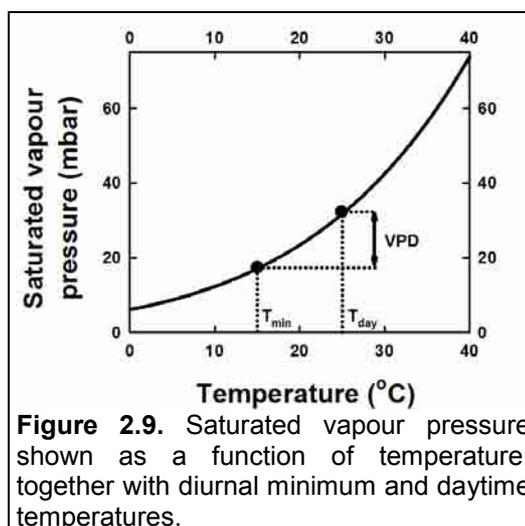


Figure 2.9. Saturated vapour pressure shown as a function of temperature, together with diurnal minimum and daytime temperatures.

Figure 2.10a shows the increase in vapour pressure deficit of the air with warming if there is no change in the diurnal temperature range. The increase in vapour pressure deficit with increasing temperature is only marginally different for different diurnal temperature ranges or daytime temperatures, with increases in VPD being between 5 and 6% °C⁻¹ over most temperature combinations. Only at very low temperatures does the increase in VPD exceed 6% °C⁻¹, or decline below 5% at very high base temperatures (Fig. 2.10a).

These changes in VPD can then be used to compute increases in transpiration rate with warming. The calculations were done with the Penman-Monteith equation (Monteith, 1965; Martin *et al.*, 1989). Canopy and aerodynamic resistances were used as given in the Figure legend to represent typical values for different canopy types.

Canopies differ in the way that transpiration rates respond to environmental drivers. Forest canopies tend to be relatively open so that transpiration rates tend to vary primarily as a function of changes in vapour pressure deficit.

Transpiration in pastures, on the other hand, is largely controlled by net radiation so that transpiration rates are less affected by variations in vapour pressure deficit or temperature than in tree canopies (Jarvis and McNaughton, 1986). These differential effects are formalised through canopy and aerodynamic resistances included in the Penman-Monteith equation.

The simulations suggest only slight increases in transpiration rate for grassland systems because of the greater control of transpiration by net radiation rather than the vapour pressure deficit of the air. Calculated increases in transpiration rate range from 1% °C⁻¹ for canopies at 40°C to an increase of about 4% °C⁻¹ at 5°C (Fig. 2.10b).

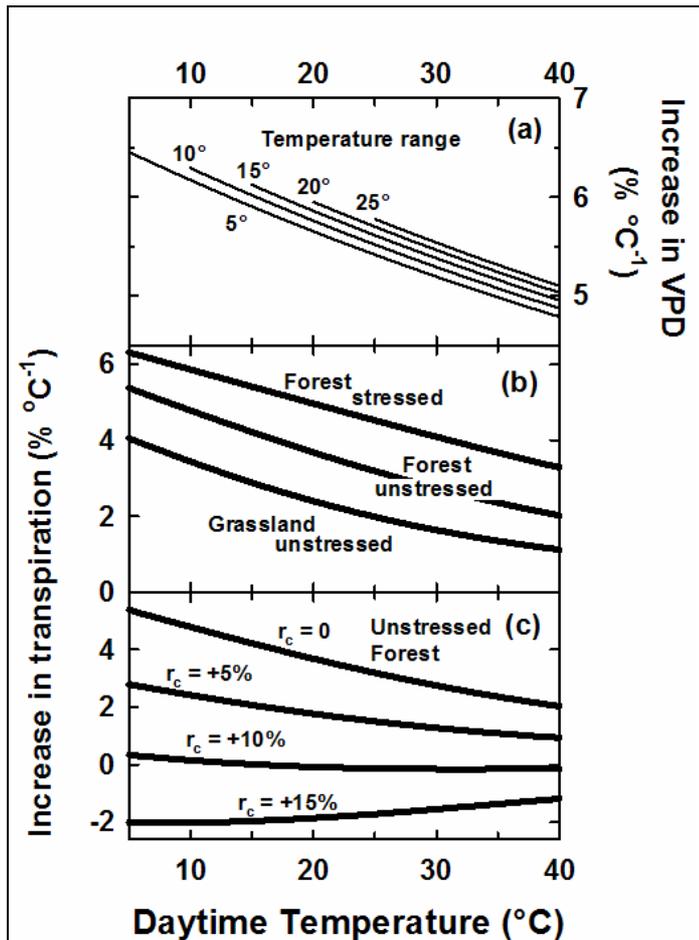


Figure 2.10. Change in vapour pressure deficit, VPD (a), transpiration rate with warming (b) and with warming plus stomatal closure due to increased CO₂ concentration (c). Change in vapour pressure is calculated as a function of daytime temperature and for a range of diurnal temperature ranges. Transpiration rate was calculated with the Penman-Monteith equation, with an assumed diurnal temperature range of 10°C, net radiation = 400 Wm⁻², and with parameters for aerodynamic and canopy resistance representative for grasslands (100; 50 s m⁻¹), unstressed forests (25; 50 s m⁻¹) and stressed forests with reduced conductance (25; 200 s m⁻¹). For calculations in (c), canopy resistance, r_c, was increased by 0, 5%, 10% or 15% as indicated in the Figure.

There is likely to be a greater increase in transpiration rate in forest systems (Fig. 2.10b) as transpiration rate in these systems is more strongly controlled by VPD and less by net radiation. Calculated increases ranged from 5% °C⁻¹ at 5°C⁻¹ to 2% °C⁻¹ at 40°C. Calculated increases in transpiration were even greater for forest systems under stress, with increases in transpiration rate ranging from 3 to 6% °C⁻¹ because the transpiration rate in stressed forests is even more strongly controlled by VPD. These calculations imply that under warmer conditions in the future, systems limited by the availability of water are likely to use available water faster. This would then result in less plant growth for the same amount of precipitation as it gets warmer. However, two other key factors also need to be considered: stomatal closure in response to increasing CO₂ concentration and changes in the diurnal temperature range.

Stomata of most plants are observed to close to varying degrees under increased CO₂ concentration. The effect of such partial stomatal closure is illustrated in Figure 2.10c for which the Penman-Monteith equation was run for a 1° increase in temperature and with some stomatal closure as indicated in the Figure. Stomatal closure by 10% would have the effect of almost completely negating the effect of increasing temperature by 1°C (Fig. 2.10c). Stomatal closure by more than 10% would lead to net reductions of transpiration rates, whereas stomatal closure by less than 10% would lead to increased transpiration rates. This increase would be more if there were no adjustments in stomatal conductance (Fig. 2.10c).

Figure 2.10 only indicates the sensitivity of the system to the indicated changes. The ultimate outcomes will largely depend on the relative rates of increases in CO₂ concentration and temperature and the extent of any physiological adjustment. Greater relative temperature increases will lead to greater increases in transpiration rate, whereas greater relative increases in CO₂ concentration will lead to less increase in the transpiration rate, or even a reduction. Similarly, species in which stomata are more sensitive to CO₂ concentration will experience a lower increase in transpiration rate than species with less sensitive stomata.

This leads to the question of how much stomatal closure can be expected in future. Stomatal adaptation has been shown on herbarium specimens in which the number of stomata on leaves has decreased with historical increases in global CO₂ concentration (Woodward, 1987; Rundgren and Björck, 2003; Kouwenberg *et al.*, 2003). The carbon isotope discrimination between ¹³CO₂ and ¹²CO₂ can also be used to infer changes in the intercellular CO₂ concentration, which can be related to historical changes in atmospheric CO₂ (Dawson *et al.*, 2002). Using this approach, Arneth *et al.* (2002) and Duquesnay *et al.* (1998) reported data that inferred some stomatal closure in response to increasing atmospheric CO₂, but Marshall and Monserud (1996) and Monserud and Marshall (2001) found no evidence of stomatal closure in their data sets.

Morison (1985), Allen (1990) and Urban (2003) compiled a range of observations from the literature, and showed that stomatal conductance across a range of species was reduced by about 40% when CO₂ concentration was doubled. Stomatal conductance in woody species, however, is generally observed to be somewhat less sensitive to CO₂ concentration, with Curtis and Wang (1998) reporting stomatal closure by just 11% and Medlyn *et al.* (2001) reporting 21% stomatal closure across a range of studies.

Medlyn *et al.* (2001) also reported that the response appeared to be weaker in older rather than younger trees and weaker for conifers than deciduous trees. Changes in stomatal conductance usually mirror changes in CO₂ assimilation rate so that the

ratio of intercellular to ambient CO₂ concentration is usually found not to change by growth in elevated CO₂ (Lodge *et al.*, 2001). Medlyn *et al.* (2001) further assessed that question by fitting the Ball *et al.* (1987) model to the available data. In the Ball *et al.* (1987) model, stomatal conductance is described explicitly as a function of CO₂ concentration and photosynthetic rate. Medlyn *et al.* (2001) found that under non-water-stressed conditions, the same model parameters held under ambient and elevated CO₂ which implied that the functional connectivity between photosynthesis and stomatal conductance is not altered by growth in elevated CO₂. Since the short-term response of photosynthesis to elevated CO₂ is well understood, the key uncertainty thus relates to the extent of a possible long-term acclimation of photosynthesis to growth in elevated CO₂, which has been discussed in a previous section of this report.

The preceding discussion has assumed that there will be the same increases in both daily minimum and maximum temperatures. However, observed temperature increases to date have been mainly due to increases in night-time temperature, with daytime temperatures having increased by only half as much as night-time temperatures (Karl *et al.*, 1993; Easterling *et al.*, 1997). This has been largely associated with an increase in cloudiness, which in turn is related to increasing industrial pollution, especially over more industrialised regions (Folland *et al.*, 2001). Simulations of future climate generally also show a decrease in diurnal temperature range, although by less than the trend observed to date (Cubasch *et al.*, 2001; Meehl *et al.*, 2007).

Any decrease in the diurnal temperature range would have the effect of reducing the expected increase in VPD, or even lead to a decrease (Fig. 2.11a). With a 0.8/1.2°C (day/night) temperature increase, there would still be a slight increase in VPD, but for an even greater disparity between daytime and night-time temperature increases, VPD would actually decrease.

Changes in transpiration rate are also less if night-time temperature increases exceed those during the day, and for 0.7/1.3°C (day/night) temperature increases, there would be almost no change in transpiration rate (Fig. 2.11b). These combinations of temperature increases are similar to the combination of temperature increases observed to date (Easterling *et al.*, 1997). In New Zealand, too, observed increases in night-time temperatures have exceeded temperature increases during the day (see the earlier section on climate), but within the inherent measurement variability, it is difficult to confidently assign ratios of increases of daytime and night-time temperatures.

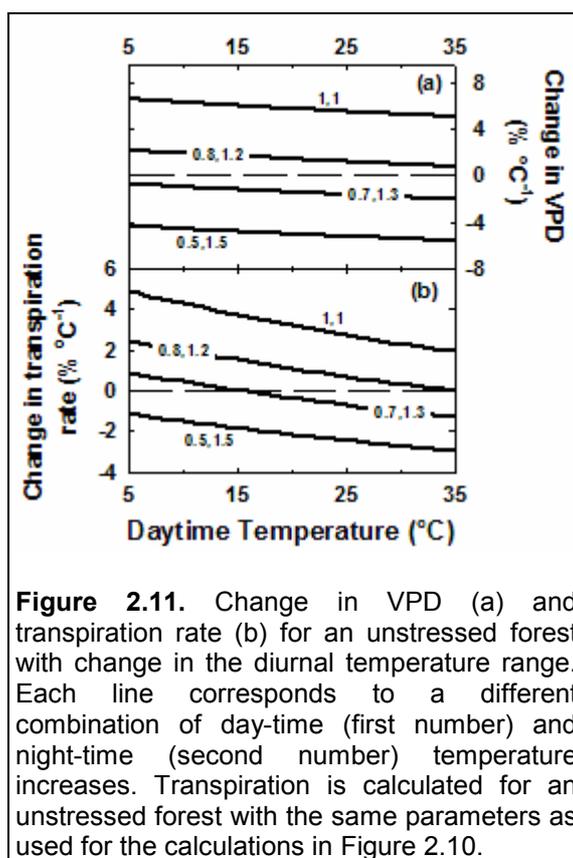


Figure 2.11. Change in VPD (a) and transpiration rate (b) for an unstressed forest with change in the diurnal temperature range. Each line corresponds to a different combination of day-time (first number) and night-time (second number) temperature increases. Transpiration is calculated for an unstressed forest with the same parameters as used for the calculations in Figure 2.10.

Taken together, the reduction in diurnal temperature range plus any stomatal closure due to elevated CO₂ might completely negate any increase in transpiration with global warming. Therefore, for the combination of climatic changes observed to date and anticipated into the future, it seems most likely that there will be some decrease rather than increase in transpiration from most plant canopies.

Changes in future water loss constitutes an important and controversial topic in the current scientific literature. As early as 15 years ago, McKenney and Rosenberg (1993) drew attention to the importance of using an appropriate function for calculating changes in transpiration rates with global warming. Potential evapotranspiration rates can be estimated using a variety of equations with varying physical rationales. Some workers have based their assessments on the widely used Thornthwaite method (Thornthwaite, 1948), which partly, although implicitly, relies on the observed correlation between evapotranspiration and temperature in the current climate.

The Thornthwaite method or derived products are frequently used because they use few parameters and they can give adequate calculations of transpiration in the current climate. The method works reliably under current conditions because the two basic drivers of transpiration, radiation and vapour pressure deficit, tend to be correlated with each other. However, that correlation is unlikely to persist into the future as temperature increases are likely to be greater than corresponding changes in radiation. Simplified approaches of calculating evapotranspiration that depend on this correlation of temperature and radiation are thus likely to have overestimated future changes in evapotranspiration (McKenney and Rosenberg, 1993).

Consequently, workers who have used the Thornthwaite method, or methods such the Palmer drought index based on the Thornthwaite method, have generally concluded that water may become much more limiting with temperature increases in the future (e.g. Gleick, 1987; Rind *et al.*, 1990; Leichenko, 1993; Dai and Qian, 2004; Burke *et al.*, 2006). Use of the Penman-Monteith equation with its more complete representation of the relevant physical processes, leads to the conclusion that increases in drought incidence will not increase by as much as would be predicted by studies that rely on these simpler approaches (McKenney and Rosenberg, 1993; see also the earlier Figs. 2.10 and 2.11).

A more solid approach is given by the use of soil water storage as a direct output from GCM simulations in which the calculation of evapotranspiration is based on physically realistic formulations. Wang (2005) and Sheffield and Wood (2008) compiled average changes in soil water from a compilation of several available GCMs and concluded that drought incidence is likely to increase globally. While the findings of these workers do not invalidate criticism of the use of the Thornthwaite method, it indicates that drought problems might have to be anticipated even after setting aside the methodological problems of many past studies. The potential for future droughts under changed climatic conditions constitutes such an important aspect of plantation performance that more work is warranted to assess whether there is likely to be a future drought risk for New Zealand, and globally. It is important for such work to be based on the best description of the relevant physical processes and include any possible plant physiological adjustments.

2.4.3 Rainfall effects

Rainfall is the main source for supplying moisture for transpiration and any changes in rainfall together with the changes in evapotranspiration discussed in the previous

section, will determine any possible changes in water availability. In addition, rainfall intensity is likely to increase because warmer air can hold more moisture (see earlier chapter) with likely increases in the risk of flooding and erosion, and as an agent for disease susceptibility (see pest and disease section). Changes in the frequency and intensity of rainfall events can also change the the amount of precipitation that can be lost by surface runoff, which can affect the replenishment of soil water even if the annual rainfall amount does not change.

In the absence of irrigation, rainfall is generally needed to meet the water requirements for trees. Rainfall contributes to the water supply that trees can access through their roots. The available water supply in soils is therefore the critical reservoir that trees can use and depends not only on water supply but also on the water storage capacity of soils in the root zone. Therefore water supply and water storage are usually linked to each other and models that estimate fractional root zone water storage or soil water balance need to be used to describe the availability of water for trees.

2.4.4 Summary of the key issues that need to be considered in assessing water-balance effects on productivity

1. Changes in water balance are affected by the combined effects of changes in inputs (i.e. precipitation) and outputs (mainly evapotranspiration), with site-specific effects modulated by variations in the water-holding capacity of specific soils.
2. Growth is influenced by water balance and a range of other environmental variables including solar radiation, temperature and vapour pressure deficit. As these variables are also likely to change in concert with soil water balance, it is important to consider their combined effects on growth. One approach is the use of process-based model that allows specific modelling of those interactions.
3. Increasing atmospheric CO₂ will increase water use efficiency which can partly or fully compensate for any decrease in water availability.
4. The effect of limited water supply is likely to vary widely between tree species and within tree species. For instance, different genotypes of *P. radiata* have been demonstrated to have a wide variation in water use efficiency.
5. Changes in the temporal patterns of rainfall in conjunction with site-specific differences in soil water holding capacity are likely to differentially affect soil-water availability at different times of the year.
6. There has not yet been an integrated quantitative assessment of likely changes in water limitation for plants in New Zealand. Such an assessment would need to be based on scenarios for rainfall changes, and use a physically-realistic estimation of changes in evapotranspiration with climate change. It would also need to include any physiological modification of stomatal conductance in response to increases in CO₂ concentration.

2.5 Climate change effects on soil processes

For growth, plants also require nutrients, especially nitrogen and phosphorus. Most nitrogen and phosphorus is derived from the decomposition of soil organic matter. In the decomposition of soil organic matter, CO₂ is released to the atmosphere and any excess nitrogen and phosphorus is mineralised and becomes available for plant uptake. If plant growth is increased by increasing CO₂ but nutrient uptake is limited, then the plant internal nutrient status declines as is usually observed experimentally (e.g. Drake, 1992; Tissue *et al.*, 1993). This provides a first negative feed-back effect on plant responses to increasing CO₂ concentration. Plants that fix more carbon also produce more litter which adds to soil organic matter and immobilises nutrients. This reduces the nutrients available for plant uptake and provides a second negative feed-back effect (Rastetter *et al.*, 1992, 1997; Comins and McMurtrie, 1993; Kirschbaum *et al.*, 1994, 1998), and process more recently termed 'progressive nitrogen limitation' by Luo *et al.* (2004).

These two processes restrict the possible positive response of plant productivity to increasing CO₂ concentration. However, temperature increases can also play a role. There are likely to be only slight direct effects of increasing temperature on plant function (as discussed above), but temperature can stimulate the rate of organic matter decomposition and the mineralisation of nitrogen (Kirschbaum, 2004). With increasing temperature, more nutrients can become available for plant uptake, and that can stimulate plant productivity independent of any direct physiological effect of increasing temperature (Schimel *et al.*, 1990).

Figure 2.12 shows the relative temperature response of organic matter decomposition rate (after Kirschbaum, 2000) and net primary production (after Lieth, 1973 – see also Figure 2.3).

The comparison shows that with increasing temperature, the rate of organic matter decomposition is likely to be stimulated much more than net primary production so that nutrients are likely to become more readily available under most circumstances. That would be at the expense of a loss of soil organic carbon. Soil carbon is important for maintaining soil structure, as a global carbon pool and in plantation forestry to supply nutrients and maintain tree growth.

If increasing temperature can improve nutrient availability, it might mean that trees will have sufficient nutrients available to make use of any potential growth enhancements through elevated CO₂. Interactions through the turn-over of soil organic matter affect only nitrogen, phosphorus and sulphur but not other potentially limiting elements, such as boron, that form no organic complexes with soil organic matter. Nutrient deficiency mapping based on foliar content identified areas for fertiliser application to enhance growth in the past and present (Will, 1985). The current foliar

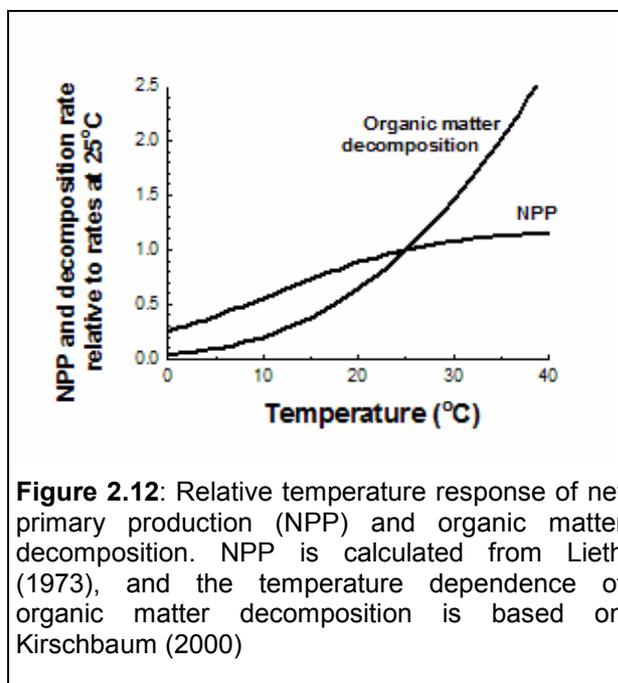


Figure 2.12: Relative temperature response of net primary production (NPP) and organic matter decomposition. NPP is calculated from Lieth (1973), and the temperature dependence of organic matter decomposition is based on Kirschbaum (2000)

nutrient status for nitrogen in exotic forests is shown in Figure 2.2 which indicates that there are only small areas with a marginal nitrogen level and that most forests have adequate levels of nitrogen in their foliage. Warming may improve this favourable status even further, but because of the current adequate nutrient status, this is unlikely to greatly affect future productivity.

2.6 Integrating responses to temperature, CO₂, nutrition and water availability in a modelling approach

Forest growth is ultimately determined by the interacting cycles of carbon, water and nutrients. Plants grow by fixing CO₂ from the atmosphere, but in the diffusive uptake of CO₂, trees inevitably lose water. Water can be replenished from the soil as long as adequate soil water is available. Otherwise, further water loss must be prevented by stomatal closure which then also prevents CO₂ fixation.

The relativities between water loss and carbon gain are affected by temperature, which affects the vapour pressure deficit and transpiration rate, and CO₂ concentration which affects the rate of photosynthesis. With increasing temperature, more water is lost per unit carbon gained, and with increasing CO₂ concentration, more carbon can be gained per unit water lost.

Plants also require nutrients, especially nitrogen and phosphorus. Most nitrogen and phosphorus is derived from the decomposition of soil organic matter. Nutrient supply is both a co-limitation that affects the primary response to aspects of climate change, and nutrient supply can be directly affected by climatic changes. Ideally, all these interactions and feedbacks should be modelled with a process-based model that can integrate these linkages and internal feed-back processes.

2.6.1 Linkages and possible effects

There has not been a fully quantitative, detailed assessment of the likely impacts of climate change on forest productivity in New Zealand. However, from an understanding of the general relationship between the key drivers of growth and productivity, the following general points can be made (Figure 2.13). Photosynthesis is likely to respond positively to the increase in CO₂ concentration, which is expected to increase the growth rate of trees. The magnitude of this response will increase with nutritional status and will be more pronounced under water-limited conditions. For New Zealand, the CO₂ fertilisation effect is, therefore, expected to be moderate. While a generally good nutritional status will allow CO₂ fertilisation to be expressed, the beneficial effect of increasing water-use efficiency, which would allow a quantitatively greater CO₂ response, is likely to be restricted to only a few areas of the country.

With expected warmer winter temperatures, the length of the growing period in New Zealand will increase. As *P. radiata* has little dormancy over winter, this is likely to result in either little change in growth or growth gains in regions with a high frequency of frosts at the current time. Warming might allow an extension of plantation forests to higher altitudes. With an overall increase in temperature and no changes in diurnal patterns the ratio between photosynthesis and respiration may remain constant.

Vapour pressure deficits will change with temperature and may lead to increased transpiration, although the relative changes in minimum and maximum temperature add an important uncertainty to this connection. Increases in transpiration may also

not eventuate if increased CO₂ lowers stomatal conductance. The main driver for changes to the water balance might then be possible changes in the amount of precipitation projected to be received under climate change. Changes in the amount and timing of precipitation will be critical and will determine whether drought and water stress may increase in frequency and/ or severity.

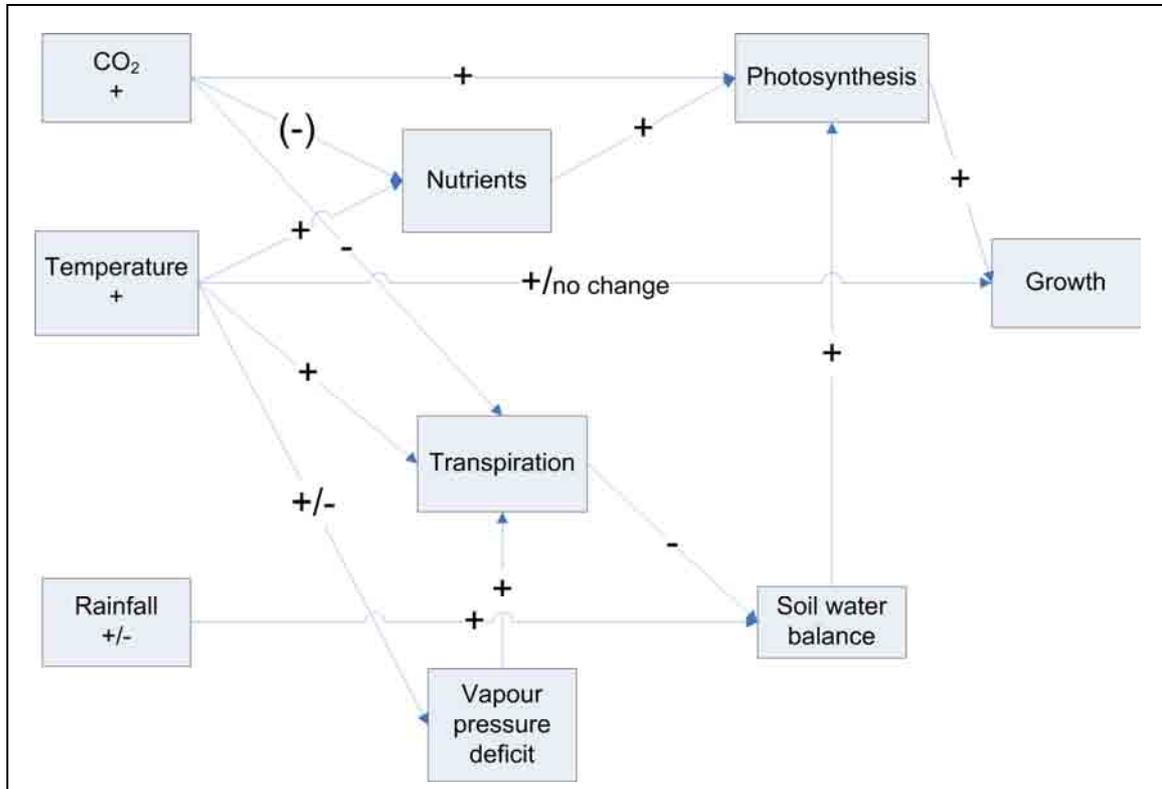


Figure 2.13. Potential impacts of CO₂, temperature, rainfall and nutrients on photosynthesis, respiration, transpiration and growth and their interaction. Indicative tendencies of the effects are given.

2.6.2 Modelling approach

Because of the importance of these various indirect and interacting effects, any assessment of climate change effects on forest productivity cannot realistically be carried out with simple models that lack most of these interactions. The integration of these multiple factors and their feedbacks therefore requires process-based or at least hybrid models that link these components. The CO₂ response of photosynthesis and feedback from CO₂ stimulated growth enhancement and modification by other factors on the strength of the CO₂ response all need to be a major part of such models. This will enable the combined effect of CO₂, temperature, nutrient and water balance changes on future plantation productivity to be captured.

Running such a model under various climate scenarios for New Zealand's conditions will allow the study of the response of our plantation forests to the expected changes. Process-based models (mechanistic models) will allow such a projection beyond our range of current experience. To gain confidence in the modelling approach, it will be essential to test such a model under current New Zealand conditions as an important first step.

One of the available models that have been developed for *P. radiata* in Australia and allows the modelling of these interactions is CenW (Kirschbaum, 1999b). The model has an extensive parameterisation requirement. Simpler models may be easier to parameterise but do not include some of the important feedbacks and may not be even be responsive to the effects of CO₂.

Whatever model is used, it needs to be parameterised for the different tree species that are of interest in the New Zealand context. Data for species other than *P. radiata* may be difficult to obtain.

It is important for a growth model to be able to be used spatially. This would allow its use at various spatial scales and with data that is spatially variable, such as climate and soils information. Currently available mechanistic models that have this functionality and are available are rare (an example is 3PGspatial). However, a few models (e.g. CenW) have been run spatially in the past and can be readily integrated into a spatial framework in New Zealand.

To apply and test a selected model for New Zealand conditions an initial step would be to run it for sites where necessary data are available. It could then be compared against observations over a wide range of climatic and edaphic conditions. This will also require the availability of the necessary spatial data to run it for a range of conditions.

In the preceding sections, we have discussed a range of issues, relationships and factors that need to be considered, both alone and in interaction with each other, in determining the effect of climate change on the productivity of New Zealand's forests. A comprehensive computer model, run spatially across the different climatic zones, and forced with different climate-change scenarios, would be one practical way of quantitatively integrating these various factors. Such a task is not trivial, but can be accomplished if it is undertaken in a coordinated manner by a team of scientists with the relevant expertise. Without such an integrated approach, climate-impact assessment will remain disjointed and able to investigate only isolated aspects of the whole picture.

3 Influence of secondary factors on planted forests

Summary

Objective

Summarise the current knowledge of likely climate-change effects on abiotic (wind and fire risks) and biotic (insects, diseases, weeds) factors, and how changes in these secondary factors are likely to influence plantation productivity. Quantify, where possible, the potential magnitude of these secondary effects on plantation productivity and describe how any interactions between secondary effects will influence plantation productivity. Identify any uncertainties in these areas, and how they can be addressed through further research.

Key results

Climate change is likely to affect many abiotic and biotic factors, which may in turn affect plantation growth and productivity. The main factors which we have considered here include weeds, insects, pathogens and the risks from wind and fire. All of these factors currently cause significant economic losses in planted forests.

Climate change is likely to increase fire risk and the incidence of damaging winds. Previous research has shown that fire danger is likely to increase significantly in most areas of New Zealand, and that the length of the fire season will probably increase. Eastern areas would be most affected, with the total number of days of Very High and Extreme forest fire danger increasing by more than 50%, or 20 days in several regions. Increases in fire risk are likely to result in an increase in the incidence of fires and plantation area burned.

It is also predicted that the westerly wind speed component will increase during the winter and spring periods, leading to an increase in the mean and extreme wind speeds for many regions of New Zealand. Increases in extreme wind speeds are predicted to occur in the east and south of the country. For the central and upper regions of the North Island, which contain a large proportion of the plantation estate, extreme wind speeds are predicted to decrease. However, the upper and eastern parts of the North Island may be subject to more severe extra-tropical cyclones. There is still considerable uncertainty about projections of the future extreme wind climate, particularly the impacts of future climate change on the magnitude and frequency of extra-tropical cyclones.

In those regions where there is a predicted increase in the severe wind climate, it is likely that the annual frequency of winds sufficient in magnitude to cause widespread damage to forests could increase substantially. Previous research and simulations indicate that the impacts from these projected increases in extreme wind speeds will vary widely, both within and between regions due to differences in the underlying vulnerability of forests in these regions. These impacts are likely to range from little or no change, to a significant increase in the risk of wind damage to planted forests.

Biotic factors are also likely to be strongly influenced by climate change. As the distribution of weeds, insects and pathogens is strongly determined by climatic conditions, changes in these conditions are likely to result in shifts in the geographic range of many species. Under climate change, global regions from which future

invasions might occur are also likely to shift. Simulations suggest that the global areas that pose an invasion risk for New Zealand could expand, and this report highlights the new areas that might currently harbour potential future invaders.

Climate change is likely to result in range expansion within New Zealand of a number of native Australian tree species, such as *Eucalyptus* and *Acacia* spp. and the proliferation of currently ornamental and potentially invasive weed species such as *Melaleuca quinquenervia* (paperbark tree) and *Pueraria montana* (kudzu). With such range expansions from their current limited distributions in the north of New Zealand, these species might become major problem weeds in the future as they are both highly invasive weeds in other parts of the world.

Climate change is also likely to affect growth rates of weeds through changes in CO₂ concentration, root-zone water storage, temperature and changing length of the growing season. It is difficult to ascertain the influence of many of these factors on weed growth without further research.

Changes to weed composition and growth rates resulting from climate change are likely to have a detrimental effect on tree growth. If dryland areas expand within New Zealand, as currently predicted, competition from weeds will increase, as weeds compete more strongly for site resources on dryland than wetland sites. As tall woody tree species from Australia compete further into the rotation than our current suite of herbaceous and shrubby weeds species, further invasion by these trees is likely to have a detrimental influence on plantation productivity. Given the large number of weed species, it is likely some will respond strongly to CO₂ fertilisation and as a result may become more invasive and compete more strongly with plantation species than at present.

Because of our limited knowledge of climate effects on forest insects in New Zealand, it is difficult to make any generalised assessment about likely climate-change effects on their abundance and distribution. However, it is likely that climate change will increase the risk of establishment of new species from warm-temperate or subtropical regions and generally result in greater abundance of insect pests due to their better survival over winter.

Without further research it is difficult to determine how changes in insect abundance and distribution will influence plantation productivity. However it is likely that in areas that become drier under climate change there will be reduced tree resistance to insect attack due to stress caused by drought or increased temperature. It is also possible that the effectiveness of natural enemies will be reduced due to increased climatic mismatches with the phenology of hosts or prey. Conversely, climate change may also lead to reduced problems with particular insect pests. For example, if growing conditions for trees improve (e.g., due to increased CO₂) this may lead to increased tree resistance to pest attack. Further research is necessary to clarify how climate change will influence insect distribution and abundance and how these changes will impact plantation productivity.

Climate change is unlikely to greatly influence the distribution of the main needle-cast diseases of radiata pine in New Zealand as these are already found throughout the country. Climate change is, however, likely to change the regional incidence and severity of some fungal diseases. It is likely that *Dothistroma* needle blight will remain rare in dry east coast regions, which are projected to become even more arid under climate change. The severity of Swiss needle cast, caused by *Phaeocryptopus gaeumannii*, which is the most widespread disease of Douglas-fir is likely to increase with climate change throughout the country as pathogen abundance is strongly

correlated with winter air temperature. Further modelling using process-based distribution models is necessary to determine the influence of climate change on the distribution and abundance of other pathogens of radiata pine and alternative species within New Zealand.

For the major diseases of radiata pine, changes to pathogen distribution and abundance will need to be quantified before the influence of these on plantation productivity can be determined. It is very likely that the anticipated increases in severity of Swiss needle cast under climate change will reduce the productivity of Douglas-fir in New Zealand. In order to more accurately determine the magnitude of these growth losses, models should be developed which link pathogen abundance to needle loss and crop productivity.

Under climate change, there are also a number of interactions between abiotic and biotic factors, which need to be considered in order to gain a full understanding of likely climate change effects on productivity. Almost all of these interactions are likely to have an overall negative impact on plantation productivity for the following main reasons. Under climate change, reduced phenological synchrony is likely to diminish the effectiveness of insect biocontrol agents on weed growth. Increased windthrow and tree damage resulting from greater extreme windspeeds are likely to trigger outbreaks of damaging insects, and provide entry points for pathogens. Similarly, increased fire risk associated with climate change could lead to insect outbreaks after areas have been damaged by fires. Increased insect attack and windthrow can also increase fire hazards by producing higher dead-fuel loads. Projected increases in the distribution of woody weeds in dryland areas could also increase fire risk as these weeds generate large amounts of highly flammable fuels.

3.1 Abiotic factors

3.1.1 Wind

3.1.1.1 Introduction

Wind is an important disturbance agent in New Zealand's indigenous forests (e.g., Reid, 1948; Shaw, 1982; Jane, 1985; Ogden, 1985). It can cause damage to isolated trees or small groups of trees, while at the other extreme it can be a major stand replacing disturbance whereby several hundreds of hectares may be blown down. This is important for promoting ecological succession in natural forests (Oliver and Larson, 1996). However, wind has also caused catastrophic damage to New Zealand's planted forest estate (e.g., Somerville, 1995) which in turn has led to direct economic losses. With the rapid expansion of the planted forest estate in the 1990s, substantial amounts of damage have also occurred to young (< 3 years old) stands, particularly those established on fertile ex-farm sites (Moore *et al.*, 2008). This phenomenon is commonly referred to as juvenile instability or toppling. While the trees generally recover, the resulting stem sinuosity often leads to logs being downgraded and hence a reduction in value.

Because of the impact that wind has historically had on the planted forests of New Zealand, it is important to have an understanding of how this risk might be expected to change in the future under possible climate change scenarios. This section reviews the historical occurrence of wind damage in planted forests and then focuses on approaches that have been used to model the risk of damage. Potential changes to the wind climate in New Zealand under the range of predicted climate change scenarios are reviewed and these are combined with models of wind damage risk to determine how this risk may change in the future. Finally, studies conducted in other countries which have investigated the potential impacts of climate change on the risk of wind damage in forests are critically reviewed in order to determine whether their findings could provide some indication as to the likely impacts of climate change on the risk of wind damage in New Zealand's planted forests.

3.1.1.2 Historical level of wind damage in New Zealand's forests

New Zealand does not have a formal scheme for reporting wind damage, unlike many European countries. Where records do exist, these have often been kept by individual forest managers, researchers or companies and generally relate to single storm events where mature or semi mature stands were totally or partially destroyed. Such records date back to the 1940s, with many of the main events documented in published accounts (e.g., Chandler, 1968; Irvine, 1970; Prior, 1959; Somerville *et al.*, 1989; Wendleken, 1966; Wilson, 1976,). Since the dissolution of the New Zealand Forest Service in 1987, records of wind damage events were kept by scientists at the New Zealand Forest Research Institute. These include numerous unpublished records, inventories and aerial photographs, however they are by no means complete and do not include damage from numerous smaller storms, much of which is undocumented. Records are also kept by those insurance companies who offer wind damage cover, particularly when a claim has been made. At present, approximately 100,000 ha of the plantation estate in New Zealand is insured against wind damage (Bevin Maybe, AON New Zealand Ltd, pers. comm.) with premiums based on historical losses. However, additional damage to young plantations, very often appearing as severely leaning stems, has largely remained undocumented. Also remaining undocumented have been on-going attritional losses in stands from

lesser storms. However, in permanent growth sample plots (PSPs), stand condition in New Zealand forests has been monitored from the onset of plantation forestry and as trees lost to wind throw are described, these records can provide some insight into attritional wind damage losses.

Available records indicate that since the 1940s, at least 50 000 hectares of wind damage has occurred in planted forests (Table 3.1), with at least 8 million m³ of timber salvaged following wind storms (A. Somerville, pers. comm.). Using data from 17 previously State-owned forests covering an area of 259 950 ha, Somerville (1995) calculated that the average overall level of damage corresponded to 12.2% of the net stocked area for a 28-year rotation. This figure is based on data from catastrophic events where areas greater than 1 ha in size were damaged and also attritional damage which was estimated from PSP records. There was considerable variation in the total amount of damage, with the least affected forests only losing 5-6% of their net stocked area, while the worst affected would lose nearly all of their stocked area.

Thompson (1976) suggested that wind damage accounts for less than 1% of the annual cut in New Zealand; however, these calculations were made prior to Cyclones Bernie and Bola which occurred in the 1980s. These calculations have been revised by assuming that on average approximately 1 000 ha are damaged in New Zealand by wind each year with the average volume per hectare in these stands being 400 m³. This means that the average annual volume damaged is roughly 1.8% of the annual cut (based on an annual cut of 22 million m³).

Table 3.1. A chronology of wind damage in New Zealand forests¹ (adapted from New, 1989a).

Date	Region	Area damaged (ha)
13 July 1945	Canterbury	1 500
July 1956	Central North Island	166
July 1958	Central North Island	150
March 1964	Canterbury	5 200
April 1968	Canterbury	1 000
20 April 1968	Nelson	963
1 August 1975	Canterbury	11 000
1979	Central North Island	600
10 April 1982	Central North Island	6 300
7 March 1988	Central North Island	19 000
7 March 1988	Northland	1 484
November 1994	Hawke's Bay	500
April 1996	Central North Island	900
February 2004	Lower North Island	1500
October 2004	Nelson	1100
Total		51 363

Prior to the late 1970s, wind damage was generally viewed as a problem that was mainly confined to Canterbury and other parts of the South Island such as Nelson and Otago/Southland. However, this view changed as a result of storms which occurred in the central and upper North Island in 1979, 1982 and 1988. The 1979 storm caused damage to 650 ha of stands in Kaingaroa Forest, while the storms in

¹These records do not include damage from many smaller storms which have resulted in considerable volumes of timber being salvaged from stands (e.g. Prior, 1959; Chandler, 1968; Irvine, 1970).

1982 and 1988 caused damage to areas of over 5 000 ha and 17 000 ha, respectively (New, 1989a).

3.1.1.2.1 Wind conditions giving rise to damage

Damage to forests has generally been the result of winds associated with sub-tropical cyclones or those enhanced by topography. Sub-tropical cyclones have caused damage in the northern and central areas of the North Island (Littlejohn, 1984; Brown and Jones, 1989; Carter, 1989), while orographically enhanced winds have caused damage to forests on the Canterbury Plains (Prior, 1959; Wendelken, 1966; Wilson, 1976) as well as in Hawke's Bay and Gisborne. The direction of damaging winds in Canterbury has usually been northwest (although in the 1968 storm and the Nelson storm of 2004 winds were from the southwest), while for Cyclones Bernie and Bola in 1982 and 1988, respectively, wind direction was southeast.

Because wind speeds are strongly affected by both large and small-scale topographic features, the wind speed at the actual location where damage occurs can be quite different to that recorded at the local meteorological station. These differences generally reflect differences in the broader scale wind speeds as well as differences in the localised exposure between the forest site and the meteorological station. Many of the meteorological stations in New Zealand with long-term records of wind speed and direction are located at or near airports, which are often considerable distances from the nearest forests. However, the close proximity of a meteorological station to the forest of interest does not necessarily guarantee that the wind speeds at this station will be representative of those at the site. Differences in topographic exposure and sheltering by other features such as buildings and vegetation can lead to large differences in the wind climate between two points which are only spatially separated by a short distance. Therefore, caution should be exercised when using data from meteorological stations to determine the magnitude of damaging winds in a particular storm event.

In general, it appears that damage to forests begins to occur when the mean wind speed exceeds approximately 25 m/s (90 km/h). However, the actual wind speed required to cause damage depends of the vulnerability of the stands in question, which is a function of their structure and the soils on which they are growing. Management activities can alter this vulnerability substantially (see Section 3.1.1.2.2). Mean wind speeds in excess of 30 m/s (108 km/h) would be expected to cause widespread and/or severe damage to stands.

There is also a considerable amount of variation in instantaneous wind speed during the averaging period. This is due to the passage of gusts, and the ratio of the maximum wind speed (over a 3 second period) within an averaging period to the mean is known as the gust factor. This gust factor is normally assumed to be approximately 1.6 in complex terrain with high surface roughness. Therefore, the maximum 3-second gusts associated with mean wind speeds of 25 m/s and 30 m/s are 40 m/s (144 km/h) and 48 m/s (173 km/h), respectively.

3.1.1.2.1.1 Quantifying the wind climate severity

Hourly² mean wind speeds generally follow a Weibull distribution (Fig. 3.1), with the location parameter indicating the most commonly occurring wind speed and the shape parameter reflecting the degree of skew at the upper end of the distribution. While the location parameter indicates the general severity of the wind climate, the shape parameter (together with the location parameter) provide an insight into the relative frequency and magnitude of extreme events. Using these parameters, it is possible to calculate the mean hourly wind speed with a given return period (e.g., 25 or 50 years) using the method described in ESDU (1988) as well as the average number of hours per year when the mean hourly wind speed exceeds a particular threshold value.

The severity of the wind climate can also be quantified using a time series of annual maximum wind speeds (either mean hourly or 3-second gusts). The distribution of annual maxima can generally be described using one of the generalised extreme value distributions, with the Fisher-Tippet Type 1 distribution often used (Cook, 1985). The two parameters describing this distribution are the mode and dispersion. The mode indicates the overall severity of the wind climate, while the dispersion parameter is a measure of the temporal variability. Using the mode and dispersion parameters for the extreme value distribution, it is possible to calculate the annual probability that a wind speed of given magnitude (V) is equalled or exceeded (AEP) and the corresponding average return period (RP) using the following equations:

$$AEP = 1 - e^{-e^{\frac{V-U}{a}}} \quad [3.1a]$$

$$RP = 1 / AEP \quad [3.1b]$$

This highly non-linear relationship between wind speed and return period is shown in Figure 3.2. From this it can be observed that wind speeds up to 15 m/s (54 km/h) occur on an annual basis, while wind speeds in excess of 30 m/s are rare and only occur at sites with more severe wind climates. At higher wind speeds, a small change in wind speed will result in a large change in average return period.

² These are generally calculated using data from the ten minute period before the hour. For example, the average wind speed during the period 1150 to 1200 is presented as the mean hourly value for 1200.

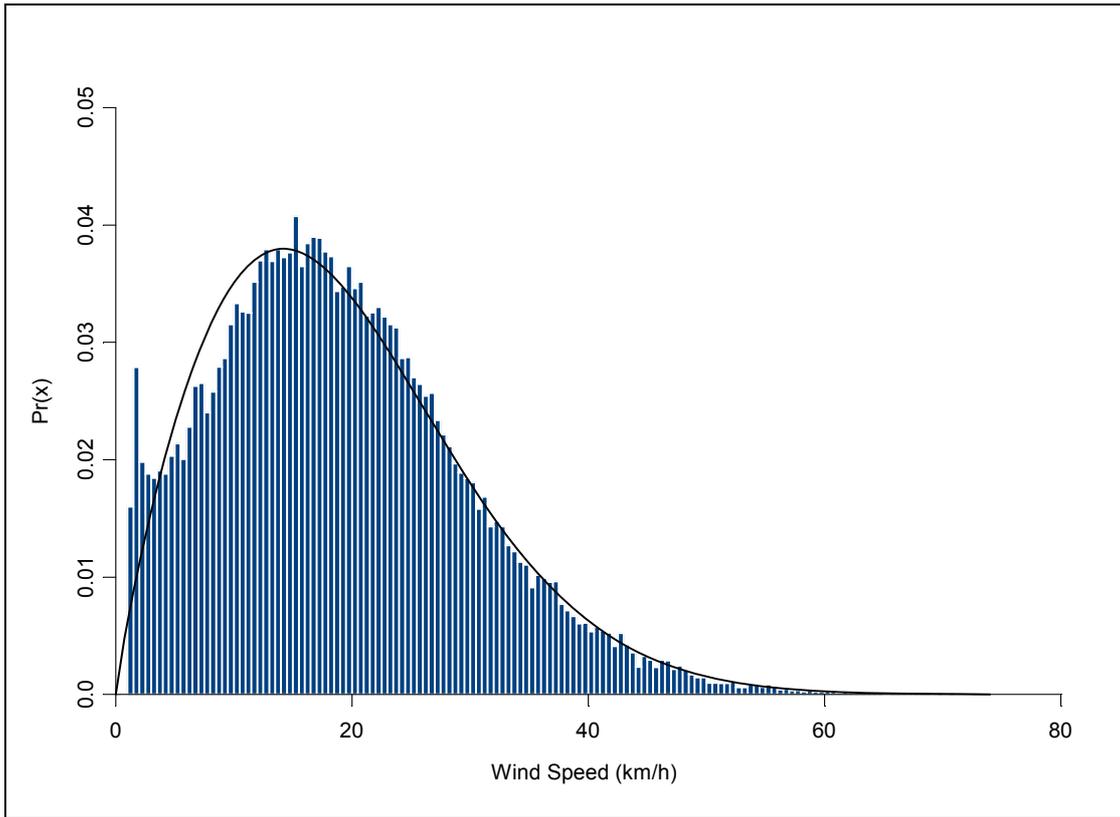


Figure 3.1. Distribution of mean hourly wind speeds from the meteorological station at Matea. The solid line corresponds to the Weibull distribution fitted to the data.

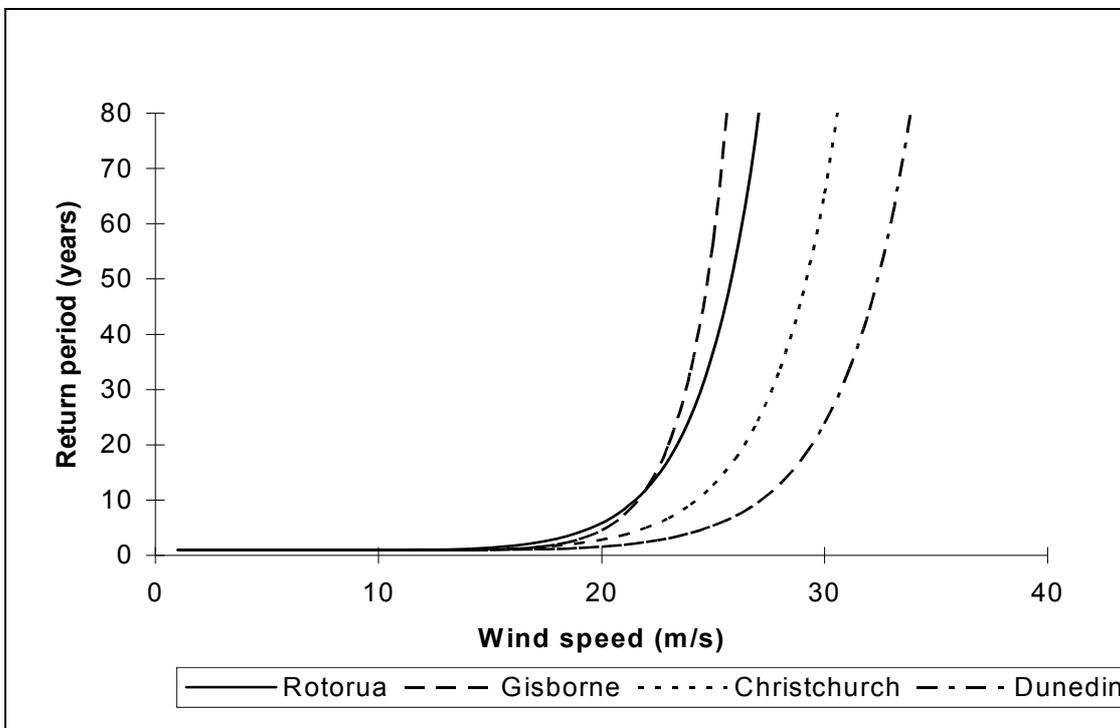


Figure 3.2. The relationship between wind speed and recurrence (expressed as return period) for four airport meteorological stations: Rotorua, Gisborne, Christchurch and Dunedin.

3.1.1.2.2 Other factors contributing to the risk of damage

While strong winds are an obvious pre-requisite for damage to occur, the risk of damage can be strongly affected by forest management activities (Somerville, 1980; Quine *et al.*, 1995). The two main activities which have the potential to increase the risk of damage are thinning and felling. Both of these practices can result in increased wind exposure to the residual trees and have been associated with a considerable amount of the past damage in New Zealand forests. Commercial (production) thinning, particularly when it is carried out in older stands (i.e., mean top height >16 m) and the residual tree density is low, can increase the risk of damage substantially. In these situations, damage could occur as a result of much lower wind speeds. Many of the stands that were severely damaged in the western part of Kaingaroa Forest during Cyclone Bola in 1988 had been production thinned (Wrathall, 1989). This practice has largely been discontinued due to concerns over this risk of wind damage. However, there is still a significant amount of damage resulting from associated upwind clearfelling. This clearfelling results in a sudden increase in wind exposure on the previously sheltered stand edge, leading to damage.

3.1.1.2.3 Economic impacts of wind damage

Wind damage can result in direct economic losses, which in some cases can be quite severe. The magnitude of these losses will generally depend on the type of damage (stem breakage or uprooting), the intensity of damage and the age of the stand at the time of damage, among other factors. Large scale wind damage events such as the 1975 storm in Canterbury or Cyclone Bola in 1988, affect wood flows by increasing short-term wood supplies (as a result of the windthrow salvage), but ultimately decrease sustainable yield due to a reduction in growing stock. The costs of salvaging wind damaged trees are higher than for conventional harvesting mainly due to reduced production rates. There can also be considerable danger to workers associated with salvaging wind damage (Childs, 1966). Revenues from salvage operations are generally reduced, particularly where there are high levels of stem breakage, but also because of fungal decay. The largest financial losses may in fact occur in mid-rotation stands where 15-20 years of costs associated with silvicultural investment have been incurred, but the value of the salvaged stand is substantially less because the trees are too young to be processed into higher value end products.

An overall analysis of the economic impacts of wind damage was conducted by Manley and Wakelin (1989) who showed that the increase in costs and reduction in revenues reduces the present net worth of a forest by up to 11% for an annual level of damage of 1%. This level of damage is similar to that assumed by Thompson (1976), but less than the 1.8% level based on the assumptions of 1 000 ha of damage per year.

3.1.1.3 Predicting wind damage

Two main approaches have been taken by researchers and forest managers in order to quantify the risk of damage and the factors that contribute to this risk. The first approach is to use data on the magnitude and frequency of wind damage to forests. Such an approach jointly analyses the two components of the risk of wind damage: the likelihood of a storm force wind occurring, and the magnitude of damage associated with that storm. While this approach is appealing because it uses actual data on the occurrence of damage, it has a number of limitations. Firstly, it requires a substantial time series of dependable data on the occurrence of wind damage. Its use is also questionable when there have been significant changes to the population

of interest. For example, the total area of plantation forests in New Zealand has increased from 356 000 ha in 1950 to 1.8 million ha in 2006 (MAF, 2007). However, within the estate many of the high risk forests on the Canterbury Plains have been converted to other land uses. In addition, the rotation length for stands has decreased from over 30 years (and in many cases over 40 years) to less than 25 years in some instances, and production thinning (a practice which has been associated with considerable amounts of damage in the past) is much less common. One of the key problems with this approach is that it is not possible to predict what will happen under a changing climate.

The second approach for predicting the risk of wind damage is the mechanistic or probabilistic approach. This uses detailed knowledge on the mechanisms of damage to calculate the threshold wind speed required to damage a stand of trees. The risk of damage is then defined as the probability that this threshold wind speed will be exceeded at a particular site. While this approach requires detailed scientific understanding of the mechanisms involved in wind damage, it does allow the forest manager to assess the implications of changing their silvicultural practice. An example of this approach is the ForestGALES model that has been developed by the British Forestry Commission's Northern Research Station (Gardiner and Quine, 2000; Gardiner *et al.*, 2000) and adapted for use in New Zealand by Moore and Somerville (1998) (Fig. 3.3). This model calculates the threshold wind speed required to damage a stand from information on the stand structure and the root anchorage strength obtained from tree pulling tests (e.g., Moore, 2000). An airflow model is then used to adjust this threshold wind speed for the effects of topography, and the average return period for this wind speed is then predicted using the recurrence functions described in Section 3.1.1.2.1.1. Because these functions are often fitted to relatively short time series of data (i.e., as short as 20 years), caution should be exercised when using them to predict wind speeds with return periods greater than the length of record used to develop them.

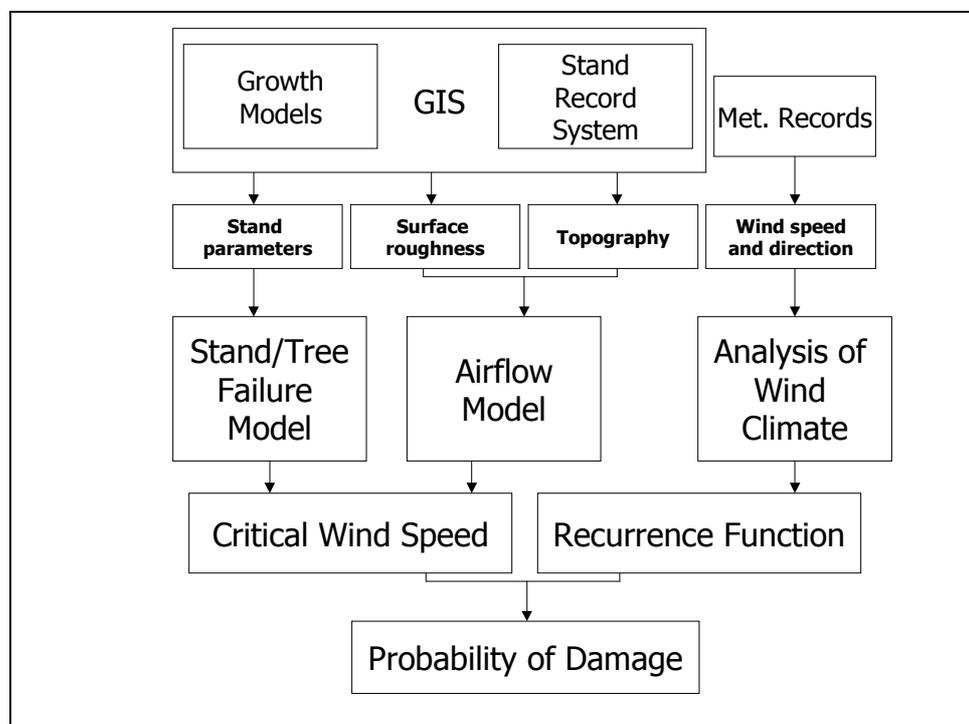


Figure 3.3. Schematic diagram of the deterministic/probabilistic wind risk analysis model.

This approach can model the effects of changes in the structure and location of forests, however the functions used to predict the return period of damaging winds are based on historical data which reflect the past wind climate. Therefore, in order for these types of mechanistic model to be used to predict the risk of wind damage under various climate change scenarios, the effect of climate change on the wind speed distribution must first be quantified.

3.1.1.4 Effect of climate change on strong wind climate

Compared with other climatic variables such as temperature and rainfall, wind has received less attention in climate change scenarios. It is expected, however, that global warming will introduce more energy into the atmosphere, which in turn will lead to greater pressure differences and increased wind speeds in certain regions of the world (Svensson, 2003). The effect of potential future changes in global circulation patterns on synoptic-scale winds have been investigated for a number of locations in the world using regionally-downscaled results from global circulation models (e.g., Hulme and Jenkins, 1998; Olofsson, 2006).

In New Zealand, the possible effects of climate change scenarios on the wind climate are discussed in Wratt *et al.* (2004) and more recently in Ministry for the Environment (2008). For mid-range temperature change projections, Wratt *et al.* (2004) indicated that the mean annual westerly wind component will increase by approximately 10% of its current value within the next 50 years. They also projected that there would be an increase in this westerly flow for all four seasons, however in the more recent report by the Ministry for the Environment (2008) there is actually a projected decrease in the summer and autumn westerlies. Winter westerlies show an even greater increase than in the original projections by Wratt *et al.* (2004). By 2080, Wratt *et al.* (2004) indicated that the mid-range projection is for a 60% increase in the mean annual westerly component of the flow, however this has been reduced to approximately 10% by 2090 in the updated projections (Ministry for the Environment, 2008). Again, the largest increase (both in absolute and percentage terms) will be in the winter months. As noted in Ministry for the Environment (2008), changes in the westerly wind component do not necessarily imply an increase in mean wind speed. Higher temporal resolution data is required to quantify future changes in wind speed. However, preliminary analyses undertaken by NIWA indicate that the winter wind speeds may only increase by 1-2% for much of the country, with maximum increases of approximately 5-6%. These relative changes are similar in magnitude to the 1-2% predicted for the United Kingdom and Ireland by Hulme and Jenkins (1998). There is also considerable spatial variation in the changes in mean winter wind speeds, with central and southern Otago and Fiordland showing the greatest projected increases by 2090, with most of the central and upper North Island projected to have a 1-2% decrease (Ministry for the Environment, 2008).

Wratt *et al.* (2004) also presented predictions on the future extreme wind climate. This was based on an analysis of the distribution of wind speeds and made the assumption that the shape of the distribution remained unchanged, but that the value of the location parameter increased. Based on this analysis, Wratt *et al.* (2004) predicted that under the mid-range temperature change scenarios the magnitude of the annual maximum wind speed could increase by 3% by 2080. In simple terrain (i.e., over the sea or flat land), they predicted that the annual frequency of occurrence of winds in excess of 30 m/s could increase by 30% by 2030 and 100% by 2080. In the updated report by the Ministry for the Environment (2008), it is projected that the greatest increases in the extreme winter wind speeds (i.e., 99th percentile wind speeds) will be in the eastern and southern parts of New Zealand, with Bay of Plenty, the Central North Island and Taranaki projected to have a decrease. The upper and

eastern regions of the North Island could be more vulnerable to extra-tropical cyclones as it is predicted that future tropical cyclones are likely to become more intense with larger peak wind speeds and more heavy precipitation. However, as Wratt *et al.* (2004) note: “*Uncertainties in these projected changes in extreme wind speeds are considerable. The IPCC in its Third Assessment Report says little about strong winds, but states that for some extreme phenomena there is currently insufficient information to assess recent trends, and climate change models currently lack the spatial data required to make confident projections*”. Therefore, one of the key priorities in order to better understand the effects of climate change on the risk of wind damage is better information on the future wind climate, particularly the magnitude and frequency of extreme events.

3.1.1.5 Possible effects of changes in wind climate on wind damage risk

A small number of studies have investigated the impact of possible future climate change on the risk of wind damage (e.g., Peltola *et al.*, 1999; Quine and Gardiner, 2002; Svensson, 2003; Olofsson, 2006). The most comprehensive of these was performed for southern Sweden by Olofsson (2006) who linked a physiologically-based tree growth model with a mechanistic wind damage model, which included a modified wind climate. In this study, Olofsson (2006) found that increased temperatures under the global warming scenarios investigated resulted in increased tree height growth, which in turn was the main factor responsible for an increase in the risk of damage. In colder Nordic countries such as Sweden and Finland, an additional aspect of climate change is predicted to be a reduction in the duration of frozen soil conditions during the winter months (Peltola *et al.*, 1999). This will lead to a reduction in the strength of the root anchorage and an increased risk of damage from winter storms. Clearly, this is not an issue in New Zealand’s mild maritime climate. A more appropriate country to compare with could be the United Kingdom, which has a maritime climate not dissimilar to New Zealand’s, but with a stronger wind climate (Moore and Quine, 2000). In that country, Quine and Gardiner (2002) used the ForestGALES model to show that the possible climate change scenarios suggested could result in substantial increases in the risk of damage. Based on these predictions, Quine and Gardiner (2002) did not recommend any change in forest management practices as these are already heavily focussed on trying to minimise the impacts of wind. The approach used by Quine and Gardiner (2002) is used here to illustrate the application of a mechanistic approach to investigate possible impacts of climate change on the risk of wind damage to forests in New Zealand.

In this analysis two stands are considered, one growing in the central North Island and the other growing in Canterbury. The silvicultural regimes employed at each site (Table 3.2) are typical of those regimes used in forests located in these areas. The stand modelled in the central North Island is the same as that used in Moore and Quine (2000), while the stand modelled in Canterbury follows the regime described by Studholme (1995).

Table 3.2. Silvicultural regimes used for radiata pine stands growing at the two different locations.

Operation	Location	
	Central North Island	Canterbury
Site index	28	20
Initial stocking	833 stems/ha	1250 stems/ha
First thin	MCH 7 m to 700 stems/ha	MCH 7 m to 600 stems/ha
Second thin	MCH 14 m to 317 stems/ha	MCH 11 m to 550 stems/ha
First prune	MCH 6 m to 2.5 m	-
Second prune	MCH 8.5 m to 4.5 m	-
Third prune	MCH 11 m to 6.5 m	-
Rotation length	28 years	32 years
Top height at time of felling	36 m	30 m

These scenarios were run using a version of ForestGALES implemented within StandPak. The stand in the central North Island was assumed to be growing on yellow brown pumice soils, while the one in Canterbury was assumed to be growing on a yellow-brown stony soil (Moore, 2000). The mode and dispersion parameters from the distribution of annual maximum hourly wind speeds were 15.263 m/s and 1.845 for the central North Island (Rotorua Airport) and 17.875 m/s and 2.023 for Canterbury (Christchurch Airport). Results of these model runs show that the threshold or critical wind speed required to damage the stand initially increases within increasing stand age before decreasing with further increases in age (Fig. 3.4).

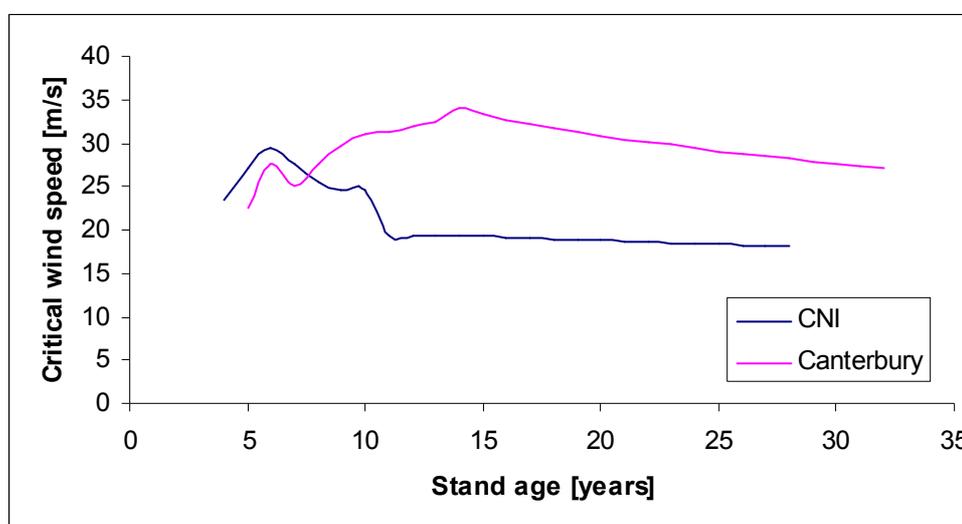


Figure 3.4. Changes in threshold wind speed for damage with stand age for the two simulated stands.

The Canterbury stand has a higher critical wind speed for damage than the central North Island stand due to the former having a higher stand density and lower tree height for a given stand age. Despite the stronger wind climate in Canterbury (Fig. 3.1), the stand located in this region is predicted to have a lower risk of damage than the central North Island stand (Fig. 3.5). In both scenarios, the risk of damage increases with increasing stand age, but this increase is more marked for the central North Island site.

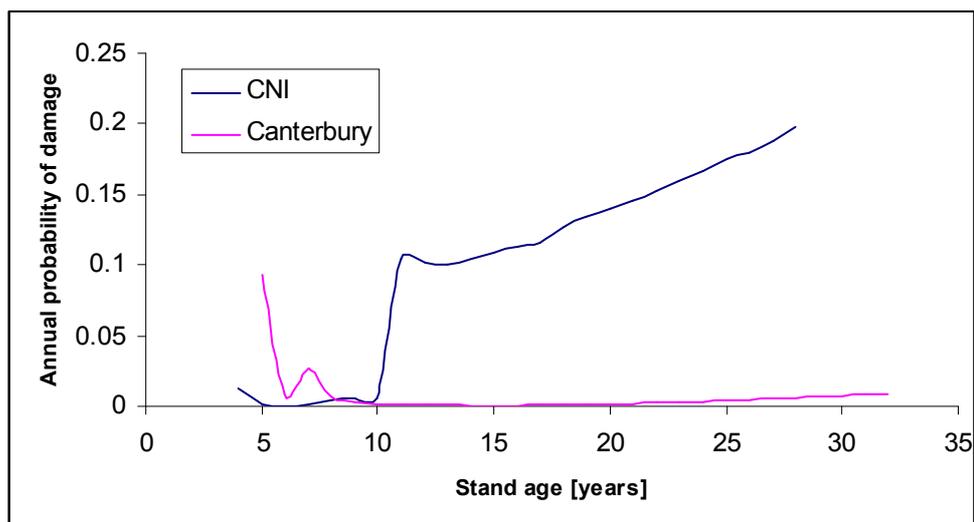
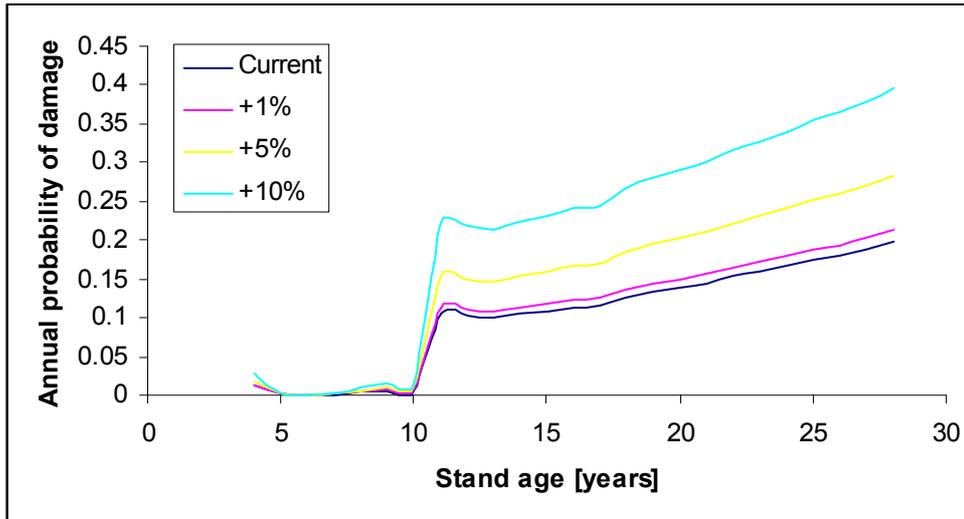
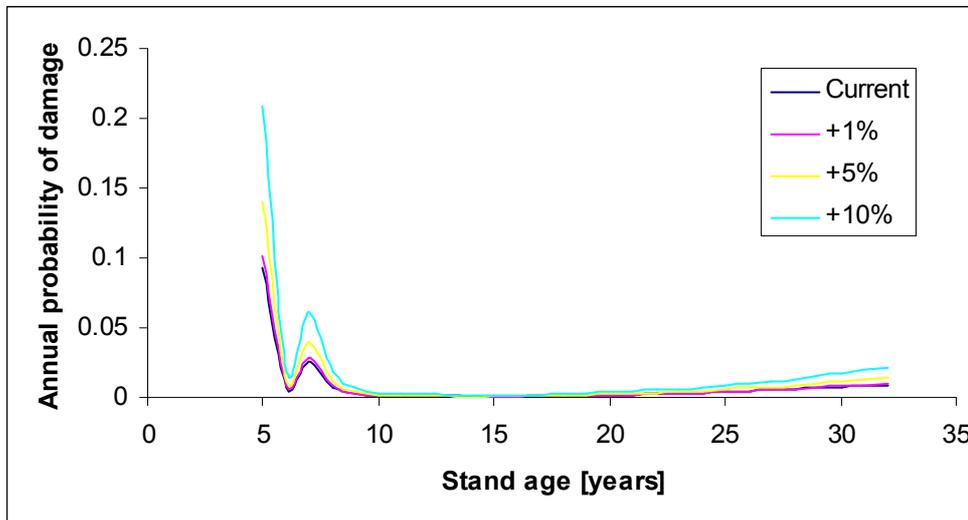


Figure 3.5. Changes in annual probability of damage for the threshold wind speed with stand age for the two case study sites.

The sensitivity to possible future climate change was modelled by increasing the mode of the distribution of extreme wind speeds by 1, 5 and 10%. The dispersion parameter could also be changed (increased) to represent a more variable strong wind climate, but that was not done in this example. For the central North Island stand, there was a large increase in the risk of damage with increasing strong wind climate (Fig. 3.6). For example, a 10% increase in the modal value resulted in a 100% increase in the annual probability of damage at age 28 years. Even for the 5% increase in modal value there was a 43% increase in the risk of damage. Given that the most recent spatially-based assessment of extreme wind speeds (Ministry for the Environment, 2008) indicates that extreme wind speeds in the central North Island could actually decrease by 3-5%, there could be a substantial decrease in the risk of damage. In contrast, at the Canterbury site there was only a small increase in the probability of damage (in absolute terms) with an increase in the strength of the wind climate (Fig. 3.6). This highlights the importance of the interaction between the vulnerability of a stand (as measured by the critical wind speed) and the wind climate in determining the impact of possible future climate change on the risk of wind damage. For stable stands with a high threshold wind speed for damage, small to moderate changes in the strong wind climate may only have a negligible impact on the risk of damage. However, for those stands which are more vulnerable, even relatively small changes in the wind climate could have a large impact on the risk of damage. If the wind climate becomes more variable (resulting in a higher value of the dispersion parameter), then the risk will increase further still.



(a)



(b)

Figure 3.6. Changes in annual probability of damage for the threshold wind speed with stand age for the (a) Central North Island and (b) Canterbury case study sites under different climate change scenarios.

3.1.1.6 Modelling approaches and future data requirements

The example contained in the previous section was intended to demonstrate an approach which could be used to assess the possible impacts of climate change on the risk of wind damage to New Zealand's forests. Because the vulnerability of forests varies with species, soil type, growth rates, stand structure and age the impact of possible future changes in the strong wind climate will not be the same for all forests. Furthermore, the risk could also be affected by changes in tree growth as a result of increased temperatures and drought. Increasing height growth would generally be expected to increase the risk of damage, while reductions in height growth will reduce the risk. Again, the degree to which the risk is changed will depend on the interaction between stand vulnerability and the strong wind climate.

In order to better understand the impacts of future climate change on the risk of wind damage, it is recommended that a mechanistic wind damage model should be used in combination with a physiologically-based tree growth model. The latter will allow

the secondary impacts of changes in climate on tree growth to be modelled, while the former can be used to test a range of scenarios.

The first step in better understanding the impacts of climate change on the risk of wind damage is to develop better predictions of the likely change in the wind climate. As noted in the report from the Ministry for the Environment (2008), substantial improvements in the predictions of the future wind climate have been made including better spatial representation of such changes. From these predictions, further predictions can be made about the future extreme wind climate using the relationships between the distributions of mean and extreme winds. Such an approach not only allows the mean wind climate to be altered, but also the variability of the wind climate. These new recurrence functions could then be applied to critical wind speed values for a range of stand types and ages in order to determine the changes in predicted risk. This would allow those combinations of stand structures and locations which exhibit the greatest change in risk to be identified.

It is important to remember that this approach is probabilistic and cannot make any predictions about individual storms, which are the mechanism by which damage actually occurs. Furthermore, the magnitude of damage will depend on the amount by which the actual winds speeds which occur during a storm exceed the threshold wind speed for damage. The greater the amount by which the threshold wind speed is exceeded, the greater the intensity of damage. Although it has limitations, in terms of being able to predict damage levels, such an approach can provide a useful framework for examining a complex phenomenon and evaluating the possible impacts of climate change in a structured and objective manner.

3.1.2 Fire

3.1.2.1 Fire risk in New Zealand

Major forest or rural wildfires can result in significant economic and property losses or environmental damage. They also pose a significant threat to the lives and safety of rural residents and the rural fire fighters that protect them. Large and devastating rural fires occur relatively infrequently in New Zealand when compared with Australia, North America and other parts of the world. However, damaging wildfires can occur in most parts of the country.

Wildfires occur when prolonged periods of hot, dry and/or windy conditions coincide with a build-up of flammable vegetation fuels and a source of ignition. Severe fire weather conditions can arise throughout New Zealand, and at any time of year due to our mild maritime climate. They are an annual occurrence in some regions, notably the eastern parts of both islands where Very High or Extreme forest fire danger can occur on more than 30 days/year (Pearce *et al.*, 2003a) (see Fig. 3.7). While other regions experience these weather conditions less frequently, severe fire weather occurs once every 3-5 years even in the lowest risk areas driven by interannual and longer term climatic patterns (Pearce *et al.* 2003a, 2007).

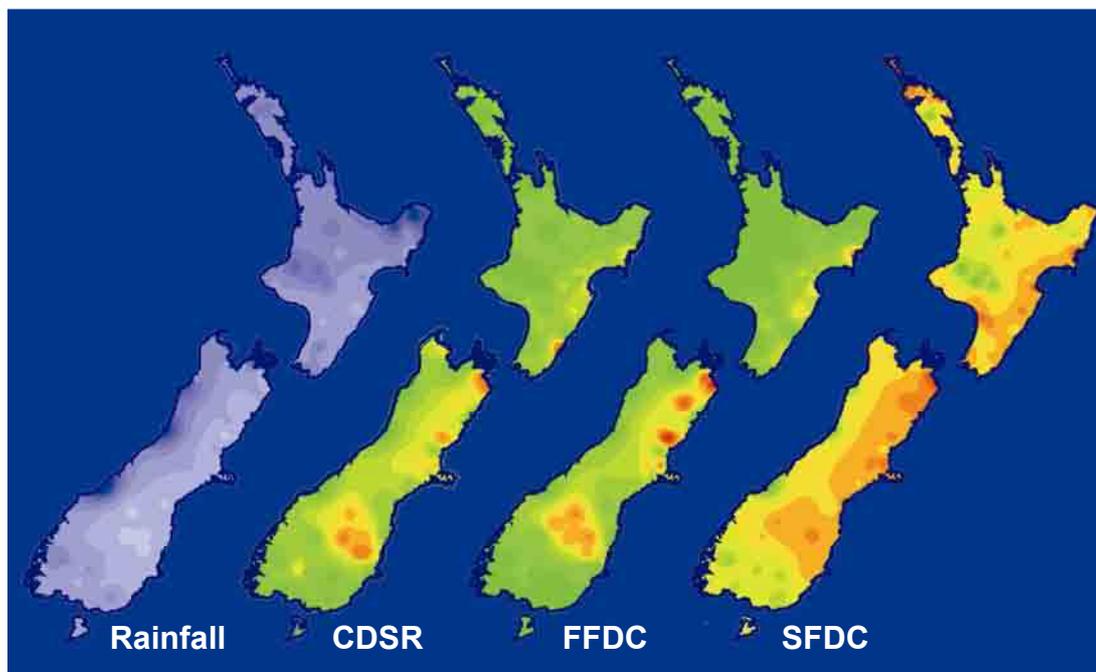


Figure 3.7. Comparisons of fire season severity across New Zealand, based on long-term average Cumulative Daily Severity Rating (CDSR) and the combined number of days of Very High and Extreme (VH+E) fire danger class for Forest (FFDC) and Scrubland (SFDC), where severity tends from high (red) to low (green) although individual scales differ. Mean annual rainfall is also included for comparison, tending from low (light blue) to high (dark blue). Maps were produced by interpolating point data for individual weather stations (using Inverse Distance Weighting), and do not include any topographic adjustment. (after Pearce *et al.*, 2003b).

In spite of considerable variability from year-to-year (Fig. 3.8) and incomplete statistics, data show that New Zealand now experiences around 3000 vegetation wildfires each year that burn around 6 500 ha of rural lands³. This includes grasslands, shrublands, indigenous and exotic (i.e., plantation) forests. Fires in native and exotic forests comprise only a relatively small proportion despite making up around 30% of New Zealand's land cover, and represent around 6% of the area burnt compared with scrublands (39% of the area burnt, from 17% of the land cover) and grasslands (54% of the area burnt, from 52% of the land cover)⁴. While the average area burned has remained relatively static, the number of vegetation fires reported has increased significantly, most likely as a result of improved reporting (e.g., through the '111' phone system and the number of incidents attended by the NZ Fire Service). However, the risks from fire have changed over time, with fire now less widely used for land clearing, and an increasing number of residential properties being developed in, or near, forested and other rural areas.

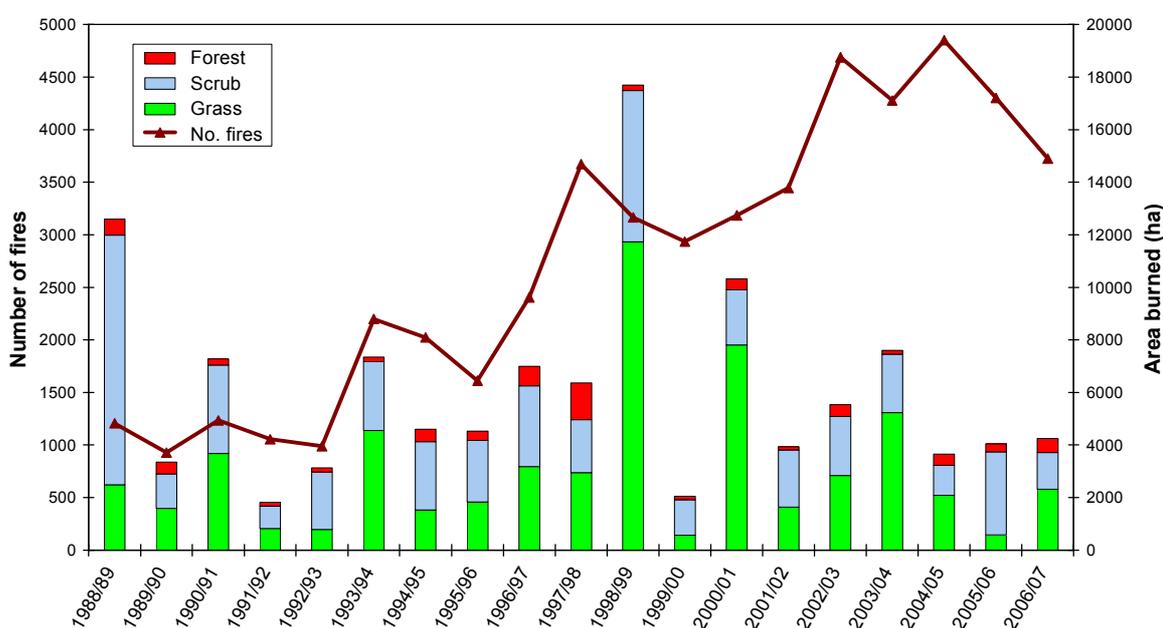


Figure 3.8. Number of fires and area burned by rural wildfires in New Zealand by vegetation type. (Source: National Rural Fire Authority Annual Return of Fires, 1988/89 – 2006/07).

Most rural fires in New Zealand are caused by people, and less than 5% of fires are started by natural causes (such as lightning strikes, volcanic activity and spontaneous combustion). In particular, escapes from land clearing burns (both permitted and unauthorised) and a steadily increasing number of malicious fires are major causes. The principal causes of fires in plantation forests are arson, escaped burns, vehicles and campfires, although forestry operations and spontaneous combustion can result in significant fires. Outside of forests, vehicles and railways are also significant fire causes. In fact, fires originating outside of forests on adjacent land present considerable (and, in many cases, the predominant) risk to production forests, with up to twice as many fires and six times the area burnt reported by forest managers on land bordering forest areas as within forests (Cameron *et al.*, 2007).

³ Based on statistics derived from the National Rural Fire Authority's Annual Return of Fires from 1991/92 – 2006/07.

⁴ Data on New Zealand land cover from Newsome (1987) and for forest cover from NZFOA (2008).

3.1.2.2 Effects of fire on forest and rural lands under current climate

Potential damage to or loss of plantations from fire is a major risk facing forest owners and, due to the value of the asset being protected, fire has been an important consideration for New Zealand forestry since plantations were first established in the mid-1890s. Despite being responsible for only 7% of the total land area, forest owners currently spend in the order of \$8-10M per annum on fire protection⁵, with a further \$22-25M per year spent by Local Government, Department of Conservation, Defence and the National Rural Fire Authority (Cameron *et al.*, 2007).

During the last 60-70 years, fires have resulted in over 40 000 ha of exotic plantation forest being burned (with a conservative value in excess of \$300M), at an annual loss of around 570 ha (0.12% of the planted estate) (Cameron *et al.*, 2007). The average annual loss over the past decade (to 2007) is somewhat lower (about 380 ha/annum or 0.02% of the plantation estate) (Fig. 3.9), largely due to improvements in fire management and a reduction in forestry prescribed burning (and associated risk of escapes). *Pinus radiata*, which makes up the majority of New Zealand's plantation forests, has very low tolerance to fire. Any heating of the cambium through the relatively thin bark or of roots through shallow soil or burning duff, or damage to live foliage from crown fire or heat scorch, is normally sufficient to kill trees. Other plantation species such as Douglas-fir and eucalypts are more fire-adapted (e.g., thicker bark, self-pruning), and will survive all but the most severe fires.

Less is known on the risk to New Zealand's native forests as historic fire statistics do not typically distinguish between exotic and indigenous forest losses, and this continues to be the case in modern fire reporting systems. However, it is a generally accepted view that in most circumstances native beech and podocarp forests are less flammable than their introduced counterparts. This is not to say that they will not burn, and major fires have occurred in the past and more recently, in both beech and podocarp/broadleaf forests, under drought conditions. The majority of New Zealand's native plant species, and forest species in particular, are not fire-resistant or fire-adapted and are severely impacted or even killed by fire. They do not tolerate heating (of stems or roots, etc.) due to thin bark, and are generally not prompted to resprout following fire or to reproduce seed by heating or fire. Most native species require undisturbed conditions over decades or centuries to establish suitable habitats for their re-establishment (e.g., species associated with mature kauri forest or dryland woodlands).

⁵ The majority of this is spent on fire administration, prevention and preparedness as opposed to firefighting and damages. While this amount includes suppression costs for the majority of minor fire events, it excludes firefighting costs and damages for major fires (>200 ha) of which there have been several in New Zealand in the last few seasons where suppression costs and damages both exceeded \$1M in each case. Forest owners also contribute a huge amount to the rural fire sector in terms of manpower and equipment, providing the equivalent of some 60 full-time positions in fire control and more than 1600 trained staff, contractors and volunteers, as well as more than 130 firefighting vehicles.

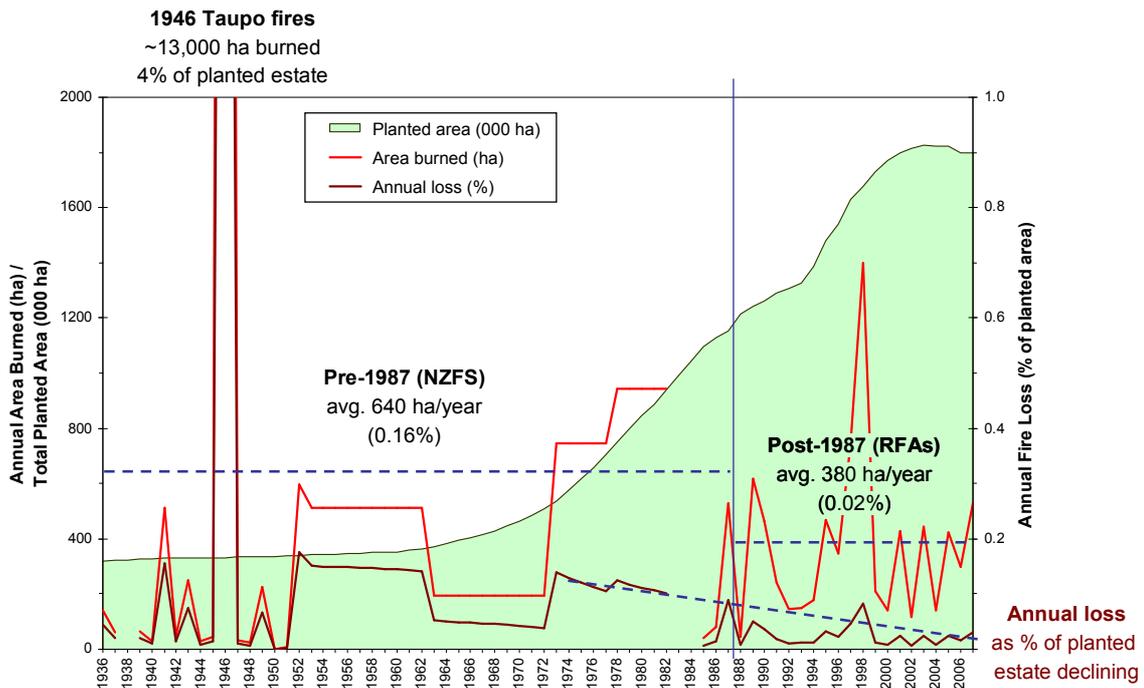


Figure 3.9. New Zealand plantation fires losses, expressed as area burned (ha) and percentage (%) of the planted estate (000's ha). Losses during the NZ Forest Service era (pre-1987) are contrasted with the post-1987 period where fire protection is provided by Rural Fire Authorities (RFAs) (after Cameron *et al.*, 2007).

The physical and economic effects of a rural fire depend on the fire intensity and area extent, as well as other factors such as the land-use or ecosystems involved. The most obvious costs are those associated with property damage, fire suppression, loss of income, and recovery and re-establishment, although these may be offset in some cases through insurance or salvage. However, fires can also have indirect costs that may be even more significant than these direct losses. These can include effects on infrastructure (road, rail or powerline damage or closures), tourism and recreation (aesthetics and access), environmental impacts (erosion, water supply contamination, nutrient loss) and effects on human health (smoke pollution, mental and emotional effects, post-traumatic stress).

Wildfires in New Zealand can therefore generally be characterised as high frequency events with low to medium severity. However, there are also periodic “catastrophic” wildfires that occur less frequently but have much more devastating consequences. The 1945/46 fire season is the worst on record, when over 238 900 ha of indigenous forest, exotic plantations, tussock and scrub were burnt in the north, east and central regions of the North Island, including the 1946 Taupo/Tahorakuri forest fire which burnt 30 700 ha; this fire represents the largest pine plantation wildfire (13 100 ha) to have occurred in New Zealand (Pearce and Alexander 1994). Canterbury also experienced major plantation fires in 1955, 1973, 1976 and 1988 (Pearce *et al.*, 2000).

One estimate of direct and indirect costs associated with another single catastrophic fire event of similar magnitude to the 1946 Tahorakuri plantation fire (Richardson *et al.*, 2003) suggested that as well as the hundreds of millions of dollars lost in terms of crop value (~\$900M) and firefighting costs (up to \$50M), the consequences for the local, regional, and national economies would be enormous. There would be in the order of 1500-2000 direct job losses, the likelihood of mill closures, a huge reduction

in export dollars, and all of the consequent effects on support industries and affected communities. It was estimated that it would take between 20 and 30 years to fully recover from such a disaster, depending on the age classes of forest destroyed.

Not all major fires have involved pine plantations, and the 1982/83 fire season saw a total of 45 000 ha burnt, including 15 000 ha of tussock and beech forest in the Ohinewairua Fire in the central North Island. More recently, the 1998/99 fire season resulted in 18 000 ha burnt, with over half this area from two major grassfires near Alexandra in Central Otago. The 2000/01 fire season resulted in 13 000 ha burnt, with 7000 ha from two major grassfires in Marlborough. The 2003/04 fire season saw four significant fires occur in Canterbury over a 3-week period involving a range of fuel types in remote high country, rural farming areas, and on 'lifestyle blocks' in the rural-urban interface.

3.1.2.3 Current and future issues regarding fire risk

With destruction of plantation forests being one of the major economic impacts of rural wildfire, it is significant that the plantation forest estate is predicted to increase to an area of about 4 million ha by 2025. There is also a trend towards less silviculture (pruning, thinning), leading to an increasing likelihood of crown fires under severe conditions. These factors increase the wildfire hazard and the potential for major economic impacts through catastrophic-scale forest fires in future.

The planted forest estate is also a considerable asset to New Zealand in terms of carbon credits under the Kyoto Protocol. Some areas of native scrub, specifically manuka/kanuka scrub established since 1990, also represent carbon credits for New Zealand's Kyoto commitments. Obviously, fire poses a significant risk to these carbon assets, and this risk will increase with climate change. Richardson *et al.* (2003) estimated that, assuming a carbon value of \$60/tonne of carbon and the national average of 400 ha of plantation forest burned annually, this represents a loss of approximately \$1.2M per year of carbon, while a catastrophic forest fire scenario could result in losses of \$140M in carbon. Losses through scrub fires in manuka/kanuka could represent a further \$0.7-1.0M per year in carbon.

New Zealand is also undergoing significant changes in land use that in many cases are resulting in more complex fuel types with higher fuel loads, more continuous fuels and increased commercial and conservation values. These changes include pasture retirement and associated reversion to tussock or scrublands (through land tenure review), invasion and spread of wilding pines, and increased planting of arable and fodder crops. Further land use change associated with carbon storage or biofuel production could also increase fuel loads and flammability that might further exacerbate fire risk in the future even without climate change. Increased fire risk could result in greater area burned and increased fire suppression costs, while higher asset and conservation values would likely mean higher fire protection costs due to an increased need for equipment, fire insurance and fuels management.

Another factor contributing to increased future risks from wildfire is the expansion of the "rural-urban interface", i.e., areas where buildings and other human development, mainly residential, adjoin or are interspersed with undeveloped rural lands containing flammable vegetative fuels. The development of communities along forest boundaries leads to a higher chance of ignition, and increased risk to forests and built assets. There is also an increased risk to the people living and working in the rural/urban interface, and to fire fighters charged with protecting life and property in these areas.

Fire continues to be widely used in New Zealand as a land management tool, to clear scrub weeds and remove tree trimmings or other heaped vegetation, to reduce logging residues and aid forestry replanting, to burn off stubble to remove waste residues and sterilise the soil. In tussock grasslands, fire is used to reduce woody weed encroachment, improve stock access and promote new pasture growth. Unlike Australia or North America, fire is currently not widely used for fuel reduction or for conservation habitat management. However, fire use has and will continue to play a major role in shaping the vegetation cover of New Zealand that we see today and in the future.

3.1.2.4 Effect of climate change on fire

Wildfire occurrence and climate are intimately linked, at daily (e.g., prevailing fire weather conditions), seasonal (e.g., drought, El Niño-Southern Oscillation events) and much longer time scales (e.g., decadal oscillations, climate change). From as early as the late 1980s, postulated increases in global temperatures associated with climate change have been related to possible increases in fire weather severity and fire danger in many parts of the world. Future climate change has been predicted to result in altered fire frequency and severity (Overpeck *et al.*, 1990; Stocks, 1993), including increased fire activity as a result of both natural (i.e., lightning) (Fosberg *et al.*, 1990; Price and Rind, 1994; Goldammer and Price, 1998) and anthropogenic (Wotton *et al.*, 2003) causes, and a greater number of escaped fires (Torn and Fried, 1992; Fried *et al.*, 2004). It is also expected to result in increases in area burned (Flannigan and Van Wagner, 1991; Flannigan *et al.*, 2005), longer fire seasons (Street, 1989; Wotton and Flannigan, 1993), and higher suppression and fire management costs (De Groot *et al.*, 2003).

The growing body of international evidence therefore suggests that future fire activity is likely to increase as a result of global warming and associated climate change (IPCC, 2007c). In many parts of the world, the warmer, drier and windier conditions associated with climate change are predicted to result in an increase in the number of fires and decreased fire return intervals. These fires are likely to spread faster and burn more intensely resulting in reduced initial-attack suppression effectiveness, a greater number of escaped fires and increased area burned. In addition to increased suppression costs and economic damages, changes in fire severity of this magnitude would have widespread impacts on vegetation distribution, forest condition, and carbon storage, and greatly increase the risk to property, natural resources and human life. However, it is also important to note that fire weather severity and the associated fire impacts may undergo little or no change, or even decrease in some areas (e.g., Flannigan *et al.*, 1998, 2000), due to the significant regional variability in predicted climate changes that in some areas include increased rainfall amounts and frequency.

Increased fire risk in many parts of New Zealand is a potential impact of future climate change (Pearce *et al.*, 2005), associated with increased temperatures and wind speeds, and lower rainfall (and humidity). The increasing frequency of extreme weather events due to long-term climate variability and climate change, is further adding to the increased risk from wildfire and the majority of experts believe this trend will continue into the future. Indications of possible future fire activity and increased suppression and management requirements associated with climate change enables New Zealand rural fire authorities to make more informed fire management decisions on fire prevention and preparedness activities now and into the future.

3.1.2.5 Modelling approaches used to determine climate change effects on fire risk

Several approaches have been utilised to determine the potential effects of climate change on fire risk. Links between historic climate and fire history identified from fire scars in tree rings (dendrochronology) and/or pollen and charcoal records from soil and lake sediments (palynology) have been used as the basis for predicting wildfire activity under future climate (Clark, 1990; Sweetnam, 1993; Flannigan *et al.*, 2001). Landscape-scale fire models where fire ignition and spread is modelled explicitly (e.g., Cary, 2002), and dynamic vegetation models that include a fire component in the model (e.g., Lenihan *et al.*, 1998; Thonicke *et al.*, 2001), have also been used to estimate future fire regimes, vegetation succession and fire impacts such as area burned with climate change⁶.

However, most studies have addressed how fire weather and associated fire season severity will change with changing climate. These studies typically use weather scenarios obtained directly from Global Circulation Models (GCM) or downscaled to the region of interest using statistical downscaling methods or, more recently, Regional Climate Model (RCM) output. Specific approaches include (after Hennessey *et al.*, 2005):

- i) applying changes in monthly average weather variables from a climate model to observed daily weather data, which are then used to recalculate fire danger under future climate;
- ii) determining changes in daily weather variability from a climate model, then applying these changes to observed daily weather data for use in calculating future fire danger; or
- iii) generating daily weather data from a climate change model for present and enhanced greenhouse conditions, then using these data as input to calculate and compare fire danger.

A key issue in the choice of methodology is the issue of future climate variability. Option i) has the advantage of avoiding biases that exist in the simulation of baseline conditions, but the disadvantages lie in assuming that existing relationships between weather variables will be maintained in future, and that there will be no change in daily weather variability. Another disadvantage of this option is the limited availability of sites with daily records of the four key weather variables (i.e., temperature, rainfall, relative humidity and wind-speed) required for fire danger computations.

While option ii) has the advantage of capturing changes in relationships between weather variables and changes in daily weather variability, it has the disadvantage of being biased by errors in the simulated baseline climate, i.e. the simulation of present climate may be too warm/cold or wet/dry on average. Cary (2002), for example, found that some errors in the simulated baseline climate were as large as the changes in climate projected due to a doubling of carbon dioxide. While corrections can be applied, residual errors remain, especially for extreme fire danger events. Use of option iii) avoids biases in the simulated baseline climate, as well as the disadvantages of the assumptions that existing relationships between weather variables will be same in future and that there will be no change in daily weather variability. Hennessey *et al.* (2005) state that including changes in daily weather variability and the behaviour of extreme events are important for determining fire-weather risk, and theirs is one of the few studies to have used this latter methodology to determine changes in fire danger with climate change.

⁶ For more detailed descriptions of landscape and vegetation fire simulation models, see Keane *et al.* (2004) and Cary *et al.* (2006).

The majority of the studies that have employed these approaches have looked at changes in fire danger ratings, fire danger class frequency and fire season severity or length using components of fire danger rating systems (such as the Fire Weather Index (FWI) or McArthur systems). However, Flannigan *et al.* (2005) used historical relationships between observed area burned and the associated weather and fire weather indices (Harrington *et al.*, 1983; Flannigan and Harrington, 1988) to estimate future area burned. De Groot *et al.* (2003) used relationships between fire weather indices, area burned and current fire suppression costs to project possible suppression costs under climate change. Flannigan *et al.* (2005) cautioned however that there is the potential problem of extrapolation of relationships beyond the range of observed values, and noted that for the future they planned to use dynamic models of climate and vegetation to estimate future fire activity and impacts.

3.1.2.6 Climate change effects on fire risk in New Zealand

Only one quantitative study has been carried out on the potential effects of climate change on fire risk for New Zealand (Pearce *et al.*, 2005). This research, carried out by Scion in conjunction with NIWA during 2004/05 for the NZ Fire Service Commission, sought to determine the potential effect of climate change on future fire risk in New Zealand. It applied regional climate change scenarios for the 2080s to long-term daily weather records for individual stations.

Two GCMs, CSIRO and Hadley, with contrasting spatial patterns of climate change across New Zealand, were used to investigate the effects on fire danger using the approach outlined in option i) above. GCM model outputs were “downscaled” to the New Zealand region using a statistical technique developed by NIWA (Mullan *et al.*, 2001b). Outputs were adapted to weather station locations from the National Rural Fire Authority’s fire weather monitoring network using a high-resolution grid over New Zealand to provide mean monthly offsets for temperature and rainfall that were used to recreate daily fire weather and fire danger records for 52 (of ~170) weather stations. High, low, and mid-range scenarios of climate change were generated for each model in an effort to cover the range of possible future climate outcomes. Summary statistics of weather inputs, FWI System components and fire danger class frequencies for each station for the range of scenarios were then compared against those for current fire climate.

Results showed that fire danger is likely to rise significantly in most areas of New Zealand, particularly the east, and that the length of the fire season will probably increase. In addition to changes in FWI System values, significantly higher fire season severity ratings and more days of very High and Extreme (VH+E) fire danger were predicted for stations in the east of both islands, the Bay of Plenty and central (Wellington/Nelson) regions under both the Hadley and CSIRO high and mid-range scenarios (see Fig. 3.10). In several cases (e.g., Gisborne, Napier and Christchurch), average seasonal severity rating values increased by 25-65%, and the total number of days of VH+E Forest fire danger by more than 20 days (>50%). Smaller, but still statistically significant, increases in seasonal severity ratings (15-25%) were found under the CSIRO high extreme scenario for stations in the west of both islands and south of the South Island (data not shown). Several stations (typically those in the south and west with low or no existing fire danger) showed little or no change in severity ratings or number of VH+E Forest fire danger days but, in one case (Tara Hills under the Hadley high extreme scenario), a very slight decrease in VH+E fire danger was found.

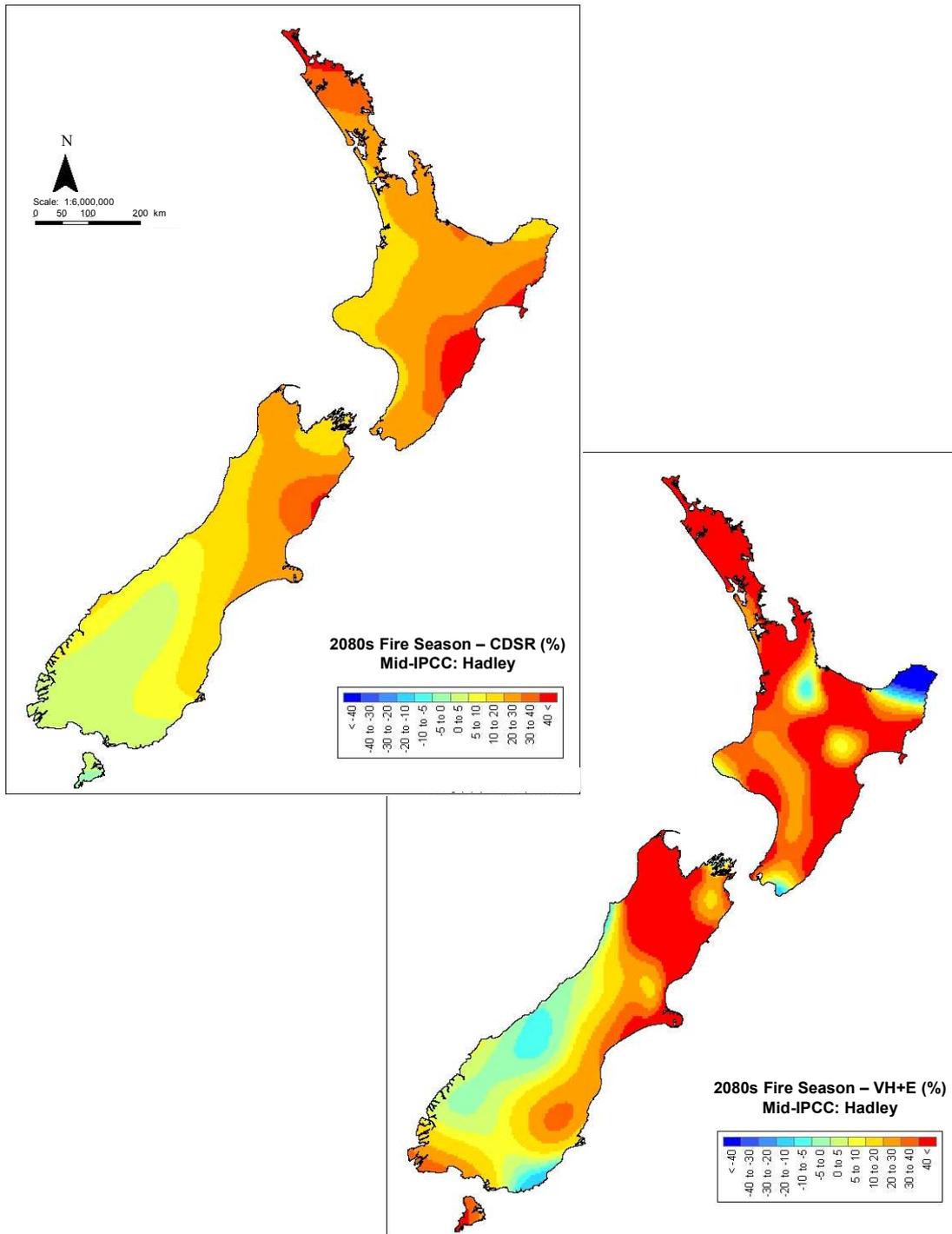


Figure 3.10. Changes in average Cumulative Daily Severity Rating (CDSR)/Seasonal Severity Rating (SSR) (%) (left), and the average number of days of Very High and Extreme (VH+E) Forest fire danger (%) (right) occurring over fire season months for the Hadley mid-range climate change scenario (from Pearce *et al.*, 2005).

Results from this study indicate that New Zealand is likely to experience more severe fire weather and fire danger under future fire climate, especially in eastern and central regions. Pearce *et al.* (2005) postulated that this would result in increased fire risk, including:

- longer fire seasons and increased drought frequency, and associated increases in fuel drying;
- easier ignition, and therefore a greater number of fires;
- drier and windier conditions, resulting in faster fire spread, greater areas burned, and increased fire suppression costs and damages;
- greater fuel availability and increased fire intensities, more prolonged mop-up, increased resource requirements and more difficult fire suppression; and
- increased frequency of thunderstorms and lightning.

They did note, however, that some of this risk might be offset by increased rainfall in some parts of the country (e.g., southern South Island).

3.1.2.7 Effects of fire on forestry with climate change

Fire can play a significant role in reducing the productivity of forest ecosystems, principally through the loss of nutrients contained in vegetation and upper soil layers to the atmosphere through volatilisation, in run-off or wind dispersion of ash (Harwood and Jackson, 1975; Raison, 1979; Dyck *et al.*, 1981). Intense fires can also result in loss of organic matter and microbial activity (Covington and Sackett, 1984; Goh and Phillips, 1991), and produce significant changes in soil structure leading to increased erosion (Leitch *et al.*, 1983; DeBano *et al.*, 1998). An increase in fire frequency and associated area burned with climate change could therefore result in reduction in forest soil productivity, and an increased need for fertiliser application for forest re-establishment following fire.

The focus on forestry and other vegetation for carbon credits and biofuel production is likely to increase into the future with climate change. As noted previously, fire poses a significant risk to these assets that is likely to increase with climate change. Increased fire risk should therefore be a key consideration in deciding where best to plant forests and other vegetation for carbon or biofuel production. On the other hand, biofuel production offers the prospect that plantations can be grown over shorter rotations, and the ability to use smaller-diameter material for biofuels creates an incentive to remove fuels from the forest floor and thus reduce fire risks.

Increased fire frequency with climate change could result in a need for greater areas of productive forest land to be set aside for fire protection works, such as firebreaks, waterpoints, or fuel breaks/buffer zones of less flammable (and less productive) species. This, combined with the likelihood of increased expenditures on fire prevention activities, provision of firefighting resources and fire suppression, could have a significant impact on the economics of forestry.

The effects of fire on forest productivity are also likely to be exacerbated through interactions with other factors that are likely to change as a result of climate change. There are obvious links between drought and fire risk, and increased drought risk with climate change (Mullan *et al.*, 2005) would result in drier and more available and continuous fuels, and therefore potentially more frequent and severe fires and larger areas burned.

Changes in the extent and distribution of weeds with climate change could lead to more flammable fuel types, especially woody scrub (e.g., gorse, broom) and grass weeds, that are more flammable and carry higher/more available fuel loads. These ignite more readily, burn more intensely and are harder to extinguish, resulting in

larger areas burned. Such changes are already being seen internationally (e.g., melaleuca (*Melaleuca quinquenervia*) in Florida, Gamba grass (*Andropogon gayanus*) in northern Australia) (Goldammer and Price, 1998), and in New Zealand with introduction of more fireprone Australian species (e.g., hakea and wattle in Northland). Increased fire frequency with climate change could also result in increased disturbance of forest soils and seedbeds, resulting in greater proliferation of early successional weeds species (e.g., gorse, broom, pampas) and, in turn, changes to more flammable fuel types, further exacerbating the problem.

There are also links between fire damage and insect attack, whereby fire-damaged forest areas can be favourable to some insect species (e.g., bark beetles; Bradbury, 1998; Suckling *et al.*, 2001). Increased fire risk with climate change could therefore lead to more widespread insect outbreaks and greater damage to forest areas adjacent to burnt areas. Conversely, insect attack can also lead to increased fire hazard by producing higher fuel loads and greater availability of dead fuels (e.g., spruce budworm and mountain pine beetle-killed forest in Canada). New insect invasions through climate change could therefore result in increased forest damage and fire-susceptibility.

An increase in the frequency of extreme climatic events with climate change, such as tropical cyclones and wind storms, could also result in greater wind damage and therefore increased fuel hazard in forest areas. More dead, downed woody fuels would potentially lead to higher risk of ignitions, more intense fires and greater areas burned.

3.1.2.8 Limitations of present modelling, and future modelling approaches

Like the majority of other New Zealand climate change studies undertaken at that time (e.g., Warrick *et al.*, 2001; Wratt *et al.*, 2004; Mullan *et al.*, 2005), the study by Pearce *et al.* (2005) on changes in fire danger used a statistical downscaling technique (Mullan *et al.*, 2001b) to downscale General Circulation Model (GCM) changes to provide the local detail required for impact studies. While this approach is a significant advancement over use of global model outputs, where a region such as New Zealand may be covered by only a very small number of GCM grid points, dynamic downscaling using a Regional Climate Model (RCM) nested within a GCM may provide more spatially accurate information on the influence of topography on local climate and fire danger, and recent international studies are increasingly using this approach (e.g., Wotton *et al.*, 1998; Flannigan *et al.*, 2001). This nested approach has the scientific advantage that it is more firmly based on atmospheric physics but requires substantially more computing power than statistical downscaling. Some work on regional modelling simulations using the nested RCM approach has been undertaken for New Zealand (e.g., Kidson and Thompson, 1997; Renwick *et al.*, 1997, 1999), and any further advances in this area should be considered in future studies of changes in fire danger with climate change.

The Pearce *et al.* (2005) study only considered the effects of changes in temperature and rainfall on future fire dangers as, at that time, possible changes in relative humidity and wind speed – the other key weather variables required to calculate fire danger – under future climate change scenarios were not well understood or could not readily be downscaled from GCM output. Relative humidity is a critical factor in fire danger rating, due to its influence on fuel moisture, ignition potential, rate of combustion and fire spread, and Beer *et al.* (1988) found it to be the most significant weather parameter affecting mean annual variations in forest fire danger. While changes in general wind flows with climate change had been modelled for the New Zealand context at the time of the Pearce *et al.* (2005) study, there was no way at

that time of inferring changes in scalar wind speed from modelled changes in these zonal wind flows. However, available indications (Wratt *et al.* 2004) suggested that the mean westerly wind component across New Zealand, for example, could increase by 60% or more with climate change, almost certainly leading to a general increase in fire dangers in the majority of model scenarios. Therefore, any future investigation of the effects of climate change should incorporate changes in wind speed and relative humidity, along with improved projections of temperature and rainfall changes, to provide more accurate of predictions of likely changes in fire danger.

The estimates of future fire climate contained in the Pearce *et al.* (2005) study were determined using offsets to current climatology in which only changes in monthly means of the underlying climatic elements were considered, as consistent projections for changes in daily and interannual variability were not available. While high, low, and mid-range scenarios of climate change were generated for each model in an effort to cover the range of possible future climate outcomes and, together with the time series of daily weather, to encompass likely daily and interannual variability in future fire climate, many experts expect climate variability to increase with climate change, so that a lack of consideration of variability will affect severity of predicted extremes (IPCC, 2001c; see Fig. 3.11). Use of a different approach, such as that employed by Hennessy *et al.* (2005), incorporating daily weather data generated from a climate change model for both present and enhanced greenhouse conditions, may overcome concerns regarding inadequate assessment of changes in extreme weather events caused by possible changes in daily weather variability.

Due to a requirement for continuous unbroken daily fire weather records, projected changes in fire danger with climate change in Pearce *et al.* (2005) were based on a relatively short historical record of available fire weather (i.e., the past 10-40 years), which was only available for a limited number of weather station locations. This historical baseline may be insufficient to capture long-term climate variability as described through decadal changes such as the Interdecadal Pacific Oscillation (IPO). This period captured just the one phase change of the IPO (in 1977 to IPO+, associated with stronger westerly winds and generally drier conditions in the east and wetter conditions in the west, than for the IPO- phase that existed earlier). Use of a longer baseline for present daily weather, potentially generated from a climate change model and validated against current station observations as in Hennessy *et al.* (2005), could encompass more long-term climate variability and enable data for a greater number of locations to be utilised.

Many international studies predict that the increased severity of future fire weather with climate change will result in increases in the number of fires and area burned, and greater fire suppression costs. These studies were only made possible by the availability of high quality data on current and historical fire occurrence, area burned, fire suppression costs and wildfire losses/damages. National fire reporting systems in New Zealand are currently inadequate to provide the quality of data necessary to carry out similar analyses, and urgent improvements to these systems are therefore required.

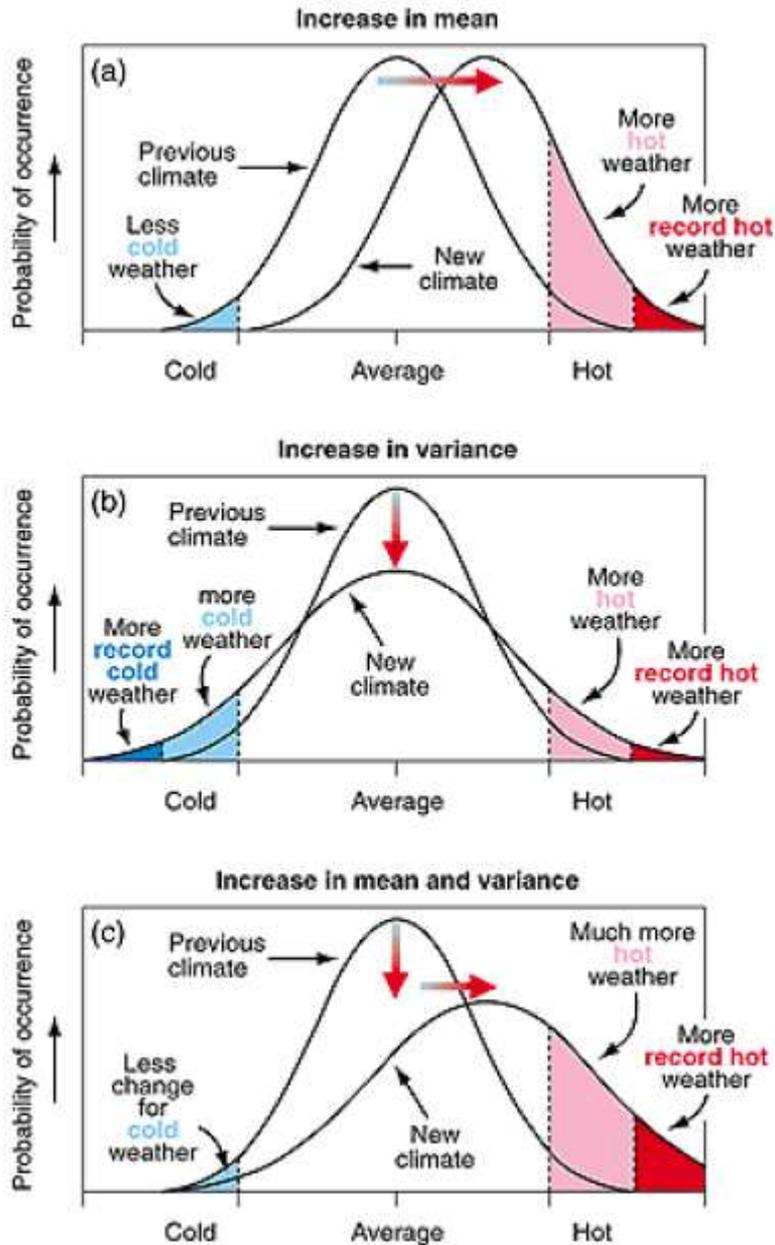


Figure 3.11. The effects on extreme temperatures when (a) the mean increases with no change in variance, (b) the variance increases with no change in the mean, and (c) when both the mean and variance increase, leading to more record hot weather. (from IPCC, 2001c).

Currently, climate change models do not typically consider the potential of wildfires to exacerbate the rate of global warming and climate change through increasing natural emissions of green gases to the atmosphere (Auclair and Carter, 1993; Amiro *et al.*, 2001a,b; De Groot *et al.*, 2007), changing vegetation cover and fire regimes (Weber and Flannigan, 1997; Cary, 2002), and reducing forest cover and global carbon storage (Kasischke *et al.*, 1995; Flannigan *et al.*, 1998, 2001). While the results of incorporating such links on the global scale circulation and climate may be almost invisible, effects at the regional scale could be much more significant.

Climate change will have direct effects on species distribution, migration, substitution and extinction, but it has been suggested (Overpeck *et al.*, 1990; Weber and Flannigan, 1997) that climate change impacts via changing fire regimes might be

more important than the direct climate change impacts on species. Fire can rapidly change a vegetation landscape that will fall more readily into a new equilibrium with climate whereas, in the absence of major disturbances, ecosystems are more likely to retain their characteristics through the inertia of internal biotic processes. Fire is a primary disturbance, and an improved understanding of the links between broad-scale fire severity and climate change will increase the ability for prediction of potential changes in ecosystem structure and function, as well as associated atmospheric feedbacks (McKenzie *et al.*, 1996). Models are available that begin to address fire disturbance and fire severity in relation to global vegetation dynamics and climate (e.g., Lenihan *et al.*, 1998; Thonicke *et al.*, 2001), and models will increasingly need to include these direct and indirect feedbacks from wildfire to more accurately predict future fire climate and associated impacts. In New Zealand, research is therefore needed to quantify the emissions of greenhouse gases from wildfires and, perhaps more importantly, from prescribed burning activities such as crop stubble and hill and high country scrub and tussock burning, for input into global and regional climate modelling.

3.2 Biotic influences

3.2.1 Framework for determining impacts

Given the complexity of species responses to the environment, and the numerous interactions involved between crop, weed and higher trophic levels, determining how climate change is likely to impact on the pest community and crop productivity is difficult task, which requires a hierarchy of analytical tools. An excellent review of the issues involved in determining how climate change may influence pests has been written by Sutherst *et al.* (2007a), from which the following section draws on many of the key points.

The effect of climate change on pests can be assessed through the concept of vulnerability, which is a measure of the potential impacts for a given change, taking into account the adaptive capacity to respond to the change, determined as

$$\text{Vulnerability} = \text{Impact} \times (1 - \text{Adaptive capacity}) \quad [3.2]$$

where adaptive capacity refers to the adaptability of management and depends on the adaptation technologies, cultural tools, and the infrastructure and resources available to implement appropriate technologies.

Impacts are determined as the product of a change in exposure of a system (in this case plantation forestry) to pests and the sensitivity of the system to that change, as,

$$\text{Impact} = \text{Exposure} \times \text{Sensitivity} \quad [3.3]$$

The following section outlines in more detail how climate change is likely to influence these terms.

3.2.1.1 Exposure

When a pest species shifts its geographical distribution, a new ecosystem might become exposed to the pest for the first time. Such changes in geographical distribution are likely to be induced by climate change as the suitability of different habitats for specific pest species will change with the changing climate. In New Zealand the threat from pests and diseases of warm-temperate and sub-tropical origin are likely to increase. Although there have been numerous interceptions and incursions of such pests, few have become established or become pests because our current climate is not suitable. Under warming scenarios such as those proposed by the IPCC, such species are much more likely to encounter suitable climatic conditions in New Zealand (see below). Climate change may also indirectly influence the risk of invasion if migrant species extend their range and move closer to New Zealand.

3.2.1.2 Sensitivity

In the context of plantation forestry, sensitivity describes how growth of trees is influenced by pests. For example, given the strong influence of weed type on tree growth, sensitivity of the plantation crop, is affected by how climate change influences exposure of a particular site to tall perennial weeds.

3.2.1.3 Impacts

The likely impact on the population, in the absence of intervention, can be derived once changes in the exposure and sensitivity of plantation forests under climate change have been defined. Determining these changes in exposure requires estimation of how climate change is likely to influence the distribution of pests. Once these changes in distribution have been determined the sensitivity of plantation growth to changes in pests needs to be established, to gauge the likely impact. Models which can be used to account for these changes in exposure and sensitivity are described in detail in section 3.2.2.

3.2.1.4 Adaptation and vulnerability

The IPCC has developed general guidelines for the design of adaptation measures to climate change and these are also applicable to other drivers of environmental change (IPCC, 2001b). Adaptation measures rely on application of innovative approaches or products to prevent or control pests in a changing environment, and can be classified as behavioural, engineering or administrative/legislative. Ideally, responses to changes in pest incidence and severity under global change should be proactive rather than reactive. Management responses should be based on a thorough understanding of the ecology of the target pest or community of species in an ecosystem at each location. The most likely pathway to developing such an understanding is to adopt a knowledge based approach to pest management with sophisticated decision support systems.

After accounting for the mitigating effects of adaptation on the impacts of pests the vulnerability of New Zealand plantations under climate change can be determined. Comparison to the current vulnerability will enable decision makers to assess how vulnerability is likely to shift under climate change.

3.2.2 Description of modelling approaches used to model climate change impacts

3.2.2.1 Determining changes in pest distribution (Exposure)

In order to assess the likely impact of climate change on the potential distribution and relative abundance of pests it is necessary to employ modelling tools. The modelling tools that are of most value are climate-based models. A comprehensive summary of these is given in Kriticos and Randall (2001). In brief, the method involves building a model of potential pest distribution using its historically observed distribution matched against climate data over that observed range. The climate dataset can then be changed to a future climate scenario and re-running the model will show the species' potentially suitable distribution under altered climatic conditions.

Future climate scenarios are novel climates as each of the climatic variables will change independently, resulting in complex change surfaces for temperature, vapour pressure and rainfall. The inferential models which are best suited to model the influence of these complex changes on species distribution are those that employ process-based mechanisms for describing how species respond to climatic variables such as air temperature and soil water balance (Kriticos *et al.*, 2006). The model which has been most widely used in this application is CLIMEX. CLIMEX can be used to model the current and future distribution of both plants and cold-blooded animals.

CLIMEX is a dynamic model (Sutherst and Maywald, 1985; Sutherst *et al.*, 1995, 2000) that integrates the weekly responses of a species to climate into a series of annual indices. The CLIMEX annual growth index (GI_A) describes the potential for population growth of the species as a function of soil moisture and temperature during favourable conditions, and up to eight stress indices (cold, wet, hot, dry, cold dry, hot dry, cold wet, hot wet) to determine the probability that the pest can survive unfavourable conditions. CLIMEX also includes a mechanism for defining the minimum amount of thermal accumulation (number of degree days) during the growing season that is necessary for population persistence. In CLIMEX, the growth and stress indices are calculated weekly, and are combined into an overall annual index of climatic suitability, the Ecoclimatic Index (EI), which ranges from 0 for locations at which the species is not able to persist to 100 for locations which are optimal for the species. Parameters for the growth and stress functions in CLIMEX are usually based on values which fit the geographical distribution of the species in its native range, and, if possible, through selected exotic ranges. Phenological observations, and relevant laboratory-based biological information can also be used to determine parameter values.

As CLIMEX is based on process-based algorithms, it can project species' potential distributions in novel climates with more confidence than regression-based models (Kriticos and Randall, 2001). Projections of the influence of climate change on future distributions are derived from future global meteorological datasets based on the standard IPCC emission scenarios (SRES). The Climate Research Unit (CRU), at the University of East Anglia, distributes a dataset of 20 future climate scenarios (five models by four emission scenarios), which are based on possible future emissions of greenhouse gases and their likely range of effects on the global climate. Predictions of future species distribution usually utilise at least two contrasting datasets, to gauge the maximum likely range in changes in potential distribution.

CLIMEX can also be used to investigate the origin of future biosecurity threats through utilisation of the region-matching algorithm. This procedure allows climate at a home location (e.g. New Zealand) to be matched with locations in the rest of world to determine where similar climates exist that may harbour future threats. By utilising the climate change databases described above, these matches can be undertaken for future climatic changes scenarios as well to determine where future threats are likely to originate from.

3.2.2.2 Modelling weed and insect population dynamics

As weeds should not be considered in isolation from potentially damaging biocontrol agents, process-based population dynamic models currently offer the best means of modelling the system. These models are also useful for investigating interactions between insect and crop species. Population dynamic models describe the key life stages of insects and weeds, how each life stage responds to environmental conditions and how they interact with each other. Given the strong linkages with actual processes and environmental variables these models can be used to explore how management intervention and climate change influence the population dynamics of both weeds and insects. This type of model, was developed for the invasive alien species *Acacia nilotica* in Australia, and used to explore how climate change and management influences population dynamics and growth of the species (Kriticos *et al.*, 2003b).

Currently, the main limitation to the development of population dynamic models lies not in the framework but in the parameterisation of the models. There is considerable uncertainty as to how changes in water use efficiency and CO₂ fertilisation are likely

to influence weed growth. As these responses are likely to vary widely between weed species, considerable work is required to accurately determine how particular weed species will respond.

Another substantial area of uncertainty relates to herbivore response to increases in C:N ratio of plant foliage, induced by increases in carbon dioxide concentration ($[\text{CO}_2]$). It has been shown that insect growth and fecundity are strongly related to the amount of nitrogen consumed during feeding. Reduced nitrogen concentration in plant material can then lead to two possible, and very different, outcomes. Herbivores may either consume more foliage to satisfy their nitrogen requirements or they may be unable to meet their nitrogen requirements so that the total herbivore population will decline. These two possibilities obviously have very different overall outcomes for ecosystem processes and functions.

3.2.2.3. Influence of weeds on tree growth (Sensitivity)

There is a need to develop accurate tree-growth models that are sensitive to competition from weed species to investigate how changes in weed growth will influence tree growth. Most models developed to date have been empirical (Mason and Whyte, 1997; Schnieder *et al.*, 1998; Knowe *et al.*, 2005). Although these empirical models are simple to parameterise and provide reasonable estimates of weed growth at a regional level, the empiricism implicit within them restricts their general applicability and reduces the confidence with which they can be applied to conditions outside the narrow range of conditions for which they were developed and parameterised. As neither tree nor weed growth or competition functions are mechanistically linked to environmental variables, these types of models are unsuitable for investigating the effect of climate change on weed competition and the growth of trees.

Development of more mechanistic and generally-applicable growth models requires specific knowledge of the mechanisms of interaction between trees and competitors. Numerous studies have substantiated the hypothesis (Goldberg, 1996) that weeds reduce tree growth through competing for site resources. Growth and succession of non-crop vegetation in relation to crop tree development has been found to have a direct bearing on the level of competition for specific resources (Richardson, 1993).

Models describing the interactions between weeds and trees require functions that quantify the effect of non-crop vegetation on site resources and how changes in these site resources influence tree growth. One simple but effective method to quantify weed competition for light is the use of competition indices, which characterise the degree to which growing space around a focal tree is occupied by neighbouring plants (Burton, 1993). Numerous studies have demonstrated that competition indices, which incorporate measurements of relative weed:tree height and percentage weed cover are closely correlated to tree growth (see Richardson *et al.*, 2006 for review). More recently it has been shown that tree growth is correlated to weed competition for water, as quantified by weed-induced changes in root-zone water storage (Watt *et al.*, 2007). To ensure that these competition modifiers are dynamic, the modelling framework needs to have growth models for both weed and tree species. As different seasonal growth patterns for weeds and trees can substantially influence the competition between them (Kimberley and Richardson, 2004), growth models need to operate at a sufficiently fine timescale to capture these changing interactions.

The effect of herbicides on tree growth can be quantified by linking these growth and competition models with dose-response models which describe the effects of

herbicide application on weeds. Combining these elements provides the necessary framework to determine how herbicide intensity and time of application influences weed growth and mortality, and how weed competition, through alteration of competition indices, affects tree growth over both the short and long term. Ideally, such a modelling framework should also include a weed succession module that describes the time delay for re-establishment of weeds, and subsequent rates of weed growth, following weed mortality from herbicide application.

Functions describing the relationships in the above three paragraphs form the basis of the decision support system Vegetation Manager (VMAN), which is fully described in Watt *et al.* (submitted c). VMAN links the modules described above through to an economic analysis, which can be used to determine how different levels of weed competition influence tree growth and crop value.

Although most aspects of VMAN, including the crucial competition functions, are well developed, a current limitation of the model is that weed growth is determined using sub-models which are empirical in nature. However, this could be readily addressed by loose coupling VMAN with population dynamic models. Using this approach, the output from population dynamic models, based on the effect of climate change on weed growth, could be used as input to VMAN, to subsequently describe how weed growth influences tree growth under climate change.

3.2.3 Impacts of climate change on pests. General observations

The key environmental factors associated with climate change that are likely to affect pest distribution and growth include increases in carbon dioxide concentration and air temperature, and changes in precipitation and root-zone water storage. There are also likely to be changes in precipitation which may lead to an increase in the frequency of extreme events, such as flooding and drought. Increasing temperature leads to earlier spring warming and delays autumn cooling which is likely to affect growth, as this will extend the growing season. Advances in spring events have already been documented on all but one continent (Parmesan, 2006), with a mean advancement by 2.3 days per decade, over time periods ranging from 16-132 years (Parmesan and Yohe, 2003).

One of the most important changes likely to result from climate change is a shift in suitable habitats for pest species, which is mainly caused by changing temperatures. Temperature influences thresholds for pest growth and survival through events such as frost frequency and the requirement for reproduction as determined through thermal accumulation. For example, these changes in thermal accumulation are likely to be very important for currently occurring 'sleeper' weeds, which are weeds that are present in New Zealand, but whose distribution or vigour is limited under current climatic conditions. If introduced beyond their thermal accumulation limits, perennial weeds can sometimes persist as 'sleeper' populations in marginally cold climates where they frequently fail to reproduce. If the heat sum required for reproduction is exceeded for these 'sleeper' weeds these populations may form foci for a potentially rapid increase in abundance.

A benchmark for the effects of temperature on changes in distribution is provided by the relationship of temperature with altitude and latitude (Linacre, 1992). Each 1°C increase in global temperature corresponds to a potential increase of ~ 170 m in a species' altitudinal distribution. As warming over the past century has been about 0.6°C (IPCC, 2001a), altitudinal range changes caused by global warming are unlikely to exceed 100 m. Similarly, in the absence of other factors, latitudinal shifts

of about 160 km can be expected for each 1.0°C increase in temperature corresponding to a shift of ~100 km with the observed temperature increase of 0.6°C. Recent research investigating biological trends in range boundaries has confirmed these shifts have occurred. Global meta-analyses document significant range-boundary changes for 279 species, which on average have shifted poleward by 6.1 km per decade over an average timespan of 66 years (Parmesan and Yohe, 2003). At the same time, these observed rates of shifts in species's distributions are less than the shift in climatic zones which indicates the delay in species' actual distributions following a change in climatic conditions.

Following entry of a pest species into a new region, such as New Zealand, the intrinsic attributes of a species and their response to a number of climatic and abiotic factors, determine the success of establishment in the new area (Baker *et al.*, 2003; FAO, 2004). As natural herbivores and competitors are generally left behind, the population of new invasions can rapidly increase unabated during the initial stages (Colautti *et al.*, 2004), although local natural enemies (i.e., herbivores, predators, parasitoids, pathogens) may eventually adapt to the invading species over time. Natural enemies from the pests' native region may also be introduced either inadvertently or as targeted biological control agents, and natural enemies already present can also 'colonise' an invader. This may reduce the competitive strength of pests over time, which was observed with spectacular results, for example, in the case of the biocontrol programme against prickly pear (*Opuntia* sp.) by cactoblastis caterpillars (*Cactoblastis cactorum*) in Australia (Culliney, 2005). A number of biological control agents have been introduced for pests in New Zealand, and some are providing successful control of their target pest (e.g., Fowler and Withers, 2006). Climate change could disturb the delicate balances between pests and their natural enemies, potentially worsening pest impacts. For example, Visser and Both (2005) documented seven cases where climate warming changed predator-prey interactions or insect-host plant interactions such that these interactions were weakened by reduced phenological synchrony.

Therefore, exposure of New Zealand plantations to different pests and different combinations of pests is very likely to occur under climate change. The combined effects of pests that are already present, those arriving in the future, and the disturbance of natural control mechanisms are most likely to worsen pest impacts in New Zealand's plantations. Although the outcomes of these direct and indirect effects are difficult to predict, negative impacts of climate change on forest weeds, insects, and diseases are generally expected. This is outlined in more detail in the respective sections below.

3.2.3.1 Effects on weeds

Changes in growth rate of weeds are likely to be determined by a number of climatic factors. Increases in [CO₂] are likely to result in greater weed growth rates, as described for six highly invasive US weed species (Ziska, 2003). Although it is possible that growth of invasive weeds will respond more strongly to increases in [CO₂] than other plants, there is currently not enough information to state that with certainty. However, as it is generally observed that some plants respond more strongly to increasing [CO₂] than others, it could be expected that at least some weed species may respond more strongly to increasing [CO₂] than the small number of economically important plantation tree species. Those more responsive weed species may thus become more invasive and dominant in the future, as speculated by Ziska (2003).

Growth will also be determined by changes in root-zone water storage and temperature and changing length of the growing season. Any reduction in growth which results from reduced root-zone water storage is likely to be offset to a limited extent by an increase in water use efficiency resulting from higher [CO₂]. The effect of changing temperature on weed growth will depend on how the temperature changes in relation to the weed optimum temperature for growth. As changes in [CO₂] have been found to increase the optimum temperature for many species, determining weed species response to increasing temperature could be complex. Lengthening of the growing season is likely to increase weed productivity.

In addition to temperature induced changes in range boundaries, local weed composition is likely to be determined by a number of interacting climatic factors. Changes in weed composition including the balance between C₃ and C₄ weeds, legumes and non-nitrogen fixing species, and woody plants vs. grasses is likely to occur on a wide scale (Farquhar, 1997; Gavazzi *et al.*, 2000). The balance between C₃ and C₄ weeds is likely to be strongly related to the interacting effects of carbon fertilisation and temperature. Increases in competitiveness of C₄ weeds due to warming (Tieszen *et al.*, 1979; Rundel, 1980; Ehleringer *et al.*, 1997) are likely to be more than offset by the reduced competitiveness which occur with increases in [CO₂] (Ehleringer *et al.*, 1997; Collatz *et al.*, 1998), which will result in C₃ plants being favoured over C₄ species under climate change (Kirschbaum, 2004). Despite the expected warming trend, it is therefore not anticipated that C₄ weeds will play a significant role in New Zealand in future.

3.2.3.2 Effects on insects

Insects are directly affected by changes in air temperature, broadly similar to plants. On the other hand, insect are generally *not directly* affected by changes in [CO₂], rainfall, vapour pressure deficit, and other climate variables that can directly influence plant growth and health. However, indirect effects of these variables can influence insects in various ways, as outlined below.

3.2.3.2.1 Effects of temperature

Most insect species occupy a relatively narrow geographic range that is partly determined by their lower and upper temperature tolerance, where the optimal temperature occurs, and how long certain temperatures prevail.

Minimum winter temperatures

In temperate and boreal regions insects are often limited by winter cold. For example, for the mountain pine beetle (*Dendroctonus ponderosae*) (Carroll *et al.*, 2004), minimum winter temperatures below -40°C lead to 100% mortality of larvae whereas about 50% survive at minimum winter temperatures of about -25°C (Safranyik and Linton, 1998; Régnière and Bentz, 2007). Consequently, such species are able to persist at higher latitudes and elevations when minimum temperatures rise.

Maximum summer temperatures

On the other hand, species can be limited if summer maximum temperatures are rising beyond a species tolerance, and such effects are expected to occur at the opposite range limits, generally at lower latitudes.

Heat sums (degree-days)

Because insects are ectothermic they require a certain heat sum to complete their development, and this is typically expressed as 'degree-days' (a measure calculated as the average daily temperature above a species-specific developmental threshold

which is accumulated for a period of interest). For example, the mountain pine beetle requires a minimum of 305 degree-days above 5.5°C from the beginning of August until the end of the growing season as well as a minimum of 833 degree-days from the beginning of August until the end of the following July in order to complete its life cycle once per year. If heat sums exceed these values, a partial or complete second generation can develop (Logan and Powell, 2004), which can either destabilise populations or contribute substantially to the occurrence of population outbreaks.

Under warmer climatic conditions, various phenological events occur earlier in spring (for example, adult emergence, peak flight, breeding, etc.) or later in autumn (for example, start of diapause, hibernation, departure of migratory species, etc.). This has already been documented for a wide range of insects and other species (e.g., Parmesan, 2006). The combined effects of increases in minimum and maximum temperatures and heat sums accumulated over time can also lead to latitudinal and altitudinal range shifts, usually pole-wards and towards higher altitudes.

3.2.3.2 Effects of changes in precipitation

While insect species are usually not directly affected by the amount and temporal variability in precipitation, the susceptibility of plants to insect attack can be greatly increased. Bark beetles and wood borers are best known to exploit drought-stressed trees that have weakened defences, and the resulting increase in available breeding material has been documented to cause outbreaks of such species. In the mountain pine beetle, outbreaks are often correlated with the occurrence of two consecutive years of below-average rainfall (Carroll *et al.*, 2004). Similar observations have been made regarding pinhole borer beetles (*Platypus* spp.) in New Zealand's *Nothofagus* forests, where outbreaks and tree mortality caused by beetle attacks follow unusually dry seasons (Milligan, 1979).

3.2.3.2.3 Effects of wind

Strong storm events causing windthrow of trees can lead to outbreaks of bark beetles. Species such as *Ips typographus* and *Ips sexdentatus*, which breed primarily in spruce and pine logs, respectively, are well-known to have outbreaks in the years following significant storm events. For example, following storm Lothar, which in December 1999 devastated large areas of forest in Europe, *I. typographus* produced enormous populations that then attacked and killed a large proportion of the surviving trees (Wermelinger, 2004). Models predict that even moderate increases in wind speed may exacerbate wind throw problems, worsening the impact of this bark beetle as a result of climate change (Schlyter *et al.*, 2006).

3.2.3.3 Effects on pathogens

Disease caused by fungal plant pathogens is influenced by interactions between the pathogen, host, and environment (Gäumann, 1950; Stevens, 1960; Tainter and Baker, 1996; Agrios, 2005). Changes in the environment can shorten the reproduction period of the pathogen and thus increase inoculum which will result in more disease. Conversely, in some circumstances, inoculum may be reduced and thus disease would be reduced. Climatic factors that influence the rate of spread of a pathogen and its growth include rainfall, temperature, humidity, wind, and radiation.

3.2.4 Origin of pests under current and future climate

Recent research has investigated how climate change will affect the likely origin of future pest invasions. The following analysis is a condensed version of the report written by Kriticos (2006).

3.2.4.1 Current origin of pests.

Under current climate, the geographic origins of New Zealand's pests are shown in Figure 3.12. Areas from which pests are most likely to currently invade are those regions with a high composite match index (CMI). Areas with the highest CMI are south-eastern Australia and Western Europe (Fig. 3.12).

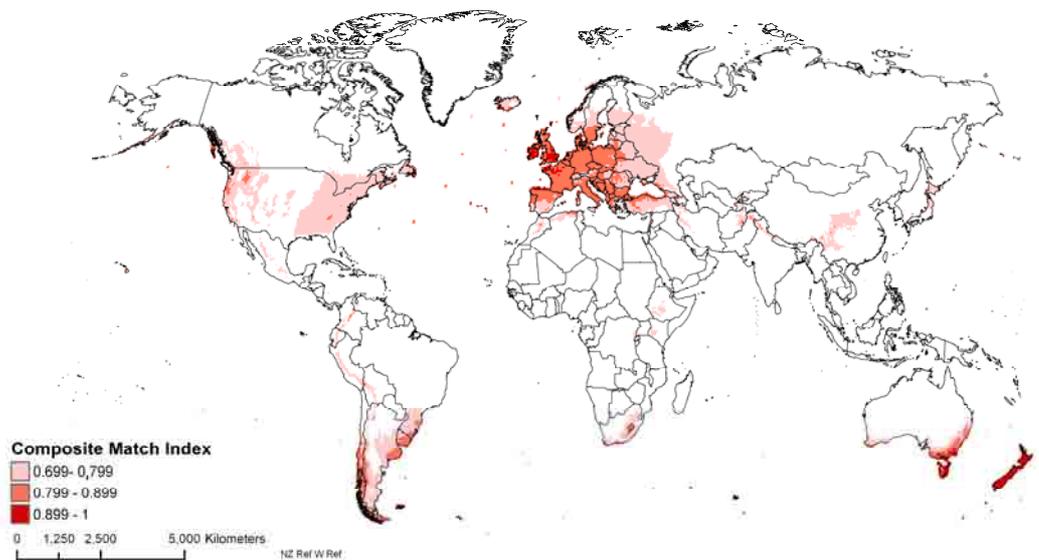


Figure 3.12. Current pest threat regions for New Zealand. Climate Match between New Zealand and the rest of the world under the 1961-1990 reference climate. The results were derived using the CLIMEX Match Climates (region) algorithm.

3.2.4.2. Potential current areas of future pests for New Zealand

A CLIMEX run was undertaken to match the climate of New Zealand in the 2080s with the current climate of the rest of the world, to ascertain where the future threats are likely to originate from (Figure 3.13), under current climate.

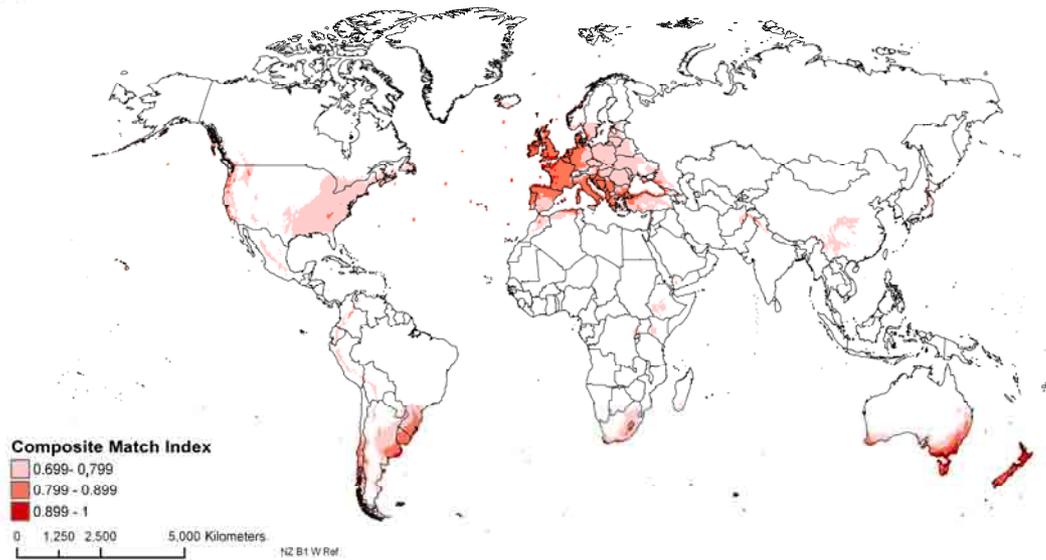
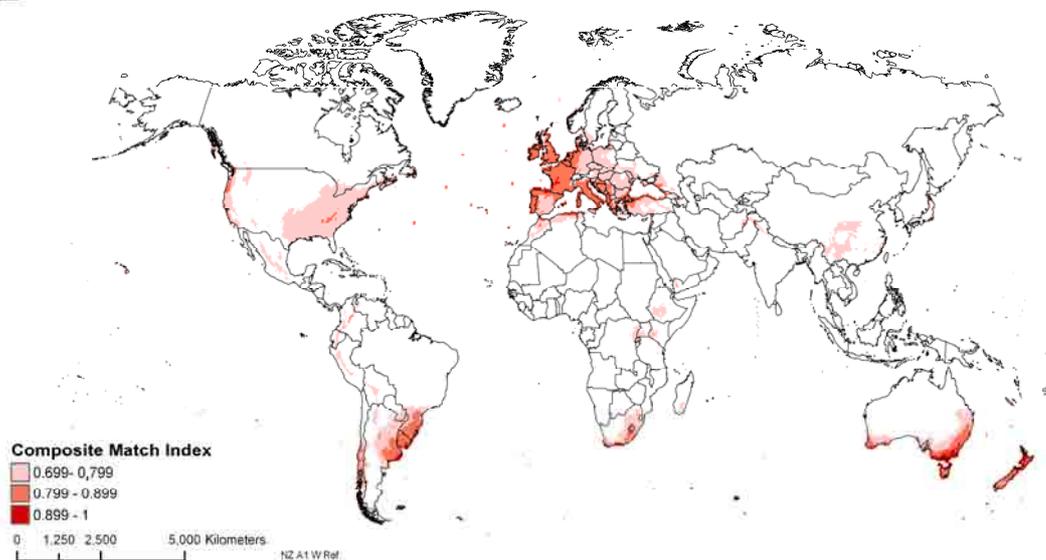
A**B**

Figure 3.13. The current geographic origin of pests that may pose a biosecurity threat to New Zealand in the future. A) comparing New Zealand's climate in the 2080s under the Hadley3 GCM run with the SRES B1 scenario with the rest of the world under the reference climate (1961-1990 average), and B) comparing New Zealand's climate in the 2080s under the Hadley3 GCM run with the SRES A1 scenario with the rest of the world under the reference climate (1961-1990 average).

Elimination of the area which current threats originate from in Figure 3.12 produces the new current areas which potential threats are likely to originate from, as shown in Figure 3.14. In geographic terms, the increased threat areas are not dramatically increased beyond the current climate threat, though some new countries may harbour emerging pests (e.g. Madagascar and Paraguay). Comparing Figures 3.14a and 3.14b, it is apparent that the regions harbouring potential new pests are not very sensitive to the degree of climate change in the range investigated here. This type of information highlights current areas which should be searched to identify new pests which could invade New Zealand in the future.

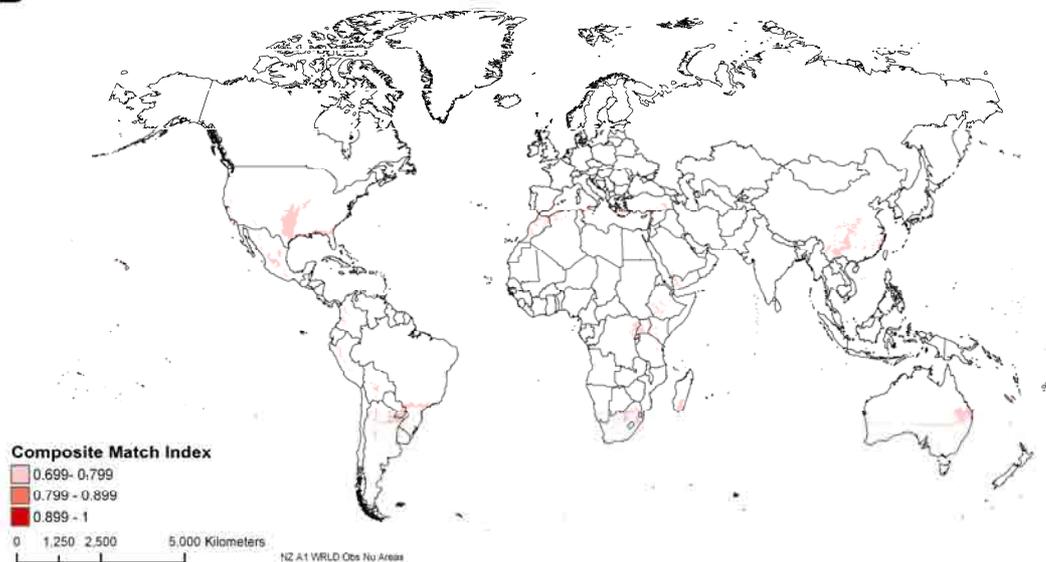
A**B**

Figure 3.14. New areas containing pests that may pose a biosecurity threat to New Zealand in the future. To focus attention on the changes with respect to the present situation, areas that, under current climate have a CMI value greater than 0.7 are excluded. A, comparing New Zealand's climate in the 2080s under the Hadley3 GCM run with the SRES B1 scenario with the rest of the world under the reference climate (1961-1990 average), and B, comparing New Zealand's climate in the 2080s under the Hadley3 GCM run with the SRES A1 scenario with the rest of the world under the reference climate (1961-1990 average).

Consideration of the pests found in the future threat areas identified in Figs 3.13 and 3.14 could reveal pest organisms not previously considered to be a threat to New Zealand. For example, the region highlighted within Queensland in Figure 3.14, includes a very high density of known occurrences for the highly invasive species *Melaleuca quinquenervia* in its native range of Australia (Watt *et al.*, submitted a). This species is currently found within New Zealand as an ornamental plant. Further research should more closely examine invasive species, within these new areas to identify any potential new invasive species, and perhaps more importantly, identify any current 'sleeping' species, such as *M. quinquenervia* currently in New Zealand, which could become invasive in the future.

The following sections describe the potential influence of climate change on weeds, insects and pathogens, and how these changes are likely to impact plantation productivity in more detail.

3.2.5 Influence of climate change on weeds

3.2.5.1 Importance of weeds in NZ plantation forestry

3.2.5.1.1 Background

Numerous studies have shown that weeds can severely reduce tree growth during the juvenile period by competing for available site resources. Consequently, weed control is the single most important establishment treatment influencing tree growth and survival (Mason and Whyte, 1997; Zhao, 1999). Management of non-crop vegetation also has a substantial influence on the economics of plantation forestry. In New Zealand if only direct costs are considered, the economic benefit to the forestry sector from sustaining cost-effective weed control is at least \$108m/yr. Without this weed control, plantation forestry would not be economically viable (Richardson and Kimberley, 2006).

Application of herbicide is widely recognised as the most cost effective means of managing non-crop vegetation. While cost effectiveness is generally the key criterion governing herbicide use, environmental constraints are becoming more important as pressure increases to reduce herbicide usage (Richardson *et al.*, 2006). Environmental certification requires that some herbicides are no longer used at all while the use of others is minimised. This may present some management challenges on the sector particularly if changes in the vigour and composition of weeds in New Zealand under climate change result in greater levels of competition from weeds.

3.2.5.1.2 Current major forestry weeds within NZ

During 2007, a survey of New Zealand forest owners was undertaken to determine the most problematic weeds within New Zealand (Watt, unpub. data). In total, there were 25 responses from twelve forest owning organisations (a number of organisations had multiple branches which responded). Respondents owned forests covering a large proportion of the New Zealand plantation estate. These forests were located across almost the entire environmental gradient over which plantations are grown in New Zealand. The companies ranked the weed species on a scale of 1 to 10, representing low to high impact, based on how difficult the weed is to control, and its influence on crop productivity. Averaging the scores, across all respondents, and by region produced the ranking shown in Table 3.3.

Table 3.3. Average scores for the top 10 exotic weed species in New Zealand, and the native species Bracken, as determined from a survey of twelve forest owning organisations. Also shown is average score for each species, by region^{1, 2}. For each region heading the number of respondents is given in brackets, after the region. For ease of interpretation regional scores of 0, are shown as a blank. Unpublished data from Watt, M.S.

Weed species		Average score by region									
Common name	Scientific name	All regions	Nthland/ Akland (4)	Waikato/ BOP (6)	Hawkes Bay (2)	Lower NI ³ (2)	Nelson/ Tasman (3)	Cant. (3)	West Coast (1)	Otago (1)	Sthland (1)
Gorse	<i>Ulex europaeus</i>	8.0	9	6	6	1	7	9	1	1	9
Scotch broom	<i>Cytisus scoparius</i>	5.9		4	9	4	9	1	8	9	1
Bracken	<i>Pteridium esculentum</i>	4.4	2	4	9		6	5	5	7	8
Blackberry	<i>Rubus fruticosus</i>	4.4	1	8	8		3	6	4	4	5
Wilding radiata pine	<i>Pinus radiata</i>	3.3	1		7	4	8	3		5	3
Buddleia	<i>Buddleja davidii</i>	3.3		8	7	2	5		3		
Pampas	<i>Cortaderia</i> spp.	3.0	6	5			4		3		
Wattle	<i>Acacia</i> spp.	2.4	6	3	1		1	6			
Woolly nightshade	<i>Solanum mauritianum</i>	2.0	7	3			1	1			
Himalayan honeysuckle	<i>Leycesteria formosa</i>	1.8		2	3		2		6	8	7
Inkweed	<i>Phytolacca octandra</i>	1.1	5	1							

¹Two respondents included in overall average scores were not included in the average scores by region, as the responses were not specific to a particular region.

²Poverty Bay was not included, as none of the respondents solely described the problematic weeds within this region.

³Includes Manawatu-Wanganui and Wellington

Table 3.3 clearly shows perennial tall species dominate the top 10 most problematic weed species. These results also indicate that a small group of weeds, which include gorse, Scotch broom, bracken and blackberry are most problematic, with all species recording average scores exceeding 4. These four species are extremely invasive and very competitive woody weeds which were listed as being problematic in almost all regions of New Zealand (Table 3.3). Also of considerable importance is wilding radiata pine regeneration which has a cosmopolitan distribution throughout New Zealand. Although buddleia and pampas are both problematic in many areas, they do not appear to be of major concern in the dry east coast regions of both Islands. *Acacia* spp. was reported as having the most detrimental impact in Northland and Canterbury. The importance of woolly nightshade and inkweed as competitors appears to be confined to the upper North Island. For inkweed this is likely to be related to the susceptibility of the species to frost (pers. comm. B. Richardson). In contrast, Himalayan honeysuckle is most problematic in both wet and dry regions in the lower South Island.

3.2.5.1.3 Influence of current New Zealand weeds on crop productivity

On wet sites where there is little competition for water, weeds mainly retard tree growth through competition for light. The magnitude of this competition for light has been found to increase as the ratio of weed height to tree height increases (Richardson *et al.*, 1996). Consequently, tall shrubby species such as Scotch broom reduce tree growth on these sites more than short species such as grasses and herbaceous species (Fig. 3.15). Although Pampas is the obvious exception to this rule, this grass species competes strongly for light as it grows as rapidly and reaches a comparable height to most tall perennial woody species. Compared to the weed-free condition the absence of weed control for a period of three years has been found to reduce tree volume growth by as little as 18% for lotus to as much as for 77% for pampas (Fig. 3.15). For trees growing with buddleia, these reductions have been reported to delay crop volume development compared to the weed-free condition by 14 months, at age 3 (Richardson *et al.*, 1996).

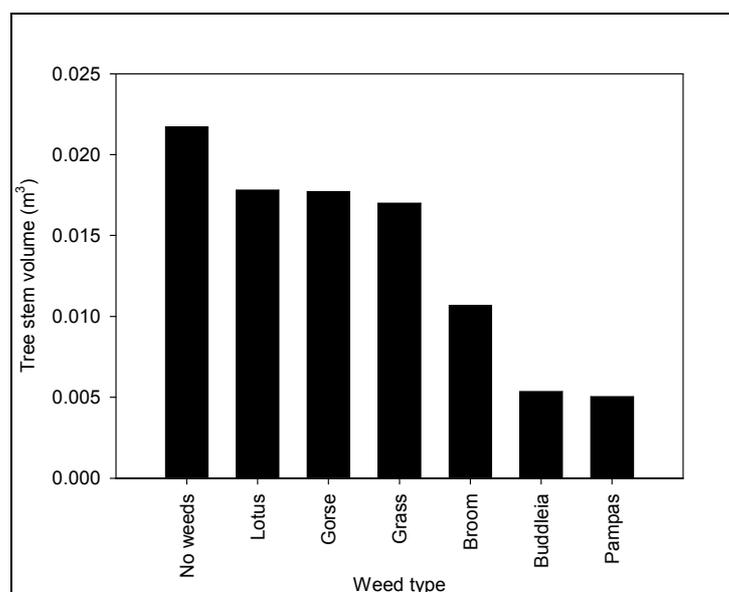


Figure 3.15. Effect of weed type on *Pinus radiata* stem volume three years after planting, on a wet site not subject to water competition. The different weed species were planted by hand as seedlings around the trees. This graphic is reproduced from Richardson *et al.* (1996).

On dryland sites, weed competition and growth reductions are more pronounced as weeds compete for both light and water. The intensity of weed competition for water can be quantified as the difference in root-zone water storage, W , between weed-free plots and plots with weeds (Watt *et al.*, 2007). In general, differences in W tend to become more pronounced as sites become more arid. Compared to trees growing with no weeds broom was found to reduce tree volume by 96 % (Watt *et al.*, 2003a) on a dryland Canterbury site at age 3 (Figure 3.16).

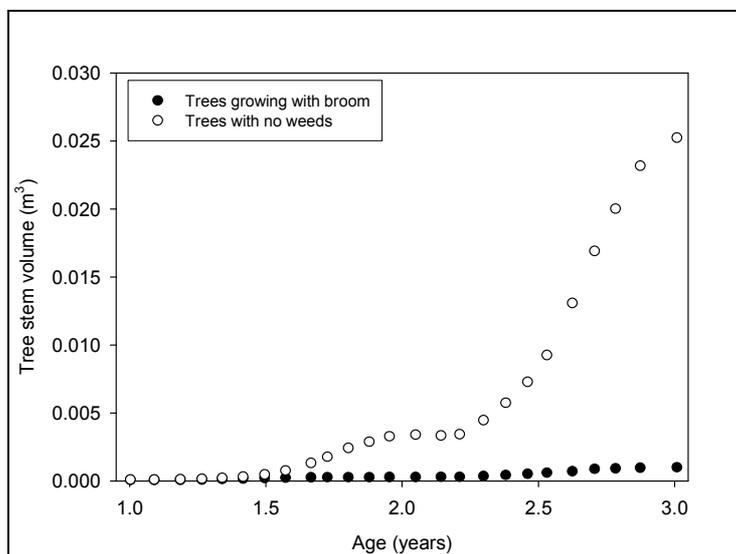


Figure 3.16. Changes in volume growth for trees in plots with broom (closed circles) and in plots with no weeds (open circles) from age one to three, at a dryland site in Canterbury. This is unpublished data taken from an experiment fully described in Watt *et al.* (2003a).

As these reductions in tree growth result from weed competition in the absence of any weed control, they illustrate the highest level of loss which can occur, compared to the weed free condition. However, it is worth noting that estimates of growth and tree mortality may be underestimated, as most studies have some form of minimal herbicide application at the start to ensure tree survival. Although they do represent an extreme level of loss, the results highlight the importance of site type and weed habit on growth losses, and how important correct application of herbicide is to achieve acceptable crop growth.

Through correct application of herbicides growth losses attributable to weed competition are usually considerably less than those described. However, as herbicide is expensive to apply there is a trade off between application rate and frequency of application and the growth loss incurred. Under cost-effective herbicide application rates, growth losses compared to the weed-free condition are typically around one to four years (pers. comm. B. Richardson).

Competition from weeds normally ceases after tree canopy closure. As most New Zealand weed species are light demanding they typically die out after the canopy closes at five to seven years of age. Silvicultural operations such as thinning and pruning can delay canopy closure. This effect may be greater on dryland sites as water stress, from weed competition can delay tree growth considerably (Figure 3.16). For example, at a dryland site in Canterbury, which was thinned and pruned at age six, broom was found to retard tree growth through competition for water up to 12 years after planting (Richardson *et al.*, 2002). From canopy closure onwards most studies have reported the trajectory of tree growth curves between treated (weed-free) and untreated (no weed control) stands are parallel, until the rotation end (Richardson,

1993). That is, following canopy closure there is usually no further loss in tree growth, due to weed competition.

3.2.5.2 Effect of climate change on weeds and plantation forestry

3.2.5.2.1 Changes in exposure of plantations to current weeds

Only a limited amount of work has investigated the future distribution of the most problematic weed species in New Zealand (Table 3.3) under climate change. Using two contrasting Global Climate Models (Hadley3 GCM and Echam4 GCM), Potter *et al.* (submitted) found that changes in climate (in 2080) will have little effect on the potential distribution of broom, with all regions remaining suitable for the species.

Although it is not currently classed as a problematic weed, the exotic tree species *Melaleuca quinquenervia*, which originates from Australia, Irian Jaya, southern New Guinea, and New Caledonia (Blake, 1968; Correll and Correll, 1986; Holliday, 1989; Geary and Woodall, 1990; Craven, 1999) has considerable potential to become invasive under climate change. This species, which is listed as one of the 100 least wanted alien species, is currently established in Auckland and Northland (Biosecurity New Zealand, Issue 59, page 5). Using CLIMEX, it was found that thermal accumulation under current climate was insufficient to allow reproduction within the current population (Watt *et al.*, submitted a). However, given that climate change will increase thermal accumulation above reproduction thresholds for northern New Zealand CLIMEX predictions suggest these 'sleeper' populations are likely to become foci for quite rapid spread of the species, as climate is projected to become suitable to optimal for the species in northern regions (Fig. 3.17). Surpassing the thermal requirement for reproduction in this species, could have significant consequences, as this species has been found to be extremely invasive and difficult to control in exotic locations, such as Florida (Myers, 1975; Austin, 1978; Wade *et al.*, 1980; Woodall, 1981, 1983).

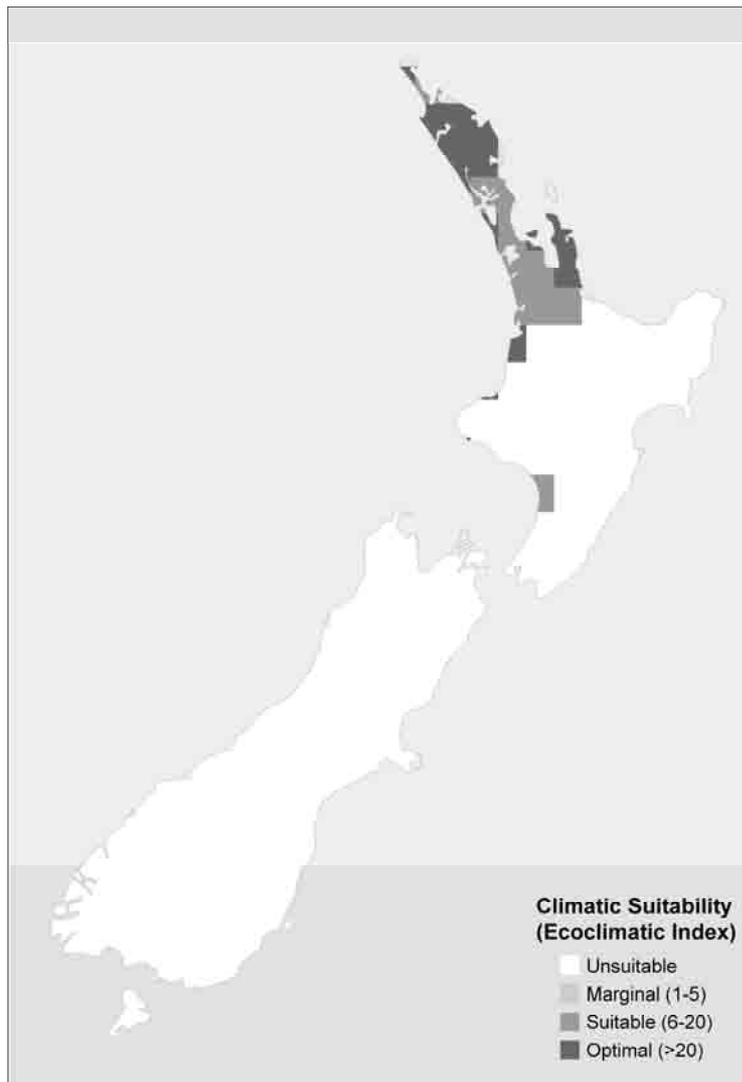


Figure 3.17. New Zealand map showing climate suitability (Ecoclimatic Index, EI) for *Melaleuca quinquenervia* in 2080, predicted using the ECHAM 4 GCM under a high emission scenario. The white areas show regions unsuitable for the tree, while the graduated grey areas highlight regions which will be suitable for the tree, expressed in terms of the Ecoclimatic index (see scale). Ecoclimatic index can be used to classify regions as marginal (EI = 1-5), suitable (EI = 6-20) and optimal (EI > 20) for *Melaleuca quinquenervia*.

Another recent exotic arrival which may cause problems under climate change is kudzu (*Pueraria montana*). This species has recently been identified growing at four sites in the Bay of Plenty (for more information see www.envbop.govt.nz/weeds/Weed357.asp). Kudzu is a perennial, semi-woody, climbing leguminous vine, which is extremely invasive in the southeastern United States (Britton *et al.*, 2002). Although we do not currently have an estimate of the potential distribution of this plant, the distribution where it is invasive, is quite similar to that of *M. quinquenervia*, which suggests that it could be more of a problem under climate change than at the present time. However, the threat of this plant should not be underestimated as it can completely replace existing vegetation (Figs. 3.18 and 3.19) and is very difficult to control (Britton *et al.*, 2002).



Figure 3.18. Landscape uses probably account for kudzu patches surrounding crumbling homesteads throughout the south of the United States. Photograph reproduced from Britton *et al.* (2002).



Figure 3.19. Kudzu kills trees by shading them and spreads inexorably, mostly through soil movement and vegetative growth. Photograph reproduced from Britton *et al.* (2002).

3.2.5.2.2 Effects of climate change on weed growth

There has been very little research investigating how climate change is likely to influence weed growth within New Zealand. Future research which should be undertaken is outlined in Section 3.2.8.

3.2.5.2.3 Likely effects of how changes in weed distribution and growth will influence plantation growth

The previously discussed general impact of climate change on weeds suggests that the following factors may have a predominantly negative influence on tree growth in New Zealand plantations.

1. In their latest report NIWA has indicated that potential evapotranspiration deficit (PED) will increase throughout almost all of New Zealand under climate change (Mullen *et al.*, 2005). If this projection is correct (see section 2.4 for limitations of the method used to make these projections), competition for water from weeds is likely to increase in most regions. This is likely to result in greater levels of weed competition for three reasons. Firstly, increased competition for water from weeds will retard tree growth, more than it currently does, given that tree growth declines concurrently with declining water storage in the root-zone (Watt *et al.*, 2007). This is clearly evident through a comparison of weed effects on tree growth on dryland vs. wet sites (see section 3.2.5.1.3 for relative differences). This effect is likely to be exacerbated as drier conditions will delay canopy closure, which may result in weed competition further into the rotation. Finally, it is possible that this increased competition may be intensified by the spread of tall woody weeds into dryland areas (Farquhar, 1997; Kriticos *et al.*, 2003a, b), as these should theoretically compete more strongly for site

resources than short non-woody species. Though competing more strongly than grasses and herbaceous species for water, there could be a positive feedback, which would further fuel the invasion by tall woody species. Although dryland sites in New Zealand are presently partially occupied by woody weeds, there is room for further expansion. Although theory suggest that woody weeds are stronger competitors than herbaceous and grass species, exceptions have been noted, and further research is needed to confirm this.

2. There is the strong risk that aggressive exotic woody tree species currently native to Australia such as *Acacia* spp., and *Eucalyptus* spp. may become more dominant competitors in New Zealand under climate change. Both *Acacia* spp. and *Eucalyptus* spp. can have very high growth rates and rapidly occupy disturbed sites, vigorously competing with *P. radiata* (Turvey *et al.*, 1983). As tree species, they can compete further into the rotation (e.g. Hunt *et al.*, 2006) than our current tall perennial weed species, which are predominantly shrubs. Some species have the ability to resprout, and seed germination is often stimulated by fire. It has been suggested that if some of the more aggressive representatives of these species become widespread in New Zealand, they would prove to be formidable competitors (Lavery, 1986). This outcome is very likely under climate change as increases in temperature and reductions in rainfall, shift the climate more closely towards the current Australian climate. This will make invasion from currently occurring members of these species more likely and also increase the risk profile for invasion of further species from Australia. In addition, the likely increases in fire frequency and severity will also make sites more predisposed to invasion from these species. Some *Acacia* species are already a problem in parts of Northland, the Bay of Plenty and Canterbury (Table 3.3).
3. Climate change may activate many vigorous and potentially invasive “sleeper weeds” currently within New Zealand. For instance, recent research (Watt *et al.*, submitted a) shows that the small ornamental population of the Australian native tree *M. quinquenervia* is likely to become naturalised and invasive very quickly under climate change, as its current steep southern range boundary is largely determined by thermal accumulation. This species is likely to be very competitive within northern New Zealand, as it is a fast growing, vigorous and tall tree, which forms extensive monocultures. It would be prudent to identify any similar potentially invasive ‘sleeper’ weed species, such as kudzu, before climate change, and investigate the feasibility of eliminating these populations.
4. Weeds by their nature are vigorous and generally have a more indeterminate growth habit than trees. Therefore it is likely that weed growth will respond more strongly to increasing [CO₂], temperature, and length of the growing season than plantation trees. Furthermore, there are numerous weed species. Natural selection will favour those that will most positively respond to changed conditions. The overall response of the weed community will be determined by those species most favourably responding to changes in conditions. While the logic of this argument is appealing, there is currently little hard evidence to support this hypothesis. Determining the relative response of trees and weeds to these climate change factors is a very important issue as even slight variation in seasonal growth patterns between trees and weeds can have a substantial effect on the overall magnitude of crop-weed interactions (Kimberley and Richardson, 2004).
5. Increases in the number of plantation fires under climate change within New Zealand are likely to have a number of effects on weed composition. Firstly, an

increased number of fires may result in greater weed invasiveness from competitive species such as pampas, *Eucalyptus* spp., *Acacia* spp., and *Melaleuca* spp., as these are all able to rapidly colonise the bare mineral soil, which results from fire. The loss of nutrients through fires also typically makes the conditions more suitable for invasion by nitrogen-fixing weed species such as gorse and broom. The increased fire risk under climate change may also have some advantages for control of weeds. Fires have been shown to reduce seed numbers for gorse (Zabkiewicz and Gaskin, 1978) and wilding pines (Minko, 1985a, 1985b), which can reduce the number of these species in the future crop. Application of herbicide following fire is generally more effective, as herbicide deposition onto organic matter is minimised. Removal of organic debris also reduces the likelihood of out of season frost damage, and makes it easier to oversow species for weed control (Richardson, 1993). However, it is unlikely that these advantages will be of significant consequence if greater fire frequency results in a vegetation composition shift towards more competitive Australian native trees.

Further research is necessary to more accurately quantify the effects of these potential changes in weed communities on forest productivity. Models are the best way forward for addressing a number of these issues. A full review of their current limitations and future research required is outlined in section 3.2.8.

3.2.6 Influence of climate change on insects

3.2.6.1 Importance of insect pests in New Zealand plantation forestry

3.2.6.1.1 Background

Data on economic impacts of pests and diseases in New Zealand's plantation forests from several sources (including Bulman, 2001; Bulman, 2002; Hood *et al.*, 2002; Turner *et al.*, 2004; Lindsay Bulman, pers. comm.) indicate that direct losses due to forest pests exceed \$160 million per year. Additional costs are incurred as a result of market access issues. For example, logs and timber exported to several countries has to be fumigated to ensure freedom from pests such as the burnt pine longhorn beetle, *Arhopalus ferox*. This exotic longhorn beetle of European origin is very widespread and abundant in New Zealand, wherever pines occur. It is often found sheltering among logs and stacks of sawn timber. Alone the cost of methyl bromide (MeBr) fumigation of sawn timber destined for export to Australia amounts to about NZ\$600,000 (Self and Turner, 2008). Adding to that the fumigation of logs and the environmental and health costs associated with the use of MeBr means that the total cost of this practise in New Zealand exceeds several hundred million NZ\$ (Self and Turner, 2008). Not included in either of these figures are costs associated with lost opportunity because certain trees cannot be grown in New Zealand as a result of the seriousness of damage sustained from pests.

3.2.6.1.2 Current major forest insects in New Zealand and overseas

Forest insect pests can broadly be classified by their feeding guild (e.g., defoliators, bark beetles, wood borers, sap suckers, seed insects). The majority of herbivorous insects are relatively host-specific, with most attacking plants within a genus or a family, although some polyphagous species have a much wider host range. Because New Zealand's indigenous flora does not include any pines or other Pinaceae, there were no insects in New Zealand's fauna specialising on pines, and consequently pines, other Pinaceae, and most other exotic trees planted in New Zealand enjoyed a

relative freedom from pests. However, a few polyphagous indigenous insects have colonised exotic trees, and there is a steady inflow of invading species from regions where our plantation forest trees originate (e.g., Brockerhoff and Bain, 2000). Until today about 150 insect pests have been recorded from pines in New Zealand (John Bain, pers. comm.), although the majority of these species do not cause economically important damage.

Historically the most serious insect pest problems in New Zealand's plantation forests were caused by the siren woodwasp (*Sirex noctilio*). This wood boring wasp, which was accidentally introduced from Europe, and its associated fungus had serious outbreaks and caused much mortality in pine stands (Zondag and Nuttall, 1977). Changed silvicultural practices and successful biocontrol eventually reduced this pest problem below an economic threshold, and today it rarely causes problems.

The native common looper, *Pseudocoremia suavis*, had a series of outbreaks that led to significant or even complete defoliation of pines in several plantations in lowland Canterbury (White, 1974). These outbreaks were treated aerially with DDT. Later outbreaks occurred on Douglas-fir in Kaingaroa Forest (Kay, 1983). The insect has colonised conifer plantations across the entire country, but further serious outbreaks have not been recorded (Berndt *et al.*, 2004).

The black pine bark beetle, *Hylastes ater*, is currently considered to be the most important insect pest in New Zealand's plantation forests. Although this beetle primarily infests stumps and logging slash, maturation feeding on seedlings causes seedling mortality which may require replanting. In addition, *H. ater* and other bark beetles and wood borer act as vectors of sapstain fungi (Reay *et al.*, 2002), reducing the time available between harvesting and processing of logs.

The Monterey pine aphid *Essigella californica* was first recorded in New Zealand in 1998. Because of its pest status overseas (e.g., in Australia and Portugal), it was initially feared that it may become an important pest in New Zealand. It is now widespread across most of New Zealand where plantation forests occur, however, in the 10 years since its discovery, it has caused only minor damage (Watson *et al.*, 2008).

Another introduced aphid, the spruce aphid, *Elatobium abietinum*, is a significant pest of spruce. Although species such as Sitka spruce have significant potential as alternative species for plantation forestry in New Zealand, the amount of defoliation and resulting growth losses caused by this insect (Zondag, 1983) largely prevent the commercial planting of spruce in New Zealand.

The species previously described above are also important as quarantine pests and most attention has been on wood borers (such as *Arhopalus fesus*) and bark beetles (such as *Hylastes ater* and *Hylurgus ligniperda*) (e.g., USDA Forest Service, 1992) (see above). In addition, wood borers and bark beetles are potentially important vectors of plant pathogens, although at present, this does not appear to be a major problem in New Zealand's plantation forest. However, if pitch canker becomes established in New Zealand the disease could have very severe impacts on the health of pines in New Zealand, and insect vectors could play an important role in the spread of this disease (e.g., Wingfield and Coutinho, 2008).

Compared with the more significant impacts of insect pests on plantation forestry in most countries overseas, the situation in New Zealand is still very favourable, highlighting the importance of biosecurity. Several examples of major insect pests

affecting trees overseas are provided below in the sections on climate change impacts on pests.

3.2.6.2 Effect of climate change on insects and plantation forestry

3.2.6.2.1 Changes in exposure of plantations to current insects

The general principles of the effects of climate change on exposure to weeds also apply to insect pests. In New Zealand, relatively little work has been done on specific forest insects with regard to differences in exposure. To our knowledge there are no documented cases of climate change-related spread of pests into previously uninfested regions in New Zealand, although there is ample evidence of such cases overseas (see above and specific examples described below). However, modelling efforts show that such effects are likely to occur in New Zealand. An example, involving a horticultural pest, concerns the oriental fruit fly *Bactrocera dorsalis* (Kriticos, 2007). This species is not present in New Zealand but has often been intercepted. Currently the climate is suitable only in northern New Zealand and some coastal regions of the North Island (Fig. 3.20). Under the 2080 climate scenario New Zealand's suitability is projected to extend over most of the North Island and parts of the South Island (Fig. 3.20).

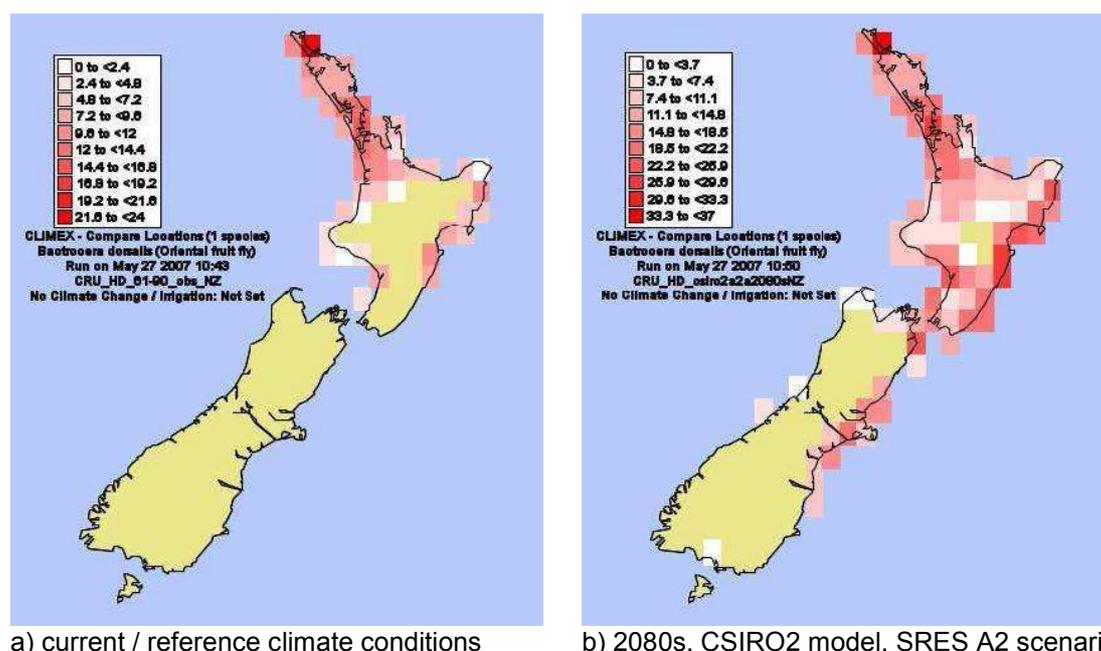


Figure 3.20. Climate suitability of New Zealand for *Bactrocera dorsalis* (a) under reference climate conditions, (b) during the 2080s based on the CSIRO2 model using the SRES A2 scenario. Taken from Kriticos (2007).

There is better evidence overseas for changes in exposure to pests due to range shifts. The pine processionary moth, *Thaumetopoea pityocampa* (Lepidoptera: Thaumetopoeidae), is a serious defoliator of pines that occurs in Mediterranean parts of southern Europe (Brockerhoff *et al.*, 2007). Apart from the damage caused by large-scale defoliation during outbreaks of this native pest (Fig. 3.21), many people suffer from dermatitis due to exposure to the urticating hairs of the caterpillars. Over the last few decades the pine processionary moth has extended its range north and upwards into regions that were previously too cold (Battisti *et al.*, 2005, Robinet *et al.*, 2007). It now occurs on the outskirts of Paris where it has previously never been observed (Fig. 3.22). The pine processionary moth could be a very serious pest in

New Zealand, if it was accidentally introduced (Brockerhoff *et al.*, 2007), and increasing winter temperatures may render our climate more suitable, although this has yet to be determined.



Figure 3.21. A stand of *Pinus radiata* with serious defoliation caused by caterpillars of the pine processionary moth (near San Roke, Basque Region, Spain). Many silken nests are noticeable as white spots. Trees have suffered approximately 80–100% defoliation. Photo by Eckehard Brockerhoff.

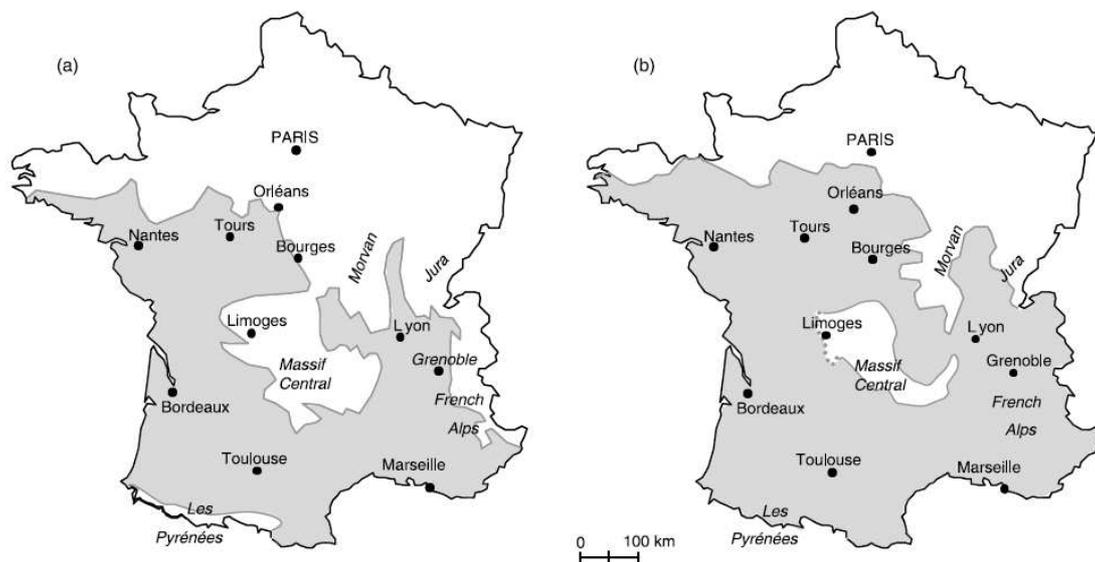


Figure 3.22. Range expansion of the pine processionary moth in France as a result of consistently warmer winters between (a) 1969-1980 and (b) 2005. From Robinet *et al.* (2007).

Possibly the highest impact range shift of any species is that by the mountain pine beetle, *Dendroctonus ponderosae*, which occurs in the Rocky Mountains region from inland California to British Columbia. It infests mainly lodgepole pine, *Pinus contorta*,

but also many other species of pine. Mountain pine beetle often mass-attacks particular trees, mediated by its use of an aggregation pheromone, which serves to overcome defences such as extensive resin production. Only dead trees can be used for breeding in the phloem zone. The most recent outbreak in British Columbia has been the biggest recorded so far, on an area exceeding the size of the North Island (Fig. 3.23). This outbreak has reached further north-east into higher elevation forests than previously recorded outbreaks, extending into regions that were previously too cold for mountain pine beetle (Fig. 3.24) (Carroll *et al.*, 2004). As a result the beetle has already been able to 'jump' over the main divide into the boreal forest of northern Alberta. A new host, jack pine, *Pinus banksiana*, is a dominant species in Canada's boreal forest which spans from Alberta thousands of kilometres east to Quebec. It is feared that a massive outbreak will take place in these extensive jack pine forests where mountain pine beetle was previously absent. This remarkable range extension is thought to be the direct result of climate change (here mainly increasing winter temperatures).

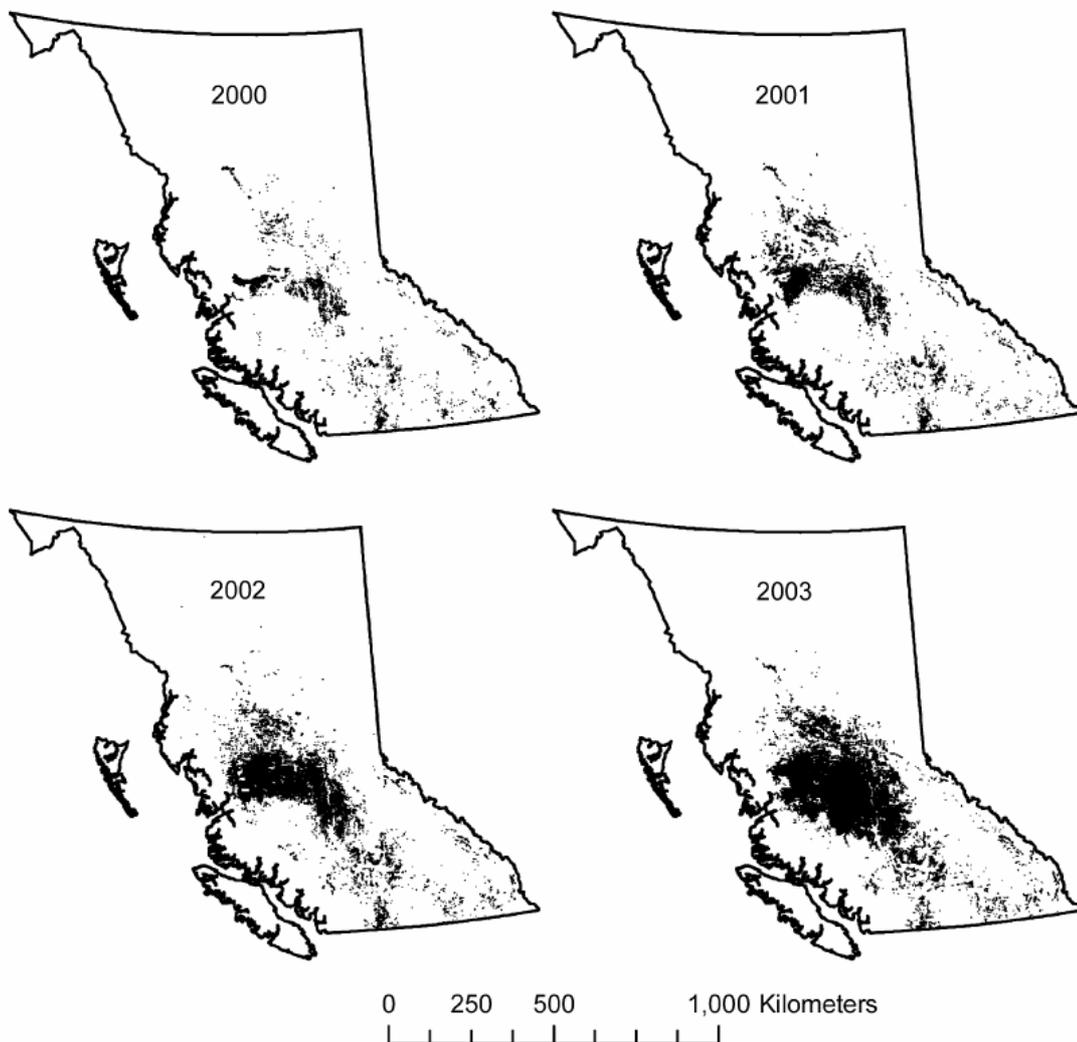


Figure 3.23. Outbreak range of the mountain pine beetle in British Columbia, Canada, showing the extent of tree mortality from 2000-2003. From Aukema *et al.* (2006).

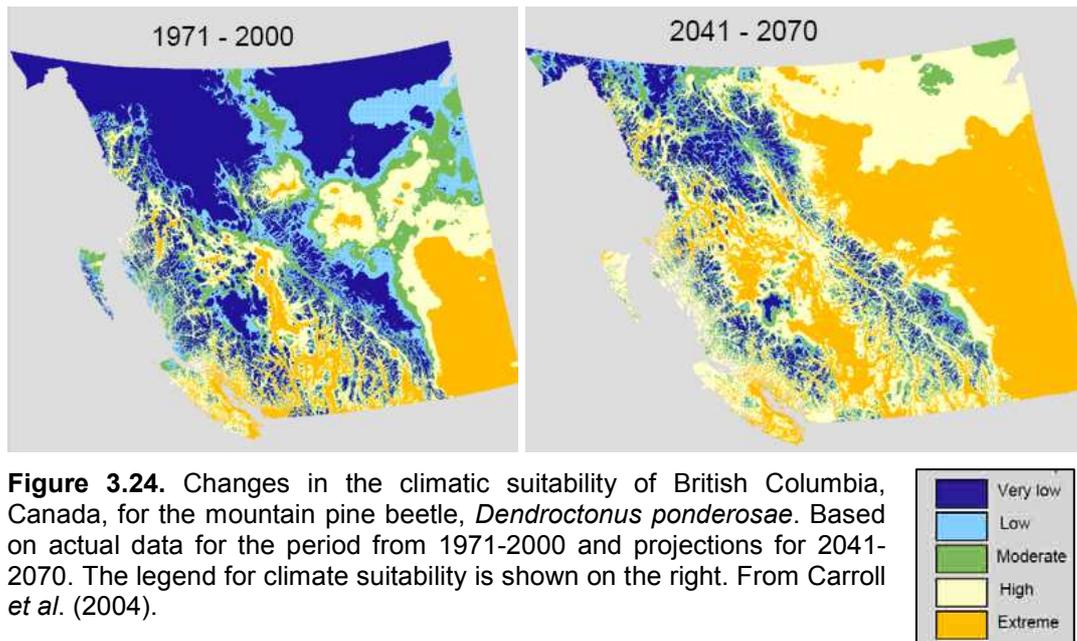


Figure 3.24. Changes in the climatic suitability of British Columbia, Canada, for the mountain pine beetle, *Dendroctonus ponderosae*. Based on actual data for the period from 1971-2000 and projections for 2041-2070. The legend for climate suitability is shown on the right. From Carroll *et al.* (2004).

3.2.6.2.2 Effects of climate change on insect populations

Little work has been done in New Zealand on the effects of climate change on insect populations but some inferences are possible based on some limited knowledge of relationships between insect damage or abundance and climate. The most severe outbreaks of a defoliator in New Zealand's plantation forests were those by the common looper, *Pseudocoremia suavis*, in Canterbury (White, 1974). White (1974) hypothesised that the combined impacts of summer drought, waterlogging in winter, and site factors that increased the trees' susceptibility to wind were responsible for the outbreaks at this location. Although this hypothesis appears to be plausible, a preliminary re-examination of the climatic conditions before, during, and after the outbreaks, indicated further research is needed to verify whether populations can in fact respond quickly to changes in such condition (Brockhoff and Watt, unpublished data). An international review of the relationship between drought and forest insect outbreaks mentions numerous cases for various insect guilds although the causal relationship with drought was not clear concerning defoliators (Martinat, 1987, but see below). However, without a thorough re-analysis of the data available on outbreaks of *P. suavis*, it will be difficult to rule out that climate change-related increases in drought frequency or severity could provoke future outbreaks.

By contrast, there is good evidence for such effects on populations of wood borers and bark beetles (Martinat, 1987). A New Zealand example has already been mentioned above, concerning the pinhole borers, *Platypus* spp., which have repeatedly risen to outbreak levels following unusually dry periods (Milligan, 1979). Although *Platypus* spp. are known to occasionally attack pine and other trees grown in New Zealand's plantations, their outbreaks are expected to affect more native trees, especially *Nothofagus*. There is general agreement that outbreaks by pine-infesting bark beetles, such as the mountain pine beetle, follow periods of below-average rainfall (e.g., Carroll *et al.*, 2004). Presently there is no aggressive bark beetle present in New Zealand that could cause tree mortality. However, damage to seedlings by *Hylastes ater* could potentially worsen, although it is unclear if seedlings show reduced resistance to bark beetle attack similar to effects observed in older trees.

A North American bark beetle, *Ips grandicollis*, which has accidentally been introduced to Australia, normally infests felled trees and logging slash but it has also been recorded to kill drought-stressed trees (Wylie, 2001). At present, this bark beetle

does not occur in New Zealand but it is frequently intercepted in shipments from Australia (Brockerhoff *et al.*, 2006). If it was accidentally introduced to New Zealand this could be a problematic pest.

The Monterey pine aphid, *Essigella californica*, could be a candidate for a climate-change induced increase in severity. This recently introduced sap-sucking insect is widespread and moderately abundant in several regions of New Zealand, but its abundance appears to be generally below a level where it causes noticeable damage (Watson *et al.*, 2008). However, in parts of Australia and in Portugal, where *Essigella* was also introduced, it is considered to have economic pest potential and it has been shown to cause growth losses (May and Carlyle, 2003; Wharton and Kriticos, 2004). A CLIMEX model indicates that the relatively cool and wet climate in New Zealand is the reason for the less severe damage observed in this country (D. Kriticos, pers. comm.). This is consistent with the observation that this aphid is generally less abundant in the Bay of Plenty region than in Canterbury where it is considerably drier. In addition to an effect of increasing drought, it is possible that an increase in air temperature could also cause greater abundance of *Essigella* because it may be able to reproduce earlier and at an increased rate. The same could apply to other species where greater heat sums can lead to an increase in the number of generations per year (Logan and Powell, 2004).

A warmer climate could also have mitigating effects on forest insect pests in New Zealand. The eucalyptus tortoise beetle, *Paropsis charybdis*, is one of the most serious defoliators of eucalypts in New Zealand, along with the gumleaf skeletoniser, *Uraba lugens*. In parts of New Zealand, *Paropsis* is under successful biological control by an introduced egg parasitoid, *Enoggera nassau*, but in colder regions only the second generation of *Paropsis* is effectively controlled. This is thought to be the result of a climatic mismatch, in the sense that the egg parasitoid is not sufficiently active during its host's spring generation, when it is too cold in inland areas (Murphy and Kay, 2000). If this is indeed the case, then a warmer climate may allow the egg parasitoid to provide better control of the spring generation.

3.2.6.2.3 Likely effects of changes in insect distribution and abundance on plantation forest growth

Because of our limited knowledge of climate effects on forest insects in New Zealand, it is not possible to predict with any certainty what the likely effects of the expected warming and reduction of precipitation will be. It is difficult to generalise, particularly concerning indirect effects between trees, herbivorous insects, and their natural enemies, and the net effects on insect pests may vary among species. However, based on observations in other countries and according to the relationships described in more detail above (see sections 3.2.3.2 and 3.2.6.2), there is a high likelihood for climate change to cause some or all of the following which would affect plantation growth:

- range shifts of insect pests that are already present in New Zealand, leading, for example, to infestations in regions that are currently climatically less suitable.
- Increased risk of establishment of species for which New Zealand's climate is presently too cold, such as pest species from warm-temperate or subtropical regions.
- Greater abundance of insect pests due to better survival during the winter (for example, if minimum winter temperatures or the frequency of frosts increase) or if increased heat sums improve the reproductive potential.

- Reduced tree resistance to insect attack due to stress caused by drought or increased temperature.
- Reduced effectiveness of natural enemies due to increased climatic mismatches with the phenology of hosts or prey.
- Conversely, climate change may also lead to reduced problems with particular insect pests, for example, if growing conditions for trees improve (e.g., due to increased [CO₂]) and if this leads to increased tree resistance to pest attack.

3.2.7 Influence of climate change on pathogens

3.2.7.1 Importance of pathogens in NZ plantation forestry

3.2.7.1.1 Background

Direct losses in productivity due to exotic forest pathogens already established in New Zealand are significant. The most damaging plantation forest pathogens of *P. radiata* are *Cyclaneusma minus*, *Armillaria* spp., *Dothistroma pini*, and *Nectria fuckeliana*. Other pine pathogens may cause sporadic or localised damage but their national impact is insignificant. Other plantation species are prone to insect and fungal attack, especially *Eucalyptus* spp., but because they comprise such a small part of the plantation forestry estate the contribution of these pests and diseases to overall economic loss is slight. However, the opportunity cost due to exotic pests is high. Poor health is one of the major reasons eucalypts and cypresses have not been planted widely in New Zealand.

3.2.7.1.2 Economic impacts of current major forestry pathogens within NZ

Estimates of annual economic loss due to *Dothistroma* needle blight in New Zealand vary. Sutton (1971) estimated that the spray programme in New Zealand at three or four operations a rotation would only increase the wholesale price of timber by 1% and FOB price of newsprint by 0.4%. Sweet (1989) estimated that increment losses from *Dothistroma* needle blight were insignificant because of regular aerial spray applications of fungicide. Therefore, he considered that economic losses from *Dothistroma* needle blight amounted to the cost of spraying only, approximately \$1.2 million per annum.

New (1989b) estimated cost of *Dothistroma* needle blight control at \$1.6 million, and conservatively using a stumpage of \$20/m³ and low overall disease levels of 10% severity resulted in losses of 225,000 m³ per annum which at a stumpage of \$20/m³ gave a loss of \$4.5 million. Bulman (2007a) stated annual loss is in the order of \$24 million per year (Table 3.4).

The most recent published estimate of loss from *Cyclaneusma* needle cast is provided in Turner *et al.* (2004) and is \$61 m per year.

Table 3.4. Estimates of direct losses attributable to disease in New Zealand plantation forests (dollar value loss is stated as at the year the study was conducted)

Disease	Source	Stumpage (\$/m ³)	Growth loss (%)	Value loss (\$ million)
Cyclaneusma needle-cast	Sweet (1989)	60	2.3	8
Cyclaneusma needle-cast	van der Pas <i>et al.</i> (1984)		2.3	
Cyclaneusma needle-cast	Bulman (1988)		5.0	
Cyclaneusma needle-cast	Bulman (2001)	50	6.6	0
Cyclaneusma needle-cast	Bulman (Unpubl. Data)	50	6.6	0
Dothistroma needle blight	Sweet (1989)	60	0.0	
Dothistroma needle blight	New (1989b)	20	2.0*	
Dothistroma needle blight	Bulman (2002)	50	4.0**	0
Dothistroma needle blight	Bulman (Unpubl. Data)	50	3.0	7
Armillaria root rot	Sweet (1989)	60	0.3	
Armillaria root rot	Hood (2002)		2.0	0
Nectria flute canker	Bulman (Unpubl. Data)	50		0
Diplodia whorl canker	New (1989b)	20		
Diplodia whorl canker	Bulman (Unpubl. Data)	50		
Other forest pathogens	Bulman (Unpubl. Data)			
Swiss needle-cast	Manley (1985)		26-40***	
Swiss needle-cast	Knowles (Unpub. data)		20	
Swiss needle-cast	Bulman (Unpubl. Data)			
Cypress cankers	Bulman (Unpubl. Data)			

* Not stated in paper, but estimated from data supplied

** Double estimate taken from New (1989b)

*** From growth loss studies undertaken in one region

Using current data, direct losses due to forest pathogens are estimated to be in the order of \$143 million per year (Table 3.5). The three most damaging diseases are *Cyclaneusma* needle-cast, *Dothistroma* needle-blight, and *Armillaria* root rot. It is probable that *Nectria* will become more damaging as it expands its range and affects a larger area. Losses from all forest pests and pathogens amount to 5.1% of the annual contribution to GDP of the forest sector.

Table 3.5. Direct losses from forest pathogens (\$/year)

Disease	Host	Loss (\$ million/annum)
<i>Cyclaneusma</i> needle-cast	<i>Pinus radiata</i>	61
<i>Dothistroma</i> needle blight	<i>Pinus radiata</i>	23
<i>Armillaria</i> root rot	<i>Pinus radiata</i>	37
<i>Nectria</i> flute canker	<i>Pinus radiata</i>	10
Others	<i>Pinus radiata</i>	4
Swiss needle-cast	<i>Pseudotsuga menziesii</i>	4
<i>Diplodia</i> whorl canker	<i>Pinus radiata</i>	2
Cypress cankers	<i>Cupressus</i> spp.	2
Total		143

Table 3.6 shows losses from specific diseases on a per hectare basis and averaged over the entire estate. Losses on a per hectare basis are presented to demonstrate that locally, or on a minor host, pathogens may have significant impact compared with that at a national scale. For instance, in cypress plantations canker causes substantial losses at about \$44/ha/year but at a national scale this amounts to only \$2 million/year due to the relatively small amount of cypress grown. *Nectria* (flute canker) causes the greatest loss per hectare but, because it has only been recorded as occurring in the

lower half of the South Island (Bulman, 2007b), it doesn't contribute as much loss as Dothistroma needle blight or Cyclaneusma needle cast at the estate level.

Table 3.6. Direct losses from forest pathogens (\$/ha/year)

Disease	Area affected (000 ha)	Loss over area affected (\$/ha/year)	Loss over entire estate (\$/ha/year)
Cyclaneusma needle-cast	940	64.9	33.9
Dothistroma needle blight	715	32.2	12.8
Armillaria root rot	870	42.5	20.6
Nectria flute canker	140	71.4	5.6
Diplodia whorl canker	100	20.0	1.1
Others	100	40.0	2.2
Swiss needle-cast	150	26.7	4.0
Cypress cankers	45	44.3	2.0
Total			79.5

Total losses from forest pathogens are estimated to be \$80 per hectare over the entire estate of 1.8 million hectares. This estimate makes no allowance for losses due to new incursions by damaging pests or diseases, or reduction of value to the urban forest and aesthetic and protection values.

3.2.7.1.3. Influence of current New Zealand forest pathogens on crop productivity

The level of Dothistroma needle blight has a strong influence on growth. Van der Pas (1981) demonstrated that the relationship between volume loss and disease level was approximately proportional where an average disease level of 50% over a period of at least 3 years resulted in volume loss of 50%. Bulman also demonstrated a negative linear relationship between diameter increment and disease level (Fig. 3.25, diameter increment 1988-1996 and average disease 1986-1996, Bulman, L. unpubl. data). When expressed in terms of foliage loss most studies have shown that up to 25% foliage loss growth does not decline (Hocking and Etheridge, 1967; Whyte, 1969). Above this amount there has been shown to be a 1% loss in volume growth for every 1% loss of foliage up to 75% foliage loss (Old and Dudzinski, 1999). *Pinus radiata* in New Zealand is affected by Dothistroma needle blight from planting up to about 15 years.

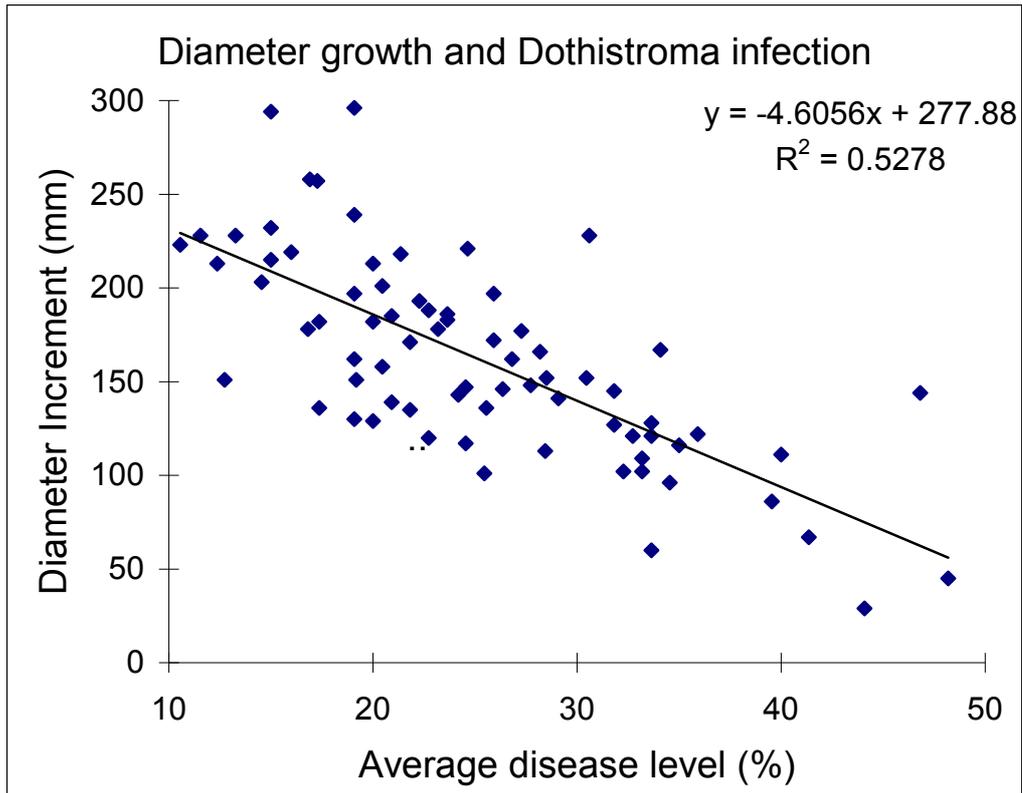


Figure 3.25. Relationship between diameter increment and disease level of Dothistroma needle blight.

Cyclaneusma needle cast has less effect on tree growth than Dothistroma needle blight. This is because Cyclaneusma needle cast affects one-year-old needles rather than the current needles attacked by Dothistroma. The relationship between disease severity and volume loss showed a reduction of the average volume increment of approximately 60% at an average disease severity of 80% (Fig. 3.26) (Bulman, 2001). Trees aged between 6 and 15 years of age are affected.

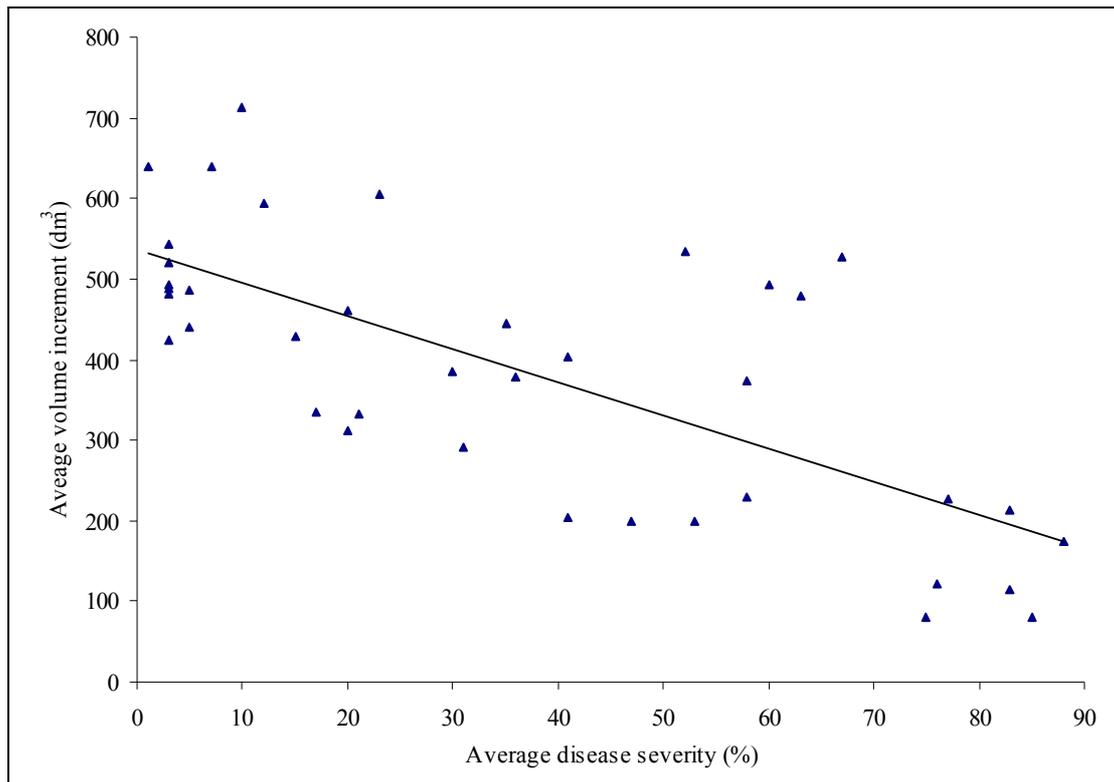


Figure 3.26. The relationship between average volume increment and *Cyclaneusma* needle cast severity.

3.2.7.2 Effect of climate change on pathogens and plantation forestry

3.2.7.2.1 Influence of climate on pathogen abundance

The significant needle cast diseases are affected more by duration and intensity of rainfall than by temperature. For *Dothistroma*, spore germination and successful infection can occur across a wide variety of temperatures. In controlled conditions infection has been shown to occur at 4.4°C (Gadgil, 1968). In field conditions a minimum of 7°C is needed (Gilmour, 1981) but infection is favoured at ranges around 16-20°C when needles are continually moist for over 10 hours (Gadgil, 1974; Forest Research Institute, 1978; Gilmour, 1981). However, even a period of 20 minutes of continuous needle moisture is sufficient for infection to occur (Gadgil, 1977; Forest Research Institute, 1978). An absence of needle wetness has been shown to inhibit production of inoculum rather than germination and penetration (Gadgil, 1977). However, *Dothistroma* can survive in foliage over long dry periods. Gibson *et al.* (1964) showed that in the laboratory *Dothistroma* remained viable in dry needles stored at room temperature for 11 months, and at 30°C survived for 5 months.

Woods *et al.* (2005) demonstrated that an outbreak of *Dothistroma* needle blight in British Columbia during the late 1990s and early 2000s was a result of increased summer rainfall. Figure 3.27 shows the relationship between severity of *Dothistroma* needle blight and summer rainfall from a forest in the central North Island. Disease was assessed in June or July, and is plotted against the preceding November to February rainfall.

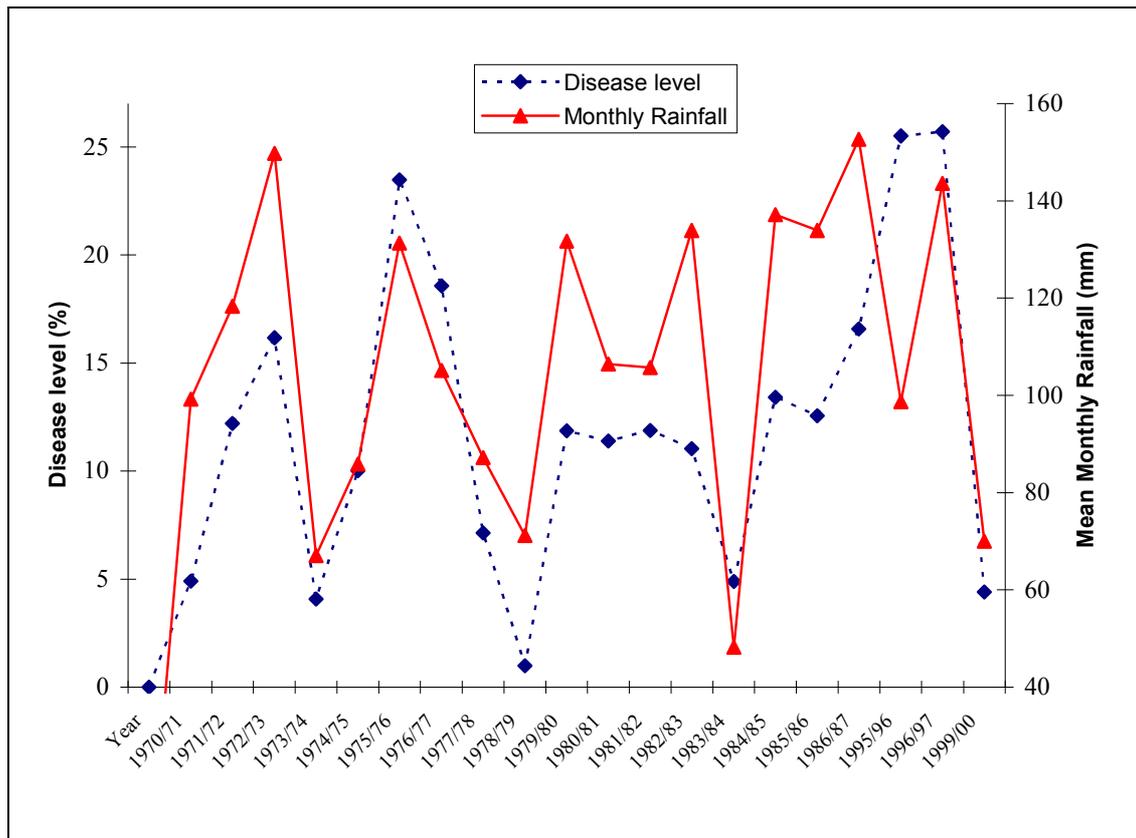


Figure 3.27. Relationship between summer rainfall and *Dothistroma* needle blight disease level.

Cyclaneusma infection occurs in autumn. Although the pathogen is common in the litter layer of *P. radiata* in New Zealand all year round, the needle-cast caused by it is severe only in years with mild wet winters (Gilmour, 1966). The weather in spring and summer when the current foliage is resistant to infection and the spore numbers are generally low cannot affect the incidence of the needle-cast. In autumn or winter, when the needles are susceptible to infection, wet periods of about five hours duration and temperatures above 10°C would provide good conditions for infection. Severe needle-cast in the spring would be expected after an autumn or winter in which such conditions prevailed (Gadgil, 1984). Such temperatures are often reached during autumn and thus temperature does not limit disease severity as much as rainfall.

3.2.7.2.2 Likely effects of climate change on pathogen distribution and growth

All important forest pathogens, with the exception of the recently introduced *Nectria fuckeliana*, are present throughout New Zealand wherever suitable hosts are grown. It is highly likely that *Nectria fuckeliana* will spread throughout the plantation forests of New Zealand, given time. Distribution is highly unlikely to be substantially affected by climate change. As stated above, *Dothistroma* can survive in dry foliage for many months and this pathogen, *Phaeocryptopus gaeumannii*, and *Cyclaneusma* are present in Central Otago and other dry areas in New Zealand. Foliage fungi and other important fungal pathogens of pines in New Zealand could not be eliminated completely by changed climate. They have already shown tolerance for a wide range of climatic conditions.

It is important to differentiate between the organism and the disease it causes. While climate change will not affect the presence of the fungus, it is likely to change the

regional incidence and severity of some fungal diseases. At present, *Cyclaneusma* needle cast is most severe in Northland and East Cape, where autumn rainfall is often plentiful and temperatures are mild. If warming does occur it may be expected that *Cyclaneusma* needle cast will become more damaging in those areas, although this increase in severity may be mitigated to some extent by the projected reductions in annual rainfall within these regions.

At present *Dothistroma* needle blight is not a problem in drier parts of New Zealand and is a rare disease in Hawke's Bay and most of the South Island, apart from Westland. Given that most currently dry areas are likely to experience less rain under climate change *Dothistroma* needle blight is likely to remain rare in these regions.

The incidence of Swiss needle cast, which is the most widespread disease of Douglas-fir (*Pseudotsuga menziesii*) and caused by *Phaeocryptopus gaeumannii*, is likely to increase with climate change. Research across a wide environmental gradient in New Zealand demonstrated a strong positive linear relationship between the abundance of *P. gaeumannii* and minimum temperature during winter (Stone *et al.*, 2007), a result which was consistent with previous findings in western Oregon (Manter *et al.*, 2005). Given that air temperature is projected to increase throughout New Zealand it is likely that there will be a significant increase in the incidence of *P. gaeumannii* and Swiss needle cast within Douglas-fir plantations under climate change.

Although the above generalisations are probable, the further research outlined in 3.2.8 is necessary to accurately determine the effect of climate change on pathogen abundance.

Increase in disease severity will reduce forest growth. For *Dothistroma* needle blight it will also result in increased cost if aerial application of fungicides is required. It is difficult to estimate the precise effect on stand growth because of compensation effects and other interactions. Productivity loss is best estimated by modelling various scenarios and effects.

3.2.8 Further research requirements

3.2.8.1 Determining changes in weed distribution

As a priority, the future potential distributions for all key weed species identified in Table 3.3 should be determined using process-based weed distribution models. Additional potential distributions should be determined for any less important weed species, with regionalised distributions under the criteria that (i) they are strong competitors (e.g. *Acacia* and *Eucalyptus* spp.) and (ii) they are currently ornamental 'sleeper' weeds, which could potentially become strong competitors if they become naturalised and if anticipated climatic changes are favourable for their growth and rapid spread. Defining the risk profile of strong competitors is of most importance, as the replacement of current weeds may not be problematic unless the replacement weed competes more strongly for site resources, than the current suite of weeds occupying the site.

Weeds which could be potentially invasive under climate change should be identified from the areas highlighted in Figures 3.13. Further distribution models should be developed for any potentially problematic weeds in order to ascertain the risk profile of the weeds.

If water use efficiency is found to influence weed growth, ideally, these simulations should also be used to explore how changes in water use efficiency are likely to influence distribution and relative abundance, using the methodology demonstrated for problematic Australian weeds by Kriticos (2008). Following the method demonstrated by Sutherst *et al.* (2007b), future models should account for the influence of interactions between weed species on their potential distributions.

3.2.8.2 Determining weed and tree growth

Sufficient information should be obtained to parameterise process-based population dynamic models and resource competition models such as Vegetation Manager (VMAN).

Parameterisation of these models will require robust information on how weed and tree growth are affected by changes in temperature, day length, water balance, carbon fertilisation and water use efficiency and how these responses may be modified through nutrient feed-backs or biotic interactions. The focus of these experiments should be in determining whether weeds and trees respond differently to climate change, as even slight seasonal variation in growth of trees and weeds over the first few years can have a substantial influence on tree growth (Kimberley and Richardson, 2004). Research needs to be undertaken to determine the response to higher [CO₂] for weeds and trees, and whether weed response may change over successive generations.

The most robust way of parameterising responses to these variables would be to undertake Free-Air CO₂ Enrichment (FACE) experiments with trees, weeds and herbivores to determine how the system responds to climate change. Given the large logistic requirements of FACE experiments, the number of important interactions to be studied and the number of weed species, for which these effects need to be described, it would be necessary to aggregate weed species into functional groupings for which broad responses can be determined.

3.2.8.3 Determining changes in insect distribution and abundance

In order to determine changes in the distribution or abundance of forest insect pests it is essential to have baseline data for comparison. However, currently there are no long-term monitoring programmes for populations of any forest insect in New Zealand, although temporary monitoring data are available for some species. This seriously limits our ability to detect climate change-related effects on insect pests, and it would be advisable to initiate such monitoring programmes as soon as possible, in order to provide baseline data. Suitable target species include *Essigella californica* and *Pseudocoremia suavis*.

Process-based distribution models and population dynamic models should be used to determine how climate change is likely to influence distribution and abundance of potentially damaging insect species such as *P. suavis*, *H. ater*, and *E. californica*.

3.2.8.4 Determining changes in pathogen distribution and abundance

Although there has been considerable research linking climate to pathogens, it is not known how abundance of the major forest pathogens is likely to alter under climate change. Further modelling should therefore be undertaken to model the influence of climate change on Dothistroma needle blight and Cyclaneusma needle cast. Process-based distribution models currently offer the best means of doing this as these models can be used to predict both potential distribution and abundance under climate

change. Further research is also needed to determine the environmental and host response factors which influence the development of *Nectria flute* canker.

It would also be prudent to undertake further research which more fully investigates how climate change is likely to influence diseases of alternative species, such as Douglas-fir, the cypresses and eucalypts. It would be useful to determine how the abundance and distribution of the main diseases of alternative species are likely to respond to climate change, and more fully define the influence their abundance has on host growth. As species selection is an important strategy for mitigating the effects of climate change such information would be invaluable in determining how best to site species, so that risks from disease can be minimised.

4 Development of a model framework

Summary

Objective

Develop an interdisciplinary framework which can be used to describe the direct and indirect effects of climate change on plantation productivity. For this framework, define the linkages and feedbacks between direct and indirect effects on plantation productivity. Using this framework, identify key knowledge gaps and uncertainties, and provide a comprehensive research plan to address these uncertainties.

Key results

The numerous interactions between plantation productivity, climatic variables and secondary effects described in this report demonstrate the complexity of the system and the necessity to model the effects of climate change on plantations in an integrated manner. Until now, there has been no integrated assessment of the likely effect of climate change on New Zealand's plantation estate, and any past assessments have only studied individual aspects of the overall response in isolation. No unifying framework exists that would allow the study of the combined effect of all likely changes, their interactions with each other and any feed-back effects that may modify the initial response to an external perturbation.

We present such a framework which could be used to determine the effects of climate change on plantation productivity. At the core of the framework is a growth model which can account for the ecophysiological effects of climatic changes on plantation productivity. This core model should be linked to sub-models accounting for the influence of additional biotic (weeds, pests, diseases) and abiotic (wind, fire) factors on plantation productivity. Interactive linkages are important as biotic and abiotic stresses not only affect plantation productivity, but ecophysiological factors such as nutrient levels and physiological stresses, and can also affect a stand's susceptibility to pests and diseases. In addition, other factors, such as stand structure, influence both fire and wind risk.

Further research is needed to translate estimates of fire and wind risk into quantitative estimates of the area or wood volume lost to these factors, which can then be incorporated directly into a spatial growth model. This will make it possible to derive overall quantitative estimates of climate change impacts on productivity and wood value under a range of assumptions, scenarios and adaptation strategies.

Under climate change, pests are likely to influence plantation growth primarily through changes in their distribution and to a lesser extent, their growth and abundance. Process-based distribution models could be used to describe such range shifts for the most important pests. Population dynamic models can be implemented to determine the effect of climate change on the abundance and growth of insects and weeds. Abundance of pathogens can be determined using a range of models, from simple climatic models through to process-based distribution models. Once the effect of climate change on weed growth has been estimated, consequent effects on tree growth can be estimated using resource competition models. For insects and pathogens, impacts of changes in their abundance on tree growth will need to be determined through the development of functions linking their abundance to plantation tree growth.

Given the diversity of potential end users of this information, the framework needs to aim for a flexible approach that can deliver relevant information at multiple scales to meet different user requirements. As the framework needs to operate at a wide range of scales, the growth model should be implemented spatially and cover New Zealand at a reasonably fine resolution so that it can be used to quantify the effects of climate change at local, regional and national levels. Although the proposed model framework provides a means of integrating the complexity of climate change on plantations, it should ideally remain modular in nature so that individual components can be studied independently, or existing formulations readily exchanged with alternative ones.

Future work

The key results and the proposed framework described above have identified a possible approach to quantify the overall effects of climate change on New Zealand's production forests in an integrative way. Key knowledge gaps have also been identified. Future work should aim to fill these gaps, quantify the potential overall impact and make it possible to develop adaptation measures to reduce the vulnerability of our plantation forests in the future.

The following tasks should be carried out as a matter of priority:

1. Test and quantify a process-based model that includes all relevant feed-back processes and has the capability to model the CO₂ and climate change response for New Zealand. Test data sets are available that could be used to run such a model under current conditions. Once the model has been run successfully under current conditions, it could be run under various climate change scenarios to quantify the effect of the expected climatic changes on New Zealand's radiata pine plantations.
2. Link the process-based growth model with modules describing the important biotic and abiotic stress factors for an integrated overall assessment of climate change impacts on the whole forest system.
3. Model changes in evapotranspiration based on the Penman-Monteith equation to assess likely changes in the incidence of drought for plantation tree species under climate change across New Zealand, and compare this assessment with previous methods that have been used for climate change assessments.
4. Little is known about the physiological response of other potential plantation tree species in New Zealand. Information should be gathered about their growth responses under different climates and from climate change experiments to identify the future potential of such species, and how their suitability might be affected by climatic changes.
5. There is still some uncertainty about the magnitude (and for some variables and locations even the direction) of climate change for New Zealand over the future decades. This uncertainty limits our ability to assess the impact of climatic changes and needs to be properly accounted for to be able to make a comprehensive assessment of the risks and opportunities associated with climate change. This uncertainty is most pronounced for an assessment of likely changes for the abiotic stressors fire and wind. There is significant potential to improve estimates of climate change impacts on wind and fire risk for New Zealand through updated analyses incorporating improved methods and understanding. In particular, better projections of wind and humidity

changes (in addition to temperature and precipitation) could be obtained through use of nested Regional Climate Models (as opposed to previous GCM downscaling).

6. Baseline information on the level of wind damage currently occurring in New Zealand's plantations needs to be recorded. The probabilistic wind damage model used to calculate the risk of damage needs further refinement and testing, including updating of the wind speed recurrence functions to reflect projected future changes in the extreme wind climate. Additional research is required to link changes in the risk of damage to likely levels of actual loss.
7. Establishment of improved fire reporting systems for recording fire occurrence and losses. Better information on current fire losses is essential so that analyses can be undertaken to identify relationships between weather and fire danger factors and fire numbers, area burned and fire suppression costs for use in estimating fire impacts with climate change.
8. In order to determine changes in the distribution or abundance of pests, it is essential to have baseline data for comparison. However, long term monitoring programmes are not currently undertaken for many key insects, weeds or pathogens. This seriously limits our ability to detect climate change-related effects on important pests, and it would be advisable to initiate such monitoring programmes as soon as possible, in order to collect key baseline data.
9. It is important to identify pests (weeds, insects, pathogens) that might pose new threats to New Zealand's plantation forests under climate change. Any potentially problematic pests should be identified from the area highlighted in this report which could currently harbour potential invaders under climate change. Once identified, further research should be undertaken using process-based distribution models to determine their potential distribution within New Zealand under climate change. Once the risk profile for these species has been ascertained this will allow a more targeted approach towards preventing their entry into the country or the development of control strategies if they are already present in the country as 'sleeper' populations.
10. Identify pests currently present in New Zealand that might pose future problems under climate change. For identified risk species, confirm their risk potential through manipulative experiments, which consider interactions between species from different trophic levels (e.g. weeds, trees and biocontrol agents), and models which describe how they are likely to impact on plantation productivity under climate change. If these methods indicate they are high-risk species, devise control measures to eradicate their current populations if feasible, or at least slow their spread under more favourable climatic conditions.
11. Implement the framework spatially and combine the multiple interactions for a range of case studies, in different regions.

4.1 Background

The numerous interactions between forest plantation productivity, climatic variables and secondary effects demonstrate the complexity of the system, and the necessity to model the effects of climate change on plantations in an integrative manner. The change in physiological responses of trees with climatic changes will result in different growth and productivity compared to the present time. There will be changes in risks associated with biotic stressors, such as pests and weeds and the frequency and intensity of abiotic stressors like wind and fire.

If these direct and indirect effects of climate change can be integrated into a interactive modelling framework which describes their overall influence on growth, it will support several important initiatives. It will firstly provide an initial understanding of the likely directions and magnitude of change of the forest system to climate change. It will also enable a sensitivity analysis to be undertaken to determine the components of global change that are likely to have the greatest impact on plantation forests, and how these impacts can be modified through various adaptation options. The identification of important feedbacks, components, and assumptions will assist in prioritising future research in terms of the importance on overall impact on plantations.

A useful framework for understanding how vulnerable plantation forests are to climate changes is given by,

$$\text{Vulnerability} = \text{Impact} \times (1 - \text{Adaptive capacity}) \quad [4.1]$$

where adaptive capacity refers to the adaptability of management which in turn depends on the adaptation technologies, cultural tools, and the infrastructure and resources available to implement appropriate technologies.

Impacts are determined as the product of a change in exposure of a system (in this case plantation forestry) to climate and secondary risk factors and the sensitivity of the system to that change, as,

$$\text{Impact} = \text{Exposure} \times \text{Sensitivity} \quad [4.2]$$

Following this framework, determination of the effects of climate change on plantations requires an integrated modelling system which can quantify the impact of climate change on plantations through accounting for both direct influences and indirect influences. The model should also be flexible enough to be able to run different scenarios to determine how adaptation can be used to mitigate impacts. After accounting for the effects of adaptation, vulnerability of the plantations to climate change can be determined. Although such a model should be integrated it would also be a key advantage if it is modular in nature, so that once key inputs and linkages are identified the model can be simplified in order to reduce the required parameterisation.

In the following, we lay out the key requirements and components of such a modelling framework. We try to highlight the important interactions that need to be included and suggest a possible approach towards developing such a framework that could be implemented through future work.

4.2 End-user requirements

There are numerous groups and individuals that require a better understanding of likely climate change effects on New Zealand's forests. They are likely to encompass a wide range of national and regional groups including government departments, regional councils, research organisations, forestry companies and local growers. At the national level, the framework needs to provide useful input to national planning for adaptation options and for modifying biosecurity protocols. Forest growers, local or regional authorities and the forest industry as a whole also need to consider any possible climate change impacts in their specific decision making about future planting of forests and the possible modification of any of their management methods.

These various end-users will have different requirements which will vary in terms of spatio-temporal scale and the required level of detail of likely impacts. A broad scale approach will allow policy makers to make decisions at the national and regional scale. Local end-users like forest owners will require finer-scale and detailed localised information for their decision making.

Given such a diversity of users, the framework needs to aim for a flexible approach that can deliver relevant information at multiple scales to meet different end-user requirements. It is important for end-users to gain an appreciation of the possible climate change impacts for their enterprise from an environmental and economic perspective. Consequently, the framework needs to indicate how changes are likely to affect the economics of plantation forestry and if they are likely to have any additional environmental consequences.

4.3 Development of an integrated modelling system

4.3.1 Growth model

The proposed modelling framework for assessing impacts is illustrated in Figure 4.1. At the core of the framework is a growth model which can account for the effects of climatic changes on plantation productivity. The model needs to be able to respond to climate and include the numerous ecophysiological feedbacks and interactions between climate and productivity that together determine the overall response to future changes in the climate. Using such a model over the wide range of conditions currently experienced in New Zealand will highlight the climatic sensitivity of specific forests or locations over the range of changes that we may encounter. At least one model with such attributes is currently available in New Zealand and could be tested in the near future (Kirschbaum, 1999b).

Ideally, as the framework needs to operate at a wide range of scales, the growth model should be implemented spatially and cover New Zealand at a reasonably fine resolution so that it can be used to quantify the effects of climate change at local, regional and national levels. This requires the linkage to climatic, soil and other environmental data at an equally fine temporal and spatial scale. Development of a spatial model will also enable a thorough analysis of the interaction of biotic and abiotic factors at the landscape level. Although these interactions add considerable complexity to the model it should be stressed that the intention here is not to incorporate all interactions, but rather to develop a modular system which allows the importance of each interaction to be tested and if found to be important included in the model.

The growth model should also be linked through to an economic analysis to allow users to determine the likely influence of climate change on measures of plantation value, such as net present value, and to what extent adaptation may avoid any adverse impacts on plantation value. There are a number of previous studies that have linked growth models to economic analyses (e.g. Watt *et al.*, submitted c). They illustrate that it is feasible to integrate the various sub-models for an analysis of the trade offs between input costs and output value.



Figure 4.1. Proposed modelling framework for assessing the impacts of climate change.

4.3.2 Integration of abiotic and biotic effects

Various abiotic and biotic factors are likely to be affected by climatic changes with both direct effects on plantation productivity and indirect effects through interactions with other factors. These abiotic and biotic factors currently have a negative influence on plantation productivity. With the possible exception of pathogens, the impact of these abiotic and biotic factors is likely to be aggravated by climatic changes and these likely adverse changes are also likely to be compounded by further synergistic effects of different factors with each other.

Ideally, models describing wind and fire risk should be interactively incorporated into the overall framework to describe the effects of abiotic factors on plantations. Stand parameters, derived from the growth model will be required as inputs for the probabilistic wind risk model, as wind risk is sensitive to stand structure and growth. Quantifying the overall risk of wind damage also requires information on topography

and soil type which could be readily accessed through digital terrain models and digital soil maps, respectively, in a spatial model.

Similarly, as fire intensity and the area burned are strongly related to vegetation structure, species and fuel loadings, including contributions from weeds and standing biomass, a fire risk model should also be coupled to the growth model and to models describing weed growth and species. Functions linking these fire risk models to the area damaged by wind and insects are also important as dead, downed woody fuels potentially leads to higher risk of ignitions, more intense fires and greater areas burned.

Further research will be needed to translate estimates of fire and wind risk into estimates of the quantitative area or wood volume lost to these factors. In the case of fire risk, for example, relationships between current fire occurrence and fire danger are required before estimates of future fire numbers, area burned and fire suppression costs can be derived from projected changes in future fire danger. Once appropriate functions have been developed, they will enable loss estimates to be directly incorporated into the spatial growth model. In turn, this will make it possible to derive quantitative estimates of climate change impacts on productivity and wood value under a range of assumptions, scenarios and adaptation strategies.

Under climate change, pests are likely to influence plantation growth primarily through changes in the pests' distribution and to a lesser extent, through changes in their growth and abundance. Process-based distribution models could be used to describe such range shifts for the most important pests. Once changes in distribution have been ascertained population dynamic models can be implemented to determine how climate change influences the abundance and growth of insects and weeds. There are a number of methods that could be used to determine pathogen abundance which range from simple climatic models (e.g. Stone *et al.*, 2007), through to more complex process-based functions typically embodied in process-based distribution models (e.g. Yonow *et al.*, 2004 Watt *et al.*, (submitted a)). Population dynamic models can also be used to determine how biocontrol agents and weeds interact under climate change.

Once the effect of climate change on weed growth has been estimated, consequent effects on tree growth can be estimated using resource competition models. As resource competition models are well developed for a wide range of current weed species (Watt *et al.*, submitted b) it should be feasible to use these models to modify productivity estimates within the growth model. For insects and pathogens, impacts of changes in their abundance on tree growth will need to be determined through the development of functions linking their abundance to plantation tree growth. These functions should be directly linked to the growth model, ideally incorporating mechanisms that describe tree compensation following insect herbivory.

There are a number of interactions between both abiotic and biotic factors which may have synergistic effects on plantations greater than the combined effect of each single factor. Most of these are likely to negatively influence plantation productivity. Although these are not discussed in detail here the proposed framework is flexible enough to accommodate functions describing such interactions.

4.3.3 Model implementation

Although the proposed model framework provides a means of integrating the complex factors and processes which govern how future climate change will affect plantations, it should also be modular in nature, so that it can be simplified as much as possible. If sensitivity analyses indicate that an effect does not markedly impact on plantations,

then this linkage can be eliminated from the overall model. For example, process-based distribution models may indicate that weed composition will change considerably through climate change. However if the new suite of weeds do not more markedly compete with trees for site resources (as defined by the resource competition model), than the existing weeds, then a number of linkages in the overall model can be eliminated. Such simplification is very important as it will greatly reduce the overall level of parameterisation required.

Many of the described sub-models which underpin the overall modelling framework are currently well developed. Mechanistic stand growth models that have the necessary feedback and responses to the changing climate are available. Process-based distribution models are currently extensively used to model how climate change influences pest distribution. Although population dynamic models have not been widely used, the necessary framework exists to determine how weeds and insect populations are likely to be affected and interact under climate change. Resource competition models describing how weed competition influences tree growth are well developed for New Zealand (Watt *et al.*, submitted c). The current model framework for abiotic risk factors is reasonably robust for both fire and wind.

4.4 Further research

The objectives described previously, including the modelling framework developed in this chapter, highlight a number of gaps in our current knowledge. Development of a robust understanding of how climate change will impact productivity will require research which addresses these issues, which are outlined in the following section.

4.4.1 Physiological responses of forests

4.4.1.1 Potential photosynthetic down-regulation in plants grown in high CO₂

Short-term plant responses to increasing CO₂ are well established, but most studies show a decline in responsiveness after some period of time (down-regulation). Some of that down-regulation has been identified as an experimental artefact of growing plants in small pots where the restricted root volume prevents plants from making full use of the additional available carbon gained from increases in CO₂. Field-grown plants show much less of the down-regulation response.

It is also clear that plants require access to nutrients to be able to make full use of additional available carbon. Hence, plants grown with access to limited nutrient supply tend to show less of a response to CO₂ than plants grown with unlimited nutrient supply. In principle, this interaction can be quantified and included in any assessment of an overall plant response to increasing CO₂. There are, however, some remaining uncertainties about the processes operating in the soil, which are discussed further below.

Finally, it is unclear whether there is some additional photosynthetic down-regulation after pot size and nutrient supply have been explicitly accounted for. Whether or not there is, however, is of great importance for the overall responsiveness of forests to elevated CO₂.

4.4.1.2 CO₂ response of older trees

For obvious logistic reasons, most studies of elevated CO₂ responses of trees have been conducted on young and small trees. The limited available evidence suggests

that older trees might possibly be less responsive to increasing CO₂ than younger trees. This has been linked to the more general notion that more actively-growing plants might be more responsive to increasing CO₂ than plants with lower growth rates.

At this stage, there is still insufficient information to settle that question. Because of the small number of observations, a number of factors are invariably confounded with each other. For instance, while in some compilations of relevant observations, there is an indication that older trees might be less responsive to high CO₂ than younger trees, the older trees are also predominantly conifers whereas the younger trees include a greater proportion of broad-leaved species. Therefore, two possible explanations for the observations from these data sets would be that either older trees are less responsive than younger ones, with different species responding similarly, or that conifers are less responsive than broad-leaved ones, with age playing no role. A third possibility might be that neither age nor species play a role, but that some other factor in this small number of experiments accounted for the difference.

Under natural conditions, the response of older trees is particularly pertinent, however most available information is based on the study of younger trees or even only non-woody species. Trends appear to indicate a lesser responsiveness of older trees, but there are too few observations, and too much confounding with other factors to be able to say that with any confidence.

4.4.1.3 Soil carbon / nutrient interactions

In an intact ecosystem, many climate change responses are determined or at least modified not by the direct physiological response of plants to changed environmental conditions, but by processes and feed-backs through the dynamics of soil organic matter. These processes can act to both enhance or inhibit plant productivity. For example, the faster turn-over of organic matter decomposition and nutrient mineralisation under warmer conditions will increase growth while the growth response to elevated CO₂ may be limited through the build-up of soil organic matter that acts to immobilise nutrients in the soil.

In principle, these soil processes are easy to understand and link to plant physiological processes. It has been difficult to verify many of these postulated processes in experimental work. This has been partly due to significant methodological challenges as relevant feed-back processes often constitute only a small fraction of total stocks and turn-over rates, and many soil processes are inherently difficult to measure. However, the paucity of information on these processes is also attributable to the fact that intensive studies of this nature have only recently been undertaken.

Consequently, there is still only poor knowledge and certainty of these key processes and feed-backs. Figure 2.1 indicated the extent by which the CO₂ responsiveness of forests can be affected by the initial environmental conditions, many of which are modified by soils processes. While the logic of these interactions is compelling, they are still only based on a theoretical assessment. While the underlying relationships are partly verified through some experimental observations, these observations are insufficient and provide incomplete evidence for these important feed-back processes.

4.4.1.4 Changes in the rate of transpiration

On current dryland areas in New Zealand, which are subject to water deficits there is considerable uncertainty as to whether climate change will ameliorate or exacerbate

the current water balance. Whether water shortages will develop depends on the balance of changes in precipitation and evapotranspiration. In addition to the well-recognised uncertainties inherent in climate-changes scenarios, there is also a difficulty of translating changes in primary drivers of evapotranspiration, such as temperature, vapour pressure deficit, etc., into changes in the actual magnitude of evapotranspiration. A full assessment of the likely changes in evapotranspiration needs to be done for different plant types and also needs to include any physiological adjustments in stomatal conductance.

As there is well-developed theory to derive evapotranspiration from its primary drivers (the Penman-Monteith equation) an assessment of likely changes in evapotranspiration and, ultimately, water balance for different crops across New Zealand is quite feasible. However to date such an assessment has not yet been done.

4.4.1.5 Responses to temperature

Physiological responses to temperature are fairly well understood, and growth models are successfully used to predict the growth of *P. radiata* across New Zealand under current conditions. Climate change, however, will introduce additional complexities primarily related to the dynamics of soil organic matter. This means that a forest for which the temperature changes to some new level will experience a different set of conditions from a forest that has long been exposed to such conditions. While it is possible to model the response to such changed conditions, there are only limited experimental data to test and verify these modelled responses.

4.4.1.6 Response of lesser-known plantation species

The vast majority of the plantation estate in New Zealand consists of *P. radiata*, and there is good understanding of the physiological response of the species to a range of environmental conditions. There is less known about the physiological responses of minor plantation species in New Zealand, such as Douglas-fir (*Pseudotsuga menziesii*), cypresses and some eucalypt species. If the climate warms significantly, forestry might also shift to more sub-tropical species, such as Caribbean pine (*Pinus caribaea*), and very little is known of the growth response of this species in New Zealand.

4.4.2 Abiotic impacts

4.4.2.1 Fire effects

The danger of fire is affected by both natural weather factors and anthropogenic factors. The natural factors are difficult to predict as fire responds to extreme weather events when high wind and temperature and low humidity follow a period without rain. The likelihood of the combination of such weather conditions is difficult to deduce from a sequence of changes in average climatic conditions that can be anticipated with somewhat greater certainty. Confidence in the prediction of weather conditions that increase fire danger is therefore lower than the confidence in the prediction of change in average conditions. However, there is significant potential for current estimates of climate change impacts on fire and wind risk for New Zealand to be improved by updating analyses to incorporate advances in climate change prediction methods and understanding. The use of Regional Climate models (RCMs) to provide an estimate of changes in local weather conditions, for example, represents a significant advancement over the GCM downscaling technique that has been used to date, and

could provide much improved estimates of wind and humidity changes as well as temperature and precipitation.

Anthropogenic factors are important as most fires are started by humans either accidentally or through malicious acts. On the other hand, the danger of fires can be largely ameliorated by management, both in the fire prevention stage through fuel management and the provision of fire breaks, and through active fire suppression if fires do occur. Future fire danger is therefore as much affected by any change in human behaviour or policies and resources towards fire management activities such as fuels management and fire suppression as it is by changes in climatic drivers. This adds a further element of uncertainty.

As highlighted previously, estimation of fire risk is also dependent on the availability of data on fire occurrence, including records of fire numbers, area burned and fire suppression costs and damages. While improvements are required to national fire reporting systems in New Zealand to provide the quality of data required for such analyses, relationships could potentially be derived between current fire occurrence and fire danger that could be used as the basis for estimates of future fire numbers, area burned and fire suppression costs associated with projected changes in future fire danger.

4.4.2.2 Strong wind effects

As for fire weather, the risk of catastrophic wind damage is strongly related to the relatively rare occurrence of severe weather events, such as the 1975 storm in Canterbury and Cyclone Bola. Changes in the magnitude and frequency of such extreme weather conditions under future climate change scenarios are more difficult to predict than changes in average climatic conditions. Therefore, confidence in the prediction of high-wind events is lower than the confidence in the prediction of change in average conditions, which in turn is lower than for other climatic elements, particularly temperature and rainfall. Future research should therefore focus on developing more robust predictions of the future wind climate for the range of potential climate change scenarios. As noted in the preceding section, the use of RCMs should lead to improvements in the quality of predictions about the future wind climate.

The vulnerability of forests to wind damage is a function of forest layout and stand structure, both of which can be influenced by management. This vulnerability can be quantified using a mechanistic model and expressed as the threshold wind speed required to damage a stand. This model requires continual refinement and validation against data from actual storm events. At present, the model can only predict the probability that a stand will be damaged, rather than the actual losses which occur. By comparing model predictions of critical wind speed against actual wind speeds which occur during damaging storms and the actual amount of damage which occurs, it may be possible to relate the level of damage to the predicted risk. It is also recommended that either one or more monitoring areas be established or that data from permanent sample plots are analysed, so that levels of attritional damage can be quantified. In the future, it is recommended that better information is kept on wind damage losses under current conditions so that any changes over time can be assessed. By combining information on catastrophic and attritional damage, the output from growth models can be adjusted to reflect losses which are likely to occur. Such information can also be used for reporting under the FAO Global Forest Resources Assessment and the UNFCCC.

4.4.3 Biotic impacts

4.4.3.1 General

In order to determine changes in the distribution or abundance of pests, it is essential to have baseline data for comparison. However, long term monitoring programmes are not currently undertaken for many key insects, weeds or pathogens. This seriously limits our ability to detect climate-change related effects on important pests, and it would be advisable to initiate such monitoring programmes as soon as possible, in order to provide baseline data.

Areas which have been identified as regions from which future pest invasions may originate (see Fig. 3.13) should be examined for potentially invasive pests, which could be problematic in the future, or may already be established in New Zealand as 'sleeper' species. Once identified further research should be undertaken to determine the potential distribution of these species under climate change in New Zealand. If these pose a future threat the necessary measures should be taken to ensure that they do not arrive in New Zealand, or if they are already in New Zealand, the population is eliminated before climate change allows their further spread.

4.4.3.2 Weeds

The future prevalence of weed problems is likely to be related to the future distribution, growth and competitive strength of the current problem weeds, and the potential of plants that are either currently present in New Zealand or could enter the country to become problem weeds, especially under changed climatic conditions.

As a priority, the future potential distributions for all key weed species identified in Table 3.3 should be determined using process-based weed distribution models. Additional potential distributions should be determined for any less important weed species, with regionalised distributions under the criteria that (i) they are strong competitors (e.g. *Acacia* and *Eucalyptus* spp.) and (ii) they are currently ornamental 'sleeper' weeds, which could potentially become strong competitors if they become naturalised and if anticipated climatic changes are favourable for their growth and rapid spread.

There is a significant potential danger that current 'sleeper' weed populations could develop into major problem weeds under changed climatic conditions. A number of species have been identified that could constitute such a danger. There is an urgent need to complete the list of potential problem species and to examine the risk of these developing into problem weeds using process-based distribution models. For species with a confirmed risk potential, there is a further urgent need to devise control strategies to eradicate 'sleeper' populations if feasible or to develop other means to prevent, or at least, slow the development of these species into weed problems.

Sufficient information should be obtained to parameterise process-based population dynamic models and resource competition models such as Vegetation Manager (VMAN), for important weed species or weed species which are likely to be strong competitors in the future. These models will require details of how weeds respond to temperature, day length, water balance, carbon fertilisation and water use efficiency and how these responses may be modified through nutrient feed-backs or biotic interactions. One means to obtain much of this information would be to undertake Free-Air CO₂ Enrichment (FACE) experiments with trees, weeds and herbivores to determine how the system responds to climate change. Given the large logistic

requirements of FACE experiments, the number of important interactions to be studied and the number of weed species, for which these effects need to be described, it would be necessary to aggregate weed species into functional groupings for which broad responses can be determined.

4.4.3.3 Insects

About 150 different insect pests have been recorded in New Zealand's pine forests. Although several of these insects are economically important, most do not currently constitute major problems. It is not known to what extent the currently present, but unproblematic, species are hindered by climatic conditions that may change in the future or whether their limited impact is due to biotic factors that might be less likely to change into the future. At this stage, only one insect species has been identified for which it seems likely that its pest potential is linked to environmental limitations, the Monterey pine aphid, *Essigella californica*. It is thought that its pest potential will increase with climatic changes, especially if this results in increases in temperature and drought frequency or severity. The same could be true for other insects, especially sap suckers and also for bark beetles such as *Hylastes ater* and defoliators such as *Pseudocoremia suavis*. Process-based distribution models and population dynamic models should be used to determine how climate change is likely to influence distribution and abundance of these potentially damaging insect species.

4.4.3.4 Pathogens

Research should be undertaken to determine how climate change is likely to affect the abundance and outbreak risk for pathogens. Process-based distribution models may provide a useful means of addressing both these issues, for different regions in New Zealand. However, additional models which account for particular importance of rainfall seasonality on abundance of important diseases such as *Cyclaneusma* needle cast should also be developed.

As most research has been directed at determining how different pathogens influence *P. radiata* the distribution, development, and influence on the host of the main growth limiting diseases is reasonably well defined. However it would be prudent to undertake further research which more fully investigates diseases of alternative species, such as Douglas-fir, the cypresses and eucalypts. It would be useful to determine how the abundance and distribution of the main diseases of these species are likely to respond to climate change, and more fully define the influence of abundance on host growth. As species selection is an important strategy for reducing the impact of climate change such information would be invaluable in determining how best to site species, so that risks from disease can be minimised.

4.4.4 Integrated impacts

This report has highlighted that many physiological and additional biotic and abiotic stress factors all interact strongly with each other, with negative effects often synergistically building on each other. While the study of individual effects and processes is relatively simple, the study of multi-process interactions is much more difficult. Yet, it is these combined effects of multiple interactions that are important in driving ultimate system responses. Without trying to capture these multiple, interacting processes, any impact assessment may omit one of the key drivers of future change. Important interactive effects should therefore be identified and research strategies should be devised which allow the influence of these effects on productivity to be quantified.

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