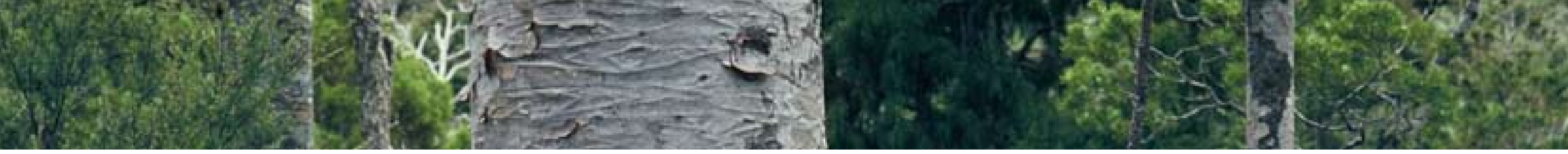

Potential effects of climate change on New Zealand's terrestrial biodiversity and policy recommendations for mitigation, adaptation and research

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Potential effects of climate change on New Zealand's terrestrial biodiversity and policy recommendations for mitigation, adaptation and research

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ABSTRACT

New Zealand's climate is changing. Mean annual temperatures have warmed by almost 1°C (with the most warming in winter) since AD 1900, but with substantial year-to-year fluctuations. Frost frequency has declined through most of the country since the middle of the 20th century, although only a small net warming (c. 0.3°C) has occurred. This study used New Zealand and international literature to review the potential effects of increasing climate change and atmospheric carbon dioxide (CO_2) concentration on indigenous terrestrial and freshwater biodiversity in New Zealand. It discusses the implications of climate change for the Department of Conservation (DOC) and for national policy, and recommends key areas for management and research, and priorities for policy development. The main recommendations for management are provision for managed adaptation to landward migration of coastal ecosystems with sea-level rise; establishing minimum flows in and riparian vegetation around streams and springs; monitoring key ecosystems (coastal, dry lowland and alpine); and surveillance of biotic distributions, particularly exotic weeds and invertebrates with known potential for spread under a warming climate. It is not considered necessary to have any active planning for climate-driven species range changes or fluctuations in abundance. While during the next 50 years climate change itself is a significant risk to biodiversity, in the short term the risks associated with combating climate change (through carbon sequestration, carbon-neutral energy development, irrigation and land use intensification) are greater. Exotic forestry for carbon capture, and more hydroelectric installations and water abstraction, carry the greatest potential risk to biodiversity. Therefore, careful consideration must be given to minimise these; in particular, to wilding spread from plantations and disruption of rivers used for hydroelectric power generation. Key research priorities are establishing effective long-term monitoring sites and obtaining a better understanding of biodiversity-climate linkages.

Keywords: climate change, biodiversity, New Zealand, greenhouse warming, conservation, management

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1. Introduction

Nearly every aspect of terrestrial ecosystem function in New Zealand will be affected by climate change and increasing concentrations of atmospheric CO_2. Further, alteration of the physical and biological characteristics of the ocean surface around New Zealand—through warming and shifting wind patterns—will affect marine-dependent biota. Rising sea levels, primarily responding to warming of the upper ocean and loss of ice from the polar ice caps, will impact coastal estuarine and terrestrial communities. It is clear that change is inevitable. Therefore, the focus has to shift onto the questions: how significant will the consequences of climate change be for New Zealand biodiversity, and what might be done to adapt to or ameliorate these changes?

New Zealand must act. We are signatories to a number of international agreements that oblige this country to make efforts to preserve its biodiversity against threats. Most important from the perspective of climate change are the Convention on Biological Diversity and the United Nations Framework Convention on Climate Change (see New Zealand's Fourth National Communication under the United Nations Framework Convention on Climate Change 2006).

As the agency responsible for managing nearly all land reserved for conservation purposes, the Department of Conservation (DOC) is responsible for monitoring, assessing, planning and reacting to current and potential climate change threats. As stated in the 2006 review of the New Zealand biodiversity strategy:

> *National and international discussions and actions on climate change issues have grown considerably since the Strategy was being developed. Consideration of climate change impacts on biodiversity, both indigenous and introduced, is absent from the Strategy and has received little consideration in the development of New Zealand's climate change policy and adaptation responses. The possible consequences of climate change for indigenous and valued introduced biodiversity are profound. This applies to all levels of biodiversity—genes, species and ecosystems—and to productive landscapes as well as indigenous ecosystems. The implications of climate change for biodiversity justify much greater consideration of the policy implications, research needs and operational management responses than they have received so far.* (Green & Clarkson 2005: 42–43)

There has been very little substantive review of potential and current impacts of climate change on New Zealand biodiversity. Brief overviews have been compiled at intervals (McGlone 1988; Hay 1990; Mitchell & Williams 1996; McFadgen 2001; Hennessy et al. 2007), as have more detailed discussions of impacts on some ecosystems (e.g. Halloy & Mark 2003). The only major attempt to review current changes and potential impacts is a report for the Ministry for the Environment (McGlone 2001). Since that report was written, projections for future climate change have improved significantly, and the amount and sophistication of scientific research published on climate and biodiversity has dramatically increased globally and locally. A further and more comprehensive review is now needed.

Over the past century, global climate has warmed by at least 0.7°C (IPCC 2007). Temperature change has not followed a steady course—there have been prolonged periods of cooling or stasis as well as sustained warmings. However, despite this short, variable record of climatic warming, global biodiversity responses have been well documented and the vast majority are consistent with warming (Parmesan 2006). On the other hand, few biodiversity changes in New Zealand have been ascribed to climate change. Hennessy et al. (2007) reported very little in the way of current changes in New Zealand and suggested few potential impacts. In part, this is because of a relative lack of long-term biodiversity records, which are critical for distinguishing episodic or stochastic events from climate-related trends. Moreover, the New Zealand climate has warmed little over the last 40 years, although frosts are less frequent in many places. For this reason, indigenous biodiversity records beginning around 1970 can be expected to show little change. However, even in the agricultural sector, where there are abundant, accurate long-term records, few climate-related trends have been commented on. A southward shift in subtropical pasture species may be an indicator of agricultural change, as may an increased frequency of warmer winters in recent decades. It is possible that a recorded halving of the planted area in kiwifruit in Northland over the last 6 years could be at least partly attributable to a warming climate leading to reduced productivity (Kenny 2001).

New Zealand researchers have used modelling approaches to project large losses of indigenous biodiversity from alpine areas (Halloy & Mark 2003) and substantial disequilibrium for forests (Leathwick et al. 1996). However, projections made on the basis of biodiversity models usually suggest large changes (e.g. Malcolm et al. 2006). A case could also be argued that New Zealand's buffered, variable oceanic climate will permit only a modest (albeit still significant) degree of biodiversity loss under moderate warming scenarios relative to more continental and northern parts of the globe.

Doubt therefore remains whether, in New Zealand's oceanic, variable climate, sufficient climate change has occurred to induce a significant number of biodiversity phase shifts, such as those when a species population becomes permanently more or less abundant, undergoes a geographical range shift, or the like. Similarly, we are uncertain as to the degree of climate change needed to induce such permanent change. Assessing the most likely biodiversity outcome in the face of uncertainty is the major task of this review.

This report reviews the past, current and potential future impacts of climate, increasing atmospheric CO_2 concentration, and climate-related anthropogenic activities on the biodiversity of New Zealand, and makes policy and research recommendations on the basis of these findings.

2. Recent and projected climate change in New Zealand

2.1 RECENT CHANGE

The cold mean annual temperatures experienced before and at the turn of the 19th century set a low departure point for temperature rise, which globally has averaged around 0.9ºC from about AD 1910. Examination of any local temperature-change record will show many departures from the trend line, and extended periods of several or more years when warming has been either much faster or slower than the long-term average. Globally, this effect is striking, with a period of rapid warming between AD 1912 and 1944, static or declining temperatures until AD 1980, followed by a relatively steady increase of 0.6ºC (Hansen et al. 2006). It is important to note that the New Zealand mean temperature change has been close to the global average, but that the timing of increases has differed.

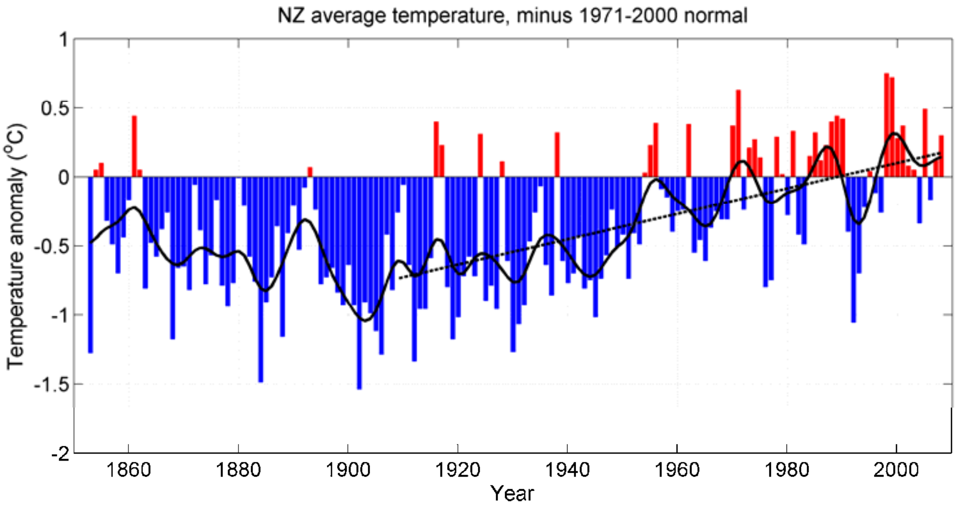
Recent (since AD 1860) changes in New Zealand's climate are summarised on a National Institute of Water and Atmospheric Research (NIWA) web page (www.niwa.co.nz/our-science/climate/information-and-resources/clivar/pastclimate, viewed 24 August 2010) and in a guidance manual for local authorities (Mullan et al. 2008). Because New Zealand is a topographically complex landmass, there are many local variations and some areas have experienced cooling for several years or more at a time. While it might be expected that New Zealand would be buffered by its oceanic setting, this seems not to be the case with regard to warming. Even oceanic island groups, such as the Chatham Islands, have experienced very strong warming over the 20th century (Mullan et al. 2005). New Zealand as a whole is already experiencing significant environmental change directly related to climate, including:

- Increasing temperatures
- Reduced frost frequency over much of the country
- Retreating South Island glaciers and snowlines
- Reduced alpine snow mass
- Rising sea level (estimated rise of 0.16 m during the 20th century)

Mean temperatures have warmed strongly since 1900 (+0.92°C from 1908 to 2007), with minimum temperatures increasing at nearly four times the rate of maximum temperatures (0.4°C v. 0.1°C per decade, respectively, between 1951 and 1998; Salinger & Griffiths 2001; see also Fig. 1). About 0.7°C of the long-term New Zealand trend is accounted for by a sustained warming in annual temperatures between 1945 and 1956. Very little change in mean temperatures (0.06°C per decade) has occurred since 1970, and the trend is not significant. Frost frequency has decreased markedly in many areas (e.g. Marlborough and Canterbury by 20–25% since 1970) but has risen slightly in few areas (Mullen et al. 2008). Diurnal temperature range and sunshine (–0.6% per decade) have decreased on average.

Precipitation changes have been complex and regionalised—a result of fluctuations in the major circulation indices: Southern Oscillation Index (SOI), Pacific Decadal Oscillation (PDO) and Southern Annular Mode (SAM). Two main

Figure 1. New Zealand mainland temperature changes, 1856–2008. Analysis and graph: James Renwick, NIWA, Wellington.



circulation changes have affected New Zealand since 1930. From 1930 to 1950, there was more south to southwest flow, but from 1951 to 1975, flow from the east and northeast increased. Then, from 1976, west to southwest flow predominated until 1998, since when there have been indications that westerly flow has weakened (Mullan et al. 2008). Rainfall has been responsive to predominant wind flow direction and thus there is no overall New Zealand trend. For instance, when years with predominately easterly flow (1960–1977) are compared with the following more westerly years (1978–1998), rainfall decreased in the north and east of the North Island but increased in the west of the South Island during the later period by as much as 15% relative to the 1961–1990 norm (Mullan et al. 2008). Western sites have shown a trend toward more daily rainfall extremes; northern and eastern sites a decrease. Between 1930 and 2004, pan evaporation has declined at 2 mm/yr from the 1970s, independent of precipitation changes (Roderick et al. 2007), and is believed to be mainly due to decreasing wind speeds or increasing cloudiness (Roderick & Farquhar 2005).

2.2 PROJECTED CHANGE

The theory of how greenhouse gas fluctuations drive climate change is now firmly established; past and present changes in climate are attributed to greenhouse gas fluctuations and modellers are now more confident of their predictions (IPCC 2007). Mullan et al. (2008) detailed the consensus view of climate change for the next 80 years based on 12 global climate models and a regional climate model for additional information. Their key points forecast:

- Increasing temperatures, with greater increases in winter and in the north. Best estimate increase of mean temperatures of 1°C by 2040 and 2°C by 2090, with projections ranging from 0.2–2.0°C by 2040 and 0.7–5.1°C by 2090 (relative to a 1990 baseline).
- Accelerating temperature rise, with the rate of increase projected to be higher than a linear trend from historical 20th-century record.
- Increasing temperatures, especially if associated with increased cloudiness, and overall rainfall leading to fewer low-altitude snowfalls, frosts, freeze-thaw cycles, and out-of-season frosts.

- Increasing frequency of extreme weather such as high-temperature and daily maximum rainfall events.
- Regional and seasonal changes in westerly winds and associated rainfall and drought patterns. In particular:
  - Increasing westerlies in winter and spring and, possibly, increased stronger winds, along with more rainfall in the west and drier conditions in the north and east of both islands.
  - Decreasing westerlies in summer and autumn, with drier conditions in the west of the North Island and possible increased rainfall in Gisborne–Hawke's Bay.
  - Increasing frequency of severe droughts across many eastern parts of New Zealand by 2080, such as inland and north Otago, eastern Canterbury and Marlborough, parts of Wairarapa, Hawke's Bay, the Bay of Plenty, Coromandel and Northland. For example, in a 'low–medium' scenario, Marlborough could experience a 1-in-20-year drought event every 3 to 5 years by 2080.
  - The possibility of droughts occurring in spring and autumn, not just in summer.

Mullan et al. (2008: xii) further commented:

> *Natural variations will be superimposed on human-induced long-term climate changes and together they will give us the extremes to which future New Zealand society will have to adapt. What currently is an unusually warm year could be the norm in 30–50 years, while an unusually warm year in 30–50 years' time is very likely to be warmer than anything we experience at present .*

Sea level has been rising at a rate of 1.7 mm per year during the last 100 years and the rate is predicted to accelerate. Best estimates predict a sea-level rise of between 18 and 59 cm by 2090–99 relative to 1980–1999 (Hume & Blackett 2007). Much less is known about how future changes may affect oceans around New Zealand. Willis et al. (2007) made the following predictions:

- **Waves:** Increasing frequency of heavy seas and swells along western and southern coasts.
- **Storm surges:** Increasing heights and extremes of storm-tide level.
- **Sea currents:** Various changes are plausible, but little research has been done and uncertainty is high. The Antarctic circumpolar current is likely to accelerate and increase flow of cool water to the Chatham Rise. Increased upwelling of cooler subsurface waters may result along the coasts.

3. Predicting biodiversity responses to climate change

Biodiversity—the sum total of the diversity, abundance and interactions of organisms and their ecosystems—is extraordinarily complex. Many species remain undescribed, even in relatively well-researched temperate areas like New Zealand, and for most of the rest, all that exists is a name, a brief diagnostic morphological description and some distributional data. Worldwide, a relatively few economically important, large or conspicuous biodiversity elements are monitored—forest trees, game animals, birds and butterflies. Direct abiotic influences on biodiversity elements and, in turn, relationships between them, are numerous and poorly understood (Tylianakis et al. 2008). Ecology is far from achieving its goal of becoming a predictive science. There are simply too many organisms, factors, variables and interactions. Hence, we are highly constrained in our ability to observe and predict changes in biodiversity.

Predicted responses to climate change fall into two categories: extrapolations from observed, experimental or theoretical understanding of biodiversity-climate linkages; and predictive modelling.

3.1 EXTRAPOLATIONS AND THEORETICAL APPROACHES

For most organisms, and for most ecosystems, we have little other option than to make broad generalisations about future biodiversity under a warming climate by using accumulated ecological observations and theory, and extrapolating from recent changes (e.g. Parmesan 2006). Palaeoecological observations help widen this evidential base (see section 5). The most common predictions made on this basis are:

- **Range and altitude change:** Many species will find, under a warming regime, suitable habitats opening up further south or higher up beyond their current geographic or altitudinal range, and some species unable to migrate will find their current range increasingly unsuitable
- **Phenological alterations:** The timing of seasonal activities such as flowering, breeding, growth and migration may alter and, in turn, disrupt relationships between species
- **Species interactions alterations:** Fluctuations in abundance and changing range limits will bring new combinations of organisms and new interactions with implications for both species and ecosystems, including disease
- **Trophic interactions alterations:** Plant productivity, below-ground processes (decomposers and mycorhyzial associations), and predator-prey interactions will be affected by climatic changes and increasing CO_2 concentrations
- **Exotic organisms are advantaged:** As climate changes, existing indigenous species may be disadvantaged relative to exotic organisms better suited to the new prevailing climates

3.2 MODELLING APPROACHES

The complex requirements of species, and spatially variable environments, make anything other than broad generalisations impossible without the use of explicit models or experiments. As experiments are costly and problematical (because experimental set-ups invariably bring with them additional artefactual change), most of our understanding of possible changes under global warming is based on models.

Two broad classes of models are currently used. The first are dynamic global vegetation models (DGVMs; Sitch et al. 2008), which use broad groups of similar species called plant functional types (PFTs) and basic physiological processes to simulate vegetation. DGVMs have not been parameterised on New Zealand PFTs, although results for New Zealand grid cells are routinely generated using global PFTs. DGVMs are primarily used to model the broad composition and structure of vegetation at a regional scale, and are hence vital in creating realistic land surface feedback for global general circulation climate models. However, as they have no species-specific information, they are of limited use for exploring plant range changes and extinctions.

Models in the second class are species-specific, and are further divided into process-based models and niche-based (or climate envelope) models (Sitch et al. 2008). Process-based models are based on parameterisation of individual species and explicitly incorporate biological processes at varying levels of realism. An example of such a model in New Zealand is LINKNZ, a forest process model that has been used to map potential vegetation in New Zealand (Hall & McGlone 2006) and past vegetation change (Hall & McGlone 2001). Another is 3-PG, which provides detailed stand information of use to both foresters and ecologists (Richardson et al. 2005; Watt et al. 2007; Waring et al. 2008). Process-based models are difficult to use for predicting future ecological communities because of the amount of data that has to be collected, and the processes that must be calibrated, for individual species. For this reason, they tend to be limited to species with well-understood demography or physiology, such as common trees.

Niche-based models establish a statistical or theoretical relationship between an environmental factor, or factors, and some measure of a species' abundance or distribution (e.g. presence or absence, abundance at random or stratified plots, and range maps). The model is then used in conjunction with climate predictions to project future species range or abundance changes. Potential range limits assessed on current species distributions invariably assume a stable equilibrium between the species abundance and some critical environmental factor or competition. As these are usually assessed on the basis of current species distribution, the ecological concept of 'fundamental niche' versus 'realised niche' is important.

A fundamental niche is the total range of habitats in which a species is capable of surviving. The realised niche is the smaller range of habitats that a species (limited perhaps by predators, parasitism, dispersal, rare events, or opportunity) actually currently occupies. In New Zealand, the realised niche is now much smaller than it would be naturally because of land use changes and spread of exotic organisms. Therefore, it is the realised niche that forms the basis for our current models and predictions. It is an everyday observation that both animals and plants can exist (and, in many cases, successfully reproduce) in the wild outside of their realised niche. Mitchell & Williams (1996) illustrate this with the case of Australian native *Eucalyptus regnans*, which, on the basis of production forestry in New Zealand, has a much larger fundamental niche than its realised niche in Australia. A further case is that of numerous New Zealand species that have extended their range overseas (Yeates & Williams 2006). For instance, karaka (*Corynocarpus laevigatus*) and New Zealand flax (*Phormium tenax*) are aggressive weeds in parts of Hawai'i, exceeding their current mean annual New Zealand temperature ranges by at least 5°C and 10°C, respectively.

Niche-based models are now ubiquitous and used routinely to generate predictions from species maps. Care has to be taken with niche-based models as many, if not most, do not perform better than chance alone. Beale et al. (2008) showed that climate envelope approaches predicted bird distributions in Europe no better than chance for 68 out of 100 species examined. Lozier et al. (2009) demonstrated that an underlying biological reality was not necessary for a niche-based approach to generate convincing results. A good fit to the observed data is no guarantee that the niche-based model is accurate.

Empirical tests of niche-based approaches have not been encouraging. In a recent study, Duncan et al. (2009) applied niche-based models to dung beetle introductions from southern Africa to Australia, and the climate matching approach failed to give adequate predictions for 60% of the cases studied. They concluded that only by identifying and quantifying the non-climate factors controlling species distributions could there be any certainty in predictions made for altered climatic states. Similar conclusions have been reached by Willis & Birks (2006). As non-climatic factors will also change in the future, this will be a challenging task, even for a single species. Given the number of significant interactions that occur in even a simple community between species, and between species and their environments, there are very real data and computational limits on how confident we may ever be in our predictions. At the moment, specific predictions of climate change effects from climate-biodiversity models cannot provide reliable explicit guidance for managers, especially when applied to range changes and extinctions.

Niche-based models tend to predict a greater degree of extinction and a greater proportion of colonisation than process-based models (Morin & Thuiller 2009). Neither take phenotypic plasticity and local adaptation into account. Therefore, niche-based predictions of massive species loss (Thomas et al. 2004) have proved controversial (Lewis 2006). Lewis (2006) outlined the complications that arise when niche-based models are used in association with species area curves to predict extinctions:

- There is no *a priori* reason why reducing habitat area should necessarily cause a decline in species richness that follows a generalised species-area curve
- The 'z' (slope of the species-area line) and 'c' background species richness values used are assumed, rather than measured, but are critical to predictions
- What constitutes 'habitat loss' is rarely defined in any operational way

Niche models are exceptionally useful and readily available tools for describing species distributions and for exploring climate distribution hypotheses. However, as prediction tools for species loss under changed climates they have serious shortcomings. Moreover, unless they explicitly incorporate a range of environmental drivers besides climate, their use is unjustified for making specific policy or management recommendations.

4. Past changes in biodiversity in response to climate change

It is important to look at past biodiversity changes in response to climate change for two reasons. First, future climatic change must be considered in the context of what has been the prevailing climate norm for our biodiversity over the last few million years; and second, changes in biodiversity in response to past climate fluctuations can give insights into future responses (MacDonald et al. 2008). As climate changes of a similar magnitude to those predicted for the coming century have occurred in the recent past, palaeoecological observations provide insights that cannot be gained in any other way.

In considering past climate change, it is useful to think in terms of two timescales. First, a short-term scale (c. 500 or so years) that accommodates the lifespan of the longer-lived elements of biotic communities, and thus can be regarded as an ecological processes scale. The second, longer term timescale (c. 120 000 years), is the average length of a glacial-interglacial cycle. Major landscape and climate cycles operate at these timescales, and it is the appropriate length of time over which to consider significant evolutionary processes (Myers & Knoll 2001). In the following two sections, the implications of climate change for both timescales are considered, but with more detailed analysis of the possible outcomes in the near future in the context of the short-term timescale.

4.1 IMPLICATIONS OF PAST CLIMATES FOR FUTURE BIODIVERSITY TRAJECTORIES

The possibility that the climate change currently being experienced will continue into the remote future is rarely considered in a policy context, but increasingly must be if adequate strategies to protect biodiversity are to be developed. Stabilisation of the concentration of atmospheric CO_2 at levels somewhat higher than present (e.g. 450–650 ppm) is predicted to lead to rising temperatures for hundreds of years due to the slow response time of the oceans (Solomon et al. 2007). Modelling of the interaction of greenhouse gas concentrations, the solar radiation cycle and the Northern Hemisphere ice sheets has indicated that the current warm earth condition (interglacial) is likely to persist for at least 50 000 years (Berger et al. 2003)—about four times longer than previous interglacials.

New Zealand is in the middle of an explosive evolutionary phase that was triggered by the emergence of the axial mountain ranges during the Pliocene and Pleistocene epochs, and climate change initiated by cooling during the Plio-Pleistocene period (McGlone et al. 2001). While New Zealand has relatively few genera and families for a landmass of its size, a disproportionately large number of species occur in a few genera that have undergone spectacular radiation. Of particular interest are radiations in the alpine regions of the southern mountains and the drier eastern areas, but some genera have also radiated in moist lowland forest environments. Many plants and animals also have highly distinctive local populations and morphs. A few groups have formed clines across the landscape, with step-wise replacement or gradual changes in morphology or ecological character of a species from area to area.

The majority of New Zealand plants and animals are adapted to cooler average conditions than those prevailing now—the result of the 2.5-million-year-long Pleistocene epoch during which climates were only for brief intervals warmer (c. 1–2°C) than present, but were for long periods 2–6°C cooler. As a result, the indigenous biota will have to adjust to conditions that may be permanently 3.5–8°C warmer than the long-term average. There will be several consequences:

- Certain habitats will be lost or permanently transformed. Eventual loss of much of the current alpine areas of New Zealand and severe reduction of cold-winter habitat seems inevitable under even moderate scenarios for rising greenhouse gas concentrations. A large part of the lowland northern North Island will effectively enter a subtropical climatic zone with mean annual temperatures between 1 and 4°C warmer than the warmest experienced over the last 3 million years.
- There are long-term consequences for soil development. Over the last few million years, the intense erosion and wind-blown loessic dust of the long glacial periods have regularly rejuvenated New Zealand soils. While erosion in mountain areas and redistribution of its products to the lowland in the form of silts and gravels has maintained this rejuvenation in many catchments, large areas of low-lying and rolling country will slowly, but surely, lose mineral nutrients through leaching of the upper soil layers but not be rejuvenated as in the past.
- The disruption of the semi-regular cycles of expansion and contraction of cool- and warm-climate-adapted species. A permanently warm climate regime, with only moderate fluctuations from the mean, will strand the cold-adapted alpine component section of the biota in a small, and probably shrinking, climatic space.
- Following from the previous three consequences—and perhaps the most important aspect of anthropogenic climate change—is disruption of evolutionary trajectories (Myers & Knoll 2001; Parmesan 2006). Current conservation effort is appropriately devoted to retaining current biodiversity in the face of predators, weeds and habitat destruction. However, these activities should take place in the context of a broader strategy that seeks to maintain evolutionary processes over the longer term. An evolutionary trajectory strategy would place a high value on the protection or re-establishment of large, extensive, connected areas of habitat, and the maintenance of a substantial indigenous presence throughout the range of disturbed landscapes.

4.2 PAST WARM EPISODES AS A TEMPLATE FOR FUTURE GREENHOUSE WARMING

Over the past 3 million years, the New Zealand region has experienced prolonged, regularly occurring episodes of warm or warmer-than-present climates. Much earlier warmer-than-now epochs, although of interest for the insights they give into how the climate system responds to changing astronomical parameters and greenhouse gas fluctuations, are of limited use for understanding biodiversity changes because of the very different biota and landscape at that time.

Previous interglacials were as warm or warmer than the present and had similar vegetation composition to the present (Bussell & Pillans 1997). Climate reconstructions (based on beetles) for the peak of the last interglacial (c. 120 000 years ago), indicated that temperatures were up to 2.5°C warmer in the summer and up to 3°C warmer in the winter than at present (Marra 2003). These results are mirrored in the current interglacial (Holocene) where quantified estimates from pollen and marine core results suggest a thermal optimum between 9000 and 7000 years ago, which was up to 2–3°C warmer than present, depending on the site (Pelejero et al. 2006; Wilmshurst et al. 2007). Cooling to present-day temperatures was well advanced by 4000–3000 years ago. There is some suggestion (lower tree lines and greater range of frost-sensitive elements) that it was winter, not summer, temperatures that accounted for the higher mean annual temperatures (McGlone et al. 2004b, 2010). For instance, mangrove (*Avicennia marina* var. *australasica*) was 140 km south of its current station in coastal central North Island 7000 years ago, and the average winter minimum temperature may have been 4°C higher (Mildenhall 2001).

The warming experienced at the height of the present interglacial between 9000 and 7000 years ago is significant, as it is in the range of increases predicted for the near future. At that time the vegetation cover of New Zealand was substantially different: southern beeches (*Nothofagus*) were uncommon, even in montane and subalpine areas, and dry forests were more widespread, especially in the east (McGlone et al. 1993). However, some aspects are not consistent with what we might consider a logical outcome of warming. Many frost-sensitive northern elements (such as kauri, *Agathis australis*) did not extend their geographical range. Non-climatic factors such as soils, ecosystem stasis and dispersal limitations probably prevented rapid spread southwards for many species, despite warming climates. More puzzling is that treelines were below current levels during the period of peak warmth. However, if, as we suspect, warming largely affected the winter months, alpine trees (which are limited by their opportunities for growth in the summer months) may have formed a lower treeline (McGlone et al. 2010). As current greenhouse warming has affected winter more than summer temperatures, it is possible that climates of the immediate future may resemble the past warm episodes in being more intensely oceanic than over the last few thousand years. That being so, biotic change cannot be predicted on the basis of mean annual temperature rises alone.

5. Potential direct effects of increased CO_2 concentration on indigenous biodiversity

Of all the predictions made for the near future, the most secure is that atmospheric CO_2 concentrations will continue to increase, perhaps to as much as between 730 and 1020 ppm by 2100 (Meehl et al. 2007). CO_2 is a key plant nutrient, and experimental studies have shown that plant growth is positively affected by increasing ambient CO_2 concentrations under non-limiting soil nutrient supply, and that water use will diminish in some species (but not, for example, in conifers) as the stomata will not have to remain open for as long (Lambers et al. 2008). Virtually all New Zealand indigenous plants use a C_3 photosynthetic pathway; the C_4 pathway is largely restricted to some exotic grass species of warm climate origin (Edwards & Still 2008). C_4 plants have a strong competitive advantage over C_3 plants under warm, droughty conditions and have a photosynthetic advantage under low CO_2. Accepted wisdom is that rising atmospheric CO_2 concentrations will reduce the current advantage of C_4 over C_3 plants, while warmer droughty conditions will increase it. However, more recent work suggests that C_4 plants will be advantaged by increased CO_2 (Leakey et al. 2004).

For C_3 plants, elevated atmospheric CO_2 concentration changes the composition of live tissue, commonly increasing carbohydrates and decreasing proteins. Some herbivorous insects will alter their behaviour in response to this change in tissue composition by, for example, consuming more, or less, or, in the case of generalists, by feeding on different species. Körner (2003) reviewed current understanding of plant and ecosystem response to elevated CO_2 and concluded that:

- Plant species respond differently to CO_2 enrichment (irrespective of the type of response involved) and these translate into ecosystem responses
- Plant responses depend on soil type, nutrition, light, water and age
- The quality of plant tissue (more carbon, less of other elements) and the amount of exudates from roots change, so plant consumers are affected
- Plant responses to CO_2 concentration are nonlinear, with the strongest relative effects underway right now, and few additional effects beyond about 550 ppm

Beier (2004: 244) stressed how limited our understanding is:

> *The effects of CO_2 alone and, to a lesser extent, warming alone each show some general and consistent patterns, but the few examples of combinations point in all directions and results are not predictable based on the individual effects. The complexity and unpredictability becomes even worse when we realise that important effects may be driven by changes in off-season processes, seasonality and extreme events …*

Dukes (2007) pointed out that the predictive power for natural settings of experiments under elevated CO_2 based on isolated plants is essentially zero. Moreover, plant responses to disease, herbivores, mutualists and other species may have important community-level effects that change conclusions based on plot-scale global change experiments.

Little experimental work has been done on indigenous New Zealand plants under elevated CO_2 (Ross et al. 2006), and none in natural settings. Therefore, we have to rely on generalised findings. Essentially, these suggest that in typical nutrient and/or climatically constrained communities, the direct effects of CO_2 fertilisation will be minor and, at this point, unpredictable. Increased CO_2 availability does not necessarily result in increased growth: availability of other nutrients, such as nitrogen and phosphorus, the soil type and status, and space for growth are more likely to be limiting. The predictability of effects on plant herbivore guilds will be even less, as the number of species involved is large and the effects recorded in experiments show both positive and negative effects depending on the specific plants and insects involved.

6. Current and potential effects of climate change on indigenous biodiversity

6.1 PHENOLOGY

Climate warming will lead to a reduction in cold-related phenomena, especially if it is accompanied by increases in cloudiness or overall precipitation (which could buffer the system). This would include less low-altitude snowfall, and fewer and less severe frosts, freeze-thaw cycles and out-of-season frosts. In turn, these should have marked effects on temperature-related phenological responses and could strongly affect ecosystem function.

Accurate phenological measurements[1] require close monitoring of phenomena. These are time-consuming to collect, particularly in New Zealand, due to our oceanic climate. Specifically, for events such as migratory arrivals and departures, emergence of insects, leaf unfolding and flowering, accurate, cost-effective measurements are easier to make if the events occur in a condensed period, with unambiguous criteria and easy access. These factors often do not apply in New Zealand. For instance, few New Zealand trees have well-defined overwintering buds and many flush leaves sporadically. Although flowering times are often better defined, many plants flower sporadically and flowering intervals are shortened later in the season in higher latitudes in New Zealand (Godley 1979). A common theme arising from most phenological studies is the flexibility of the temperature-related responses. For instance, deciduous plant species in New Zealand tend to lose their leaves as a response to cold temperatures, and not so much as a set response to day length (McGlone et al. 2004a). Nevertheless, although cold-related biological phenomena such as deciduousness, diapause[2] in invertebrates, hibernation, and chilling requirements for germination are not as widespread as in other temperate zones (Dumbleton 1967; Morris 1989; Wardle 1991; McGlone et al. 2004a), they are important factors in the biology of the species concerned.

Many observations have been made on the phenology of New Zealand species, mainly as parts of descriptive studies of individual species ecology, broader studies of community functioning, or for specific purposes—such as determining the seasonal availability of nectar in island reserves (Perrott & Armstrong 2000) or to show how life-cycle factors may affect distribution (e.g. alpine forget-me-nots, *Myosotis* spp.); Dickinson et al. 2007). Other sets of observations are specifically directed towards teasing out the relationship of temperature to the phenomenon of interest (e.g. flowering times of the cabbage tree, *Cordyline australis*); Harris et al. 2006).

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[1] Measurements of periodic biological changes (e.g. seasonal or migratory) in relation to climate.

[2] Diapause is a delay in the development or physiological activity of insects in response to adverse environmental conditions.

There are only a handful of long-term phenological studies in New Zealand that could provide a basis for an analysis of climate effects. The most significant of these relate to seeding (discussed in section 8.1). An ambitious set of studies was initiated in 1966 at the Orongorongo Research Station in the Rimutaka Ranges, lower North Island, which recorded many phenological phenomena for many taxa (plants, invertebrates, birds, and mammals) over an extended period (Brockie 1992). Seed-fall monitoring continues, but activity at the site has been downscaled considerably, and no comprehensive broad-based phenological monitoring is currently underway.

There is no doubt that long-term studies can yield insights obtainable in no other way (e.g. those from central England, where several hundred species were observed on a daily basis over 50 years; Fitter & Fitter 2002). Unfortunately, the data for such a study do not exist in New Zealand, nor do we have the formal monitoring networks that would make it possible. Why has New Zealand invested so little in documenting phenological and migratory phenomena? Leaving aside the fact of a small scientific and amateur workforce available for the task, and the challenges faced in making meaningful measurements in an oceanic environment, the most likely explanations are:

- The rather muted phenological change that is typical in New Zealand has not attracted much scientific attention
- The butterflies, migratory birds, deciduous trees, large conspicuous flowers, and hibernating animals that have been focussed on elsewhere (such as in the Northern Hemisphere temperate zone) are under-represented in New Zealand
- The relative lack of indigenous ecosystems and species (birds in particular) close to major population centres

Despite these constraints, efforts continue. A number of tree seed sites are regularly monitored. The recently developed New Zealand Biodiversity Recording Network (www.nzbrn.org.nz; viewed 1 December 2009) registers species observations made by professional and amateur ecologists, and in 3 years has added over 260000 plant observations and over 43000 bird observations. However, in the short term, comprehensive phenological and migratory data for a range of New Zealand taxa extending over a long time period will remain rare.

6.2 TERRESTRIAL BIRDS AND BATS

Birds have been better monitored than most groups of organisms—bird banding has been systematically carried out since 1948 (Froude 1999) and some of the longest duration data sets in the world are for New Zealand birds (e.g. Australasian gannet, *Morus serrator*), since late 1880s; Darby 1989). With a few exceptions, native birds have been comprehensively displaced from modified habitats by introduced birds, and even in their remaining natural habitat are greatly reduced. The inability of native birds to persist is largely the result of loss of preferred food in modified landscapes and predation by introduced mammals in natural habitats. There is little evidence that introduced birds have had a direct adverse impact on indigenous bird species in less modified natural areas (Clout & Lowe 2000), aside from the myna (*Acridotheres tristis*); Tindall et al. 2007). Wasps have greatly reduced food resources for some indigenous birds, particularly in beech honeydew areas (Beggs 2001).

A study of climate and the distribution of European birds showed that winter cold, seasonal warmth and moisture availability were the most important factors controlling distributions (Huntley et al. 2006). In contrast, climate does not seem to be a major controlling factor for indigenous birds in New Zealand settings—habitat modification and predation are much more important. Thus, there are no obvious indigenous bird species richness gradients related to climate, nor do they show biodiversity hot spots (Robertson et al. 2007). Therefore, there is no basis for assuming that warming climates will necessarily induce range changes in indigenous bird species. Of the exotic birds, there is evidence that myna distributions are controlled by warmth because, from initial introductions in the South and North Islands in the 1870s, the species' range rapidly contracted to the northern half of the North Island (Tindall et al. 2007).

While average climate may be of little consequence for indigenous bird distribution and abundance, severe cold and consequent reduction of food supply can affect bird mortality and egg laying, and there is some evidence for these factors being important for some species in some locations. Fantail (*Rhipidura fulignosa*) populations decline after severe winters (e.g. in Canterbury/Nelson in 2002; D. Kelly, University of Canterbury, pers. comm. 2009) and local extinctions of fantails through climate-related reduction of insect abundance have occurred on offshore islands (Miskelly & Sagar 2008). A record cold event in Southland in July 1996 reduced mohua (yellowhead, *Mohoua ochrocephala*) numbers by 66% in the Eglinton Valley (Dilks 1999). Furthermore, during that same winter, takahē (*Porphyrio mantelli*) in the Murchison Mountains also suffered a significant population decline (Maxwell 2001). Many of New Zealand's braided river birds species (e.g. wrybill, *Anarhynchus frontalis*, black-fronted tern, *Sterna albostriata,* and black stilt, *Himantopus novaezelandiae*) are now forced to nest on islands in the middle of braided rivers as a result of predator pressure and river-edge habitat modification (i.e. from farming; Keedwell 2005). Not surprisingly, the major cause of nest failure (and egg/chick mortality) is spring floods. Under the predicted climate change scenarios, spring flooding is expected to increase (Mullan et al. 2008), which will result in recruitment failure for braided river bird species.

With regard to exotic birds, a significant trend towards an earlier egg-laying date (30 days between 1962 and 1995) for welcome swallows (*Hirundo tahitica*) has been tentatively attributed to global warming (Evans et al. 2003). Starlings (*Sturnus vulgaris*) in New Zealand have been shown to lay eggs early under high (El Niño) and low (La Niña) values of the SOI, which, in turn, is linked to El Niño Southern Oscillation-related changes in food supply (Tryjanowski et al. 2006). However, local food supply conditions mean that variation in the response from district to district can be expected (Flux et al. 2009).

New Zealand bats are unusually active in winter compared with other temperate bats (Sedgeley 2001) and are therefore sensitive to warmer winters, becoming more active and thus experiencing an increased demand for insect food at a time when it may be scarce (Pryde et al. 2005).

While climatic factors will have a secondary effect on bird and bat populations and ranges when compared to predator pressure and habitat modification, they will still be significant. It can be expected that insectivorous and fruit- and nectar-eating birds will be advantaged by warmer winter climates, along with the possibility for some counter-intuitive outcomes (as with bats) or impacts of extremes (as with river-nesting birds).

6.3 MARINE AND COASTAL BIRDS

Marine and coastal birds are abundant in the New Zealand region, especially close to the coast and along the subtropical convergence where nutrient-rich waters upwell. The New Zealand region has the most diverse seabird assemblage in the world—about 80 species breed here—and it is the most significant area globally for seabird biodiversity. Climate change is likely to affect sea and shorebirds through altered sea conditions reducing the abundance of marine food or the birds' ability to access it. Predicted changes in the New Zealand region are for warmer sea surface temperatures (there has already been an increase of 0.6°C in the upper 700 m since 1950), an increase in westerly winds in winter/spring, and a strengthening of northeasterly winds in summer (Willis et al. 2007). More La Niña-like conditions are predicted to prevail, although it is unclear what will happen in the long term to important ocean-atmosphere processes reflected by indices such as SAM, SOI or PDO. Major oceanographic features such as the position of maximum upwelling will not alter, but changes in intensity of the processes will. Net primary production in the oceans of the New Zealand region is close to the global average and has not changed markedly in the recent past (Willis et al. 2007).

New Zealand has a number of long-term records of seabird abundance. Predation on land is by far the most important predictor of success in seabirds (Jones 2000), but, where that factor is absent, marine states—and, in particular, the ENSO cycle—are often claimed to explain fluctuations in seabird numbers (e.g. Scofield & Christie 2002). Australasian gannets (a large predator of small pelagic fish) have increased markedly in Australasian waters for a range of reasons, including more frequent and stronger ENSO events, and increased discard from fishing boats following the introduction of quota-based harvesting regulations (Bunce et al. 2002). A 41-year record of the endemic red-billed gull (*Larus novaehollandiae scopulinus*, a medium-sized inshore feeder on krill) shows that its breeding success is positively related to krill abundance, which is, in turn, positively related to the La Niña (or positive phase of the ENSO cycle) when winter conditions are stable, and upwelling associated with northeasterly winds in late spring (Mills et al. 2008). Recent changes in the PDO and El Niño (negative) phase ENSO events led to a more than 50% decline in red-billed gulls between 1983 and 2003.

Decline in yellow-eyed penguins (*Megadyptes antipodes*) has also been shown to correlate to climate variables (rainfall and temperature). Although a mechanistic link has not been demonstrated, it is theorised that ocean productivity is likely to be the key influence (Peacock et al. 2000). Sooty shearwater (*Puffinus griseus*) populations have declined in recent years both at breeding colonies in New Zealand and at wintering grounds in the eastern North Pacific. This trend coincides with increases in oceanic temperatures and southwards movement of the Subantarctic Front, which may have limited productivity (Lyver et al. 1999; Shaffer et al. 2006; Scott et al. 2008).

6.4 REPTILES

New Zealand has an unusual and ancient reptile fauna. Two species of tuatara (*Sphenodon*), sole representatives of the order Sphenodontia, survive on offshore islands; and there is a lizard fauna of possibly more than 80 spp. in the genera *Hoplodactylus*, *Naultinus*, *Cyclodina* and *Oligosoma* (Towns et al. 2001; Chapple et al. 2009). All but one of the lizard species bear live young and an unusual number are adapted to cool environments, including the black-eyed gecko (*Hoplodactylus kahutarae*), whose habitat is alpine bluffs 1300–2200 m above sea level in the north of the South Island. Over half the species are rare or endangered—a situation largely brought about through predation (Towns et al. 2001).

Given the sensitivity to ambient temperatures of all reptiles, and the cool and variable environment inhabited by many New Zealand species, warming temperatures may lead to both positive and negative population change, depending on the species involved. In New Zealand, skink behaviour during pregnancy has been shown to relate strongly to attainment of ideal temperatures for embryo development, and temperature during pregnancy of viviparous lizards can have a wide range of effects, including changes in offspring body size, shape, locomotor speed, scale pattern and sex (Hare et al. 2009). Tuatara provide a well-researched example: tuatara sex ratios are determined by burrow temperatures experienced by the eggs. On one of their major refuge islands, the sex ratio is heavily biased towards males (1.7 M : 1 F; Nelson et al. 2002). Results of mechanistic modelling suggest that these particular sex ratios will increasingly tilt towards males due to rising temperatures until, with a mean annual temperature rise of 4°C, all will be born male (Mitchell et al. 2008). There is some reason to be sceptical that this scenario will eventuate, as the tuatara lineage has survived warmer temperatures in the past—until recently, tuatara thrived in Northland where mean summer temperatures are about 6°C warmer than the southern tuatara islands. Additionally, there are a number of readily available interventions such as plantings, shade cloth, and translocation that could help to address the impact of temperature rise on tuatara (Huey & Janzen 2008).

Regardless of whether the climate warming threat to tuatara is real or not, these studies demonstrate how potent climate can be as an ecological factor for reptiles. Given the number, importance and threatened status of many reptiles, it would be prudent to ensure that research into potential climate change effects and responses is carried out alongside the more immediately relevant predation studies.

6.5 INVERTEBRATES

Little New Zealand research has been specifically directed towards effects of climate change on invertebrates, which, being ectotherms[3], could be expected to show major responses. A recent review of potential change in insect pests in planted forests in New Zealand concluded that it was difficult, even in such

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[3] Ectotherms are animals whose body temperature depends on that of their environment.

well-studied systems, to make any generalised assessment as to likely effects of climate change on their abundance and distribution (Watt et al. 2008). The most probable outcome was considered to be a greater abundance of insect pests due to improved over-winter survival. Laird (1995) suggested that climatic warmth 'windows' in the early 1970s and late 1980s may have assisted the exotic mosquitos *Culex quinquefasciatus* and *Aedes notoscriptus* in their southwards dispersal within New Zealand. International agencies are incorporating invertebrates in monitoring schemes (e.g. Doran et al. 2003), and observations have been made on changes in invertebrate distributions and abundance in response to climate (Parmesan 2006), but it is also recognised that there has been little progress in generalising impacts of global change on insects (Harrington et al. 2001; Dukes et al. 2009). A number of studies directed towards understanding past climates and their effects on the biota have used climate niche approaches to characterise limits of beetles (see Marra et al. 2004; Burge & Shulmeister 2007), chironomids (Woodward & Shulmeister 2007) and stick insects (Buckley et al. 2009). The only significant recent invertebrate-climate work in New Zealand has been carried out on freshwater and alpine ecosystems (see sections 7.6 and 7.9).

6.6 AQUATIC ORGANISMS AND FRESHWATER ECOSYSTEMS

New Zealand freshwater ecosystems are relatively simple, and food chains are generally short. There are just 39 currently recognised indigenous species of fish (although the number is likely to increase) and 16 exotic species (McDowall 2010). None of the freshwater invertebrate groups are particularly rich in species (Winterbourn 1987).

Of the indigenous fish fauna, about half (18/39) are diadromous, meaning individual fish make dual regular migrations between marine and freshwater environments in the course of their life cycle (McDowall 2010). Diadromous fish, with few exceptions, span the full latitudinal range of New Zealand and thus their species diversity is more or less the same north to south (McDowall 1998). These species are therefore unlikely to be much affected by warming per se (Glova 1990). On the other hand, non-diadromous species tend to be much more restricted and it can be postulated that their inability to use marine pathways to access new freshwater systems will make them more susceptible to changing environments.

The thermal tolerances of the vast majority of freshwater organisms in New Zealand are unknown (Quinn et al. 1994). Richardson et al. (1994) experimentally showed for eight common native fish species that they were able to thrive within a wide temperature range. A subset of the fish and invertebrate fauna are adapted to cool waters (peak temperatures <16–18°C). Leathwick et al. (2005) found that January air temperature, upstream average temperature and rain days, river flow, and riparian shade were all significant factors in a predictive model for indigenous diadromous fish distribution. While little is known about the temperature tolerances of aquatic invertebrates (McDowall 1992), stoneflies have been shown to have low tolerance of warm temperatures. Quinn & Hickey (1990) noted that stoneflies had restricted distributions compared with mayflies in streams, and Quinn et al. (1994) showed that a representative stonefly of cool waters was unable to acclimate to higher temperatures. These studies support the concept that warming waters in the North Island lowlands could lead to the thermal tolerances of some organisms being exceeded, because temperature stress already probably curtails some distributions. Collier & Smith (2000) showed that lethal water temperatures for stoneflies occur much more commonly in streams flowing through pasture rather than forested landscapes. Eels (*Anguilla* spp.) are almost completely inhibited from migrating up rivers where the temperature exceeds 22°C (August & Hicks 2008). More recent work suggests seasonal recruitment of eels may have advanced by a few weeks and is also declining, possibly as a result of changing thermal fronts in the Tongan spawning areas (Jellyman et al. 2009). Of particular concern is the possibility that, as glaciers vanish and alpine streams warm, alpine mayflies (*Deleatidium* spp.) may suffer reduced ranges and local extinction (Winterbourn et al. 2008).

New Zealand's lowland and eastern freshwater ecosystems are already under considerable pressure from deforestation, abstraction and pollution. Historic deforestation may have already seriously curtailed ranges or even caused extinction of freshwater invertebrates, so maintenance of forest cover is of high importance (Harding 2003). Climate change is likely to exacerbate these pressures through warming. River temperatures are also correlated with the SOI, as are other standard measures of water quality variability including drought-related changes (Scarsbrook et al. 2003). In eastern areas, the effects of the predicted increased incidence and severity of droughts will likely be compounded by increased abstraction for irrigation (see also section 8.3). There is the potential for cool-adapted freshwater organisms to be affected under warmer climates, especially in streams that have lost tall shading vegetation along their banks, and if water flows drop through drier conditions and greater abstraction of water for agriculture (Larned et al. 2004; Barquin & Scarsbrook 2008).

While for many freshwater organisms there may be movement within catchments and between habitats in response to temperature change, a significant number of range-restricted species are trapped in lakes or east- or west-flowing river systems (Allibone 1999), meaning they are unlikely to be able to move southwards to cooler waters.

Drought, combined with low water levels in rivers, poses major risks to aquatic life (Palmer et al. 2008). In agricultural and pastoral catchments, lack of dilution and flushing results in higher loadings of nitrogen and phosphorus and increased bacterial contamination (Caruso 2001). Fish kills occur when populations are stranded from the main flow with no deep pool refuges. Invertebrate communities are best sustained by steady water flows (Scarsbrook 2002). Adverse effects from drought and high temperatures (>25°C) can be widespread in some districts and locations but recovery is usually rapid (Caruso 2001). The indigenous fish community in New Zealand gravel-bed rivers appears to be resilient to both flood and droughts as long as refuges are available during low water flows (Jowett et al. 2005; Davey et al. 2006).

Many of the generalised impacts suggested for flowing water systems apply to lakes: pollution, abstraction of water, drought lowering of water levels and an increase in water temperature (Meyer et al. 1999). Heating of the upper lake water layers has a large potential to create biotic change because of increased lake stratification and warmer temperatures in the upper few metres. In large, deep lakes this can have a positive effect on fish, but in smaller lakes, reduction in dissolved oxygen and deepening of the thermocline can reduce or eliminate thermal refuges for cool or cold water species (Meyer et al. 1999). Wind has a marked effect in reducing lake stratification, hence the predicted increase in wind in some districts in New Zealand could offset the heating effect, but will be very much site dependent.

Fish are often sensitive to water temperature, so warming can be expected to advantage temperate invasives, and disadvantage thermophilic species. While at present the risk of aquarium escapes is moderated by the lack of tropical or subtropical character in our waterways, this risk will grow with rising temperatures (Koehn & McDowall 2004). Introduced trout (*Salmo*, *Onchorhychus* and *Salvelinus* spp.) are having a major impact on the abundance and distribution of native galaxiid fishes (McIntosh et al. 2010). Modelling of exotic fish populations suggests that there could be a generalised shift southwards for brown and rainbow trout (Glova 1990; Scott & Poynter 1991), with possible benefits for the indigenous fish they compete with and prey on in the North Island (McDowall 1992).

Interestingly, environmental stressors such as floods, low flows and natural acidity can help native galaxiids survive in the presence of trout, as they are much less affected (McIntosh et al. 2010). Therefore, predicted increases in rainfall variability may not have a negative effect on indigenous fish.

6.7 PLANTS AND FORESTED ECOSYSTEMS

Many plant species have range boundaries that can be correlated either with a macro-environmental variable such as soil or climate, or are clearly linked with a certain habitat type. For instance, Northland has a rich assemblage of plants that are not found further south. Some, if not most, of the endemic northern flora have attributes that are clearly suited to warm rather than cool climates, such as higher photosynthesis rates under warm conditions (e.g. kauri) or lack of cold tolerance (e.g. pūriri, *Vitex lucens,* or mangrove); Wardle 1991). Warming or changing climates may eventually lead to a mismatch between the climate envelope for a given species and its current distribution.

However, Wardle (1991) suggested that a large number of New Zealand vascular species had ranges smaller than their potential climate envelopes, implying that historic or soil factors played a role. Other studies of past distribution of plants (trees in particular) over the last glacial-interglacial cycle, do not give strong support to the concept of long-distance movement of plants across the landscape in response to glacial-interglacial cycles (McGlone 1997). New Zealand's oceanic climates may have a great deal to do with this relative lack of mobility, because the lack of climatic extremes combined with relatively large climate fluctuations within a given season should promote generalist climate adaptations. The wide latitudinal ranges of some of New Zealand's most abundant organisms such as rimu (*Dacrydium cupressinum*), which occurs from Stewart Island/Rakiura to North Cape as a dominant, and many terrestrial bird species, are consistent with this expectation. Moreover, our mountainous landscapes provide large thermal and precipitation gradients over short spatial scales. Species that are poor competitors under warming climates therefore usually have options close by and will probably survive *in situ*. The exception, as always, are those species confined to small isolated patches of habitat, as with many endangered species.

Niche-based models are widely used internationally, because of the ready availability of mapped environmental variables and species distributions. In New Zealand, they have had relatively limited use apart from predicting tree distributions (Mitchell & Williams 1996; Leathwick 2001). For some species, large range changes have been predicted (e.g. tawa, *Beilchmiedia tawa*); Leathwick et al. 1996); other species appear either to have complex interactions between climate factors or to be also limited by non-climate-related factors and thus their ranges are unlikely to change. For instance, in the case of kauri, major population declines have been modelled but the geographical range is predicted not to change by much (Mitchell & Williams 1996; Mitchell 2009).

Malcolm et al. (2006) recently modelled species extinction under various climate change scenarios, and made estimates for 25 biodiversity hotspot regions globally, including New Zealand. They used an integrated approach of using a DGVM to predict biomes under a climate with double the concentration of CO_2, species biome specificity (essentially niche-based), a species-area curve assessment of how many species would be lost, and assumptions about migration. They predicted a loss of about 40% of New Zealand's endemic species under assumptions of zero migration and narrow species specificity, but only 2.5% loss under a scenario of perfect migration and broad specificity. Such modelling exercises are useful contributors to an ongoing debate, but should be cautiously regarded as a basis for policy. For example, Leathwick & Austin (2001) demonstrated that interactions between southern beech and other tree species changed the outcomes of predictions based on environmental factors alone. Further, they argued that landscape models of future vegetation will need both environmental and species interaction rules to be successful. This is not the only caveat: herbivory, disease, unusual climatic extremes and novel interactions mean that even with reliable climate projections, our biotic projections will have wide error margins.

It is not clear whether ecosystem responses to climate change will be incremental, abrupt or a mixture of both. However, abrupt changes are now commonly seen in the Northern Hemisphere (Jepsen et al. 2008) and abrupt community changes as a result of discrete climate events have occurred here occasionally. For instance, warm temperatures and drought promote excessive honeydew production in mountain beech (*Nothofagus solandri* var. *cliffortioides*, Dungan et al. 2007) and kāmahi (*Weinmannia racemosa*) forest, which, in turn, promotes outbreaks of *Platypus* beetle damage leading to die-back of mature trees, which can ultimately determine the forest structure (Wardle 1984). The 2-week period of record below-freezing temperatures in Southland and Otago in July 1996 gave rise to severe cold-related damage and death among a wide range of indigenous and exotic trees, shrubs and herbs (Bannister 2003). Neither of these abnormal events could have been easily foreseen, especially given the general warming trend of the 20th century, showing that some consequences of changing climates are likely to be unpredictable.

6.8 IMPACTS OF FRAGMENTATION AND FIRE ON FORESTED ECOSYSTEMS

Climate change does not occur in an ecological vacuum, and the most mentioned exacerbating factor in climate change literature is the compounding effect of habitat loss and the associated breaking up of native habitat into isolated patches (Heller & Zavaleta 2009). Fragmentation reduces beetle populations in forest fragments up to 1 km from the forest edge (Ewers & Didham 2008). Reduction of population size is one of the key factors increasing susceptibility to other shocks, and has been a factor in local extinctions of native plants (Duncan & Young 2000). Even when existing populations within a fragment are not reduced, it is often claimed that climate change will be much more damaging if plants and animals lack continuous tracts of undisturbed habitat along which to locate their preferred climate space (de Chazal & Rounsevell 2009). For instance, in the United Kingdom, even though butterfly species are advantaged by climate warming, over three-quarters declined between 1970 and 2000, mainly as a result of previous (and ongoing) human-induced habitat fragmentation (Warren et al. 2001).

Because of the overwhelming impact of invasive mammals on native vertebrates and large invertebrates in New Zealand, fragmentation has been seen, until recently, as a secondary issue (but see Ewers et al. 2006). Climate change may alter this, in that climate stress will impact fragmented patches in which some indigenous populations will be already reduced, making spread to more climatically favourable locations difficult. The north of the North Island is one of the most likely areas for fragmentation to negatively affect outcomes for biodiversity under projected climate change. Much of the northern landscape will become a frost-free zone by the end of this century, hence fragments will be under heavy invasive weed pressure. Another area is the dryland zone, east of the main axial ranges of both islands, where continuing land development and consequent reduction or elimination of existing fragments of habitat will be exacerbated by projected increases in drought (Rogers et al. 2002).

New Zealand's native species evolved under conditions of highly infrequent natural fire, and are generally poorly adapted to survive fire and to recover in its wake (Ogden et al. 1998). New Zealand has around 3000 rural fires per year that burn about 7500 hectares of vegetation. While affecting only a very small proportion of the total land area, these fires are often in areas with already highly fragmented and vulnerable remnants. The fires, although largely controlled, continue to have a negative effect on the biodiversity values of fragments by encouraging weed spread and reducing margins (Espie & Barratt 2006). Pearce et al.'s (2005) report on fire within warming climates concluded:

- Warmer temperatures, stronger winds, lower rainfall and more drought will prevail by 2080
- The fire season will probably be longer through starting earlier and finishing later
- More thunderstorms and lightning will increase ignitions
- Fuel will be easier to ignite
- Drier and windy conditions will result in faster fire spread and greater areas burned
- More severe fire weather and fire danger is likely throughout eastern districts, Bay of Plenty and Wellington/Nelson regions

Most New Zealand indigenous ecosystems will be unlikely to be adversely affected by fire under a greenhouse scenario, but fire will intensify in those eastern areas where indigenous biodiversity is most vulnerable. Increased natural spread or planting of exotic conifer forests in these zones is likely to exacerbate fire incidence and consequent damage to vulnerable biodiversity (see also section 8.1.2).

6.9 SUBALPINE AND ALPINE ECOSYSTEMS

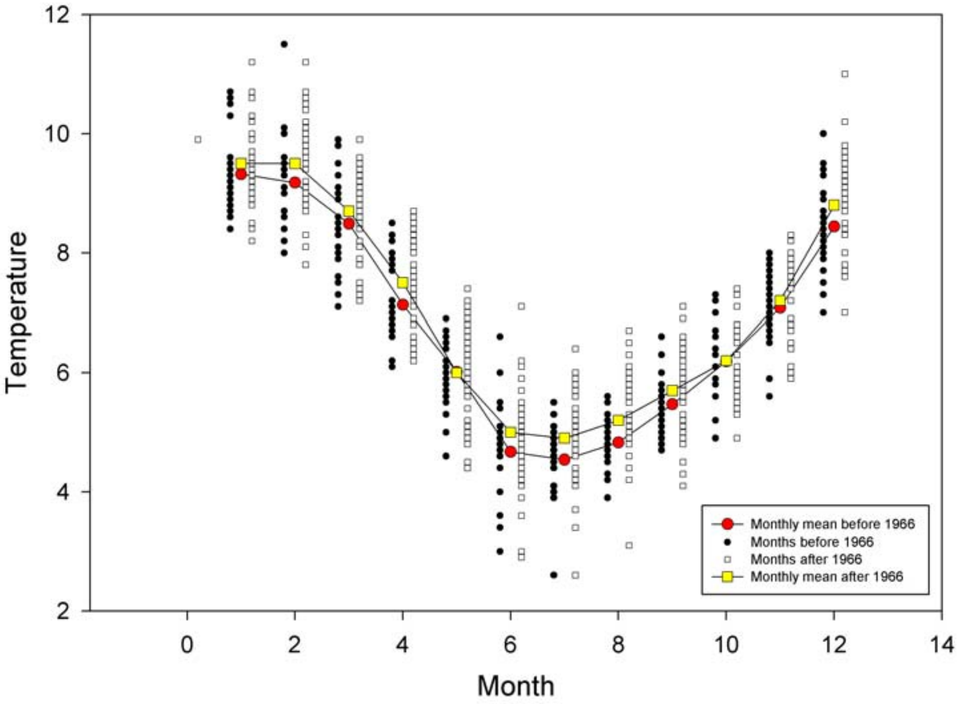
6.9.1 Tree line

Tree lines represent the altitudinal limit to tree growth (that is, woody plants 3 m or more in height) and, in New Zealand, coincide with the tree species line and are typically abrupt (Wardle 2008). They are a temperature-controlled phenomenon, and their position tends to coincide with a mean average growing temperature of about 7°C globally (Körner & Paulsen 2004). As subalpine forest is often species-depauperate, and the alpine zone species- and endemic-rich, the potential for rapid upward movement of the tree line ecotone with warming will have potentially severe biotic consequences. Halloy & Mark (2003) have made calculations of potential biodiversity loss for alpine communities in New Zealand. If the current warming is maintained, they estimate that between 40 and 70 species of indigenous plants will be at risk of extinction due to rising tree lines and spread of closed woody vegetation. If mean annual temperatures reach 3°C above present, they predict loss of about 80% of the discrete alpine areas in New Zealand, and extinction of between 200 and 300 species of indigenous vascular plants—or up to half the alpine total. Loss of a proportionate number of alpine invertebrates, as well as invasion by exotic alpine species, would also seem inevitable.

New Zealand tree lines were investigated in 1989/90 at 18 sites, and the results suggested that an 'advance zone' of seedlings and saplings of several species had established during the previous 60 years (Wardle & Coleman 1992). However, in most cases, the advance zone of seedlings was less than 7 m in altitudinal extent, while 100 m might have been expected on the basis of climatic warming to that date. Follow-up measurements at transects in six localities 11 years later showed that, despite impressive continued growth of established saplings at tree line edge over that time, the average altitudinal gain was less than 0.5 m for saplings and 1.6 m for forest canopy margins from the original measured boundaries (Wardle et al. 2005). In an independent study of silver beech (*Nothofagus menziesii*) tree lines in North Westland in the late 1990s, Cullen et al. (2001) concluded that recruitment of trees was controlled by disturbance and that, despite the warming that had occurred since 1950, no net movement of the tree line was apparent.

Wardle et al. (2005) considered explanations for the relatively small altitudinal extension of tree lines in New Zealand, and attributed this either to a lag in which environmental hurdles such as winter frosts or lack of mycorrhizal infection prevented seedling establishment or to the current tree line having established during a warmer-than-19th-century interval some centuries ago (i.e. the tree line is already adjusted to the current 20th-century warming). Figure 2 shows the temperature observations from the tree line at subantarctic Campbell Island/ Motu Ihupuku, where tree line expansion has been comprehensively studied (Bestic et al. 2005; McGlone et al. 2007). Although mean annual temperatures

Figure 2. Seasonal temperature changes: subantarctic Campbell Island/Motu Ihupuku. A split was made in the series to contrast the warmer decades after 1966 with the cooler preceding decades. Analysis and graph: James Barringer 2009.



have increased by about 0.4°C since 1966, the monthly variability perhaps explains why the tree line has not responded: that is, the increase in growing season temperature is still small compared with year-to-year variability, and not biologically significant enough to lead to an upward extension of the tree line. A global survey of tree lines (Harsch et al. 2009) has shown that only half are advancing (although none appear to be retreating). A possible explanation given by these authors is that infrequent outbreaks of cold weather causing mortality in young plants, especially in winter, may prevent many tree lines from showing altitudinal advances, despite an overall warming trend (Holtmeier & Broll 2005; Harsch et al. 2009). In other words, tree lines may not be controlled by mean growing season temperature.

Although tree lines in New Zealand can be convincingly demonstrated not to have advanced, or to have risen by only a very small fraction of the altitudinal range suggested by increasing mean annual temperatures, there is no reason to assume this state of affairs will continue if mean temperatures continue to rise. However, the critical climate parameter about which little is known, is how the variability around that mean will change.

6.9.2 Alpine

Globally, arctic and alpine communities are at risk through increasing shrubby growth and loss of herbaceous taxa (Wilson & Nilsson 2009). The New Zealand alpine zone is extensive, species rich and has many range-limited and endemic plants and animals. Many alpine groups have radiated extensively in the alpine zone (e.g. buttercup, *Ranunculus* spp., and stoneflies, *Plecoptera* spp.). New Zealand has many unusual alpine organisms including a mountain parrot, the kea (*Nestor notabilis*), and high-altitude skinks and cicadas. Flatworms occur in alpine zones with 3 months' snow cover (Johns 1998). Warming that results in a loss of alpine environment could result in serious loss of indigenous biodiversity, particularly if it eliminates isolated patches of alpine terrain or opens them up to easier access or population increase by invasive species (Halloy & Mark 2003).

The alpine zone environment is highly variable: bare rocky sites, bogs and scree slopes exist in close juxtaposition to forest, tall shrubland, closed grassland and herb fields. Snow cover is highly variable from site to site, in the course of a year and between years. Many plants and animals are either adapted to snow cover or strongly affected by it. In turn, snow cover interacts with the temperature regime to control freeze-thaw cycles, avalanches and snow melt freshets.

As described above, tree lines have not moved significantly on the New Zealand mainland despite the warming of last century. However, the retreat of glaciers over that time has been consistent, with a 0.6°C rise in mean annual temperature (Chinn 1996; Hoelzle et al. 2007), although with considerable fluctuation around the trend (Chinn 1995). Snow accumulation is highly variable, with far less during positive, La Niña ENSO phases (McKerchar et al. 1996). There has been little seasonal trend in snow storage since 1930 (Fitzharris & Garr 1996), but recent surveys have indicated that the end-of-summer snowline is 130 m higher than the position needed to maintain glacier ice mass, which has shrunk 50% over the last century (www.niwa.co.nz/our-science/climate/news/all/2008-09-15; viewed 1 February 2010) and is now the lowest on record. End-of-summer snowline is a composite of winter snow accumulation and summer ablation and therefore not a simple metric. Ablation is approximately twice as significant as accumulation in determining snowline. Moreover, the complex interaction of atmospheric pressure, prevailing wind flow, and temperature that controls snowfall means that future winter snow lie may not necessarily be related strongly to the glacier line. It is possible that, aside from annual temperature, the snow and frost aspects of the alpine environment that are crucial to many alpine organisms may have changed little.

Not all organisms found in the alpine zone belong to species found only in that zone. For instance, about 40% of the plant species that regularly occur in the alpine zone are also found below tree line, and a significant number occupy structurally similar habitats at sea level (Mark & Adams 1995). The true cold-specialised alpine species occur in the higher, more extensive alpine areas where, even with considerable warming, open habitat is likely to remain. In contrast, many restricted alpine endemics tend to be found in isolated upland sites and, while potentially threatened by increased woody growth, are often confined to habitats such as bogs, shingle slides, bluffs and outcrops, which may persist even under an upward moving tree or shrub line. Vegetation change is slow in montane grasslands (Lee et al. 2000; Lloyd et al. 2003) and in high alpine cushion communities (Mark & Wilson 2005). Therefore, it is by no means clear whether warming has already affected the distribution of alpine plants or whether, within limits, continued warming will do so.

Little work has been done to examine the effects of climate change on alpine animals. The few examples studied suggest their habitat requirements will be complex if for no other reason than the year-to-year fluctuations in seasonal cover of snow (Sinclair et al. 2001). A study of alpine grasshoppers (White & Sedcole 1991) demonstrated how complex the effect of warming could be on organisms (e.g. grasshoppers) with variable-length life cycles and different morphological stages; but their conclusion was that warming would lead to populations shrinking and being displaced to higher altitudes and marginal habitats. A study of microhabitats of an alpine cockroach showed that decreased snow cover led to increased freeze-thaw cycles in the alpine zone and thus, paradoxically, warmer winters with less snow resulted in a much more stressful climate (Sinclair 1997, 2001). Overall, there is no basis for simplistic assumptions about the results of warming on alpine organisms, and counter-intuitive outcomes are possible (Bannister et al. 2005; Sinclair & Byrom 2006)

6.10 SUBANTARCTIC AND SOUTHERN OCEAN ECOSYSTEMS

The southern ocean has warmed by 1.0–1.3°C between AD 1915 and 2000 (Hilton et al. 2006) but has not warmed substantially since the mid-1980s, although variability remains high. Primary productivity of the open ocean may have fallen in concert with rising temperatures (Gregg et al. 2003). However, both warming and biological responses are spatially variable (Whitehouse et al. 2008), with most warming in the New Zealand sector since 1950 occurring south of 50°S (Gille 2002). Predictions are for major warming of the southern ocean and for southwards movement of the westerlies (Gille 2008). Bergstrom & Chown (1999) suggested that climate change effects will be profound in southern oceanic islands; in particular, through altering conditions to favour invasive organisms.

The New Zealand subantarctic islands are biotically important as they are home to a number of endemic organisms, including the endangered New Zealand sea lion (*Phocarctos hookeri*) and substantial breeding populations of seabirds. The argument has been made that ocean warming has led to a decrease in productivity in subantarctic waters, resulting in declining population size of most monitored seabird and seal breeding sites since the 1960s (Weimerskirch et al. 2003; Hilton et al. 2006; Le Bohec et al. 2008). Close observations of breeding success in the New Zealand sea lion population on Auckland Island since 1995 have shown a 31% decline in pup production since 1998, but this is attributed largely to epidemics and bycatch of females in fishery operations (Chilvers et al. 2007). Even during poor years for pup production, starvation does not appear to play a role in their deaths (Childerhouse et al. 2005). A decline in wandering albatrosses (*Diomedea exulans gibsoni*) on Auckland Island since the 1970s has been attributed to mortality from fishing operations (Walker & Elliott 1999), as have declines in both black-browed (*D. melanophrys*) and grey-headed (*D. chrysostoma*) albatrosses on Campbell Island/Motu Ihupuku (Waugh et al. 1999). This seems likely, given that other albatross populations on Macquarie Island, unaffected by fishing, have remained stable (Terauds et al. 2005). On the other hand, no fishery or terrestrial influence on breeding has been reported for rockhopper penguins (*Eudyptes chrysocome*) on Campbell Island/Motu Ihupuku, yet their numbers have fallen by 94% since the early 1940s. Their greatest decline occurred with sustained warming between 1945 and 1955, and reduced food abundance along with warmer oceans is implicated (Cunningham & Moors 1994). A detailed study of rockhopper penguins (a species that has shown marked declines at nearly all breeding sites) from around the southern ocean showed significant changes in their diet from AD 1840 through to the present (Hilton et al. 2006). But, as the authors concluded, the population declines do not agree perfectly with shifts in productivity or temperature, and thus there is no single, global explanation for them.

Scrub expansion has been very noticeable on Campbell Island/Motu Ihupuku since the late 1940s and was originally attributed to the warming that accelerated at about that time (Rudge 1986; McGlone et al. 1997). However, recent more detailed work has shown that scrub is merely reoccupying the prominence it had before farming in the late 19th and early 20th century and, as it has not exceeded its previous pre-farming altitude, warming is unlikely to have driven the expansion (Bestic et al. 2005; McGlone et al. 2007). Whether there will be other climate-related changes in the plants of the subantarctic islands remains an open question. Whinam & Copson (2006) reported a substantial decline in moss (*Sphagnum*) on Macquarie Island, which was attributed to higher wind speeds and warming, but most other changes seem to be the predictable results of elimination of herbivores such as goats (*Capra hircus*), cattle (*Bos taurus*), rats (*Rattus* spp.), and rabbits (*Oryctolagus cuniculus*).

6.11 SEA-LEVEL RISE AND COASTAL ECOSYSTEMS

Sea-level rise will be the most important consequence of global climate change for coastal ecosystems, although warmer ocean temperatures may also have marked effects.

Soft shores (beaches and mudflats) are likely to be more severely affected by sea-level rise than hard (rocky) shores. Where coastlines are in a more or less undeveloped state, soft shore coastal ecosystems are likely to adjust naturally to rapid sea-level rise, albeit with some spatial reconfiguration. New areas of estuary and marshland habitat will generally be created and replace the inundated areas. However, this is unlikely to occur along most New Zealand soft shorelines. The extensive development of land close to sand dunes, marshes and estuaries means it is unlikely that people will readily allow new areas of dunes, marshland or estuary to form behind those now present. The most probable human response to sea-level rise will be to protect assets and infrastructure by erecting new hard barriers to prevent erosion, planting sand dunes to stabilise them, and infilling encroaching wetlands and installing new drainage. This scenario (often termed 'coastal squeeze' in international literature) means that rising sea levels will probably remove large areas of the rich biological habitat represented by existing coastal dunes, estuaries and marshlands.

Ecosystems typical of dune and dune slack systems, estuaries, and a range of saline, brackish (mixed saline and fresh water) and freshwater lagoons, shallow lakes and marshes will be most affected. Changes in estuaries will affect mangroves, salt marshes, and seagrass meadows, while in shallow lakes and marshes the changes will impact various macrophyte[4] assemblages (of emergent, submergent or floating aquatic plants). These coastal ecosystems are highly productive, and are also exceptionally important breeding, resting and feeding sites for indigenous marine mammals, fish and birds (Burns et al. 1990; Turner & Schwarz 2006). Several New Zealand sites are also important habitats for migratory birds and are of international ornithological significance (e.g. Miranda and Kaipara Harbour).

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[4] A macrophyte is a large aquatic plant as distinct from microscopic algae.

Estuarine systems and coastal brackish lakes and marshes are already under severe and increasing stress from human activities because their catchments are a focus for human settlement and the ultimate repository of the sediment created by land uses in the catchments of the waterways that feed them. Sedimentation, eutrophication and pollution by waste resulting from human activities and development are important current drivers of biotic depletion. Mangroves have been expanding seawards in New Zealand as a result of increased sediment influx, largely from intensification of human activities in the coastal zone (Harty 2009). Coastal dune systems have been widely stabilised by exotic species and developed for agriculture, forestry and settlement.

Sea-level rise will compound these effects. Loss of productive estuarine habitats and biota is likely to accelerate, with the more visible ecological effects being reduced populations and altered migratory patterns of coastal birds, and declines in certain marine fishes. Economic impacts of current estuarine quality decline will also be accentuated (e.g. through depletion of commercial and recreational fishery stocks such as snapper (*Pagrus auratus*), whitebait and some shellfish). Removal of species' habitats by coastal squeeze is already apparent in coastal dunes in some places.

General warming and extreme warm events will also affect coastal ecosystems, and, in particular, estuaries. Warming could extend the potential range of mangroves, but threaten biologically important seagrass (*Zostera* spp.) meadow habitats, which are possibly sensitive to low salinities at high temperatures (Burns et al. 1990).

7. Climate change and exotic pests, weeds and disease

7.1 MAST SEEDING AND PREDATOR IRRUPTIONS

Mast seeding is the intermittent production of large seed crops by a population of perennial plants (Kelly 1994), which creates a variable supply of resources that can have marked downstream effects on ecological communities. The phenomenon is especially common in New Zealand plants. The intensity of masting is commonly measured and expressed in terms of the interannual coefficient of variation (CV, the standard deviation divided by the mean) of reproductive output (e.g. seed, fruit or flower production).

Various theories are advanced to explain both short-term patterns and the evolutionary selection of mast seeding. For New Zealand species, the best supported explanation of short-term temporal patterns of masting combines resource accumulation minima in the plants with the existence of a synchronising environmental cue (Kelly 1994; Rees et al. 2002). Resource accumulation increases with time since the last mast seeding event, and may be enhanced by greater photosynthate accumulation or additional soil resources (e.g. higher nitrogen mineralisation rates; Richardson et al. 2005) in warm summers. Schauber et al. (2002) showed that out of 18 species with long seeding records, 17 were synchronised (*Dacrydium cupressinum* being the exception). Synchronising cues vary among New Zealand's masting genera (Schauber et al. 2002): a warmer-than-average summer seems to cue mast flowering and seeding the following year in tussock grass (*Chionochloa* spp.) and southern beech species (Rees et al. 2002; Schauber et al. 2002).

Both wind pollination efficiency and predator satiation may select for evolution of masting (Kelly 1994). Mast seeding increases pollination efficiency in southern beech, but makes little difference to tussock grass pollination (Kelly et al. 2001). However, masting increases predator numbers in tussock grass (Kelly & Sullivan 1997; Kelly et al. 2000; Kelly & Sork 2002). Hence, mast seeding seems to arise from different types of evolutionary fitness benefits in different genera.

7.1.1 Influence of mast seeding on predator cycles

Masting has an important influence on New Zealand's mammalian predator cycles and, ultimately, on their native vertebrate prey, especially native birds, but also lizards (Hoare et al. 2007), bats (Pryde et al. 2005) and large native invertebrates such as ground wētā (Smith & Jamieson 2003). Stoats (*Mustela erminea*) are a major concern because of their severe effect on many indigenous birds; but rodents (rats, *Rattus* spp. and mice, *Mus musculus*) are also predators, in their own right, of small birds, nestlings, eggs, lizards and invertebrates.

The general pattern is now fairly well understood (Ruscoe et al. 2006; Kelly et al. 2008a). First, rodent numbers commonly increase steeply following mast seeding. In general (exceptions exist), populations of mice, and sometimes rats, increase rapidly following the seeding of beech (Choquenot & Ruscoe 2000; Blackwell et al. 2003; Fitzgerald et al. 2004). Rats increase following the fruiting of rimu in podocarp forests (Harper 2005; Murphy et al. 2008). An outbreak of mice has also been measured following mast seeding in alpine tussock grassland (Wilson et al. 2006; Wilson & Lee 2010). Second, top predators usually increase in number in response to the higher availability of rodents as prey. Such irruptions are especially striking in stoats, because precocious fertilisation enables very rapid population responses to food availability (King & Murphy 2005). Finally, the exotic predator (stoat), which is sustained by the exotic prey (rodents), can place unusually high predation pressure on native vertebrates—that is, on birds and lizards. This pattern is called *hyperpredation* (Courchamp et al. 2000) and is a special form of *apparent competition*. The general pattern can, however, become very complex. For example, mice and rats are both preyed on by mustelids and are also major predators of invertebrates, eggs and lizards. Rats also prey directly on birds. The control of mustelids can result in rapid population increases (release) in these mice and rats (mesopredators[5]), with devastating outcomes for native birds and lizards (Tompkins & Veltman 2006; Ramsey & Norbury 2009).

In New Zealand, some studies suggest that the greatest impacts on native prey species occur when rodent abundance is declining after major irruptions, because there is a functional response whereby top predators (i.e. stoats) increasingly turn to feed on secondary (native) prey (e.g. Murphy et al. 2008). This 'prey-switching' is thought to occur because mammals are forced to increase their consumption of other prey when they are abundant and when the preferred plentiful food is exhausted. However, most research and, in particular, that from the South Island, provides little evidence of a functional prey-switching response in top predators, but suggests the greatest impacts on native prey occur when top predator abundance is highest. For example, native prey (birds) may form a relatively constant proportion of the diet of stoats throughout the mast cycle (Murphy & Dowding 1995). In other words, impacts on native prey may be related to absolute numbers of top predators rather than the availability of alternative exotic prey. Both prey-switching and predator abundance situations probably occur, but in different ecological circumstances (Kelly et al. 2008a). However, the distinction may be important for predicting the consequences for native prey of potential climate change-driven alterations in plant masting patterns (as discussed below).

7.1.2 Will mast seeding become more frequent?

Recent publications discuss the possibility that more frequent mast seeding events, and/or more common and less-variable seed production may result from the warmer summer days, and, perhaps, more intense ENSO events predicted in New Zealand under climate change (Yeh et al. 2009). There are two implications for biodiversity:

1. Management of predator cycles on native prey could become more difficult and demanding. Although mast cycles are increasingly predictable as their drivers become better understood, and flowering observations can, to some extent, give warning of heavy seed yields, it remains extraordinarily difficult to avoid the negative impacts on native prey species.

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[5] A mesopredator is a middle-ranked predator in a given food web.

2. More frequent seeding, and associated reduced variability in seed volume per seeding event (because plants will have less time between flowering events to accumulate reserves), may reduce the reproductive success and, hence, fitness of populations. For example, some studies suggest that reduced seeding variability may increase seed predation in alpine tussock (McKone et al. 1998; Rees et al. 2002).

The potential effects of warming depend to a great extent on whether or not a given species can alter the temperature at which a seeding cue is generated (Kelly et al. 2008b). Both beech and alpine tussock exhibit more pronounced masting (higher variability) in cooler, higher-elevation environments (Sullivan & Kelly 2000; Kelly et al. 2001). In alpine tussock species, higher altitude populations flower more heavily at any given temperature—evidence that species can adjust to site temperatures. Consistent with this prediction, Richardson et al. (2005) reported an increase in the frequency of mast seeding events in high-elevation mountain beech communities in recent decades, and also a decrease of years with near-total failure of seeding. However, such changes in mast seeding are unlikely to be large in magnitude. If resource accumulation minima are needed for reproduction, plants will not be able to respond to every higher-than-long-run-average temperature cue. A short-term trade-off between reproduction and resource accumulation will limit the ability of individual plants to increase heavy seed crop frequency in response to rising temperature. This could mean that the level of synchrony among plants will decrease, because more frequent warm signals will allow each plant to flower as soon as it reaches its internal threshold (McKone et al. 1998; Rees et al. 2002). The population level consequence of this reduced level of synchrony is less extreme variability among years in the available seed crops, making it easy for seed predators to maintain higher population densities. However, the effect may not be initially all that great. As an example (assuming lapse rates are 1°C per 100 m elevation), a 2°C increase in mean temperature might lower flowering variation to those now seen at elevations 200 m downhill; this difference is relatively subtle (e.g. Richardson et al. 2005). Such an adaptation of a masting response to new local conditions was demonstrated in Mark (1965) and Greer (1979), who reciprocally transplanted and 14 years later re-transplanted two snow tussock species (*Chionochloa macra* and *C. rigida*) between lowland (10 m) and mountain (910–1590 m) sites in Otago.

7.1.3 What would be the effect of more frequent mast seeding?

Even if masting regimes were to radically alter, the likely impacts on native prey species are uncertain. There might be different consequences depending on the system, and when the greatest impact on native prey occurs. If most of the impact on native prey occurs when rodent abundance is declining after major irruptions (i.e. there is a prey-switching functional response by predators), more frequent mast events will logically increase the frequency of episodes of intense predation on native prey. Alternatively, a change bringing less-intense (although more frequent) flowering episodes may also reduce the intensity of prey-switching functional responses. In other words, the surplus of hungry stoats would be smaller as rodents declined from lower peaks. On the other hand, a change to lower masting variation could bring more continuous flowering and resource inputs for rodents, and could lead to more constant and higher base populations of rodents and stoats, including more ship rats (*Rattus rattus*) in South Island beech forests, which are mainly free of them at present. If predation on native species is more closely related to predator numbers than cyclical changes in exotic prey abundance, impacts on native prey may increase.

In summary, we suggest that the climate change-driven alterations in the mast seeding patterns of native plants will be subtle, rather than dramatic. Such alterations are unlikely to dramatically reduce the fitness of populations of mast seeding native plants in the short or medium term. However, the consequences for native prey species in forests and grasslands are potentially complex and impossible to predict. Changes in mast seeding patterns may well increase pressure on native prey in indigenous forests and grasslands through their effects on predator cycles. Many native species exist in small isolated populations (e.g. long-tailed bats, *Chalinolobus tuberculatus*, kākā, *Nestor meridionalis,* mohua) and are particularly vulnerable to predation by stoats and rats. Even small alterations in predator population sizes and distribution may be enough to extirpate some of these threatened species. This would add to the generally increased pest pressure that climate warming is likely to bring.

7.2 MAMMALIAN PREDATION

Predation by introduced predators has been the major, or at least proximate, factor in driving numerous birds, reptiles, amphibians and invertebrates to extinction or severely restricting their ranges in New Zealand (Clout & Lowe 2000). Fast-breeding mammals, particularly rats, stoats, ferrets (*Mustela furo*), possums (*Trichosurus vulpecula*) and cats (*Felis catus*) have been the main culprits, although hedgehogs (*Erinaceus europaeus*), pigs (*Sus scrofa*), dogs (*Canis familiaris*), wasps (*Vespula* spp. in particular) and some birds (e.g. thrush, *Turdus philomelos*) have had effects on some populations at various times.

The primary predators in New Zealand are mammalian and therefore little affected directly by temperature in comparison to food supply—hence the wide distribution of many of these animals (rats, possums and stoats). However, warmer, drier winters are thought to extend the breeding seasons of rodents, goats, pigs and possums, and while this alone would not necessarily increase their population sizes in the absence of increased food, it would permit them to recover more quickly from control operations (Hay 1990). Nearly every part of the lowland and montane environment is saturated with mammalian predators and, as discussed above, while climate-driven fluctuations in food supply may initiate destructive predator-prey cycles, it is not clear whether climate change *per se* is going to increase their pressure on vulnerable indigenous organisms. A study of survival of long-tailed bats in southern New Zealand (Pryde et al. 2005) indicated that survival was lower when predator numbers were higher and winter temperatures warmer. The authors offered two explanations: the warm winter temperatures could have increased bat activity at a time when food was scarce, thus depressing survival; or, warmer temperatures may have affected rodent (the key predator) survival and food supply (beech seed), thereby increasing predation intensity.

The alpine zone provides habitat for a number of threatened indigenous birds, reptiles and invertebrates. It seems possible that the prevailing cool alpine climates, high rainfall and snow have limited mammalian predation on these species. Certainly, the survival of the takahē at high altitudes in Fiordland and the kākāpō (*Strigops habroptilus*) in the cool, very high rainfall areas of the same region—both species which, in pre-human times, had a widespread distribution throughout lowland New Zealand—suggest that these areas offer some protection.

Warming of the alpine zone may reduce this limited protection for alpine animals. As an example, rock wrens (*Xenicus gilviventris*) live in areas between 900 and 2500 m in altitude, and are highly vulnerable to predation by stoats, rats and mice (Leech et al. 2007). Stoats have been shown to be better suited to alpine grasslands than once thought, and it has been demonstrated that they can subsist on an almost exclusive invertebrate diet (Smith et al. 2008), meaning that a rapid decline of vertebrate prey would not necessarily release survivors from predation pressure. Stoats are thin with short fur, and thus vulnerable to cold and wet conditions and need warm, dry nests (King & Murphy 2005). It is theorised that ship rats are limited by mean monthly temperatures below 2°C, which may restrict foraging for food (Studholme 2000). They are abundant throughout New Zealand forests but become increasingly uncommon with altitude (Innes 2005), although they are abundant in alpine grassland on Macquarie Island (Shaw et al. 2005) and have been occasionally recorded in alpine grassland on the New Zealand mainland. If the current trend of large fluctuations in temperatures on a seasonal or multi-annual basis continues in the alpine zone, stoats and ship rats, which are fast-breeding and short-lived, and disperse rapidly, would be ideally placed to create damaging predation pulses in warm years. If beech and alpine tussock mast seeding years coincide with warm years, the combination of abundant food at the tree line, and warm conditions above it, could prove devastating. Sinclair & Byrom (2006) gave an example of a decline to near extinction of an alpine marmot in British Columbia through inverse density-dependent predation. They concluded that it was the original reduction of the population through loss of alpine habitat due to climate change that made it hyper-vulnerable to predation.

Rabbits are favoured by warm, dry climates and sustain damaging levels of predators such as feral cats and ferrets in dry tussock grasslands, as well as posing significant threats to native herbs and shrubs (Nugent & Flux 2005). Predicted warmer temperatures and declines in rainfall to the east of the Southern Alps/ Kā Tiritiri o te Moana will consequently increase overall rabbit and associated predator pressure on indigenous ecosystems.

7.3 INVERTEBRATE PREDATION AND HERBIVORY

New Zealand already has more than 2000 exotic invertebrates (Brockerhoff et al. 2010). In contrast to mammals, invertebrate predators are directly sensitive to climate. Given that the vast majority of invertebrates occur in the tropics, subtropics and warm temperate zones of the world, warming in New Zealand cannot help but increase the range and pressure exerted by invertebrate pests. Nevertheless, the projected warmer climate of New Zealand in 2080 will only slightly expand the area of the globe from which new pests might invade (Watt et al. 2008).

Relatively few invertebrates appear to have invaded natural ecosystems in New Zealand (Ridley et al. 2000). Invasive invertebrates have inflicted major damage on agricultural and horticultural systems, but there is little evidence that natural ecosystems have suffered much. However, not much research has been carried out on this topic. If there has been little damage, it may be, in part, because of the inherent ability of native vegetation to resist invertebrate herbivory (Brockerhoff et al. 2010). It is also possible that invasion of complex natural ecosystems may be slow, resulting in a long lag time before serious effects become apparent.

The major introduced invertebrate predators in New Zealand are the social and paper wasps and ants. Wasps are implicated in reduction of indigenous invertebrate abundance through direct predation, and in reducing bird populations through competition for nectar, honeydew and invertebrates, and attacks on nestlings (Clout & Lowe 2000; Beggs 2001). Common wasps (*Vespula vulgaris*) in New Zealand attain some of the highest densities recorded anywhere in the world (Barlow et al. 2002). Wasps are highly responsive to climate conditions: wet winters with flooding do not favour nest survival and can lower populations, while warm, dry conditions are ideal for explosive population growth (Beggs 2001). However, a crucial factor appears to be the negative effect of late-September rainfall (Barlow et al. 2002), underlining how sensitive pests can be to aspects of the annual weather cycle.

Ants are a major biodiversity threat and are abundant in warmer areas overseas (Dunn et al. 2009). The big-headed (*Pheidole megacephla*) and Argentine (*Linepithema humile*) ants, which are regarded as among the worst invasive species in the world as they have the capacity to wreak havoc on the native arthropod fauna, are already present in the exotic fauna. Continued warming and drying eastern climates are likely to encourage their spread. The temperature match between New Zealand climates and the current global distribution of an ant species is a good predictor of its invasion potential (Lester 2005). On this basis, at least six or more species are likely to invade (Harris & Barker 2007).

7.4 HERBIVORY BY INTRODUCED MAMMALS

Herbivory, mainly from introduced deer (*Cervus* spp., *Odocoileus virginianus* and *Dama dama*), possums, goats and hares (*Lepus europaeus*), as well as cattle, sheep and rabbits (in modified areas), has been responsible for reducing palatable plant species and encouraging the proliferation of browse-resistant species, both exotic and native. Although browsing pressure has yet to drive any known plant to extinction, extreme range reduction and modification of ecosystem functioning is occurring (Nugent et al. 2001), although not in any straightforward or predictable way (Wardle et al. 2001). Over 50 years of study of the major mammalian herbivores—deer, possums and rabbits—has come up with few generalisations as to herbivore influence on the vegetation, since impact is variable from year to year and place to place. Herbivore impact is modified by soil fertility, and other influences on vegetation cover. Impact is therefore often patchy, with major destruction of palatable species in some areas and little in others (Payton 2000).

CO_2 changes are likely to alter the relative abundance and foliage palatability of plants (Körner 2003). In general terms, plant species will respond individually to CO_2 enrichment, and their growth and distribution changes will depend on factors such as soil nutrients and climatic preferences. But, overall, carbon-based nutrients will tend to increase while nutrients associated with other elements will decrease, so foliage quality will usually fall. Whether introduced herbivores will respond by consuming more of the low-quality foliage or focus on higher quality foliage is unclear.

With respect to the broader issue of climate change and herbivore pressure, there is little information suggesting one way or another that average climate is the significant controlling factor on mammal population size (although long, cold, wet winters tend to lead to increased mortality among most mammalian species; see entries in King 2005). Changes in climate that result in alteration of the abundance of plant food are likely to be more significant than direct effects of climate on survivorship. There is some evidence that mammals will take advantage of warmer conditions to extend their ranges. The alpine zone may come under increased pressure as mammal populations are more likely to respond dramatically at the extremes, and reduced snow lie may make larger areas vulnerable to browsing pressure for longer. For instance, at present, brown hares are found sparsely in the alpine zone up to 2000 m. They currently have a variable influence on herbs and woody plants (Norbury & Flux 2005), but a surge in numbers may significantly increase the threat they pose to certain endangered plants (Gruner & Norton 2006). A population of rabbits has established at an altitude of 1800 m on Mt Ruapehu, well above the normal altitudinal limit of 1000 m for this species, and it is suggested that reduced snow lie may have led to this range expansion (Flux 2001).

7.5 NATURALISATION AND SPREAD OF WEEDS

The arrival of new weeds and increased invasiveness of existing weeds is one of the most troubling likely consequences of climate change. All other factors being equal, the warmer a region, the greater the number of plant species. For instance, in the Americas, a latitude shift from 40°S to 25°S results in more than an order of magnitude increase in the number of woody species (Weiser et al. 2007). As frost is a key factor separating low- and high-diversity regions (Woodward 1987), once it ceases to be significant, a much greater range of plant species will be able to compete with local species.

At a broad scale, local species richness appears to be no barrier to further invasions as native and exotic richness is correlated (Sax 2002). Plant and animal communities should be regarded as undersaturated with species.

The net outcome of CO_2 increase and global warming is predicted to be higher local plant diversity, nearly entirely driven by naturalisation of cosmopolitan weeds (Woodward & Kelly 2008). From a conservation point of view, weed species number *per se* is not the primary problem as, theoretically, a very large number of species can coexist. An ecosystem in which native species dominate biomass and processes, but in which a large number of exotic species find a home, is not ideal, but tolerable. Much more important is the handful of ecosystem transforming weeds, which dominate to the extent that previous native occupants are reduced to low numbers and all essential ecosystem processes are controlled by non-indigenous plants. Here we can expect a knock-on effect in which native specialist animals (both above and below ground) are also reduced to a remnant. One needs no further example than what has happened in Hawai'i, Mauritius and other oceanic islands where tall woody exotic shrubs and trees completely dominate large areas of the landscape (Denslow 2003). In Hawai'i, expansion of exotic trees such as *Psidium guajava*, *P. cattleianum*, *Melastoma candidum* and the nitrogen-fixing *Falcataria moluccana* have created a situation where even without disturbance 'the native components of [lowland forests] may soon no longer be self-sustaining' (Zimmerman et al. 2008). Clearly, this is about as bad a biodiversity outcome as can be imagined.

Nearly as many naturalised vascular plant species occur in New Zealand as named native species (c. 2200), and there is a much larger (> 30 000) reservoir of species grown in gardens and as crops that potentially could naturalise (Williams & Cameron 2006). A survey of aquatic plants within the New Zealand aquarium and nursery trade revealed over 180 species (many of which are weeds elsewhere) that have yet to naturalise. The majority are tropical species, unlikely to survive, but some temperate environments pose high potential risk (Champion 2004). The rate of naturalisation is not slowing; therefore, the pool of genotypes from which the weeds of the future will appear continues to grow. Weeds are hard to manage and control operations tend to be expensive. Broad-spectrum herbicides used on an infested area without targeting will do as much or more damage to native plants as to the weeds and, given the propensity of weeds to expand into areas where competitors have been eliminated, such control is often ineffective. On the other hand, locating and killing individual plants by physical or targeted chemicals, and then monitoring and returning to ensure regeneration from seeds, underground structures or fragments of clonal plants is prevented, is very expensive. Successful total elimination of weed species in New Zealand has been confined to limited incursions. Aquatic weeds pose as many challenges in control as terrestrial weeds and successful control is notably rare (Willby 2007).

Biocontrol has been deployed successfully against a few weeds and has reduced the danger posed by others. However, some weeds are both devastating ecosystem-transforming invaders, and highly unlikely candidates for biocontrol. Prime among these are those conifers (e.g. *Pinus contorta*, *Pseudotsuga menzesii* and *Larix decidua*), which have the potential to extend closed forest above the native tree line, and markedly higher growth rates than native tree competitors (Ledgard 2001; Waring et al. 2008). The current outlook in New Zealand is not good.

Weed assessment frameworks and bioclimatic and plant trait models are now widely used with good success (> 80%) to predict which plants are likely to become invasive (Gordon et al. 2008; Potter et al. 2009). The key criterion is whether a plant species in similar environments elsewhere has become invasive. However, totally accurate prediction of naturalisation and invasiveness is difficult, as there are many influential non-bioclimatic factors such as genetic adaptation, intensity of introduction effort, and interaction with local pathogens or pests. Some weeds that were major New Zealand weeds or considered likely to become so (for instance, sorrel, *Rumex acetosella*, in the 1960s) are no longer regarded as serious weeds, while others have become much more prominent (e.g. hawkweeds, *Hieracium* spp.).

Not all weeds currently invasive in New Zealand will benefit from climate change (e.g. exotic broom, *Cytisus scoparius*, which favours cooler climates; Potter et al. 2009), but the large number of cold-sensitive species already in the country, and the very large numbers that could invade, argue for the balance being towards more and more highly invasive weeds. The warmer and more northerly areas of New Zealand are where the phenomenon of 'greenhouse weeds' should first manifest. Williams (2008) suggested that there are possibly 100 weeds that potentially could threaten Northland, and 17 weed threats (including five trees) in Northland that are already in the exponential growth phase. While not strictly weeds, C_4 pasture grasses have undesirable characteristics in New Zealand and there is evidence that some of them are spreading. The most convincing observations are for paspalum (*Paspalum dilatatum*), which is expanding from Northland southwards. Field & Forde (1990) suggested climate warming between 1978 and 1990 may have expanded the region in which 40% or more of the pasture contains this species by over 1.5° of latitude.

The most cost-efficient way of containing weeds is to prevent them establishing or, failing that, preventing them reaching the exponential spread phase (Mack et al. 2000). An example of how difficult this is to achieve is provided by woolly nightshade (*Solanum mauritianum*), an important woody weed in warm-oceanic climates. A review of control efforts undertaken in the Bay of Plenty (Stanley 2003) concluded that its biology was poorly known, control efforts had been unsuccessful and expensive ($6m), the plan of eradicating or substantially reducing the weed unrealistic, and monitoring and follow-up control after operations hit-and-miss. Additionally, in focussing on a single weed, incursions of other important weeds (e.g. kiwifruit, *Actinidia* spp., and lantana, *Lantana camara*) were not discovered soon enough, and their control was delayed.

Whole-landscape approaches to weeds are the only ones likely to succeed and, in particular, strategies that are focussed on controlling or eliminating weeds in sensitive areas or preventing them becoming established. Most weeds likely to become greenhouse weeds are already in the country, and have been introduced for ornamental purposes or as warm climate crops (e.g. kiwifruit).

The social and economic resistance to pre-emptive control should not be underestimated. For instance, in Auckland, mere discussion of listing bangalow palm (*Archontophoenix cunninghamiana*, distribution Queensland to just south of Sydney) in the Auckland Regional Pest Management Strategy because of its preference for habitats also preferred by New Zealand's endemic palm nikau (*Rhopalostylis sapida*) led to organised protests and legal threats from nursery owners (Thompson 2006).

There has been much emphasis in the recent literature on the need to develop better tools (often statistically based) to identify weeds with a high propensity to become invasive (Gordon et al. 2008). However, this presupposes an effective surveillance mechanism to both keep weeds out and to identify and react quickly to the early stages of spread, along with widespread public acceptance of the need to eradicate potential weeds ahead of necessity. Past history and current practice does not provide any comfort that this phase of the operation will be effective, because prevention of a problem rarely gains any notice (aside from a disquieting trend to curtail efforts too soon) and lessons do not seem to be learned from heroic but pointless attempts to control invasions. We must continue to invest in the development of tools that can contain damaging environmental weeds (such as biocontrol and improved conventional approaches for area operations) and to develop a better social awareness of the highly threatening future of today's valued garden ornamentals.

7.6 DISEASE, PATHOGENS AND THEIR VECTORS

New Zealand may come under pressure from novel diseases and vectors as the climate warms. For example, New Zealand had only 12 mosquito species before human settlement (compared with 32 in England and 67 in Japan, both countries of similar climate and size). Four exotic, potentially disease-spreading mosquitoes have since established, but at least 30 other species have been intercepted at national entry ports (Derraik & Slaney 2007).

On the other hand, infectious diseases are part of complex networks of species and their ultimate effects depend on their virulence, levels of resistance and adaptation in the affected populations, the transmission rate of the disease, and population density, among other factors. The link between climate and infectious disease is therefore complex. Lafferty (2009) argued that climate change may not lead to a net increase in the geographic distribution of infectious disease because:

- Diseases and pathogens have lower and upper climate bounds, so the most likely outcome is range shifts, not range expansions
- A reduction in habitat will reduce a disease or pathogen range, while increase may not, because other factors such as dispersal barriers, competition and predation affect the realised niche
- Habitat degradation and species loss may prevent transmission of infectious diseases that depend on other species as intermediate hosts or for vectoring

Therefore, infectious diseases could, in fact, be highly sensitive to biodiversity loss through reducing pathogen spread, and the geographic distribution of some infectious diseases may decline with warming, while others may be promoted by proliferation of their infective stages (Poulin 2006). For instance, while much of the indigenous avifauna have had no exposure to avian malaria,[6] some exotic birds (e.g. blackbirds (*Turdus merula*) and sparrows (*Passer domesticus*)) have, and can provide a reservoir for infection of native birds whose numbers are too low to sustain a disease outbreak (Tompkins & Gleeson 2006).

Ecological understanding of vectors and reservoirs may therefore be a key to predicting effects of climate change (Lafferty 2009). Invertebrates are the animal group most sensitive to changes in temperature, and small changes in temperature, for example 1–2°C, can result in changes in population distribution and abundance (Poulin 2006). There are two key vector mechanisms in New Zealand:

- Invertebrate vectors, e.g. biting insects
- Invertebrate intermediate hosts, e.g. aquatic snails

As with other species, distinct diseases may react differently to increases in temperature. For instance, some trematodes from South Island locations release more infective stages at elevated temperatures, and some fewer (Dukes et al. 2009; Koprivnikar & Poulin 2009). Movement of species results in ‘new encounters’ that could be particularly hazardous because native species which have not previously been exposed to the disease may have little resistance. Native species may become new vectors. As an example, the indigenous mosquito *Culex pervigilans* is the likely vector of avian malaria parasites in mohua (Derraik et al. 2008).

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[6] Some migratory bird species have had exposure to avian malaria.

8. Biodiversity consequences of climate change mitigation and adaptation activities

New Zealand needs to act to reduce its greenhouse gas emissions and adapt to the alterations climate change will bring. Some mitigation and adaptation activities in New Zealand could also produce benefits for biodiversity (e.g. well-planned and resourced indigenous afforestation projects and control of herbivores). However, many (perhaps the majority) pose additional threats to biodiversity. Indeed, it seems likely that damage to New Zealand's biodiversity from its mitigation and adaptation activities will be greater than damage from climate change itself over the next 50 years. New Zealand contributes about 0.3% of global greenhouse gas emissions (MfE 2009), but makes a proportionately larger contribution to global biodiversity. While reducing national greenhouse gas emissions and adapting to climate changes impacts are vital for national and global wellbeing, these activities should not result in adverse impacts on national and global biodiversity values. Below we outline potential benefits and negative impacts on biodiversity associated with reforestation, energy generation and irrigation.

8.1 REFORESTATION

Establishing a forest on non-forested land as a means of carbon sequestration is central to the New Zealand mitigation effort. Its effects on indigenous biodiversity will vary according to where it is done and whether indigenous or exotic species are used and, in the case of indigenous reforestation projects, how well the reforestation projects mimic natural forest composition and structure.

8.1.1 Native forest restoration and regeneration

Numerous private organisations and local bodies are already engaged in forest restoration. Although the total contribution from restoration is likely to be small in area, its impact at the local ecosystem level could be considerable, especially if it manages to restore indigenous ecosystems in areas where they are largely missing. The greatest benefits for biodiversity from reforestation will arise if new native forests are established in lowland areas, particularly on fertile arable land that is connected to steeper, higher, less-fertile land. This will best improve overall biodiversity and ecological processes, especially for larger mobile organisms with varied habitat requirements, and for narrow-range habitat specialists of fertile warm sites. The lowest cost and ecologically most desirable approach is to encourage regeneration of existing shrublands through to tall forest. Increasing the amount of land in regenerating native forest should aim to increase connectivity between current forest patches, because small isolated fragments of forest suffer exaggerated losses of biodiversity through stochastic effects, increased predation, higher levels of weed invasion, and reduced genetic diversity and interchange compared with continuous tracts (Timmins & Williams 1991; Ewers & Didham 2008). Doing so will also provide for more and larger populations of many species, which will permit them to adapt and evolve in the future (Woodruff 2001). Although larger areas of forest will possibly only delay rather than prevent loss of many indigenous vertebrates and large invertebrates through predation, the prospects for other biotic groups (e.g. plants and small invertebrates) are better.

8.1.2 Exotic forests—planted and wilding

Mature exotic plantations nearly always contain indigenous species, including birds, invertebrates and plants (Allen et al. 1995; Ogden et al. 1997). Exotic forests are likely to support more indigenous species than some other land use types, such as intensively managed exotic pasture, cropping or horticulture. However, the evidence for the contribution of exotic plantations to indigenous biodiversity is patchy, and often depends on limited data, such as counts of species from a few groups.

Exotic plantations have several important drawbacks for biodiversity:

- Isolated forest plantations with few linkages to larger areas of indigenous forest achieve only limited diversity as they are too small and uniform to ensure the full range of biodiversity-supporting biological processes
- Commercial exotic forestry plantations are periodically felled, negating most interim biodiversity gains
- There is loss of indigenous biodiversity where exotic forests are planted (or wildings spread) into indigenous communities such as tussock grasslands or shrublands
- Such plantings also preclude or defer the potential future biodiversity gains to be made through natural regeneration of native forest

Many exotic tree species are novel functional types in New Zealand and are ‘ecosystem transforming’. Their effects include:

- Reducing catchment water yield, with negative effects on stream flow and freshwater biodiversity, and the functioning of seasonally wet ecosystems (e.g. ephemeral wetlands; Johnson & Rogers 2003)
- Stabilising previously dynamic systems (e.g. pines on coastal dunes and broom on river beds) with consequent loss of indigenous flora
- Invading areas (alpine and drylands) where native forest was either absent or limited
- Creating flammable (pyrophylic) forest communities, leading to a higher risk of wildfire than from pasture or native shrubland and forest (exotic conifer and gum plantations create the equivalent of North American and Australian forests), and consequent destruction of exposed native vegetation remnants

Persistent offsite effects on biodiversity are possible, especially from wilding tree spread from exotic plantation forests into native grassland and shrubland plant communties (including above the natural tree line), where they displace a range of native fauna and alter ecosystem processes. Wildings generate a substantial ongoing drain on public budgets: the cost of control increases rapidly with time, and liability for control is difficult to establish and enforce. Sterile tree crops that can be planted without wilding risk have been slow to emerge.

8.1.3 Domestic biofuel plantations and crops

Intensification of land use contributes significantly to biodiversity decline, especially in lowland to montane landscapes. Several tree crops (e.g. willows, *Salix* spp., and sycamore, *Acer pseudoplatanus*) are being contemplated for New Zealand's domestic biofuel manufacture on 'marginal lands' where, like other exotic forestry plantations, they will often conflict with maintenance of biodiversity. Remnants of undeveloped marginal lands (e.g. rough pasture, mixed exotic/native shrubland or steep riparian margins) close to transport corridors in more developed landscapes are likely to be targeted for such crops, because the crops are bulky and must be transported and processed once harvested. Although often highly modified, semi-natural communities, they nonetheless provide refuges for remnant populations of indigenous lizards, birds and plants. For example, old willows in South Canterbury provide critical roost sites for the only regional population of long-tailed bats (Sedgeley & O'Donnell 2004). Like exotic forestry plantations, biofuel tree species may also generate wilding tree problems unless limited to sterile crops. Non-tree plants are being contemplated as biofuels. For instance, elephant grass (*Pennisetum purpureum*; a highly invasive species in other countries) has been recently approved by the Environmental Risk Management Authority (ERMA) for release (sterile varieties only).

8.2 ENERGY GENERATION

As part of a transition to a decarbonised energy future, development of renewable energy sources is likely to increase. Construction and operation of generation facilities and transmission lines will increase pressure on biodiversity.

8.2.1 Hydroelectricity

Hydroelectricity development through dams has permanent impacts on biodiversity. Original habitat is lost in the flooded zone behind the dam. Draw-down zones of low productivity unsuitable for many biota are created by lake level fluctuations behind the dam (although lakes may create new habitat for some species). Indigenous diadromous fish populations (e.g. whitebait and long-finned eel, *Anguilla dieffenbachii*) are depleted through loss of connectivity between feeding and breeding habitats. Current mitigation technologies are, at best, partially effective, and effective ones may be economically unfeasible (e.g. plant shut down and free spilling in the eel migration season), especially for enabling downstream migration.

Loss of habitat below a dam can be more of a problem. Impeded or pulsed water flows along parts of the river, termed 'hydropeaking', effectively decrease the bed width of habitat for biota within the river. Less sediment deposition and entrapment from dam to river mouth reduces fine sediment deposition rates in particular. This causes riverbed 'armouring', which reduces three-dimensional habitat structure and hence productivity within the riverbed below the variable zone. Other types of hydroelectricity development such as diversion races also have an impact on biodiversity through alteration of lake and river levels. Many new hydroelectricity developments will also add to demand and likely reduce the amount of water to available for ecosystem needs.

In Doubtful Sound/Patea, the tailrace from Manapouri Power Station altered natural freshwater inputs and created intertidal habitats unsuitable for populations of bivalves (Tallis et al. 2004). Increased juvenile mortality in the bottlenose dolphin (*Tursiops truncatus*) population in the Sound corresponded with the opening of the second tailrace tunnel in Deep Cove, which brought additional cold freshwater inputs. Although the cause of mortality is unknown, it may have been a consequence of thermal stress (Currey et al. 2007; Rowe et al. 2009).

8.2.2 Land-based wind power

The environmental impacts of land-based wind generation are often thought to be less severe and permanent than hydroelectricity development, but the potential impacts of wind power stations on birds, both migratory species such as cuckoos and local populations of species such as falcon (*Falco novaeseelandiae*), remain largely unquantified and are potentially severe (Powlesland 2009). Also, wind turbines are generally individually very large, and require construction of large platforms and major roading to erect and service them, so construction impacts on local biodiversity can be considerable (e.g. through erection of footings, transport and storage of materials and cut and fill for roading). Wind power stations in natural areas can fragment very large areas of hitherto intact indigenous habitat, provide permanent routes and vectors for weed and pest invasion, and introduce constant human disturbance (e.g. maintenance traffic), which can result in a range of negative effects on ecosystems and species. Effects on biodiversity may be disproportionately large where turbines or associated infrastructure (e.g. roading) are positioned along bird migratory routes or are sited on or near rare ecosystems of high significance for biodiversity (e.g. proposed wind power stations on Epakauri Conservation Area in Northland and on the Mt Cass ridge in Canterbury).

8.2.3 Energy from marine sources

Marine energy development potentially involves wind, wave and tidal energy devices. All are still relatively new, so little is yet known about their potential and realised effects on biodiversity. Most attention in other countries has been on potential wildlife impacts, especially on cetaceans and seabirds, and very little work has yet been done in New Zealand.

In Europe, marine wind power stations (usually horizontal axis turbines with 2–3 rotor blades 20–40 m long, mounted on tubular, 60–90-m-tall towers embedded in the sea floor) are furthest ahead of the three development types (Wright et al. 2009) and are hazardous to seabirds (Evans 2008). Cetaceans will potentially be displaced or harmed during exploration and construction phases through intense noise during pile-driving, drilling and dredging operations (Weilgart 2007), increased vessel activities and increased turbidity and re-suspension of polluted sediments. Other impacts may occur through ongoing operation of marine wind power stations through noise and vibrations emanating from the turbines, contamination of water (e.g. by lubricant leaks), entrapment or entanglement in or collision with the structures; changes to prey and food webs; electromagnetic impacts due to cabling that may be hazardous for elasmobranchs (sharks, skates and rays); and increased vessel traffic from maintenance operations (Jefferson et al. 2009).

Tidal power can be either tidal stream power or tidal range power. Tidal stream power uses completely submerged turbines and hydrofoil devices located in fast tidal currents where sea flows are compressed (e.g. at bay entrances, around headlands or between islands). The turbines generate high levels of underwater noise which likely to displace cetaceans and fish, and may cause physical hazards. Tidal range power uses tidal barrages (a type of dam spanning an estuary) to exploit the water level difference between high and low tides. These barrages destroy estuarine functioning (e.g. altered hydrological and sediment flow regimes disrupt estuarine ecosystems and hence food sources for fish and wading birds). Like other dams, they also disconnect feeding and breeding grounds for migratory fish and other biotic groups that depend on upstream or downstream penetration.

Overall, development of marine energy resources will add to existing pressures on coastal ecosystems and introduce new ones. Although the risks are unquantified, such devices are likely to have adverse effects on marine mammals, fish, seabirds and estuarine biodiversity.

8.2.4 Geothermal energy

Geothermal energy development in the Waikato and Bay of Plenty regions has already had marked environmental effects. Because of heat and fluid extraction for power generation and use of overlying terrain for infrastructure, three-quarters of New Zealand's geysers have been lost, with flow-on effects for their ecosystems (Environment Waikato 2006). The main concerns about increased geothermal generation with regard to biodiversity relate to effects on communities of thermophylic plants, animals and micro-organisms.

8.3 IRRIGATION

There is likely to be increased demand for irrigation from drier eastern regions (and for cheap electricity to power it). Abstraction for irrigation reduces water flows; reduces quantity and quality of feeding habitats for wading birds; allows predators access to islands previously safe for fauna; and increases vegetation cover, which, in turn, destroys open gravel nesting habitats and provides better cover for predators. The impacts of irrigation dams on biodiversity are similar to those constructed for hydroelectricity generation (discussed above) with an added risk of increased spread of invasive species.

9. Management recommendations

9.1 CONTEXT

In New Zealand, a 0.9°C rise in temperature and substantial rainfall alteration since the early 1900s has led to surprisingly few observed biotic or ecosystem changes that can be attributed to climate change (see Table 1). Nevertheless, there can be little doubt that climate change will have a significant effect on New Zealand biodiversity by the end of this century. Observations, experimental studies and models show that climate change, acting through a multitude of pathways including directly through temperature and CO_2 physiology, via timing and intensity of

TABLE 1. RECENT CLIMATE-RELATED BIODIVERSITY CHANGES.

TAXA	OBSERVED CHANGES	REFERENCES
Mammals	Long-tailed bats (*Chalinolobus tuberculatus*) declining possibly due to warm winter effects	Pryde et al. 2005
Mammals	Rabbits (*Oryctolagus cuniculus*) established above tree line on Ruapheu as a result of reduced snow cover	Flux 2001
Birds	Earlier (by 30 days) egg laying in welcome swallow (*Hiroundo tahitica*) between 1962 and 1995	Evans et al. 2003
Birds	Population decline in red-billed gull (*Larus novaehollandiae scopulinus*) linked to changes in circulation indices (PDO and ENSO) between 1983 and 2003	Mills et al. 2008
Birds	Decline in sooty shearwater (*Puffinus griseus*) populations throughout Pacific due to warming oceans and loss of productivity	Lyver et al. 1999; Shaffer et al. 2006; Scott et al. 2008
Birds	Declines in yellow-eyed penguins (*Megadyptes antipodes*) since 1980 linked to long-term climate-related trend towards lower ocean productivity	Peacock et al. 2000
Birds	Decline rockhopper penguins (*Eudyptes chrysocome*) by 94% since early 1940s linked to rising ocean temperatures between 1945 and 1956	Cunningham & Moors 1994
Fish	Decline in recruitment of *Anguilla* spp. and several weeks' advance of main migration over last 30+ years possibly linked with changing thermal fronts in spawning grounds	Jellyman et al. 2009
Reptiles	Tuatara sex ratios increasingly male with increasing temperature	Mitchell et al. 2008
Insects	Mosquitos (*Culex quinquefasciatus* and *Aedes notoscriptus*) may have been aided in southwards dispersal within New Zealand by two warm climatic 'windows' (early 1970s and late 1980s).	Laird 1995
Plants	Seed production increased in mountain beech (*Nothofagus cliffortioides*) along elevation gradient related to warming during flower initiation (1973–2002)	Richardson et al. 2005
Plants	Spread of paspalum (*Paspalum dilatatum*) southwards from the northern North Island by 1.5° latitude between 1978 and 1990	Field & Forde 1990

extremes (such as frosts), and indirectly via food supply and altering competitive interactions, has the potential to markedly alter the abundance and interactions between indigenous organisms, to change optimal ranges and thereby affect community composition and distributions. Warmer temperatures will increase the potential for invasion by exotic biota and naturalisation of some already in the country, and alter range sizes and abundance of many already here.

There are large uncertainties around all predictions of biodiversity scenarios in future climate-change affected environments. These uncertainties arise from three sources: uncertainty around future climate states (Meehl et al. 2007); weak ability to predict biodiversity reaction to climate change (Tylianakis et al. 2008); and uncertainty around anthropogenic influences. It is well understood that annual averages for temperature and rainfall are not as important for biodiversity as seasonal averages, and that infrequent climatic extremes can exert a strong influence. Climatic indices such as SOI, SAM and PDO have marked effects on local climates and can be demonstrated to affect growth and abundance of organisms (e.g. Mills et al. 2008). None of these are well characterised in climate scenarios for greenhouse futures. It follows that models, scenarios and predictions for biodiversity will be even more poorly characterised. Even if climate models could produce more precise seasonal and index forecasts, the complexity of biodiversity interactions defies the production of anything other than rather coarse community-level scenarios. Regardless of the amount of effort devoted to biodiversity future casting, it is unlikely to be more than an input into scenario generation, and its main function to scope the range of possible futures. Similar constraints apply to forecasting anthropogenically driven changes to landscape and biota. However, despite these uncertainties, we can be sure that climate change will be pervasive throughout New Zealand and affect all biological systems at all levels.

Actions taken to adapt to, or mitigate, the climate change threat may inadvertently increase biodiversity loss if they involve ill-considered changes in land use. New Zealand's actions to combat the current biodiversity crisis are insufficient (Green & Clarkson 2005). Government actions over the past decade clearly show that preventing biodiversity loss takes second place to economic growth. Regional councils have a mandate for biodiversity conservation, but have insufficient funding to allow them to undertake more than a limited range of actions. The provisions of the Resource Management Act do not encourage a wider national view and are very much focussed on resolving local issues. Increasingly, the conservation of biodiversity is being evaluated, both in New Zealand and worldwide, through a lens of 'ecosystem services'—in particular, the contribution of biodiversity towards sustaining tourism, recreation, water resources and carbon sequestration—not just for its intrinsic values. The underlying premise is that where protecting components of biodiversity results in ecosystem services that have a significant financial value, then this will provide a strong basis for funding the maintenance and enhancement of these components.

This relative inaction, or apparent lack of concern, in the face of ongoing biodiversity loss, makes climate change-specific recommendations for the distant future (c. 50 years) often appear naive. If New Zealand is not investing sufficient resources or political capital in the fight to stop ongoing loss, why would it invest further to avert potential, but uncertain, future loss? Policy recommendations for mitigating or adapting to the biodiversity consequences of future climate change therefore have to be made with regard to current expenditure limits and, where possible, based on redirection of existing effort.

A recent review of 22 years of scholarly articles on climate change and conservation makes sobering reading. Heller & Zavaleta (2009) concluded that about 70% of the 113 types of recommendations made in these articles were of a general nature, based on ecological reasoning rather than specific research, case studies, or empirical data, and are largely non-specific with regard to biome or region. Climate change recommendations made in the literature generally posit a much greater degree of national awareness, investment, coordination and management than those already deployed in response to the current biodiversity crisis. There is a fair amount of scepticism from some leading ecological laboratories as to the urgency of the threat relative to others, and our ability to do anything useful about it:

> *There is unnecessary emphasis on climate change. It is one of several major threats and it is doubtful if it is the major threat. Ironically, it is the threat we can do least about! Our country's* [Australia's] *obsession with predicting the impact of climate change on biodiversity rather than using decision-making tools to choose between actions to adapt to climate changes will leave us little time to act.* (Possingham 2009: 2)

We agree with this scepticism to some extent. The current climate change scenarios for New Zealand are not particularly troubling from either a human or biodiversity point of view, and major impacts are possibly 50 years in the future. However, abrupt, significant climate change is still possible in the near future, as the rapid warming of 0.7°C in a single decade (1945–55) of the last century demonstrates. Therefore, there is no option but to accommodate this reality into our planning (Heller & Zavaleta 2009). Most of our recommendations centre on preparedness rather than action. That is, on how to strengthen our understanding and monitor change, rather than undertake mitigation or adaptation.

The best defences against biodiversity loss as a consequence of climate change are exactly those actions that we are undertaking right now to prevent biodiversity loss due to ongoing pressures of pests, weeds, and land use change. Therefore, it is not our intention here to canvas all the possible actions that could be taken because most are ones that we should be undertaking in any case. We also argue that from a research point of view, an excessive focus on climate and biodiversity is also misplaced. What is needed is a functional understanding of species responses to a wide variety of pressures and a much better understanding of how biodiversity maintains itself at a landscape level. We therefore only mention those few actions that are justified now as part of a climate-change-specific policy.

Our analysis of the situation leads us to divide policy and management recommendations into two groups: those that are specifically targeted towards the direct consequences of climatic change and more generally help 'future proof' our biodiversity against a range of threats including climate change; and those that are directed towards reducing the potentially negative biodiversity effects of activity leading towards reducing greenhouse gas emissions.

9.2 RECOMMENDED RESPONSES TO MANAGE THE DIRECT EFFECTS OF CLIMATE CHANGE

9.2.1 Coastal ecosystems

Most natural threats to coastal ecosystems, including sea-level rise, are greatly exacerbated by the fact that intensive, ongoing developments are located too close to the sea to accommodate natural changes. When a hard inland boundary (natural or anthropogenic) prevents coastal ecosystems from responding to habitat loss brought about by erosion and/or sea-level rise, the situation is termed 'coastal squeeze' (Schleupner 2008). As sea level rises, the incidence of coastal squeeze will accelerate. Policy focus must therefore be on reducing or preventing anthropogenic hardening of vulnerable coastlines. Purchase of land for coastal reserves ahead of the need, removing inappropriate anthropogenic hard structures, and prevention of further hardening of the coastal area through new construction of housing, marinas, roads and sea walls are clearly priorities. While this involves a daunting level of planning and coordination, there is a high degree of certainty as to the timing and rate of increase of sea level, and little doubt as to the subsequent biodiversity effects (Rosier 2004; Hume & Blackett 2007; Ramsey 2007).

Local governments (particularly district councils) currently have the key role in implementing adaptation and mitigation policies for coastal ecosystems, nominally under the guidance of the New Zealand Coastal Policy Statement (NZCPS, prepared by DOC). Actions recommended to councils to mitigate coastal hazards by Jacobson (2004) and in the proposed Coastal Policy Statement (DOC 2008) are broadly appropriate for mitigating and avoiding adverse impacts on biodiversity. They include avoidance of hard structures, planned and managed retreat from advancing coastal zones, and retirement of infrastructure.

These actions are technical challenges, but a far greater challenge lies in managing the human dimension of the problem. Inserting a greater degree of consideration for future biodiversity values into coastal planning is not without cost, and occurs in an already highly regulated and contested context. Planning conflict along coastlines is usually sparked by individuals and corporations asserting or claiming exclusive access to coastal resources, or wishing to modify valued components of the coast, such as dunes or beach morphology, for individual benefit. Although making place for movement of coastal ecosystems will ensure public access and *should* therefore have a measure of inbuilt, widely based support, trends are currently going the wrong way. Coastal development grew over the last 50 years, but escalated very rapidly in the last decade (Rosier 2004; Ramsey 2007) when pressure to develop the coast, coastal property prices and the value of associated infrastructure increased markedly (Hume & Blackett 2007). This significantly increased the financial implications of coastal hazards from sea-level rise, and stiffened resistance from private vested interests to the implementation of adaptation and mitigation policies by public agencies.

There is little evidence to date of an inclination for councils to tackle either the technical or human challenges. Therefore, there is an urgent need for central agencies to:

- Establish the importance for biodiversity of estuarine, marsh and dune coastal ecosystems in the minds of councils and the public

- Increase understanding of the current pressures on coastal biodiversity, and the compounding impacts of climate change-induced sea-level rise and likely human adaptation responses

A programme of public land purchases and/or public funding for land retirement agreements will likely be needed to enable managed retreat from advancing coastal zones and retirement of infrastructure. Coastal land is generally expensive, and resistance and holdouts from private parties can be anticipated. Therefore, successful programmes will require strong cooperative partnerships between conservation agencies and local government regulators as well as careful planning, coordination and management, including (and perhaps especially) the social dimensions.

A systematic appraisal of the location and intensity of threats to coastal biodiversity from sea-level rise is urgently needed to guide and justify actions and programmes to mitigate them. A priority is to map future coastlines (based on high-resolution digital elevation models (DEMs) and geological estimates of uplift/subsidence) and areas of coastal squeeze in relation to infrastructure and land use.

Recommendation 1: Systematic appraisal of coastal ecosystems by councils or central government is required to determine where threats will be most intense.

Recommendation 2: Acquire, reserve or place conditions on land to prevent hardening of vulnerable areas of coastline, and to make space for inland migration of coastal ecosystems.

9.2.2 Freshwater ecosystems

While the immediate biodiversity threats relate to dams, water abstraction, and waterway and lake pollution, climate change will severely impact compromised elements of freshwater biodiversity through increasing heat stress, drought, changing the season of maximum water flow and increasing the need for irrigation. New Zealand is already facing severe energy and water allocation issues, and further biodiversity degradation of waterways in future is therefore likely.

While many lakes and rivers are buffered by their very size from some of these impacts, much freshwater biodiversity is contained in relatively small streams, seepages and springs that are at high risk. Nevertheless, some of the worst climate impacts can be avoided if minimum water flows are maintained and riparian vegetation cover encouraged (Maxted et al. 2005; Barquin & Scarsbrook 2008).

Recommendation 3: Maintain minimum water flows in small streams and springs and encourage riparian vegetation cover.

9.2.3 Alpine habitat reduction and change

Because of the large uncertainty concerning tree line response, the current need is not so much for planning for rising tree lines, but research into the factors that maintain them, and monitoring to ensure early warning signals of upwards movement are not missed. All growth in the alpine zone is slow, so monitoring does not have to be frequent and the current 5-yearly remeasurement of transects seems sufficient (Wardle et al. 2005; Mark et al. 2006).

Halloy & Mark (2003) made brief mention of two mitigation measures to counter shrinking alpine habitat effects. The first is to remove trees and tall shrubs to maintain open habitat; the second is to transfer endemic species trapped in shrinking patches to other more secure habitats in the same region. It is easy to envisage that both might become necessary if and when habitat reduction becomes apparent. As discussed, tree line movement and shrub growth in the alpine zone is likely to be slow, so there will be enough time to assess, plan and act. However, some key aspects of the alpine habitat might change without any warning whatsoever. Snow lie is highly variable from year to year (Clare et al. 2002), and is important for some snow-bank and ice patch specialists. Likewise, annual minimum temperatures are highly variable, are likely to be a key influence on the distribution of some indigenous specialists, and are a key restraining influence on the spread of exotic weeds and pests.

While it is not difficult to imagine managers keeping a few key sites free of competing indigenous woody growth where endangered species are at risk (as similar activities are carried out already to prevent loss of certain plants), wholesale loss of snow and ice across a region cannot be compensated for by habitat manipulation. Many alpine species appear to have low dispersal rates (McGlone et al. 2001) and exacting habitat requirements. If it is deemed necessary to move species to a more secure location because of widespread habitat loss, or invasion by exotic predators or weeds, the undertaking will be expensive and success uncertain. As much warning as possible must be allowed for. It would therefore be prudent to maintain early warning sites where the possibility of loss of habitat could be assessed on a regular basis, and to document the biodiversity elements of most value and most at risk (because of specialist requirements and restricted range; for example see Dickinson et al. 2007). A start on such a system has begun in New Zealand—Global Observation Research Initiative in Alpine Environments (GLORIA; Mark et al. 2006).

Recommendation 4: Increase monitoring of alpine and upper montane zones (including on subantarctic islands) to identify potentially rapid zone changes, vegetation changes, and pest and weed pressures.

9.2.4 Indigenous biota and ecosystem range changes

Novel climates are predicted for the end of the 21st century, and ecosystem disruption and species extinctions are likely as new species enter and current occupants vanish (Williams et al. 2007). Much of the international concern has focussed on the probability that warming will occur too rapidly for many species to be able to migrate to their preferred niche, leading to suboptimal outcomes for some ecosystems, which will lack their characteristic complement of species, or for species stranded in increasingly unsuitable environments (Brook et al. 2008). The fear is that this will lead to widespread disruption, exacerbated by other drivers to create enhanced extinction risk. Recommendations for adaptation in Northern Hemisphere regions have therefore emphasised provision for closely juxtaposed or connected habitat pathways to permit climate-driven movement (Heller & Zavaleta 2009).

Range change poses less risk to New Zealand than in many other biological regions because of the country's variable, oceanic climate and mountainous topography. The biota is adapted to marked seasonal fluctuations, in particular drought, has relatively few species in the lowlands adapted to extreme seasonal cold, and projections for the medium-term future do not suggest prolonged heat waves that could directly impact species. It is more likely that competitive relationships will change, affecting local abundances, but even this should be ameliorated by continuing variability. Restricted range, cool climate specialists are almost without exception positioned on topographically controlled climate gradients. There are a number of cool climate outliers—for instance in the Waikato Basin on isolated mountain peaks—but unless they prove to be unusually distinctive and valuable genotypes, there will be little point in making provision for them. What seems more likely is a pattern of patchy but dramatic change as a consequence of extreme events impacting areas and ecosystems in a vulnerable state. We anticipate that the most important changes of this type will occur in drier, more heavily deforested lowland regions such as the eastern coastlines of both the North and South Islands.

A consequence of continuing natural and human-mediated dispersal within New Zealand is that indigenous plants and animals are now occurring well outside their natural ranges. With continuing warming, it is highly likely that such 'naturalisations' will accelerate, and are likely to involve spontaneous self-introductions from Australia and the Pacific as well. For instance, a large number of plants and animals have invaded unaided since human settlement including orchids, 16 birds (e.g. silvereye, *Zosterops lateralis*, and welcome swallow; Tennyson 2010), and butterflies and dragonflies (McGlone 2006).

Some thought should be given to the effects of indigenous natural movement, indigenous translocation, and unassisted immigration in the context of climate change. It is often tacitly assumed that we should allow nature to take its course with regard to natural fluctuations in temperature but intervene in the case of anthropogenic warming. However, these are now not so easily distinguished. Moreover, we have so transformed the environment that any climatic change—natural or anthropogenic—can have detrimental effects. As we have seen, the potential environmental niche of most species is far wider than the realised niche. Novel species interactions may therefore occur that are without historical precedent, and thus the potential for biodiversity loss exists (through competitive exclusion, hybridisation, disease, predation or parasitism). We now have the potential for three distinct situations with regard to native species:

- Spread of anthropogenically translocated species, with or without warming
- Natural spread of species, with or without warming
- Anthropogenic translocation of species in the belief that they will fare poorly unless moved to a new location

There is a well-entrenched concept in plant restoration circles termed 'ecosourcing', which asserts that only species from within the local area should be used to enrich or restore plant ecosystems (Godley 1972). Problems envisaged are that species and individuals from more remote locations may be more aggressive, displacing local species or ecotypes and, more importantly, that new genetic ecotypes or species may interbreed with the same or closely related species, thus swamping local genetic diversity (Molloy 1995). Introduction of indigenous plant species from outside their current range is therefore regarded as something to be avoided (Simpson 1992) and use of individuals obtained more than a few kilometres from the restoration site is discouraged, even if the species range encompasses the site. Such fears have some foundation: for instance, in the Wellington district, the strictly northern trees karaka, pōhutukawa (*Metrosideros excelsa*) and karo (*Pittosporum crassifolium*) are all aggressively spreading in local coastal forests (Simpson 1997). Costall et al. (2006) recommended control of karaka in the southern North Island outside of its natural range on the grounds that it reduces understorey diversity. It is striking that such considerations have been rarely applied to birds, where the main concern now is to achieve genetic diversity in small populations (Robertson et al. 2009).

There are several core questions:

- Do we endeavour to restrict or maintain species distributions that may have been formed by historical events or dispersal limitations when climatic parameters clearly suggest broader ranges?
- Should we make a distinction between natural and anthropogenic movements?
- Should species be translocated to follow their shifting environment niche when the result may easily be weed- or pest-like consequences in their new range?
- Should species that have dispersed unaided from areas outside of the New Zealand region remain unmolested?

These questions cannot be answered by appeal to principles, but only by practical considerations. Unless range changes, unaided or anthropogenic, seem likely to do permanent and substantial harm to the biodiversity of New Zealand, they should be ignored. In effect, this is already the implicit policy with regard to exotic biota, and there is no reason why it should not apply to native biota.

Recommendation 5: Large climate-driven range or population changes of indigenous species have not yet been observed, and there is no reason as yet to suspect they will occur in the medium term. No action is called for aside from ongoing monitoring. Should such change begin, only when there is a serious threat to indigenous biodiversity (extinction or loss of unique ecosystems) should management responses be considered.

9.2.5 Invasive species

Climate change will put more pressure on the environment through permitting many more already introduced or naturalised species (for the most part weeds) to expand in range, and make New Zealand more vulnerable to invasive organisms not already in the country. Risk managers should take into account the high probability of warmer (and in the southeast) somewhat drier conditions in their deliberation as to the risk posed by potential and present invasive species. A list of species already present that are likely to be advantaged by warmer/drier climates should be assembled, and consideration given to risk reduction actions such as education campaigns, banning sale and proscribing the planting of high-risk plants or legislating for their removal. Examples are spot occurrences of invasive invertebrates such as ants or the presence in gardens of plants with major weed potential under a warming climate such as strawberry guava (*Psidium cattleianum*) or palms. Perhaps the greatest immediate risk is that of continued spread of lodgepole pine (*Pinus contorta*) above the tree line, as it is already well established above the limit to native tree line species in numerous locations (Ledgard 2001) and is likely to respond well to continued warming (Wardle 1985); and the growing threat posed by Douglas fir (*Pseudotsuga menziesii*).

Recommendation 6: Prepare a list of weed and pest species likely to become invasive when the climate warms, and a list of at-risk habitats, followed by pre-emptive action (banning, control and removal) with regard to high-risk pests or weeds.

9.3 RECOMMENDED ACTIONS TO MINIMISE THE NEGATIVE EFFECTS OF CLIMATE CHANGE MITIGATION AND ADAPTATION ACTIONS ON BIODIVERSITY

9.3.1 Reforestation

While reforestation is a potentially low-impact and low-cost means for New Zealand to comply with international agreements to reduce greenhouse gas emissions, the use of exotic forest species poses risks. Exotic plantations are generally cheaper and easier to establish than native forests, and they sequester carbon more rapidly (Davis et al. 2009). They are vastly more profitable. Given the serious threat to biodiversity from an increase in exotic afforestation, a clear case exists for specific interventions by government to provide:

- Incentives for the establishment of native rather than exotic forest
- Greater information on indigenous afforestation options for 'marginal lands' and the relative benefits of native reforestation for biodiversity
- Binding national policy obliging councils to address threats and risks of wilding trees for New Zealand flora, fauna, ecosystem services and landscapes
- For exotic plantation owners to bear the full cost of their negative externalities, including the onsite and offsite damage to biodiversity resulting from plantations

If enforced, these would reduce the financial incentive for exotic plantations (and provide a greater incentive for native plantations by comparison). Genetic methods now provide a cost-effective method to trace wildings to particular plantations and thus establish liability. However, a clearer legal process and precedents may be needed, including new or amended legislation.

Recommendation 7: As there is likely to be an expansion of forests in response to the need to sequester carbon (and the mechanisms to encourage this), existing regulations, permits and liabilities regarding exotic afforestation need to be strengthened to minimise impacts on native biodiversity, and more incentives given for native reforestation.

9.3.2 Expansion of renewable energy generation

Renewable energy is often presented as abundant, inexhaustible and environmentally responsible, but it will sometimes have severe and irreversible impacts on biodiversity. To minimise these impacts, comprehensive national assessment of the threats from different development types of energy development, and appropriate high-level planning and guidance to avoid areas important for biodiversity is needed.

Currently, a policy vacuum exists. There is no nationally consistent, comprehensive information about the likely scale and types of impacts from different energy developments, and there is no high-level policy that directs developments to where they will likely do least harm to biodiversity and other environmental services and features. The general pattern is for each controversial development to be fought case by case through the Environment Court. This situation is inefficient for all parties, imposing high costs on national conservation and developers alike.

Recommendation 8: Create a national-level set of guidelines to establish how renewable sources may be developed without unduly compromising indigenous biodiversity. This could possibly be achieved through the National Policy Statement on Biodiversity.

10. Underlying research needs

Maintaining and strengthening of the research effort currently going into biodiversity and ecosystem research will go a long way towards meeting the research requirements around understanding and mitigating climate change impacts. However, when we examine international research efforts directed specifically towards climate change, they have focussed to a large extent on three areas:

1. Demonstrating the reality and magnitude of biodiversity responses
2. Experimental manipulations of small plots (by warming the environment or increasing ambient CO_2)
3. Modelling potential biodiversity responses to future change

As a result, there is a large and rapidly growing literature concerning biodiversity and climate change. It has been effective in raising awareness of the likelihood of devastating biodiversity consequences of uncontrolled greenhouse gas emissions. Naturally, there is now an expectation that researchers will give some lead as to what might be done to mitigate and adapt to biodiversity consequences. With regard to mitigation and adaptation, the biodiversity and climate change work done to date is of only marginal use to managers (Heller & Zavaleta 2009; and see discussion in section 10.1).

Research recommendation 1: Undertake research monitoring of vulnerable ecosystems (in particular, coastal, alpine and drier lowlands) and consider establishing long-term ecological research sites/observatories (LTERs).

Demonstration of changes is useful, but it leaves open the question of what can or should be done to counter them. Experimental manipulations are necessary, if only to parameterise models, but the research situations are often highly constrained in size and scope by the experimental set-up, and results may translate poorly to a broader scale and longer durations. Modelling is also informative, but the models mainly project some way into the future (typically 20–100 years) under some form of equilibrium and thus have only limited relevance for the typically shorter timescale ecosystem that managers work at. Moreover, other potential or likely changes such as land use changes, spread of invasive species, species interactions, climate extremes and variable climate trends are not incorporated. Ecosystem managers mainly deal with current problems and are unlikely to intervene to create better outcomes in the remote future without substantial immediate pay-off. They need a better and more holistic understanding of the potential impacts, and better tools to achieve outcomes.

Although research directed towards informing and enabling practical steps to be taken to counter potentially damaging effects of climate and CO_2 changes is necessary, we should not neglect the huge scientific opportunity created by our inadvertent experimentation with the atmosphere. It is difficult to predict what the biodiversity consequences might be, but we can be sure they will occur and that they almost certainly will be large. It would be remiss of the scientific community not to set up comprehensive long-term observation sites, and other nations are undertaking ambitious schemes to do just this (e.g. Australia; Doran et al. 2003). Proposals in the past have foundered on the familiar problems of funding shortages, competition for funds, coordination problems and the propensity of scientific groups to set up new sites and facilities rather than endure the strains of working collectively. It would be hoped, given this is one of the three major and unprecedented ecological traumas of the last 100 years (the others being land use change and exotic invasions), that a way will be found to overcome these difficulties and work towards a common goal. DOC and other core government environmental agencies are key to the establishment of LTERs—only they have leverage to ensure that they are established.

Research recommendation 2: develop process models and their supporting parameter databases to improve understanding and predictions regarding population fluctuations in both declining indigenous species and exotic predators and herbivores.

In the case of individual species, process models, which use a mechanistic basis for their predictions, will be superior to ecological niche models for evaluating the effects of climate change. This is because the models can account for changing, novel combinations of environmental factors that are likely to be an increasingly important issue in New Zealand with the simultaneous impact of climate change, CO_2 increase, land use intensification and pests (Ramsey & Norbury 2009). Without a firm understanding of how populations are responding simultaneously to multiple factors, it is likely that control or restoration efforts will be inefficient.

Research recommendation 3: Develop tools for control of wilding spread from exotic reforestation plantation.

More than a century of intensive research into exotic forestry has given the best possible basis for understanding how to establish exotic plantations, but little as to how to cost-efficiently prevent spread into adjacent conservation land. Cost-effective tools to suppress exotic spread and development of sterile cultivars will be needed if more extensive plantations are to be pushed into land hitherto regarded as uneconomic for commercial forest as a response to Kyoto incentives.

Research recommendation 4: Develop scenario modelling to project and quantify the biodiversity effects of renewable energy schemes.

If the worst biodiversity impacts of renewable energy are to be avoided, nationwide whole-landscape approaches are needed. Some tools and much of the data needed for this work are already available. For instance, with regard to hydroelectricity development, scenario modelling can be readily used to assess the future impact of individual or multiple dams on the contribution of rivers to national biodiversity. It will be possible for New Zealand to avoid the largest impacts of new hydroelectricity development if it generally avoids rivers of highest value for biodiversity, and focusses on those systems that have already had their connectivity and flow regimes modified by the effects of dams.

Research recommendation 5: Develop decision-making tools for resource allocation to biodiversity conservation that include specific climate change components.

There is now a considerable amount of research going into creation of tools that assist in making decisions on the most efficient use of resources combined with securing the best biodiversity outcomes (Chee 2004; Wilson et al. 2009). It is likely that research effort will need to create new, New Zealand-specific tools if any investment is to be made to counter likely but uncertain effects of climate change.

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What are the likely impacts of climate change on New Zealand's terrestrial biodiversity?

Nearly every aspect of terrestrial ecosystem function in New Zealand will be affected by climate change in coming years. It is clear that change is inevitable. Therefore, we need to know how significant the consequences of climate change wil be for New Zealand biodiversity, and what might be done to ameliorate or adapt to them. This report reviews the past, current and possible future impacts of climate, increasing atmospheric CO_2 *concentration, and climate-related anthropogenic activities on New Zealand's terrestrial biodiversity, and makes policy and research recommendations on the basis of these findings.*

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