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Provision of Natural Habitat for Biodiversity: Quantifying Recent Trends in New Zealand

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1. Introduction

1.1 Biodiversity and habitat provision in New Zealand

The Millennium Ecosystem Assessment (MEA) found that over the past 50 years, natural ecosystems have changed more rapidly and extensively than in any other period of human history (Millennium Ecosystem Assessment, 2005). In the 30 years after 1950, more land was converted to cropland than in the 150 years between 1700 and 1850, and now one quarter of the earth's surface is under cultivation. In the last decades of the twentieth century, approximately 20% of the world's coral reefs have disappeared and an additional 20% show serious degradation. Of the fourteen major biomes in the world, two have lost two thirds of their area to agriculture and four have lost one half of their area to agriculture. The distribution of species has become more homogeneous, primarily as a result of species introduction associated with increased travel and shipping. Over the past few hundred years, the species extinction rate has increased by a thousand times, with some 10–30% of mammal, bird, and amphibian species threatened with extinction. Genetic diversity has declined globally, particularly among cultivated species.

A framework of ecosystem services was developed to examine how these changes influence human well-being, including supporting, regulating, provisioning, and cultural services (Millennium Ecosystem Assessment, 2003). While overall there has been a net gain in human well-being and economic development, it has come at the cost of degradation to many ecosystem services and consequent diminished ecosystem benefits for future generations. Many ecosystem services are degrading because they are simply not considered in natural resource management decisions. Biodiversity plays a major role in human well-being and the provision of ecosystem services (Diaz et al., 2006). For example, natural ecosystems provide humans with clean air and water, play a major role in the decomposition of wastes and recycling of nutrients, maintain soil quality, aid pollination, regulate local climate and reduce flooding.

New Zealand has been identified as a biodiversity hotspot (Conservation International, 2010). Located in the Pacific Ocean, south east of Australia, New Zealand covers 270 thousand square kilometres on three main islands (North, South and Stewart Island). It has a wide variety of landscapes, with rugged mountains, rolling hills, and wide alluvial plains. Over 75 percent of New Zealand is above 200 meters in altitude, reaching a maximum of

3,700 meters on Mount Cook. Climate is highly variable and has played a key role in biodiversity distribution (Leathwick et al., 2003).

As New Zealand has been an isolated land for more than 80 million years, the level of endemism is very high, with more than 90% of insects, 85% of vascular plants, and a quarter of birds found only in New Zealand (Ministry for the Environment, 2007). One of the most notable characteristics of New Zealand's biodiversity is the absence of terrestrial mammals, apart from two bat species, and the dominance of slow-growing evergreen forest. New Zealand's indigenous biodiversity is not only unique within a global context – it is also of major cultural importance to the indigenous Maori people. Maori have traditionally relied on, and used, a range of ecosystem services including native flora and fauna for food, weaving, housing, and medicines.

The isolation of New Zealand has preserved its unique biodiversity, but also rendered the biodiversity vulnerable to later invasion. When Maori migrated from the Pacific Islands, circa 700 years ago, predation upon birds began and much lowland indigenous forest was cleared, especially in the South Island. Rats and dogs were also introduced. The birds, having evolved in an environment free of predators, were susceptible to disturbance and many began to decline to the edge of extinction. When Europeans arrived in the early 19th century, they extensively modified the landscape and natural habitats. Large tracts of land were cleared and converted into productive land for pastoral agriculture, cropping, horticulture, roads, and settlements. Only the steepest mountain land and hill country was left in indigenous forest and shrubland. Swamps were drained and tussock grasslands were burned. Not only was the natural habitat significantly altered, but a large range of exotic species were introduced, including deer, possums, stoats, ferrets, and weasels, causing a rapid decline in native birds and degrading native forest. Other introduced plants and animals have had significant effects in the tussock grasslands and alpine shrublands, most notably rabbits, deer, and pigs, and the spread of wilding pines, gorse, broom, and hieracium. Despite significant efforts to control weeds and pests and halt the loss of natural habitat, around 3,000 species are now considered threatened, including about 300 animals, and 900 vascular plants (Hitchmough et al., 2005).

The Economics of Ecosystems and Biodiversity study (TEEB) suggested that it is difficult to manage what is not measured (TEEB, 2010). To prevent further biodiversity loss, decision-makers need accurate information to assess and monitor biodiversity. However, biodiversity assessment is not a trivial task. As defined by the Convention on Biological Diversity (CBD), biodiversity encompasses "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (CBD, 1992). Conceptually, biodiversity is a nested hierarchy comprising genes, species, populations, and ecosystems. In order to assess status and trend, these multiple levels need to be assessed simultaneously. Noss (1990) suggested a conceptual framework with indicators providing measurable surrogates for the different levels of organisation. Loss of extent is one of the many indicators in this framework, and it has been widely used internationally in reporting to the CBD (Lee et al., 2005). It is relatively easy to report, and has been recognised as one of the main drivers for biodiversity loss (Department of Conservation [DOC] and Ministry for the Environment [MFE], 2000).

1.2 Previous assessments in natural habitat

Several national surveys of vegetation cover have been completed. The New Zealand Land Resource Inventory was derived by stereo photo-interpretation of aerial photographs

combined with field work (Landcare Research, 2011). The survey scale was approximately 1:50,000 and had a nominal date of mid-1970. The legend included 42 vegetation classes, of which six were indigenous forests (coastal, kauri, podocarp-hardwood (lowland or mid-altitude), *nothofagus* (lowland or highland), and hardwood) and three were indigenous grass classes (snow tussock, red tussock, and short tussock). The Vegetative Cover of New Zealand was produced at the scale of 1:1,000,000 primarily from the NZLRI (Newsome, 1987). The small scale required mixed vegetation classes to be used, such as "grassland-forest" or "forest-scrub".

The Land Cover Database (LCDB) was derived by photo-interpretation of satellite imagery and has nominal dates of 1995–96 for LCDB1 and 2001–2002 for LCDB2 (Ministry for the Environment 2009). Indigenous classes included tussock grassland, manuka/kanuka, matagouri, broadleaved hardwoods, sub-alpine shrubland, and mangroves; however, different indigenous forest classes were not delineated and were lumped into one class of indigenous forest. Walker et al. (2006) used the LCDB to look at changes to natural habitat between 1995–96 and 2001–2002. They concluded that much of the highland natural habitats had been preserved since pre-Maori times, but also that much of the natural habitat of lowland ecosystems had been lost and continues to be lost. Limitations in the LCDB prevented reliable analysis of the changes in indigenous grassland, wetlands, and regeneration of shrublands to indigenous forest.

The recently completed Land Use Map (LUM) has extended the date range for indigenous forest to between 1990 and 2008 (Ministry for the Environment, 2010). LUM is primarily helping New Zealand meet its international reporting requirements under the Kyoto Protocol. It tracks and quantifies changes in New Zealand land use, particularly since 1990. For this purpose, it produced national coverages for 1990 and 2008 of five basic land cover classes (indigenous forest, exotic forest, woody-grassland, grassland, and other), from satellite imagery.

1.3 Proposed assessment of natural habitat provision

More recent work by Weeks et al. (in prep) has improved the accuracy and extended the analysis to between 1990 and 2008 on tussock grasslands. Ausseil et al. (2011) have improved the accuracy of wetland mapping and identified changes since pre-European time. These recent analyses, together with the LUM, permit a synthesis of information for assessing recent trends of natural habitat provision in New Zealand. This chapter presents this synthesis and describes a national measure of habitat provision for biodiversity. We look at New Zealand's natural habitat changes from pre-Maori to the present, and also at recent trends. We will focus this chapter on three natural ecosystems: indigenous forest, indigenous grasslands, and freshwater wetlands. The measure of habitat provision will combine information on current and historical extents with a condition index to quantify stress and disturbance.

2. Indigenous forests

Indigenous forests in New Zealand are generally divided into two main types. The first is dominated by beech trees (*Nothofagus*), and the second generally comprises an upper coniferous tier of trees with a sub-canopy of flowering trees and shrubs (the broadleaved species) (Wardle, 1991). However, these two types are not mutually exclusive and mixtures are common. Lowland podocarp-broadleaved forests are structured like forests of the

tropics. Kahikatea (*Dacrycarpus dacrydioides*) and Kauri (*Agathis australis*) are the tallest trees in New Zealand, and can reach up to 50 metres in height. At maturity these trees tower above the broadleaved canopy with other emergent podocarps like rimu (*Dacrydium cupressinum*), totara (*Podocarpus totara*), matai (*Prumnopitys taxifolia*), and miro (*Prumnopitys ferruginea*), to give the forest a layered appearance. Below the upper canopy many shorter trees, shrubs, vines, tree- and ground-ferns compete for space, and below them, mosses. Beech forests tend to be associated with southern latitudes and higher elevations, such as in mountainous areas, and are generally sparser than the podocarp-broadleaf forests. Their understory may contain only young beech saplings, ferns, and mosses.

Indigenous forests provide unique habitat for a large range of plants, animals, algae, and fungi. Since the arrival of Maori, circa 700 years ago and the subsequent burning of large areas of forest, and then Europeans from ~1840, who cleared large areas for farming and settlement, the extent of indigenous forest has significantly declined and, in combination with many introduced pests, has placed enormous pressure on the survival of many species. MfE (1997) reported that 56 of the listed threatened plant species are from indigenous forest habitats. Also, many of the seriously threatened endemic birds are forest dwellers: wrybill, kiwi, fernbird, kokako, kakariki, saddleback, weka, yellowhead, kaka, and New Zealand falcon.

The extent of indigenous forest in 2008 can be mapped using a combination of LCDB2 and LUM. Theoretically, the LUM contains a recent extent of indigenous forest. However, because the class definitions are land-use rather than land-cover based (for Kyoto Protocol), the indigenous forest class is not the same as the standard definition in LCDB2 and contains much indigenous shrubland yet to reach the maturity of a forest. Hence the LUM should only be used to report on changes to forest if the LCDB definition of indigenous forest is to be used. We therefore combined all the changes "from" or "to" forest in the LUM with LCDB2 to produce a recent extent of indigenous forest.

Figure 1 compares the extent of indigenous forest and shrubland in 2008 with the estimated pre-Maori historic extent, derived by combining LCDB2 and a historic map of New Zealand (McGlone, 1988). In the North Island, the area of indigenous forest has reduced from 11.2 million hectares to 2.6 million hectares. Most remaining indigenous forest is in the hills and mountains. In contrast to indigenous forest, indigenous shrublands have now become extensive, comprising over 1.0 million hectares. These shrublands often comprise a wide variety of indigenous shrub species and could naturally regenerate to indigenous forest if left. In the South Island, the area of indigenous forest has reduced from 12.0 million hectares to 3.9 million hectares, and, similar to the North Island, the remaining forest is mainly in the hills and mountains. At 0.6 million hectares, the area of indigenous shrublands in the South Island is as large as in the North Island.

The loss of indigenous forest between 1990 and 2008 may be assessed directly from the LUM. In the North Island, 29 thousand hectares of indigenous forest have been lost, and in the South Island, 22 thousand hectares of indigenous forest have been lost. The spatial location of this loss is important as some types of forest are better represented than others. We follow the method of Walker et al. (2006) who considered the area of indigenous forest remaining in land environments. The land environments are defined by unique combinations of climate, topographic, and soil attributes, and are a surrogate for unique assemblages of ecosystems and habitats (Leathwick et al., 2003). Four levels of classification have been defined with 20 level I, 100 level II, 200 level III and 500 level IV environments.

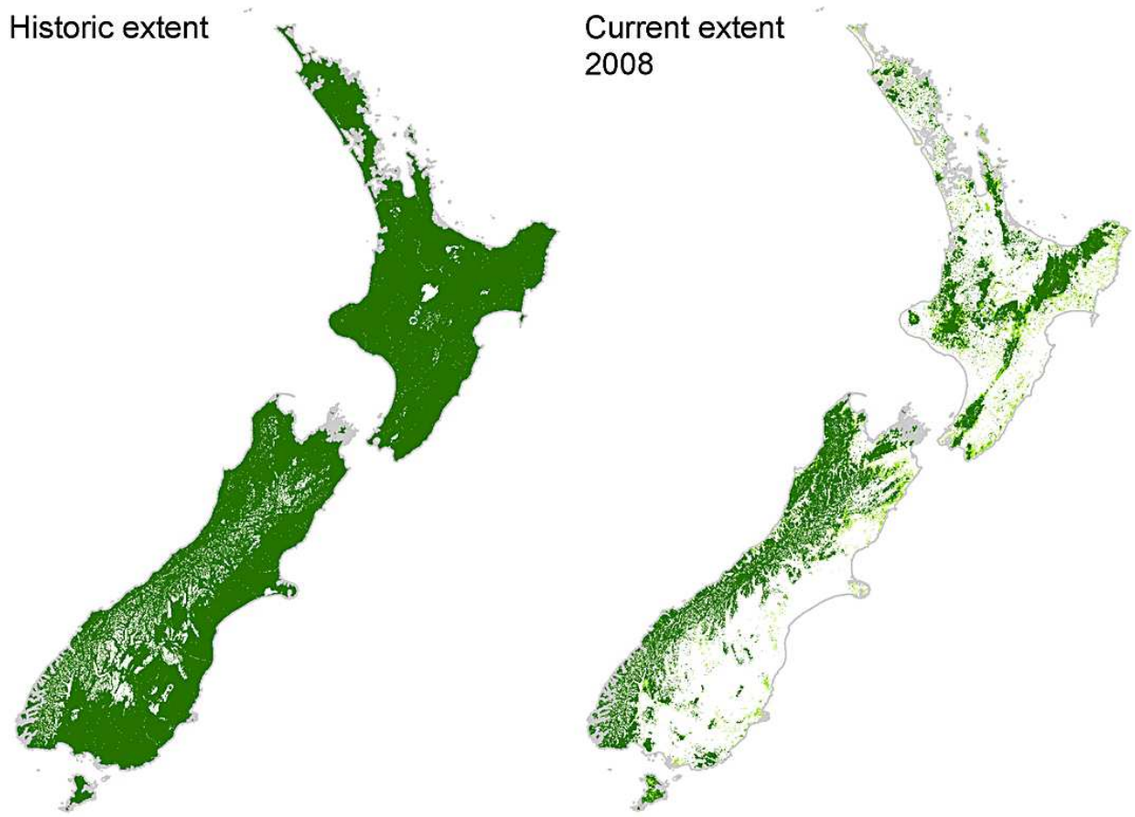

Fig. 1. Historic land cover (1000 AD) compared with recent land cover (2008). Dark green is indigenous forest, light green is indigenous shrubland.

Figure 2 shows the loss of indigenous forest in each of the Level II land environments over the last 18 years. Loss is still evident in many of the land environments. Indeed, nine land environments have lost more than 5% of their remaining indigenous forest. This could be critical, given that eight of those have less than 5% of the land environments remaining in indigenous forest.

3. Indigenous grasslands

Approximately one half of New Zealand's land area is made up of a variety of exotic and indigenous grassland ecosystems. Approximately one-fifth of these grasslands comprise modified indigenous short and tall-tussock communities, which are mostly located on the South Island. Unlike many other indigenous ecosystems in New Zealand, they have a unique, partially human-induced origin. Once largely distributed in areas of lowland montane forest and shrubland, large regions of grassland were created through burning by Maori, especially for moa hunting and for encouraging bracken fern (*Pteridium aquilinum*), an important food source (Stevens et al., 1988; Ewers et al., 2006). Lowland podocarp forests hosting such species as totara (*Podocarpus totora*) and matai (*Prumnoptiys taxifolia*) were replaced by a variety of fire adapted grassland species, in particular the short tussock species *Festuca novae-zelandia* and *Poa cita*. Some 200 years later these species were progressively replaced by taller large grain *Chionochloa* spps (McGlone, 2001).

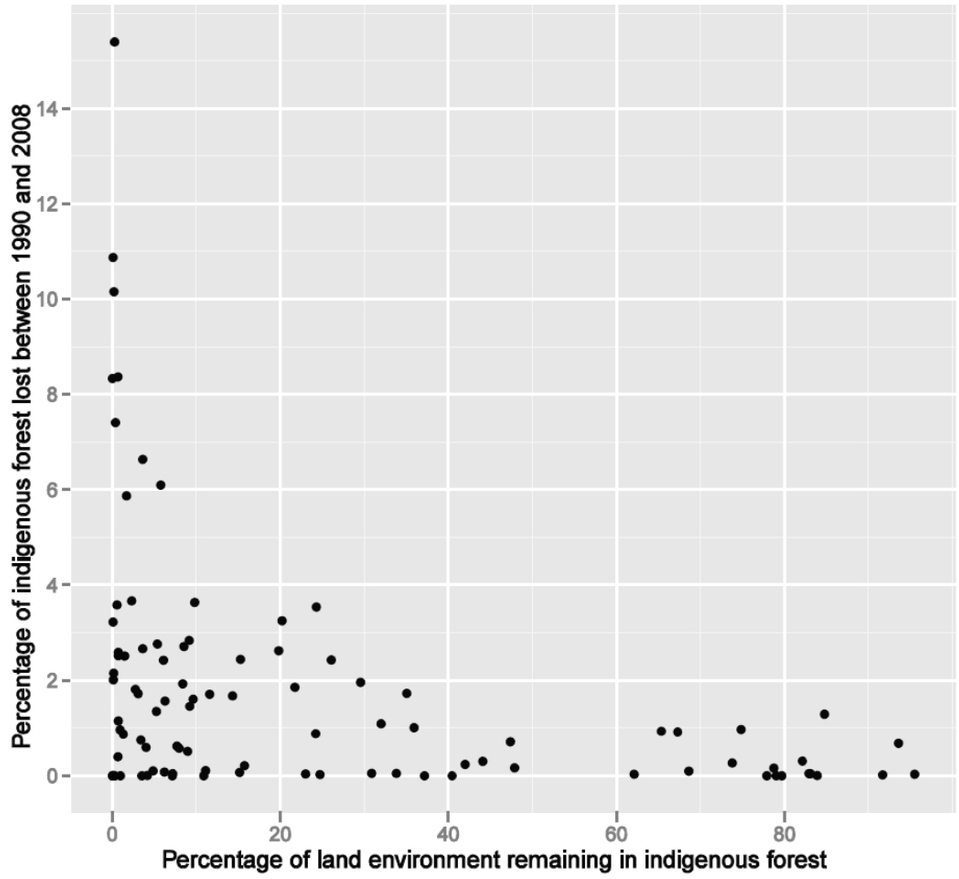

Fig. 2. Percentage of indigenous forest lost between 1990 and 2008.

New Zealand's tussock grasslands have undergone a variety of transformations. In the South Island, much of the high country (tussock grasslands) was acquired from the Maori between 1844 and 1864 (Brower, 2008). During this time, pastoral licenses were granted for 1 year in Canterbury and 14 years in Otago, and the tussock landscape rapidly began to change. Lease holders used fire both to ready land for grazing and to facilitate travel. The result was a huge reduction in area of lowland and montane red tussock grasslands, the elimination of snow tussock from lowland eastern parts, and the reduction of snow-tussock found near settled areas. By the 20th century there was substantial loss of native species through conversion to vigorous exotic grasses maintained by the widespread use of fertilizers and herbicides.

Today, New Zealand's indigenous grasslands are dominated by grass species (Poaceae family) characterised by tussock growth (elsewhere known as "bunch grasses") (Ashdown & Lucas, 1987; Levy, 1951; Mark, 1965; Mark, 1993). The plant communities, however, vary from highly modified to areas with no exotic species (predominantly at elevations above 700 meters (Walker et al., 2006; Cieraad, 2008). Though tussock species *Chionochloa, Poa,* and *Festuca* are the dominant species in the landscape, numerous woody species are also present. At higher and more exposed sites with shallow soils and less available moisture, shrubs including the species of *Brachyglottis*, *Coprosma*, *Dracophyllum*, *Carydium*, *Hebe*, *Podocarps* and other *Olearia* spp dominate; at lower altitudes native shrub species such as manuka

(*Leptospermum scoparium*) and kanuka (*L. ericoides*) are more common and through time have established themselves among the grasses (Newsome, 1987).

Though most New Zealand's indigenous grasslands have been modified to varying degrees by the indirect and direct effects of human activity, they continue to support a rich flora and fauna and are characterized by high species diversity (Dickinson et al., 1998; McGlone et al., 2001; Mark et al., 2009; Walker et al., 2008). However, recent changes in land-use activities have led to further fragmentation. An increasing area of indigenous grasslands (in the South Island), formerly used for extensive grazing, is being converted to intensive agriculture and areas once covered by indigenous grassland species are being progressively replaced with exotic pasture, forestry plantations, and perennial crops.

Mark and McLennan (2005) assessed the loss of New Zealand's indigenous grasslands since European settlement, comparing the Pre-European extent of five major tussock grassland types with their current extent (using LCDB1). They estimated that in 1840, 31% of New Zealand was covered by tussock grasslands dominated by endemic tussock grass species. In 2002, however, just 44% of this area of indigenous grasslands remained, of which most was in the interior areas of the South Island. Of this, approximately 28% was protected with a bias towards the high-alpine areas. Remaining subalpine grassland communities (i.e. short tussock grasslands) still persisted, but were severely degraded and/or modified and under protected. Figure 3 illustrates the change in extent from pre-human to pre-European to current times.

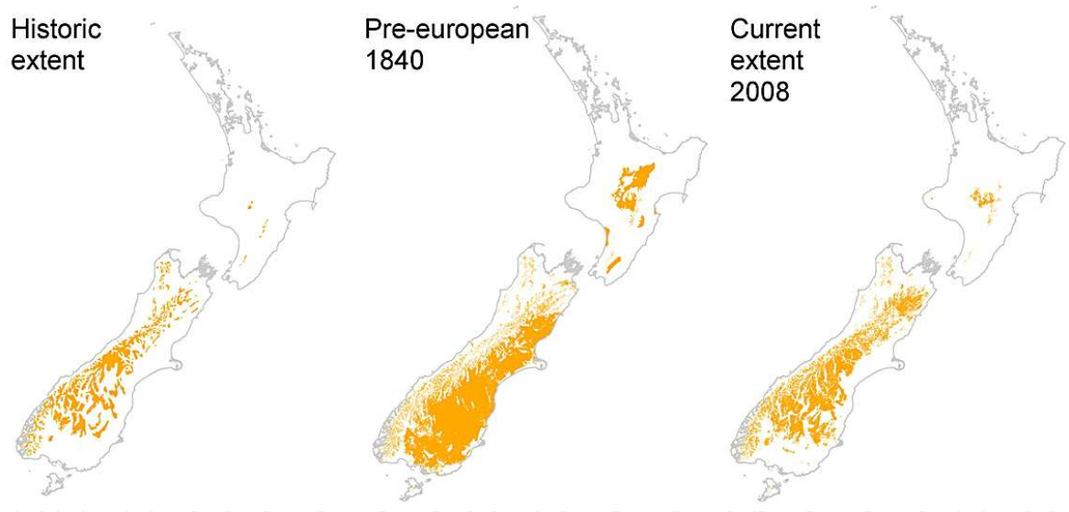

Fig. 3. Changes in the extent of New Zealand's indigenous grasslands since the arrival of humans.

Recent trends in land-use change suggest a movement towards increased production per hectare of land. Weeks et al. (in prep) estimated the current (2008) extent of indigenous grasslands and compared it with grassland in 1990. In 1990, 44% of New Zealand's indigenous grassland remained, by 2008 this was reduced to 43%. During this time there was an accelerated loss from 3,470 ha per year between 1990 and 2001 to 4,730 ha per year between 2001 and 2008. The majority of this change took place at lower altitudes (in short tussock grasslands) and on private or recently free-hold land. Most of the land-use change has been incremental and occurred at the paddock scale (less than 5 hectares).

Continued impacts and reduced indigenous biodiversity are expected over the next century. In grazed areas, plant community composition should continue to alter gradually depending on stocking rates and variability in climate and disturbance regimes. As for areas that are completely converted to new land cover types, changes should be much more immediate. These conversions are likely to have significant impacts on the ecosystem structure and provision of ecosystem services.

4. Freshwater wetlands

Wetlands are defined as permanently or intermittently wet areas, shallow water and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions. They support a wide range of plants and animals. In New Zealand, wetland plants include 47 species of rush and 72 species of native sedge (Johnson & Brooke, 1998). Many of these plants have very specific environmental needs, with a number of plants species adapted to wet and oxygen deprived conditions. Wetlands support a high proportion of native birds, with 30% of native birds compared with less than 7% worldwide (Te Ara – the Encyclopedia of New Zealand, 2009). For instance, the australasian bittern (*Botaurus poiciloptilus*), brown teal (A*nas chlorotis*), fernbird (*Bowdleria punctata*), marsh crake (*Porzana pusilla)*, and white heron (*Egretta alba*) rely on New Zealand's remnant wetlands. Migratory species also depend on chains of suitable wetlands. Wetlands are also an essential habitat for native fish, with eight of 27 native fish species found in wetlands (McDowall, 1975). Among those are shortfin eel (*anguilla australis*) and inanga (*galaxias maculatus*), the major species in the whitebait catch, and species from the Galaxiid family like the giant kokopu (*galaxias argenteus*), which is usually found in swamps (Sorrell & Gerbeaux, 2004). Apart from provision of habitat for biodiversity, wetlands offer other valuable ecosystem services such as flood protection, nutrient retention for water quality, recreational services (Mitsch & Gosselink, 2000), and important cultural services for Māori, including food harvesting and weaving materials. The importance of wetland ecosystems is recognised internationally, and New Zealand is a signatory to the Ramsar Convention on Wetlands of International Significance. Six sites are currently designated as Wetlands of International Importance, with a total area of 55 thousand hectares.

In less than two centuries, New Zealand wetlands have been severely reduced in extent, particularly with the conversion to pastoral agriculture from the mid 19th century. The loss is attributed to human activities through fires, deforestation, draining wetlands, and ploughing (Sorrell et al., 2004; McGlone, 2009). Further degradation of the habitat has occurred since the introduction of livestock with consequent increases in nutrient flows, changing the fragile equilibrium in the wetlands and altering species composition (Sorrell & Gerbeaux, 2004). The loss of local fauna and flora has also been dramatic. Fifteen wetland birds species have become extinct (with 8 out of 15 being waterfowl species) (Williams, 2004), and ten species are on the list of threatened bird species (Miskelly et al., 2008). Among the plants, 52 wetland taxa species have been classified as threatened (de Lange et al., 2004). The decline in many native freshwater fish is also attributed to the loss and degradation of wetlands (Sorrell & Gerbeaux, 2004).

Ausseil et al. (2011) estimated that the pre-human extent of wetlands was about 2.4 million ha, that is, about 10% of the New Zealand mainland. The latest extent (mapped in 2003) was estimated at 250,000 ha or 10% of the original coverage.

Figure 4 compares the current extent of freshwater wetlands with its historic extent. The greatest losses occurred in the North Island where only 5% of historic wetlands remain compared with 16% in the South Island. The South and Stewart Islands contain 75% of all remaining wetland area, with the highest proportions persisting on the West Coast of the South Island and on Stewart Island. The remaining wetland sites are highly fragmented. Most sites (74%) are less than 10 ha in size, accounting for only 6% of national wetland area. Only 77 wetland sites are over 500 ha, accounting for over half of the national wetland area.

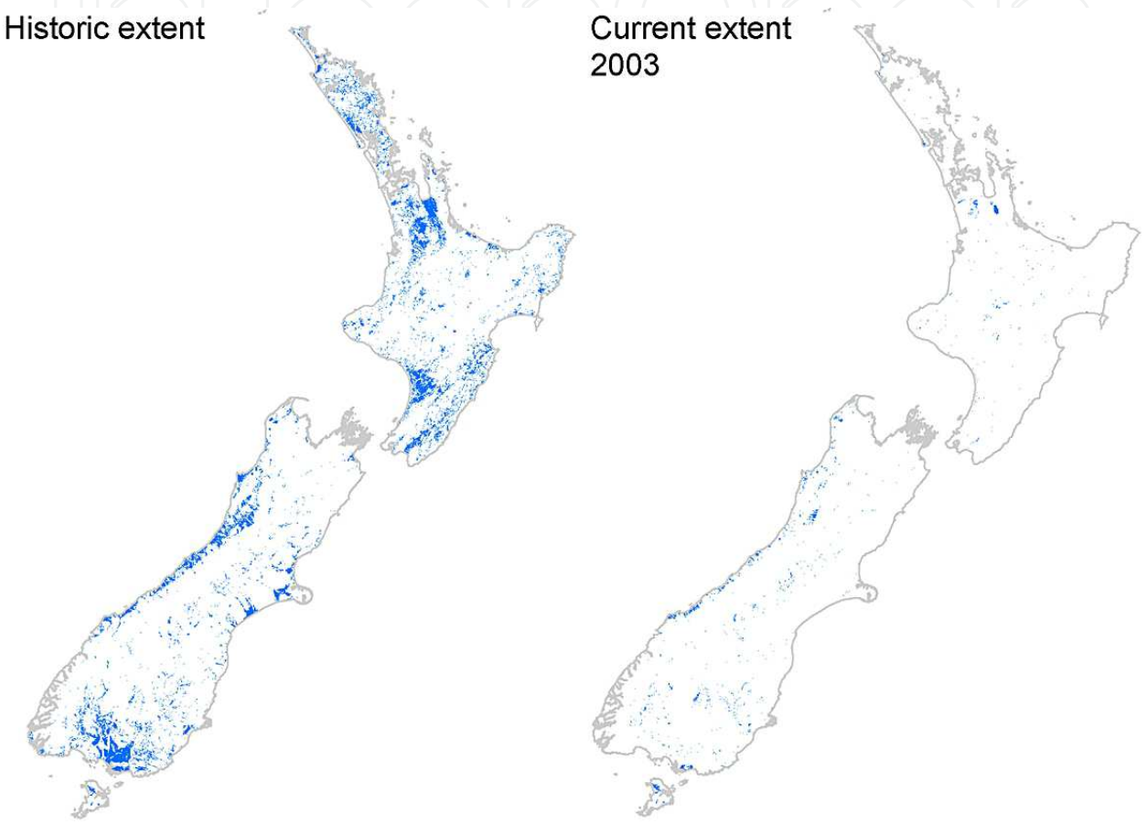

Fig. 4. Map comparison of current and historic extent of freshwater wetlands (blue areas) in New Zealand.

Classification of wetlands can be a challenge as they are dynamic environments, constantly responding to changes in water flow, nutrients, and substrate. Johnson & Gerbeaux (2004) clarified the definitions of wetland classes of New Zealand such as bog, pakihi, gumland, seepage, inland saline, marsh, swamps, and fens. By using GIS rules, it was possible to classify wetlands into their types and follow the trend of extent since historical times (Ausseil et al., 2011). Swamps and pakihi/gumland are the most common wetland types found in New Zealand. However, swamps have undergone the most extensive loss since European settlement, with only 6% of their original extent remaining (Figure 5). This is due to swamps sitting mainly in the lowland areas where conversion to productive land has been occurring.
Unlike indigenous forest and indigenous grasslands, there is no national study describing recent loss over the last ten to twenty years for wetlands in New Zealand. However, some

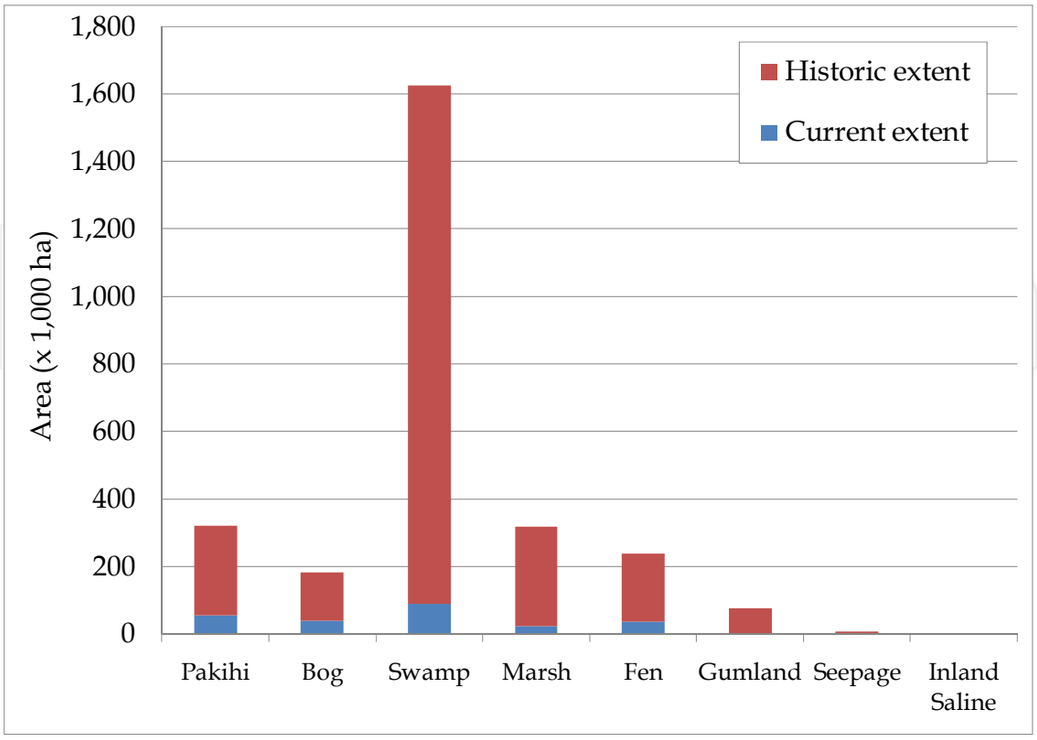

Fig. 5. Current and historic extent of wetland per class.

regional analyses suggest that wetland extent continues to decline, although at a slower rate, as land drainage and agricultural development continue (Grove, 2010; Newsome & Heke, 2010). Wetland mapping is a challenging task as wetlands are sometimes too small in area to be identified using common satellite resolution. Their extent can vary seasonally (e.g., dryness, wetness) and therefore can change markedly at the time of imagery acquisition. While satellite images are useful for providing information at national scale, automatic classification is not possible as vegetation types in wetlands are so variable, making them difficult to characterise through spectral signature. Thus wetlands have been mapped on a manual or semi-automated basis (Ausseil et al., 2007), and this requires a significant amount of effort for all of New Zealand.

5. Measure of natural habitat provision

Measures of habitat provision need to account for different types of habitat and their associated biodiversity. Dymond et al. (2008) showed how proportions of unique habitat remaining may be combined to give a national measure of habitat provision. The habitat measure is based on the contribution it makes to the New Zealand Government goal of maintaining and restoring a full range of remaining natural habitats to a healthy and functioning state. For measuring indigenous forest and grasslands, the historical unique habitats come from Land Environments New Zealand (LENZ) (Leathwick et al., 2003). Wetlands are at the interface of terrestrial and freshwater habitats, and therefore another habitat framework representing both aquatic and terrestrial biota (Leathwick et al., 2007) is used. As such, the measure of habitat provision for wetlands is applied separately from the indigenous forests and grasslands measure.

5.1 Indigenous forest and grasslands

We used LENZ at level II (suitable for national to regional scale) and the most recent land cover (2008) to characterise historic and present habitats. The measure of habitat provision for a land environment is defined as:

$$H_i = P_i \left(\frac{a_i}{A_i} \right)^{0.5} \tag{1}$$

where
a_i is the area of natural habitat remaining in land environment i,
A_i the area of land environment i, and
P_i is the biodiversity value of the ith land environment when fully natural.
The 0.5 power index is used to produce a function monotonically increasing from zero to one with a decreasing derivative in order to represent the higher biodiversity value of rare habitat. In the absence of comprehensive and detailed biodiversity information, Dymond et al. (2008) suggested using species-area relationships (Connor & McCoy, 2001) to estimate P_i as the land environment area to the power of 0.4. The varying condition, or degree of naturalness, of individual sites also needs to be taken into account in the habitat measure:

$$H_i = P_i \left(\frac{\sum_{j=1}^{n} c_{ij} b_{ij}}{A_i} \right)^{0.5} \tag{2}$$

where
c_{ij} is the condition,
b_{ij} is the area of of the jth habitat site in the ith land environment, and
n is the number of habitats in the ith land environment.
The condition of indigenous forest, subalpine shrublands, alpine habitats, and tussock grasslands above the treeline, are assumed to have a condition of 1.0. Tussock grasslands below the treeline have a condition of 0.8 and indigenous shrublands have a condition of 0.5. All other landcovers have a condition of 0.0.
Figure 6 shows the input layers (current land cover and land environments at level II) and the resulting habitat provision map. This map shows the contribution per hectare to the habitat measure (i.e. each pixel represents $\frac{c_{ij} b_{ij}}{\sum_{j=1}^{n} c_{ij} b_{ij}} H_i$).

5.2 Freshwater wetlands

Wetlands are at the interface between water and terrestrial dry environments. They have been categorised with freshwater environments in the past, and as such require a different definition of biogeographic units than the terrestrial environments. We replaced land environments data with biogeographic units defined by climatic and river basin characteristics (Leathwick et al., 2007). This framework was used to define priority conservation for rivers (Chadderton et al. 2004) and wetlands (Ausseil et al., 2011). A condition index for wetlands, similar to c_i in equation (1), was calculated for all current wetland sites in New Zealand (Ausseil et al., 2011). This condition index reflects the major

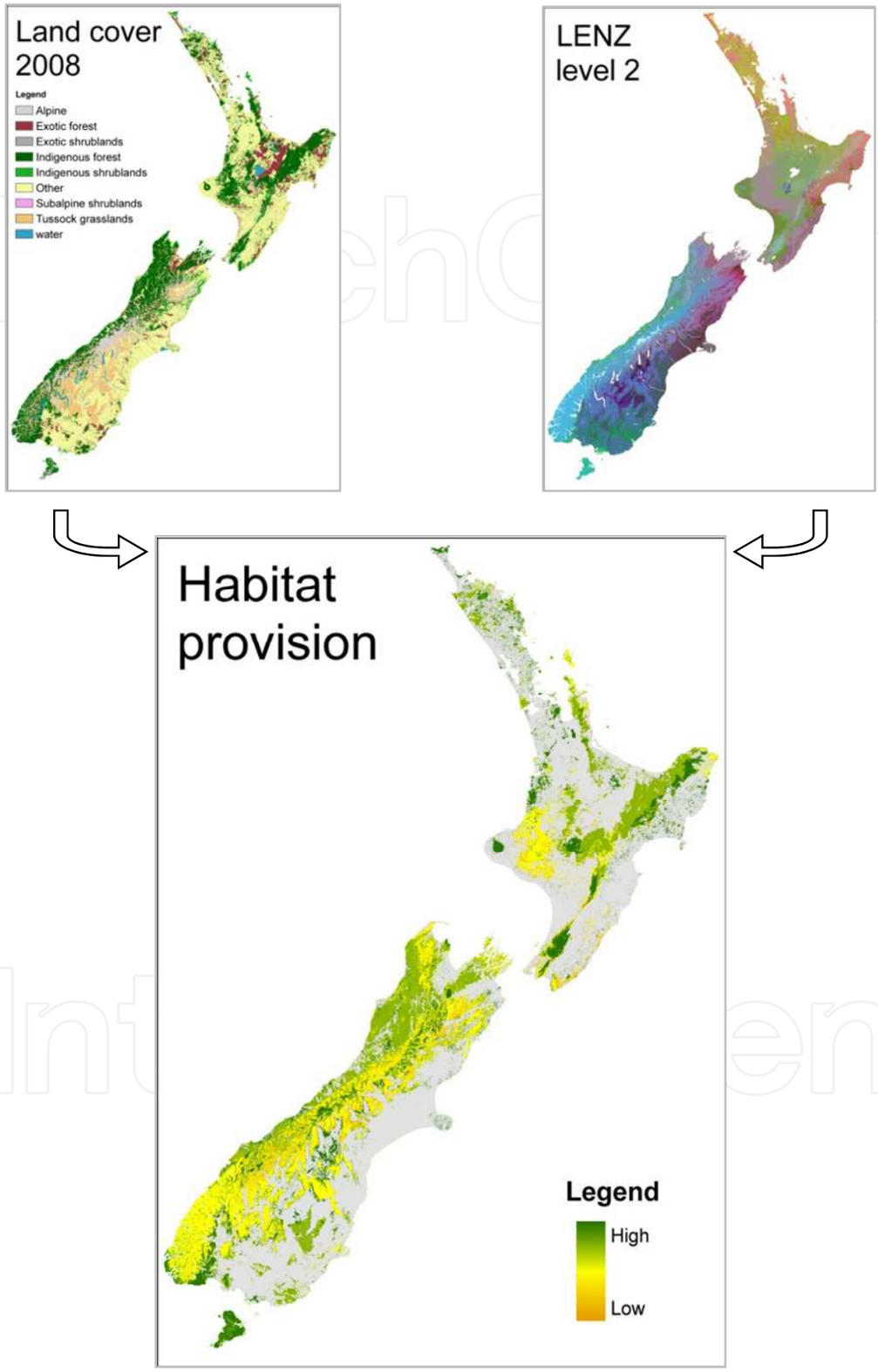

Fig. 6. Habitat provision per hectare from forests and grasslands.

anthropogenic pressures on wetlands, including nutrient leaching, introduced species, imperviousness, loss of naturalness, woody weeds, and drainage pressure.
The measure of habitat provision for wetlands in a biogeographic unit now needs to account for different wetland classes, so is defined as

$$W_i = \sum_{k=1}^{m} P_{ik} \left(\frac{\sum_{j=1}^{n} c_{ijk} b_{ijk}}{A_{ik}} \right)^{0.5} \quad (3)$$

where
c_{ijk} is the condition index of wetland site j in class k in biogeographic unit i,
b_{ijk} is the area of wetland site j in class k in biogeographic unit i,
A_{ik} is the historic area of class k in biogeographic unit i,
m is the number of wetland classes, and
n is the number of class k wetland sites in biogeographic unit i.
Wetland habitats are defined at the class level (m=8 classes) using the wetland classification of Johnson & Gerbeaux (2004). As with P_i in equation (2), P_{ik} is defined as the historical area per wetland class per biogeographic unit to the power of 0.4. The sum $\sum_{j=1}^{n} c_{ijk} b_{ijk}$ reflects the total area of class k wetlands in biogeographic unit i, weighted by the condition index c_{ijk} for each wetland site. If all the wetlands were in pristine condition, the sum would equal the total areal extent in that class.
Figure 7 shows wetland habitat provision for each biogeographic unit. The colours represent the value W_i from equation (3).

6. Discussion

Though there are still large areas of natural habitat remaining in New Zealand, there continues to be ongoing loss. Prior to the settlement of humans, there were 23 million hectares of indigenous forest. Today, only 6.5 million hectares of indigenous forest are remaining. While the total area remaining is large, little of that is in lowland forest ecosystems, and over the last 20 years more lowland ecosystems have been lost. Despite continuing losses in lowland ecosystems, the net area of indigenous forest may well be increasing due to regeneration of indigenous shrublands in marginal hill country. Indigenous grasslands have a similar pattern of change. Over the last 170 years, 4.7 million hectares of indigenous grasslands have been lost. Though the total area of remaining grasslands is large, little of that is in lowland ecosystems, and over the last 20 years more lowland ecosystems have been lost. Wetlands are the most severely impacted ecosystems. Of the 2.4 million hectares of wetlands existing pre-Maori, only 250 thousand hectares are remaining – that is, only 10% of what was there originally. Again, lowland wetlands are mostly affected, with a higher proportion of swamps lost. Recent trend analyses shown in this chapter reveal that loss is still continuing, and is a precursor to negative impacts on provision of ecosystem services and subsequent human well-being.
The habitat provision map for indigenous forest and grasslands show large spatial variability. High values are usually associated with rarer habitats in good condition, but also with habitats in very small land environments. For wetlands, the habitat provision map is

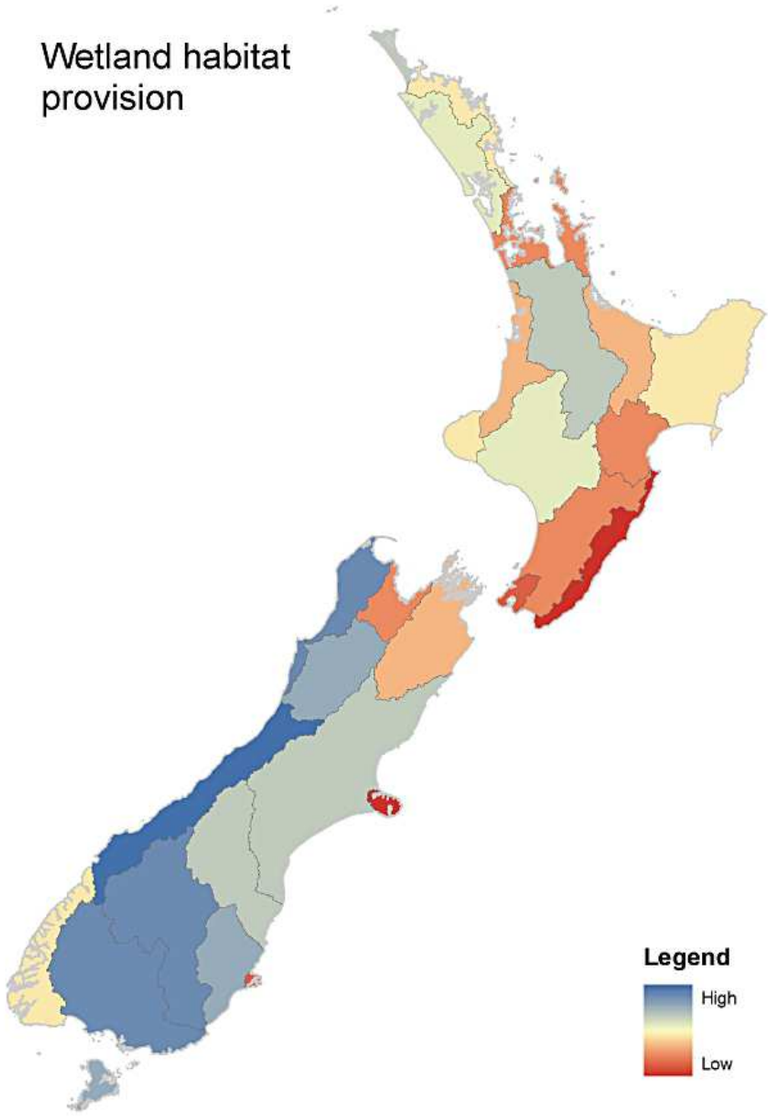

Fig. 7. Habitat provision for freshwater wetlands in each biogeographic unit.

shown at the biogeographic unit level, mainly because wetland boundaries are difficult to depict at the scale shown here. The contribution to the national habitat measure comes mostly from biogeographic units with minimal conversion to productive land. Low values represent units where wetland areas have depleted or where wetlands have been degraded. This information can be used by decision-makers to prioritise the allocation of conservation funds. For example, the maps can be intersected with legally protected areas, like those from Walker et al. (2008), which target areas under private ownership with high natural values. Several legislative tools can be used to protect remnant habitats, including the establishment of conservation covenants like the Queen Elizabeth the Second National Trust (QEII), Nga Whenua Rahui, and the National Heritage Fund.

QEII's goal is to help New Zealand farmers protect open space on private land for the benefit and enjoyment of the present and future generations of New Zealanders. The

covenant is registered against the title of the land in perpetuity and there are obligations to manage the land in accordance with the covenant document. Over 70,000 ha are now protected by QEII covenants (Ministry for the Environment, 2007). Nga Whenua Rahui is a contestable fund to negotiate the voluntary protection of native forest on Maori-owned land. Legal protection is offered through covenants, setting aside areas as Maori reservations or through management agreements. About 150,000 ha of native ecosystems are now protected under this fund. The Nature Heritage Fund (NHF) is a third contestable fund for voluntary protection of nature on private land. Its aim is to add to public conservation land those ecosystems important for indigenous biodiversity that are not represented within the existing protected area network. Since 1990, the fund has protected over 100,000 hectares of indigenous ecosystems through direct land purchases, covenants on private land or fencing.

The information on habitat provision could feed into the Department of Conservation (DOC) management system. DOC is responsible for managing biodiversity on the conservation estate, and is developing the Natural Heritage Management System (NHMS). DOC's statement of intent is to legally protect the best possible examples of each native ecosystem type, by fencing, reinstating water levels, replanting, controlling pest animals and weeds, and reintroducing native species to restore and maintain natural ecosystems. The framework proposed in this chapter is envisaged to help achieve this goal through accurate information on habitat extent and ecosystem loss, and provides a measure for comparing habitats within and across land environments where species level assessments may not be possible.

Continuing loss of natural habitat may be due to a lack of market prices for associated ecosystem services (TEEB, 2010). Monetisation of habitat provision could partly redress this. In New Zealand, Patterson & Cole (1999) estimated natural forests and wetlands to both have total economic values of approximately 6 billion dollars in 1994. From this, it is possible to convert the units of habitat measure to economic value in dollars per year – Dymond et al. (2007) estimated this as 60 units to one (NZ) dollar per year (assuming areas are in hectares). This monetisation would permit the comparison of changes in habitat alongside changes in other ecosystem services in the same units. This reduces the complexity of results when analysing impacts of different land-use decisions. Using dollars also provides context for stakeholders unfamiliar with biodiversity impacts associated with habitat loss. The negative side of monetisation is that some stakeholders may be encouraged to make trade-offs on the basis of the monetisation alone, not realising the assumptions and limitations involved, or being aware of environmental bottom-lines. Indeed, the risk of valuation is to get the figure very wrong. There are numerous valuation methods, often based on subjective, hypothetical, and questionable assumptions, which can all give vastly different values (Spangenberg & Settele, 2010). Altogether, monetisation, although easy to comprehend, can be misleading and should be used with caution. It should be used in close consultation with decision-makers, so that they are fully aware of the pitfalls and assumptions behind the valuation, to avoid misallocation of resources.

The measure of habitat provision is a landscape approach which makes several assumptions. First, it uses particular GIS databases, each of which has a certain level of sensitivity and accuracy. Land environments has been tailored to forest ecosystems, and does not encompass the full breadth of other ecosystem types. Biogeographic units were used for wetlands, assuming that freshwater species would be concentrated within defined hydrological boundaries. Second, it assumes that landscape morphology and pattern can be used as a surrogate for species. Though this overcomes the issues surrounding availability

of data, application is limited at the various levels and components of biodiversity. In other words, provisions can not be assessed at multiple scales (i.e. habit, community and/or species). Third, the condition of indigenous forest and grassland assumes that all sites are characterised by one condition, though condition could vary within large sites. For wetlands, the condition does vary per site, but it is based on landscape indicators. It is appropriate for a rapid assessment of sites, and can help for prioritising field visits (Ausseil et al., 2007), but does not necessarily reflect the true condition in the field.

The loss of indigenous forest is well characterised by the habitat provision analysis, but the gain of indigenous forest from regenerating indigenous shrublands is not. This is because both the LCDB and the LUM datasets focus on mapping change primarily between woody and herbaceous vegetation, and the subtle changes in the spectral signature of regenerating indigenous forest and mature forest are not accurately characterised or determined, making it difficult to decide whether indigenous vegetation is mature enough to be classified as forest. This is important because there are large areas of indigenous shrublands in New Zealand, approximately 1.6 million hectares. Much of these shrublands are currently regenerating to forest and could make a significant contribution to the areal extent of indigenous forest if this trend continues. If we assume a conservative time of 100 years to reach forest maturity and a uniform distribution of shrubland age, then we would expect about 1% of the shrubland area to change to indigenous forest each year – this amounts to 16 thousand hectares per year. Over 18 years this would equate to approximately 300 thousand hectares, which is six times the estimated current loss of indigenous forest. This fills an important information gap in our understanding of the changing areal extent of indigenous forest and indicates the importance of using objective mapping techniques to monitor change.

Conservation management in New Zealand is becoming increasingly strategic, systematic, and reliant on accurate information on which to plan and prioritise goals and actions. A range of sophisticated tools and approaches have been developed to support these efforts in the past ten years. These include measuring Conservation Achievement (Stephens et al., 2002), the Land Environments of New Zealand (Leathwick et al., 2003), and measuring provision of natural habitat (Dymond et al., 2008). In addition, these efforts have spawned considerable activity for acquiring underlying data, such as biodiversity value (Cieraad, 2008), land cover (the LCDB3 project), and threats to biodiversity (Overton et al., 2003; Walker et al., 2006). However, a national coordinated approach to conservation management taking into account species distributions is required. Overton et al. (2010) are developing a tool called Vital Sites to assess ecological integrity. This incorporates current and natural distributions of native species based on a modeling approach, pressures (e.g., pests or habitat loss) on biodiversity, and the effects of management on relieving pressures. It operates at two levels (species and landscape) and assessments of significance and priorities can be made at each separate level or by combining the two levels. This research tool will provide another step to helping achieve goals towards identifying the most vulnerable ecosystems in New Zealand requiring urgent protection and management.

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Every ecosystem is a complex organization of carefully mixed life forms; a dynamic and particularly sensible system. Consequently, their progressive decline may accelerate climate change and vice versa, influencing flora and fauna composition and distribution, resulting in the loss of biodiversity. Climate changes effects are the principal topics of this volume. Written by internationally renowned contributors, Biodiversity loss in a changing planet offers attractive study cases focused on biodiversity evaluations and provisions in several different ecosystems, analysing the current life condition of many life forms, and covering very different biogeographic zones of the planet.

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