INTENSIVE DEVELOPMENT OF NEW ZEALAND'S INDIGENOUS GRASSLANDS: RATES OF CHANGE, ASSESSMENTS OF VULNERABILITY AND PRIORITIES FOR PROTECTION

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ABSTRACT

Research was undertaken in the indigenous tussock grasslands of South Island of New Zealand in order to quantify past rates of conversion to agricultural land use and to develop vulnerability models to predict future conversion spatially and temporally. The study area was delineated using the median spectral reflectance of indigenous grasslands and included the largest extent of unprotected contiguous grasslands concentrated in the central South Island. Conversion from indigenous grasslands to a non-indigenous cover was quantified by comparative mapping over three intervals (1840-1990, 1990-2001, and 2001-2008). The basic premise in using satellite imagery to detect changes in land-use/cover is that these are revealed by changes in spectral signature. However, New Zealand's indigenous and non-indigenous grasslands have overlapping spectral trajectories and high inter-annual variability, therefore contextual information was needed in order accurately map conversion from indigenous grassland cover to exotic pasture.

Within the study area around the time of European settlement (1840) there were approximately 3.3 million hectares of indigenous grasslands. Between 1840 and 1990 around 1 million hectares of indigenous grasslands were converted to a non-indigenous cover. The extent of conversion during the preceding time period (1990-2008) was approximately 71,261 ha, of which 72% was converted to pasture and cropland and the remaining 28% to mining, urban settlements and exotic forestry. Although the overall rate of grassland conversion decreased relative to the period of European settlement and 1990, the proportion of remaining indigenous grasslands converted each year increased. Almost two-thirds of post-1990 conversion has occurred in environments with less than 30% indigenous cover remaining, and much is in land classified as non-arable with moderate to extreme limitations to crop, pasture and forestry growth.

To assess the relative vulnerability of remaining areas of indigenous grassland to intensive land use (mainly intensive pasture production but also exotic conifer plantations, urban use and mining), spatial predictions using Generalized Additive Models (GAMs) were used to establish relationships between two different types of dependent (response) variables (presence or absence of conversion) and

potential environmental and proxy socio-economic explanatory variables. The chosen predictors for the final model were used to map conversion probabilities in geographic space. The selected GAMs showed the mean probability of conversion in remaining indigenous grasslands was 0.15 and the mean area of conversion was 116 ha. Habitat that was most vulnerable to conversion was at moderate elevations and on medium slopes, and had previously been classified as being of low suitability for production.

To interpret the regression models, plots of the partial response curves resulting from the model, and overall contributions of variables to the model, were used. The most important explanatory variables for predicting the probability of conversion in order of ‘alone contribution (the potential for each variable alone to explain conversion) was slope, rainfall, land tenure, distance to roads, proximity to existing agricultural, regional council, and mean annual temperature. Interpretation of the GAMs showed that conversion was negatively related to: slope, rainfall and distance roads; positively related to mean annual temperature; higher in the Otago and Canterbury regions and on privately owned or recently privatized lands, and peaked at intermediate proximity to roads.

The prediction of the probability of conversion model was cross-validated both spatially and temporally. Temporal cross-validation compared predicted probabilities of conversion against reference maps of observed ‘current’ conversion. Spatial cross-validation evaluated model discrimination between ‘converted’ and ‘not converted’. Temporal and spatial performance was measured using the Receiver Operating Characteristic (ROC), a graphical plot of the true positive rate (sensitivity) as a function of the false positive (1-specificity) for different probability thresholds. For temporal cross-validation there was high correlation between ‘predicted’ and ‘observed’ (ROC = 0.913), and for spatial validation the relationship between the fitted and observed was also high (ROC= 0.921), indicating there was good discrimination between ‘converted’ and ‘not converted’.

Integrating validated estimates of the probability of conversion (vulnerability) into conservation planning tools is an important component of conservation planning. Comparison of conservation prioritisation outputs with validated estimates of

vulnerability of New Zealand's indigenous grasslands showed variable effectiveness of vulnerability surrogates; one surrogate performed most poorly where vulnerability of grasslands to conversion was greatest and realized probability of protection was lowest. Furthermore, estimates of vulnerability using surrogates underestimated vulnerability on flat land that was closer to roads and overestimated areas on steeper land that was topographically invulnerable to conversion.

There is an increased disparity between patterns of protection and patterns of conversion indicating that existing conservation planning tools are not effectively targeting the most vulnerable areas of remaining indigenous grasslands. An up-to-date validated vulnerability assessment offered a practical and a responsive technical bridge for the gap between science and implementation. This approach can be applied more widely to provide national models of vulnerability from representative samples of conversion.

Key words conservation planning; indigenous grasslands; receiver operating characteristics; remote sensing; agricultural conversion; New Zealand

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1 Introduction

1.1 Research Topic

This research focuses on developing methods for monitoring habitat loss, and predicting conservation priorities in New Zealand's indigenous grasslands. The focus on the indigenous grasslands resulted from the recognition that rapid land-use changes were taking place and information about recent changes and trends in the extent and condition of New Zealand's indigenous grasslands was limited. Existing information on land use change was not of high enough resolution to perform this analysis, so the first step was to develop methods and detect change from a combination of satellite imagery, aerial photographs and extensive field work. This provided improved data on the remaining and current extent and rates of conversion of New Zealand's indigenous grasslands. To identify major environmental, proxy socio-economic correlates of land use conversion and habitat loss, I used spatial regression models to model patterns of loss and predict the vulnerability of remaining indigenous grassland habitat to conversion. This resulted in temporally and spatially validated models and predictions of vulnerability to land use conversion. This study then explored the implications of these patterns of loss to conservation of New Zealand's grasslands and the significance of validated vulnerability assessments to existing conservation prioritisation tools.

1.2 Background

Over the past 50 years, ecosystems have changed more rapidly than any other period of human history (Millennium Ecosystem Assessment 2005). Considerable proportions of the world's thirteen terrestrial biomes are being converted to less ecologically diverse ecosystems (Hoekstra et al. 2005). Such a high degree of conversion is leading to extensive changes in biodiversity composition and ecological processes resulting in the diminishing of ecosystem services that help sustain biological diversity and human populations.

Internationally, some of the largest changes in biodiversity have occurred and are expected to occur in grasslands, yet they continue to remain one of the least protected ecosystems (White et al. 2000). Most of the world's indigenous grasslands have been converted for agricultural activities (Goobridge 1992). Areas with better soils and more frequent rainfall have been mostly cleared for crops, while poorer quality grasslands have been left for rearing stock (Suttie et al. 2005). Globally there is limited information on the rate, type, and amount of change that is occurring in grassland ecosystems (White et al. 2000), and New Zealand is no exception. Without fundamental information on trends occurring in grasslands, researchers are unable to assess potential effects on habitat and their associated biodiversity and ecosystem services, and policy makers lack the evidence needed to assess effects of land management and legislation, and to inform sound policy formation (Gluckman 2011).

Given the biological and cultural significance of indigenous grasslands and the ongoing change in land use in these areas, it is important to monitor and measure land use change that is taking place in these ecosystems. However, recent attempts to do so have faced several challenges. First, there is no universally accepted definition of grasslands (Bailey 1989, Scholes and Hall 1996, House and Hall 2000). Second, there is little agreement on the methods to determine boundaries between native grasslands and agricultural land/permanent pasture and between grasslands and forests (White et al. 2000). Finally, there is limited data available for evaluating historical change. Therefore, there is still a need to develop standardised methodologies that are useful for detecting land-use change in grasslands.

New Zealand's indigenous tussock grasslands provide a range of important ecosystem services (i.e. water regulation and soil formation), including significant cultural values to New Zealanders. Unlike many other indigenous ecosystems in New Zealand, the tussock grasslands have a unique and partially human induced origin. Once largely in forest and shrubland, regions of tussock grassland were created by the aftermath of Maori burning and clearing for hunting moa and encouraging the growth of bracken fern (*Pteridium aquilinen* L.) (Stevens et al. 1988 as cited by Ewers et al. 2006).

Post-European settlement the grassland systems underwent a variety of transformations. In the South Island, between 1844 and 1864, much of the indigenous grassland was acquired from the Maori (Brower 2008). During this time variable pastoral licenses were granted (ranging from 1 year in Canterbury to 14 years in Otago), and the tussock landscape was rapidly transformed. Lease holders used fire 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 us of fertilizers and herbicides and the introduction of rabbits (the exotic species *Oryctolagus cuniculus .L*) contributed to additional degradation particularly in the drier parts of the Mackenzie Basin and Central Otago.

Today, New Zealand's indigenous grassland system remains not only a highly modified landscape but also a continuously changing landscape. Invasion exotic species such as gorse and hieracium continue to threaten the re-establishment of native vegetation. Furthermore, recent changes in land-use activities have led to further fragmentation. An increasing number of indigenous grasslands (in the South Island), formerly used for extensive grazing, are being replaced with exotic pasture, forestry plantations, and perennial crops. However, though most New Zealand's indigenous grasslands have been modified to varying degrees by indirect and direct effects human activity, they have continued to support a rich flora and are characterized by high species diversity (Dickinson et al. 1998, McGlone et al. 2001, Walker et al. 2008, Mark et al. 2009).

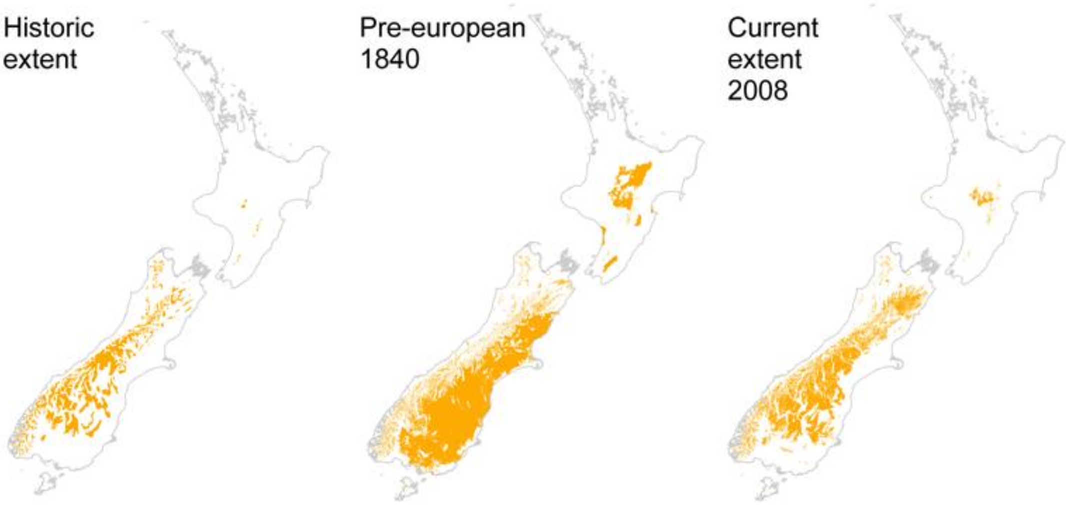

Figure 1.1 Changes in the extent of New Zealand's indigenous grasslands since the arrival of humans

Though expert-opinion-based estimates of the extent of the remaining grassland cover have been made (Mark & McLennan 2005), quantifying the true extent of grassland biodiversity continues to be a challenge (Walker et al. 2006). Several land-use/cover maps have been developed for New Zealand (Newsome et al. 1987, Thompson et al. 2003). These methods relied heavily on field observation, making data collection time-consuming and economically inefficient for regular updates of large areas. Furthermore, vegetation cover mapped by Newsome et al. (1987) was mapped at a coarse 1:250,000 scale and the New Zealand Land Cover Database produced by Thompson et al. (2003) primarily targeted woody ecosystems. These maps have therefore not been reliable for accurately detecting changes in New Zealand’s grassland ecosystems (Walker et al. 2006).

Internationally, the use of remotely sensed imagery is becoming a cost-effective method to identify and map land-use/cover changes in grasslands. Substantial improvements to the standardization, illumination, and viewing geometry, (Liang et al. 2005, Dymond and Shepherd 2004) along with enhanced spatial resolution (as much as 5 metre pixels), has significantly improved the ability to detect changes in vegetation cover. Therefore satellite imagery provides a viable source of data which can be used to efficiently and accurately detect land-use changes in grasslands.

Satellite imagery is useful for monitoring change in land-use at a global, regional or national scale by virtue of its large areal extent and high resolution. Substantial improvements to the standardization, illumination, and viewing geometry, (Dymond and Shepherd 2004, Liang et al. 2005) along with enhanced spatial resolution, has significantly improved the ability to detect changes in vegetation cover. Therefore satellite imagery provides a viable source of data which can be used to efficiently and accurately detect land-use changes in grasslands.

There are a variety of change detection techniques that can be used to assess land-use change. These can be summarized into two broad categories: change measurement (stratification) methods versus classification approaches (Malila, 1980, Coppin et al., 2004). Change measurement method involves the use of algorithms and thresholds to determine the changed areas (Singh 1989b). Commonly used change measurement methods include image differencing, image regression, image ratioing, change vector analysis (CVA), and vegetation index differencing (Malila 1980, Coppin et al. 2001, Lillesand et al. 2004, Ding et al. 2007, Lin et al. 2009). In comparison, classification approaches, which include post-classification comparisons, are based on independently classified images (Lu et al. 2004). These images can be classified using a variety of techniques including: unsupervised classifications, supervised classifications, and manual digitizing. Classification approaches to detecting land-use change has the advantage of being able to provide a matrix of change information and has the ability to reduce the impact from radiometric calibration between the two dates of imagery (Coppin et al. 2004, Lu et al. 2004).

Not all methods have the same ability to detect and monitor change in all ecosystems. Different change detection methods can yield different change maps. This is because the ability to detect change is a function of the class definitions, the spatial extent, and the context of the change (Khorram et al. 1988, Brockhaus and Khorram 1992). The selection of the appropriate method is therefore important. Though, numerous studies have compared change detection methods, few studies have compared methods for grasslands (Brockhaus and Khorram 1992, Cohen and Spies 1992, Mas 1997, Bucha and Stibig 2008, Berberoglu and Akin 2009). Most comparisons of change detection methods have focused on land-use change in woody ecosystems (Mas 1997, Bauer et al. 2004, Dymond et al. 2008b). In fact, little progress has been made in the application of remote sensing technology for monitoring change in grassland ecosystems (Buffing and Herbel 1965, Cayrol et al. 2000, Burba and Verma 2001). Furthermore, monitoring land cover change with remote sensing data can be unreliable when the process of interest operates at a scale below the spatial resolution of the sensor, as in patches of tussock grasslands converted to pasture.

Nevertheless, monitoring of reflectance through time in grassland or semi-arid ecosystem is possible through recent remote sensing developments that standardise satellite images for atmosphere, illumination, and viewing geometry (Dymond and Shepherd 2004). Berberoglu (2009) used NDVI and image differencing of Landsat TM imagery to detect change in semi-arid landscapes. Other studies successfully used spectro-radiometer and satellite data to estimate and assess biophysical characteristics of grassland ecosystems including biomass and leaf area index (Briggs and Nellis 1989, Friedle et al. 1994, Chen and Brutsaert 1998). In addition, remotely sensed data have been used to discriminate among land cover and grassland types (Price et al. 1993), and textural algorithms have been used to discriminate among grassland communities (Lauver and Whistler 1993). Studies have also shown the usefulness of high spectral satellite imagery (Bradley and Mustard 2005) and multi-temporal imagery (Langley 2001) for detecting changes in grassland vegetation. In New Zealand, Vescovo et al. (2009) conducted a preliminary study of mapping biomass and cover in New Zealand Grasslands using 2003/2004 Landsat imagery. They found marked variability between different grassland types, though they noted that indigenous tussock grasslands showed a “very similar” spectral signature to depleted low-productive areas. Other studies have found single-date Landsat TM data provided a reliable method for mapping vegetation cover in semi-arid regions (Langley 2001).

1.3 **Research Questions**

The main research questions addressed in this thesis are:

- How can we monitor and quantify trends in indigenous grasslands?
- What are the recent and current patterns and rate of conversion in New Zealand's indigenous grasslands?
- What are the environmental, social, and economic correlates of conversion?
- Can past patterns of conversion be used to predict future patters of conversion?
- What are the likely impacts of validated vulnerability assessments to current prioritisation tools?

1.4 **Goals**

The goals of this thesis are to:

- Address the information gaps in (1) remote sensing technological developments and national land cover data and (2) knowledge of the status and trends in indigenous grasslands.
- Map grassland types using fine-scale spatial satellite imagery.
- Detect changes using stratified spatial sampling of remote sensing data and ground truth outputs using stratified spatial sampling methods.
- Apply spatial regression and modelling approaches to identify the key predictors of habitat conversion.
- Compare and validate patterns of conversion as predicted by past patterns of change, to actual observed patterns of conversion.
- Assess the likely impacts of validated vulnerability assessments to current prioritisation tools.

The thesis comprises of four main chapters (2-5) that have been accepted by, or submitted to four international journals: *New Zealand Journal of Geography* (published in New Zealand), *New Zealand Journal of Ecology* (New Zealand), *Environmental Conservation* (UK), *Environmental Management* (USA). While each chapter is self-contained with an introduction and background literature review, and formatted according to the relevant journal style, each chapter builds on the results of the previous chapter to develop an overview of the patterns of loss and conservation implications of New Zealand's indigenous grasslands.

Chapter 2 evaluates a selection of remote sensing based land-use change detection methods. It addresses the information gap between remote sensing technology and land use change detection in New Zealand's indigenous grasslands. It also includes an analysis of temporal profile of different grassland covers to explain the performance of the different change detection methods. This research has been submitted to *New Zealand Journal of Geography* as, Weeks E.S., Dymond J.R., Shepherd J.D., and Aussiel A.E. Remote sensing methods to detect land-use changes in New Zealand's indigenous grasslands.

In Chapter 3 the most appropriate land-use change detection method is adopted to evaluate conversion in grasslands in the South Island of New Zealand during two consecutive time periods (1990-2001 and 2001-2008) spanning 18 years, using satellite imagery. It also identifies types and patterns of conversion that result in the loss of habitat for indigenous species in different ecological districts, land environments, land-use capabilities, and administrative districts. This research has been accepted by the *New Zealand Journal of Ecology*: Weeks E.S., Walker S., Dymond J.R., Shepherd J.D., and Clarkson B.D. Patterns of recent and past conversion of New Zealand's indigenous grasslands.

Chapter 4 describes a method for an assessment of the vulnerability of remaining areas of indigenous grassland. Quantitative spatial models to predict the vulnerability of remaining indigenous grassland to conversion were created, based on new mapping of past and current land use in relation to patterns of climate, topography, soils, and proximity to infrastructure (i.e. roads) or existing development. Furthermore model validation techniques were developed to measure the ability of vulnerability of predictions based on past conversion to predict current and future conversion. This research has been accepted by *Environmental Conservation*: Weeks E.S., Overton J.M., and Walker S., Estimating dynamic patterns of vulnerability in a changing landscape: a case study of New Zealand's indigenous grasslands.

The final research paper, Chapter 5, builds on Chapter 4 in which the likely impacts of validated vulnerability assessments to current prioritisation tools are addressed. This chapter considers the importance of using validated vulnerability data in conservation planning and assesses the congruence of realised protection outcomes with apparent conservation priorities from simple and more complex planning tools, and those using surrogate and validated vulnerability data. This research has been accepted subject to changes by *Environmental Management* as: Weeks E.S., Walker S., Overton J.M., and Clarkson B.D. The value of validated vulnerability data in conservation planning.

Chapter 6 is a synthesis of the research presented in the preceding four chapters. It highlights the key findings of this research and summarises general trends in conservation planning in New Zealand. It also makes recommendations for future research needed to improve conservation planning in New Zealand.

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2 Remote sensing methods to detect land-use changes in New Zealand's indigenous grasslands[1]

2.1 Abstract

In order to improve biodiversity management in New Zealand's indigenous grasslands, it is necessary to monitor land-use/cover change trends. We evaluated a selection of change detection methods (image differencing, NDVI differencing, post classification, and visual interpretation) to determine the most accurate method for detecting land-use change in New Zealand's indigenous grasslands. Our results demonstrated the difficulties of detecting change in New Zealand's indigenous grasslands. In the grassland landscape, automatic detection methods were not able to differentiate between variations of soil moisture and vegetation phenology from variations in land-use change. This, in combination with topographic effects, which have hampered the automated mapping of vegetation, is the main reason why visual interpretation of high-resolution imagery is still needed. Operator-assisted interpretations of high-resolution imagery were able to detect change at 98% accuracy. This surpassed all other methods, which were unable to achieve an overall accuracy greater than 56%.

Key words: remote sensing, land-use change, indigenous grasslands, New Zealand

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2.2 **Introduction**

Land use has been recognized as one of the major drivers of global change in ecological systems over the past several decades (Coupland 1994, Vitousek 1994, Sala *et al*. 2000, Weng 2002). Rapid and sizeable changes, brought about primarily through the demand for productive land, are occurring across different ecosystems, including grasslands. Recent estimates indicate that 41% of all temperate non-woody grasslands, savanna and shrublands have been converted to agricultural land (White *et al*. 2000). Many grasslands continue to remain under threat to further conversion for intensive land uses, and it is uncertain how best to monitor them, particularly in New Zealand (Walker *et al*. 2006).

Approximately 60% of New Zealand's land area is made up of a variety of grassland ecosystems comprising either introduced or indigenous grassland species (Wardle 1991). Approximately one-fifth of these grasslands are modified indigenous short and tall-tussock communities, mostly located on the South Island (Mark and McLennan 2005). They were created around 800 years ago through the burning of lowland forest by Maori for Moa hunting and to encourage the growth of bracken fern (*Pteridium aqulinum* L.) (Stevens *et al*. 1988, Ewers *et al*. 2006). After the arrival of Europeans (circa 1840), most of the indigenous grasslands in the South Island were acquired from the Maori by the British Crown and pastoral licenses were granted for up to 33 years (Brower 2008). This led to rapid changes in the landscape. Lease holders used fire to ready land for grazing and to facilitate travel. By the 20^{th} century there was substantial loss of native species through conversion to vigorous European seeding exotic grasses maintained by the widespread use of fertilizers and herbicides.

Informal observations and (albeit limited) quantitative data suggest that in the last decade conversion ("development" and/or "improvement") for dairy intensive grazing of dairy stock is proceeding rapidly in New Zealand's remaining indigenous grasslands. This has led to the introduction of exotic European seeding grass species, such as short rotation rye grass, white clover and red clover, which were better suited to high stocking rates and intensive grazing. Type 1 discretionary consents[2] were issued on 17 pastoral leases in the period April 2002 to March 2003 and on 35 and 42 leases in the subsequent 12-month periods (Walker pers. com.). Land reform (colloquially known as "Tenure Review") is gathering pace, dividing former pastoral leases into separate freehold (privatized) and conservation parcels, concentrating pastoral production within the freehold portion, usually on lower elevation land, and enabling new land uses such as subdivision, dairying and viticulture. This appears to be leading to accelerated loss of indigenous grasslands (i.e. low producing, depleted and tall tussock grasslands) on land transferred to freehold through Tenure Review.

Spatially explicit information describing the extent, condition, protection status and trends in New Zealand's indigenous grasslands is a critical requirement for assessing the impacts of current land management practices and conservation initiatives. It is necessary to determine where, and how urgently, changes in regulations, management and conservation practices are required for sustainability of indigenous grassland ecosystems. Though several land cover maps have been developed for New Zealand, including New Zealand Land Resource Inventory (NWASCO 1975-79) The Vegetative Cover of New Zealand (Newsome *et al*. 1987), and the New Zealand Land Cover Database 1 (1996/1997) and 2 (2001/2002) (Thompson *et al*. 2003). These maps were developed using intensive field observation, making data collection time-consuming and economically inefficient for regular updates of large areas. Furthermore, vegetation cover mapped by Newsome *et al*. (1987) was mapped at a coarse 1:1,000,000 scale, and the New Zealand Land Cover Database produced by Thompson et al. (2003) primarily targeted woody ecosystems. These maps have therefore not proved useful for detecting changes in New Zealand's grassland ecosystems (Walker *et al*. 2006).

Internationally, the use of remotely sensed imagery has become a cost-effective method to identify and map land-use/cover changes in grasslands. Berberoglu (2009) used NDVI and image differencing of Landsat TM imagery to detect change in semi-arid landscapes. Other studies successfully used spectro-

[2] Type 1 consents are issued for Burning, Clear Scrub, Cultivation, Earth Disturbance, Fertiliser, Maintain/upgrade tracks, Plant Trees, Soil Disturbance, Sow Seed, Topdress, Tracking, or Trenching

radiometer and satellite data to estimate and assess biophysical characteristics of grassland ecosystems including biomass and leaf area index (Briggs and Nellis 1989, Friedle *et al*. 1994, Chen and Brutsaert 1998). In addition, remotely sensed data have been used to discriminate among land cover and grassland types (Price *et al*. 1993), and textural algorithms have been used to discriminate among grassland communities (Lauver and Whistler 1993). Studies have also shown the usefulness of high spectral satellite imagery (Bradley and Mustard 2005) and multi-temporal imagery (Langley 2001) for detecting changes in grassland vegetation.

While previous studies have used high resolution spectral data from satellites and spectro-radiometers to estimate biophysical characteristics of grasslands and to discriminate among major grassland cover types in New Zealand (Vescovo *et al*. 2009), little research has quantitatively compared different methods for detecting changes in between grassland cover types. The objective of this study therefore is to compare a selection of different land-use/cover change methods (image differencing, NDVI differencing, post-classification, and manual mapping) to detect changes between different grassland cover types (low producing grasslands, depleted grasslands, tall tussock and exotic pasture) found in New Zealand. For the purpose of this study low producing, depleted and tall tussock grasslands are considered 'indigenous' grasslands (visualised as greenish brown, because of the high reflectance of red and medium-infrared bands) because they are extensively managed grasslands dominated by endemic tussock grass *Chionochloa, Poa,* and *Festuca* species. The non-indigenous grasslands include high producing grasslands (visualised orange-red in the false colour visualisation because of their high reflectance in the near-infrared band) which are intensively managed grasslands characterised by exotic European seeding grass species, such as short rotation rye grass, white clover and red clover. We measured the accuracy of each method to detect change between the indigenous grasslands and the non-indigenous grasslands. We also analyse temporal profiles of different grassland covers in order to try to understand the difficulties described in the literature of separating the four different grassland cover types.

Study Area

The study area is the Mackenzie Ecological Region (51,377km^2) in the centre of South Island, east of the Southern Alps (Figure 2.1). The Mackenzie Ecological Region has a semi-arid climate with high velocities of wind and highly variable seasonal temperatures. The high mean summer temperature is 20ºC and the average low winter temperature is –1ºC. Precipitation also widely varies between the seasons. The average annual precipitation is between 500 and 1000 mm, with most rainfall falling in the winter (June to September) (Leathwick 2003).

According to the New Zealand Land Cover Database (LCDB) approximately 91% of the study area is in 'indigenous' grasslands (depleted, low producing and tall tussock grasslands). The remaining 9% of the study area includes lakes and rivers (4%), exotic pasture (seeding European species) (4.6%), settlements (0.01%) and high alpine herbs (0.4%).

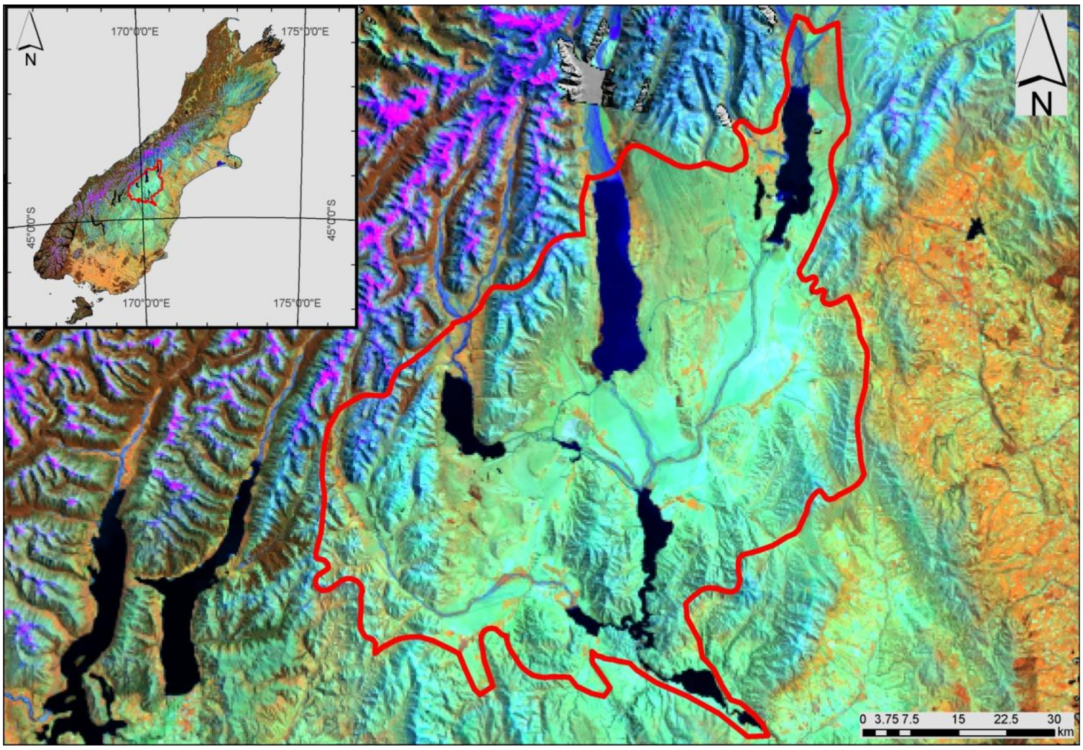


Figure 2.1 Location of the study area, the Mackenzie Ecological Region, is in south-west Canterbury in the South Island in New Zealand (Landsat 7 ETM+ bands 4, 5 and 3 mapped to red, green and blue).

2.3 Methods

Satellite data & pre-processing

The dataset for this study comprised Landsat 4 ETM+ and Landsat 7 ETM+ ortho-rectified satellite images, taken during the summers of 1989/1990 and 2001/2002. We removed the confusing effects of topography (Dymond and Shepherd 1999) by processing the imagery to standardised spectral reflectance, that is, reflectance on a flat surface, viewed from zenith, with a standard solar elevation (Dymond and Shepherd 2004). The ETM+ bands were also pan sharpened to 15m pixels using a local correlation filter (Dymond and Shepherd 2004) to retain the integrity of the original spectral signatures.

Four images were mosaicked to get a 95% cloud-free coverage of the Mackenzie Ecological Region. The images were geometrically corrected and geo-referenced to the New Zealand Map Grid (NZMG) coordinate system by using 2.5 meter pixel black-and-white ortho-photographs as reference. Approximately 25 evenly distributed ground control points (GCPs) pairs were selected to produce a mapping transformation with a root mean square (RMS) mapping error of 20 meters. Re-sampling was performed using cubic convolution.

The final mosaicked images of the Mackenzie Ecological Region were then masked to exclude areas that were not grasslands. A mask of grassland cover was created using the four grassland covers described in New Zealand’s Land Cover Database (low producing grasslands, depleted grasslands, tall tussock grasslands, and high producing grasslands) (Thompson 2003). The final images used for change detection included areas of grassland cover only, and excluded all other cover types.

Image differencing

The image differencing method resulted in a residual image which represents the change resulting from the subtraction of the two dates (nominally 1990 and 2002). Band 4 was used for the change detection because it is one of the most useful bands for detecting vegetation change (Singh and Yadava 1974, Singh 1986, 1989). Using ERDAS Imagine 9.2, a temporal difference image was derived using the standard formula:

$$Dx^{k}_{ij} = x^{k}_{ij}(t_2) - x^{k}_{ij}(t_1) \qquad (1)$$

where $x^{k}_{ij}(t_2)$ is the reflection of the ij th pixel in band k at time t_2.

Pixels of no reflectance change were distributed around the mean, while pixels of reflectance change were distributed in the tails of the distribution (Singh and Yadava 1974, Singh 1986). Large negative or large positive values corresponded to probable change. A thematic image of ‘change’ and ‘no change’ was produced by thresholding the difference image. A crucial component of this change detection method is the selection of a threshold value between ‘change’ and ‘no change’. Numerous techniques have been used in selecting thresholds (Stow *et al*., 1997, Phinn *et al*., 1999, Rogerson, 2002). For this study, we adopted the interactive approach used by Woodwell *et al*. (1983). Various standard threshold levels were applied to the lower and higher tail of each distribution in order to find the threshold value that produced the highest classification accuracy.

NDVI differencing

The NDVI is a widely used spectral vegetation index that has been correlated to biomass, plant productivity, and a variety of other vegetation parameters (Rouse *et al*. 1974, Tucker 1979). The NDVI is calculated from the red and near-infrared standardised reflectance images:

$$NDVI = (nir - r) / (nir + r) \qquad (2)$$

where nir is the standardised reflectance in the near-infrared band and r is the standardised reflectance in the red band. We calculated NDVI for both dates and then differenced them to create a change map (Nelson 1983, Singh 1986). We then selected the optimum threshold values of change by maximizing the classification accuracy associated with a given number of standard deviations.

Post-classification

We used the matrix operation tool from GIS Analysis in ERDAS Imagine to compare the two land cover thematic images, Land Cover Database 1 (LCDB1) and Land Cover Database 2 (LCDB2). LCDB1 was developed in 1997 using SPOT imagery, and updates where made in 2002/03 using Landsat 7 ETM+

satellite imagery, to create Land Cover Database 2 (LCDB 2) (Thompson *et al*., 2004). Each LCDB map consisted of a vector-based thematic classification of 43 land cover/uses, four of which were considered for this study: low producing grassland, tall tussock grassland, exotic pasture, and depleted grasslands. The resulting thematic image classified ‘change’ as any change between the above classes. We then used the matrix operation tool enable thematic recoding. The resulting file was a binomial thematic image with two classes, ‘change’ and ‘no change’.

Manual mapping and visual interpretation

Land-use change, from indigenous grassland (low producing grasslands, tall tussock and depleted grasslands) to a non-indigenous grassland cover (exotic pasture) was manually mapped using visual interpretation. Satellite imagery was used for interpretation and was supplemented with ortho-rectified aerial photography. Using ERDAS Imagine 9.1, each polygon of change was digitized at a display resolution 1:10,000. Digitizing was conducted using the area of interest (aoi) tool. The aoi file was then converted to a vector file. This file was then converted to a binomial raster layer of ‘change’ and ‘no change’.

Accuracy Assessment

Each change detection method was checked for accuracy using stratified random sampling. The change detection layers consisted of two strata, ‘change’ and ‘no change’. Within each stratum, at least 75 random samples were selected (Congalton 1991). Actual change was determined by visually examining the area around the selected points in a sequence of ortho-photographs at a 1:1,000 scale, using the three dates of imagery (1990, 1996, and 2002).

Classification accuracy was assessed using the ERDAS Imagine Accuracy Assessment utility. The overall classification accuracy was calculated from the error matrix by dividing the correctly classified samples (sum of the values in the main diagonal) by the total number of samples. The producer’s accuracy (errors of omission) and user’s accuracy (errors of commission) were also derived from the error matrix. The producer’s accuracy is calculated by dividing the number of correct pixels in one class by the total number of pixels as derived from reference data; the more errors of omission, the lower the producer's accuracy (Banko 1998). The user's accuracy measured the reliability of the map by dividing the correct classified pixels in a class by the total number of pixels. The Kappa coefficient was also calculated to provide an additional measure of the overall accuracy; it measured the proportion of agreement after chance agreements have been removed from consideration (Rosenfield 1986). For example, when the Kappa coefficient is zero the agreement between classified data and verification data equals chance agreement.

Trend Analysis

To explore the phenology of the four grassland covers, a time series of remotely sensed data was collected from SPOT 4 VEGETION (1km resolution) throughout the growing season. The VEGETATION sensor was selected because its wide swath provides daily coverage of the study area. We derived the average NDVI for each of the grassland cover types in the study area, and a time series between 1998 and 2007 of averaged ten-daily NDVI was produced.

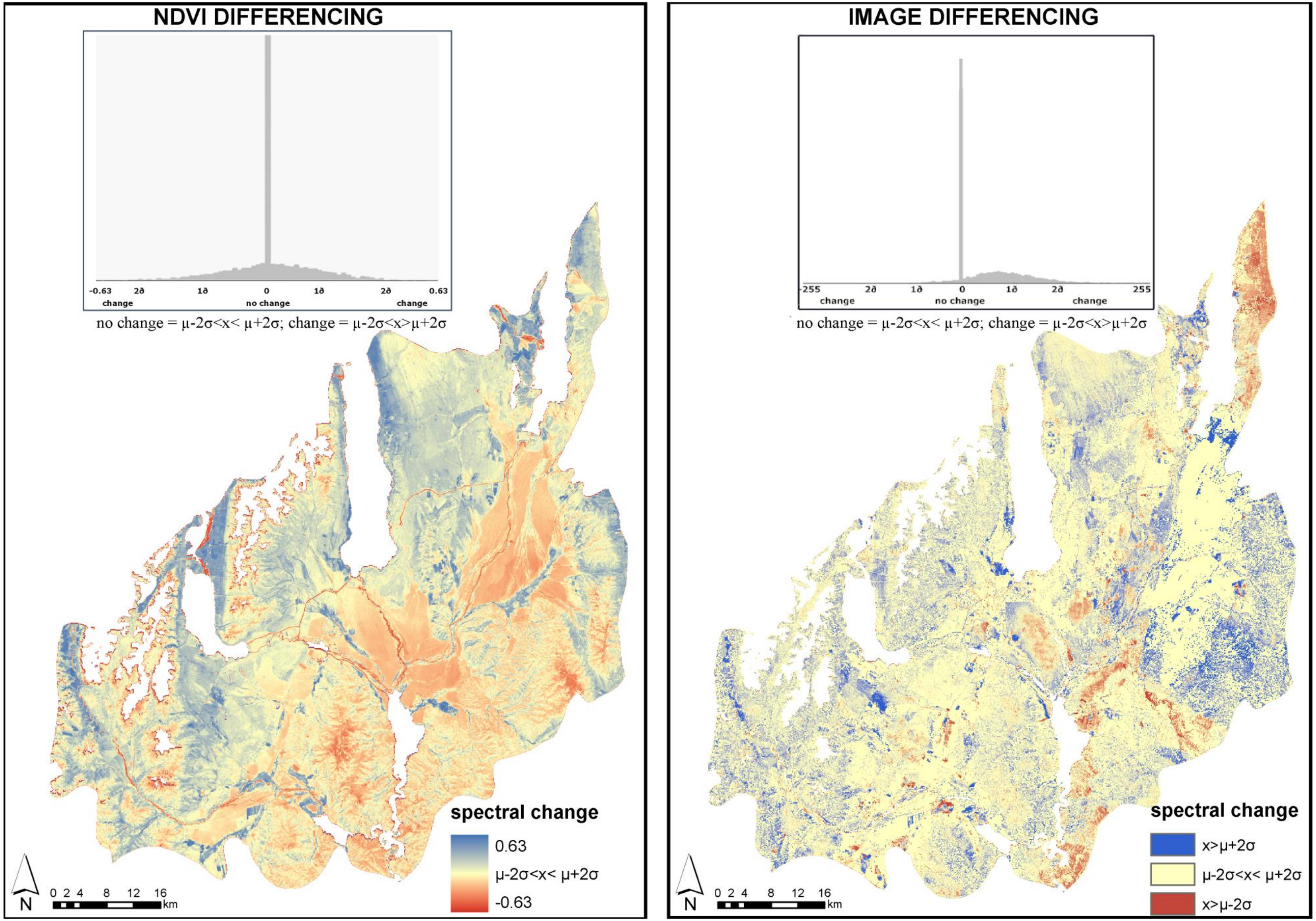
2.4 **Results**

Thresholds

Standard deviations of 1σ, 2 σ, 3σ and 4σ was tested for both the NDVI differencing and Image differencing data to define the most suitable threshold. As result of this assessment 2σ was the most accurate one among others and showed more important spectral variation between the two dates. Figure 2.2 shows the spatial distribution of spectral change for the two change detection methods, and the corresponding histogram. Difference between NDVI images ranged from -0.25 and -0.92 with and the range of difference between images using the image differencing method was -85 and 112. Threshold application was performed for each change detection method using the following formulas:

$$\text{no change} = \mu-2\sigma<x< \mu+2\sigma \text{ ,and change} = \mu-2\sigma<x>\mu+2\sigma \qquad (3)$$

This image was reclassified which resulted in a new image with the value of '0' assigned for 'no change' and '1' for changed areas. The total set of 'changed' and 'unchanged' pixels resulting from the above reclassification were used for the accuracy assessment.



2.2 Spatial distribution of spectral change between images and corresponding histogram used to define change threshold.

Comparison of change detection methods

Table 2.1 shows the accuracy of the four change detection methods. NDVI differencing had higher user's accuracy (97%) for the 'no change' class than for the 'change' class (11%). In addition, the producer's accuracy for the 'change' class was higher (81%) than the 'no change' class (52%). There was a noticeable difference between the accuracy of 'no change' and 'change' detection using the NDVI differencing method. With the 'no change' classification, the user's accuracy 98% compared to 13% for the 'change' classification. There were more errors of omission in the 'change' class than errors of commission, resulting in a user's accuracy of 53%. In comparison, the post-classification method and visual interpretation also had high user's accuracy (98%, and 99% respectively) for the 'no change' detection. However, the user's accuracy for 'change' was the lowest (4%) for the post-classification and highest (97%) for visual interpretation.

Table 2.1 Accuracy (%) of the change detection methods (image differencing, NDVI differencing, post classification, and visual interpretation).

Class Name	Producers Accuracy	Users Accuracy	Overall Accuracy	Kappa
		Image differencing		
no change	81	97	54	0.09
change	52	11		
		NDVI differencing		
no change	95	98	56	0.12
change	53	13		
		Post-classification		
no change	45	98	47	0.04
change	100	4		
		Visual interpretation		
no change	97	99	98	0.97
change	98	97		

Visual interpretation attained the highest overall accuracy of 98% and post-classification had the lowest overall accuracy (47%). Of the two automatic detection methods, NDVI differencing attained a slightly higher accuracy (56%) than image differencing (54%). The Kappa statistics for all methods other than visual interpretation was low. Post-classification had the lowest Kappa statistic (0.04), followed by image differencing (0.09) and NDVI differencing (0.12). Visual interpretation had the highest kappa statistic (0.96).

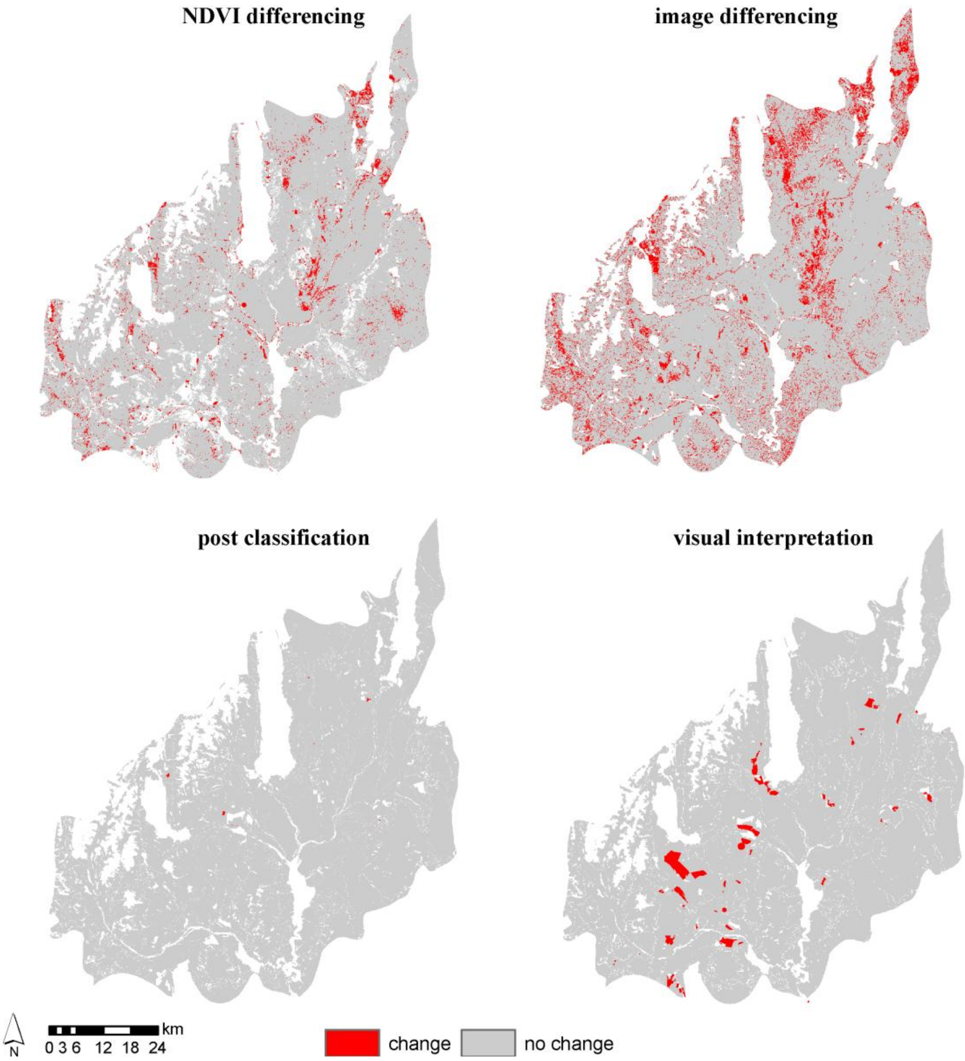


Figure 2.3 A comparison of the distribution of 'change' and 'no change' detected using four different change detection methods (image differencing, NDVI differencing, post classification, and visual interpretation).

Image differencing, NDVI differencing, and post-classification methods were unable to detect all land cover changes (Figure 2.3). Image differencing, however, was able to detect more (87,200 ha) 'change' than that of post classification (387 ha) or NDVI differencing (20,430 ha) (Figure 2.4). Though image differencing detected more change, much of the 'change' detected was not true 'change'.

Attempts to reduce these inaccuracies by adjusting the threshold proved unsuccessful.

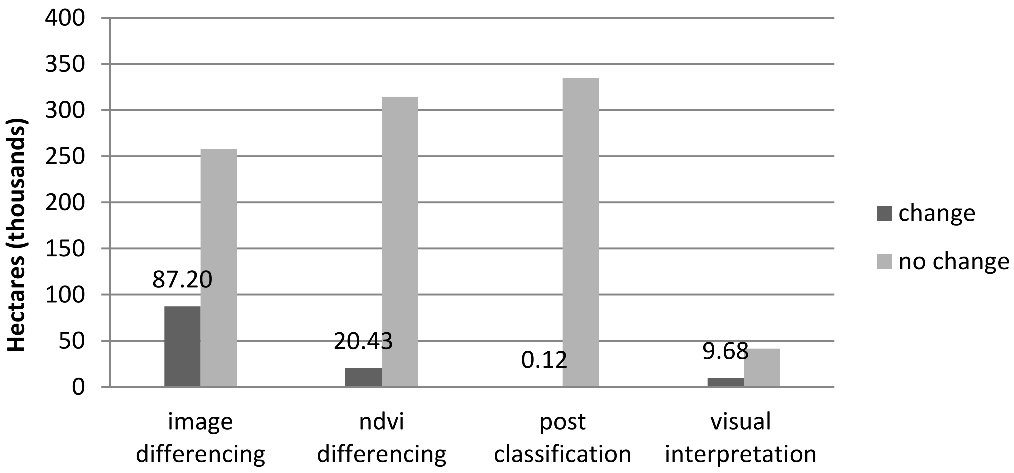


Figure 2.4 A comparison of the area of change (measured in thousands of hectares) detected for each change detection method (image differencing, NDVI differencing, post-classification, and visual interpretation). The reference data was collected using arial photographs and ground-truthing.

Figure 2.5 highlights the differences between the four change detection methods. The visual interpretation method produced the most accurate map (Figure 4f). Image differencing (Figure 4c) resulted in a change map with a scattered distribution of change. Extensive areas were mapped as land use/cover ‘change’. NDVI differencing (Figure 4d) produced a similar map to that of image differencing; however, unlike image differencing, it underestimated the distribution of change. The post-classification (Figure 4e) method produced a change map with little detected change.

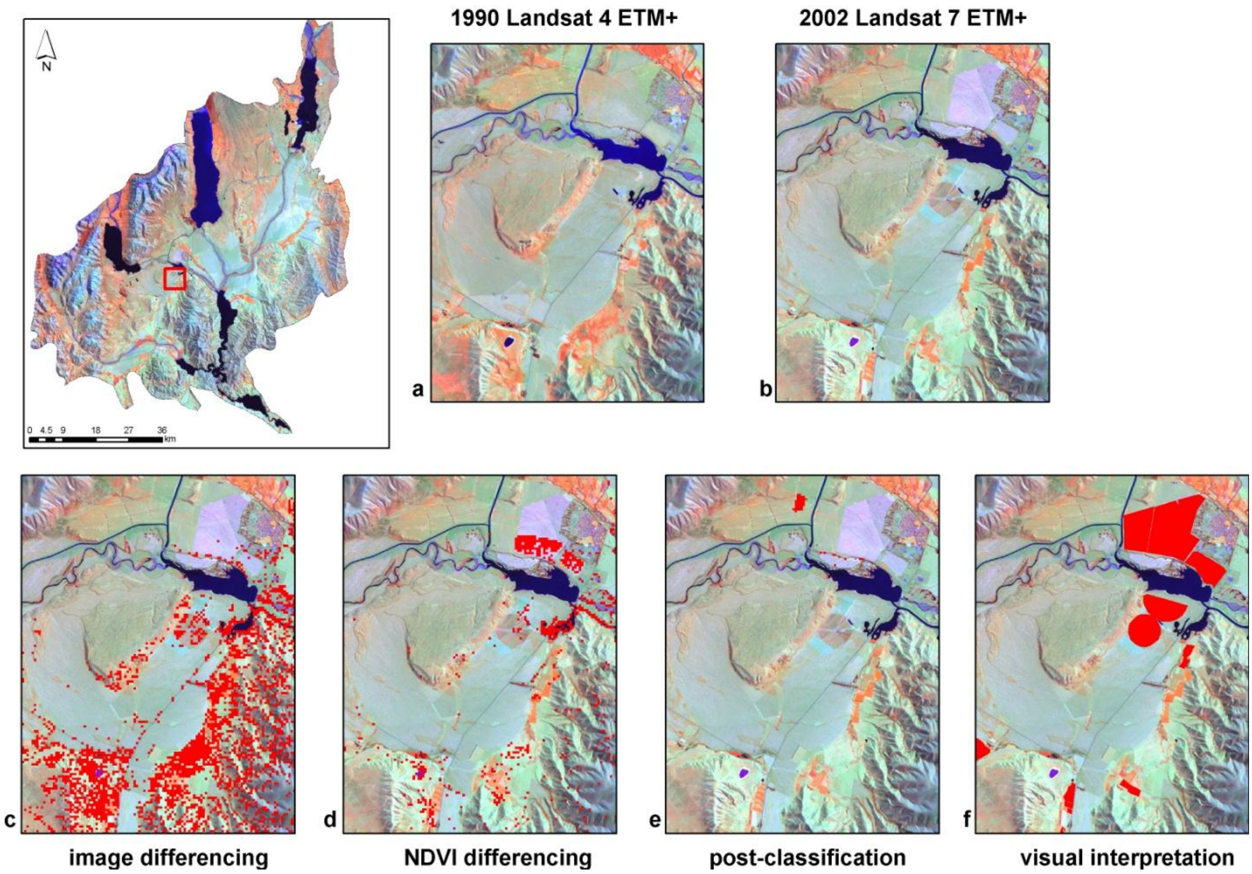


Figure 2.5 A comparison of the distribution of change for four change detection methods (image differencing, NDVI differencing, post-classification, and visual interpretation). Figure (a) illustrates the land cover in 1990 and (b) in 2001 (Landsat ETM+ bands 4,5 and 3 mapped to red, green, and blue). Figures c-f illustrates the changes (in red) detected for each change detection method.

Trend Analysis

Figure 2.6 shows the time series plot of NDVI for the five grassland cover types. There is significant inter-annual and seasonal variation in spectral response. The low producing and depleted grasslands tend to have greater within-class variability than the more homogenous tall tussock and high producing exotic grasslands. This illustrates the effect of changes in climatic and vegetation condition from season to season. While the spectral response for high producing exotic grasslands and tall tussock is more consistent from year to year, there remains a high variability in low low producing and depleted grasslands due to various factors including atmospheric conditions, soil moisture, vegetation phenology, and the extent of bare ground.

Though there is inter-annual and seasonal variability in spectral response, there is a predictable pattern of spectral change from one date to the next that gives rise to different spectral "starting points" along the different spectral trajectories. For example, tracking the mean NDVI response of each grassland type shows high producing exotic grasslands move in opposite directions, in spectral space, than tall tussock (indigenous vegetation cover). More specifically, during the summer months the NDVI of tall tussock increases from 0.35 to 0.5 while exotic grasslands decrease from 0.8 to 0.55. The opposite happens during the winter months, where the NDVI of exotic grassland increases and the NDVI of tall tussock decreases. The other grassland types (low producing and depleted grasslands), which consist of a mixture of exotic and indigenous species, follow the same seasonal trends as exotic grasslands.

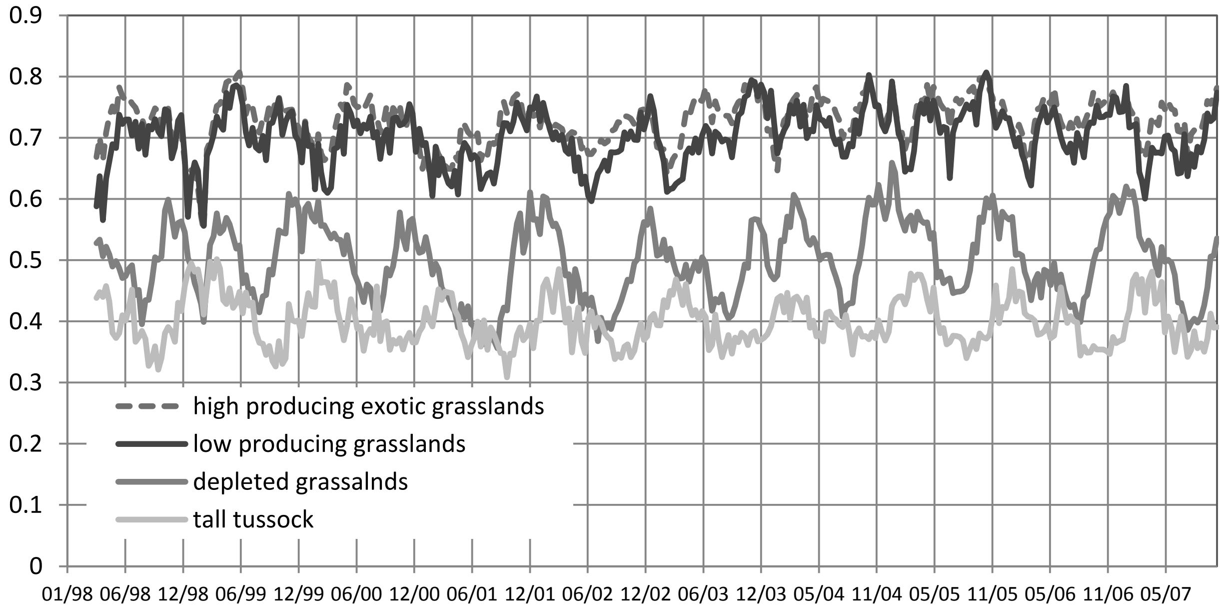


Figure 2.6 Daily average NDVI of four different grassland covers (high producing exotic grasslands, low producing grasslands, depleted grasslands, and tall tussock (see Thompson 2003) plotted between January 1998 and May 2007.

While the grassland types generally had distinct spectral trajectories, they showed overlapping ranges (Figure 2.7). The NDVI of low producing grasslands and high producing grasslands pasture ranged between 0.54 and 0.81, with both having a similar seasonal variation. The NDVI of depleted grasslands ranged between 0.35 and 0.66, and the upper extreme values overlapped with the lower extreme values of low producing grasslands and high producing grasslands. Tall tussock had a lower NDVI range than the other grassland covers (0.31 to 0.50). The upper and lower quartile overlapped with the lower extremes of depleted grasslands.

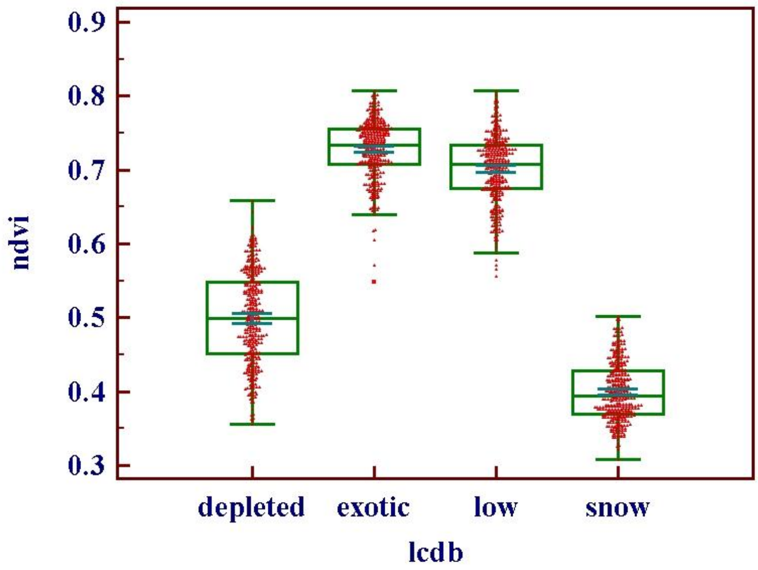


Figure 2.7 Box-and-whisker showing the distribution of NDVI for each grassland cover type: low (low producing grasslands), exotic (exotic grasslands), depleted (depleted grasslands), snow (snow tussock grasslands). The central box represents the values from the lower to upper quartile (25-75 percentiles). The middle line is the median, and the horizontal line extends from the minimum point to the maximum value, except outliers (red box) which are displayed as separate points.

2.5 Discussion

Current remote sensing methods have enabled the application of land-use/cover change in wide variety of temperate ecosystems (Lu *et al.* 2004). However, detecting land-use change in the New Zealand indigenous grasslands proved difficult. This is due to (i) the high temporal variability of the spectral properties of major grassland covers causing within-class spectral variability; (ii) the spectral reflectance of major grassland covers (the “to” class) overlaps with that of tussock grasslands (the “from” class) and, (iii) the varied spatial frequency of the landscape results in complex scenes.

The basic premise in using satellite imagery for change detection is that changes in land-use/cover result in changes in spectral reflectance. Therefore change detection for natural ecosystems generally requires the phenology of the different vegetation covers to be distinct. For example, the conversion of anthropogenic materials through urbanization can be readily identified and mapped (Masek *et al.*

2000), and land cover dynamics in tropical regions such as cutting forest for pasture have well documented trajectories of spectral properties over time (Adams *et al.* 1995, Skole and Tuker 1993). However, in New Zealand's grasslands the baseline conditions from which measures of change are difficult to define. Here, changes in remotely sensed surface properties of grassland cover can be a result of high degree of species elasticity in response to rainfall, not land-use/cover change, and because it is difficult to distinguish between different grassland cover types.

Automatic classification methods (i.e. image and NDVI differencing) have proved useful for estimating change at a single date, either directly (Dymond *et al*. 2008a) or as a stratum in random sampling (McRoberts *et al*., 2002, Czaplewski and Patterson, 2003), but for estimating change in New Zealand's grasslands the accuracy is low. This was due to the complexity and variability in the spatial patterns of the grassland ecosystems, making the spectral reflectance indistinct. In addition, the temporal variability makes it difficult to decipher between land-use/cover change and seasonal change. As a result, none of the automatic change detection methods were able to reliably detect actual land-use/cover change. Furthermore, these methods were unable to provide adequate qualitative information regarding the nature of change.

Previous comparisons of change detection methods found that post-classification methods resulted in reasonably accurate land-use/cover change maps (Mas 1999, Berberoglu and Akin 2009). While this method is not influenced by the variations of soil moisture and vegetation phenology, the accuracy is completely dependent on the accuracy of the initial classification (Coppin *et al*. 2001). Thus, mis-classifications and mis-registration errors present in the before and after maps are compounded.
In this study, the post-classification method detected land-use change in grasslands with low accuracy. This is because the maps that were compared (LCDB 1 & 2) for this method was generated by looking specifically at changes in woody vegetation. Even though grassland cover classes existed in the LCDB 1 & 2 classifications, changes between them were not really considered, unless very obvious. Therefore, it must be noted that though the accuracy for mapping grassland change using this method is low, it does not reflect the overall quality of the land cover databases, nor the accuracy of post-classification as a method. It

does, however, highlight the importance of selecting the appropriate thematic map for the post-classification method to be informative.

Vescova *et al*. (2009) conducted a preliminary study of mapping biomass and cover in New Zealand Grasslands using 2003/2004 Landsat imagery. They found marked variability between different grassland types, though they noted that indigenous tussock grasslands showed a "very similar" spectral signature to depleted low-productive areas. Other studies have found single-date Landsat TM data provided a reliable method for mapping vegetation cover in semi-arid regions (Langley 2001). Our study shows that further research is needed to develop a method to detect changes in grasslands in New Zealand. For example, it may be possible to adapt a hyper-spectral method, or a regression method exploiting several narrow bands, to detect temporal land-use changes in grasslands.

For a new change detection method to be useful it needs to be easily implemented and accurate. Although a variety of change detection methods have been successfully implemented across different landscapes, it is still difficult to match suitable methods to specific ecosystems. At present there is an urgent need for a method suitable to detecting land-use/cover change in New Zealand's grasslands. Rapid changes in land use are expected to have continued impacts in grassland biodiversity over the next century (White 2000, Walker *et al*. 2006). At present, in the absence of an accurate automated method, we recommend using visual interpretation and manual mapping to monitor land-use/cover change in New Zealand's indigenous grasslands.

2.6 **Conclusion**

Our results demonstrate the limitations to using image differencing and NDVI differencing for detecting change in New Zealand's indigenous grasslands. In addition this study highlighted some of the pitfalls when using New Zealand's existing land cover maps for post-classification analysis. Visual interpretation resulted in the highest accuracy for detecting change, suggesting that contextual information is needed to determine changes from one grassland cover to another. Further research is needed to explore alternative methods for detecting land-use change in New Zealand's grasslands. Therefore at present, for the most cost effective method to quantify accurately the extent of land-use/cover change in

New Zealand's grasslands, we suggest using a combination of visual interpretation and field work.

2.7 Acknowledgements

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3 Patterns of past and recent conversion of indigenous grasslands in the South Island of New Zealand[3]

3.1 Abstract

Although the area of formally protected temperate grasslands in New Zealand has increased in recent decades, low to mid-altitude systems continue to be poorly protected and land use intensification has accelerated in recent years. Recently acquired satellite imagery enables improved estimates of the extent, type, and rate of conversion of New Zealand's remaining indigenous grasslands. Remaining indigenous grassland in the South Island was reduced by 3% between 1990 and 2008 and replaced with exotic pasture, forestry plantations, and perennial crops. Although the overall rate of grassland conversion has decreased relative to the period between human settlement and 1990, the proportion of remaining indigenous grassland converted each year has increased. Almost two-thirds of post-1990 conversion has occurred in environments with less than 30% of indigenous cover remaining, and much is in land classified as non-arable with moderate to extreme limitations to crop, pasture and forestry growth. Two-thirds was recorded in the Waitaki, Mackenzie and Central Otago administrative districts. Opportunities to further protect more of the full range of indigenous grasslands lie where the government tenure review process continues in these districts.

*Keywords***:** loss of indigenous habitat, South Island, remote sensing, rate of habitat loss, land tenure

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[3] Accepted as Weeks E.S., Walker S., Dymond J.R., Shepherd J.D., Clarkson B., in *New Zealand Journal of Ecology*

3.2 **Introduction**

Most of the world's indigenous grasslands have been converted for agricultural activities (Goobridge 1992). Areas with better soils and more frequent rainfall have been mostly cleared for crops, while poorer quality grasslands have been left for rearing stock (Suttie et al. 2005). Globally there is limited information on the rate, type, and amount of change that is occurring in grassland ecosystems (Pearson & Ison 1997; White et al. 2000) and New Zealand is no exception. Without fundamental information on trends occurring in grasslands, researchers are unable to assess potential effects of land conversion on habitat and their associated biodiversity and ecosystem services, and policy makers lack the evidence needed to inform sound policy formation (Gluckman 2011).

Since European settlement more than 60% of all New Zealand's indigenous habitats have been converted for agricultural and forestry (McGlone 2001). In the past the most threatened ecosystems have been considered to be lowland forests, coastal dunes and wetlands (Stevens et al. 1988; Ogden et al. 1998; Leathwick 2001; McGlone 2001), but remaining indigenous grasslands are also under threat from expansion of intensive agricultural land uses (Ewers et al. 2006; Walker et al. 2006). Over 95% of New Zealand's remaining indigenous grasslands are located in the South Island. These indigenous tussock grasslands have a partially human induced origin and provide a range of important ecosystem services, i.e. water regulation and soil formation, including significant cultural value, to New Zealanders (McAlpine & Wotton, 2009). Most of the lowland and montane regions of tussock grassland were created by Māori burning and clearing of forest and shrubland, for hunting moa and encouraging the growth of bracken fern (*Pteridium esculentum*)[4] (Stevens et al. 1988; Ewers et al. 2006). Initially the short tussock grasses (*Festuca* and *Poa* species) dominated, but within about 200 years were replaced by taller large *Chionochloa* species (McGlone 2001).

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 against their current extent using New Zealand

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[4] For the purpose of this paper the species names are based on nomenclature used by Landcare Research New Zealand Plants Database.

Land Cover Database 1 (LCDB1; Thompson et al. 2003). They estimated that in 1840 (the beginning of formal European settlement), 31% of New Zealand was covered by indigenous grasslands dominated by endemic tussock grass species, but that just 44% of this area remained in 2002, mainly in the interior areas of the South Island. Of this remaining area, approximately 28% had statutory protection with a bias towards the high-alpine areas. Mark and McLennan (2005) noted that remaining subalpine grassland communities (i.e. short tussock grasslands) still persisted but were severely degraded and/or modified, and very poorly protected.

With the release of Land Cover Database 2 (LCDB2) (Thompson et al. 2003), more recent land cover change could be detected. However, though the automatic detection technology used to identify likely change between the two dates provided reliable estimates of change in woody vegetation, it was not informative for non-woody vegetation change. Identifying change in herbaceous vegetation is difficult because of temporal variability in soil moisture which has a greater effect than on woody vegetation (Dymond et al. 2006). Consequently, estimates of areas and rates of change in grasslands derived from comparisons of LCDB1 and LCDB2 are conservative and misleading (Walker et al. 2006).

Informal observations and limited quantitative data suggest that the conversion (land-use change) of New Zealand's indigenous grasslands is proceeding rapidly in the South Island. In addition, a process of Crown land reform (called tenure review) has led to the division of substantial areas of former pastoral land into separate private and conservation parcels (Brower 2008; Walker et al. 2008; Mark et al 2009). On land transferred to private ownership, reduced legislative constraints on vegetation clearance and/or reduced property size (Brower 2008) may be accelerating loss of habitat of indigenous species, including indigenous tussock grasslands.

Aerial photography and satellite imagery provide appropriate data for monitoring conversion of land cover at regional or national scales. Substantial improvements to the standardization, illumination, and viewing geometry (Dymond & Shepherd, 2004), along with enhanced spatial resolution have significantly improved the ability to detect changes in grassland cover. We used Landsat 7 ETM+ images taken in the summers of 1989/1990, and 2000/01, and SPOT-5 imagery taken during the summer of 2007/08 to quantify the extent of conversion of indigenous grassland habitat in the South Island between 1990 and 2008, and estimate the current (2008) extent of remaining indigenous grassland cover in the South Island within the study area. Using this data we compare the rate of past (1840-1990) and recent (1990-2008) conversion, and identify the types and patterns of conversions that result in loss of habitat for indigenous species in different ecological districts, land environments, land use capabilities, and administrative districts.

Study Area

To objectively delineate the study area covering indigenous grasslands in the South Island, we created a median reflectance image using SPOT 4 satellite imagery (1 km spatial resolution, daily coverage from the VEGETATION instrument) that was collected between 1990 and 2003. Spectral reflectance at each pixel was ranked from highest to lowest and the median value pixel extracted. Median reflectance was preferred to mean to eliminate the effects of cloud coverage which skews the average reflectance. Using ERDAS Imagine 9.1 (Leica Geosystems Geospatial Imaging 2003) the study area was then specified by applying the automatic region growing tool. A seed point of indigenous grassland was selected from ground truth data collected during the summer of 2008. From the seed point, all pixels with a median spectral Euclidean distance within 0.7 (7% spectral variation), and that were considered contiguous, were accepted. The total study area included the largest continuous extent of unprotected indigenous grasslands in the South Island and amounted to 4.3 million hectares (Fig 3.1).

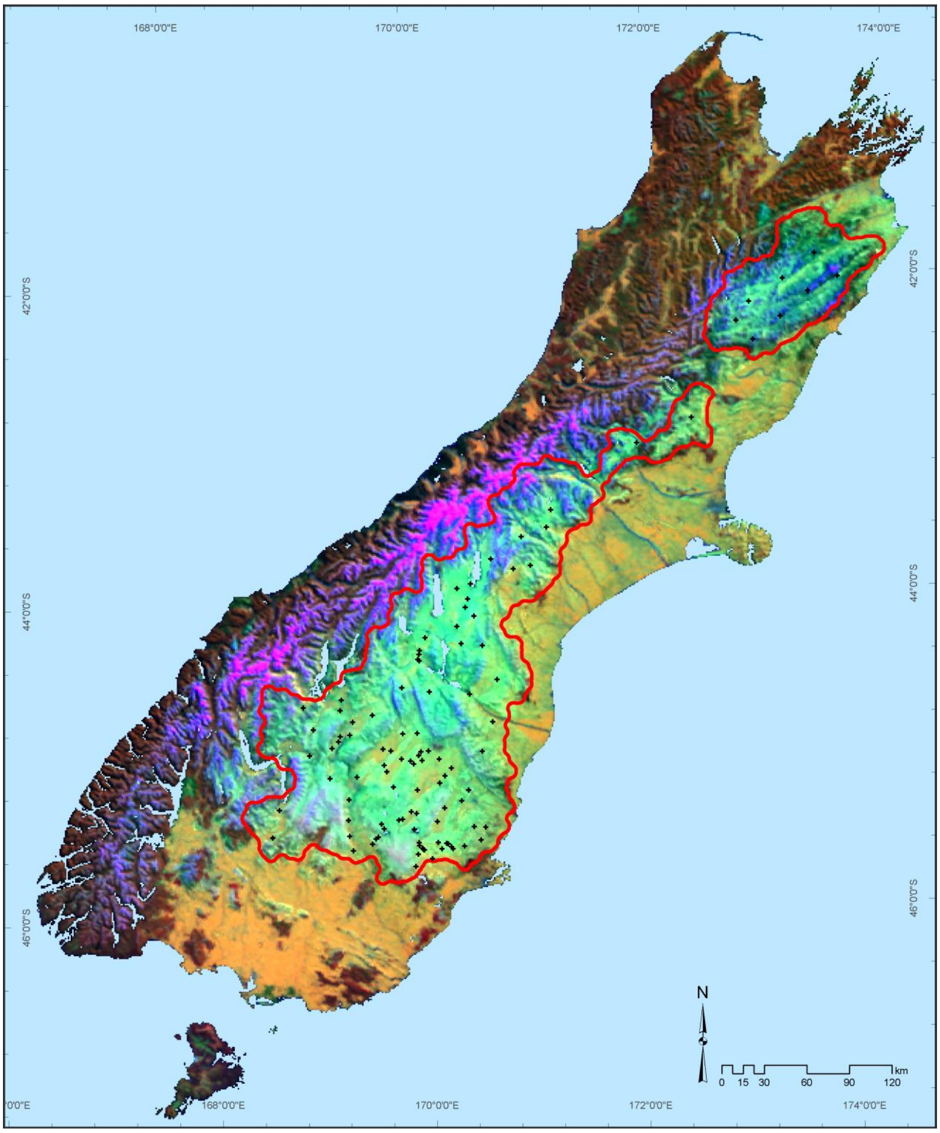

Figure 3.1 Study area (within red border),and distribution of ground field checks (+) over laid onto the median reflectance image created using (1990-2003) imagery from the VEGETATION sensor from SPOT-4 imagery (bands 3, 4 and 2 mapped to red, green, and blue).

3.3 Methods

Data pre-processing for detection of conversion

We used three satellite images: Landsat 7 ETM+ images taken in the summers of 1989/1990, and 2000/01, and SPOT-5 imagery taken during the summer of 2007/08. The Landsat 7 ETM+ imagery was orthorectified and standardised for reflectance using methods described by Shepherd and Dymond (2003). ERDAS Imagine 9.1 was used to geometrically correct the raw SPOT-5 digital imagery. Each image was ortho-rectified to the New Zealand Map Grid using the SPOT-5 orbital pushbroom model and 15 m digital elevation model (DEM) (Shepherd & Dymond, 2003). The standardised spectral reflectance was calculated assuming a nadir-viewing satellite sensor, and a 50 degree Sun elevation. The 6S code was used to model irradiance and transmission of light through the atmosphere, and the WAKII model (Dymond et al., 2001) was used to describe and standardised the directional reflectance properties of the land-cover and terrain. The standardised reflectance of the SPOT-5 imagery (bands 1, 2, 3, and 4) of invariant targets after application of cross-sensor response function calibration agreed with the standardised reflectance from 2001 ETM+ imagery (bands 2, 3, 4 and 5) to within ± 0.01 or within 6% of the reflectance for slope angles up to 45 degrees (excluding sun incidence angles less than 5 degrees).

Mapping of conversion

We mapped grassland conversion in three time intervals: 1840-1990, 1990-2001, and 2001-2008. To map conversion from 1840 to 1990 we used Mark and McLennan's (2005) estimate of the extent of indigenous grasslands in 1840, and compared it with the 1990 extent of remaining indigenous grasslands, within our study area. The 1990 indigenous grassland extent layer was estimated by combining the polygons of grasslands from the New Zealand Land Resource Information (NZLRI) vector layer and the woody vegetation from Eco Sat 1990 vector layer (Dymond & Shepherd 2004), and checking each polygon against the existing 1990 satellite imagery. Once the basic land cover map was corrected for digitizing errors it was aggregated into a land cover classification system described in Appendix 3.1.

Conversion from 1990 to 2001 and from 2001 to 2008 was mapped using three satellite images. All conversion from an indigenous grassland cover type (as defined by the 1990 land cover map) to a non-indigenous cover type was manually digitized at 10 meter resolution using the following band combination: Landsat (band_4, band_5, band_3) and for SPOT-5 (XS3, SWIR, XS2), which provided for easy identification of a clear distinction between indigenous grassland cover (i.e. snow tussock) and exotic cover (i.e. pasture). We mapped non-indigenous cover in six classes: planted forest, wilding forest, exotic pasture, cropland, settlement, and open-pit mining (Table 3.1).

Table 3.1 Description of the non-indigenous land cover types considered in the study.

Land cover type	Sub-type	Description
Afforestation	Planted forest	Radiata pine, Douglas Fir or other planted forestry tree species
	Wilding forest	wilding (not intentionally planted), Radiata pine, Douglas Fir, or other forestry tree species
Agriculture	Exotic pasture	Grasslands with exotic species (i.e. rye grass, clover)
	Cropland	Perennial and annual crops including cultivated bare ground
Barren Land	Settlement	Built-up areas and impervious services; grasslands with settlements including recreational areas
	Open-pit mining	Open pit mining.

To map conversion (changes in land-use) we used multiple sources of evidence, including information from satellite images, photographs, land-use databases, local knowledge, and field inspection. Satellite imagery was the primary source used for interpretation but was supplemented with existing land cover information, including aerial photography supplied by Terralink International. The process used multiple (up to 6) ERDAS Imagine viewers, each containing one of the three dates of satellite imagery, the 1990 indigenous grassland cover map, and aerial photographs. Each polygon of change was digitized on top of the satellite imagery, using the area of interest (aoi) tool in ERDAS Imagine, and all polygons were converted to a single vector layer. The area of conversion during each time period for each conversion type was then calculated.

Groundtruthing (field inspections) was used both to train the mapping process and also to confirm conversion between 1990 and 2008. For training, the operator recorded 250 different GPS points of land-use/cover in the field. At each GPS point photographs were taken, the land-cover/use was recorded, and the corresponding visualized spectral signature was identified in the satellite imagery. A laptop was connected to a GPS unit allowing for continuous tracking of the current position against the background of the satellite image. This was achieved using ArcView software, the Digital topographic database (Land Information New Zealand) and GPS Utility (GPS Utility Limited, UK) software.

The process of confirming conversion involved operator investigation of each of the 375 conversion polygons in the field and systematically traversing approximately 10 000 km of no-conversion from the ground and in the air, throughout the study area (Fig. 3.1). During the groundtruthing process we identified both errors of commission (whether the conversion identified in the satellite imagery was in fact present on the ground) and errors of omission (whether the conversion observed on the ground was correctly captured from the satellite imagery). Panoramic photographs were taken from the ground and oblique or vertical photos from the air. The aerial photographs were taken at 2 000 meters above sea level. In total these photographs amounted to approximately 1 500 location-specific photographs of different examples of conversion (Fig. 3.2). Once returning from the field, all mapping errors were corrected.

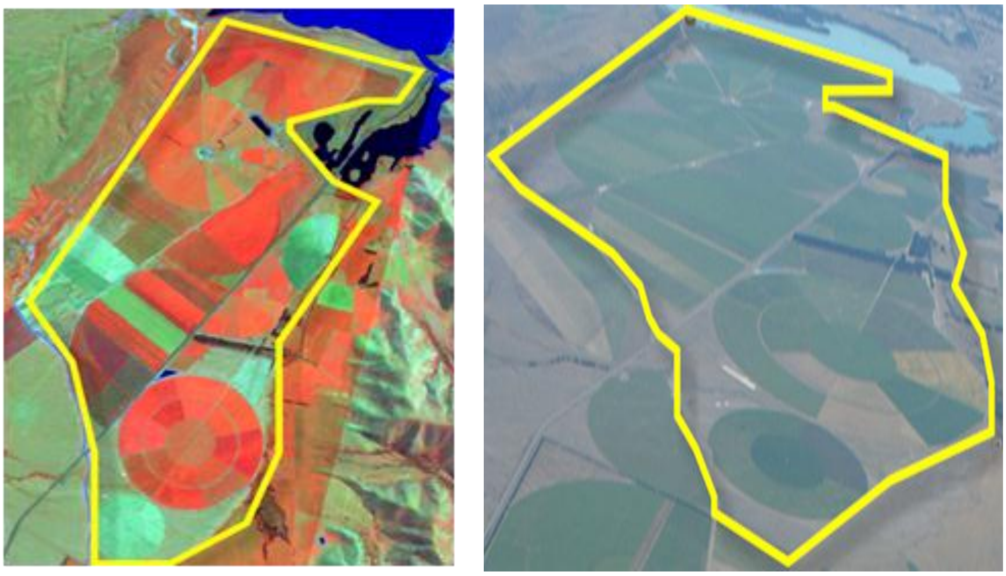

Figure 3.2 SPOT-5 imagery (left), showing the spectral signature of an irrigated pasture using band combinations XS3, SWIR, and XS2, and corresponding oblique photo (right) taken at 2,000 meters above sea level using a Canon A640 7.1 mega pixel compact camera.

Accuracy Assessment

Once the final map of conversion was completed the mapping accuracy of the operator was then assessed using a purpose built software package designed to manage on-screen ERDAS Imagine viewer content and enable a rapid pixel by pixel assessment (Shepherd JD unpublished software application). Five hundred points were randomly sampled in areas mapped as “converted” and two thousand points were randomly sampled in areas mapped as “non-converted”. A second operator assessed whether conversion was correctly or incorrectly identified in order to produce a confusion matrix (Congalton, 1991). From the confusion matrix the mapping accuracy of “conversion” was calculated and the mapping accuracy of “no-conversion” was calculated. The overall classification accuracy was also estimated from the confusion matrix.

Rates of conversion

For each time period we estimated an average rate of conversion. The rate of conversion, or loss, per year (r) was calculated as an average conversion for each time period using the formula:

$$r = \frac{(A_0 - A_1)}{(t_0 - t_1)}$$

where A_0 = area at time t_0, and A_1= area at time t_1 .

Analysis of patterns of conversion

As the basis for describing patterns of grassland conversion we used a combinatorial analysis of datasets, run in GIS using the ArcSampling program developed by Landcare Research. The digitized vector layer was converted to a 25 m raster layer and was combined with the Land Environments of New Zealand (LENZ) (Leathwick et al., 2003), Land Use Capability (LUC) from Land Resource Information System (LRI) (Newsome et al., 2000), a classification of ‘Threatened Environments’ (Appendix 3.2; Walker et al. 2006), digital elevation model (DEM) for New Zealand (Barringer et al. 2002), protected areas of New Zealand (PANZ), crown owned leased land, a selection of environmental layers (rainfall, slope and mean annual temperature) used for LENZ (Leathwick et al. 2003), regional council and district boundaries (Newsome et al. 2000), and ecological districts (McEwen 1987). The ArcSampling program created a raster layer of all unique combinations of input classes, and tabulated the area of each unique combination for import into Microsoft Access. Microsoft Access and Excel (Microsoft 2007) were used for subsequent calculations and tabulations.

3.4 **Results**

Remaining indigenous grasslands

The dataset adapted from Mark and McLennan (2005) suggests that in 1840 there were approximately 3.3 million hectares of indigenous grasslands within the study area. By 1990, 30.5% of these grasslands were converted to a non-indigenous cover type (Table 3.2). The remaining 2.3 million hectares of indigenous grasslands provided a baseline for detecting grassland conversion between 1990 and 2008.

Table 3.2 Original extent of indigenous grasslands in 1840 (adapted from Mark & McLennan 2005) and the remaining extent in 1990, 2001 and 2008. The total loss (in hectares) in the preceding time period is shown in the second column and the percentage loss of the remaining grasslands from the previous time period is show in the last column. The total hectares and percentage loss since 1840 are shown in the last row.

Year	Remaining grasslands (hectare)	Loss in preceding time period (hectares)	% loss of remaining area in preceding time period
1840	3,318,991	-	-
1990	2,307,691	1,011,300	30.5
2001	2,269,566	38,125	1.65
2008	2,236,430	33,136	1.46
TOTAL		1,082,561	33.58

Indigenous grassland cover has continued to decline since 1990. Between 1990 and 2001, 38 125 hectares of indigenous grasslands were converted to a non-indigenous cover type. An additional 33 136 hectares were converted between 2001 and 2008. By 2008, 3% of the 1990 indigenous cover was converted, leaving 2.2 million hectares of indigenous grasslands within the study area (Fig. 3.3).

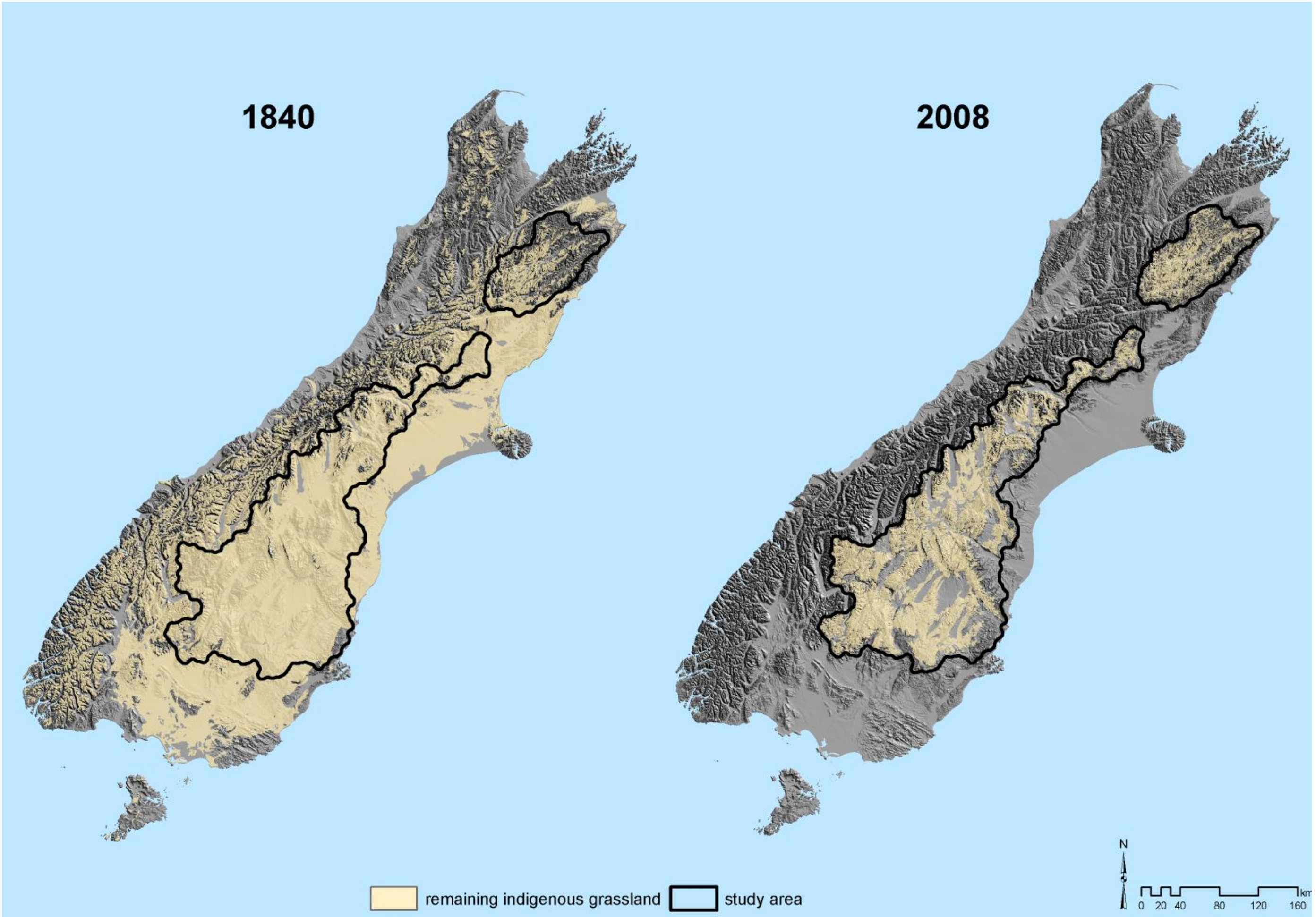

Figure 3.3 The original extent of indigenous grasslands in 1840 (left) (adapted from Mark & McLennan (2005), and the extent of remaining indigenous grasslands in 2008 (right).

Conversion between 1990 and 2008

Grassland conversion maps for 1990, 2001, and 2008 are displayed in Figure 3.4. Mapping accuracy may be determined from the confusion matrix shown in Table 3.3. The conversion mapping accuracy was 97.4% and the no-conversion mapping accuracy was 99.85%.

Table 3.3 Confusion matrix showing accuracy (%) of mapping.

	Mapped as conversion	Mapped as no-conversion
Observed conversion	487	3
Observed no-conversion	13	1997
Percentage correct	97.4%	99.85%

Of the 71 261 hectares of indigenous grasslands converted between 1990 and 2008, 50 314 hectares (71%) were converted for agriculture (Table 3.4). This included 47 656 ha for pasture and 3 903 ha for cropland. The remaining area was converted for: afforestation (17 637 ha), mining (1 688 ha) and urban development (174 ha). Though most of the afforestation was from planted trees (15 887 ha) some resulted from the spreading of wilding trees (1 750 ha).

There were some differences between the types of changes during the two time periods. In the 11 years from 1990 to 2001, agriculture (25 361 ha) followed by afforestation (11 066 ha) accounted for the majority of the 38 125 ha of indigenous grasslands converted to a non-indigenous cover type, and mining converted a further 1 668 ha. Very little grassland conversion for urban development (29 ha) was detected. There was less (33 136 ha) change in total in the shorter (7-year) period from 2001 to 2008. During this time, 24 603 ha of grassland were converted for pasture, almost as much as in the preceding 11 year period. More land was converted for urban development (145 ha), and less land to exotic forest (6 571 ha) and for mining (195 ha) than in the preceding 11-year period.

Table 3.4Areas of grassland conversion by land cover type (areas in ha) from 1990 to 2001, and from 2001 to 2008. Non-indigenous land cover is grouped into three types: Afforestation, Agriculture, and Barren land (see also Appendix 3.1 for land use associations for each of these cover types).

	Afforestation		Agriculture		Barren land		
	Planted	Wilding	Pasture	Cropland	Mining	Urban	Total
1990-2001	9,724	1,342	23,053	2,308	1,668	29	38,125
2001-2008	6,163	408	24,603	1,622	195	145	33,136
Total	15,887	1,750	47,656	3,903	1,863	174	71,261

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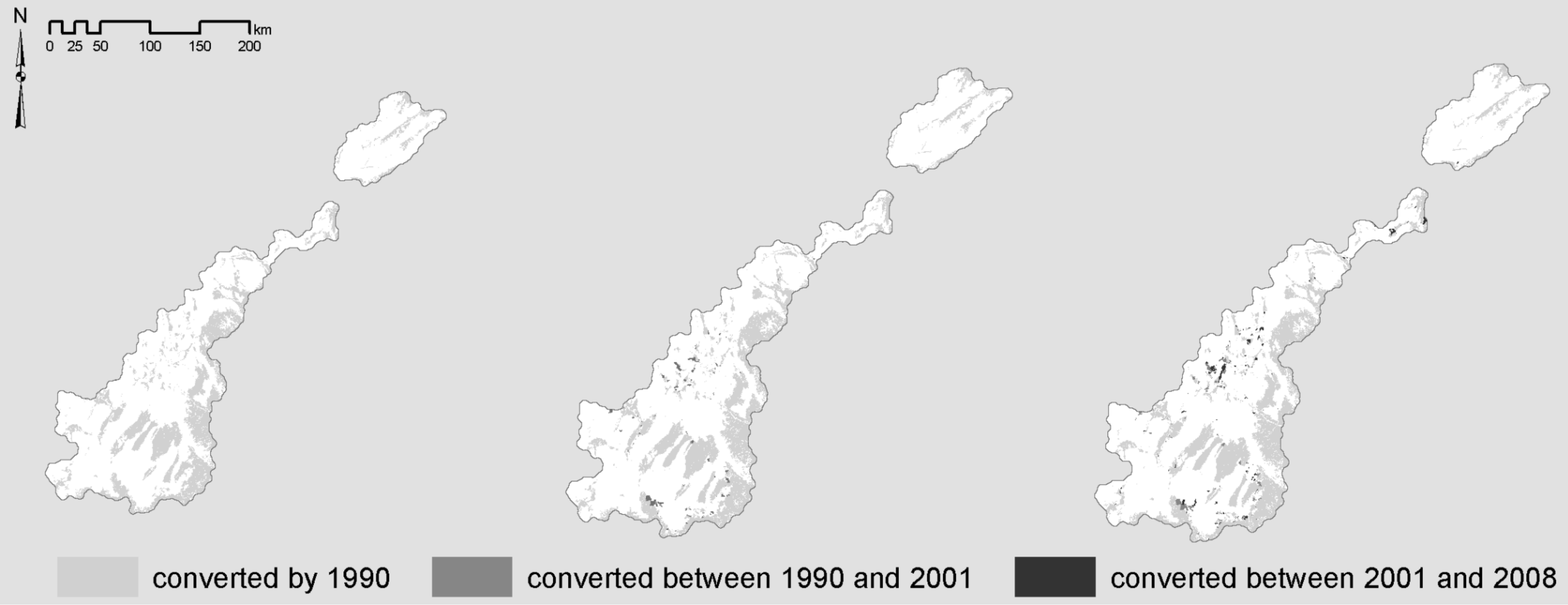


Figure 3.4 Progressive conversion of grasslands (light grey) by 1990 (left), from 1990 to 2001 (dark grey; middle), and from 2001 to 2008 (black; right). White areas (the background) represent areas not converted.

Rates of change

The rate of grassland conversion (ha yr^{-1}) has decreased relative to the period between European settlement and 1990. However, the proportion of remaining indigenous grassland converted each year has increased (Fig. 3.5). Between 1840 and 1990, 6 742 hectares of indigenous grasslands were converted each year on average (0.20% loss per year of remaining). Between 1990 and 2001, the rate of conversion was reduced to 3 466 hectares each year on average, (0.15% loss per year of remaining). Between 2001 and 2008, the rate of conversion increased to 4 734 hectares each year on average (0.21% loss per year of remaining).

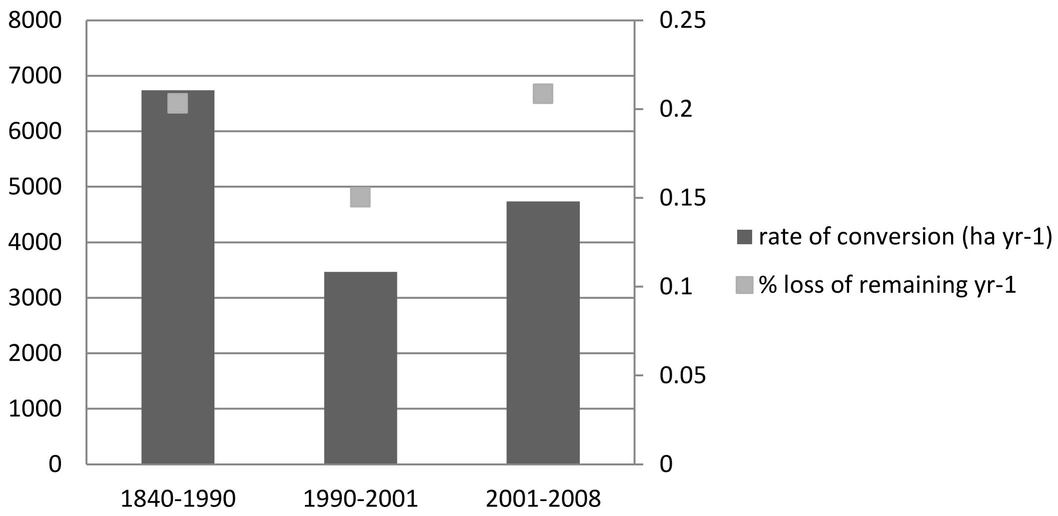


Figure 3.5 A comparison of the rate of conversion (ha yr^{-1}), columns, increments on left vertical axis), and the percentage loss (square grey symbols, increments on right vertical axis) of remaining grasslands per year during three time periods (from 1840 to 1990, from 1990 to 2001, and from 2001 to 2008).

Area, slope, elevation and rainfall

The area of individual conversion polygons from 1990 to 2001 and 2001 to 2008 ranged from 0.062 ha to 5 507ha (Figure 3.6). The median area of each polygon was 53 ha with a mean of 190 ha and a standard deviation of 462 ha. The largest polygons of conversion (>2 000 ha) were recorded during the 2001 and 2008 time period.

Grassland conversion took place at moderate elevations, slope and rainfall (Figure 3.6). The highest point of conversion was 1546 meters and the lowest was 192 meters with a median of 864 meters. These areas had a slope between 10º and 35º (median of 21°). There was no change found at slopes greater than 42º. Rainfall in these locations ranged from 337 mm to 2774 mm; however three-quarters of conversion had an average rainfall between 400 and 1600 mm (median of 1397 mm).

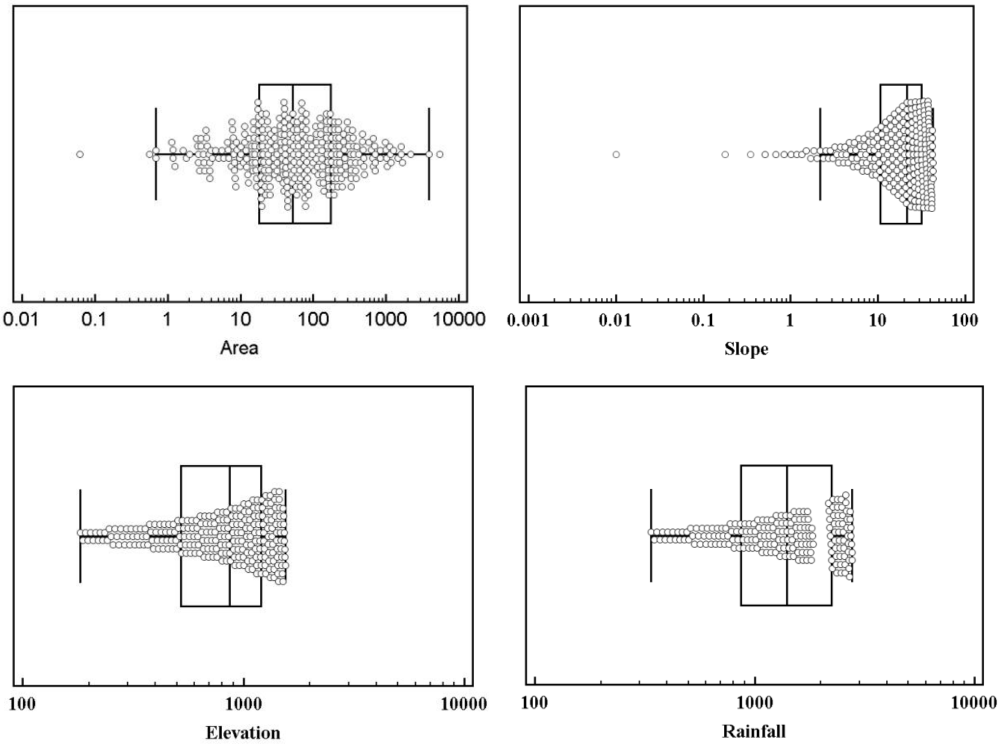


Figure 3.6 Box-and-whisker showing distribution of area, slope, elevation, and rainfall of conversion polygons on a log scale. Each dot (○) represents each individual polygon of conversion. The central box represents the values from the lower to upper quartile (25-75 percentile). The middle line is the median, and the horizontal line extends from the minimum point to the maximum value, except outliers which are displayed as separate points.

Administrative regions and districts

The majority (65 521 ha) of grassland conversion from 1990 to 2008 was concentrated in Canterbury and Otago administrative regions. Marlborough and Southland made up a small portion (<5%) of the study area, and less than 2% of grassland conversion was recorded in these regions.

Grassland conversion took place in 13 different districts (Table 3.5). Two-thirds of recorded conversion of indigenous grassland from 1990 to 2008 occurred in the Mackenzie (11 442 ha), Waitaki (22 159 ha), and Central Otago (15 850 ha) districts, and the majority (65%) of this conversion was on non-arable land with moderate (LUC 6) to extreme limitations (LUC 8) to crop, pasture and forestry growth. The most conversion on non-arable land was recorded in Mackenzie (7 474 ha), Waitaki (13 466 ha), and Central Otago (12 006 ha).

The Mackenzie and Waitaki district (both within the Canterbury region) also showed recent increases in the rate of conversion per year, which approximately doubled during the second period (2001-2008). Although the rate of conversion per year increased in most districts in the second time period (2001 to 2008), in Central Otago, Queenstown, Clutha and Marlborough districts there was a decrease.

The districts with the largest extent of remaining grassland in 2008 were Mackenzie (299 759 ha), Waitaki (259,521), and Central Otago (635 152) (Table 3.4). These districts also had the most remaining grasslands under lease from the crown in 2008 (166 759 ha in MacKenzie, 107 794 ha in Waitaki, and 265 199 in Central Otago), and the least area (in proportion to remaining grasslands) under protection (40 748 ha in MacKenzie, 43 093 ha in Waitaki, and 55 394 ha in Central Otago). Districts with the least remaining grasslands in 2008 were Selwyn (43 184 ha), Waimakariri (19 754 ha) and Clutha (23 185 ha), which also had <35% of their remaining grasslands protected.

Table 3.5 Hectares of remaining indigenous grasslands in 1990 and 2008, and recorded grassland conversion from 1990 to 2008 in council districts within our study area. The table shows conversion within all land and no-arable land (classes 6-8 in the LUC), and 2008 remaining grasslands in total and with two tenure categories (protected private and crown land and crown pastoral lease). The average rate (ha/yr) of conversion in each district in each of the two time periods (1990-2001 and 2001-2008) is shown on the right.

	Area of conversion (hectares)			Remaining Area (hectares)			Remaining (%)			Rate of conversion (hectares/year)	
District Council	Total 1990	Conversion 1990-2008	Conversion non-arable land 1990-2008	Protected 2008	Crown leased 2008	Total 2008	Protected 2008	Crown leased 2008	Total 2008	1990–2001	2001–2008
Canterbury											
Mackenzie	311,201	11,442	7,474	40,748	166,759	299,759	14	56	96	448	931
Waitaki	281,680	22,159	13,466	43,093	107,794	259,521	17	42	92	930	1,704
Waimate	117,312	752	331	11,058	54,192	116,560	9	46	99	28	64
Ashburton	116,989	518	442	36,831	64,228	116,471	32	55	99	13	53
Selwyn	44,065	881	477	5,908	8,692	43,184	14	20	98	25	86
Waimakariri	21,944	2,190	14	2,214	4,649	19,754	11	24	90	140	160
Hurunui	102,968	2,424	2,116	6,408	29,091	100,544	6	29	98	136	210
Otago											
Central Otago	651,002	15,850	12,006	55,394	265,199	635,152	9	42	98	985	716
Dunedin City	109,134	5,832	2,340	16,383	9,241	103,302	16	9	95	219	490
Queenstown	190,928	2,592	403	26,909	10,479	188,336	14	6	99	200	56
Clutha	24,066	881	926	4,486	8,320	23,185	19	36	96	62	29
Southland											
Southland	130,225	504	97	18,428	97,672	129,721	14	75	99	12	54
Marlborough											
Marlborough	217,127	5,268	317	32,604	115,094	211,859	15	54	98	359	190

Ecological districts

The study area includes parts of eleven Ecological Districts (ED; McEwen 1987) and all of the Mackenzie, Waitaki and Central Otago districts (Table 3.5). Most EDs showed an increase in percentage loss of remaining cover from 2001 to 2008 compared with 1990 to 2001, but in Central Otago and the Lakes District the rate of loss decreased (Table 3.6).

The greatest recent increase in loss of remaining indigenous grasslands was in Lowry where 0.7% was converted from 1990 to 2001, and71% from 2001 to 2008. Large increases in the rate were also seen in the Puketeraki, Canterbury Foothills, and Mackenzie EDs where the percentage loss of remaining grasslands per decade doubled.

Table 3.6 Percentage of remaining indigenous grasslands in each ecological district (McEwen 1987) found within the study area.

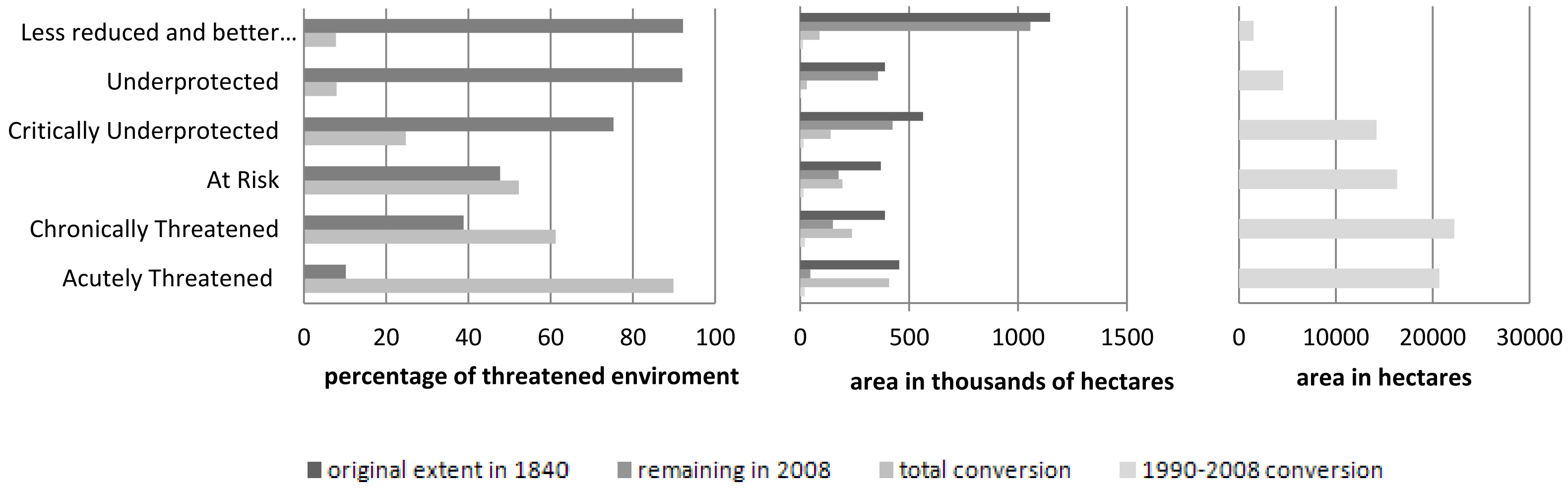
	% loss of			
Ecological Districts	**remaining 1990-2001**	**remaining per decade**	**remaining 2001-2008**	**remaining per decade**
Clarence	0.01	0.01	0.61	0.87
Lowry	0.69	0.62	71.46	100
Puketeraki	0.47	0.43	2.10	3.01
Canterbury Foothills	0.25	0.23	9.51	13.59
Heron	0.11	0.10	0.41	0.58
Tasman	0.01	0.01	0.05	0.08
Mackenzie	3.60	3.27	4.75	6.78
Pareora	0.16	0.15	0.20	0.28
Waitaki	0.12	0.11	0.49	0.71
Lakes	0.65	0.59	0.05	0.08
Central Otago	1.73	1.57	0.95	1.36
Lammerlaw	5.13	4.66	4.52	6.45
Mavora	0.02	0.02	0.19	0.27
Waikaia	0.01	0.01	0.05	0.07

Threatened Environments and protection status

A quarter of total indigenous grassland conversion from 1840 to 2008 has occurred in environments that are mapped in the threatened environment classification as having less than 30% of indigenous cover remaining (Figure 3.7). The largest total area of conversion (409 070 ha) was in ‘Acutely Threatened’ environments (those with <10% indigenous cover left; Walker et al. 2006). The second largest area (238 278 ha) was in ‘Chronically Threatened’ (10-20% of indigenous cover remaining), followed by ‘At Risk’ environments (those with 20-30% indigenous cover remaining).

Environments most prone to conversion were also the environments with the least remaining indigenous grasslands. By 2008, 10% of grasslands found in ‘Acutely Threatened’ environments remained, and 38% of those in ‘Chronically Threatened’ environments remained. Between 1990 and 2008, 20 706 ha were converted in ‘Acutely Threatened’ environments, followed by 22 234 ha in ‘Chronically Threatened’, and 16 377 ha of ‘At Risk’ environments. A very small proportion (1.4%) of conversion was found in the two environments with >30% remaining (Underprotected and Less Reduced and Better Protected).

Figure 3.7 Areas and percentages of grassland conversion, and remaining indigenous grassland within each 'Threatened Environment' category (Walker et al. 2006).



3.5 **Discussion**

Quantifying grassland conversion

A critical step in managing ecosystems is to take stock of their extent, condition, and capacity to continue to provide natural services. Our analysis quantifies the extent to which recent changes in land use activities have further reduced and fragmented indigenous grasslands, and how the pattern of land conversion has changed. We show that although large areas of indigenous grasslands remain in New Zealand, there continues to be on-going loss. About one third (34%) of indigenous grasslands have been converted in the South Island in the last 168 years.

Before 1990 conversion took place in lowland environments that were most suitable for production. Our study shows that in the last two decades, more non-arable land (as defined by Land Use Capability in the New Zealand Land Resource Inventory; Newsome et al. 2000) has been converted in environments characterized by mid to low slopes and elevation, summer droughts, extreme winter and summer temperatures, high winds, and limited annual rainfall. Many of these grasslands are on relatively infertile and/or porous erosion-prone soils with degraded vegetation cover (due to overgrazing by rabbits and livestock) (Hewitt 1998).

Although several national inventories of remaining indigenous grasslands have been completed in recent years, these have relied on national land cover databases (LCDB). Specifically, use of LCDB1 (Mark & McLennan 2005), or comparisons of LCDB1 and LCDB2 (Walker et al. 2006) have led to underestimates of grassland conversion. For example, comparisons of LCDB1 and LCDB2 suggest that between 1996 and 2001 there were 2 486 ha of change from tall tussock grasslands to a non-indigenous cover class, for the entire country (Walker et al. 2006). We found there to be twice as much as this, within our study area alone, by 2001. Furthermore, Mark and McLennan (2005) estimated that 77% and 82% of tussock grasslands remained in the McKenzie and Waitaki ecological districts, respectively, in 2002. Our data suggest about 7% more conversion had taken place in each of these districts by 2002.

Our findings confirm limitations in LCDB for detecting changes in grasslands as noted by Walker et al. (2006), and explained by difficulties associated with automatic detection of non-woody vegetation change (Dymond et al. 2006) which indicate that coarsely assigned land cover classes remain inadequate to assess biodiversity loss. Increasingly available higher resolution satellite data are permitting more accurate interpretations of land cover, but measurements of grassland conversion will remain varied unless: (1) a more universally accepted set of definitions of grasslands is established, and (2) there is greater consistency in methods used to determine boundaries between forests, and grasslands and agricultural land/permanent pasture and grasslands. We suggest improvements in estimates of grassland land cover and conversion, and better representation of the heterogeneity of grasslands types could be made by defining classes based on the structure and floristic composition of the vegetation rather than land uses (i.e. low producing grasslands). Ideally, grassland classes should be defined using robust field sampling to establish the biotic component of the class (Newsome 1987), and complemented by remote sensing technology that matches spectral signatures with each grassland class (Ferreira et al. 2003).

Types and rates of conversion

Our results suggest a trend towards increased production per hectare of land, within the South Island indigenous grasslands, and particularly towards more productive pasture. Two-thirds of the conversion we recorded between 1990 and 2008 was to exotic pasture. Methods of conversion usually involve over-sowing with legume species (mostly white clover *Trifolium repens*) and exotic grass forage species, often accompanied by installation of irrigation infrastructure and increased application of fertilizers to attain desired productivity levels. Within these same areas the rate of conversion (expressed as a proportion of remaining grasslands) has increased noticeable in the last decade. Between 1990 and 2008 the average rate of conversion across the study area increased by 1 267 hectares per year.

Although we did not address the causes in our study, it seems likely that the driving forces for the types and increase in rate of land conversion are linked to growing international demand for products of New Zealand’s high value more customized primary industries. In particular, land-based primary industries (i.e. dairying) have recently expanded and increased production nationally (Ministry of Agriculture and Forestry 2008). The largest growth in dairying has taken place in the South Island and has occurred largely through conversions of sheep-and-beef farms in response to low land prices, high per-cow productions, and in some cases access to irrigation. To enable estimates of the extent and location of future conversion, it may be helpful to identify the economic drivers of land conversion in the South Island more precisely, for example by modelling the economic structural process that underlies land-use changes (Veldkamp & Lambin 2001). Most case studies highlight the importance of policies in driving land use change (Lambin et al. 2001). The current spatial distribution of grassland conversion might also be better explained by modelling the underlying temporal dynamic processes and spatial interactions associated with economic agents (Irwin & Geoghegan 2001).

Incremental cumulative loss

While our study recorded loss of grasslands to intensive land use throughout the South Island, most of the change was found in the three administrative districts: Waitaki, Mackenzie. and Central Otago. The most noticeable increases in the rate of conversion were in the Waitaki and Mackenzie districts, where the rate of land conversion doubled in last decade. The majority of individual changes were incremental and less than 140 ha in size, yet over the long term their cumulative effect was significant, particularly when combined with the few larger developments, such as the 2 000 to 5 500 ha changes recorded in the Waitaki district.

In addition to loss of habitat for indigenous grassland species, an important cumulative effect of multiple incremental changes in land use and land management practices may be further fragmentation of the landscape, which in turn is linked to changes in the attributes of biodiversity (Bascompte & Sole 1996). Small scale conversions along with the building of roads, fences, power lines and other infrastructure provide opportunities for semi-natural vegetation to develop and form a network of corridors that facilitate dispersal of organisms, particularly invasive exotic species, throughout the landscape. These changes can

lead not only to increased weed and pest invasion of remaining indigenous grasslands, but also to modification of ecosystem processes including changes in the decomposition rates, and transference of nutrients and soil erosion (Wolters et al. 2000).
Our results suggest the scale of grassland conversion is such that the cumulative effects of land intensification on biodiversity loss and ecosystem services deserve greater attention in planning decisions under the Resource Management Act 1991 (RMA). At present, land clearance and other resource use decisions associated with grassland conversion are usually assessed on a case-by-case basis (Heitzmann 2007). There may be a need to complement such decisions, for example with regulatory limits that take into account the cumulative effects of land intensification on biodiversity loss and ecosystem services.

Need for increased protection

The current global extent of grasslands has been identified as having significance in relation to the future of global biodiversity and ecosystem services. More than one quarter (26%) of the terrestrial Eco regions selected as "outstanding examples of world's diverse ecosystems and priority targets for conservation action" by the World Wildlife Fund in 2000 (White et al. 2000) were grassland ecosystems. Recent research in New Zealand also indicates that indigenous grasslands play an important role in carbon sequestration (Mark &Dickinson, 2008).

Our study quantifies the extent of recent conversion in New Zealand's grasslands for the first time. Documentation of effects of recent conversion on biodiversity and ecosystem services requires further analysis and study. However, it seems likely that on-going conversion is contrary to New Zealand's Biodiversity Strategy state outcome that by 2020 there will be "A net gain in the extent and condition of natural habitats and ecosystems important for indigenous biodiversity" (Department of Conservation/Ministry for the Environment 2000). Furthermore, future land management policies should stand by New Zealand's commitments and responsibilities under the 1992 "Convention on Biological Diversity".

Although most of New Zealand's remaining indigenous grassland has been modified to varying degrees by the indirect or direct effect of human activity, they have continued to support a rich flora characterized by high species diversity (Duncan et al. 1997; Dickinson et al. 1998; Duncan et al. 2001; Walker et al. 2008; Mark et al. 2009). Recent grassland conversion is concentrated in environments that are poorly protected and with less than 30% of the total land environment remaining in indigenous cover. Although the total area extent of protection of indigenous grasslands has increased through the tenure review process in recent decades, most of the new conservation land is at high elevations, and the protection of these low-mid elevation remains inadequate (Walker et al 2008; Mark et al. 2009). These low to mid elevation highly modified ecosystems, support high numbers of the South Island's threatened plant species (Walker et al. 2008; de Lange et al. 2009). Our study identifies that grassland habitats that are most reduced and poorly protected in New Zealand are also those being most rapidly transformed by intensification. Perhaps the best remaining opportunities to protect these grassland habitats exist where the government land reform (tenure review) process continues.

Internationally, some of the largest changes in biodiversity have occurred and are expected to occur in grasslands, yet they continue to remain one of the least protected ecosystems (Hoekstra et al. 2005). There have been substantial improvements to reporting and analysis of grassland land conversion yet there remains a major and widespread disparity between habitat loss and protection. The lack of protection of New Zealand’s most threatened environments exemplifies this global trend. Though New Zealand has a much greater proportion of protected grasslands than most countries, there continues to be inadequate retention of representative indigenous biodiversity (Mark et al. 2009). Given the extent of the remaining indigenous grasslands there is ample opportunity for New Zealand to make a major contribution to the conservation of global grassland biodiversity.

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Appendix 3.1 Land cover classes used for the 1990 base map. Classes and descriptions adapted from Ecosat 1990 (Dymond & Shepherd 2004), and New Zealand Land Resource Information (NZLRI) (Newsome et al. 2000).

Land cover classes and descriptions		
Land cover	**Class**	**Description**
Forest	Indigenous forest	Tall or short forest (>30% cover)
	Planted forest	Radiata pine, Douglas Fir, eucalypts, or other planted species
		Roads/tracks within forest area
		Wilding pines
		Shelterbelts
Grassland	Exotic grassland	Grassland dominated by exotic species Exotic grassland with linear shelterbelts
	Indigenous grassland	Grasslands dominated with Festuca, Poa and Chinocloa species
		Tussock grasslands Alpine herbfields
		Grassland with woody species (<30% cover)
Cropland	Cropland-perennial	Orchards
		Vineyards
	Cropland-annual crops	All annual crops Cultivated bare ground
Settlements	Settlements	Built-up areas and impervious surfaces
		Grasslands with settlements including recreational areas
Bareground	Other land cover	Montane rock/scree Largely bare soil (if not cropland)
		Roads
		Open pit mines
		Any other remaining land
Water	Open water	Rivers, riverbeds, streams, ponds, natural lakes
		Man-made lakes and reservoirs
Shrubland	Native or exotic shrub	Broadleaved hardwood shrubland, manuka/kanuka shrubland, and other woody shrubland (>30% cover)
		Mategori and sweet brier
Wetland	Vegetated non-forest	Herbaceous and/or non-forest woody vegetation: periodically flooded Estuarine/tidal areas

Appendix 3.2 Threatened Environment categories and descriptions from Walker et al. (2005).

Category	Criteria	Category name
1	<10% indigenous cover left	Acutely Threatened
2	10–20% left	Chronically Threatened
3	20–30% left	At Risk
4	>30% left and <10% protected	Critically Underprotected
5	>30% left and 10–20% protected	Underprotected
6	>30% left and >20% protected	Less reduced and better protected

4 Estimating patterns of vulnerability in a changing landscape: New Zealand's indigenous grasslands[5]

4.1 Abstract

Effective conservation planning must anticipate the rates and patterns of dynamic threats to biodiversity, such as rapidly changing land-use trends. Poor understanding and prediction of the drivers and patterns of change has made it difficult to assess the relative vulnerability of areas of remaining indigenous habitat, and thus identify those in most immediate need of protection. Here we use quantitative spatial models to assess and predict the vulnerability of remaining indigenous grassland habitat in New Zealand to land use conversion. We used model validation techniques to measure the ability of vulnerability predictions based on past conversion patterns to predict modern conversion. Although the area of formally protected temperate grasslands has increased in recent decades, low to mid-altitude systems continue to be poorly protected, land-use intensification has accelerated in recent years, grassland vulnerability patters are changing rapidly. Indigenous grassland habitat that most vulnerable to conversion was at moderate to high elevations and have previously been classified as being of low suitability for intensive pastoral use. Models based on earlier conversion patterns performed more poorly in predicting modern conversion. Up-to-date land conversion data appear crucial for accurately predicting future conversion patterns and assessing vulnerability.

[5] Accepted as Weeks E.S., Overton J.Mc., Walker S., in *Environmental Conservation*

4.2 **Introduction**

Conversion of indigenous species habitat is one of the leading causes of global biodiversity loss (Houghton 1994; Ricketts and Imhoff 2003; Hoekstra *et al*. 2005; Reidsma *et al*. 2006), and continues worldwide despite conservation efforts (Sala et al. 2000; Mottet *et al*. 2006; Brown *et al*. 2007; Kangalawe 2010). Increasing global population and greater demand for food, fodder, fibre and fuel is leading to rapid changes in land use patterns, and areas once considered impervious to human activity are increasingly coming under threat (Parks 1995; de Koning *et al*. 1999; Rouget *et al*. 2003; Wilson *et al*. 2005a).

Assessment of the vulnerability of species and habitats to imminent proximate threats such as habitat conversion is a fundamental component of conservation management and planning (Margules and Pressey 2000; Wilson *et al.* 2005b). Spatial statistical models provide a tool for predicting where habitat conversion is most likely to take place (Hall *et al.* 1995; Pontius *et al.* 2001). However, models based on patterns of past habitat conversion will not necessarily provide reliable predictions of future vulnerability, because exhaustion of formerly suitable areas, and changes in global markets, technology and crops can alter both the distribution and rate of habitat conversion over time (Wilson 2005b; Pressey *et al.* 2007). The reliability of habitat-conversion vulnerability predictions is therefore likely to decrease as they are projected further into the future, risking misallocation of scarce conservation resources (Wilson *et al*. 2005b).

Because habitat conversion is a dynamic threat, it is important that practitioners keep abreast of change and regularly validate the utility of their vulnerability assumptions and models (Pressey *et al*. 2007). Validation requires testing the predictions of independent data (i.e. not those used in model parameterisation) to ensure that the relationships inferred by a model are robust and the predictions reliable. Yet absence of validation is a common weakness of habitat-conversion models (Pontius *et al*. 2004), and the robustness of vulnerability models used in conservation planning is seldom assessed (Wilson *et al.* 2005a; Pressey *et al.* 2007). Few studies have used a rigorous model validation procedure to quantify changing patterns of vulnerability to habitat conversion in a landscape of rapid land-use change.

This paper describes an assessment of the vulnerability of remaining areas of indigenous grasslands in New Zealand to conversion for intensive land uses (mainly intensive pasture production, but also exotic conifer plantations, urban use and mining). We modelled conversion of indigenous grasslands (hereafter 'grassland conversion') in relation to a range of potential environmental and socio-economic explanatory at different time periods, and tested and compared their performance in predicting recent (2001–2008) grassland conversion. Our results highlight changing patterns of grassland conversion in New Zealand, and have implications for informed prioritisation of vulnerable grasslands, including identification of remaining sources of uncertainty. We discuss the implications of our results for international conservation practitioners predicting vulnerability to future habitat conversion.

Study Area

Our study area covers approximately 4.2 million hectares between latitudes 41º and 46º south in the centre of the New Zealand's South Island, east of the Southern Alps (Figure 4.1). It includes the majority (3.3 million hectares) of New Zealand remaining indigenous grasslands, which span an elevation range from 10 to 2749 m above sea level, and experience average annual rainfall ranging from 285 mm at lower elevations to 7018 mm at the higher elevations.

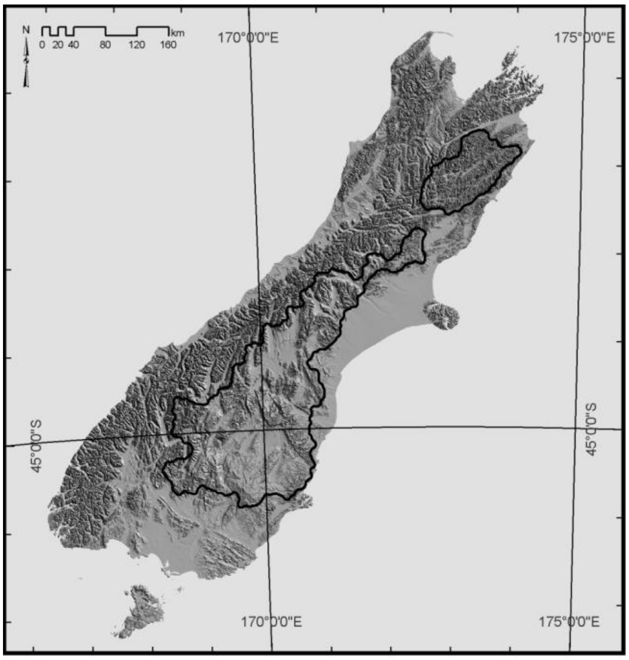

Figure 4.1 Location of study area in the interior South Island, New Zealand.

Indigenous grasslands in New Zealand are dominated by tussock growth forms (elsewhere known as "bunch grasses") (Levy 1951; Mark 1965; Ashdown and Lucas 1987; Mark 1993) of *Chionochloa, Poa,* and *Festuca* species. Unlike many other indigenous ecosystems in New Zealand, grasslands below the natural tree line have a unique, human-induced, origin, and were created in the last 800 years by Maori burning (Stevens et al. 1988; McGlone, 2001; Ewers et al. 2006) (Figure 4.1). The grasslands were further modified and reduced by European pastoral use (McGlone, 2001; Mark and McLennan 2005) after much of the inland South Island "high country" was acquired from Maori around 1840 (Brower 2008).

Today, although large areas of indigenous grassland habitat remain in New Zealand, there is ongoing loss (Figure 4.2). Mark and McLennan (2005) estimated that in 1840, 31% of New Zealand was covered by tussock grasslands dominated by endemic tussock grass species. They calculated that by 2002 just 44% of this area of indigenous grasslands remained, mostly in the interior areas of the South Island, and of this, 28% was legally protected, with a bias towards the high-alpine areas. Aussiel et al. (2012) estimated that by 2008 an additional 3% of remaining grasslands had been converted.

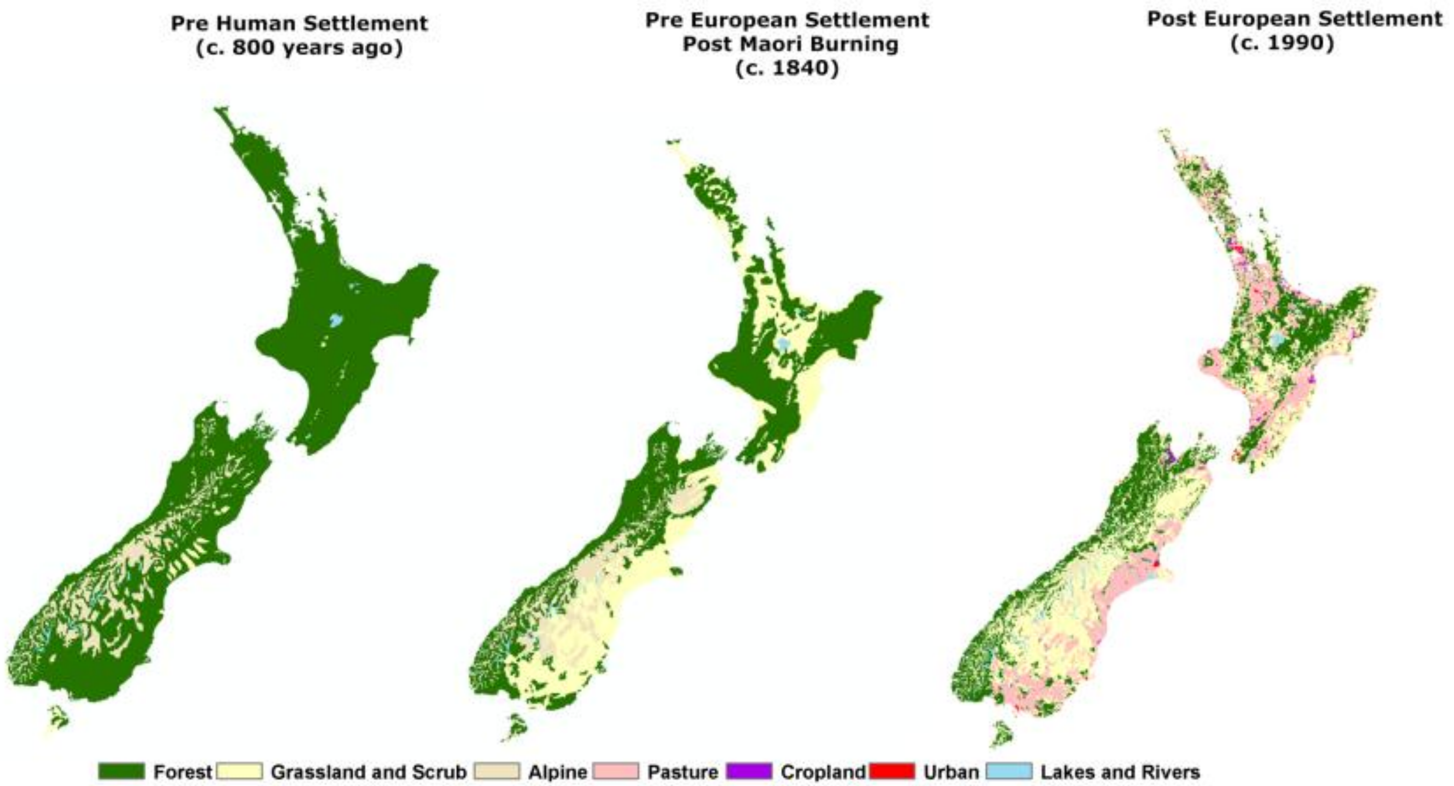

Figure 4.2 Land cover in New Zealand from pre-human (Maori) settlement (circa 800 years ago) (left) (McGlone 2001), and post-Maori and pre-European settlement (circa 1840) (middle) (McGlone 2001; Mark and McLennan 2005), to post-European settlement (circa 1990) (right) (Newsome1987).

Conversion of indigenous grasslands in New Zealand appears to be proceeding most rapidly at lower elevations, and on private land (Walker *et al*. 2008; Mark *et al*. 2009). Here, indigenous tussock grassland plant communities are most severely modified and invaded by exotic species, and are poorly protected (Mark and McLennan 2005; Mark *et al*. 2009), but nevertheless retain important residual indigenous biodiversity, including many threatened plants (de Lange *et al*. 2009; Walker *et al*. 2008) and endemic lizards and invertebrates (Patterson 1992; Patrick and Dugdale 2000).

Two trends may be elevating the vulnerability of this residual indigenous biodiversity. First, while extensive pastoral grazing enabled persistence of indigenous biodiversity in some places, there has been a recent trend to more intensive agricultural land use (cultivation and irrigation), most notably in response to international demand for dairy exports (Ministry of Agriculture and Forestry 2007). Second, in 1992 a voluntary land-reform programme (colloquially called 'tenure review') began splitting 2.4 million ha of Crown-owned pastoral leasehold land within our study area into private and conservation land parcels (Walker *et al*. 2008; Brower 2008; Mark *et al*. 2009). Walker *et al*. (2008) observed that land most vulnerable to habitat conversion and rich in threatened plant species is being privatized by land reform, while land at least risk of biodiversity loss is protected.

Until recently, quantification of vulnerability in remaining indigenous habitat grass has been hampered by poor national data on rates of grassland loss (Walker *et al*. 2006), and better data are urgently needed to guide conservation and development decision-making. This study is based on new mapping of grassland conversion and provides the first validated assessment of indigenous grassland vulnerability to conversion in New Zealand.

4.3 **Methods**

Data

Response variables and models

Three recent maps of grassland conversion served as data for conversion from 1840 to 1990 (pre-1990), from 1990 to 2001, and from 2001 to 2008 (Figure 4.3).

Each 25-m-grid cell in a map represented either grassland that had not been converted or grassland that had been converted for forestry, agriculture (cropland or exotic pasture), or urban development. These maps were used to generate three different binary response variables (conversion or no conversion from 1840 to 1990, from 1990 to 2001, and from 1990 to 2008), and one continuous response variable (area of conversion from 1990 to 2008). Figure 4.3 shows the progressive expansion of grassland conversion within the study area.

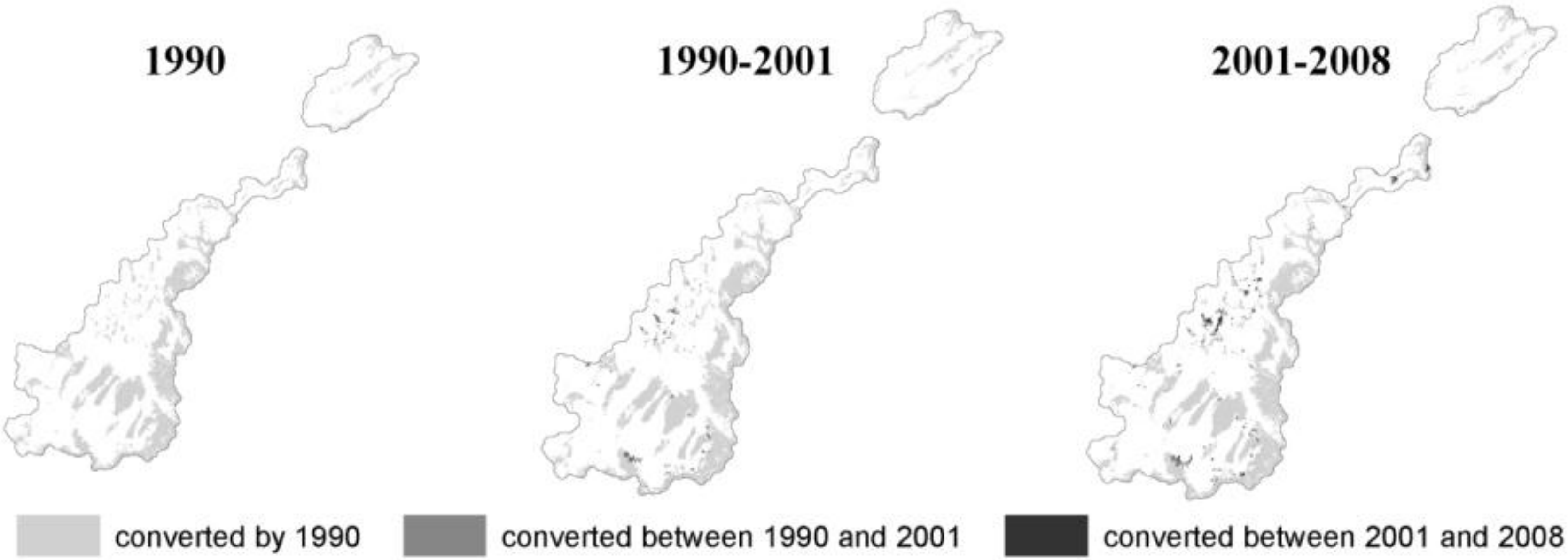

Figure 4.3 Progressive expansion of grassland conversion in our study area..

We created four different models. Our first two models predicted 'past' (1840–1990) and 'recent' (1990–2001) probabilities of conversion. Our third model was based on both recent and 'current' (2001–2008) patterns of conversion, and is used to predict future (post-2008) probability of conversion. Our fourth model used patterns of conversion across the study area from 1990 to 2008 to predict the area covered by an individual conversion event in future.

Explanatory variables

As potential predictors of conversion, we collated a comprehensive set of the environmental and socio-economic variables currently available in New Zealand (Table 4.1). As environmental predictors, we used 11 climate, substrate and landform variables developed for the Land Environments New Zealand (LENZ) database (Leathwick *et al*. 2003). We compiled 12 socio-economic predictor variables from a variety of sources, which we categorize as governance, land tenure, infrastructure or productivity.

As proxies for governance we used local government administrative regions and water catchments derived from topographic maps. We used seven categories of land tenure derived from maps supplied by the New Zealand Department of Conservation. Four categories represent different designations on former Crown pastoral lease land that completed land reform by 2008 (conservation land, conservation land with a grazing licence, unencumbered privatised land, and private land encumbered by a conservation easement (colloquially a 'covenant'). Three further categories represent remaining Crown pastoral leases, public conservation land, and private land respectively. We represented the spatial interactions of land-use decisions by calculating the maximum neighbourhood of a 2-km distance from land converted to agriculture before 1990.

Infrastructure variables were derived from national digital topographic databases of range of infrastructure (e.g. roads, power lines, and irrigation). We created 25-m raster layers of the log of distance to each type of infrastructure. We used two different layers as indices of agricultural productivity. Land Use Capability (LUC) was retrieved from the New Zealand Land Resource Inventory data layers (SCRCC 1971). LUC classifies land areas according to their capability to sustain continuous production into categories ranging from class ‘1’: “land with virtually no limitations for arable use and suitable for cultivated crops, pasture or forestry", to class '8': "land with very severe to extreme limitations or hazards that make it unsuitable for cropping, pasture, or forest" (Newsome et al. 2000). For pasture productivity we used the index created by Baisden (2006), which has values ranging from 41 for low to 2038 for high productivity.

Table 4.1 Explanatory (predictor) variables used for the models.

Name of variable	Abbreviation	Definition	Units	Category	Source
Mean annual temperature	mat	Mean annual temperature	C	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Mean annual solar radiation	mas	Mean annual solar radiation	$MJ/m^2/d$	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Evapo-transpiration	r2pet	Ratio to the annual potential evapo-transpiration	ratio	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Vapour pressure deficit	vpd	The annual vapour pressure deficit	kPa	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Annual water deficit	deficit	The annual water deficit	mm	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Rainfall	rain	Mean annual rainfall	mm	Climate	Land Environments of New Zealand(Leathwick et al. 2003)
Substrate age	age	Estimated age class of substrate	class	Substrate	Land Environments of New Zealand(Leathwick et al. 2003)
Soil calcium	calcium	Estimated class of soil calcium	class	Substrate	Land Environments of New Zealand(Leathwick et al. 2003)
Acid soluble phosphorous	acidp	Estimated class of acid soluble phosphorous	class	Substrate	Land Environments of New Zealand(Leathwick et al. 2003)
Elevation	elevation	Elevation above sea level	m	Landform	New Zealand Digital Elevation Model (Barringer et al. 2002)
Slope	slope	Slope estimated from DEM	degrees	Landform	New Zealand Digital Elevation Model (Barringer et al. 2002)
Catchment	catchgroup	River catchment	class	Governance	NZMS260 series topographic map (Land Information New Zealand)
Regional Council	region	Regional council, where 1 = Otago, 2 = Southland, 3 = Marlborough , 4 = Canterbury	class	Governance	NZMS260 series topographic map (Land Information New Zealand)
Land tenure	land tenure	Land tenure based on seven categories: 1. Former Crown pastoral lease (FCPL) conservation land, 2. FCPL conservation with grazing licence, 3. FCPL privatised, 4. FCPL private covenant, 5. Other conservation land, 6. Other private land, 7. Current Crown pastoral lease.	class	Land tenure	Department of Conservation
Distance to water	water	The distance of each pixel to a pixel of water	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Distance to roads	roads	The distance of each pixel to a pixel of roads	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Distance to irrigators	irrigators	The distance of each pixel to a pixel of irrigators	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Distance to roads	roads	The distance of each pixel to a pixel of roads	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Distance to towns	towns	The distance of each pixel to a pixel of towns	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Distance to power	power	The distance of each pixel to a pixel of power lines	m	Infrastructure	NZMS260 series topographic map (Land Information New Zealand)
Proximity to agriculture	pad	The proportion of pixels that are within 2km of land cleared for agriculture by 1990	m	Infrastructure	N/A
Land use capabilities	luc	Land are classified according to their capability to sustain continuous production, where class 1 has the highest capability, and class 8 the lowest.	class	Productivity	NZ Land Resource Information System (Newsome 2000)
Pasture productivity index	pastprod	Net primary productivity	$g\ m^{-2}$	Productivity	Pasture Productivity Index (Baisden 2006)

Sampling design

Each model used a stratified random sampling design with two strata: ‘converted’ and ‘not converted’ A total of 5,000 point observations were allocated to the calibration data set generated across the study area in GIS, and information extracted for each explanatory variable at each point. Based on the ratio of the area ‘converted’ to ‘not converted’ we chose to have 1,000 points randomly distributed in the ‘converted’ stratum and 4,000 in ‘not converted’. For the post-1990 models we masked out areas that had changed before 1990. We used the same sampling scheme for the three probability-of-conversion models and the area model. In the area model, a binary response variable (0 = converted and 1 = not converted) was replaced by the area of a converted polygon of grassland.

Identification of important explanatory variables

We identified important explanatory variables using Generalized Additive Models (GAMs) to fit relationships between two different types of dependent (response) variables (presence or absence of conversion, and area of conversion) and our potential explanatory variables. We used Generalized Regression Analysis and Spatial Prediction (GRASP) set of functions (Lehmann *et al*. 2002) in S-PLUS software (MathSoft 1997) for these models.

Probability of conversion was modelled as a binomial variable, and area of conversion as a Poisson variable. In all models, a starting model including all continuous and categorical predictors smoothed with 3 degrees of freedom was fitted first and significant predictor variables selected thereafter by backward and forwards stepwise procedure using the Bayesian Information Criterion (BIC) for variable selection. The chosen predictors for each final model were used to map predictions in geographic space.

To interpret the regression models, we used plots of the partial response curves that resulted from the model, and the overall contribution of the variables to the model. Partial response curves allowed visualization of how the response variable varies as a function of the predictor variables, while the contributions allowed us to assess the relative importance of the predictor variables in explaining the variation in the response variable.

Model predictions were also exported to GIS as lookup files and mapped as raster surfaces. Cell values in probability of conversion maps show likelihood that the cell was converted independent of other cells, while cell values in the area of conversion map show predicted area of conversion (in hectares) in that cell. We made a uniform scaling adjustment to probability predictions (multiplying by $\frac{\overline{x}\,oberved}{\overline{x}predicted}$) so that the mean probability was similar to the observed probability of conversion.

Model calibration and validation design

Calibration makes a model as consistent as possible with the data set from which the parameters are estimated (Pontius 2004). In contrast, validation uses statistical techniques to determine acceptable levels and costs of Type I and Type II errors (Mayer 1993). Separating calibration process from validation processes assures that the model is not over-fitted.

In our first two models we used one set of data to calibrate the model and a second set to validate the model. We modelled conversion from 1840 to 1990 and from 1990 to 2001, and used fitted parameters from these models to predict conversion from 2001 to 2008. The validation process compared predicted probabilities of conversion from these two models against reference maps of observed 'current' conversion (i.e. conversion from 2001 to 2008).

In absence of post-2008 conversion data, we calibrated our third (‘probability of conversion’) and fourth (‘area of conversion’) models using a single set of data, and validated them using correlations between predicted and observed conversion in the same time period.

Model validation

For validation of the pre-1990 and 1990 –2001 models we first used Receiver Operating Characteristic (ROC) curves (Swets 1988; Pontius and Schneider 2001; Pontius and Batchu 2003) to evaluate model discrimination between ‘converted’ and ‘not converted’. The true positive rate (sensitivity) was plotted as a function of the false positives (1-specificity) for different cut-off points. The area under the ROC curve measures how well the model can distinguish between ‘converted’ and ‘not converted’ and is calculated using integral calculus trapezoidal rule (an area of 1 represents a perfect model).

Second, we assessed the accuracy of each (i.e. pre-1990 and 1990 – 2001) model using interactive dot diagrams, which separate ‘not converted’ (0) and ‘converted’ (1) pixels on the horizontal axis and display probabilities of conversion from the models on the vertical axis. Thresholds identified in these dot diagrams indicate cut-off (threshold) points of the best separation (minimal false negatives and false positives) between ‘converted’ and ‘not converted’ (Schoonjans *et al*. 1995).

Our models ‘probability of conversion’ and ‘area of conversion’ from 1990 – 2008 were validated by plotting observed values against the values predicted by the model, and also by cross-validation. The ROC test (Fielding and Bell 1997) was used to test how well the 1990–2008 probability of conversion model distinguished between ‘converted’ and ‘not converted’. Goodness of fit in the area of conversion model was assessed by correlation of predicted values with observed values.

Vulnerability comparison

Finally, we also compared predicted vulnerability based on the pre-1990 and 1990–2008 conversions. Each map of vulnerability was scaled to span a range of 0 to 1 (by dividing by maximum probability), and the difference between the two maps calculated and mapped (scaled vulnerability based on 1990–2008 data minus scaled vulnerability based on pre-1990 data). A high positive difference would indicate that pre-1990 vulnerability underestimated future vulnerability, while more negative values indicate overestimates. We sampled the difference map at 5,000 random points across the study area, and modelled difference in relation to our suite of predictor variables using a GAM (Table 4.1).

4.4 **Results**

Model performance

Our cross validations returned ROC values of 0.913, 0.916, and 0.921 for our pre-1990, 1990 –2001 and 1990 – 2008 conversion probability models, respectively, indicating very good accuracy and stability. The correlation between observed and predicted conversion was highest in the 1990–2001 model (Spearman correlation coefficient $r = 0.708$), followed by the 1990–2008 model ($r = 0.693$), and the pre-1990 model ($r = 0.666$). Correlation between observed and actual area of conversion was low ($r = 0.467$), suggesting this model explained less than a quarter of observed variation in conversion area ($R^2 = 0.22$).

Table 4.2 Contribution (rounded to the nearest whole number) of selected predictors in the pre-1990, 1990 –2001, 1990 –2008 and area model. The drop contributions indicate the marginal contribution of each variable and are obtained by dropping each explanatory variable and calculating the associated change in deviance. The alone contributions reflect the potential of each variable. They are calculated by creating new models with only one predictor.

Predictors	**Pre-1990**		**1990 to 2001**		**1990 to 2008**		**Area**		**Difference**	
	Drop	Alone	Drop	Alone	Drop	Alone	Drop	Alone	Drop	Alone
Mean annual temperature	551	1550	123	1001	17	973	5940	7210	2	27
Rainfall to evapo-transpiration	19	510	121	1305	-	-	-	-	-	-
Annual water deficit	25	500	74	751	-	-	-	-	10	66
Rainfall	25	-	151	1251	143	1366	8321	4550	-	-
Elevation	-	-	205	1750	-	-	-	-	5	49
Slope	173	700	651	2021	540	1962	14053	8251	27	102
Pasture productivity	-	-	-	-	-	-	6100	4856	-	-
Catchment	251	451	-	-	-	-	-	-	-	-
Regional council	-	-	-	-	58	392	-	-	8	22
Land tenure	-	-	125	1152	143	1195	9532	6102	-	-
Distance to roads	-	-	98	1456	97	1433	-	-	11	65
Proximity to agriculture	-	-	51	603	59	660	13062	14005	-	-
Land Use Capability	-	-	-	-	-	-	-	-	11	61

Important explanatory variables

The ‘alone’ contributions of variables (the potential for each variable alone to explain conversion) differed among our three probability of conversion models (Table 4.2). Mean annual temperature played a dominant role in the pre-1990 model. It was the best ‘alone’ predictor of conversion, followed by slope, rainfall, catchment group, vapour pressure deficit, and annual water deficit. When dropping each predictor from the final model, mean annual temperature was the only variable whose contribution could not be compensated for by any of the other variables.

Partial response curves for selected predictors in each model are collated in Appendix 1. They show that pre-1990 conversion was positively related to mean annual temperature and negatively related to slope, soil moisture deficit, and vapour pressure deficit; and to rainfall <1000 mm. A wider selection of variables explained significant variation in conversion after 1990 (i.e. in the 1990–2001 and 1990–2008 models), and the variables were differently ranked in their ability to explain observed variation in conversion ('alone' contributions in Table 2). After 1990, elevation became a significant predictor of conversion in addition to temperature, slope and water balance, and significant differentiation between types of land tenure and effects of distance to roads and proximity to existing agricultural development also became apparent.

Slope was the dominant variable in both the 1990–2001 and 1990–2008 models. Between 1990 and 2001, slope was the best ‘alone’ predictor of conversion, followed by elevation, distance to roads , rain, ratio of rainfall to potential evapo-transpiration, land tenure, mean annual temperature, annual water deficit, and proximity to existing agricultural development (Table 2). In the 1990–2008 models, the ranking was first slope, then rainfall, land tenure, distance to roads, proximity to existing agriculture, administrative region, and mean annual temperature.

Conversion from 1990 to 2001 was positively related to mean annual temperature and elevation, and negatively related to slope, rainfall, soil moisture deficit, and distance to roads (Appendix 4.1a). Probability of conversion peaked at intermediate proximity to existing agriculture (Appendix 4.1b). Among land tenure categories, the probability of grassland conversion between 1990 and 2001 was highest on private land (category 6 in Appendix 4.1b), followed by former Crown pastoral lease land was privatised by 2008 (category 3) and then by existing Crown pastoral lease land (category 7). A small portion of conversion was also predicted on former Crown pastoral lease land that was privatised and covenanted between 1992 and 2008 (category 4); this reflects our input data showing where a ski field had been developed and wilding conifers spread (Appendix 4.1b). Probability of grassland conversion on recent (categories 1 and 2) and pre-existing public conservation land (category 6) was negligible.

Probability of conversion in the 18 years from 1990 to 2008 was also negatively related to slope, rainfall, and distance to roads, and positively related to mean annual temperature, and showed the same peak at intermediate proximity to existing agricultural development as in the 1990–2001 period (Appendix 4.1c). Probability of conversion was also higher within two regions (Canterbury and Otago). As with the 1990–2001 model, conversion probability was highest on private land from 1990 to 2008, followed by recently privatised former Crown pastoral leased land and then existing Crown pastoral lease land, and was negligible on public conservation land.

Proximity to existing agricultural development was the highest ranked predictor in our model of area of conversion from 1990 to 2008 ('alone' contribution in Table 2), followed by slope, land tenure, October vapour pressure deficit, annual rainfall, and pasture productivity. Larger areas of grassland were converted in places further from existing agricultural development (low pad; Appendix 4.1d), and on flat and highly productive land (although large areas were also converted on some steep land). Area showed a similar response to land tenure as it did to conversion probability, with the smallest areas on public conservation land (category 6) and the largest on private land (category 7). Conversion on conservation and private land created by recent land reform (categories 1 and 3) also tended to cover relatively large areas.

Model validation

A comparison of the area under the ROC curves for our pre-1990, 1990–2001 and 1990–2008 conversion probability models are displayed in Figure 4.4. In the pre-1990 model a randomly selected sample from ‘converted’ has a probability-of-conversion value larger than that for a randomly selected sample from the ‘not converted’ group 83% of the time, while in comparison, for the 1990 to 2001 model, it was 93% of the time. Although both models’ performance was significantly ($p< 0.0001$) different from 0.5 (i.e. both models had the ability to distinguish ‘converted’ from ‘not converted’ between 2001 and 2008), the 1990–2001 model provided a better estimate of recent conversion. Interactive dot diagrams which display the accuracy of the two models (Appendix 4.2) similarly showed the 1990–2001 models had higher sensitivity (93%) and specificity (80%) than the pre-1990 model (86 and 71%, respectively).

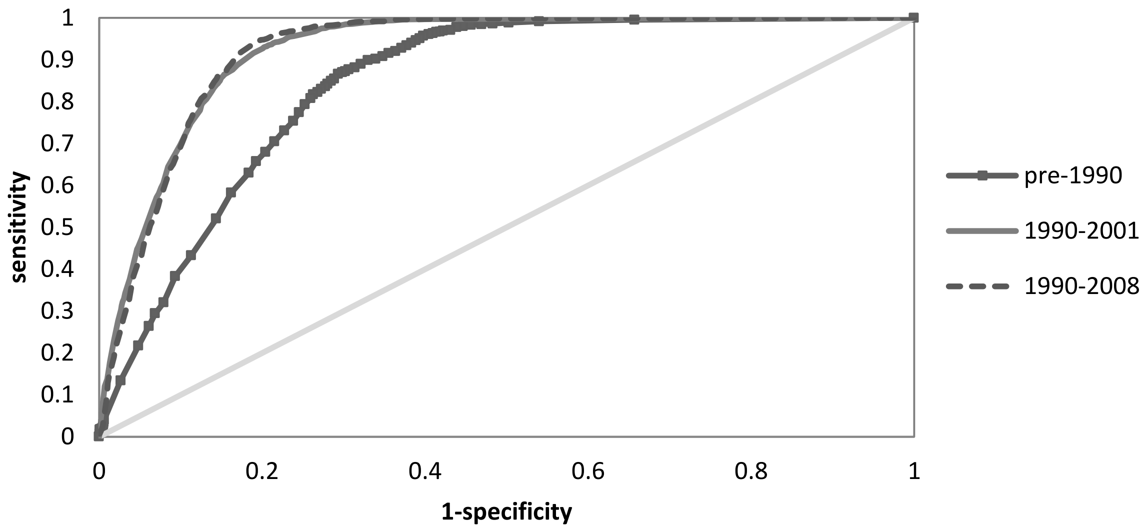

Figure 4.4 ROC curves used to validate predicted probabilities of grassland conversion for each model: pre-1990 conversion (lower curve, ROC (area under the curve) =0.841), 1990–2001 (middle ROC=0.913), and 1990–2008 (top ROC=0.921). The pre-1990 and 1990–2008 model was compared to the observed conversion between 2001 and 2008 and the 1990–2008 model validation represents the relationship between fitted and observed data. Each curve plots sensitivity (proportion of positives correctly identified, or ‘true positives’) against 1–specificity (the complement of proportion of negatives correctly identified, or the false-positive rate). A ROC value of 1 indicates perfect sensitivity and specificity, and the diagonal line represents a model providing no discrimination.

Predicted vulnerability

Figure 4.5 shows the vulnerability of remaining indigenous grasslands in the study area, as predicted by our different models. Probability of conversion is predicted for 150 years (Figure 4.5a), 11 years (Figure 4.5b), and 18 years (Figure 4.5c) and area of conversion for 18 years (Figure 4.5d) but legends are scaled so that spatial patterns of vulnerability can be directly compared.

In all three probability-of-conversion models, the predicted probability of conversion of indigenous grasslands that remain in 2008 is lowest in steep, mountainous land and higher on inland basin floors and lower range slopes (Figures 4.5a, b, c). There are differences in the amount of land predicted to be highly vulnerable to conversion, however, with the least land predicted to be vulnerable by the pre-1990 model (Figure 4.5a), and the greatest area of land of high vulnerability to grassland conversion predicted by the 1990–2008 model (Figure 4.5c).

The spatial distribution of sites predicted to be most vulnerable varied between the three probability-of-conversion models, but most strikingly between the pre-1990 model and the two models based on conversion after 1990 (Figures 4.5b, c). Comparison with the 1990–2008 model showed that predictions based on pre-1990 data underestimated indigenous grassland vulnerability on gentle slopes (<10°) and at elevations 500 and 1100 m, and overestimated vulnerability at elevations above and below this range and on steeper slopes ('Difference' columns in Table 4.2). Sites predicted to be more vulnerable after 1990 also had lower mean annual temperatures, were in land-use capability classes with lower capability for cropping or pasture, were closer to roads, and were more likely to be within the Canterbury region.

Predicted area of conversion from our fourth model ranged from 4 ha to 2147 ha (Figure 4.5d). Large areas of conversion were predicted at lower elevations with low slopes, and small areas (4–116 ha) of conversion were predicted at higher elevations. In general, areas with the highest probability of conversion predicted by the 1990–2008 model of vulnerability were also the places predicted to have the largest areas of conversion. However, some places where large areas of conversion were predicted at higher elevations (Figures 4.5d) had low probabilities of conversion (i.e. low vulnerability; Figures 4.5c). These places appear to represent the predicted locations of large conifer plantations or wilding conifer spread, rather than agricultural conversion.

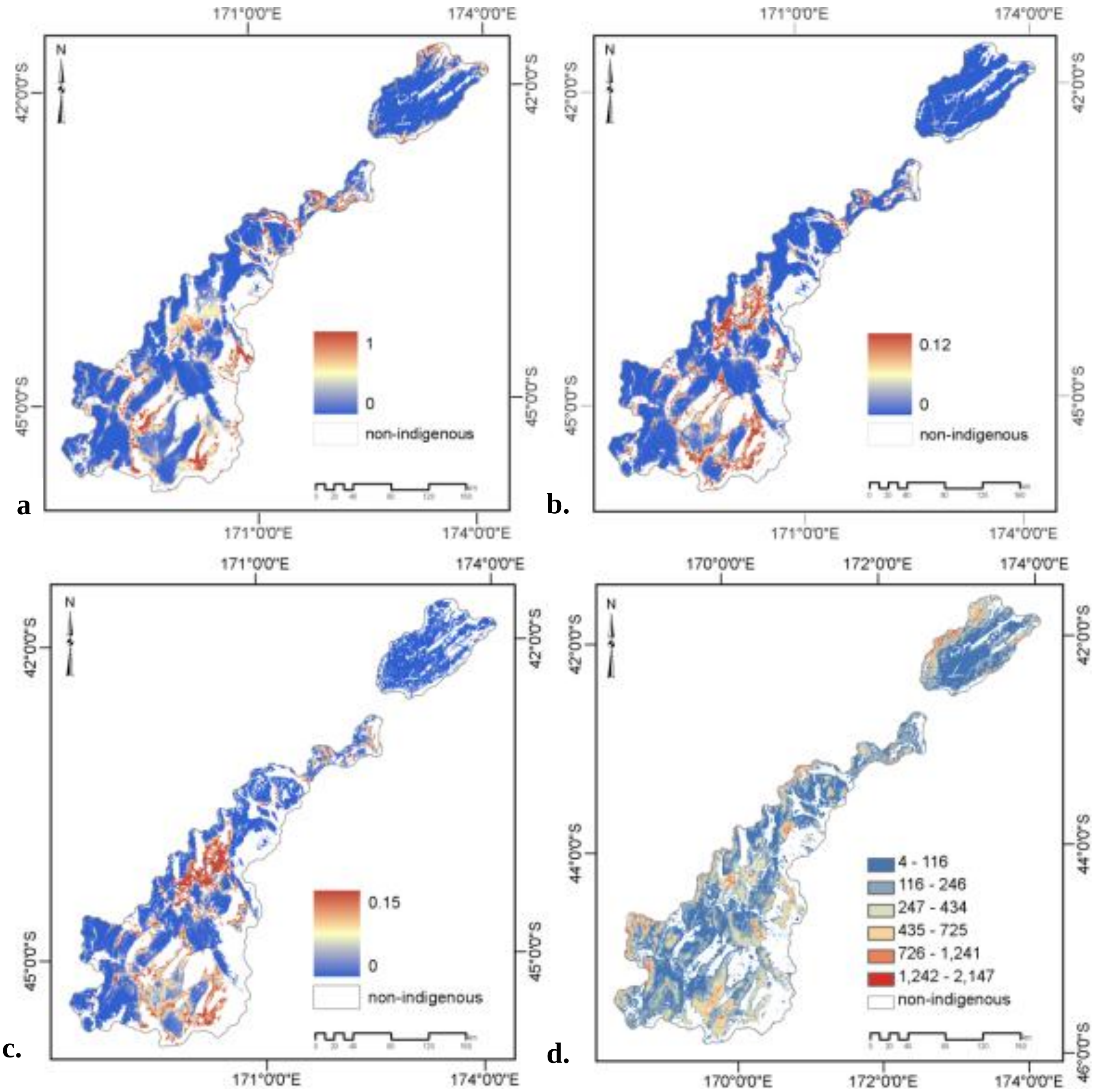

Figure 4.5 Vulnerability of remaining indigenous grasslands to future conversion predicted by four different models. Probability of conversion (a, b and c) was predicted based on observed patterns of conversion (a) pre-1990, (b) from 1990 to 2001, (c) and 1990 to 2008. Area of conversion (d) is based on a model of observed conversion from 1990 to 2008. Areas of conversion prior to 1990, and areas that are not indigenous grasslands are labelled 'non-indigenous' (white).

4.5 Discussion

Implications of validated vulnerability models

Our results measure the extent to which observations of past conversion can be used to predict future patterns of land conversion and highlight the importance of using a robust validation process in estimating vulnerability (i.e. likelihood of future loss) in conservation planning.

A common weakness of land conversion modelling is the use of the same data for both calibration (making the model as consistent as possible with the data from which the parameters were estimated) and validation (assessment of the predictive power of the model) (Pontius *et al*. 2004). Lack of consideration of model uncertainty through rigorous validation has been shown to result in inaccurate and over-confident predictions. Here, we calibrated our model using one set of data and compared the results to another set, to determine how well the model used the general pattern in the calibration data to extrapolate a pattern of future conversion. Such model validation is a critical aspect that is often overlooked in predictions of vulnerability in conservation planning (Wilson *et al*. 2005b). Vulnerability assessments based on past observations assume that past patterns of conversion will remain the same in the future. However, it is well accepted that factors that are important for explaining patterns of conversion for a past period may not necessarily predict future landscape changes (Wilson et al. 2005a). Here, we tested the ability of models of past grassland conversion to predict conversion in a future time period using rigorous statistical validation procedures. In doing so, we demonstrated how rapidly patterns of vulnerability may change over time, using the indigenous grassland landscape of New Zealand as a test case.

By comparing patterns of conversion over time, we found that conversion patterns show a trend in New Zealand, and consequently that historical (pre-1990) patterns of conversion inaccurately predict current (2001–2008) and future (post-2008) vulnerability. Spatial patterns of conversion have also changed, with sites most vulnerable to conversion before 1990 being different to those most vulnerable between 1990 and 2001 and different yet again to those most vulnerable between 2001 and 2008. Indigenous grasslands have been increasingly converted on gentle slopes at higher elevations, at sites with less extreme annual moisture deficits but lower mean annual temperatures, and lower inherent capability for sustained production.

Clearly, more recent conversion data will provide the best estimates of future conversion. We caution, however, that vulnerability estimates based on too narrow a time range may also provide less accurate forecasts, because they are based on a small sample of conversion events. We therefore suggest that our composite model of conversion from 1990 to 2008 will probably provide more accurate forecasts of grassland vulnerability to future conversion than models based on the 1990–2001 or 2001–2008 periods.

Past and present vulnerability to conversion in New Zealand indigenous grasslands

Indigenous grasslands least vulnerable to conversion, both at present and in the past, are those on steep slopes, at high elevation, and that experience the lowest temperatures and highest rainfall. The distribution of more vulnerable grasslands has changed considerably, however. Historically (until 1990) in New Zealand's South Island grasslands, the lowest, warmest, flattest, and driest land was converted most rapidly for production. These places were also among the driest in our study area, with relatively low rainfall, high annual water deficits and high October vapour pressure deficits. The current vulnerability of New Zealand's remaining indigenous grasslands is also predicted by variables that determine the suitability of sites for intensification such as gentle slope and high mean annual temperature. However, the grasslands now most vulnerable to conversion are at moderate rather than low elevations and on more marginal land than those most vulnerable before 1990. Proximity to roads and to existing agricultural development, and land tenure have also became important predictors of grassland conversion.

The greatest increases in vulnerability have been seen on land above 500 m that is classified as suitable mainly for low productivity extensive grazing, and in one administrative region in particular. The transition in grassland vulnerability in New Zealand from the warmest, lowest, and driest environments to higher, cooler and more marginal land fits the 'maximum power principle', that people will use the most economically productive land first (Odum 1983; Hall *et al*. 1986). Our results reflect that in New Zealand, the most economically viable land is becoming less available, while demand is rapidly increasing due to an increased demand for dairy exports (Ministry of Agriculture and Forestry 2007).

Although we could predict conversion probability relatively accurately, our model predicting area of conversion was weak. Nevertheless, our models did show that since 1990 the proximity of existing agricultural activity influenced patterns of the size of conversion. Small-scale incremental conversion is taking place near existing agriculture activity while large areas of conversion are further away. The relatively poor predictive ability of our area model may be because environment is less important than some socioeconomic drivers (such as land cost) that may be poorly represented in our models. Another complicating factor may be divergent predictors of large agricultural developments and large conifer plantations and/or areas of wilding conifers.

Ultimate and proximate threats in modelling vulnerability

Uncertainties caused by changes in ultimate threats underlie the limitations of models of vulnerability based on proximate threats (Pressey *et al*. 2007). Because land-use systems respond to a combination of proximate (biophysical) and ultimate (socio-economic) drivers, modelling vulnerability to conversion (intensification) ideally requires a multidisciplinary approach (Veldkamp and Fresco 1996; Lambin *et al*. 2001; Rounsevell *et al*. 2006). Our models attempt to incorporate some dimensions of the socio-economic environment as well as the biophysical environment. However, incorporating socio-economic data was challenging. Because these data are mainly held as aggregated national datasets in New Zealand, they do not represent patterns at regional and local levels. Furthermore, few are readily translated into spatial layers. In our models we therefore used proxy variables such as administrative districts, distances to infrastructure, and land tenure to represent dimensions of the socio-economic environment such as governance, social contagion, and law. These variables proved useful (and indeed necessary) for accurate predictions of conversion, especially after 1990, but did not allow us to explore causality (Veldkamp and Lambin 2001). Therefore explanations for the correlations we observed need to be further explored, and models adopting different approaches to our biophysical models are likely to be needed. For example, models using system, actor-based and narrative approaches to incorporate endogenous variables (e.g., macroeconomic, land management technology, infrastructure and land use policy changes) often highlight the important role of economic opportunities and policies in driving land conversion (Lambin *et al*. 2001).

Our incorporation of socio-economic proxy variables into biophysical models did, however, provide some insights into potential higher-level drivers of grassland conversion. For example, we showed that grasslands on privately owned land, and on land that was formerly Crown pastoral lease but was privatised through land reform ('tenure review') between 1992 and 2008, had a high probability of conversion and is currently highly vulnerable. Current Crown pastoral lease land was also vulnerable, but less than private and recently privatised land. This is consistent with privatisation through land reform increasing the vulnerability of remaining indigenous grassland habitats, as predicted by Walker *et al*. (2008). Our results also highlight that public conservation land appears to have provided complete protection against grassland conversion, except in a few cases where wilding conifer spread was already advanced when the land was acquired for conservation through land reform. Our models also clearly showed an increase, after 1990, in intensive agricultural development on land previously considered suitable only for extensive grazing. This trend suggests a recent increase in the economic viability of irrigation on marginal land as a second potential higher-level driver.

Although spatial regression models such as ours are useful for predicting the vulnerability of grasslands to conversion because of their robustness, we caution they do not account for temporal heterogeneity. Land-use decisions are often triggered by single events such as economic fluctuations or crises, often remote in space and time, which operate at a higher hierarchical level (Houghton 1994). Where temporal heterogeneity is high, process-based models or models using economic frameworks might be more appropriate, and yield better representations of the decision making process. However, for systematic conservation planning (Margules and Pressey 2000; Margules *et al*. 2002; Wilson *et al*. 2007) the two different types of model (process based and spatial regression) may complement each other. The strength of spatial regression models is to effectively identify the areas vulnerable to threatening processes such as conversion, which could be targeted for increased protection, while process-based models foster understanding of the drivers of these changes, which could potentially be addressed through socio-economic policy and instruments.

Incorporating vulnerability into conservation prioritisation

Absence of robust vulnerability assessment such as this study provides has likely hampered protection of New Zealand's more vulnerable indigenous grasslands. However, a measure of vulnerability to conversion alone is insufficient for identifying priority areas for conservation, which must take into account not only vulnerability, but also relative conservation value such as irreplaceability or significance. In applying the results of this study to conservation planning, it would also be useful to consider not only vulnerability to conversion, but also other types of vulnerability, (such as to invasive species and climate change) and the effectiveness and costs of different management approaches and activities (Carwardine *et al*. 2009).

4.6 **Conclusion**

A variety of methods have been developed to model vulnerability to conversion. These models vary in their complexity and applicability to conservation management. Generalized additive models provide a simple robust method to explore predictors and patterns of land-use change. Our analysis demonstrates that they can also be feasibly used to predict future patterns of conversion using recent land conversion data.

Our results provide the first data-derived and statistically validated measurement of the vulnerability of New Zealand's indigenous grasslands to conversion, and show a trend to greater agricultural conversion on higher, more marginal land. They also show that models based on earlier conversion patterns performed more poorly in predicting modern conversion. Up-to-date land conversion data therefore appear crucial for accurately predicting future vulnerability to habitat conversion.

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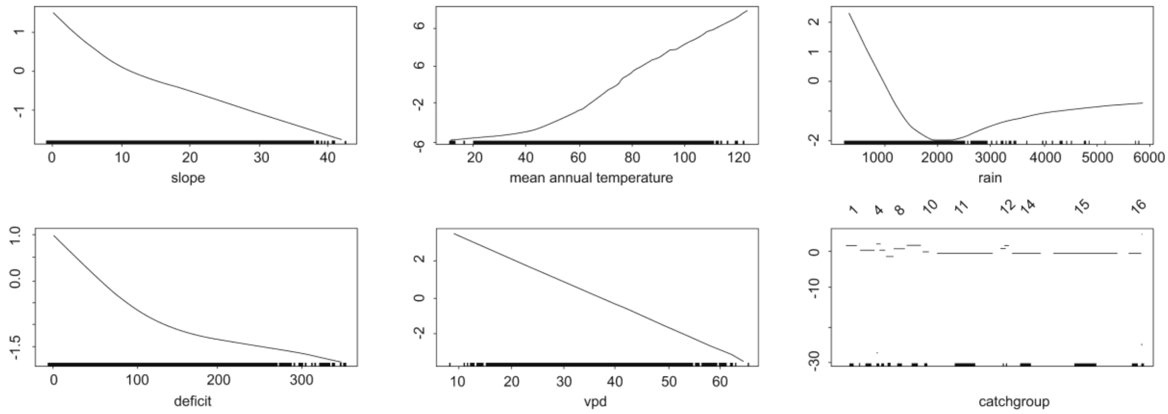
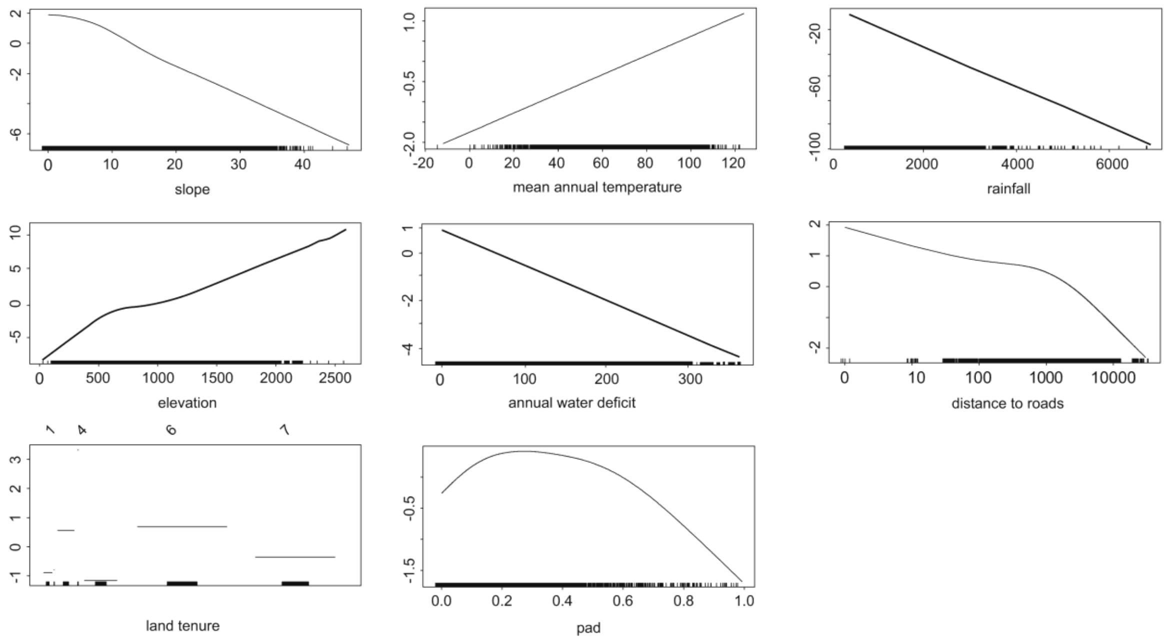
Appendix 4.1 Partial response curves of selected predictors in the (a) pre-1990 conversion model and (b) the model of conversion from 1990 to 2001. The y-axis shows the partial contribution of each predictor variable (the relationship of the response variable to the predictor considering the other variables in the model). The x-axis shows the selected predictor variable. For each continuous predictor, the response is represented by a fitted non-parametric smoothing function. Partial response curves of selected predictors in (c) the 1990–2008 conversion model, (d) the area model and (e) the difference between pre-1990 vulnerability and 1990–2008 vulnerability. Table 1 describes each predictor, including those represented by abbreviations.

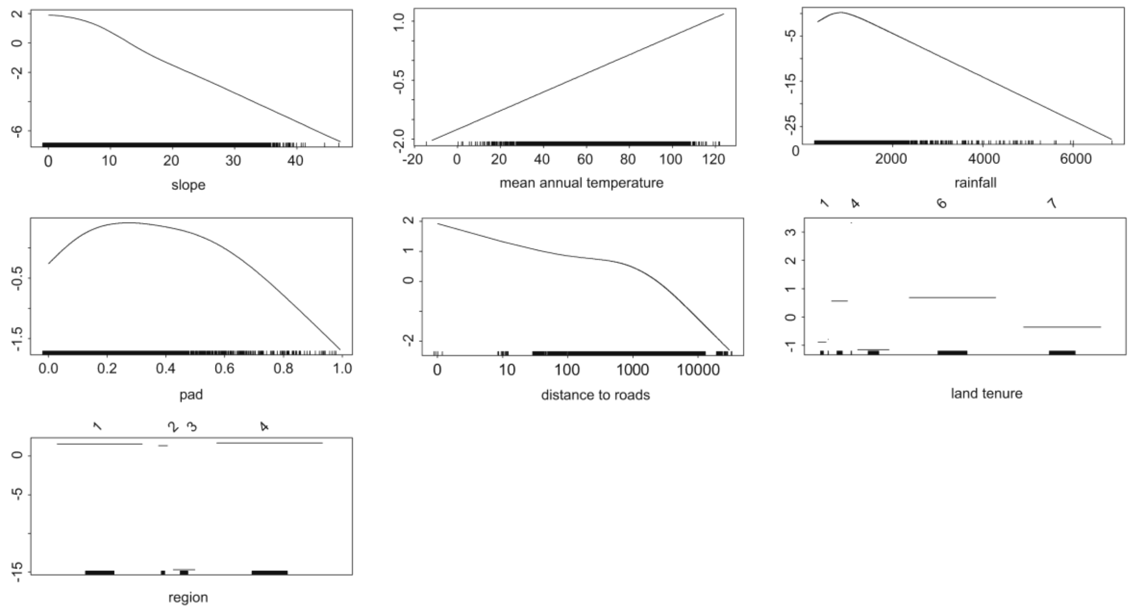
(a)



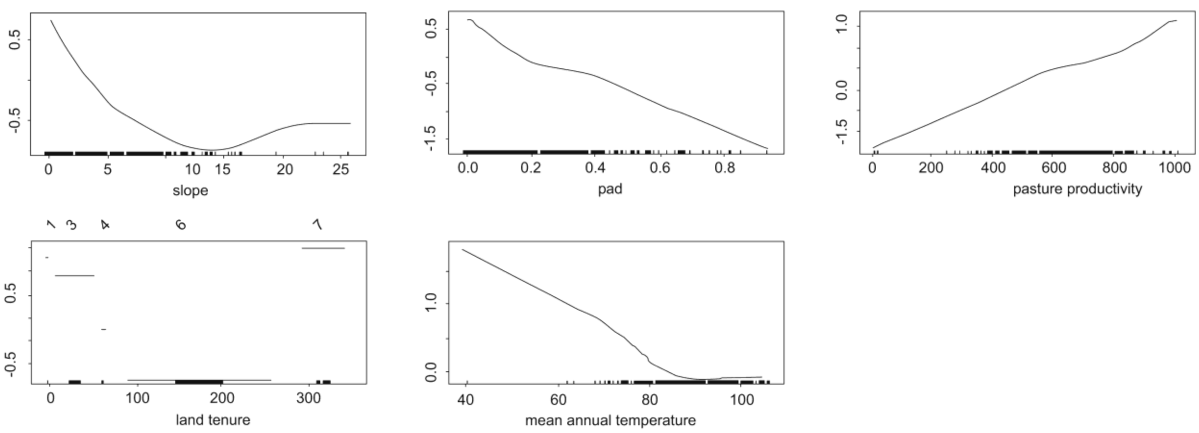
(b)



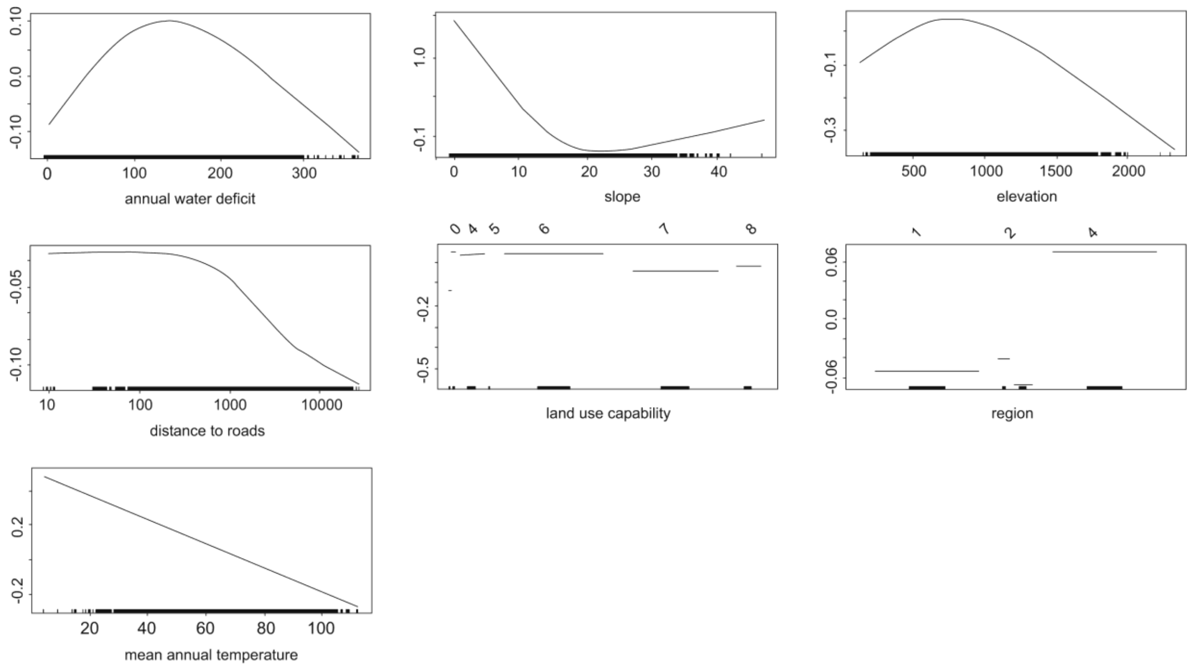
(c)


slope
mean annual temperature
rainfall
pad
distance to roads
land tenure
region

(d)

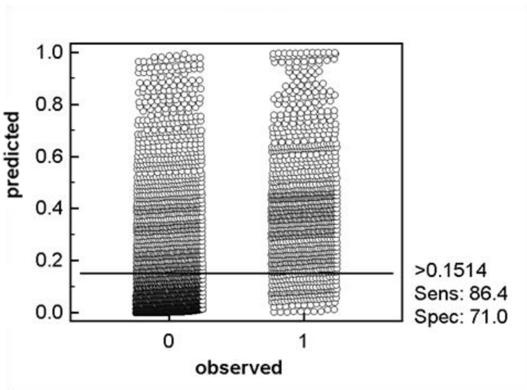

slope
pad
pasture productivity
land tenure
mean annual temperature

(e)

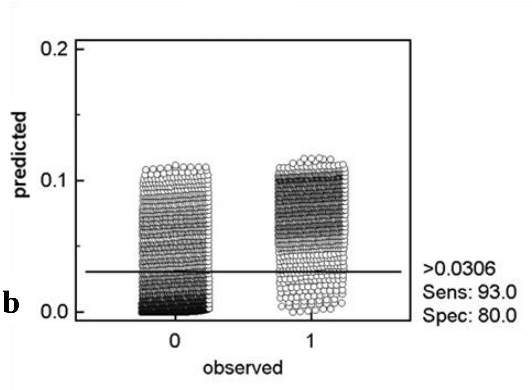

0.10
0.0
-0.10
0
100
200
300
annual water deficit
1.0
-0.1
0
10
20
30
40
slope
-0.1
-0.3
500
1000
1500
2000
elevation
-0.05
-0.10
10
100
1000
10000
distance to roads
0
4
5
6
7
8
-0.2
-0.5
land use capability
1
2
4
0.06
0.0
-0.06
region
0.2
-0.2
20
40
60
80
100
mean annual temperature

Appendix 4.2 Interactive dot diagrams used to validate predicted probabilities of conversion in models based on pre-1990 conversion (a), and 1990–2001 conversion (b). Negative and positive observations of conversion between 2001 and 2008 are separated on the horizontal axis and predictions from the models are shown on the vertical axes and displayed as dots in the diagrams. The horizontal line indicates the best separation (minimal false negative and false positive results) between the two groups. The corresponding test characteristics sensitivity (proportion of actual positives which are correctly identified), and specificity (the proportion of negatives correctly identified) are shown at the right side of the display.

a



b



5 The value of validated vulnerability data in conservation planning[6]

5.1 Abstract

Data needed for informed conservation prioritisation are generally greater than the data available, and surrogates are often used. Although the need to anticipate dynamic threats is recognised, the effectiveness of surrogates for vulnerability is seldom tested. We test the effectiveness of surrogates for the vulnerability of New Zealand's indigenous grasslands, and consider their application in conservation prioritisation tools, in a situation of rapid expansion of land use intensification and an active land reform program. Our comparisons of conservation prioritisation outputs with validated estimate of conversion-vulnerability show variable effectiveness of vulnerability surrogates; one surrogate performed most poorly where vulnerability of grasslands to conversion is greatest and realised probability of protection is lowest. We conclude that dynamic planning need not be complex, but validated vulnerability assessments may be crucial. Simple tools integrating irreplaceability and up-to-date validated vulnerability estimates may offer a practical and responsive technical bridge for the gap between science and implementation.

[6] Accepted subject to changes as Weeks E.S., Walker S., Overton J.Mc. to *Environmental Management*

5.2 **Introduction**

Systematic conservation planning is the process of identifying and configuring complementary actions required to achieve conservation goals (Margules and Pressey 2000; Moilanen 2008). Since the 1980s, numerous spatial approaches have been developed for identifying priority areas for conservation (Moilanen 2008). These systems require various forms of data, including information on the distribution of biodiversity (e.g. Ferrier and Drielsma 2010), the distribution and effects of pressures on biodiversity (such as pest, weeds, pollution and land clearance) and consequent vulnerability (Wilson and others 2005b), and the effects, and costs, of potential management of pressures (Wilson and others 2007; Underwood and others 2008). In all these areas, the data needed for informed prioritisation of conservation actions are generally greater than that currently available. Conservation organisations must therefore invest resources that could potentially be spent on other conservation activities into gathering and developing data. There is a growing literature on cost-effectiveness and optimality of data-gathering for guiding conservation planning (Cleary 2006; Bottrill and others 2008; McDonald-Madden and others 2008; Grantham and others 2008; 2009). A key message from this work is that diminishing returns are inherent in data gathering for conservation planning; at some point, it becomes more effective for conservation organisations to stop data gathering and instead to implement protection, albeit with imperfect information.

When data or data-gathering resources are scarce, surrogates are often used. There has been considerable assessment of, and debate about, the effectiveness of surrogates (such habitat maps) for the distribution of species and taxa (Rodrigues and Gaston 2002; Brooks et aland others 2004; Lombard and others 2003; Pressey and others 2004; Rodrigues and Brooks 2007; Grantham and others 2010). Most conservation planning approaches use surrogates for mapping pressures on biodiversity and vulnerability (Wilson and others 2005b), yet relatively little attention has been paid to their relative effectiveness. Surrogates for vulnerability in planning land protection reviewed by Wilson and others (2005b) included tenure and land use, environmental or spatial variables correlated with past conversion, threatened species distributions, and maps compiled from expert judgement. Many assumptions are inherent in the application of these surrogates.

For example, use of land tenure as a surrogate assumes vulnerability can be estimated from the associated permitted land uses; surrogates based on past conversion and threatened species patterns assume that future distributions and impacts of threatening processes are indicated by those in the past (Wilson and others 2005a).

Recently, attention in conservation planning research has shifted from techniques to produce static reserve blueprints, such as those produced by optimisation, to solving the challenge of conservation planning in the context of dynamic threats (e.g. Higgins and others 2000; Pressey and others 2007). These approaches acknowledge that threats are dynamic in most conservation planning situations, that prioritisation that ignores dynamism can be ineffective, and that the need for dynamic updating of conservation priorities is based on updated vulnerability data. However, a major implication of dynamic prioritisation is that solutions may be more demanding of data, and more complex to produce, than those that assume stasis (Pressey and others 2007).

This situation brings trade-offs between data gathering and conservation effectiveness into stark relief. Clearly, as with data on biodiversity distribution, there will be diminishing returns inherent in the gathering and validation of accurate data on expanding threats. Conservation tools based on less accurate data and more simple solutions may be more effective for conservation, once the cost and flexibility limitations of incorporating more accurate data are accounted for. For example, Meir and others (2004) demonstrated that comprehensive reserve network design may be counterproductive in situations where site availability is uncertain, reserve acquisition is protracted, and rates of biodiversity loss are high. They suggested that in these situations, simple decision rules, such as protecting the available site with the highest irreplaceability or with the highest species richness, may be more effective for protecting biodiversity than a static blueprint.

Importantly, prioritisation solutions that incorporate dynamic threats may not only have greater information needs, but could also be more difficult to communicate to practitioners, and to embed and implement within operational conservation organisations. Pressey and others (2007) acknowledge that while there is a need for science to solve the problems of dynamic planning, there is also a pressing need for policy and practice to catch up with science (Brooks and others 2004; Knight and others 2006; 2008). Fitting the solution to the practical situation is a challenge that has seldom been addressed in the literature (but see Ferrier and Drielsma 2010).

This paper considers the importance of using validated vulnerability data in conservation planning tools that assist prioritisation of conservation land acquisition in New Zealand's indigenous grasslands. These grasslands are subject to an active and on-going programme of land reform, which is splitting former Crown land leased for grazing into conservation and privatised land parcels. The grasslands are also subject to dynamic threats, especially a recent expansion of land use intensification.

First, using both simple and more complex conservation planning tools, we investigate how apparent conservation priorities based on surrogate information conform to priorities assessed on the basis of recently collected and validated vulnerability data. We also investigate how apparent conservation priorities change when validated vulnerability data are used in place of a simple land tenure surrogate in the more complex conservation planning tool. Second, we assess the congruence of realised protection outcomes of land reform program with the apparent conservation priorities from simple and more complex conservation planning tools, and those using surrogate and validated vulnerability data. We apply our results to discuss practical and potential value of incorporating of validated vulnerability into conservation planning tools in the context of New Zealand's indigenous grasslands.

Study area

About 4.4 million hectares indigenous grasslands remain in New Zealand, spanning an elevation range from 10 to 2749 m above sea level, with gradients of annual rainfall from 285 mm to 7018 mm at higher elevations, and mean annual temperatures from -6.9 to 16.2 °C. The majority of these remaining grasslands are contained within a contiguous area of 3.3 million hectares, which we treat as our study area.

New Zealand's indigenous grasslands in are dominated by tussock growth forms (elsewhere known as “bunch grasses”) (Levy 1951; Ashdown and Lucas 1987; Mark 1993) of *Chionochloa, Poa,* and *Festuca* species. Below treeline, they have a unique, human- induced, origin, and were created in the last 800 years by Maori forest burning (Stevens and others 1988; McGlone 2001; Ewers and others. 2006). Although large areas of indigenous grassland habitat still remain, there is ongoing loss, which appears to be proceeding most rapidly at lower elevations (Walker and others 2008a; Mark and others 2009). Lowland indigenous tussock grassland plant communities are severely modified by past fire and pastoral grazing, are invaded by exotic species, and are poorly protected (Mark and McLennan 2005; Mark and others 2009). Nevertheless, they retain important residual indigenous biodiversity, including many threatened plants (Walker and others 2008b; de Lange and others 2009) and endemic lizards and invertebrates (Patterson 1992; Daugherty and others 1994; Patrick and Dugdale 2000; Patrick 2004).
In 1992 a land-reform program (colloquially called 'tenure review') began splitting 2.4 million ha of Crown-owned pastoral leasehold properties within New Zealand's indigenous grassland zone into private and conservation land parcels (Brower 2008; Walker and others 2008a; Mark and others 2009). In this same period, there has also been a significant trend to more intensive agricultural land use (cultivation and irrigation) in New Zealand, most notably in response to international demand for dairy exports (Ministry of Agriculture and Forestry 2007). Walker and others (2008a) observed that land most vulnerable to habitat conversion and rich in threatened plant species was being privatized by land reform, while land at least risk of biodiversity loss had been protected.

5.3 **Methods**

Data

Available conservation planning tools

Two conservation prioritisation tools are available in New Zealand that could be implemented to improve land protection in indigenous grassland land reform. The first is the Threatened Environment Classification (hereafter 'TEC') which categorizes New Zealand's land environments (Leathwick and others 2003; used as surrogates for potential ecosystem pattern) into six categories based on the proportion of indigenous land cover remaining and the proportion legally protected for conservation (Walker and others. 2006). The two highest-risk categories of the TEC (land environments with <20% indigenous cover remaining) are incorporated into national biodiversity guidance as 'national priorities' (Ministry for the Environment, 2007).

Recently, a more complex framework (Vital Sites and Actions, hereafter 'VSA') has been developed, in which priorities are assessed based on a combination of 'Significance' (an estimate of irreplaceability, i.e. the marginal contribution if a site to a conservation goal) and expected loss (i.e. vulnerability). While both TEC and VSA planning tools incorporate and rely on surrogates for irreplaceability and vulnerability, in VSA the two are explicitly distinguished. Furthermore, unlike the TEC, VSA is a flexible framework (*sensu* Ferrier et al. 2010) in which estimates of vulnerability (expected loss) can also be directly and readily updated.

Validated vulnerability

As 'validated vulnerability' we use a spatial prediction of the probability of land conversion between 1990 and 2008. This spatial layer was chosen from a suite of models of past and recent grassland conversion in relation to a suite of 23environmental and socioeconomic predictor variables, and is the first validated model of patterns of grassland conversion developed for New Zealand grasslands.

Land tenure

Spatial data depicting land tenure were compiled from the most up-to-date digital spatial data government agencies could supply. Protected land was compiled from data depicting public protected land administered for natural heritage purposes by the Department of Conservation, regional parks administered by regional territorial authorities, and covenants (private land administered by the Department of Conservation, Nga Whenua Rahui, or the QEII National Trust) in 2008. A second land tenure category (Crown grazing land, comprising land under a perpetually renewable pastoral lease or fixed-term pastoral occupation licence and owned by the Crown) was also sourced from the Department of Conservation. Remaining land was categorised as private land.

Analyses

All our GIS procedures used ESRI's ArcView 3.2. For data Generalized Additive Models and regression models we used S-Plus (MathSoft 1997) and the Generalized Regression Analysis and Spatial Prediction (GRASP) set of functions (Lehmann and others 2002).

Correlation between the TEC and validated vulnerability

We first sampled validated vulnerability and the TEC at 5000 randomly-placed sampling points across New Zealand grasslands, and regressed and plotted validated vulnerability on the TEC categories, assuming that the six categories represent a progression from the most vulnerable to the least vulnerable environments. We then built a generalized additive model of residual vulnerability in the TEC categories in relation to the same suite of predictor variables as used to predict validated vulnerability.

VSA priorities with surrogate and validated vulnerability

The modelling process involves predicting current and natural distributions of native species in response to threats (e.g. pests or habitat loss) on biodiversity, and the effects of management on relieving threats. It operates at two levels: species and ecosystem, and assessments of priorities can be made at each separate level or by combining the two levels. We used the model at ecosystem level of analysis and for one threat (vegetation clearance), and will only describe the methods for this level of analysis.

Next, we ran the VSA model twice, first with 'surrogate' vulnerability and the second time with 'validated vulnerability'. In the first run, surrogate vulnerability was a probability of clearance assigned by expert judgment to our three different types of land tenure. We regressed priorities based on validated vulnerability against priorities based on surrogate vulnerability. Next, we ranked priorities from the two different VSA runs, and scaled them so that 1000 represented the highest and 0 the lowest priorities, and subtracted ranked priorities based on surrogate data from those based on validated data. This produced a 'difference' map depicting the spatial distribution of changes in priority across indigenous grasslands remaining in the study area in 1990. We sampled this map at 5000 random points, and built a generalized additive model of difference in priority rank in relation our suite of environmental and socioeconomic predictor variables.

Realised protection outcomes, vulnerability, and conservation priorities

We then also built a generalized additive model of probability of protection in relation to our environmental and social predictor variables (Appendix 5.1). The model was cross-validated (using ROC; Fielding and Bell 1997), interpreted using contributions and partial response curves, and its predictions were mapped across indigenous grasslands remaining in the study area in a GIS. We sampled this layer at 5,000 points and regressed the probability of protection on TEC categories and validated priorities.

Finally, we further investigated patterns of realised protection in relation to validated vulnerability, TEC categories, and VSA priorities within a subset of data. The subset covered only land within the study area that had been identified as worthy of protection in field survey within the spatial boundaries of 66 pastoral leasehold properties, and that had completed land reform between 1992 and 2008. These further analyses test whether patterns of realised protection observed across all indigenous grasslands are repeated in a subset of areas for which 'significance' has been validated. We divided the data subset into four land reform outcome classes: these were public conservation land protected from stock grazing, public conservation land with a short-term stock grazing license (i.e. a less rigorous level of public protection), privatised land encumbered by a conservation easement or 'covenant' to limit threatening land uses, and private land unencumbered with any protective mechanism. To display patterns of protection outcomes, we graphed the distribution of validated vulnerability, TEC categories, and validated priority from VSA within each outcome class.

5.4 **Results**

Conservation priorities and validated vulnerability

There was a monotonic decrease in the probability of grassland conversion between 1990 and 2008 across categories of the TEC from the category of highest risk (Category; <10% indigenous cover remaining; Acutely Threatened) to lowest risk (Category 6; >30% indigenous cover remaining and >20% of land environment protected) (Figure 5.1a) Linear regression suggested that TEC explained 57% of variation in validated vulnerability. There was considerable scatter in the intermediate categories, however, with the fourth category in particular including much land that was both highly vulnerable to land clearance and not vulnerable to land clearance at all. Slope played a dominant role in predicting residual vulnerability from the linear regression trend of vulnerability on TEC categories, it was the only variable that, when dropped from our model, could not be compensated for by the combination of other variables. Ranked contributions of variables followed the order slope, distance to roads, mean annual temperature, elevation, region, and annual water deficit. Therefore, vulnerability to clearance is likely to be lower than predicted by the TEC on steep slopes in remote areas than on gentle slopes, near to major roads.

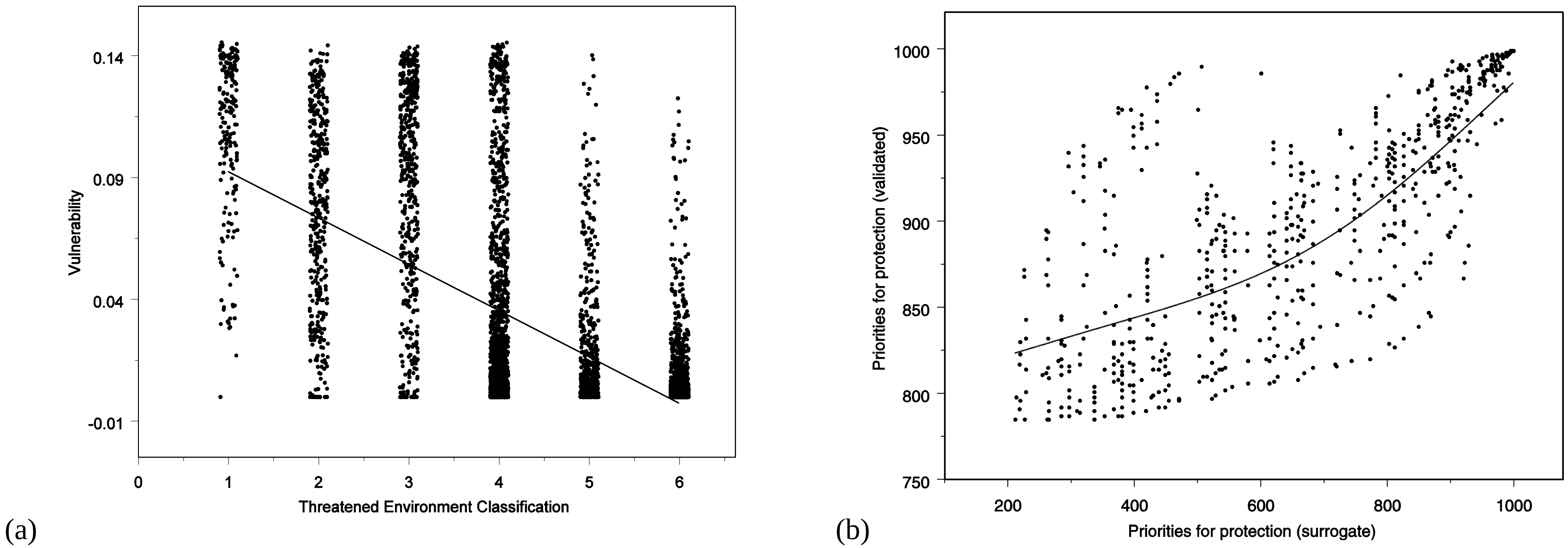


Figure 5.1 The regressions use 5,000 sampling points placed randomly across the study area (the line is a local regression spline smoother) comparing (a) vulnerability in six categories of the threatened environment classification and; (b) priorities based on validated vulnerability on priorities based on surrogate vulnerability from the VSA model. The 'Threatened Environments' classification assumes category 1 (Acutely Threatened) is the most vulnerable and category 6 (Less Reduced and Better Protected) is least vulnerable: Priorities for proteciton rank from 750 (low prioirty) to 1000 (highest priority).

Priorities estimated by VSA based on surrogate and those based on validated vulnerability were also positively correlated (Figure 5.1b). There was good agreement between the two runs on the very highest priorities, but considerable scatter elsewhere. e.g. pixels given low priority based on validated probability of clearance that were ascribed very high priority. Priorities based on validated vulnerability spanned a narrower range (750 - 1000) than priorities based on surrogate vulnerability, suggesting validated vulnerability provided greater discrimination between high and low priority.

Figure 5.2 shows that validated priority (Figure 5.2b) shows a similar spatial pattern to validated vulnerability (Figure 5.2a). The grasslands most vulnerable to conversion, and with the highest priorities for protection, are largely clustered around the fringes of masked areas that represent low-elevation grasslands that were converted before 1990. Importantly, Figure 5.2c shows that the greatest positive disparity between priorities was in the grasslands that are most vulnerable to conversion: surrogate vulnerability led to underestimations of their vulnerability and priority.

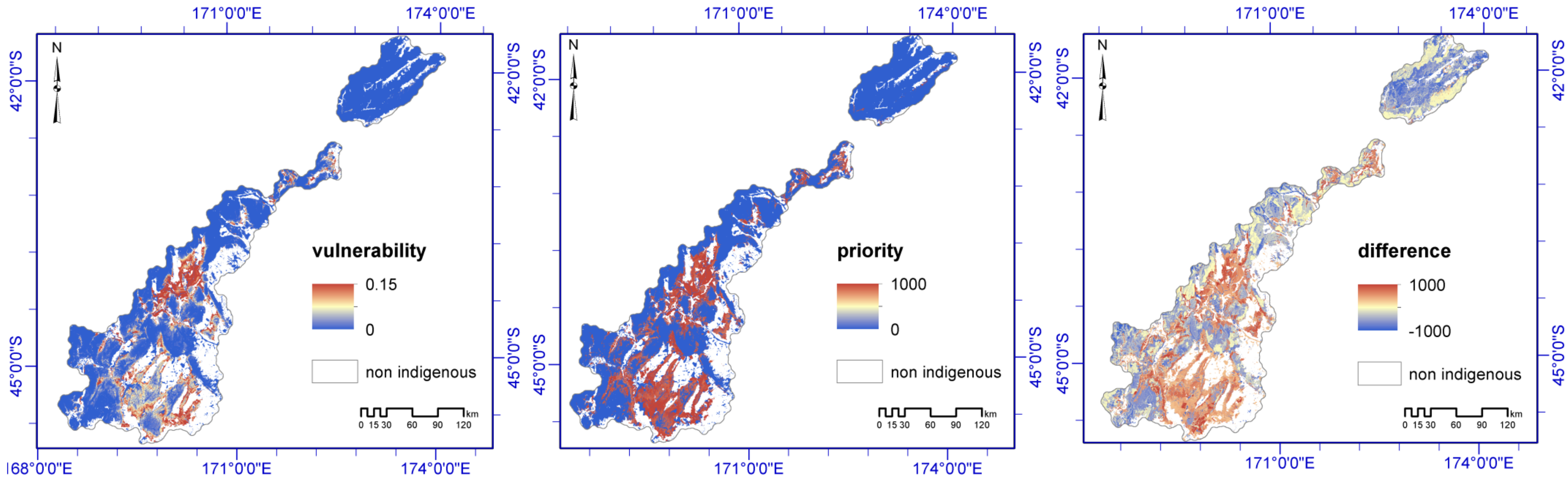


Figure 5.2 Map of (a) validated vulnerability, (b) priority from the VSA model based on validated vulnerability, and (c) difference in ranked priorities from two runs of the VSA model (priority based on surrogate vulnerability subtracted from priority based on validated vulnerability). In (c) high positive values indicate areas where priority was most greatly underestimated when surrogate vulnerability was used.

Two environmental predictors –elevation and slope– explain the disparity in priorities (Table 1). The residual vulnerability (deviance from the linear regression trend) in the TEC categories is related to lower slopes and higher elevation. VSA gave lower priority to lower elevation and flatter land when surrogate vulnerability was used in place of validated vulnerability.

Table 5.1 Contribution (rounded to the nearest whole number) of selected predictors to three GAM models: (a) partial response curves of residual vulnerability from the TEC, (b) priorities for protection validated vulnerability and using surrogate data to estimate vulnerability, and (c) priorities for protection using validated vulnerability and the probability of protection based on recent patterns of protection. The drop contribution indicates the marginal contribution of each variable and is obtained by dropping each explanatory variable and calculating the associated change in deviance. The alone contribution reflects the potential of each variable, and is calculated by creating a new models with only one predictor.

	TEC residual		**VSA**		**Priorities for protection**	
Predictors	**Drop**	**Alone**	**Drop**	**Alone**	**Drop**	**Alone**
Distance to roads	26	68	12	52	10	56
Elevation	27	39	37	155	36	153
Slope	115	172	22	150	109	164
Annual water deficit	20	15	42	152	25	18
Mean annual temperature	3	46	10	15	9	38
Region	15	39	19	20	12	29

Realised protection outcomes, vulnerability, and conservation priorities

A model of protection probability on our environmental and socioeconomic predictors had a cross validation of 0.83 showing good model stability. Predicted and observed protection were positively correlated, with predicted protection explaining about half (R^2= 0.514) the variation in the observed location of protected areas. Protection probability was greater for higher elevation land. Contributions of predictors to the protection probability model showed elevation played a dominant role it was the only variable that, when dropped from the model, could not be compensated for by the combination of other variables.

Ranked contributions of variables followed the order elevation, soil moisture deficit, mean annual temperature, distance to roads, region, and finally slope.

Figure 5.3 shows the probability of protection of remaining indigenous grasslands in the study area between 1990 and 2008 in relation to vulnerability, TEC, and protection priorities from VSA based on validated vulnerability. Across all remaining indigenous grasslands, the probability of protection was inversely related to vulnerability (Figure 5.3a); places with the highest probability of protection from 1992 to 2008 were the least vulnerable. Probability of protection also increased across the six TEC categories (i.e. from the most to the least vulnerable) (Figure 5.3b); most areas in the four most vulnerable TEC categories had very low probabilities of protection. Probability of protection showed a less distinct relationship with priority from the VSA model, and there was considerable scatter (Figure 5.3c). On average, areas that were moderately low priorities in the VSA model had the highest probabilities of protection from 1992 to 2008 and areas that VSA gave the highest priority had the lowest probabilities of protection.

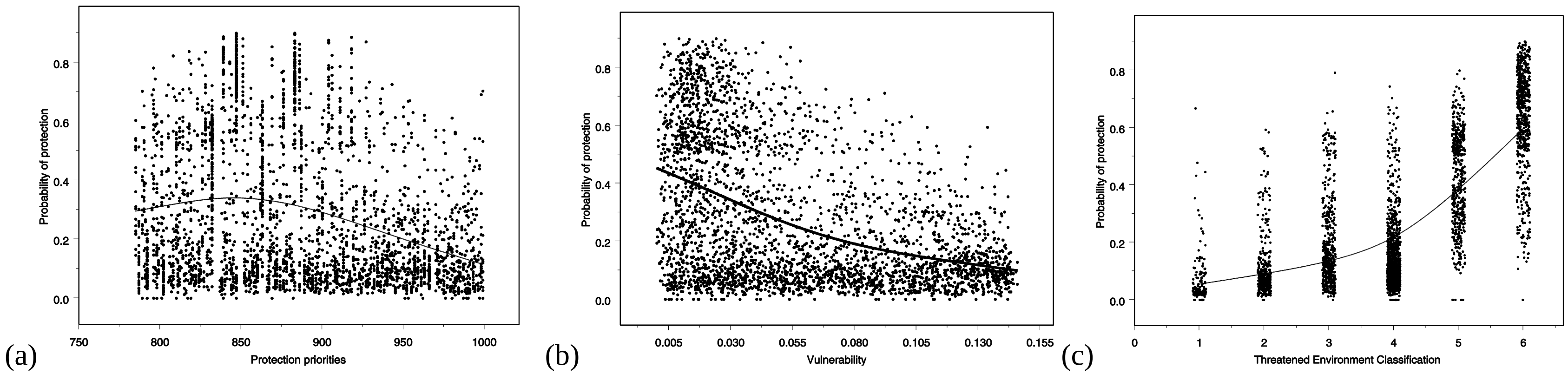


Figure 5.3 Probability of protection of remaining indigenous grasslands in the study area between 1990 and 2008 in relation to (a) validated vulnerability, (b) the Threatened Environments Classification (TEC) and (c) protection priorities from the VSA model based on validated vulnerability. Each graph shows 5,000 randomly-placed sampling points across remaining indigenous grasslands. Solid lines are local regression spline smoothers.

Figure 5.4 shows land designations in land reform for only those areas that were identified as having 'significant inherent values'. Within this subset of land, almost all that was designated as new public conservation land (categories PCL and PCL_G) was at high elevations, and relatively little was in places that are highly vulnerable to conversion. The subset of grasslands with 'significant inherent values' that were most vulnerable (Figure 5.4a), and most threatened (according to the TEC classification; Figure 5.4b) tended to be privatised in land reform (category Pvt_U), while those that were less vulnerable, and in the least threatened TEC category were most consistently protected. The pattern of designations in relation to 'priority' as attributed by the VSA model was less clear, however, and all four land designation categories included a considerable range of VSA priorities (Figure 5.4c).

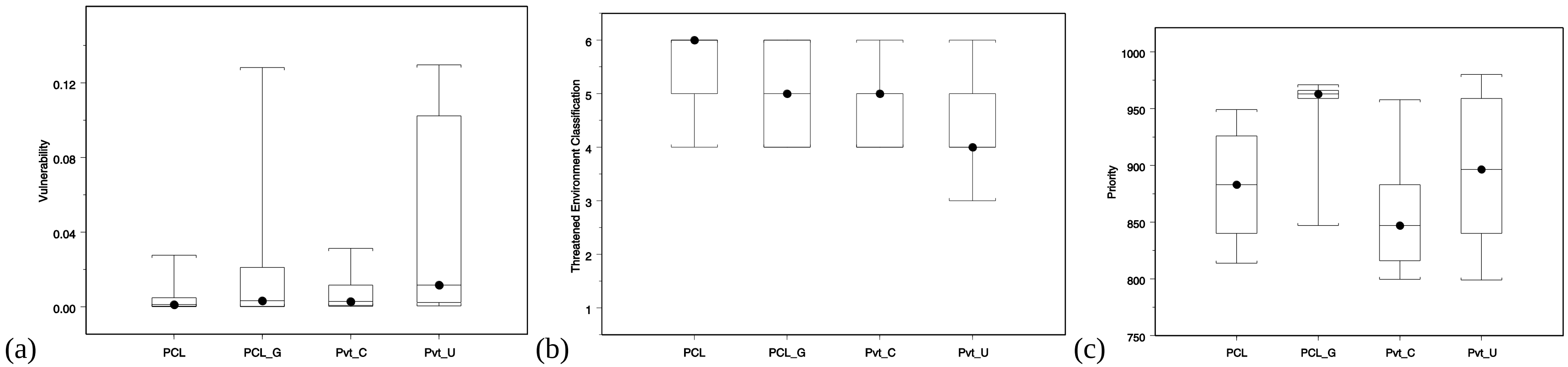


Figure 5.4 Land designation outcomes of land reform for areas with identified 'significant inherent values' on a subset of pastoral leases in relation to (a) vulnerability (b) classes of the threatened environment classification (TEC) in which 1 is most vulnerable and 6 is least vulnerable, and (c) protection priority from the VSA model based on validated vulnerability. The four categories show public conservation land (PCL), public conservation land with a grazing licence (PCL-G), land privatised with a conservation easement (Pvt-C) and land privatised with no encumberance.

5.5 **Discussion**

Variable performance of vulnerability surrogates

Most quantitative methods for identifying conservation priorities require detailed knowledge about the extent and distribution of biodiversity and the dynamic nature of the conservation problem than is currently available, and surrogates are often used. Some surrogates for species and habitat distributions may be cost effective (Cowling and Heijnis 2001; Grantham and others 2008; Lombard and others 2003), and others less useful (Brooks and others 2004; Grantham and others 2010). Our study suggests that some estimates of vulnerability based on surrogates are relatively reliable (such as the TEC), but also demonstrates that others (such as land tenure in this instance) can provide relatively misleading assessments of conservation priorities.

The two different surrogates had different weaknesses. Relative to a validated estimate of the rate and spatial distribution of indigenous grassland habitat loss, the TEC tool underestimated vulnerability on flat land that was closer to roads in intermediate TEC categories. The TEC also overestimated vulnerability on steeper land that was topographically invulnerable to clearance in 'national priority' environments. Validated vulnerability made a significant difference to the conservation prioritisation output of a recent prioritisation model (VSA) for New Zealand’s indigenous grasslands. Whereas use of a land tenure surrogate gave greater priority to land at higher elevations and with steeper slopes that was not vulnerable to conversion, validated vulnerability data enabled VSA to more consistently prioritise areas most vulnerable to conversion at lower elevations and on flatter land.

Contrasting patterns of protection and vulnerability

In New Zealand's grassland land reform, the most vulnerable indigenous grasslands are being privatised, while the least vulnerable are being protected. Our results show that the majority of New Zealand's indigenous grasslands protected over the last two decades are at relatively high elevations and on steeper slopes; places that have very low vulnerability to perhaps the most relevant and active threatening process in this biome, which is grassland conversion for intensive agriculture.

The TEC tool's first four categories effectively targeted the low elevation places that are highly vulnerable to clearance and poorly protected in land reform, with the proviso that indigenous grasslands on steep slopes within these four categories were relatively invulnerable to clearance. In contrast, the pattern of conservation priorities produced by the land tenure vulnerability surrogate in VSA not only failed to prioritise grasslands that were actually most vulnerable, it also produced patterns of priority that were similar to recent observed patterns of protection.

Our result supports the prediction of Walker and others (2008) that extensive habitat loss may result from New Zealand's land reform. The pattern of protection also follows global preference, described by Pressey and others (2004), for protecting ecosystems that are ‘cheap’ because they are residual from economic land use, and also happen to be scenic and pristine because they are less threatened.

Prioritisation and implementation

The observed pattern of land allocation outcomes in New Zealand's grassland land reform suggests tools such as the TEC and VSA model developed for more efficient reserve selection are not being applied; there appears to be a gap between science and implementation. It is increasingly recognised that systematic conservation planning must be complemented by an implementation strategy (Knight and others 2006), or at least consider implementation issues in its design; ultimately an effective conservation plan is one that translates science into action (Groves and others 2002). Design of conservation planning tools for ready uptake and application has been little discussed in the literature. However, Ferrier and Drielsma (2010) noted that practitioners preferred not to use static optimised maps, and required tools that help them make quick decisions for estimating marginal benefits. Even earlier, Meir and others (2004) also demonstrated convincingly that simple decision rules may have greater practical utility than detailed optimised plans when degradation rates and uncertainty are high, and implementation is carried out over a number of years. All these conditions apply to the case of New Zealand's indigenous grasslands.

Although outcomes of grassland land reform do not appear to have implemented the simple TEC tool, the tool has otherwise been quite widely adopted in New Zealand to inform national guidance and regional policies for biodiversity. The simplicity of the tool may have facilitated this uptake. While it is reassuring to note the strong correlation between the TEC and validated indigenous grassland vulnerability, our results suggest that high vulnerability to conversion extends beyond the first two TEC categories (which have been adopted as 'national priorities' for biodiversity protection on private land). The high recent rates of indigenous vegetation clearance in intermediate TEC categories suggests they too may have less than 20% remaining indigenous cover soon. Our analysis suggests national priority for biodiversity protection should be extended to indigenous grasslands remaining on all gently sloping land in the first four TEC categories.

In the process of land reform on Crown land in New Zealand's indigenous grasslands, ecological values, and their significance, are assessed by survey on a property-by-property basis, often for the first time. A limitation of implementing the VSA conservation prioritisation model in this circumstance is its requirement for *a priori* irreplaceability (significance) data to estimate priorities. There are likely to be high cost, and social and practical barriers to implementing and updating such survey in New Zealand's indigenous grasslands. The suitability of the naturalness surrogate in VSA that informs significance, and hence priority, is not directly tested in this paper. However, Walker et al. (2008) showed that the majority of threatened plant species in interior New Zealand are in highly modified grasslands of low naturalness. Such modified places would generally be regarded as being of high irreplaceability, but are given low significance and hence priority by the VSA. The TEC, on the other hand, disregards naturalness, and tends to target modified grasslands that hold the majority of threatened plant species. Therefore, although realised protection patterns in land reform outcomes appear less discouraging in relation to VSA priorities that incorporate validated vulnerability than they in relation to validated vulnerability and TEC categories, this result may be an artifact created by inappropriate use of surrogate information in another aspect of VSA.

Our study suggests that in land reform, once significant values had been confirmed to be present, land allocation decisions could be practically guided by either a validated vulnerability map, or the simple TEC tool (which is based on surrogate data but shows a strong correlation with validated vulnerability, particularly in more vulnerable places). The contrary patterns of vulnerability and protection probability we have shown here suggest application of either of these simple tools might substantially improve conservation land allocation decisions. This simple, case-by-case integration of significance and vulnerability may also be practical in other New Zealand planning situations, such as indigenous vegetation clearance consent decisions on privatised land, which are applied for sporadically.

5.6 **Conclusion**

The integration of validated estimates of vulnerability into conservation planning tools is an important component of dynamic conservation planning. We conclude, moreover, that dynamic conservation planning need not be very complex and unwieldy. An advantage of using a stand-alone conversion-vulnerability map as a decision tool is that it is relatively cheap and straightforward to estimate, validate, and hence regularly update from remote sensing data. It is also relatively simple to understand, which may facilitate uptake. Regular updates of land cover data, would mitigate the recognised risk that past patterns of vulnerability to conversion may not indicate future patterns, due to extrinsic forces such as changes in global markets, development of new technologies, and shifts in land use decisions (Wilson and others 2005a).

However, we also recognise that New Zealand’s land reform decisions, like those that determine conservation land allocations internationally, are only partially driven by science (Pressey 2004). Indeed, they may be relatively little influenced by modern understanding of conservation priorities and conservation planning tools. In New Zealand's indigenous grasslands, Brower (2008) illuminated key socio-political drivers of the conservation and privatisation outcomes that interact with, and are likely to be more influential than technical conservation planning tools. To make progress, it is probably essential that conservation planners engage with and design tools and implementation strategies that anticipate and respond to these other drivers.

5.7 Acknowledgements

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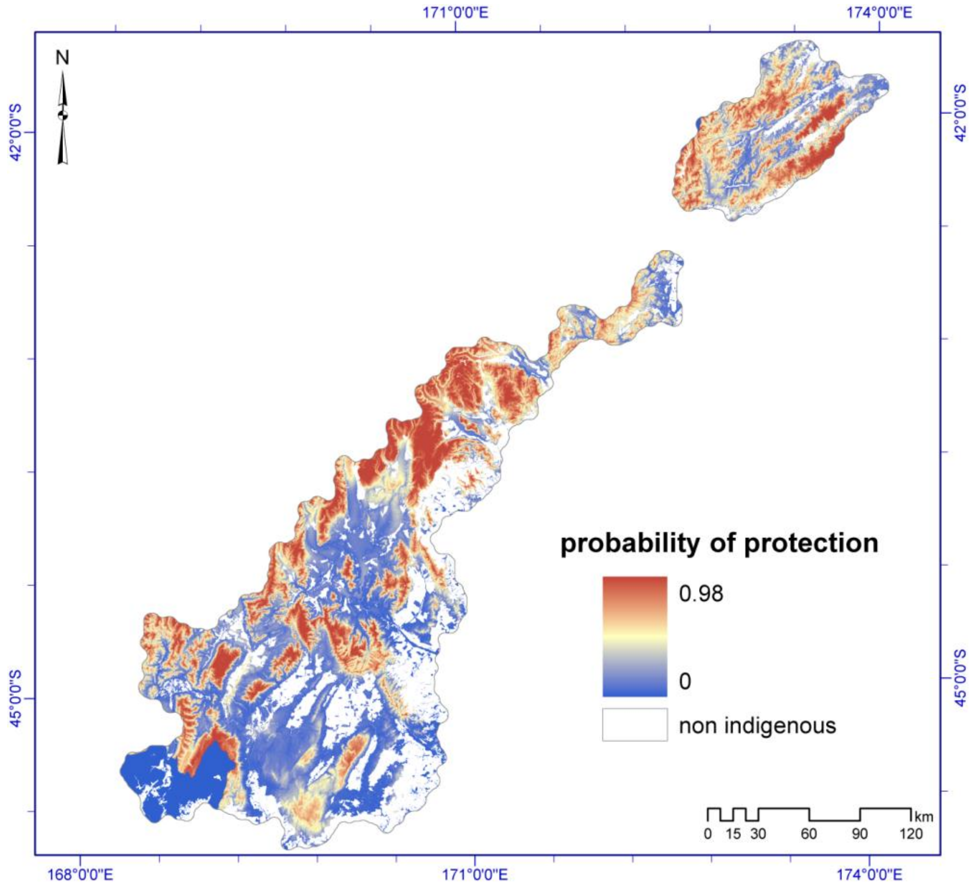
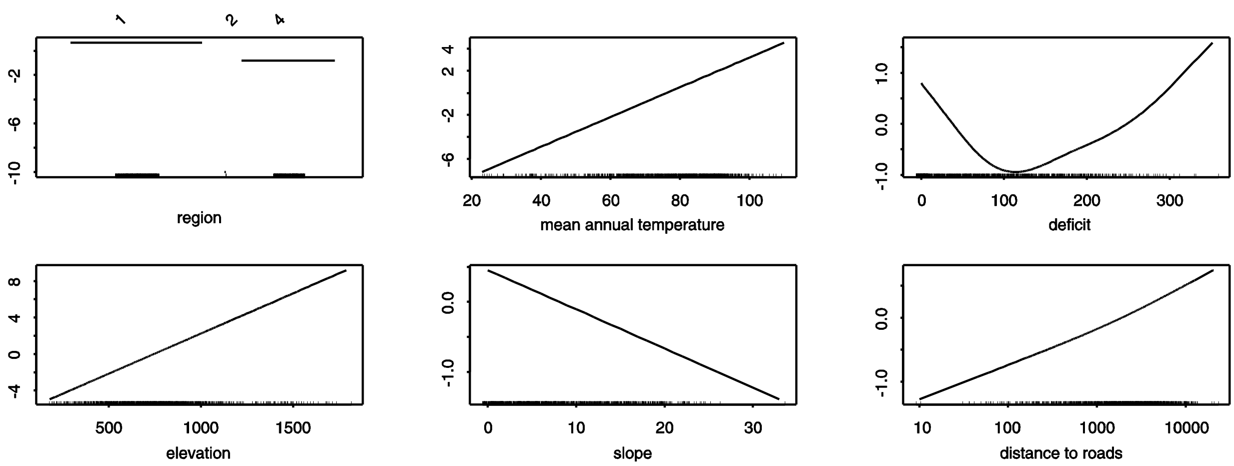
Appendix 5.1 Probability of protection of remaining indigenous grasslands within the study area based on a model of current patterns of protection.



Appendix 5.2 Partial response curves of selected predictors in the GAM of probability of protection. The y-axis shows the partial contribution of each predictor variable (the relationship of the response variable to the predictor considering the other variables in the model). The x-axis shows the selected predictor variable. For each continuous predictor, the response is represented by a fitted non-parametric smoothing function.



6 Synthesis

6.1 Discussion

This research has added to our understanding of developing methods for monitoring habitat loss, and understanding patterns of biodiversity loss and conservation priorities in New Zealand's indigenous grasslands. It addresses the information gaps in remote sensing technological developments and national land cover data (Chapter 2), and has quantified the rates and types of land-use conversion in New Zealand's indigenous grasslands, using satellite imagery (Chapter 3). It has also identified the major environmental correlates of habitat loss and developed temporally and spatially validated models of land conversion over time (Chapter 4). Lastly, it has outlined gaps in existing conservation prioritisation tools in New Zealand (Chapter 5).

To evaluate change in grassland cover (habitat loss) three types of data were used in this study: field observations, aerial photography, and a time series of Landsat ETM+and SPOT 4 and 5 data centered on the South Island of New Zealand during an 18-year period (1990-2001 and 2001-2008). As the basis for describing patterns of conversion I developed spatially explicit models using a regression based approach to analyse historical, current and future patterns of conversion against a number of candidate environmental and socio-economic predictors.

Spatially explicit information describing the extent, condition, protection status and trends in New Zealand's indigenous grasslands is a critical requirement for assessing the impacts of current land management practices and conservation initiatives. This study confirms limitations in existing Land Cover Database (LCDB2) for detecting changes in grasslands (Walker et al. 2006), and indicates that coarsely assigned land cover classes remain inadequate to assess biodiversity loss. Automatic detection technology can be used to provide reliable estimates of change in woody vegetation (Adams et al. 1995; Skole & Tuker 1993; Dymond 2007) however it is not able to provide reliable estimates for grasslands and other herbaceous vegetation types. Temporal variability in moisture content influences the spectral signature of herbaceous vegetation far more than that of woody vegetation which makes identification of incremental changes in grassland cover increasingly difficult.

Characterising the patterns and rates of loss of biodiversity is crucial for estimating overall biodiversity loss and designing management to mitigate loss. In New Zealand's indigenous grasslands 3% of the area assessed had changed to a non-indigenous cover. The change was not uniform throughout the study area but was concentrated in areas which indigenous vegetation cover was already very low and poorly protected. The scale of grassland conversion and widespread disparity of between habitat loss and protection suggest that the cumulative effects of land intensifications on biodiversity loss and ecosystem services deserve greater attention in planning decisions under the Resource Management Act 1991.

Modeling patterns of loss of biodiversity is an important component of conservation management and planning (Margules and Pressey 2000, Wilson et al 2005). Generalized Additive Models can provide a tool for capturing the essence of where loss of biodiversity is likely to take place, what is driving these patterns, and thus determine vulnerability to future loss. This study adopted a model using a statistical deduction approach (using the tool GRASP in S-Plus) that analysed observed patterns of recent conversion and compared them to user-supplied map layers of physical (environmental) and socio-economic attributes. The model chose the predictors of biodiversity loss based on the best fit patterns. The results from the models provided the first data-derived and statistically validated measurement of vulnerability of New Zealand's indigenous grasslands to conversion, and show a recent trend to greater loss of biodiversity on higher more marginal land. They also show that up-to-date land conversion data were better at predicting current conversion and future patterns of conversion.

Environmental decision making is based on the quantification of environmental properties such as vulnerability. The integration of validated estimates of vulnerability into conservation planning tools is an important component of conservation planning. Most quantitative methods for identifying conservation priorities require detailed knowledge about the extent and distribution of biodiversity and the dynamic nature of the conservation problem, than is currently available. As a result surrogates are often used. This study demonstrates that estimates of vulnerability based on surrogates provide relatively misleading assessments of conservation priorities. In New Zealand's grassland habitat existing conservation priority tools based on surrogates underestimate priority on flat land and overestimates priority on steep land at higher elevations. Conservation prioritisation that is based on validated vulnerability data consistently prioritises areas more vulnerable to conversion at lower elevations and on flatter land. Land allocation decisions for protection could be practically guided by a validated vulnerability map, and regular updates of land cover data would mitigate the recognized risk that past patterns of vulnerability to conversion may not indicate future patterns of vulnerability to conversion.

6.2 Directions for future research

This thesis has identified the following areas for future research:

- This study highlights the limitations to using coarsely assigned land cover classes (such as those used in LCDB1 and 2) and suggests improvements in estimates of grassland land cover and better estimates of the heterogeneity of grassland types are needed. A challenge to land cover change detection science is distinguishing change in land cover from variability within land cover types. In New Zealand's grasslands inter-seasonal variability makes identification of land cover change difficult. In order to identify change it is important to have a baseline. Analysis of a time series of satellite imagery of four different grassland types suggest that the temporal pattern of reflectance throughout the seasons for exotic grasslands (improved pasture) is very different from indigenous grasslands (snow tussock and short tussock). This trend suggests improved pasture spends considerable periods of time through the year with much green biomass, while tussock grasslands spend most time with little green biomass. A preliminary study of mapping biomass and cover in New Zealand's grasslands using multi-spectral narrow-band data (Vescova et al. 2009) confirm that there is the potential to up-scale the biophysical parameters estimate from ecosystem to landscape level.
- Over the past two decades a range of spatially explicit land use change models have been developed to meet land management needs and to better

assess the vulnerability to habitat loss (Veldkamp and Lambin 2001, Wilson 2005). Most models of land use change (or habitat loss) can identify the location of future change (or habitat loss) and quantify that change. A prerequisite to the development of realistic models is the identification of most important correlates (or predictors) of change. This study used a regression based approach (using the tool GRASP) to understand historical, current and future patterns of land use change (conversion) by establishing relationships between a wide range of socio-economic and environmental proxy variables (i.e. distance to roads). The advantage of using a statistical based model is that it is easier to implement, produces robust and transparent results, and provides a quantitative measure to validate results. However, a model using proximate causes of land conversion may obscure causality. Most case studies highlight the need to account for the endogeneity of variables such as land management technologies, infrastructures or land use policies. To account for endogeneity a process based model that is able to deal with temporal changes in driving forces or processes and represents the decision-making process by actors needs to be adopted. Further developments include: modeling the quantity of future land conversion in addition to the spatial distribution; modeling drivers of conversion; and incorporation of biophysical feedbacks.

- A comprehensive assessment of vulnerability would consider all of the threats affecting an ecosystem and also the dynamic responses of threats to conservation actions (Wilson et al 2005). This study only assessed one component of vulnerability but combining other threats into prioritisation models is recommended. There are a wide variety of threats that could be considered including invasive flora species, introduced pest species, and the impact of a threatening process on individual species. In the case of grasslands, this analysis was limited by the availability of relevant data. Substantial efforts have been put into collecting flora biodiversity data for tall tussock grasslands but there is a lack of sufficient vegetation plot data for short tussock grasslands (Cieerad 2008), which are the most vulnerable to conversion.

- A range of tools and approaches have been developed in the past 10 years to support systematic efforts including measuring Conservation Achievement (Stephens et al. 2002), Project Prioritisation Protocol (Moilanen et al. 2008), and measuring provision of natural habitat (Dymond et al. 2007), however, a simple inexpensive conservation planning tool is still needed to guide conservation planning. This study highlights the importance of integrating validated assessments of vulnerability into conservation planning. It also suggests implementing this method across all ecosystems, but also requires developing a framework that can be used to allocate limited funds to threat specific conservation actions in areas where they are likely to achieve minimal loss in national ecological integrity (Lee 2007).

6.3 **References**

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