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Remotely sensed landscape heterogeneity as a rapid tool for assessing local biodiversity value in a highly modified New Zealand landscape

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Abstract. The widespread conversion of natural habitats to agricultural land has created a need to integrate intensively managed landscapes into conservation management priorities. However, there are no clearly defined methods for assessing the conservation value of managed landscapes at the local scale. We used remotely sensed landscape heterogeneity as a rapid practical tool for the assessment of local biodiversity value within a predominantly agricultural landscape in Canterbury, New Zealand. Bird diversity was highly significantly correlated with landscape heterogeneity, distance from rivers and the Christchurch central business district, altitude and average annual household income, indicating that remotely sensed landscape heterogeneity is a good predictor of local biodiversity patterns. We discuss the advantages and limitations of using geographic information systems to determine local areas of high conservation value.

Introduction

Humans have converted 36% of the Earth's land surface area to agriculture at the expense of natural habitats (Morris 1995). As a consequence, land use change is expected to be the primary driver of population reduction and species loss for the foreseeable future (Vitousek 1994). Because land use change and landscape context are critical determinants of population viability (Hanski and Ovaskainen 2000; Vandermeer and Carvajal 2001), it is important to move beyond simply protecting natural habitat remnants, to a better recognition of the role that highly modified landscapes play in maintaining native biodiversity (Pimentel et al. 1992; Moguel and Toledo 1999; Jensen 2001; Dolek and Geyer 2002). As Novacek and Cleland (2001, p. 5468) point out, "we are obviously past any point where strategies that focus on conservation of 'pristine' habitat are sufficient for the job". There is a clear need to integrate the conservation values of agricultural landscapes into priority management strategies.

Frequently, conservation goals and management strategies are accomplished by maximising the diversity and complementarity of species within conservation areas of minimal size (Balmford and Gaston 1999). High species richness is one measure that is commonly used to determine priority areas for conservation protection at regional (Wickham et al. 1995; Barnard et al. 1998; Wessels et al. 2000; Sfenthourakis and Legakis 2001; Simonson et al. 2001) and global (Williams et al. 1997; Myers et al. 2000) scales, despite the obvious problem that diversity is not always a good surrogate for uniqueness or conservation value (Prendergast et al. 1993). Clearly, the optimal method for locating these species-rich hotspots is to conduct detailed primary surveys to quantify native species richness (Balmford and Gaston 1999), but this can be costly and time-consuming and is usually only feasible at coarse resolution. In practice, the same economic and logistic constraints that determine the scope and extent of conservation action, also limit the feasibility of conducting surveys. Furthermore, there are lost opportunity costs in the time and money spent identifying priority areas, at the expense of active conservation management.

In managed landscapes, one problem with taking a similar approach to maximising the diversity and complementarity of species is that native species are often much rarer and more widely dispersed than in natural landscapes. Consequently, conservation priorities for managed landscapes might well have to be resolved at local, rather than regional scales. Unfortunately, the direct and indirect costs of primary surveys become exponentially greater with increasing spatial resolution, making it difficult for conservation managers or urban planners to obtain empirical data for setting local conservation priorities within regions. Here, we assess the degree to which remotely sensed landscape diversity metrics can be used in combination with other digital data sets to overcome this problem in identifying areas of high native species richness at a local scale.

A common strategy to minimise the costs of identifying regions of high species richness has been to establish whether there are correlations between landscape structure, plant diversity and animal diversity, with the goal of using an easily sampled landscape metric as a predictor of overall biodiversity (Crisp et al. 1998; Duellie and Obrist 1998; Howard et al. 1998; Simonson et al. 2001). Debate over the degree of spatial autocorrelation in taxon diversity continues to be the subject of numerous empirical (Oliver et al. 1998; Tardif and DesGranges 1998; Allen et al. 1999; Barker and Mayhill 1999; Allen et al. 2001; Negi and Gadgil 2002; Vessby et al. 2002; Hawkins and Porter 2003) and experimental (Siemann et al. 1998; Haddad et al. 2001) studies. Most investigations find positive correlations between the diversity of different taxa (Duellie and Obrist 1998; Howard et al. 1998; Lawton et al. 1998; Niemelä and Baur 1998; Blair 1999; Allen et al. 2001), but the correlations are often weak and, more importantly, the spatial locations of hotspots for different taxa frequently do not overlap (Prendergast et al. 1993). Despite these apparent problems, several recent studies have attempted to use remotely sensed landscape heterogeneity as a general predictor of biodiversity hotspots. Surprisingly, perhaps, there has been a remarkable degree of success in establishing relationships between landscape diversity and the diversity of plants (Debinski et al. 1999; Lehmann et al. 2002; Luoto et al. 2002; Moser et al. 2002), invertebrates (Debinski et al. 1999; Cowley et al. 2000; Fleishman et al. 2001; Kerr et al. 2001; Mac Nally et al. 2003) and birds (Nøhr and Jørgensen 1997; Debinski et al. 1999) at regional scales. Most recently, Hope et al. (2003) showed socioeconomics to be an important determinant of urban plant diversity. We extend the approaches taken in these studies to examine the possibility of using remotely sensed landscape heterogeneity in combination with a socioeconomic indicator as a tool for rapidly assessing local biodiversity value in heavily modified, managed landscapes in New Zealand.

For local-scale analyses, limitations on spatial resolution (Tobalske and Tobalske 1999) and the degree to which different land-use types can be differentiated with remotely sensed data, restrict the accuracy of geographic information system (GIS) analyses in conservation management. Furthermore, local landscape diversity assessments are likely to be particularly sensitive to the availability of precise, up-to-date, spatial land-use information. However, the attraction of using GIS to determine local diversity hotspots is the ease and cost-effectiveness with which it can be employed, relative to the alternative of costly and time-consuming primary surveys. Such an advantage makes GIS a powerful tool for conservation managers and urban planners faced with the challenge of prioritising local conservation areas.

In New Zealand, the Land Cover Database (LCDB) maps land-use types from a nationwide, satellite-derived data set that is updated regularly. We used the LCDB to determine spatial patterns in landscape diversity with regard to the dominant natural and anthropogenic features of the environment. The degree to which GIS analyses can be used to predict local biodiversity value was assessed by analysing the spatial congruence between landscape diversity, a socioeconomic indicator and bird species richness.

Methods

Study area

We measured landscape heterogeneity in a representative portion of the coastal Canterbury Plains, South Island, New Zealand (Figure 1A). The area comprised 457,500 ha of highly modified landscape, including Christchurch City, its surrounding suburbs and hinterland containing several rivers, a large area of intensive agriculture, and the mountainous Banks Peninsula. Historically, the area was almost entirely forested before conversion to agriculture (Knox 1969; McGlone 1989). Today, only 51 forest fragments remain as small, isolated patches on Banks Peninsula (total area = 351 ha), in the foothills of the Southern Alps (total area = 193 ha), and Riccarton Bush (7.7 ha) near the centre of the city. The LCDB recognised 14 different land-use types within the study area (Table 1), of which pastoral land was dominant, covering 82 % of the total area (Figure 1B).

GIS analyses

Digital information from the LCDB, NZ Digital Elevation Model (NZDEM) and NZ Statistics Department Survey data were analysed using ArcView 3.2 and Patch Analyst GIS software. The study area was divided into 1×1 km grid squares for analysis.

Biodiversity value within the landscape was estimated by compiling a list of native bird species that utilise (defined as feeding or breeding within) each of the 14 land-use types in Canterbury (Appendix 1). Bird diversity value was assigned to each land-use type according to the total number of bird species within each land use (Appendix 1). The bird diversity value of each grid square was determined using the following equation:

$$\mathrm{BD} = \sum (pA_i \times i)$$

where pA_i is the proportion of grid square area occupied by the ith land-use type and S_i is the number of species that utilise the ith land-use type.

Spatial patterns in bird diversity were investigated in relation to five variables: (1) landscape diversity, (2) altitude, (3) distance from the

Table 1. The 14 land-use types that occurred in the study area, and the area (ha) and proportional area (%) of each.

Land-use	Area (ha)	Area (%)
Pastoral	376,555	82.30
Scrub	28,002	6.12
Urban	14,957	3.27
Planted forest	14,924	3.26
Bare ground	8454	1.85
Tussock	5521	1.21
Urban open space	2746	0.60
Coastal wetlands	2719	0.59
Inland water	994	0.22
Horticulture	987	0.22
Coastal sands	793	0.17
Indigenous forest	576	0.13
Inland wetlands	306	0.07
Mines and dumps	10	0.00
Total	457,545	100.00

Horticulture includes market-gardens, pastoral includes all exotic grasslands, bare ground includes riverine gravel beds and urban open space includes parks and playing fields.

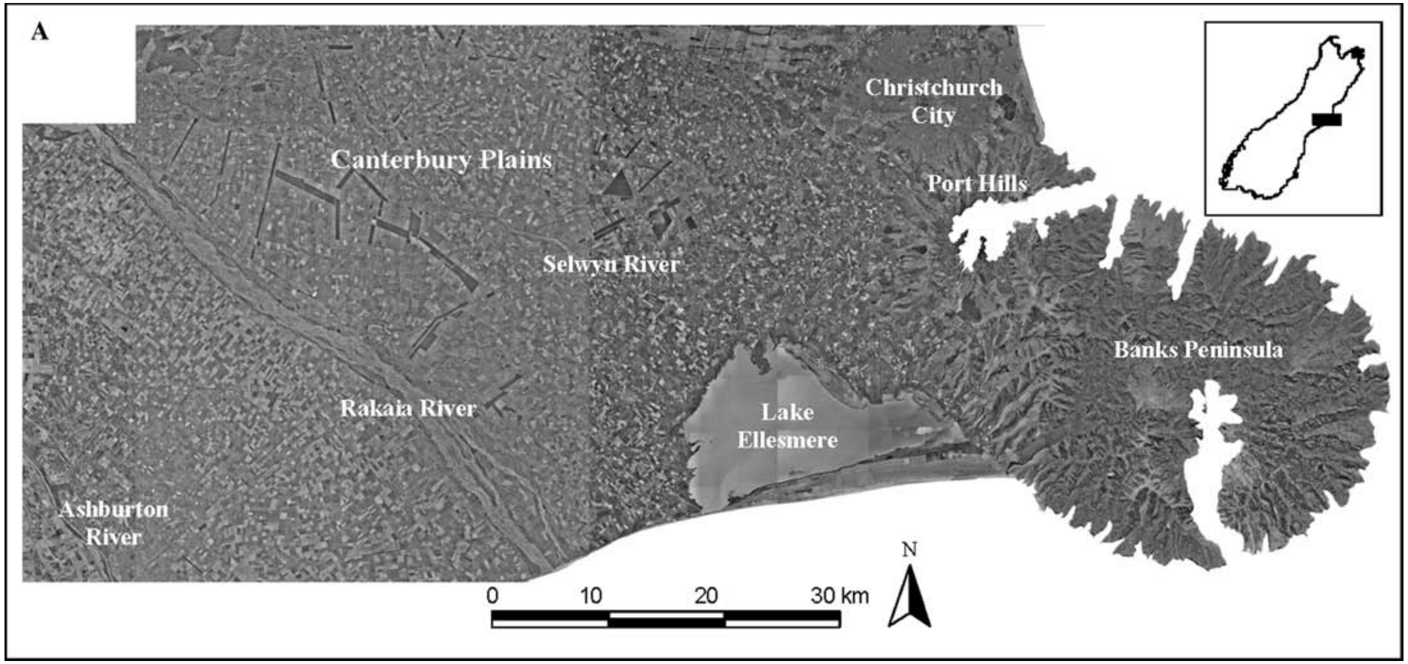

Figure 1. (A) Aerial photograph of the study area showing major natural and anthropogenic features; (B) spatial distribution of land-use types from the Landcover Database. The concentric circles radiate from the Christchurch Central Business District (solid circle) at 10 km intervals.

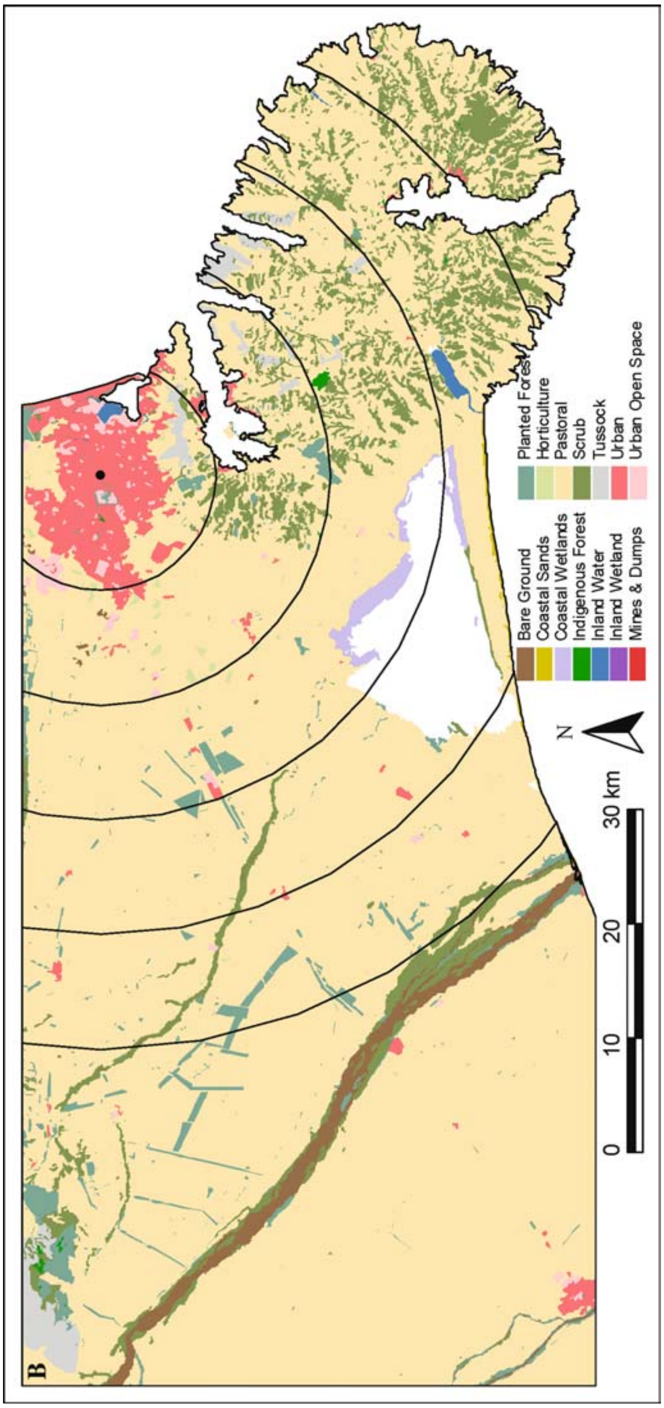

Figure 1. Continued.

Christchurch Central Business District (CBD), (4) distance from rivers and (5) average household income. First, landscape diversity was calculated within each grid square using the Shannon (Shannon and Weaver 1949) and Simpson (Simpson 1949) diversity indices. Second, average altitude (m) was obtained from the NZDEM. Elevation within the study area varied from sea level to 930 m. Third, distance from the centre of each grid square to the CBD and the nearest river (km) were calculated using Patch Analyst. Finally, average household income (NZ $ p.a.) per grid square was derived from the most recent (2001) national survey data. Values for average household income per grid square varied from $ 0 (no households within grid) to $ 92,000 p.a.

Data analysis

Variables with non-normal distributions were converted to rank data for analysis. To account for spatial autocorrelation between adjacent grid squares, we included grid square latitude and longitude as covariates in all analyses. Values of the Shannon and Simpson indices were compared with a Spearman rank correlation. Relationships between bird diversity and the five predictor variables were tested with linear regression.

Results

The two landscape diversity indices were highly significantly correlated with each other (Spearman rank correlation: $r = 0.999$, $df = 4821$, $p < 0.001$). Given the high degree of concordance in index values, we restricted further analyses to Shannon landscape diversity. Spatial patterns in Shannon landscape diversity were apparent with regard to three variables (Figure 2A). First, diversity in the city centre was extremely low, but a strong peak in landscape diversity was apparent at 10 km from the CBD. The surrounding, predominantly agricultural landscape, exhibited intermediate-to-low diversity. Second, land-use diversity was high along rivers and declined rapidly over a distance of 4–5 km. Third, high land-use diversity was observed near sea level and at medium altitudes, whereas the low-lying Canterbury Plains had very low diversity.

Areas with highest bird diversity were apparent in estuaries and in the suburban green-belt of Christchurch city (Figure 2B). Bird diversity was lowest along the major rivers and in the CBD. The monocultural, pastoral regions that dominated much of the Canterbury Plains had moderate values of bird diversity.

Significant relationships existed between bird diversity and all five GIS predictors (Table 2). The model explained a total of 53% of the variation in bird diversity, with landscape diversity alone explaining 42%. Surprisingly,

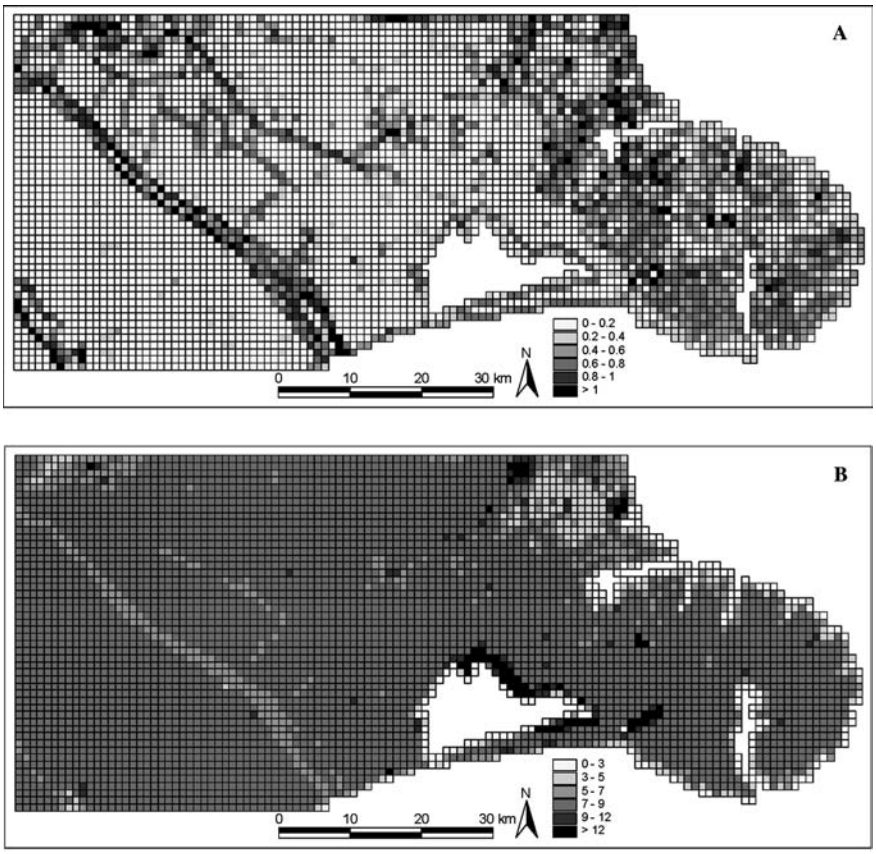

Figure 2. Diversity indices in the landscape: (A) values of the Shannon landscape diversity index; (B) values of the Bird Species diversity index.

Table 2. Results of multiple regression of bird diversity on Shannon landscape diversity, altitude, distance from Christchurch Central Business District (CBD) and nearest river, and average household income. Model $r^2 = 0.533$.

Variable	Df	SS	MS	*F*-value	*p*
Rank(Shannon)	1	2,972,593,370	2,972,593,370	4200.96	< 0.0001
Rank(river)	1	469,357,932	469,357,932	663.31	< 0.0001
Rank(CBD)	1	324,588,576	324,588,576	458.72	< 0.0001
Income	1	16,112,493	16,112,493	22.77	< 0.0001
Rank(altitude)	1	3,284,764	3,284,764	4.64	0.03
Residual	4690	3,318,637,401	707,599		
Total	4695	7,104,574,536	3,786,644,734		

bird diversity and Shannon landscape diversity were negatively correlated, as were distance to CBD and household income. Distance to river and altitude were positively correlated with bird diversity.

Discussion

Remotely sensed landscape variables were highly significantly correlated with bird diversity at the local scale relevant to conservation managers and urban planners. The amount of variance explained by the model was comparable to that found in other studies of a similar nature (Nøhr and Jørgensen 1997; Anderson 2001), indicating GIS analyses can successfully characterise areas of high local biodiversity in dissimilar environments. Therefore, GIS analysis provides a fast, cost-effective predictor of local biodiversity hotspots that may be appropriate for targeted conservation efforts.

Surprisingly, our analysis showed bird and landscape diversity to be negatively correlated, indicating that uniform landscapes apparently have higher biodiversity value than heterogeneous landscapes. This is because the Canterbury region is dominated by pastoral land, which supports a relatively large number of native species compared to some other urban or industrial land uses (Appendix 1), so the majority of homogeneous grid squares had moderately high bird diversity. Although natural habitats such as wetlands and indigenous forest supported considerably more species, these habitat types occurred rarely in the landscape and were typically small, isolated patches that were unable to support the full complement of species known to utilise them. Therefore, grid squares containing natural habitats also contained larger areas of species-poor landuses, leading to low average bird diversity in grid squares with high landscape heterogeneity. This result does not mean that we are suggesting pastoral lands are more appropriate sites for conservation action than natural habitats. An important component of biodiversity value that was not measured in this study is beta diversity, or species turnover between sites. It is reasonable to assume that beta diversity would be much higher between heterogeneous grid squares containing natural habitats than between uniform grid squares composed entirely of the same pastoral land use.

Within the Canterbury region, local areas of high biodiversity value were located in estuaries and some city suburbs. Both landscape and bird diversity were surprisingly high within a zone between 7–13 km from the Christchurch city CBD. This was probably due to the intersection of human-modified habitats with natural and semi-natural habitats such as coastal sands, lakes and wetlands in the city ‘green-belt’, which together support large numbers of native bird species. The high diversity value of New Zealand city suburbs has also been shown for native plant and beetle diversity. For example, Given and Meurk (2000) found that Christchurch city supports a minimum of 350 native vascular plant species (out of a total flora of ca. 2500 species) and an exhaustive survey by Kuschel (1990) recorded 753 native beetle species in a single suburb of another New Zealand city, Auckland (out of a total fauna of ca. 5000 species). In contrast, agricultural monocultures typically support a depauperate native flora and fauna, as highlighted by Sivasubramaniam et al. (1997) who found just 44 species of beetles in Canterbury carrot fields, and Kuschel (1990) and Harris and Burns (2000) who found that agricultural landscapes in the North Island supported predominantly exotic beetle species and few natives.

Bird diversity was negatively correlated with annual household income. A recent study by Hope et al. (2003) showed income to be positively correlated with plant genera richness because native plants were replaced with a larger number of exotic genera in residential gardens. If the trend described by Hope et al. (2003) is also true of New Zealand cities, we would expect to see a decline in bird species richness with increasing income as native animal diversity is typically positively correlated with the amount of native vegetation (Crisp et al. 1998).

Although the strong relationship between landscape and bird diversity implies a high degree of predictive power, two problems are apparent in using remotely sensed landscape diversity as a surrogate for bird species diversity. First, the method emphasises areas with a high proportion of edge habitat, where two land uses are immediately adjacent to one another. Grid squares with the highest landscape diversity occur where the edges of several land-use types occur in close proximity. Given that ecosystem dynamics are altered at habitat edges (Murcia 1995; Fagan et al. 1999) and some native species are obligate core-dwellers, any bias toward edge habitat is a serious problem for conservation managers. However, in highly modified landscapes like Canterbury, the problem may be minimal relative to that experienced by managers working in natural landscapes. Managed landscapes typically support a large proportion of invasive species (Lonsdale 1999), whereas many native species survive in remnant natural habitats (Kuschel 1990). Of the native biodiversity that survives in managed habitats, most are either generalist species that are tolerant of different land uses or highly dispersive species that are able to cross areas of unsuitable land use in search of favourable habitat patches. So, the predominance of edge habitat in modified environments is unlikely to further reduce native biodiversity.

The second problem associated with using remotely sensed surrogate measures of biodiversity is that the LCDB is derived from satellite imagery using 100×100 m resolution. Consequently, fine scale habitat heterogeneity is not adequately represented in the data set. For example, recent efforts by the Christchurch City and Regional Councils to improve biodiversity conservation along riparian strips within the city bounds (Meurk and Swaffield 2000; Environment Canterbury 2001) and the establishment of native shelterbelts (Meurk 2003) are unlikely to appear on future updates of the LCDB, because the width of these strips is typically less than 50 m. This is problematic, given the importance of narrow, linear features such as shelterbelts, hedgerows and road verges in providing breeding habitat and dispersal corridors for birds (Skagen et al. 1998; Estrada et al. 2000), small mammals (de Lima and Gascon 1999; Laurance and Laurance 1999) and invertebrates (Hill 1995; Fournier and Loreau 2001; McLachlan and Wratten 2003) in human-dominated landscapes. Some of these features could be incorporated into the GIS by using digitised 1:50,000 topographical maps of New Zealand. However, the data used in creating the TopoMap series were collected prior to 1989, so many features on the maps may have been removed and others added. Future development of the LCDB can address this problem by increasing resolution and categorising these fine-scale land-use types.

Despite these apparent drawbacks, the LCDB provides a valuable resource for the identification of conservation values at a local scale. Such a rapid method of landscape assessment means that organisations tasked with prioritising areas for conservation action are able to do so without resorting to costly site surveys. The value of the LCDB is further enhanced by the production of regular updates. By repeating analyses using future updates of the LCDB, managers will be able to ascertain areas of rapid changes in land-use diversity. Temporal changes in land use are increasingly being recognised as important drivers of current species composition and diversity patterns (Danielsen 1997; Harding et al. 1998; Grove 2002), so combining landscape history with current landscape composition will provide a powerful, complementary method for targeted conservation action.

Acknowledgements

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Appendix 1. List of native (including self-introduced) bird species from the Canterbury region and the land-use types within which they feed or breed.

Bird species		Land-use type													
Common name	Latin name	Pastoral	Scrub	Urban	Planted forest	Bare ground	Tussock	Urban open	Coastal wetlands	Inland water	Horticulture	Coastal sands	Indigenous forest	Inland wetland	Mines and dumps
Great spotted kiwi	*Apteryx haastii*												×		
Southern crested grebe	*Podiceps cristatus*									×					
Little penguin	*Eudyptula minor*								×			×			
Great cormorant	*Phalacrocorax carbo*								×	×		×		×	
Pied cormorant	*Phalacrocorax varius*								×	×		×		×	
Little black shag	*Phalacrocorax sulcirostris*								×			×			
Little cormorant	*Phalacrocorax melanoleucos*							×	×	×		×		×	
Spotted shag	*Stictocarbo punctatus*								×			×			
White heron	*Egretta alba*								×					×	
Reef heron	*Egretta sacra*								×			×			
White-faced heron	*Ardea novaehollandiae*	×							×	×		×		×	
Australasian bittern	*Botaurus poiciloptilus*													×	
Royal spoonbill	*Platalea regia*								×	×				×	
Paradise shelduck	*Tadorna variegata*	×						×		×				×	
Blue duck	*Hymenaolaimus malacorhynchos*									×			×		
Grey duck	*Anas superciliosa*							×	×	×		×		×	
Grey teal	*Anas gracilis*							×		×				×	
New Zealand scaup	*Aythya novaeseelandiae*							×	×	×				×	
Australasian shoveler	*Anas rhynchotis*								×	×				×	
New Zealand falcon	*Falco novaeseelandiae*					×	×						×		
Australasian harrier	*Circus approximans*	×	×		×		×					×	×	×	
Marsh crake	*Porzana pusilla*													×	
Pukeko	*Porphyrio* sp.	×												×	

Pied oystercatcher	*Haematopus ostralegus*	×							×			×		×	
Variable oystercatcher	*Haematopus unicolor*								×			×			
Pied stilt	*Himantopus himantopus*					×			×					×	
Spur-winged plover	*Vanellus miles*	×					×	×	×					×	
Black-backed gull	*Larus dominicanus*	×		×		×		×	×			×		×	×
Red-billed gull	*Larus novaehollandiae*	×		×		×		×	×	×		×		×	×
Black-billed gull	*Larus bulleri*								×	×		×			×
White-fronted tern	*Sterna striata*								×	×		×		×	
Black-fronted tern	*Sterna albostriata*					×			×	×		×		×	
Caspian tern	*Sterna caspia*								×	×		×	×	×	
Kereru	*Hemiphaga novaeseelandiae*							×					×		
Kaka	*Nestor meridionalis*												×		
Kea	*Nestor notabilis*						×								
Red-crowned parakeet	*Cyanoramphus novaezelandiae*												×		
Yellow-crowned parakeet	*Cyanoramphus auriceps*												×		
Shining cuckoo	*Chrysococcyx lucidus*		×		×			×					×		
Long-tailed cuckoo	*Eudynamys taitensis*												×		
Morepork	*Ninox novaeseelandiae*														
Sacred kingfisher	*Todiramphus sanctus*							×	×			×	×	×	
Welcome swallow	*Hirundo tahitica*	×		×				×			×				
Rifleman	*Acanthisitta chloris*												×		
New Zealand pipit	*Anthus novaeseelandiae*	×				×	×								
Brown creeper	*Mohoua novaeseelandiae*		×										×		
Grey warbler	*Gerygone igata*		×		×			×			×		×		
Grey fantail	*Rhipidura fuliginosa*		×		×			×			×		×		

Appendix 1. Continued

Bird species		Land-use type													
Common name	Latin name	Pastoral	Scrub	Urban	Planted forest	Bare ground	Tussock	Urban open	Coastal wetlands	Inland water	Horticulture	Coastal sands	Indigenous forest	Inland wetland	Mines and dumps
Yellow-breasted tomtit	*Petroica macrocephala*												×		
South Island robin	*Petroica australis*												×		
Silvereye	*Zosterops lateralis*		×	×	×			×			×		×		
New Zealand bellbird	*Anthornis melanura*		×		×			×					×		
Tui	*Prosthemadera novaeseelandiae*												×		
Total		10	7	4	6	6	5	16	24	17	4	19	21	24	3

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