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A spatial analysis of indigenous cover patterns and implications for ecological restoration in urban centres, New Zealand

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Abstract High levels of endemism, the sensitivity of species that have evolved without humans, and the invasion of exotic species have all contributed to severe depletion of indigenous biodiversity in New Zealand. We considered the contribution that urban restoration can make to maximising biodiversity by analysing landcover patterns from two national databases along an urban–rural gradient. Thirteen of 20 land environments in New Zealand are represented in cities, and nearly three-quarters of all acutely threatened land environments are represented within 20 km of city cores nationally. Despite this, remaining indigenous landcover is low within urban cores, with less than 2% on average, but increasing to more than 10% on average in the periurban zone. Threatened lowland environments are most commonly represented within cities, and least represented within protected natural areas. Restoration of existing urban habitat is insufficient to halt biodiversity loss. Ecosystem *reconstruction* is required to achieve a target of 10% indigenous cover within cities. A co-ordinated national urban biodiversity plan to address issues beyond a local and regional focus is needed. Analysis of national patterns of urban land environments, indigenous cover and remnant ecosystems will support action at a regional and local level while enhancing national and global biodiversity goals.

Keywords Biodiversity · Urban · Periurban · Restoration · New Zealand

Introduction

Nearly half the human population is now urbanised (Crane and Kinzig 2005), and in New Zealand, a southern hemisphere archipelago and biodiversity hotspot (Mittermeier et al. 1999),

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more than 78% of the population are urbanised (Statistics New Zealand 2005). Ecological research on species loss frequently focuses on human mediated impacts such as the effects of invasive species (e.g.,Vitousek et al. 1997; Hobbs and Mooney 1998; Sax and Gaines 2003) and habitat modification and fragmentation (e.g., Hobbs and Mooney 1998; Marzluff and Ewing 2001; Williams et al. 2005; Parris 2006). Habitat destruction is the primary environmental cause of biodiversity decline at local, regional, and global scales (Dobson et al. 1997). However, some researchers have also noted a cycle of biological impoverishment and human apathy to biodiversity loss in affluent urbanised societies, initiated by the homogenisation and reduction in local flora and fauna (Blair 2001; Marzluff and Ewing 2001) and the 'extinction of experience' (Miller 2005) although others have debated this view (Mehtala and Vuorisalo 2006). Researchers have argued that interactions with nature in close proximity to places where people live and work can strengthen human connections to the natural world (Miller 2005). Indeed, Crane and Kinzig (2005) suggested that what remains of habitats and biodiversity in the city is of disproportionate importance. On one level, protection of fertile lowland ecosystems where many cities occur is often lacking, with nature reserves biased towards other kinds of wildland ecosystems (e.g., Scott et al. 2001).Yet urban environments offer opportunities to reverse ecosystem degradation and biodiversity loss in a meaningful way through human engagement in ecological restoration.

The impact of urbanisation is partly a function of the original composition of the landscape (Dunning et al. 1992). We need, then, to further examine the relationship between biodiversity and the urban environment. The challenge for conservation is to manage these complex landscapes in a way that retains and enhances biodiversity value (Sutherland et al. 2006). An important first step is to quantify urban landscape patterns including patterns of indigenous landcover that can be used as a surrogate for biodiversity. Species richness indices may not reflect the quality of indigenous biodiversity: many studies do not specify the relationship, or the desired relationship, between indigenous and introduced biodiversity. In New Zealand, however, this difference is of critical importance, as adventive introduced species are now as numerous as native species (Lee et al. 2000; Sullivan et al. 2005). There is widespread recognition that spatial patterns affect ecological processes (Pickett and Cadenasso 1995; Wu and Hobbs 2002). Further, the importance of restoration across whole landscapes rather than at particular sites has also been highlighted as an issue (Hobbs and Mooney 1998).

Case studies form the vast majority of urban ecology research (e.g., Luck and Wu 2002; Baker et al. 2003; Grimm and Redman 2004; Stewart et al. 2004). Biodiversity data has rarely been analysed at a broad scale to determine how restoration efforts can best benefit threatened ecosystems. A variety of ecological studies (e.g., McIntyre and Hobbs 1999; Drinnan 2005) suggest that below 10% relictual habitat cover in a landscape may trigger a decline in many species with severe fragmentation effects, although other researchers suggest even greater cover is required (e.g., Rutledge 2003). As well, our understanding of the nature of threshold effects, where they lie and detecting their approach is limited. Ecosystems may respond to biotic and abiotic pressure in a non-linear, stepwise manner (Balmford and Bond 2005). Although the issue may be further confused by considerations of scale (e.g., Turner 2005), it is useful to examine the consequences of a 10% threshold in relation to urban indigenous landcover and urban landscape ecology.

Both population and species loss can be driven by land transformations associated with urbanisation (e.g., McDonnell et al. 1997; Luck and Wu 2002) yet many cities plan independently for their perceived biodiversity needs without reference to national patterns. The current upsurge of urban restoration activity in New Zealand suggests it is timely to consider the potential of urban centres to successfully contribute to *national* biodiversity goals. Analysing patterns of urban indigenous cover, urban land environments and restoration efforts in relation to national biodiversity is an essential part of that planning process.

In this research, we investigated land environment patterns across New Zealand cities to understand their differentiation in relation to geographic and climatic/abiotic factors. A second key aim was the identification of the degree of remnant indigenous vegetation (measured as indigenous cover) in and around New Zealand cities. Further, we assessed the size and extent of these indigenous cover fragments. This allowed us to determine the persistence of ecological features in the urban core and surrounding areas. These cities are small by world standards, ranging in population size from 40,000 to 400,000, but are typical of the size of many cities which will rapidly expand within the next 50 years. An analysis of both the existing resource potential within the urban core, and within buffer zones around this core, allowed us to consider the resource potential which can potentially be incorporated within the city as it expands. New Zealand cities are highly variable in both landform and level of biodiversity resource. We have set out to determine the range of urban native ecosystems that are (or can potentially be) represented within the broader environmental profile of New Zealand, and hence the restoration potential of New Zealand cities.

Methods

Two databases were analysed—The Land Environments of New Zealand (LENZ) database (Leathwick et al. 2003) developed by Landcare Research, and the Landcover Database 2nd edition (LCDB(2)) data set (Terralink 2004). LENZ uses environmental variables such as rainfall, water vapour pressure, mean annual temperature and minimum annual temperature to build a picture of the land environment which existed prior to human settlement in around 1200 A.D. It has four levels which vary from the general to the detailed. LENZ level 1 is useful for identification of general patterns of which land environments are represented in cities, while LENZ level 4 is the most detailed level of analysis which is most appropriate for environmental profiles and the examination of land environments in cities.

LCDB(2) is a database which quantifies current landcover across New Zealand. It is derived from LANDSAT 7 imagery with a 15 m pixel resolution. We have used data on indigenous landcover from this database as a surrogate for biodiversity. The Landcover classification has approximately 50 classes, such as broadleaved indigenous hardwood, tall tussock grassland, and subalpine shrubland. A generalised landcover data set was developed from the initial landcover data set by combining classes. Appendix 1 lists these generalised classes and describes how these classes were generalised from the initial LCDB (2) data set. Although there are some discrepancies and incorrect assignment of LCDB(2) classes (mainly confusion of natural and planted forest) this database is the best available in New Zealand. We have focused on general trends in land categorisation rather than absolute values, so that the requirement for data accuracy is more relaxed than might otherwise be the case. We therefore consider the results robust.

Urban areas were initially identified using Statistics New Zealand's urban regions with a population greater than 40,000 based on the 2001 population census. This compares with American models of urban areas which may have a unit of 50,000 people as an integral part of the definition; or a density of more than 3.8 people per hectare as defined by the US Census Bureau (Theobold 2004). Twenty cities were identified for analysis (Fig. 1). We then classified urban as an urban area based on a contiguous algorithm to construct an

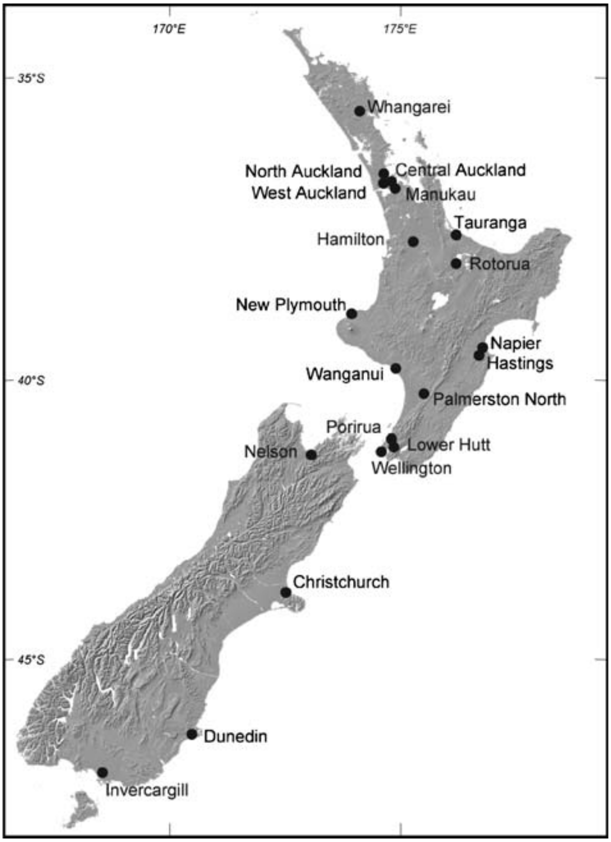

Fig. 1 Spatial distribution of the 20 largest New Zealand cities included in this analysis of urban biodiversity

'urban core', to allow us to measure urbanisation in space in a similar way to McDonnell et al. (1997). The aim was to compare structurally/ecologically equivalent parts of the city rather than just variable historical administrative units, hence precluding the use of city boundaries.

The 'urban' area defined by the city boundary is administered by city councils and district councils (as defined by their legal obligations in the case of city councils and the District Plan for each district council) and may include large amounts of rural land. These are not useful for analysis of urban–rural gradients. However, complex heterogeneous habitats such as cities can be usefully ordinated along conceptual 'urban–rural'gradients (McDonnell and Pickett 1990). Our approach was initially one of data exploration, to investigate the connectedness of the urban and periurban areas. In this case, the analysis involved identifying the contiguous urban core, and the surrounding rural areas of each city using a range of proximity (buffer) distances—0, 5, 10, and 20 km (Fig. 2). Each buffer region extended from the inner core to the outer limit of 5, 10 or 20 km respectively, thus encompassing the closer buffer zones. Small satellite urban areas associated with a city can distort urban analyses because such areas add large buffer regions: to prevent this distortion, only the core part of the urban area was used. This deleted small urban areas that were not contiguous to the core urban area. The contiguous urban core (also referred to as the 0 km buffer zone) was defined as an urban area over 300 ha, and was derived by deleting urban patches less than 300 ha in size. The four buffer regions of an urban area did not extend into

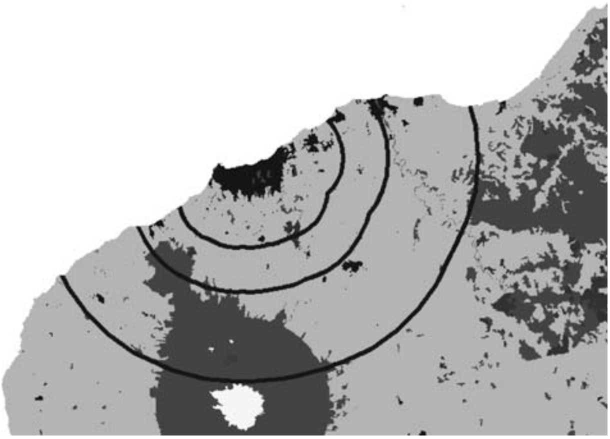

Fig. 2 Centroid buffer zones of 5, 10, and 20 km around urban cores were used to analyse urban biodiversity in the 20 largest New Zealand cities. In this case, the buffer zones are shown for New Plymouth (39.04S, 174.05E), with the urban core shown in *black*. A substantial part of Mt Egmont National Park is included in the southern part of the 20 km buffer zone to the *bottom* of the picture

the buffer region of a neighbouring urban region to prevent double counting of LENZ or LCDB (2) classes. These buffers defined the analysis area for summarising the LENZ classifications (levels 1, 2, 3, and 4), LCDB (2) classification and the generalised version of LCDB(2). Statistics on the extent and proportions of the different LENZ classes and LCDB(2) classes were produced. The analysis area varies between the LCDB(2) and LENZ summaries because LENZ excludes the sea and internal water bodies (e.g. lakes). Internal water bodies in the LENZ classification are represented by "NULL." STATISTICA was used to compile and analyse descriptive statistics and to carry out a cluster analysis of land environments in New Zealand cities (StatSoft Inc. 2006). This analysis is based on average distance between linkages using the Manhattan city block index of similarity (StatSoft Inc. 2006).

Results

Underlying urban land environments

Thirteen of 20 land environments at the most generalised level of LENZ occur within the urban cores of the largest 20 cities. Most cities have more than one land environment represented, with the most common number of land environments being two or three ($\bar{x} = 2.8$, range 1–5; Table 1). Nonetheless, one land environment frequently tends to predominate in each city, so that, for example, the northern lowlands (A) form the most commonly found environment with more than 90% of the land area in each of the seven northern cities. The central hill country and volcanic plateau (F) and western and southern north Island lowlands (C) are also well represented in urban centres. On the other hand, the central dry foothills (E) occur in only one city, and also occupy the smallest area, while environments L (the southern lowlands), N (the eastern South Island plains), and Q (the southeastern hill country and mountains) are similarly sparsely represented in an urban setting (Table 1).

Analysis of urban land environments shows four main groupings at a linkage distance of 1,000, and the effects of latitude can be clearly seen in these groupings (Fig. 3). Although Christchurch has the most unique set of land environments and is alone in group one, this is largely due to the large representation of environment N that occurs in only one other city (Dunedin, in small amounts). The second group comprises mainly southern and central cities.

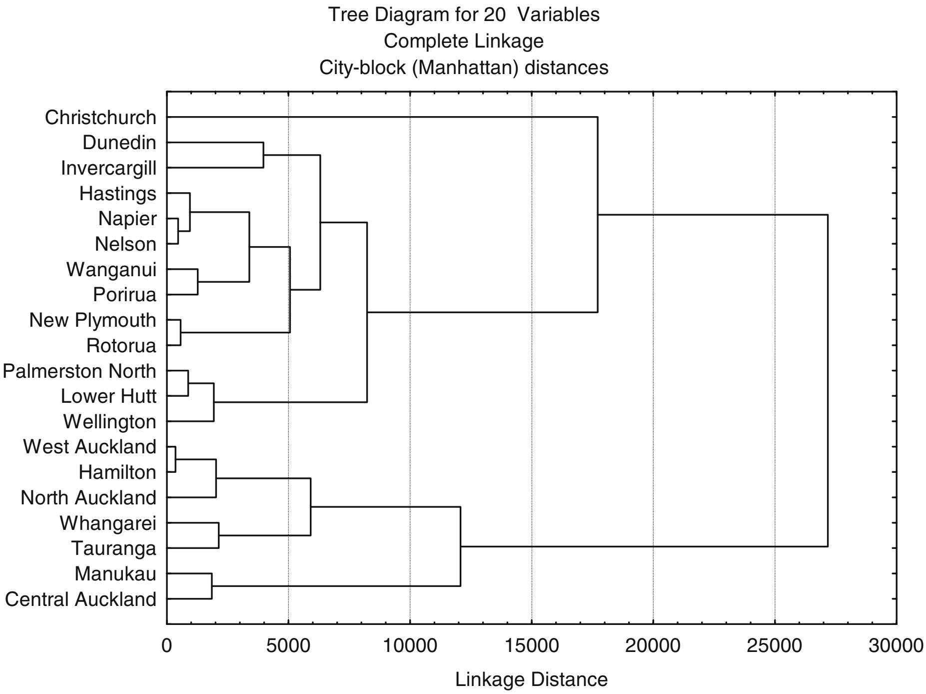

Fig. 3 Dendrogram using city-block (Manhattan distances) and complete linkages to analyse the land environment relationships in the 20 largest New Zealand cities from the LENZ Level 1 database

Northern cities are represented in the third and fourth groups; West Auckland and Hamilton cities are closely grouped because of the large amount of environment A in both. Collaborative biodiversity strategies could be useful where such similarities exist; for example, between Hamilton (which has low indigenous cover in all urban buffer zones) and West Auckland. The range and diversity of environments evident in Table 1, however, also emphasises the need for each city to put in place individual solutions to biodiversity retention.

A more detailed analysis of land environments at Level IV of LENZ shows that 100 of 500 identified land environments lie beneath core urban areas, increasing to 181 in the 5 km buffer zone, and 214 in the 10 km buffer zone. Using the five threat categories for indigenous land environments in New Zealand that have been identified (acutely threatened, chronically threatened, at risk, critically underprotected, underprotected, and a further category of no threat (Walker et al. 2005), the urban cores comprise 63 different acutely threatened land environments (which make up 66% of the land area) and 13 chronically threatened environments. Only 10 of the 100 land environments in the urban core are regarded as having 'no threat'.

Within the 5 km buffer zone around the urban core, the number of acutely threatened land environments increases to 83, representing 52.5% of all acutely threatened environments on a national scale (Table 2). Further, the number of acutely threatened land environments represented continues to increase with buffer zone distance around the urban cores (although they form the highest proportion of the land environment within the urban cores). Sixty acutely threatened land environments (38% of all acutely threatened land environments) have more than 10% of their potential land area within the total urban and periurban area defined at

Table 1 Total land area (hectares) of the underlying land environments represented beneath the core urban areas (n=20)

	No. land environments	A	B	C	D	E	F	G	H	I	J	L	N	Q
Whangarei	3	1,517			21			3						
North Auckland	1	7,403												
West Auckland	2	5,597			219									
Central Auckland	2	13,559			48									
Manukau	3	11,844			39			128						
Tauranga	1	3,626												
Hamilton	3	5,560					25	73						
Rotorua	5	186		96			2,456	27	18					
New Plymouth	5	3			41		2,228	20	6					
Napier	2		2,183							96				
Hastings	2		1,732								550			
Wanganui	4		513	1,388			5				13			
Palmerston North	3			2,803						19	5			
Porirua	2			874			236							
Lower Hutt	3			2,669			664		55					
Wellington	2			2,870			1,837							
Nelson	3		1,972			36					121			
Christchurch	5		110				280			484	971		11,722	
Dunedin	3											2,589	291	646
Invercargill	2											354		2,092

Thirteen of 20 land environments are represented across all 20 cities. Water bodies are excluded from the analysis. Cities are arranged approximately north to south. The first 16 cities are all in the North Island. Blank cells indicate a zero value for that land category. A full list of descriptions for land environment categories can be found in Leathwick et al. 2003 (see Appendix 2)

its outer limits by the 20 km buffer zone. Twenty-two acutely threatened environments have more than 50% of their possible area represented within these urban and periurban zones, and six acutely threatened land environments have more than 90% of their total land extent within 20 km of the urban cores of cities. The high proportion of acutely threatened land

Table 2 Patterns of indigenous biodiversity presence and threat in relation to buffer zones at increasing distances from the urban core of urban centres, across all cities (n=20)

Distance from urban core (km)	0	5	10	20
No. acutely threatened land environments (LENZ IV)	63	83	93	114
Acutely threatened land environments as a percentage of the total area in the buffer zone	66.2	58.7	53.5	47.4
Indigenous cover (%). Mean ± SE	1.96±0.5	9.9±2.6	12.8±3.2	15±2.8
Actual remaining indigenous cover on acutely threatened land environments (ha)	598	7,467	14,136	34,798
Total no. of indigenous cover types (richness)	5	10	11	14
Mean no. indigenous patches ± SE	42±9	196±28	373±52	864±149

environments indicates enormous potential to contribute to the protection, restoration and reconstruction of threatened environments in cities.

Despite human modification of land environments in cities, 598 ha of indigenous cover currently exists on these acutely threatened land environments within the urban cores (an average of 29.9 ha per city), and nearly 35,000 ha remains within a 20 km buffer zone around the urban core. For some acutely threatened environments such as I3.2b and I5.2a, which have more than 90% of their potential land environment within the 20 km buffer zone of cities, the remaining indigenous cover is in critical need of protection (consisting of only 12 and 11 ha indigenous landcover respectively). Other land environments with more than 90% of their potential extent within the 20 km urban buffer zone fare slightly better, but prompt action to protect these remnants is warranted. That is, although indigenous cover on these land environments is relatively low, the vast majority of land environments have some remaining indigenous biodiversity. Only 9 of the 114 acutely threatened land environments within 20 km of the urban core of cities have no indigenous cover remaining in this zone.

Remaining indigenous landcover in the urban and periurban zones

The percentage of indigenous cover remaining within the urban core of cities ranged from 0–8.9% ($\bar{x} = 1.96\%$; Fig. 4). As expected, the amount of remaining indigenous cover increased with distance from the urban core, with the greatest amount of indigenous cover, on average, in the 20 km zone. Many New Zealand cities are coastal, and water is hence also represented strongly within the buffer zones (Fig. 4). Nine of the 20 largest New Zealand cities reach a threshold of 10% indigenous cover approximately 5 km from the urban core, but at a distance of 10 km from the urban core, only one other city can be added to this list (11 in total if we

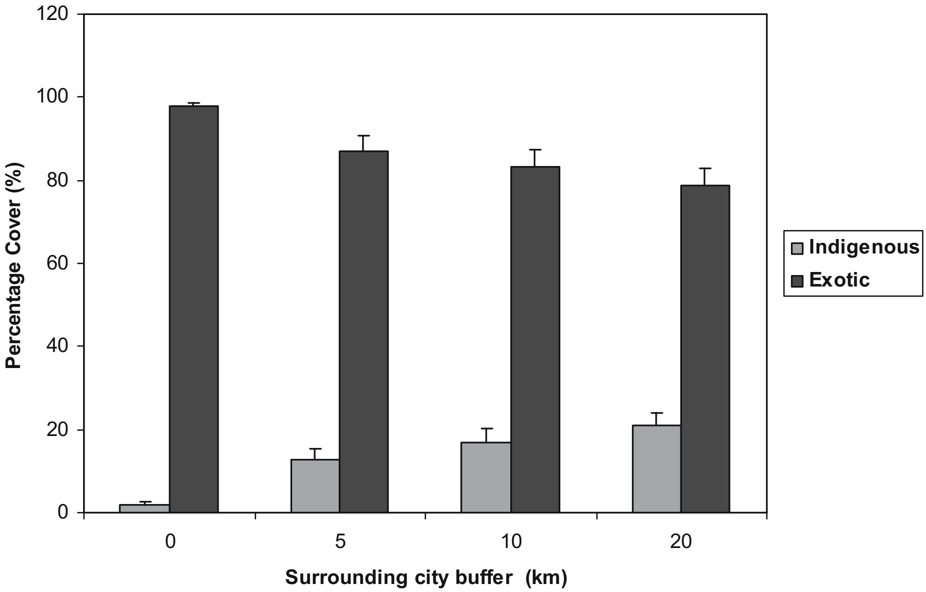

Fig. 4 Mean percentage of indigenous and exotic cover in each buffer zone surrounding the inner urban core of cities. Mean±SE for each zone (n=20)

exclude water bodies). At 20 km from the urban core, only 12 cities have indigenous cover at or above the 10% threshold, although two cities are just below this threshold. If water bodies are excluded, however, 16 cities comfortably reach the 10% threshold 20 km from the urban core.

The pattern of indigenous cover through the urban and periurban zones from the central urban core varies. Three distinct patterns of increasing indigenous cover were revealed along the buffer zone gradient from the urban core to 20 km. Napier is typical of cities with a low relief, farmed periurban zone and little remaining indigenous biodiversity (Fig. 5a). Nelson's indigenous cover is representative of the second pattern; it has a green belt on the outer edge of the city and gradually increasing areas of remnant indigenous ecosystem further away from the city (Fig. 5b). Wellington is typical of cities built in hilly environments around a harbour, with the influence of an inner city green belt, but decreasing indigenous cover within the 20 km buffer zone (Fig. 5c). New Plymouth has a distinct pattern of indigenous cover that did not fit one of the other three patterns (Fig. 5d). New Plymouth is the only New Zealand city with this curve shape—This results from a combination of history and topography with many reserves established in the early 1900s (Clarkson and Boase 1982) and eventually incorporated within the urban centre. In the periurban zone indigenous forest was rapidly and efficiently converted to grassland for dairy farming but a significant forest reserve was created in 1881 in the adjoining upland, the proto Egmont National Park (Clarkson 1986).

The number of indigenous landcover types represented in each zone significantly increased with distance from the city centre, with only five types represented in total in the core of urban centres, increasing to 14 in the 20 km buffer zone (Table 2). These indicate that relatively high diversity can be found within 20 km of the urban core in many cities, although one urban centre has no recorded indigenous vegetation in the urban core at all. Of the indigenous terrestrial landcover, only broadleaved hardwoods and forest occur in more

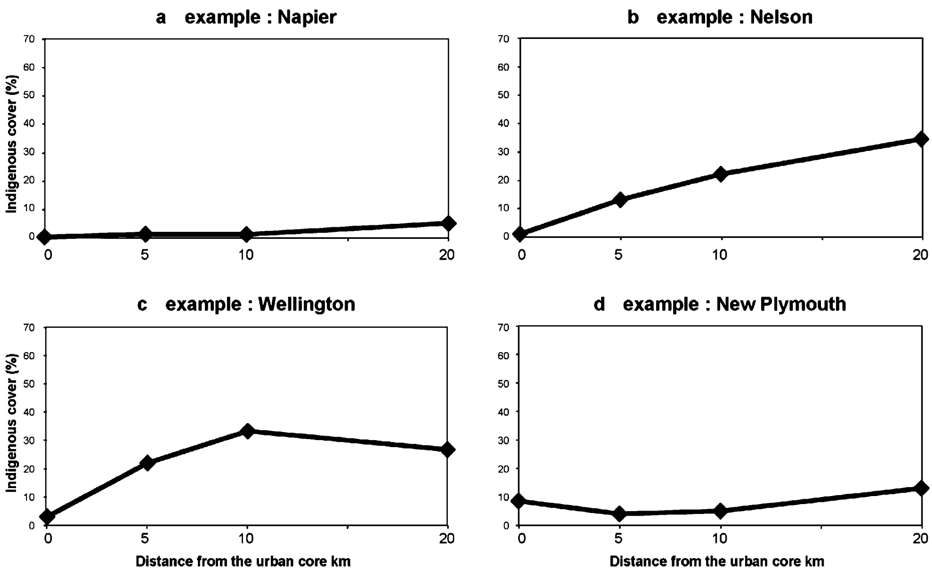

Fig. 5 Patterns of indigenous land cover over a gradient of concentric buffer zones from an urban core through distances of 5, 10 and 20 km. Pattern **a** is represented by six urban centres, **b** by seven centres and **c** by six centres. New Plymouth is the only city with a periurban indigenous cover of this curve type **d**

than half of the urban centres; at the other extreme, flaxland and herbaceous freshwater vegetation are both recorded in only one city each. Unusual indigenous biodiversity is represented within 5 km of the urban cores, with fernland in New Plymouth's environs, tall tussock grassland near Dunedin, and gravel and rock field in that of Lower Hutt. As well, indigenous hardwoods and indigenous forest occur in the 5 km buffer zone of all 20 cities.

In a postal survey of the 20 largest New Zealand cities, city council staff indicated that there is little remaining cover which could be restored to either indigenous or mixed indigenous–exotic cover within cities (Clarkson and Wehi, unpublished data). Yet existing indigenous cover (including that currently under restoration) falls far short of a 10% indigenous biodiversity threshold. Most cities would have to increase indigenous cover dramatically to meet this overall target as they have less than 2% indigenous cover within their urban cores. On average, 395.8 ha of indigenous cover is required for each city to reach such a 10% threshold. There is a great deal of variation within this figure: New Plymouth would have to add only 34.5 ha; a city such as Christchurch which has a severely depleted indigenous environment would need to add 1,365 ha. However, most cities meet an overall 10% indigenous cover target approximately 10 km out from the urban core (although this figure encompasses a number of diverse types of ecosystems).

Discussion

This is the first meta-analysis of indigenous landcover in New Zealand cities and reveals national patterns of landcover which have not previously been identified, including the degree of remnant indigenous landcover in urban centres. Urban centres are situated most often in the lowland coastal environments which are known to have the greatest number of threatened species and environments (Rogers and Walker 2002). The majority of land environments in New Zealand are represented to some degree within cities, including 215 different acutely threatened environments. Three well defined generalised gradient patterns of increasing quantity of indigenous cover occur from the urban core to the periurban areas and the number of indigenous cover types also increases significantly within a 20 km buffer zone around the centre.

Most New Zealand cities occur within the North Island, with most large cities occurring in the North Island northern lowlands environment. The New Zealand flora has a high degree of rarity (ca. 22%; de Lange et al. 1999), and there are unexpectedly large numbers of threatened taxa in the lowland zone (Rogers and Walker 2002). This contrasts with some overseas examples, where cities occur in low productivity lands (Collins et al. 2000). Moreover, 92 taxa are restricted to the northern North Island (Rogers and Walker 2002) where urban growth is fastest. Despite this, these lowland environments have low levels of statutory protection. Walker et al. (2005) found that the risk of habitat loss is highest on unprotected lands. Considering that Hobbs and Mooney (1998) have argued that habitat loss can be used as a surrogate measure for population extinctions, this suggests a poor prognosis for biodiversity in many urban areas in New Zealand.

The high number of landcover types represented within 20 km of cities, and the small extent of indigenous landcover that remains on acutely threatened land environments, suggests that particular care is needed to protect these remnants. In addition, although a recent survey in the Waikato indicates that some ecosystems may have higher demand for protection, such as freshwater rivers and water bodies (Environment Waikato 2006), biodiversity values are not necessarily greater in ecosystem types preferred by urban dwellers. More research on the underlying land environments of cities worldwide could point to trends in biodiversity loss from urbanisation, and accompanying opportunities. As yet the value of natural ecosystems in cities is poorly known, as many cities do not have inventories or databases which quantify these areas. We suggest that active partnerships to manage nature in the city are vital to the persistence of biodiversity. An integrated science of urbanisation is woefully inadequate.

The administrative boundaries between urban centres and adjoining district councils occur at different points along this gradient. This has major implications for the protection and restoration of indigenous landcover, and hence biodiversity, in different cities. Regional councils may currently be responsible for large tracts of indigenous resource which may have an important role in the cities of the future. Conversely, city boundaries frequently include large rural areas which may be intensively used for farming, but may equally include representative indigenous remnants. City boundaries also vary markedly in the extent of actual land area included within them. Moreover, low density rural development in the periurban zone is the fastest growing form of land use in the US since 1950 (Hansen et al. 2005). Real opportunities exist for planners to protect ecological areas with high indigenous biodiversity values, or enable purchase of land suitable for ecosystem reconstruction. Co-operative partnership with regional administrators and planning within a national urban biodiversity framework has the potential to significantly enhance the national biodiversity resource.

Currently, large remnant areas tend to be close to the city boundary, and hence are rarely situated within the urban core. For example, according to a report summarising Palmerston North's biodiversity resource, Palmerston North has only around 300 ha of ecological sites within the city, or less than 1% of the city area, but large forest catchments close to the city boundary (Boffa Miskell 2002). This has implications for future urban growth; these areas with large indigenous patches tend to be in 'dynamic' landscapes where the landscape continues to be modified and communities may be in transition (Hansen et al. 2005). The city centre however, requires a different approach for restoration of biodiversity with its more 'static' landscape and larger degree of fragmentation. It is notable, nonetheless, that several cities have as many as five indigenous landcover types remaining within the heart of the city.

The curve types shown in Fig. 5 suggest that different approaches to urban biodiversity conservation may also be based on links between remaining landcover and topography. The six cities with curve type (a) tend to have limited relief, and five have large agricultural tracts surrounding the urban core. Restoration may therefore be largely limited to stream verges, gully systems, or acquiring land for reconstruction. All those cities with curve type (b) have large tracts of native forest within the 20 km buffer zone, which could potentially be used as a protected breeding source for indigenous species. Cities with curve type (c) tend to be hilly, with indigenous landcover, as well as exotic forest, remaining on steep hills surrounding the cities. Solutions for individual cities should reflect these differences. However, corridors linking large indigenous tracts and other remnants are essential for all curve types if urban biodiversity is to thrive. One proposed example is the construction of corridors between remnant kahikatea stands in the periurban plains surrounding Palmerston North (Janssen 2002). Similarly, the establishment of corridors to connect New Plymouth's significant reserves in the urban core to the nearby national park are, in our view, a priority.

Species movements between urban and periurban space demonstrate some of the dynamic exchanges between urban and rural areas, and the need for local authorities to coordinate restoration action. The movement of indigenous birds such as the kereru (*Hemiphaga novaeseelandiae*) and tui (*Prosthermadera novaeseelandiae*) between different zones in the city demonstrates the need for co-ordinated management of biodiversity around cities (Innes et al. 2005; Powlesland 2006). Kereru nest in cities such as New Plymouth with high indigenous cover, but in other cities such as Invercargill and Hamilton with reduced indigenous landcover in the central city, they are absent or visit seasonally from the edges (Powlesland 2006). Other species such as endangered diadromous native fish (*Galaxias* spp.) access the many gully streams in Hamilton city where they live and breed via the Waikato River which runs through the centre of the city; the gully systems extend beyond city boundaries into the periurban zone (Aldridge and Hicks 2006). City restoration can therefore become a key component or even a driver for a regional restoration. In this case, both city and rural authorities are moving to restore the gullies adjoining the city and enhance other habitats such as the peat lakes, and without this integration between administration agencies at the city/district interface biodiversity loss would be inevitable. Similarly, the goal of returning tui to cities will depend largely on the development of appropriate linkages with the periurban zone.

Given the low percentage of land available for urban restoration, and low percentage of indigenous cover, we must consider how we can create viable ecosystems within cities if we are indeed to enhance all indigenous biodiversity which could be represented. Urban parkland often contains elements of indigenous biodiversity. Parkland can have negative impacts in terms of increased human disturbance, but also positive benefits such as additional variety and seasonal offering of flowers and fruits for native birds. Most importantly urban parkland can be a potential additional resource for reconstruction of depleted ecosystems. Urban greenspaces offer a multitude of untapped conservation opportunities (Meurk and Hall 2006). Protection of large fragments in the periurban zone is clearly the best way to ameliorate some of the effects of small fragment size in the urban core, for example by allowing breeding zones from which birds for example can travel for feeding in smaller fragments. Efforts to meet a target of 10% as minimum indigenous cover are also more likely to meet with success by including and expanding protected indigenous remnants within the 5 and 10 km buffer zones. Careful location of rural residential development may also be important in determining its influence on invasion rates and ecological processes within this zone (Downs et al. 2000; Hansen et al. 2005; Sullivan et al. 2005). As well, McIntyre and Hobbs (1999) have pointed out that landscape “variegation”, which describes modified habitats, is not necessarily hostile to all native organisms. Management of exotic cover can therefore have beneficial effects for indigenous species. We consider, however, that restoration or protection of existing indigenous fragments is not enough to protect biodiversity. Reconstruction of indigenous ecosystems will be required.

In Hamilton, a North Island city, management of the biodiversity resource has evolved over the last 30 years. At first, emphasis was on protection and management of tiny remnant reserves, with key sites averaging 1.1 ha (Downs et al. 2000). However, an extensive gully network through the city of around 750 ha is currently under restoration by the city council and private landowners alike. Management focus has thus gradually shifted to revegetation and now reconstruction with the establishment of the 55 ha Waiwhakareke Natural Heritage Park (Clarkson and McQueen 2004). With the reserves, gully restoration and development of Waiwhakareke combined, Hamilton may meet a target of 10% indigenous biodiversity resource within the city in the next 20 years. Each city has its own setting, physical and natural resources but it is likely that the approach adopted for Hamilton will have relevance for other cities, especially those represented by curve (a) (Fig. 5) that have had a severely depleted biodiversity resource from the early stages of city development. It is not yet clear where these reconstructed landscapes will fall on the continuum of variegated landscapes, or alteration states as described by McIntyre and Hobbs (1999).

Cities offer special opportunities for conserving biodiversity that are not available elsewhere (Clarkson and McQueen 2004; Kilvington et al. 1998), and these opportunities are particularly pertinent for areas with highly vulnerable endemic flora and fauna which are susceptible to foreign invasion. The concentration of people makes for a huge potential volunteer base; the complete absence of grazing animals enables undergrowths and ground covers rare in many wildland ecosystems; the lack of grazing and other threats provides opportunities for establishing populations of threatened plants struggling in their natural habitat; and the juxtaposition of people and biodiversity resources enables more efficient and effective activities such as education about biodiversity and the environment, cultural harvesting by native peoples and so on. Despite many species being poorly adapted to human dominated systems, solutions to some of these problems are also achievable with further urban ecological research.

Modern urban planning which clearly separates rural and urban areas has developed from European notion of cities based on densely populated core (often a castle) surrounded by moat and with agricultural land outside, in contrast to a Japanese model of mixed agricultural and urban land (Yokohari and Amati 2005) on a similar time span. A new model which incorporates indigenous cover within the matrix of urban planning would be a desirable alternative approach supplemented of course by green belts and peri-urban conservation.

To meet the goals of the New Zealand Biodiversity Strategy (Anonymous 2000), we need to more explicitly address the need for a representative set of ecosystems and healthy populations of characteristic and iconic indigenous plants and animals, including the development of a nationally co-ordinated plan for urban areas. Because of the vulnerability of our fauna and flora to invasive species, the focus must remain on indigenous biodiversity rather than species richness per se, as frequently considered by other ecological research (e.g., Findlay and Houlahan 1997; Grayson et al. 1999). In Europe for example, it is often difficult to differentiate archeotypes and natives (Wittig 2004), leading to different management considerations than those required here. Furthermore, we do not have a history of managed urban woodlands, as is the case in parts of Europe (Gundersen et al. 2005), so that management of enrichment planting and restoration areas is still relatively new. However, acceptance of mixed origin native and exotic urban forest as a goal is growing with the recognition that management rather than extermination of invasive species is the reality (Stewart et al. 2004). To manage all of this effectively will require further capability development beyond management of utilities and infrastructure into the realms of ecosystem management. This includes different organisational scales for action (Savard et al. 2000) including strengthening relationships between municipal and regional authorities, in particular because of the need for integrated urban–periurban actions, and the inconsistencies between the city and district boundaries in relation to the urban core. As well, reconstruction of ecosystems requires a different approach from restoration, including research direction. We emphasise the need for an explicit analysis of the indigenous resource available as opposed to a laissez faire or ad hoc approach. New Zealand has a strong record in wildland ecology research which can be reconfigured to meet these issues. As yet, only a quarter of cities have inventoried indigenous and special ecological sites within their boundaries (Wehi and Clarkson unpublished data). People–wildlife interactions are crucial in shaping people's views of the environment and environmental issues (see Savard et al. 2000). McDonnell (2005) argued that maintaining a diversity of indigenous organisms and ecosystem processes in cities is critical to the ecology of a region. Understanding the landscape mosaic at a national level further informs this process. The analysis presented in this study indicates that biodiversity decline must be addressed in a systematic and considered way if substantial progress towards the retention of indigenous biodiversity is to occur.

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Appendix 1

Table 3 Generalisation of the Landcover 2 data set

Landcover 2 class	Generalised class	Indigenous
Built-up area	Urban	
Urban parkland/open space	Urban	
Surface mine	Mine or dump	
Dump	Mine or dump	
Transport infrastructure	Unclassified	
Coastal sand and gravel	Coastal sand	
River and lakeshore gravel and rock	River or unclassified	
Landslide	Unclassified	
Alpine gravel and rock	Alpine rock	Yes
Permanent snow and ice	Permanent snow	
Lake and pond	Lake	
River	River	
Estuarine open water	Sea	
Short-rotation cropland	High producing grassland	
Vineyard	Horticulture	
Orchard and other perennial crops	Horticulture	
High producing exotic grassland	High producing grassland	
Low producing grassland	Low producing grassland	
Short tussock grassland	Tussock	Yes
Tall tussock grassland	Tussock	Yes
Depleted grassland	Low producing grassland	Yes
Herbaceous freshwater vegetation	Freshwater wetland	Yes
Herbaceous saline vegetation	Saltwater wetland	Yes
Flaxland	Freshwater wetland	Yes
Fernland	Indigenous scrub	Yes
Gorse and broom	Exotic scrub	
Manuka and or kanuka	Indigenous scrub	Yes
Matagouri	Indigenous scrub	Yes
Broadleaved indigenous hardwoods	Indigenous scrub	Yes
Sub alpine shrubland	Sub alpine scrub	Yes
Mixed exotic shrubland	Exotic scrub	
Grey scrub	Indigenous scrub	Yes
Minor shelterbelts	Unclassified	
Major shelterbelts	Unclassified	
Afforestation (not imaged)	Exotic forest	
Afforestation (imaged, post LCDB 1)	Exotic forest	
Forest—harvested	Exotic forest	
Pine forest—open canopy	Exotic forest	
Pine forest—closed canopy	Exotic forest	
Other exotic forest	Exotic forest	
Deciduous hardwoods	Exotic forest	
Indigenous forest	Indigenous forest	Yes
Mangrove	Saltwater wetland	Yes

Appendix 2

Table 4 Land environment categories at level 1 of LENZ (taken from Leathwick et al. 2003)

	Land environment categories
A	Northern lowlands
B	Central dry lowlands
C	Western and southern North Island lowlands
D	Northern hill country
E	Central dry foothills
F	Central hill country and volcanic plateau
G	Northern recent soils
H	Central sandy recent soils
I	Central poorly-drained recent soils
J	Central well-drained recent soils
K	Central upland recent soils
L	Southern lowlands
M	Western south Island recent soils
N	Eastern South Is plains
O	Western South Is foothills and Stewart Is
P	Central mountains
Q	Southeastern hill country and mountains
R	Southern alps
S	Ultramafic soils
T	Permanent snow and ice

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