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Backcountry Huts as Introduction Points for Invasion by Non-native Species into Subalpine Vegetation

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Abstract

The role of backcountry huts as introduction points for the establishment and spread of non-native plants into remote natural areas has received little attention. We surveyed soil and vegetation around 25 backcountry huts in the subalpine landscape of the Australian Alps to examine the role that such huts play in acting as foci for invasion by non-native species into remote mountain areas. We found that the hut surroundings were characterized by greater soil compaction, lower vegetation height, and more bare ground relative to the native plant community located 100 m from huts. At the landscape-scale, a total of 32 non-native species were recorded within 100 m of huts. Seven species were found at greater than 50% of huts (*Hypochoeris radicata*, *Taraxacum officinale*, *Acetosella vulgaris*, *Trifolium repens*, *Cerastium glomeratum*, *Agrostis capillaris*, *Poa annua*), and these tended to be the species that have (a) been long-established (>50–100 yrs) and (b) are the most frequent in the broader landscape. Hence, huts act to promote these ruderals by providing opportunities for their establishment, but such opportunities are not confined to hut surroundings. Several other non-native species, however, were common around huts but largely absent from the wider landscape (e.g. *Anthoxanthum odoratum*, *Plantago major*, *Polygonum aviculare*, *Stellaria media*). This suggests that some species are advantaged by the disturbance and dispersal opportunities provided by hut recreational activities in a way that is not catered for elsewhere in the Australian Alps. A weak negative relationship between non-native species richness and increasing altitude was found, but native species richness and distance of hut from access road were poor predictors of non-native species richness. Our study highlights that recreational activities may provide opportunities for the establishment of non-native plant species in remote high mountain areas, some of which are novel to the landscape, and that these may form the basis for further invasion into adjoining native vegetation.

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Introduction

Infrastructure associated with recreation, along with the recreational activities themselves, may provide opportunities for invasion by non-native plant species into natural ecosystems because they can modify habitat via disturbance while also overcoming dispersal barriers (Lonsdale and Lane, 1994; Parendes and Jones, 2000; Dickens et al., 2005). In subalpine and alpine areas, for example, ski resort gardens and revegetation practices are two means by which non-native species are often introduced into high mountain environments (McDougall and Appleby, 2000; Johnston and Pickering, 2001a; McDougall, 2001; Bear et al., 2006), and these may become the propagule source for invasion into adjoining native vegetation (McDougall et al., 2005). It is not surprising, therefore, that non-native plants abound in areas of high human activity (Lonsdale, 1999) and that areas where human impacts are lower, e.g. remote area wilderness, tend to have fewer non-native species in their flora. The role of recreational use in facilitating these invasions, however, is not well understood.

Dispersal opportunities are likely to improve when visitation rates increase or concentrate in remote areas as visitors can aid propagule dissemination in their vehicles (Lonsdale and Lane, 1994), clothing, and equipment (Whinam et al., 2005). Opportunities for establishment of non-native species might also increase with increasing recreational pressure as trampling damages vegetation and can increase resource availability (Hobbs and Huenneke, 1992; Cole, 1995; Marion and Cole, 1996; Cole and Monz, 2003). This pattern is particularly evident close to campsites and along the walking tracks leading to them (Marion and Cole, 1996; Leung and Marion, 1999; Gibson et al., 2000; Jesson et al., 2000; Johnston and Pickering, 2001a; Dickens et al., 2005). The role of backcountry huts, used by bushwalkers in summer and skiers in winter, as focal points for non-native plant establishment and spread into remote natural areas, however, has received little attention (Lloyd et al., 2006).

Despite their remoteness from large population centers, the Australian Alps have relatively high visitation rates for their restricted area (Pickering and Buckley, 2003), potentially enhancing opportunities for non-native propagule dispersal and establishment. More than 64,000 people per annum visit Mt. Kosciuszko (Australia's highest mountain summit) (Pickering and Buckley, 2003) and, while there is no specific data to quantify

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visitation to areas outside of alpine resorts (Department of Natural Resources and Environment, 2000), anecdotal evidence suggests that visitation may be concentrated in areas such as backcountry huts, the focus of year-round recreational activities. Hut logbooks indicated that several hundreds of people visit them each year, with more entries at huts located close to major access roads (authors, personal data).

Flora in high mountain regions, however, is subjected to abiotic conditions characterized by low growing season temperatures, shortened growing season lengths, and severe temperature extremes (Korner, 1999). These factors can limit the establishment and survival of species maladapted to such conditions. Hence, despite the potential effects of recreational activities on non-native invasion in high mountain areas, alpine ecosystems have generally been rated as amongst those least vulnerable to invasion by non-native species (Humphries et al., 1991). In general, non-native species diversity declines with increasing altitude in alpine Australia (Mallen, 1986; Johnston and Pickering, 2001a; McDougall et al., 2005; Bear et al., 2006) and elsewhere (Jesson et al., 2000).

Many of the huts in the subalpine landscape of Australia are visited year-round and hence, these areas receive a disproportionately high visitation rate in the landscape relative to their accessibility. As such, levels of physical disturbance (to the soil and vegetation) are likely to be high, and accidental (and deliberate) introductions of exotic organisms are likely to be centered on such places. An assessment of the current condition of the vegetation at backcountry huts, and those factors that affect plant invasions by non-native species, is generally lacking in Australia and elsewhere. Godfree et al. (2004) showed that in the Kosciuszko National Park, Australia, some non-native species were found only near alpine huts ($n = 4$ huts), suggesting that non-native species have either been dispersed by human activity or advantaged by the disturbances associated with the huts. In mountains of New Zealand, Jesson et al. (2000) found that important predictors of non-native species presence were altitude and proximity to a river, hut, or road, these being sites of disturbance, with a negative relationship between the number of exotic species and distance from huts and roads (i.e. propagule source). Elsewhere in New Zealand, Lloyd et al. (2006) found that non-native species were centered on a single remote hut and did not extend far from the hut surroundings. Hence, it appears that backcountry huts are potentially important (but largely unrecognized) sites for invasion by non-native plant species into remote areas.

This study aimed to (a) quantify the level of plant invasion around backcountry huts in a range of landscape settings in the Australian Alps and (b) determine whether the number of non-native species present at huts could be predicted in relation to the landscape setting (altitude) and accessibility (distance from road) of the huts.

Methods

STUDY AREA

Alpine and subalpine vegetation in Australia is limited to areas above 1500 m in mainland southeastern Australia. The area is a series of undulating and dissected plateaus (Costin, 1957), extending from Baw Baw National Park in Victoria (37°52′S) to Namadgi National Park (35°25′S) in the Australian Capital Territory, a distance of about 350 km. The climate is characterized by 1–6 months of snow cover, depending on aspect and altitude, low annual minimum and mean temperatures, and precipitation between 1200 and 2200 mm per annum (Williams, 1987). Soils are highly organic and acidic, having formed on a range of lithologies including basalt, schist, gneiss, limestone, and granite (Rowe, 1972). Plant communities range from snow gum (*Eucalyptus pauciflora*) woodlands in subalpine tracts to heathlands, grasslands, peatlands, and herbfields in alpine areas (Costin et al., 2000). Over 700 native taxa from 82 families have been recorded from treeless areas of the Australian Alps (McDougall et al., 2005).

Landscape-scale disturbance is infrequent in the Australian Alps. Fire occurred in 1939 over much of the study area (Lawrence, 1994) and during 2003, fire affected large areas of the Australian Alps, burning the surroundings of some huts in this study (Williams et al., 2006). Grazing by domestic stock has been a feature of alpine areas since the 1850s and has been gradually removed from high mountain catchments since the 1950s (Lawrence, 1994).

Tourism and recreational activities have been part of the Australian Alps since the 1860s when the first ski lodge was built (Good, 1992) and has increased since the 1960s, with a corresponding alteration in soils, hydrology, and vegetation (McDougall, 2001). Today, bushwalking, car touring, horse riding, and skiing are significant activities in many areas of the Australian Alps, as is hydroelectricity production.

McDougall et al. (2005) reviewed the non-native flora of the Australian Alps (using available quadrat data, personal observations, and herbarium specimens) and found a total of 128 species had established in treeless vegetation. Most were herbaceous and most were uncommon and had low cover. Most species were considered unintentional introductions as seed attached to vehicles, animals, and humans.

STUDY SITES

Twenty-five huts in the Alpine National Park, Victoria, were selected for study during February–May 2005. The huts sampled the range of physical settings (lithology, aspect, slope, altitude), remoteness (distance from access road), and disturbance histories (grazing, fire) found in the region. All huts were located within the subalpine *Eucalyptus pauciflora* woodland with a grass- to shrub-dominated understory. Huts (or the original structures) and their associated pit toilets have been established for varying periods, ranging from decades to over a century, many of them built as refuges for high mountain graziers. Today, huts are mostly visited by day walkers, campers, horse riders, and, in winter, cross-country skiers, although no data exists to quantify these visitation rates.

ENVIRONMENTAL CHANGES AROUND HUTS

To determine whether soil compaction varied with distance from hut, soil bulk density was measured in May 2005 at 10 randomly selected huts. Three soil samples, each 120 cm^3, were taken randomly from each of three locations around huts (0, 50, and 100 m from hut). Soil samples were dried at 105 °C for 24 hrs and weighed. The data are reported as bulk density (weight of dry soil divided by volume of soil). The effect of distance from hut on mean soil bulk density was compared by analysis of variance, using square-root transformed data to improve normality, followed by a Tukey's post-hoc test.

VEGETATION SURVEY

Vegetation surveys were conducted from February to April 2005 to document the non-native species surrounding huts. All plant species rooted in 10 m radius circular quadrats (area = 314 m^2) at three locations (0, 50, and 100 m from hut, each quadrat being located in a random direction) were recorded to examine effects of distance from hut on invasion. Cover of individual species was not recorded. Instead, we focused on presence/absence data and frequency of occurrence across the landscape. Nomenclature follows Ross and Walsh (2003). The effect of distance from hut on non-native species richness was compared using one-way analysis of variance followed by Bonferroni post-hoc tests, using all huts as replicates.

To determine the factors that affect total non-native species richness at huts (pooling species for the three quadrats per site) at the landscape level, generalized linear models were used with the explanatory variables altitude (m), distance from road/access point (km), and total native species richness as predictors. As the dependent variable (i.e. non-native species richness) consisted of discrete data (counts), a Poisson error distribution was assumed in all regression models and a logarithmic link function was used in all fitted models (McCullagh and Nelder, 1989). A forward stepwise procedure was used to enter the explanatory variables in the model and only variables that accounted for a significant change in deviance ($p < 0.05$) were retained. Analyses were conducted using GenStat8.

GROUND COVER CONDITION—POINT QUADRATS

Point quadrats were used to determine how the percent cover of non-native species, native species, and bare ground changed from near (0 m) to far (100 m) from huts. Point quadrats were also used to quantify the height of vegetation (i.e. structure) relative to distance from huts. A single 100 m transect was established at each site running away from huts in the direction of greatest visible degradation. Using a 4 mm diameter pin, vegetation was recorded as native, non-native, bare ground, or non-natural bare ground at 20 cm intervals. Bare ground refers to that found naturally such as inter-tussock or inter-shrub space, or the bare ground created by natural fire events. Non-natural bare ground refers to that in the form of tracks, and to that between patches of the flat, sparse ground cover found in high-use areas (e.g. surrounding huts). Maximum vegetation height was recorded as 0 (bare ground), 1 (0–5 cm), 2 (6–15 cm), 3 (16–30 cm), 4 (31–50 cm), 5 (51–100 cm) or 6 (>100 cm) at 1 m intervals. Height data are presented as percent frequency of each height class in 10 m sections of the transect. Percent cover of bare ground, non-native and native species were compared across all distances from hut in 10 m sections. Data could not be transformed to normality, so the non-parametric Kruskal-Wallis test was used.

PLANT TRAITS OF INVADERS

To determine whether non-native species invasion at huts was associated with common life-history strategies, plant traits for each non-native species found in the vegetation survey were assigned using local floras and weed identification books (e.g. Halfiger and Brun-Hool, 1975; Auld and Medd, 1987). All species were scored for life-form (annual, perennial), growth-form (grass, herb, shrub, tree), and mode of dispersal (wind, water, vertebrate animal, ant).

BACKGROUND LEVELS OF NON-NATIVE SPECIES INVASION IN THE LANDSCAPE

Two sources of information were collated to assess the frequency of non-native species in the Australian Alps. Firstly, using quadrat and point location data collected by state government botanists and universities and stored in a database managed by the Victorian Department of Sustainability and Environment (Flora Information System, unpublished data), we retrieved data ($n \approx 1300$ records) above 1400 m in our study area that were collected from subalpine and alpine plant communities (excluding wetlands). Using this database, we calculated the landscape-scale frequency of non-native plants to derive a "natural" hierarchy of abundance in the region. We then compared the frequency of species in the hut non-native flora with that observed in the landscape matrix to determine whether species found around huts were generalist invaders of the alpine environs in Victoria, or whether their frequency of occurrence was much higher than might be predicted from their background levels of invasion. Secondly, we used abundance categories defined by McDougall et al. (2005) for non-native species in natural vegetation across the entire Australian Alps to further confirm our hierarchies of abundance in the landscape. Under this scheme, non-native species were ranked as "rare," "locally common," or "widespread" in the landscape.

RESIDENCE TIME

To determine if the species that were most common around huts were also those that have been in the landscape the longest, we used data provided in Johnston and Pickering (2001b) to estimate the timing of first incursion into the broader Australian Alps landscape. These authors summarize plant invasions into high altitude (>1500 m) areas of Australia based on herbarium and quadrat information from the earliest records of the government botanist J. H. Maiden in 1896 (restricted to the Kosciuszko National Park) to more recent surveys (1999). While the data is spatially and temporally restricted, it does provide a useful indication of the relative residence time of species. We use this data to identify species that have been in the landscape for an extended time (these species were first collected between 1898 and 1954), for a moderately long time (first collected between 1954 and 1985), or are species that appear to be recent introductions (>1985).

Results

SOIL COMPACTION AROUND HUTS

Soil was more compacted immediately around huts (0 m) than 100 m from them ($p = 0.007$) (Fig. 1).

VEGETATION CHANGES AROUND HUTS

Vegetation was predominantly absent (0 cm height class) for the first 10 m near huts (Fig. 2). There was a significant relationship between the proportion of bare ground and distance from hut (Kruskal-Wallis: $P < 0.001$). Vegetation >100 cm in height increased from 0 to 20% cover in the first 40 m. Other vegetation height classes gradually increased (i.e. 31–100 cm categories) or decreased (i.e. 1–30 cm categories) in frequency along the transect with increasing distance from hut (Fig. 2).

Ground cover was composed of primarily non-natural bare ground and non-native species within 20 m of huts (Fig. 3). Non-natural bare ground cover then sharply decreased and native cover increased. After 20 m, non-natural bare ground and non-native cover slowly decreased, and bare ground and native cover slowly increased. By 100 m from huts, native species and bare ground

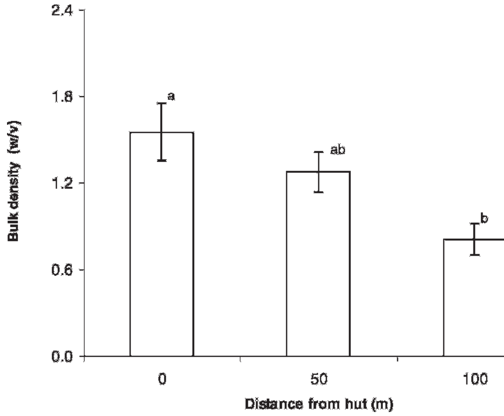

FIGURE 1. Mean soil compaction (±1 SE), measured as bulk density, at 0, 50, and 100 m from backcountry huts located in the Alpine National Park, Victoria. Different letters represent significant difference between group means.

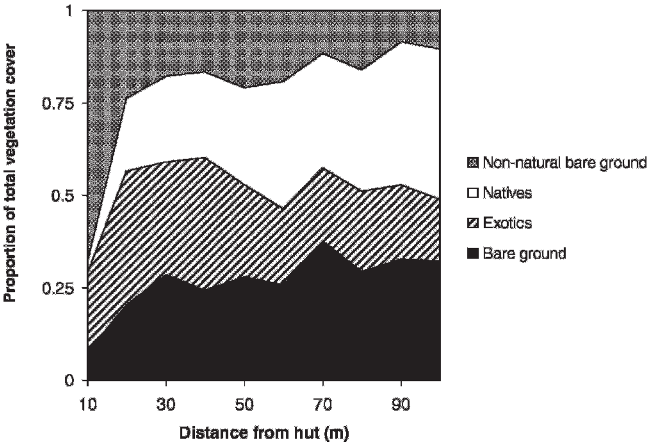

FIGURE 3. Mean abundance of ground cover classes along a 100 m transect running from near backcountry huts (0 m) into native vegetation.

became the most abundant cover types. There was a significant difference in cover class abundances across all distances (Kruskal-Wallis: $P < 0.001$); non-native ($P = 0.003$), natives ($P < 0.001$), and non-natural bare ground ($P < 0.001$) were all affected by distance from hut, but not bare ground ($P = 0.411$).

NON-NATIVE SPECIES AROUND HUTS

A total of 32 non-native species from 11 families were detected in the vegetation surrounding huts (Table 1). Dominant families were Poaceae (35% of species), Asteraceae (16%), Rosaceae (13%), and Polygonaceae (10%). Seven species were found at greater than 50% of huts: *Hypochoeris radicata*, *Taraxacum officinale*, *Acetosella vulgaris*, *Trifolium repens*, *Cerastium glomeratum*, *Agrostis capillaries*, and *Poa annua*. Four species were new to the study region (Table 1), having not previously been recorded in the matrix of native vegetation (using FIS data as our baseline). At the landscape-scale (using data from McDougall et al., 2005), only *Triticum aestivum* has not been previously recorded in subalpine vegetation in Australia.

Species that were most frequent in the landscape were also most frequent around huts (Table 1). Indeed, frequency hierarchies among the most common species were not different between the hut flora and landscape flora for the five commonest species (Table 1). However, there were many exceptions beyond this point. Species such as the grasses *Agrostis capillaris*, *Anthoxanthum odoratum*, and *Poa annua*, as well as the forbs *Polygonum aviculare* and *Stellaria media*, were observed very frequently around huts (>30% of huts) but have rarely been observed in the natural landscape (Table 1).

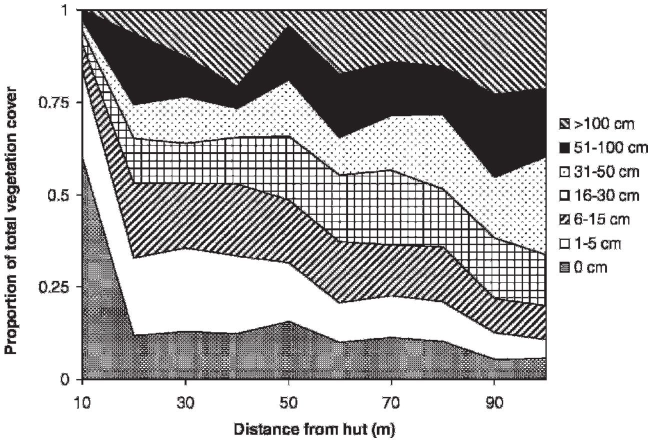

FIGURE 2. Mean ($n = 25$) proportion of vegetation in seven height classes along a 100 m transect from backcountry huts (0 m) into native vegetation.

Most (69%) of the non-native species we encountered around huts were first noted in the broader alpine landscape between 1954 and 1985 (Table 1). This suggests that many species have had a relatively long association with high mountain areas. Only three of the species (9%) found around huts are known to have been long-established in alpine areas (i.e. before 1954; *Acetosella vulgaris*, *Hypochoeris radicata*, *Taraxcum officinale*) while 18% are thought to be recent invaders, appearing only after 1985 in surveys of alpine vegetation (Table 1).

Non-native species were common at all huts; 54% of huts had between 5 and 12 non-native species in their surrounds (Fig. 4). Significantly higher numbers of non-native species were recorded in the immediate vicinity of the hut (within 10 m) ($F = 20.42$, $p < 0.001$) relative to 50 and 100 m from huts (Fig. 5). The total number of non-native species around huts also varied depending on whether the hut landscape had been recently burnt (in 2003), where richness was significantly lower ($p = 0.002$), or had remained unburnt (Fig. 4).

SPECIES RICHNESS—GENERALIZED LINEAR MODELING

As indicated by the results from the first pass of the model fitting procedures, no single variable satisfactorily explained exotic species richness at huts. When variables were fitted individually, the greatest change in deviance in exotic species richness was accounted for by altitude (11.58%, $p < 0.01$). Native species richness and distance from main road were inadequate predictors of exotic species richness (change in deviance of 1.99% and 0.99%, respectively, $p > 0.05$).

Scatterplots of richness and altitude (the only significant predictor) showed data points evenly distributed across the range of the explanatory variables; the relationship was therefore not driven by outliers (Fig. 6).

When all three explanatory variables were added to the model, total variation explained was substantially improved in the final pass model, although the explanatory power was still low (total change in deviance = 21.45%).

TABLE 1

Frequency of non-native species recorded at backcountry huts ($n = 25$), within the study landscape (FIS database, $n \approx 1300$ quadrats and point localities) and ranked abundance as per McDougall et al. (2005). The "residence time" of non-native species was assessed by reference to Johnston and Pickering (2001b), which contains a listing of the period that non-native species were first recorded in the Australian Alps between 1898 and 1998. Life/growth form: AH = annual herb, PH = perennial herb, AG = annual grass, PG = perennial grass, PS = perennial sedge, S = shrub, T = tree. Mode of dispersal: A = ant; B = animal; H = water; V = vegetative; W = wind.

Species	Frequency (%) at huts	Frequency (%) in landscape	"Abundance" in the Australian Alps	Period first recorded in alpine area	Life/growth form	Mode of dispersal
Acetosella vulgaris	100	41	widespread	<1954	PH	B, H, W
Achillea millefolium	8	0.1	common	1954–1985	PH	A
Agrostis capillaris	68	2	widespread	1954–1985	PG	B, V, W
Anthoxanthum odoratum	32	0.8	widespread	1954–1985	PG	B, H, W
Arrhenatherum elatius	4	0	rare	1954–1985	PG	B
Capsella bursa-pastoris	4	0	rare	1954–1985	AH	B, ?W
Cerastium glomeratum	72	11	widespread	1954–1985	AH	?B, ?H
Cirsium vulgare	8	7	widespread	1954–1985	AH	B, H, W
Crataegus sp.	8	0.4	—	>1985	T	B
Dactylis glomerata	20	0.7	rare	1954–1985	PG	B, H, ?W
Festuca rubra	16	0.3	widespread	1954–1985	PG	B, ?H
Holcus lanatus	28	4	widespread	1954–1985	PG	?W
Hypochoeris radicata	88	39	widespread	<1954	PH	B, W
Juncus effusus	12	0.5	widespread	1954–1985	PS	B, H, ?W
Lolium perenne	20	0.3	rare	1954–1985	PG	B, H, W
Lotus corniculatus	8	0.7	widespread	1954–1985	PH	B, H
Malus × domestica	24	0.2	widespread	>1985	T	?B, ?H
Phleum pratense	28	0.5	rare	1954–1985	PG	B, H, W
Plantago major	20	0.5	rare	>1985	PH	B, H
Poa annua	56	0.5	rare	1954–1985	AG	B, H, W
Polygonum aviculare	36	0.2	rare	1954–1985	PH	B
Rosa rubiginosa	12	3	common	1954–1985	S	B, H
Rubus spp.	20	3	rare	>1985	S	B, V
Rumex crispus	16	0.5	rare	1954–1985	PH	B, H, W
Sonchus asper	4	0.7	rare	1954–1985	AH	B, H, W
Stellaria media	44	0.3	—	>1985	AH	?W
Taraxacum officinale	76	8	widespread	<1954	PH	B, H, W
Trifolium repens	88	19	widespread	1954–1985	PH	B, H
Triticum aestivum	4	0	—	>1985	AG	B
Urtica urens	12	0	rare	>1985	AH	B

PLANT TRAITS OF INVADERS

Half (50%) of all invasive taxa recorded around huts were perennial herbs, while grasses also comprised a large proportion of the species seen (38%) (Table 1). Annual species contributed 25% of species. The most common mode of dispersal by non-native species recorded in this study was by animals/humans (69% of species), followed by water (56%) and wind (50%). Two species (apple [*Malus × domestica*] and wheat [*Triticum aestivum*]) could be considered with certainty as introductions due to bushwalkers.

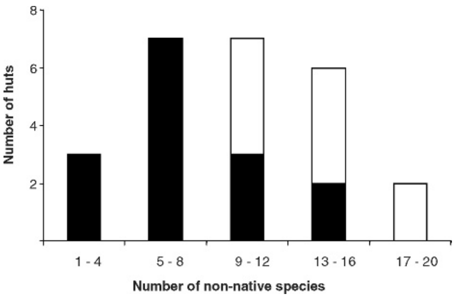

FIGURE 4. Total number of non-native species observed at huts that had been either burnt (■) or unburnt (□) by wildfire in 2003.

Discussion

KEY FINDINGS

Plant invasion into mountain regions of Australia have previously been attributed to the disturbances created by cattle grazing (van Rees, 1982), revegetation (McDougall, 2001), roads (Godfree et al., 2004), and ski resorts (McDougall and Appleby, 2000; Johnston and Pickering, 2001a). Our study confirms that non-native plant species have established in subalpine vegetation

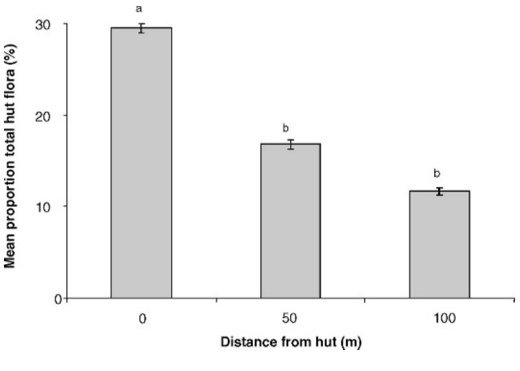

FIGURE 5. Mean (±1 SE) total hut flora (%) constituted by non-native species at 0, 50, and 100 m from hut. Different letters represent significant difference between group means.

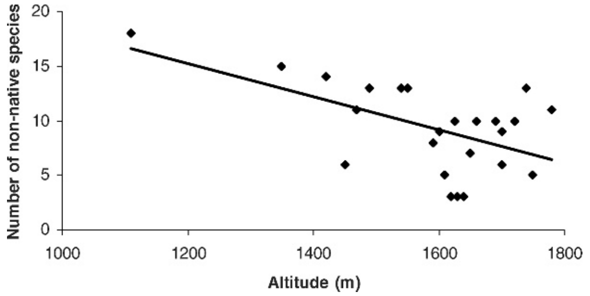

FIGURE 6. The relationship between non-native species richness around backcountry huts and altitude in the Australian Alps.

around backcountry huts used for recreational purposes, with some species much more frequent there than expected given their distribution in the landscape matrix. The environment immediately surrounding backcountry huts was highly modified, with more compacted soil close to huts than further from them. Vegetation (both native and non-native) was shortest immediately surrounding huts despite most not having been grazed for some time, with more disturbance-tolerant growth forms closer to the huts. Non-native species were more abundant close to huts, possibly a reflection of the high levels of disturbance observed there. Hence, recreational activities are capable of causing substantial modification to natural vegetation and may provide (a) the vectors necessary for dispersal into remote areas, (b) the disturbance necessary for initial establishment, and (c) the changes in the native plant community structure necessary to permit dominance by non-natives. This, in turn, might ultimately enable invasion by non-native species into more distant native vegetation given enough time and/or appropriate disturbances such as fire (Walsh and McDougall, 2004).

VEGETATION AT HUTS

High country huts were found to be a key point for environmental change in the subalpine landscape. Soil was significantly more compacted closer to huts (0 m) than far from them (100 m) and we ascribe these changes to recreational impacts of bushwalkers and campers, as has been found by Grabherr (1982). Impacts in natural areas tend to be focused around facilities such as shelters, picnic tables, and toilets, and the subsequent changes in vegetation composition and structure have been well-documented (Marion and Cole, 1996; Leung and Marion, 1999; Marion and Farrell, 2002; Bennett et al., 2003; Johnston and Johnston, 2004; McDougall and Wright, 2004; Roovers et al., 2004).

Vegetation was shortest immediately around huts, probably due to trampling and shear-stresses. Given the abundance of shrubs in the native woodlands of the study region, we believe that the vegetation absent from hut surroundings was likely to include shrubs such as *Podolobium alpestre*, *Bossiaea foliosa*, *Grevillea australis*, and *Hovea montana* (Costin et al., 2000). Shrubs are generally resistant to low intensity trampling, denoting their ability to resist initial change when damaged, but are not tolerant of repeated trampling, denoting their inability to tolerate a cycle of disturbance and recovery (Cole, 1995). As height, cover, and biomass of woody plants generally decreases with recreation-associated trampling (Whinam and Chilcott, 1999; Gibson et al., 2000; Roovers et al., 2004), this may explain the lack of taller vegetation types found within 20 m of huts. By comparison, grasslands are generally much more resistant to trampling stresses (Edwards, 1977; Cole, 1995; Marion and Cole, 1996; McDougall, 2001; Roovers et al., 2004) and less likely to degrade even when subject to repeated trampling.

Significantly more non-native species were found, on average, immediately around huts than 50 or 100 m from them. Higher non-native richness directly surrounding huts may be a result of the greater soil disturbance that occurs there (Hobbs and Huenneke, 1992; Parendes and Jones, 2000; Johnston and Johnston, 2004). Alternatively, it may be due to enhanced dispersal, with seeds attached to the clothing and equipment of hikers (Whinam et al., 2005) or horses (St. John-Sweeting and Morris, 1990), or greater modification of the native plant community structure. It remains unclear which factor(s) are most responsible for invasion by non-native species into these areas.

Comparisons with other vegetation surveys conducted in the Alpine National Park (e.g. FIS unpublished data; McDougall et al., 2005) indicate that some non-native species were at much greater frequency surrounding huts than in the wider landscape (e.g. *Achillea millefolium*, *Taraxacum officinale*, *Polygonum aviculare*, *Agrostis capillaris*, *Poa annua*, and *Anthoxanthum odoratum*). This suggests that some species are advantaged by the disturbance and dispersal opportunities provided by hut recreational activities in a way that is not catered for elsewhere in the Australian Alps. Godfree et al. (2004) similarly found evidence for a hut-specific non-native flora in the Australian Alps. Hence, huts are points for the introduction of new species into the alps, promoting a suite of species that are not characteristic ruderals of roadsides and other highly disturbed areas in alpine landscapes (Mallen, 1986; Mallen-Cooper and Pickering, 2008). That 18% of the hut flora we identified had not been recorded in the Australian Alps prior to 1985 (Johnston and Pickering, 2001b) strengthens the view that recreation around huts enhances opportunities for invasion of novel species.

Many non-native species that were common around huts, however, were also common in the alpine landscape matrix (e.g. *Acetosella vulgaris*, *Hypochoeris radicata*, *Trifolium repens*). Such species may be capable of long-distance dispersal (Jesson et al., 2000) and may have taken advantage of the disturbances created by decades of stock grazing that has occurred in Australian alpine landscapes. These species have long been recorded in the Australian Alps (e.g. Mallen, 1986, notes that *A. vulgaris* and *H. radicata* were two species of widespread distribution in the Kosciuszko National Park in 1898) and perhaps this long residence time and subsequent propagule pressure, co-regulated by disturbance (Britton-Simmons and Abbott, 2008), explains their ubiquity in this landscape. Huts act to promote these ruderals by providing opportunities for their establishment, but such opportunities are not necessarily confined to hut surroundings. Some of these species (e.g. *T. repens*) are also common in other remote areas of the world (e.g. U.S.A. [Dickens et al., 2005], New Zealand [Jesson et al., 2000], and subantarctic islands [Mallen, 1986]), suggesting a cosmopolitan distribution.

We expected that non-native richness would be higher at huts closer to access points (e.g. nearby roads) due to the effects of higher disturbance due to greater visitation (Parendes and Jones, 2000) and potentially greater propagule pressure. There was no significant relationship, however, between the distance from access road to hut and non-native species richness. This seems puzzling but the effects of isolation may be accentuated when distances exceed the relatively short distances from point source covered in this study (i.e. we sampled sites that were only <10 km from access roads). This highlights for us that much of the Australian Alps is not very remote from access tracks and hence, isolated from potential sources of invasion. Similarly, Dickens et al. (2005) failed to find a decline in non-native species invading recreational portage trails in Minnesota with increasing distance from access point, suggesting that degree of disturbance, not propagule dissemination *per se*, might drive invasions in remote areas.

Sites which were burnt in the 2003 fires were found to have significantly fewer non-native species than those which were not. Lower non-native richness at these sites may have resulted because propagules did not survive the fires or post-fire climatic conditions were not conducive to germination. This, however, may not always be the case. Walsh and McDougall (2004) noted that, two years after fire in nearby subalpine grasslands, non-native species richness was higher than pre-fire richness. Hence, it is worth revisiting some of the huts surroundings of areas burnt in 2003 in future years to monitor the rate of invasion with post-fire succession rather than assume that it will always be lower there.

General linear modeling showed that non-native species richness decreased with increasing altitude. This is likely due to the decreased favorable growing conditions associated with altitude (Choler et al., 2001) and is a widely observed phenomenon in alpine areas (Mallen, 1986; McDougall et al., 2005; Bear et al., 2006; Mallen-Cooper and Pickering, 2008). It would appear reasonable, therefore, to suggest that with increases in global temperatures due to climate warming, invasion of alpine areas by non-native species might occur more readily. Given that negative relationships between altitude and non-native richness might also result from factors such as mass-area relationships and declines in disturbance intensity, such hypotheses need more rigorous testing.

Conclusion

The invasion by non-native species into the subalpine landscape of the Australian Alps is enhanced by recreational activities that improve dispersal opportunities and increase soil and vegetation disturbance. However, the ecological impacts of such invasions on native plant communities have yet to be quantified and this remains an important topic for future research. It will be important to model future invasion pathways into alpine vegetation to understand where, and why, non-native species will spread beyond the point of introduction into the landscape. Identifying the factors that facilitate this process is necessary if conservation managers are to effectively control future invasions around backcountry huts. In those areas already invaded, the ability of the native vegetation to recover, and the factors that limit these transitions of recovery, should be identified. Finally, it is important to develop a seed dispersal and plant functional trait database of non-native species in alpine areas to help understand which plant types might be associated with future invasions into remote areas with anticipated global warming and increasing recreational pressures.

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