Impacts of tourism hotspots on vegetation communities show a higher potential for self-propagation along roads than hiking trails

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ABSTRACT

Vegetation communities along recreational tracks may suffer from substantial edge-effects through the impacts of trampling, modified environmental conditions and competition with species that benefit from disturbance.

We assessed impacts on trackside vegetation by comparing high and low usage tourism sites at a 1 −10 m distance from recreational tracks in a popular arid-lands tourism destination in South Australia. The central aim was quantification of the strengths and spatial extent of tourism impacts along recreational tracks with a qualitative comparison of roads and trails.

Track-distance gradients were most prevalent at high usage sites. There, species community composition was altered, total plant cover decreased, non-native species cover increased, plant diversity increased or decreased (depending on the distance) and soil compaction increased towards recreational tracks.

Roadside effects were greater and more pervasive than trailside effects. Further, plant diversity did not continuously increase towards the road verge as it did along trails but dropped sharply in the immediate road shoulder which indicated high disturbance conditions that few species were able to tolerate.

To our knowledge, we are the first to demonstrate that the access mode to a recreation site influences the potential of certain impacts, such as the increase of non-native species, to self-perpetuate from their points of introduction to disjointed sites with a predisposition to disturbance. Due to this propulsion of impacts, the overall spatial extent of roadside impacts was far greater than initially apparent from assessments at the road verge. We discuss possible means of mitigating these impacts.

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1. Introduction

An intricate network of recreational tracks admits visitor traffic to many natural tourism areas worldwide. Tracks traverse the landscape and fragment plant habitat which exposes abutting plant communities to edge-effects (reviewed by Murcia, 1995): The physiognomy of tracks as well as their perpetual use and maintenance may interfere with vegetation via a direct mechanical disturbance, a modification of the abiotic plant habitat and the facilitation of non-native and native species that thrive in disturbed areas. With a growth in tourism numbers, the expectation is that recreational demand will intensify conflicts with the protection of vegetation and other habitat resources. Thus we need to elucidate how vegetation varies in relation to visitor usage along travel corridors in order to mitigate adverse effects. Given the popularity of roads and hiking trails for facilitating easy access throughout protected areas, our key objective is to determine whether trackside impacts and their spatial extent differ between sites with vehicle and hiker access. So far, impacts of both modes of access on the total abundance of plants, the composition and diversity of plant communities have mostly been investigated independently. A direct comparison can aid tourism management in making informed decisions about access options to tourism sites.

Plant abundance next to recreational tracks may be lower (Cole, 1978) or greater (Hall and Kuss, 1989) than in less disturbed sites. Differences in abundance are dictated by complex processes that govern the trackside environment. For example, trampling (reviewed by Cole, 2004) may damage plant tissue (Meinecke, 1928) and cause an overall reduction in plant vigour and reproductive output (Liddle, 1997) which may in turn lead to a reduction in the total cover, height and biomass of vegetation (Cole, 2004). Plant abundance may further be affected indirectly through

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changes in the micro-environment next to tracks: soil abrasion and compaction are prominent examples of habitat modification that accrue from trampling (Belnap, 1998). Compression of the soil structure leads to a reduction in air and water movement, reduced water infiltration (Hammitt and Cole, 1998) and a decreased water retention, except for coarse-textured soils (Gallet and Rozé, 2002). Such conditions are inhospitable for root (Bhuju and Ohsawa, 1998) and vegetative growth (Settergren and Cole, 1970). Compaction also increases the force required for plant roots to penetrate the soil, restricting root elongation and soil pervasion (Materechera et al., 1991). However, plant species vary in their tolerance of soil compaction (Bassett et al., 2005; Kyle et al., 2007). Parker et al. (2010) for instance found that non-native but not native plant species richness in North American forests was positively correlated with increasing soil compaction.

Notwithstanding, the track shoulder may retain higher quantities of moisture due to an increased water runoff from the compacted and barren centre of most tracks which may stimulate vegetation growth (Amor and Stevens, 1976). The continuous use of roads by vehicles may further modify the plant habitat by impacting the trackside environment with emissions from exhaust fumes (Morgan, 1998), or raised dust may cover plants and inhibit various physiological processes (Farmer, 1993).

Under the semi-arid conditions in our South Australian study area, alterations in the physical and chemical environment next to recreational tracks are particularly potent at instigating changes in the community composition as native plants are adapted to a normally very limited water and nutrient supply (Friedel et al., 1993). Recreational tracks may therefore facilitate the establishment of invasive, non-native species that are well-known for their proficiency to withstand modified environmental conditions (Liddle and Greig-Smith, 1975), particularly if competition with other disturbance-sensitive species is alleviated (Frenkel, 1970). In Australian grasslands, for instance, a greater richness of exotic species adjacent to roadsides has been attributed to higher nutrient concentrations from vehicle emissions which fostered non-native species growth and suppressed the growth of native species (Morgan, 1998). Likewise, trailsides were susceptible to species invasions (Hall and Kuss, 1989; Tyser and Worley, 1992).

Propagule pressure, defined as the quality, quantity and frequency of invading organisms (Groom et al., 2006), was proposed as a key deciding factor for the successful establishment of introduced populations (Lockwood et al., 2005). Propagule sources accrued locally were considered important in some cases where the abundance of non-native species along transportation corridors may not have imposed sufficient pressure to sustain colonising populations (Levine, 2001; Sullivan et al., 2009). In our study area we suspected that nearby pasturelands exposed to intense grazing, logging and land development provided an abundant local supply and constant influx of non-native species via transport corridors and waterways.

We conducted our study in selected gorges in the Flinders Ranges, a popular tourism destination in South Australia where some of the gorges are accessible via unpaved but well-maintained backcountry roads and others are restricted to hiker access. The central question was how tourism impact indicators (Belnap, 1998) respond to an increase in usage along roads or hiking trails. To address our question we assessed vegetation variables and soil compaction at the 1—10 m distance to roads or hiking trails at high and low usage tourism sites. We focused on this particular distance band as we expected that it would encompass the zone of greatest environmental change due to road (Godefroid and Koedam, 2004) and trail (e.g., Benninger-Truax et al., 1992) usage.

Even though many of the described impact mechanisms may affect vegetation communities equally along roadsides and trailsides in our study area, we suspected that changes in plant metrics and soil compaction emanating from an increase in usage would be more severe and more pervasive along roads than along hiking trails for several reasons. Most visitors explore vehicle-access gorges throughout their entire length but camp or stop only at some of multiple sites (Wolf et al., 2012). In contrast, hikers concentrate their activities (such as hiking and break stops) around their entry from favoured access points through to the middle of hiker gorges and rarely pursue any camping activities whatsoever. Consequently, high usage regions of roads result from greater camping usage and stopping of visitors whereas they result from a greater number of passing and stopping visitors along trails. Camping, being a temporally extended and physically more involved form of usage, should aggravate impacts. Impacts on plants and their habitat may further be exacerbated along roads because vehicles cause heavy-weight trampling and pollution; particularly in high usage sites where camping or stopping visitors manoeuvre their vehicles more in order, for instance, to seek optimum parking. Roads that receive frequent usage also require maintenance (e.g., grading) which typically affects the surroundings more than the maintenance needed for well-used trail sections.

The following non-exclusive hypotheses were tested: (1) Plant metrics and soil compaction will differ between high and low usage sites. (2) Changes in univariate plant metrics and soil compaction emanating from an increase in usage will be more severe and more pervasive along roads than along hiking trails. (3) Species' reactions to tourism usage will manifest through multivariate compositional changes between high and low usage sites. (4) Impacts such as an increase in species that thrive on disturbed sites will self-perpetuate from tracks to other sites where conditions are naturally disturbed. This will increase the ecological effect zone to the banks of neighbouring creek beds that traverse recreation sites.

2. Methods

2.1. Study area

This study was conducted in a popular tourism destination in South Australia, the central and northern Flinders Ranges, from the Flinders Ranges National Park (Wilpena: lat. 31° 30′ S, long. 138° 30′ E) into the Vulkathunha-Gammon Ranges National Park (Balcanoona: lat. 30° 30′ S, long. 139° 30′ E). The geomorphologically diverse Flinders Ranges encompass six bioclimatic regions (Nix, 1982) and provide a versatile mixture of habitats for a rich vegetation community with a record of 1233 native plants, including more than 200 species under conservation threat and 14 endemic taxa (Brandle, 2001). Given the sporadic rainfalls that vary from approximately 200 to 500 mm per annum (Brandle, 2001) much of the vegetation is typical of semi-arid communities (Kuchel, 1980). The vegetation of the Flinders Ranges is adapted to sporadic and unpredictable rainfall and low nutrients as typical of the southern Australian rangelands (Caughley, 1987). The ephemeral vegetation is fast-growing and short-lived but the perennials are typically slow-growing and long-lived (e.g., saltbushes and bluebushes) and have low resilience to degrading impacts (Freudenberger et al., 1997) which may manifest from tourism use.

Our study focused on gorges as they attract intense visitor traffic due to their iconic and scenic values. Moreover, they support high plant species richness due to their propensity to retain water and to provide shady refuges from the drier, surrounding plains. With an average of approximately 70 plant species per site, gorges hosted by far the richest plant community compared to 14 other landform elements that were assessed in a comprehensive, biological survey of the Flinders Ranges (Brandle, 2001). Morton et al. (1995) in their

analysis of refugia for biological diversity in arid and semi-arid Australia listed the Flinders Ranges as a refuge of "*extremely significant*" quality.

Typically, gorges are traversed by intermittently flowing creek beds that are fringed by riparian woodlands of *Eucalyptus camaldulensis var.* and common understory species such as *Melaleuca glomerata* or *Myoporum montanum* as well as numerous ephemeral herbs. There are two varieties of *E. camaldulensis* present including *E. camaldulensis var. camaldulensis* and *E. camaldulensis var. obtusa*. These riparian eucalypt communities provide habitat for species in an otherwise inhospitable environment. The community was described by Neagle (2003) as a *E. camaldulensis* woodland group that occurs on levees and along drainage lines scattered across the Flinders Ranges. Though the associated vegetation was noted to be variable, some shrubs are ubiquitous including *Acacia victoriae*, *Senecio magnificus*, *Enchylaena tomentosa*, *Solanum sturtianum* and *M. montanum*. The wetter conditions of some areas attract sedges such as *Cyperus gymnocaulos* and rushes such as *Cyperus vaginatus*. The community occurs on a variety of loam soils in the higher rainfall areas of the Flinders Ranges (Brandle, 2001) and on clays in outwash areas from the ranges (Brandle, 1998).

Occasional, heavy flows of water can sweep through the watercourses out to the surrounding plains. During our study period from July to December in 2006 and 2007 no substantial rainfall events occurred and the creek beds were largely dry except for some pockets of water where drainage was impeded. In these damp areas sedges and rushes proliferated. Much like the rest of the Flinders Ranges with their 300 non-native plant taxa (Brandle, 2001)—including proclaimed weed plants such as *Echium vulgare*—the gorges have been subjected to the invasion of non-native species such as *Rumex vesicarius*, *Sisymbrium erisimoides* and *Cirsium vulgare*. The nomenclature and taxonomy for plant species follows Barker et al. (2005).

2.2. Study design

Visitors experience the gorges from a network of roads and trails that provide access to a variety of official and unofficial campsites. We selected seven major gorge systems, three of which were mainly accessed by vehicles and four were exclusively accessible to hikers. In either gorge type, we established transects at 40 sites, a minimum of 250 m and usually not more than 500 m apart. The chosen areas (Fig. 1) were on average (±1SE) 92.5 ± 3.6 m wide and had to contain a distinctly marked recreational track (roads: 5 ± 0.2 m wide; trails: 1.2 ± 0.15 m wide) and a creek bed (11.6 ± 0.9 m wide), which were by at least 30 m (46.9 ± 4.5 m) from the track. Roads were unpaved but graded and composed of gravel, dirt or a mix of the two.

Because we sought a comparison between lightly and heavily used areas, we first classified our study sites as low or high usage ($n_{(low\ usage\ roads)}$ = 21; $n_{(high\ usage\ roads)}$ = 19; $n_{(low\ usage\ trails)}$ = 22; $n_{(high\ usage\ trails)}$ = 18) based on differences in the number of passing tourists, their passing speed, the percentage of stopping tourists, their stop time, the number of day- and night-campers and the average camp-time by day which we had estimated in transects ("visitor census plots") throughout our gorges (Wolf et al., 2012). Further, we included proxies which reflect usage (inferred from personal observations): the size of traversing and adjacent (boundaries situated within 30 m to the visitor census plots) campgrounds, the numbers of fire places, trash items and interpretation signs. Visitor census plots, which contained the plant census plots (see next paragraph), were placed so there was not more than approximately 10% of overlap with traversing campgrounds as we did not intend to measure the effect of camping per

se but of recreational tracks. On either side of the recreational track (Fig. 1b,c) and the creek bed (Fig. 1d) at these pre-determined sites, two belt transects of 50 m × 10 m were established. Within each transect, ten 1 m × 1 m frame quadrats (Kent and Coker, 1995) were sampled at a 1−10 m distance to the track or creek bed bank, so that each distance was replicated (*n* = 2) at each transect (Fig. 1a). Within each distance band sampling quadrats were located randomly. The sampling was restricted to a distance of up to 10 m on either side of tracks due to (1) sampling constraints imposed by the large number of sampling quadrats and geographically dispersed locations which required us to focus on the zones where impacts were most likely; (2) knowledge that effect-distances of trackside impacts, especially for trails, are typically ≤15 m (see Section 4.2). We recognised that impacts may extend beyond 15 m along roads (see Section 4.2) and so we undertook a pilot sampling and sourced knowledge from local park management. These indicated that distance gradients from roads were locally restricted and similar to that found along trails. We also sampled creek beds and these were separated by at least 30 m from tracks and so one distance >30 m to tracks was also included in the design.

Although the creek beds were mostly dried out, the marked succession from pure rock to mineral soil and the presence of plants was used to demarcate the bank for the placement of the first sampling quadrat. A single observer visually estimated the species-cover of all living, vascular plant species 1.5 m or less in height as the area of the sampling quadrat covered by the vertical projection of the above-ground parts of the plants. The majority of vegetation in our study area, with the exception of trees and a few larger shrubs, did not exceed 1.5 m in height and plant cover could be measured reliably up to this height using our approach. Further to this we had specific hypotheses about the effects of tourism usage and underlying mechanisms such as trampling which were most relevant to lower vegetation strata.

The percentage of total plant cover at each quadrat was calculated as the sum of individual species covers. As vegetation was layered, percentages of overlapping cover may sum to more than 100% (Sutherland, 2006). Species diversity was expressed through the heterogeneity diversity index Simpson's $E_{1/D}$ (advantages of this measure over its alternatives were reviewed by Magurran, 2004). Nomenclature and status as native or non-native species to the Flinders Ranges followed Barker et al. (2005).

In each transect of 50 m × 10 m, three measurements of soil compaction were taken with a hand-held dial penetrometer (Pocket Geostester, Zoli Maurizo, Italy) at each distance from the track or creek bed. The first measurement was taken in the centre of each transect and two further measures were taken on either side at a 10 m-distance to the centre point. The penetrometer measured the force (in kg cm^{-2}) necessary to penetrate the upper soil stratum (Lipiec and Hatano, 2003). Recorded compaction values refer to the maximum value read from the dial of the instrument as it was pressed onto the soil surface until fully compressed or the full length of the shaft (approximately 50 mm) was inserted into the soil. The average value of the multiple readings was used in the statistical analyses.

2.3. Data analyses

We used multivariate analysis to examine differences in composition in relation to the usage intensity and distance from roads and trails, and visualised existing differences with nMDS plots, followed by Indicator Species Analysis (Dufrêne and Legendre, 1997) to identify species indicative of high or low usage conditions. Univariate analyses were used to compare mean values of the percentage of total plant cover and of non-native species
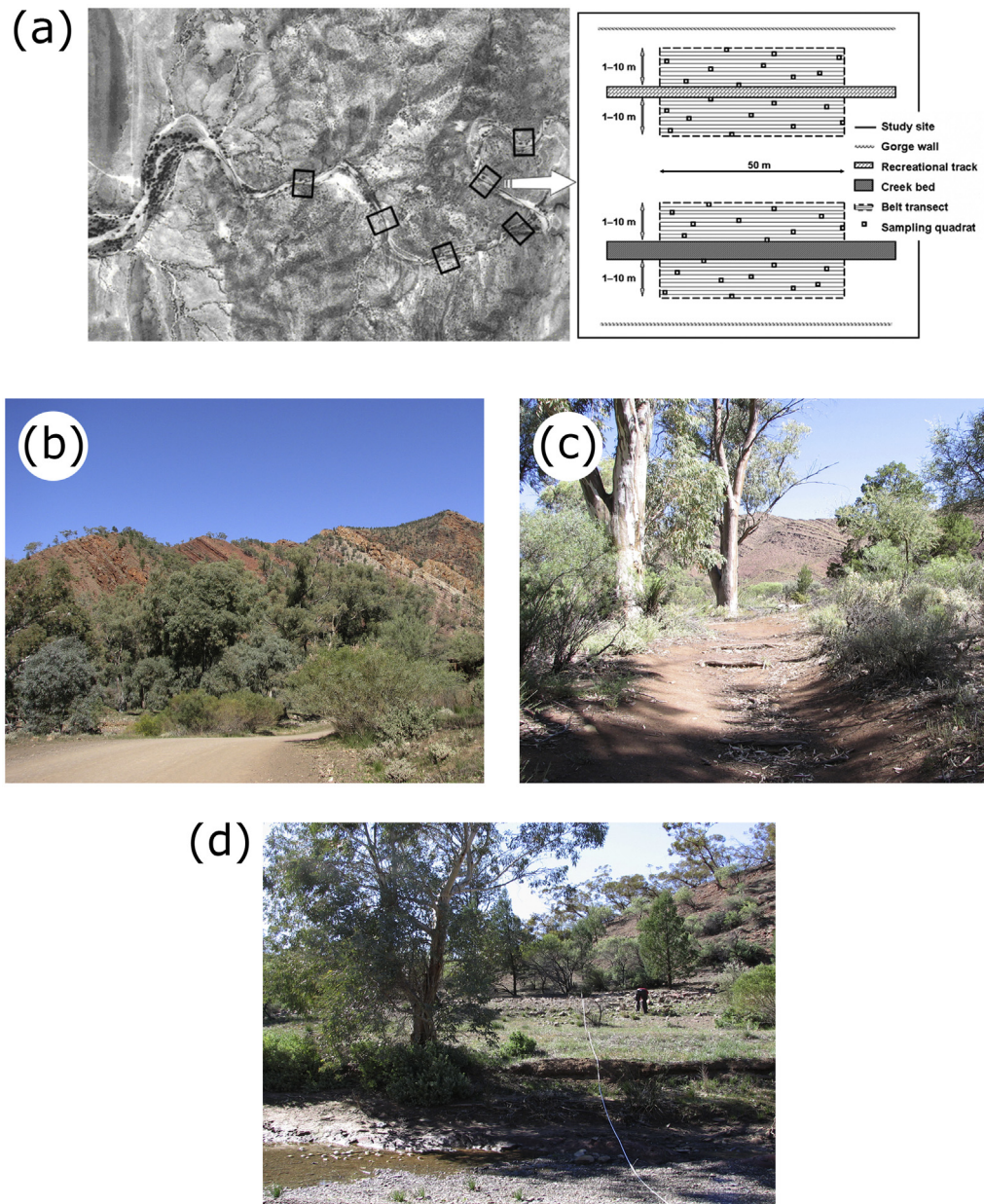
Fig. 1. Illustration of the experimental set up by (a) a sample of study sites in Brachina Gorge (lat. 31° 34′ S, long. 138° 56′ E; aerial photograph, survey/frame no. 2320/24, reproduced with permission of the South Australian Department of Lands) and the distribution of sampling quadrats within each study site. On either side of the recreational track and the creek bed, two belt transects of 50 m × 10 m were established. Within each transect, ten 1 m × 1 m frame quadrats were distributed. Representative sections of (b) a road verge, (c) a trail verge and (d) a creek bed.

cover, Simpson's diversity index $E_{1/D}$ and soil compaction in relation to the usage intensity and distance from roads and trails.

To determine the variation in plant community composition adjacent to roads and trails and in relation to usage intensity and distance from such tracks, species cover data along tracks were analysed with permutational multivariate analysis of variance+, PERMANOVA+ in PRIMER v6 (Clarke and Gorley, 2006). Prior to analysis data were square-root-transformed to downweight high-abundance species in accordance with Clarke and Gorley (2006). No further transformations were necessary as PERMANOVA+ does not impose parametric assumptions. A four-way factor design was performed on a Bray–Curtis similarity matrix of the species data with 999 permutations and type III sum of squares.

The experiment was examined as a split-plot design (Quinn and Keough, 2004). Study sites (random, vehicle gorges: 40 levels, hiker gorges: 40 levels) were treated as the experimental whole-plot units/subjects that were nested in the between-subject treatments 'gorge' (random, vehicle gorges: 3 levels, hiker gorges: 4 levels) and 'usage intensity' (fixed, 2 levels), the latter two factors being crossed with each other. The within-subject treatment 'distance' (fixed, 10 levels), which was crossed with all other factors, was 'applied' to sampling quadrats (i.e., sub-plot units; $n = 800$) that were randomly distributed within the belt transects to ensure an equal correlation of all pairs of measures in the same whole-plot unit. The two samples from the same distance to the track at each site represent replication ($n = 2$) at the lowest design level. The random factors, 'gorge' and 'site', allowed us to generalize about spatial variation in response to disturbance created by recreational tracks. Final models were extracted by excluding factors with P-values > 0.25 (Underwood, 1997) from initial models in a manual,

stepwise backward selection procedure (Crawley, 2007). Differences among levels of a factor were identified with post hoc pairwise tests (999 permutations). These also allowed specification of simple main effect tests (Field, 2005) as a follow-up to significant interactions to identify the level of one factor at which significant differences of the other factor occurred.

Multivariate patterns were visualized by ordinating the sampling quadrats in a two-dimensional species-space with non-metric multidimensional scaling (nMDS) in PRIMER v6. To reduce the number of displayed sampling quadrats, we averaged the distance replicates and randomly chose one vehicle and one hiker gorge to represent community patterns at the 1–5 m and 6–10 m distance from tracks in relation to usage intensity. The pattern of distinction of sites by their species community based on high and low usage that we discuss in the results was present in all gorges. Here and in the following, we selected two distance bands of 1–5 m and 6–10 m, respectively, for analysis because effect zones of tourism usage were apparent up to a threshold at 5 m and by using this we simplified the analysis in order to distinguish the key effects.

To examine which species were consistently associated with high or low usage conditions at the 1–5 m and 6–10 m distance, we conducted an Indicator Species Analysis (Dufrêne and Legendre, 1997) from the PC-ORD package (McCune and Mefford, 1997). The perfect indication value of 100 (with 0 being the minimum) occurs for a species that is present in all sites belonging to one particular factor level, and absent in all others. A Monte Carlo randomization procedure tested the indicator values for statistical significance.

Univariate analyses of the same relationships that had been investigated for the multivariate data were conducted by fitting four-way factor ANOVA models on the mean percentage of total plant cover and of non-native species cover, the mean Simpson's diversity index $E_{1/D}$ and mean soil compaction. Initial models were reduced following the same procedure applied for the selection of the final PERMANOVA+ models. Significant interaction effects were followed-up with simple main effect analyses. For consistency, we have presented the results of the simple effects for usage intensity × distance even when this interaction was not significant, which was the case for one model where the P-value had just exceeded the level of significance.

To test whether the univariate effects of the usage intensity and distance were not only prevalent adjacent to tracks but also creek beds, univariate variables recorded close (1–5 m) to tracks were compared with those recorded close to creek beds and contrasted with the data gathered far (6–10 m) from tracks and creek beds. Therefore, a conjoint factor 'section/block-distance' (fixed; 4 levels: close/track, close/creek, far/track, far/creek) was created. This factor was used to replace the factor 'distance' in the design that was consistently applied in the previous analyses. A significant interaction of the section/block-distance with usage intensity was followed-up with simple effect analyses.

Data were transformed as needed to approximate the assumptions for ANOVA (Quinn and Keough, 2004). Denominator degrees of freedom that are not integers indicate a numerical approximation with the Satterthwaite (1946) method. All transformations ($x' = \sqrt{x}$; $x' = \log 10(x + 1)$) and univariate statistical analyses were carried out with SPSS for Windows 17.0. We accepted the 5%-level of alpha as a threshold to decide whether to reject or fail to reject a null hypothesis. Means ± 1 SE are presented unless otherwise indicated.
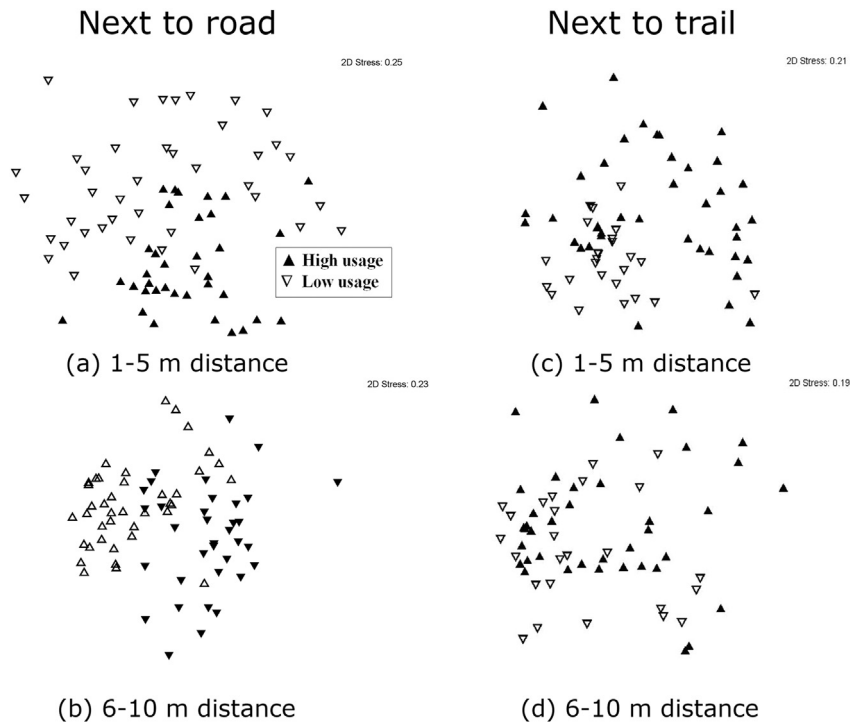
3. Results

3.1. Species community and indicator species

A total of at least 126 plant species belonging to 38 families and 86 genera was recorded in the 1600 1 m × 1 m sampling quadrats with two genera (Bromus sp., Juncus sp.) not identified to species level. The three most common Families were the Chenopodiaceae, Fabaceae and Asteraceae with 16, 16 and 13 species, respectively. A fifth (20.6%) of the recorded species were not native to the Flinders Ranges.

The overall community composition of vegetation growing along roads (Table 1.1) and trails (Table 1.2) was influenced by the interaction between the usage intensity and distance. At the 2–3 m distance plant assemblages supported along both high usage roads and trail sections were different from those growing along low usage sections, which was consistent across gorges. Assemblages inhabiting the 1st metre from roads were different from those growing between the 5th and the 10th metre; along two roads, even the vegetation communities growing at the 2–3 m distances differed from those growing further away. Trailside gradients were somewhat more uniform and impacts on plant communities reached further adjacent to high usage trails: Assemblages growing at 1–3 m from high usage trail sections were different from assemblages growing at the 9–10 m distance whereas assemblages growing at 1–3 m from low usage trails were different from assemblages growing at any further distances. The nMDS ordination based on floristic composition of the two gorges we have chosen to display clearly stratified sites by usage intensity both at 1–5 and 6–10 m from the road (Fig. 2a,b) and at 1–5 m from the trail (Fig. 2c,d). Thus, the nMDS results illustrated a more extreme difference in communities based on usage intensity than the PERMANOVA+ as the differences also persisted further from the tracks. However, these results need to be interpreted with caution as the stress of the nMDS plots was high (≥0.2).

There was a considerable overlap in high (27.1%) and low (41.6%) usage indicator species between roads and trails. Very few indicators (Dufrêne and Legendre, 1997) were found for low usage conditions, whatever the distance or the access mode to the gorge, compared to the greater number of species that were indicative of

Table 1
Final PERMANOVA+ models including all main terms and interactions which significantly (bold values) explained variation in plant community composition adjacent to (1) roads and (2) trails.

	(1) Next to roads			(2) Next to trails		
	df	Pseudo-F	P(perm)[a]	df	Pseudo-F	P(perm)[a]
Usage intensity	1, 2.01	1.42	0.196	1, 4.10	2.62	**0.010**
Gorge	2, 34	10.90	**0.001**	3, 32	2.28	**0.002**
Usage intensity × gorge	2, 34	1.71	**0.011**	3, 32	0.81	0.740
Site(usage intensity × gorge)	34, 724	4.47	**0.001**	32, 715	5.04	**0.001**
Distance	9, 18.05	2.07	**0.004**	9, 32	2.41	**0.001**
Usage intensity × distance	9, 724	1.39	**0.004**	9, 715	1.49	**0.001**
Gorge × distance	18, 724	1.61	**0.001**	27, 715	1.19	**0.009**

Note: Terms for which P(perm) > 0.25 (Underwood, 1997) were excluded from final models (denoted as 'NA') unless they figured in higher order or nested terms. The factors 'site(usage intensity × gorge) × distance' and 'usage intensity × gorge × distance' are not listed because of P(perm) > 0.25 for all effects.
[a] In PERMANOVA the distribution of the pseudo-F statistic is obtained by using a permutation procedure.

Fig. 2. NMDS plots of sampling quadrats based on floristic composition at the (a, c) 1–5 m vs. (b, d) 6–10 m distance showing differences in plant assemblages between high (▲) and low (▽) usage sections of a road in Brachina Gorge (lat. 31° 34′ S, long. 138° 56′ E) and a trail in Weetootla Gorge (lat. 30° 49′ S, long. 139° 24′ E). No scales are shown on the axes as the orientation of a nMDS diagram is arbitrary (Clarke and Gorley, 2006).

high usage conditions, particularly at the 1–5 m distance from tracks (Table 2.1a, b; Table 2.2a, b). Very conspicuous was the high number of non-native species in high usage conditions of roads and trails; notably among them, were two proclaimed weed species, *Asphodelus fistulosus* and *Marrubium vulgare*. Species with preference for high usage conditions, however, also included native species. In fact, at the further distances and along trails, the disturbed 'niches' of the plant habitat were increasingly occupied by native species (fraction of native vs. non-native species at 1–5 m from roads: 0.2; at 6–10 m: 0.6; at 1–5 m from trails: 0.7 and at 6–10 m: 1.7). Other native species such as *Rhagodia parabolica*, *Rhagodia spinescens* and *Swainsona phacoides* preferred the presumably least disturbed sites at the 6–10 m distance from low usage recreational tracks. Further notable was that all low usage indicator species were perennial whilst at least half of the high usage indicator species were annual, the majority of them being non-native species. This was expected as disturbance is known to favour annuals (McIntyre et al., 1999).

3.2. Total plant and non-native species cover, plant diversity and soil compaction

The univariate vegetation variables and soil compaction also strongly depended on the interaction between the usage intensity and the distance from roads and trails (Table 3, Fig. 3).

The fact that none of the dependent variables exhibited an ostensible gradient with distance from low usage trail sections suggested that low usage had a negligible effect on the trailside environment. Only the percentage of non-native species cover decreased slightly with distance from low usage trails (Fig. 3f). In contrast, as one moved away from high usage trails the percentage of total plant cover increased and the percentage of non-native species cover, plant diversity and soil compaction decreased (Fig. 3e–h). The gradients were strongest within the first 4 m.

However, soil compaction was only significantly higher directly beside trails compared to other distances. Significant differences between high and low usage trail sections were also confined to the first 4 m where high usage conditions were associated with a reduced percentage of total plant cover (1st to 3rd m) and increased percentage of non-native species cover (1st to 4th m), plant diversity (1st to 3rd m) and soil compaction (1st m) (Fig. 3e–h).

Likewise, towards low usage roads there was a comparatively weak increase in the percentage of non-native species cover but also in soil compaction (Fig. 3b and d). However, vegetation growth was substantially enhanced as indicated by the marked increase in the percentage of total plant cover at 2–4 m from low usage roads compared to any other distances (Fig. 3a). In contrast, at the verge of high usage road sections no increase in plant cover was discernible but rather there was a slight decline in the 1st m compared to the other distances. Thus unfavourable conditions may have prevented vegetation benefiting from the growth-enhancing conditions prevalent close to low usage road sections. Further, the mean percentage cover with non-native species was significantly greater up to 4 m from high usage road sections compared to further away (Fig. 3b). Plant diversity increased over the first 3 m up to a maximum, where it was distinctly higher than along low usage road sections; after this point it dropped to significantly lower values, similar to the ones observed in the vicinity of low usage road sections (Fig. 3c). Soil compaction decreased with distance to high usage road sections over the first 5 m (Fig. 3d). Significant differences between high and low usage road sections occurred up to 5 m with a reduction in the percentage of total plant cover (2nd to 5th m) and an increase in the percentage of non-native species cover (1st to 4th m), plant diversity (1st to 5th m) and soil compaction (2nd to 4th m) at high usage sections (Fig. 3a–d). Similar plant cover and soil compaction values were recorded directly beside low and high usage road sections and thus suggest that the former zone had suffered from a similar degree of

Table 2

Plant species that were significantly indicative (Dufrêne and Legendre, 1997) of (a) high or (b) low tourism usage intensity at a 1–5 m vs. 6–10 m distance from (1) roads or (2) trails (IV = indicator values = % perfect indication; ranging from 0 to 100).

Scientific names	(1) IV next to roads		(2) IV next to trails	
	1–5 m	6–10 m	1–5 m	6–10 m
a) High usage indicators				
Ajuga australis		2.6		
Anagallis arvensis[a]	4.9		3.6	
Asphodelus fistulosus[a]	30.6	16.5		
Atriplex stipitata			3.7	
Carthamus lanatus[a]	4		3.1	
Cassinia laevis			2.8	2.2
Centaurea melitensis[a]	19.2		13.9	
Citrullus colocynthis[a]	2.6			
Echium plantagineum[a]	22.3	11.5	17.5	13.3
Fumaria muralis[a]				2.8
Marrubium vulgare[a]	7.4			
Medicago minima var. *minima*[a]	8.3	3.7	6.6	
Medicago praecox[a]	14.5	3.7	9.5	
Nicotiana velutina	2.1			
Portulaca oleracea	4.6			
Ptilotus obovatus var.[b]		15.6	11.1	16.2
Salvia verbenaca[a]		2.6		3.3
Sclerolaena cuneata				11.2
Solanum ellipticum			12.4	9.2
Sonchus tenerrimus[a]	5.9		4.5	
Zygophyllum apiculatum	8.3		4.7	9.2
(b) Low usage indicators				
Cymbopogon ambiguus				12.4
Cyperus alterniflorus				7.7
Cyperus gymnocaulos	27.3	10.2	25.4	10.2
Eremophila freelingii		2.9		3.2
Olearia decurrens	3.3		3.1	
Rhagodia parabolica		4.3		4.9
Rhagodia spinescens		11	6.1	10.9
Rumex brownii	4.7		4.5	
Swainsona phacoides		7.1		7.3

[a] Non-native species; denoted in bold if 'proclaimed' in South Australia; i.e., landholders are legally obliged to control them (http://www.wmssa.org.au/weeds.htm).

[b] Both *Ptilotus obovatus var. obovatus* and *Ptilotus obovatus var. griseus* occur in the study area and were not further differentiated for the purpose of this study.

disturbance to the latter despite the overall lower usage prevailing at the whole site.

Even though the interaction effect of usage intensity and distance on the percentage of non-native species cover (Table 3.1b) and plant diversity (Table 3.1c) was inconsistent across vehicle gorges, as inferred from the triple interactions, a closer inspection revealed that the patterns were similar across all gorges albeit more or less pervasive. For instance, the increase in the percentage of non-native species cover from low to high usage sites affected the roadsides up to 7 m at Brachina Gorge (lat. 31° 34′ S, long. 138° 56′ E) whereas the other two vehicle gorges were affected only up to 4 m.

Tourism impacts were not confined to the trackside but also exerted their effect to the banks of intermittently flowing creek beds neighbouring to high usage road sections (Table 4.1, Fig. 4). Both the percentage of non-native species cover (Table 4.1b; Fig. 4b) and plant diversity (Table 4.1c; Fig. 4c) were greater within 1–5 m from creek beds under high usage conditions consistent with the findings from the roads. The fact that the effects were much smaller at 6–10 m from roads and creek beds implied that the ecological effect zone of tourism impacts was not continuous but prevailed at two disjointed areas (Table 4.1b, c; Fig. 4b, c). Interestingly, plant cover along creek beds showed the opposite trend to roads with a somewhat higher mean percentage of total plant cover under high usage conditions (Table 4.1a; Fig. 4a). Along creek beds

neighbouring high usage trail sections, plant diversity was also slightly higher compared to low usage trail sections but the effect size was very small (Table 4.2c; Fig. 4g).

4. Discussion

4.1. Effect of usage intensity and distance from roads and trails on trackside vegetation and soil compaction

Our results show that the tourism usage intensity influenced the condition of trackside vegetation and soil compaction depending on the distance to roads or hiking trails. Next to low usage road sections plants grew substantially better at the 2–4 m distance than closer or further away. Greater water runoff from roads towards their edges (Norton and Smith, 1999) and a higher nutrient availability from exhaust emissions (Angold, 1997) may have stimulated plant growth as both water and nutrients are limited resources in arid ecosystems (James et al., 1995). Even though an increased water runoff from trails is frequently reported as an adverse condition that promotes trail degradation and soil erosion (Deluca et al., 1998), it has only marginally been discussed as a promoter of vegetation growth at trailsides (Bright, 1986). The mere existence of a track did not however ostensibly change vegetation or soil compaction, as evidenced along low usage hiking trail sections. These were narrow and not sufficiently compacted to facilitate neighbouring plant growth. Further, the input of additional nutrients from the sporadic hiking traffic was likely negligible compared to inputs from vehicle emissions.

Although roadside conditions imposed on by tourism usage were not so adverse as to override the above discussed growth-enhancing effects, they clearly affected the vegetation closest to the road. Trampling stress, for instance, exerted from passing or parking vehicles, foot traffic and maintenance activities may affect vegetation in the vicinity of roads and to a lesser extent along trails (reviewed by Liddle, 1997). These studies conform with our findings in that the difference in total plant cover between high and low usage conditions was somewhat greater and more noticeable further from roads than trails. Another consequence from trampling is soil compaction which may have impeded plant growth in high usage areas and up to 5 m to tracks. Our findings match the spatial extent of increases in soil compaction reported by Godefroid and Koedam (2004) along forest roads. Unsurprisingly, soil compaction changes were greater and more pervasive near roads than trails as compaction strongly depends on the amount of pressure applied (Liddle, 1997). Exhaust emissions and dust pollution may have further compromised plant growth (Bignal et al., 2007). In fact, we witnessed numerous patches along roads where plants were entirely covered in dust, sometimes up to several millimetres in depth.

High levels of tourism usage substantially changed the floristic composition with different species thriving along the gradient of disturbance. In particular, non-native species cover increased towards high usage roads and trail sections although the effect was greater and more pervasive near roads. A similar though weaker trackside gradient in non-native species cover existed under low usage conditions. Apparently, disturbance along low usage tracks was too weak to substantially enhance growth of non-native species. Our findings are consistent with others who have reported changes in community composition, increases in non-native species cover, and diversity along roads (Morgan, 1998), trails (Benninger-Truax et al., 1992) and other types of edges (Fox et al., 1997; Luken et al., 1991). Tracks may act as conduits for the spread of exotic species (Lonsdale and Lane, 1994) as human objects become vectors for reproductive units of invasive plants. The trackside edge then favours the establishment and growth of those

Table 3
Final ANOVA models including all main terms and interactions which significantly (bold values) explained variation in (a–c) vegetation variables and (d) soil compaction adjacent to (1) roads or (2) trails.

	(1) Next to roads			(2) Next to trails		
	df	F	P	df	F	P
(a) Total plant cover (summed % of overlapping cover per sampling quadrat)						
Usage intensity	1, 2.02	15.01	0.060	1, 3.67	5.29	0.089
Gorge	2, 1.98	3.97	0.203	3, 3.73	0.75	0.578
Usage intensity × gorge	2, 34	0.52	0.599	3, 32	0.50	0.688
Site(usage intensity × gorge)	34, 742	6.61	**<0.001**	32, 715	8.56	**<0.001**
Distance	9, 742	8.18	**<0.001**	9, 30.18	0.39	0.930
Usage intensity × distance	9, 742	4.11	**<0.001**	9, 715	2.53	**0.007**
Gorge × distance	NA, NA	NA	NA	27, 715	1.83	**0.007**
Usage intensity × gorge × distance	NA, NA	NA	NA	NA, NA	NA	NA
(b) Non-native species cover (% of total plant cover)						
Usage intensity	1, 2.01	8.83	0.096	1, 3.86	10.58	**0.033**
Gorge	2, 0.06	201.38	0.764	3, 2.59	3.37	0.194
Usage intensity × gorge	2, 40.86	0.38	0.686	3, 32	0.39	0.760
Site(usage intensity × gorge)	34, 706	6.03	**<0.001**	32, 742	5.40	**<0.001**
Distance	9, 18.41	27.53	**<0.001**	9, 742	12.32	**<0.001**
Usage intensity × distance	9, 18.06	2.24	0.070	9, 742	2.75	**0.004**
Gorge × distance	18, 18	0.15	1.000	NA, NA	NA	NA
Usage intensity × gorge × distance	18, 706	2.89	**<0.001**	NA, NA	NA	NA
(c) Simpson's diversity index E$_{1/D}$						
Usage intensity	1, 2.01	12.49	0.071	1, 3.39	1.87	0.255
Gorge	2, 1.10	10.16	0.196	3, 3.96	1.54	0.336
Usage intensity × gorge	2, 36.46	0.80	0.456	3, 32	0.85	0.479
Site(usage intensity × gorge)	34, 706	4.13	**<0.001**	32, 715	4.73	**<0.001**
Distance	9, 18.17	6.30	**<0.001**	9, 30	3.94	**0.002**
Usage intensity × distance	9, 18.09	2.89	**0.026**	9, 715	3.34	**0.001**
Gorge × distance	18, 18	0.50	0.922	27, 715	1.83	**0.001**
Usage intensity × gorge × distance	18, 706	2.06	**0.006**	NA, NA	NA	NA
(d) Soil compaction						
Usage intensity	1, 2.10	48.26	**0.018**	1, 3	1.18	0.349
Gorge	2, 1.96	226.76	**0.005**	3, 2.8	1.11	0.474
Usage intensity × gorge	2, 34	0.09	0.910	3, 32	0.82	0.501
Site(usage intensity × gorge)	34, 742	6.92	**<0.001**	32, 742	12.71	**<0.001**
Distance	9, 742	19.05	**<0.001**	9, 742	6.07	**<0.001**
Usage intensity × distance	9, 742	4.22	**<0.001**	9, 742	2.19	**0.021**
Gorge × distance	NA, NA	NA	NA	NA, NA	NA	NA
Usage intensity × gorge × distance	NA, NA	NA	NA	NA, NA	NA	NA

Note: Terms for which $P > 0.25$ (Underwood, 1997) were excluded from final models (denoted as 'NA') unless they figured in higher order or nested terms. The factor 'site(usage intensity × gorge) × distance' is not listed because of $P > 0.25$ for all effects.

species that can cope well with disturbance which is a typical trait of non-native species.

Native species that colonize early successional stages may also be adapted to these conditions, and it was therefore not surprising to detect a considerable number of natives among the high usage indicator species. Godefroid and Koedam (2004), for instance, reported an increase in the number of ruderal species, disturbance indicators, nitrogen-demanding species and indicators of basic conditions near forest paths which included native and non-native species. They too noted very few indicator species growing in the less disturbed zones compared to the great number of species that typically inhabited the immediate areas near their roads. Interestingly, the percentage of native species indicators for high usage conditions increased with distance to the track and in hiker gorges. Presumably native species that colonize disturbed zones have advantages over non-native species when the disturbance level is intermediate rather than high.

The changes in the plant community resulted in a pronounced track-distance gradient of the plant diversity along high usage sections. The behaviour along high usage roads was unimodal over the 1–4 m distance, consistent with the predictions of the Intermediate Disturbance Hypothesis (IDH) (Connell, 1978). Presumably, only the more disturbance-tolerant species were able to survive directly beside roads whilst intermediate levels of

disturbance at a 3 m distance facilitated the invasion of non-native species and native coloniser plants without excluding native, disturbance-sensitive species. Further away, plant diversity declined sharply likely because the disturbance-sensitive species regained their competitive superiority and outcompeted those species that had invaded the more disturbed zones. Along high usage trails, plant diversity decreased with distance, which indicates a continuous decline in the disturbance gradient consistent with the downslope part of the IDH curve. Thus, except for in the first m, diversity patterns matched between roads and trails. The application of IDH may explain the opposite trends regarding the response of plant diversity when others compared more or less disturbed tourism areas or different distances from recreational tracks evidenced by decreases (Andrés-Abellán et al., 2005), increases (Bright, 1986) or a unimodal pattern consistent with the IDH (Kobayashi et al., 1997).

In some instances the causal relationships between tourism usage and some of the low usage indicators may be reversed and not an indication of a plant species' susceptibility to disturbance: the sedges *Cyperus alterniflorus* and *C. gymnocaulos* as well as *Rumex brownii* are likely associated with low tourism usage because their presence indicates conditions such as damp or swampy grounds which tourists, especially when camping, consistently avoid.
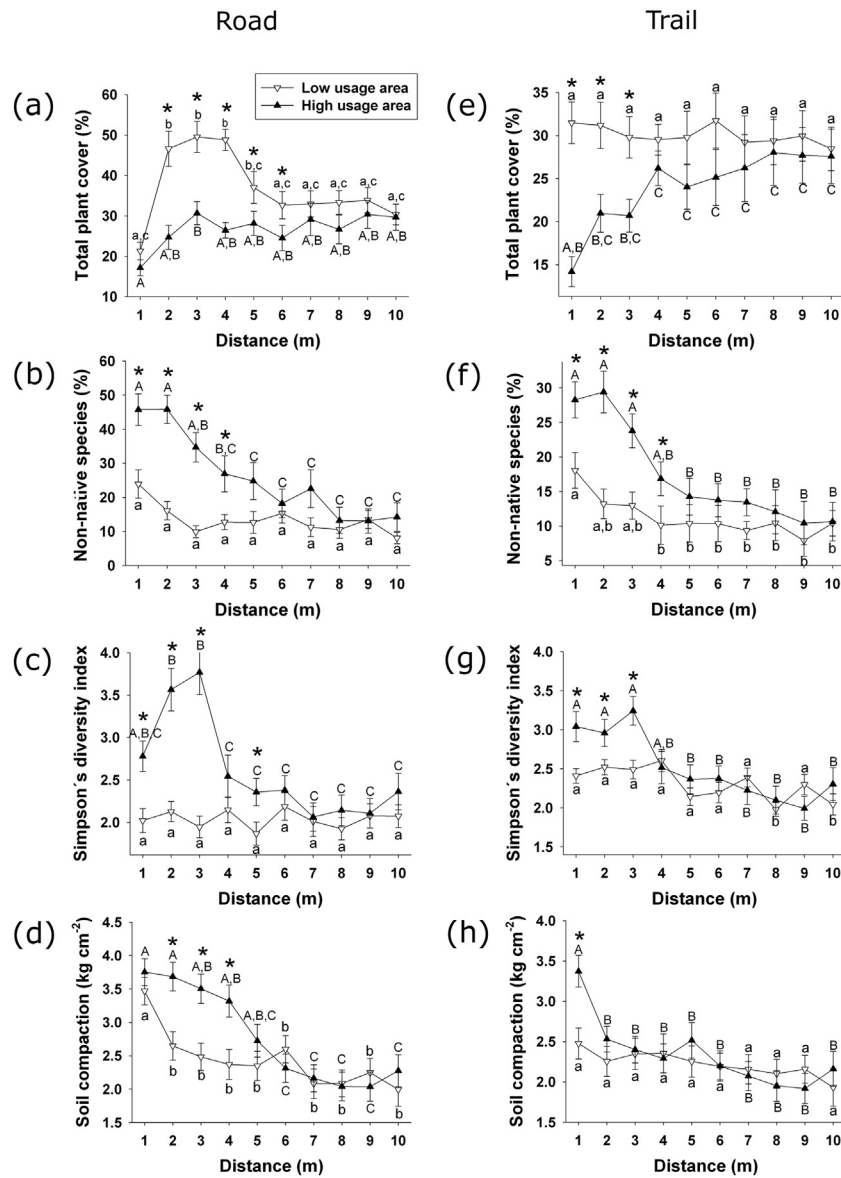
Fig. 3. Mean (±1 SE) (a, e) percentage of total (overlapping) plant cover (per 1 m^2), (b, f) percentage of non-native species cover (of total plant cover), (c, g) Simpson diversity index $E_{1/D}$ and (d, h) soil compaction in relation to the tourism usage intensity and distance to roads or trails. Asterisks indicate significant simple effects of usage intensity at a particular distance. Bars that do not share a common letter are significantly different at a particular level of usage intensity.

4.2. Spatial extent of road- and trailside impacts

Knowing the spatial extent of trackside impacts is of decisive importance for sustainable tourism management. Usually, the 'effect zone'—the area in which substantive, ecological changes take place in terms of species, soil and water (Forman and Deblinger, 2000)—extends clearly beyond the boundaries of access routes which may substantially reduce the size of the functional interior of the neighbouring ecosystems (Fraver, 1994). The spatial extent of the trackside effect zone is a convoluted function of environmental factors including topography, weather conditions, vegetation type as well as disturbance properties such as the usage intensity, itself a function of visitor numbers, individual user behaviour and type of activities. Further, track properties such as width, surface type, presence of shoulders and level of improvement are influential (Brooks and Lair, 2005). Given this complexity, it is not surprising that the width of the effect zone largely varies between studies.

Still, we noticed two major trends in the literature: short effect-distances of ≤15 m and long effect-distances of ≥50 m. Trailside effects have usually been restricted to the vicinity of the track (Benninger-Truax et al., 1992; Chizhova, 2004; Dale and Weaver, 1974; Hall and Kuss, 1989) whereas roadside effects have extended over a short (Godefroid and Koedam, 2004; Morgan, 1998; Olander et al., 1998; Watkins et al., 2003) or a long distance (Angold, 1997; Forman and Deblinger, 2000; Gelbard and Belnap, 2003; Gelbard and Harrison, 2003; Hansen and Clevenger, 2005).

In our study, trail-distance gradients were exacerbated or became noticeable only under high usage conditions. Along high usage trails, impacts mainly percolated a short distance into the trailside environment but this depended on the variable that was measured (Chen et al., 1992; Watkins et al., 2003). Our initial analysis indicated short-distance effects of roads. Unpaved roads with light usage such as ours (Godefroid and Koedam, 2004; Watkins et al., 2003), tend to exhibit narrow effect zones whilst

Table 4

Final ANOVA models including all main terms and interactions which significantly (bold values) explained variation in (a–c) vegetation variables and (d) soil compaction adjacent to creek beds inside (1) vehicle or (2) hiker gorges.

|	(1) Along creek beds inside gorges with vehicle access			(2) Along creek beds inside gorges with foot access		
	df	F	P	df	F	P
(a) Total plant cover (summed % of overlapping cover per sampling quadrat)						
Usage intensity	1, 2	2.50	0.254	1, 3.13	0.04	0.855
Gorge	2, 5	2.44	0.182	3, 2.89	1.82	0.323
Usage intensity × gorge	2, 34	0.59	0.562	3, 32	1.81	0.165
Site(usage intensity × gorge)	34, 108	3.37	**<0.001**	32, 114	2.60	**<0.001**
Sect./block-dist.	3, 6	2.39	0.168	3, 114	0.44	0.723
Usage intensity × sect./block-dist.	3, 108	9.08	**<0.001**	3, 114	5.37	0.002
Gorge × sect./block-dist.	6, 108	3.60	**0.003**	NA, NA	NA	NA
Site(usage intensity × gorge) × sect./block-dist.	108, 1440	2.64	**<0.001**	114, 1440	3.09	**<0.001**
(b) Non-native species cover (% of total plant cover)						
Usage intensity	1, 2	4.59	0.165	1, 4.75	13.92	**0.015**
Gorge	2, 2	2.14	0.289	3, 1.77	7.76	0.136
Usage intensity × gorge	2, 34	3.87	**0.031**	3, 32	0.14	0.933
Site(usage intensity × gorge)	34, 108	3.80	**<0.001**	32, 114	5.39	**<0.001**
Sect./block-dist.	3, 6	6.75	**0.024**	3, 114	13.10	**<0.001**
Usage intensity × sect./block-dist.	3, 108	8.89	**<0.001**	3, 114	2.43	0.069
Gorge × sect./block-dist.	6, 108	2.77	**0.015**	NA, NA	NA	NA
Site(usage intensity × gorge) × sect./block-dist.	108, 1440	2.11	**<0.001**	114, 1440	1.80	**<0.001**
(c) Simpson's diversity index $E_{1/D}$						
Usage intensity	1, 2	15.36	0.059	1, 3.28	2.84	0.183
Gorge	2, 4	1.85	0.267	3, 2.90	1.86	0.316
Usage intensity × gorge	2, 34	1.48	0.241	3, 32	2.38	0.088
Site(usage intensity × gorge)	34, 108	2.39	**<0.001**	32, 105	2.48	**<0.001**
Sect./block-dist.	3, 6	5.15	**0.043**	3, 9.85	2.94	**0.037**
Usage intensity × sect./block-dist.	3, 108	11.47	**<0.001**	3, 105	2.27	0.084
Gorge × sect./block-dist.	6, 108	3.21	**0.006**	9, 105	1.52	0.150
Site(usage intensity × gorge) × sect./block-dist.	108, 1440	1.61	**<0.001**	105, 1440	1.46	**0.002**
(d) Soil compaction						
Usage intensity	1, 2	1.76	0.315	1, 3.32	1.58	0.290
Gorge	2, 2	29.48	**0.033**	3, 2.73	3.20	0.196
Usage intensity × gorge	2, 34	0.98	0.385	3, 32	0.72	0.546
Site(usage intensity × gorge)	34, 114	3.12	**<0.001**	32, 114	6.72	**<0.001**
Sect./block-dist.	3, 114	12.57	**<0.001**	3, 114	5.00	**0.003**
Usage intensity × sect./block-dist.	3, 114	3.69	**0.014**	3, 114	1.04	0.379
Gorge × sect./block-dist.	NA, NA	NA	NA	NA, NA	NA	NA
Site(usage intensity × gorge) × sect./block-dist.	114, 1440	3.14	**<0.001**	105, 1440	3.20	**<0.001**

Note: Terms for which $P > 0.25$ (Underwood, 1997) were excluded from final models (denoted as 'NA') unless they figured in higher order or nested terms. The factor 'site(usage intensity × gorge) × distance' is not listed because of $P > 0.25$ for all effects.

multi-lane, substantially improved roads exert a far-reaching influence (Angold, 1997). Not only are the effect zones of paved and busier roads larger but the acuity of the impacts is often greater (e.g., Gelbard and Belnap, 2003).

As initially described we suspect that the much higher camping activity, the greater potential of vehicles to alter their environment and the more extensive maintenance required at high usage road sections compared to trails were the reasons for the greater magnitude and larger effect zone of the impacts following an increase in usage. Whilst camping most likely was a major determinant for the greater effect zone of tourism usage along roads compared to trails, intermittent breaks of tourists must have had a considerable influence as well. This is demonstrated by the fact that low usage roads received similar mean visitor numbers as high usage trails but showed weak trackside gradients—except for the plant cover changes that were best explained by water runoff rather than usage. The major discriminator between the two was the substantially higher number of visitors that were stopping on roads compared to low usage trails. This likely had an effect along high usage roads too but our design could not separate it from the camping influence.

Importantly, in our further analysis we found evidence that recreational use of roads affected the banks of creek beds even though they were on average almost 50 m away: Close to creek beds, non-native species cover and plant diversity were greater if in the neighbourhood to high usage road sections. We propose two possible explanations. Firstly, some visitors may have been travelling off-road along creek beds. Tyser and Worley (1992), for example, found unexpectedly high levels of non-native species 100 m from the closest trail which to them suggested that non-native species may have been introduced in off-trail areas either by hikers or horse riders. However, if this mechanism was relevant in our study one would have expected to also witness evidence of trampling impacts from high usage which we did not find along creek beds. In fact, plant cover near creek beds showed the opposite trend of more cover under high usage conditions. Secondly, high propagule pressure exerted from the prospering source populations of disturbance-increaser plants near high usage roads may have driven their establishment along the naturally disturbed creek banks where they met favourable growing conditions. Vegetation communities in riparian zones are particularly susceptible to infestation with non-native species as they are exposed to "*fluvial disturbance from floods and the nonfluvial disturbance regimes of adjacent upland areas*" (Gregory et al., 1991: 543). High propagule pressure combined with adequate growing conditions may even explain why the total plant cover increased
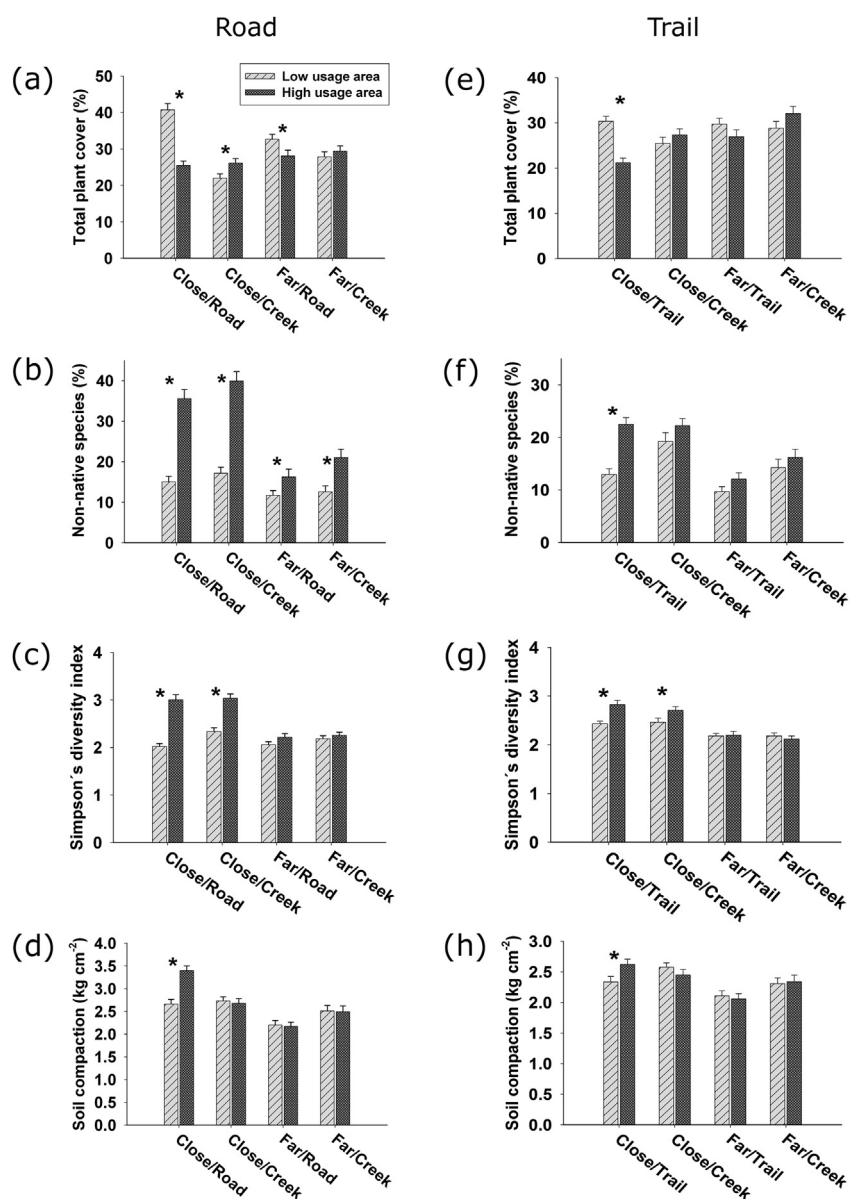
Fig. 4. Mean (±1 SE) (a, e) percentage of total (overlapping) plant cover (per 1 m^2), (b, f) percentage of non-native species cover (of total plant cover), (c, g) Simpson diversity index $E_{1/D}$ and (d, h) soil compaction in relation to the tourism usage intensity and section/block-distance (close = 1–5 m vs. far = 6–10 m from recreational tracks or creeks) to roads or trails. Asterisks indicate significant simple effects of usage intensity at a particular section/block-distance.

with high usage along creek beds where trampling impacts were not a hindrance.

To our knowledge, we are the first to demonstrate that the potential of certain impacts to self-perpetuate from their points of introduction to other, disjointed sites with a predisposition to disturbance may depend on the type of access track to a recreation site. The presence of multiple disturbances is a common phenomenon in many ecosystems (Hobbs and Huenneke, 1992) and additive as well as synergistic effects between different types of disturbance have been observed (Hodgkin, 1984; Noy-Meir, 1988). Parendes and Jones (2000: 70) have alluded to a similar finding as ours in an experimental forest in Oregon, U.S.A., where an interaction between clearcuts and roadsides were suggested as the prerequisite for the establishment of exotic species. Strikingly, we did not encounter the same pattern of impact dispersion along creek beds neighbouring trails, presumably because propagule pressure was less intense.

5. Conclusions and management implications

This study has shown that recreational tracks affect plant cover along their verge into the hinterland and contribute to a shift in species composition towards a higher proportion of non-native and native plants that thrive under disturbed conditions. Importantly, we presented evidence that the effect zone of roads was greatly enhanced because non-native species cover and plant diversity increased along the shallow banks of creek beds that neighboured high usage roads, despite an average separation of nearly 50 m. Such impacts are likely to be overlooked and consequently remain underestimated since the expectation is that disturbed vegetation reduces in a continuum from roadsides into adjacent ecosystems. In contrast, our study uncovered changes that spread between disturbed areas without manifesting in the less disturbed areas in between. This has important implications for monitoring of recreation sites which needs to consider a larger spatial extent that

encompasses other disturbance-prone sites. Native species were shown to indicate disturbance further away from tracksides which needs to be considered when selecting indicator species for monitoring of impacts.

It is typically very difficult to eradicate non-native species that are already widely dispersed in a disturbed system and exert intense propagule pressure. Notwithstanding, measures need to be taken to reduce disturbance to support a healthy and competitive, native plant community that can oppose or at least maintain a balance with intruding species. For example, visitors need to be educated and enticed by superior infrastructure to camp at a few places, preferably near the access points to recreation areas, where enforcement is low. Vehicles may be directed to sites where stopping is encouraged, for instance, by providing parking bays or information signs along the route; along hiking tracks picnic tables may serve the same purpose. However, such actions must be subtle and not attract campers to these bays or detract from the wilderness experience by creating obtrusive infrastructure or crowding.

More extreme measures such as the closure of track sections for vehicle access have successfully been implemented in one of our study areas. Along hiking trails, the middle or opposite ends of more accessible entrance points are typically substantially less frequented as we observed that visitors tend to remain within a few kilometres from where they have accessed a track. Hiker access could be restricted to one access point and then some zones would be exempted from all but minimal usage. We suspect a high visitor acceptance along tracks with one favourite access point.

In our study, a conglomerate of passing, stopping and camping usage determined the classification into high or low usage. When usage is complex and different types of visitor behaviour are interspersed, this may be the most feasible way of classifying sites and establishing dose-dependence relationships. However, where possible, independent mechanistic relationships between the amount of passing, stopping or camping usage with environmental impacts need to be established based on a continuous gradient of usage intensity, so thresholds for restricting usage can be ascertained.

Acknowledgements

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