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Comparing the impacts of different types of recreational trails on grey box grassy-woodland vegetation: lessons for conservation and management

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Abstract. Tourism and recreation are popular in natural areas but can damage plant communities, including those of high conservation value in protected areas. This includes impacts from recreational trails, but what type of trail has the most impact and why? We compared the impacts of five different trails (narrow, intermediate and wide bare earth trails, intermediate gravel trails and wide tarmac trails) on the endangered grey box grassy-woodland (*Eucalyptus microcarpa* (Maiden) Maiden) in Belair National Park near Adelaide in South Australia. First, the extent, width and area of recreational trails in the remnant woodland were mapped. Then, vegetation parameters were recorded in quadrats at three distances from the edge of trails in the woodland, with 10 replicate sites per trail type and single quadrats at 10 control sites (i.e. total 60 sites, 160 quadrats). All trails resulted in vegetation loss on the trail surface and along the edges of the trails, as well as changes in vegetation composition, including reductions in shrubs and bulbs close to the trail. The most common types of trail were bare earth trails with an average width of 2.5 m (50% of trails) which resulted in the greatest soil loss (>88 000 m^3) and vegetation loss (33 899 m^2 or 3.4 ha) in the 167 ha woodland remnant overall. Wider (5.4 m) hardened tarmac trails, however, were associated with low species richness, high cover of exotic grasses and few herbs, shrubs and bulbs compared with vegetation away from trails and closer to other trails. Therefore a mixed approach to the provision of trails may be most appropriate, with hardened trails used in areas of highest use, but in some circumstances leaving trails unhardened may be more appropriate where they are likely to remain narrow and where there is less likely to be erosion and/or safety issues.

Additional keywords: compositional changes, edge effects, fragmentation, recreational infrastructure, remnant vegetation, trail impacts.

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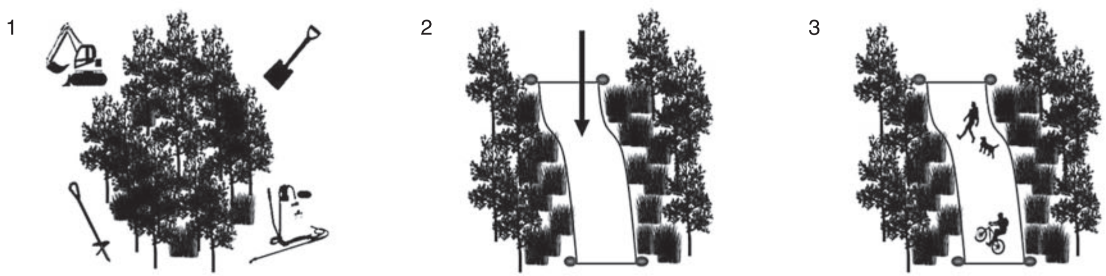
Introduction

Globally and in Australia, nature-based tourism and recreation is increasing including trail-based activities such as hiking and mountain-biking (Kuenzi and McNeely 2008; Balmford *et al*. 2009; Monz *et al*. 2010; Newsome *et al*. 2013; Rossi *et al*. 2015). As a result, the creation of recreational trails in natural areas is becoming more common, especially in countries such as the USA and Australia (Monz *et al*. 2010; Ballantyne *et al*. 2014; M Ballantyne, pers. comm.). While providing access to natural areas is important for human health and wellbeing (Swanwick *et al*. 2003; Lee and Maheswaran 2011) and fosters support for nature conservation (Spenceley *et al*. 2015), the infrastructure required for nature-based tourism and recreation has a range of environmental impacts including on plant communities of high conservation value (Matlack 1993; Stenhouse 2004; Godefroid and Koedam 2004; Hill and Pickering 2006; Olive and Marion 2009; Marion and Leung 2011; Ballantyne *et al*. 2014).

Recreational trails vary in their design, construction, maintenance and use. They can have hardened or unhardened surfaces, they can be narrow arteries or wide road-like developments, they can be formally created by land managers or informally created by users, and they can be single trails or a proliferation of dense networks (Marion and Leung 2001). Associated with the diversity of trails is variation in the severity and types of their impacts on vegetation (Godefroid and Koedam 2004; Hill and Pickering 2006; Monz *et al*. 2010; Ballantyne and Pickering 2015*a*). They can damage vegetation: (1) during their construction and maintenance, (2) as a result of changes in abiotic conditions caused by the presence of the trail, and/or (3) from the use of the trail for different types of recreational activities (Fig. 1). These impacts can occur at the



Stage	Construction/maintenance	Presence	Use
Activities	Clearance, grading, digging, irrigating, channeling, levelling, drilling, tempering, surfacing.		Hiking, dog-walking, mountain-biking, dirt-biking, 4 × 4 driving, horse -riding, racing and geocaching.
Impacts	Trampling, crushing, compaction, removal, noise, litter, pollution, weed spread, pathogens and surface runoff.	Changes in ultra violet and infra-red, photo-synthetically-active radiation, precipitation, temperature, moisture, wind, and organic matter.	Trampling, plant collecting, compaction, erosion, noise, litter, pollution, weed-spread and pathogens.

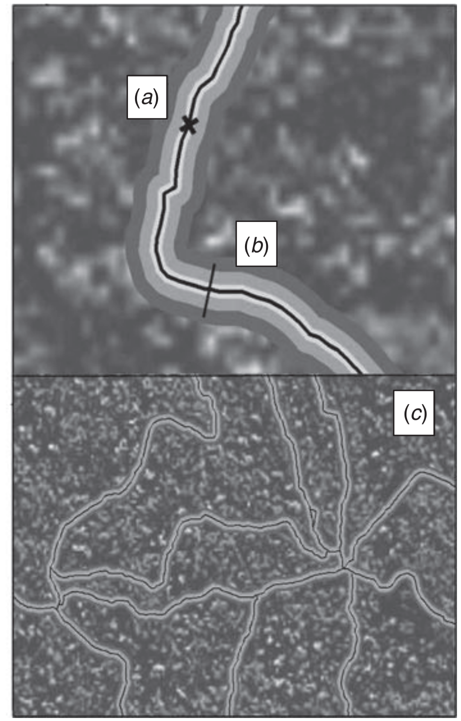


Fig. 1. Recreational trails impact natural vegetation in three main ways: through the construction and maintenance of the initial trail corridor: (1), through maintained presence of the exposed trail corridor leading to changes in local abiotic conditions (2) and finally, through the use of the corridor by recreational activities (3). These processes cause impacts at various scales from direct local effects on the trail (*a*), to indirect local effects at the trail edge (*b*) and to dispersed landscape scale effects that occur cumulatively across the trail network (*c*).

direct local scale (along the trail corridor itself) and at the indirect local scale (as edge effects away from the trail), as well as cumulatively across the landscape (Brooks and Lair 2005) (Fig. 1).

Impacts of trails can vary depending on the methods used in their construction and maintenance, but clearing and disturbing vegetation along the trail corridor and adjacent to the trail is common, especially where heavy machinery is used (Buckley 2004; Ledoux 2004). There are often changes in soil conditions including compaction, soil removal and introduction of new soil or other substrates to create the trail (Marion and Leung 2004; Müllerová *et al*. 2011). Impacts from the presence of the trail include changes in light regimes, precipitation, wind exposure and temperature as well as altered water flow and reduced organic matter along edge gradients away from the trail (Laurance *et al*. 1998; Potito and Beatty 2005; Pohlman *et al*. 2007). Finally, the use of the trail has additional impacts such as soil erosion and compaction often on the trail surface, along with damage to vegetation on the trail edge from trampling, changes in hydrology and soil movement as a result of soil erosion on the trail (Cole 2004; Marion and Leung 2004; Pescott and Stewart 2014) with more intensively-used trails often experiencing more damage than trails with limited use (Kutiel *et al*. 1999; Potito and Beatty 2005; Hamberg *et al*. 2008; Kim and Daigle 2012). The use of trails by plant collectors, arsonists, as well as the spread of weeds, pathogens and feral animals is also of concern (Benninger-Truax *et al*. 1992; Matlack 1993; Buckley *et al*. 2004; Baret and Strasberg 2005; Pickering and Mount 2010).

There is an increasing body of literature comparing the impacts of different trail-based activities such as hiking and mountain biking (Cole and Bayfield 1993; Kutiel *et al*. 1999; Talbot *et al*. 2003; Cole 2004; Hill and Pickering 2009; Monz *et al*. 2010; Pickering *et al*. 2010*a*). These studies have found that the timing, type and intensity of use and the tolerance of ecosystems affect the severity and types of impacts on vegetation (Cole 2004; Marion and Leung 2004; Pickering 2010). Fewer studies have assessed impacts of the trail infrastructure itself (Ballantyne and Pickering 2015*a*), and only a handful compare impacts among different types of trails (Godefroid and Koedam 2004; Müllerová *et al*. 2011) with just two from Australia (Hill and Pickering 2006; Ballantyne and Pickering 2015*b*). This is despite the hundreds of kilometres of formal trails (e.g. visible maintenance, signage and/or access infrastructure and/or mapping by land-owners for public use), and informal trails (those created and maintained by visitors for recreation that are outside of the formally-managed trail system) in Australia many of which traverse plant communities of high conservation value.

The results from the few studies that have compared among trails have found that the severity of impacts can vary among trails. Vegetation along the edge of formal hardened trails, for example, can be affected by disturbance to vegetation and soils during construction and maintenance resulting in the colonisation of trail edges by weeds and ruderal species (Hill and Pickering 2006), decreases in native plant cover, especially disturbance-sensitive species (Hill and Pickering 2006), changes in soil nutrients (Müllerová *et al*. 2011) and ultimately longer-term changes in plant composition and structure (Hill and Pickering 2006; Malmivaara-Lämsä *et al*. 2008; Ballantyne and Pickering 2015*b*). Informal, unhardened and narrow trails, in contrast, may have fewer and less severe impacts along the trail edge as there is often less disturbance to existing vegetation during their creation, but there can be continuing impacts on the trail itself (Marion and Leung 2004; Olive and Marion 2009; Marion and Leung 2011; Wimpey and Marion 2011). This includes soil erosion and trail widening when the trail surface is not stabilised by hardening which can affect vegetation on the trail edge (Olive and Marion 2009; Wimpey and Marion 2010).

The aim of the current research was to assess how recreational trails differ in the severity and types of impacts they have on plant communities of high conservation value. This involved comparing impacts of different recreational trails in a large remnant of grey box grassy-woodland (*Eucalyptus microcarpa* (Maiden) Maiden) in Belair National Park. The woodland is listed as an endangered plant community under Australian legalisation (Australian Government 2010) and as critically endangered within the World Wildlife Fund's terrestrial ecoregions database (World Wildlife Fund 2014). Specifically we: (1) mapped the extent and types of recreational trails in the woodland, (2) compared on-trail impacts between trail types (direct local scale), and (3) compared the impacts of different trail types on the composition of vegetation along the edges of the trails (indirect local scale). Based on the results of this study, we then provide recommendations for the management of different types of recreational trails in this and other plant communities that are of high conservation value, and popular sites for trail based recreation.

Materials and methods

Study ecosystem

Grey box grassy-woodland (*Eucalyptus microcarpa* (Maiden) Maiden) is restricted to a series of scattered remnants across south-eastern Australia from central New South Wales through north and central Victoria and into South Australia (Australian Government 2010, 2012). Before European settlement the ecosystem covered around 3.4 million ha, but today occupies just over 0.5 million ha; a loss of 85% (Australian Government 2010, 2012). The primary causes of this decline have been land clearing for agriculture, development and resource extraction with ongoing pressures from weed invasion, grazing and altered fire regimes (Yates and Hobbs 1997; Spooner *et al*. 2002; Australian Government 2010).

In South Australia, near the capital city of Adelaide (population 1.25 million people), the loss of grey box grassy-woodland has been particularly severe declining from 20 000 ha to just 2000 ha (90% loss) (Kraehenbuehl 1996; Paton 2008). The woodland typically occurs on undulating plains and slopes with productive soils derived from alluvial or colluvial deposits with rocky outcrops and clay pans (Australian Government 2010). Canopy cover is slightly denser than elsewhere in its range, likely as a response to higher rainfall in the region which averages 375–700 mm year^{-1} and Mediterranean temperatures (mean summer maximum 27°C; mean winter maximum 12°C) (Paton 2008; Australian Government 2012). The canopy is dominated by around 50% cover of the dominant tree

E. microcarpa often intermixed with stands of *Eucalyptus camaldulensis* Dehnh. and *Eucalyptus leucoxylon* F.Muell. (Australian Government 2010). The shrub layer provides around 30–40% cover and is dominated by species such as *Acacia pycnantha* Benth. and *Dodonaea viscosa* Jacq. The understory is highly diverse and dominated by grasses such as *Rytidosperma caespitosum* (Gaudich.) Connor & Edgar and *Poa labillardieri* var. *labillardieri* Steud. with numerous herb, bulb, chenopod and cryptophyte species. There is also high cover of invasive European grasses such as *Avena barbata* Pott ex Link, *Brachypodium distachyon* (L.) P.Beauv. and *Briza major* L. (Australian Government 2010). Total plant species richness often exceeds 200 even in small remnants (J Quarmby, pers. obs.) and includes numerous nationally-threatened species such as the pink-lipped spider-orchid (*Caladenia behrii* D.L.Jones) and plum leek-orchid (*Prasophyllum pruinosum* R.S.Rogers). The ecosystem is also a stronghold for some International Union for the Conservation of Nature (IUCN) Red-listed species such as the swift parrot (*Lathamus discolor*) and Flinders Ranges worm-lizard (*Aprasia pseudopulchella*).

Study site

The Adelaide grey box grassy-woodland is highly fragmented and consists of numerous small remnants, the largest of which is 190 ha, with most <100 ha. Most remnants are within public reserves and private land between Adelaide and the southern Flinders Ranges (Australian Government 2010). These remnants are increasingly used for a range of recreational trail-based activities including hiking and mountain biking, including the large 167 ha of grey box grassy-woodland within Belair National Park (835 ha) (Fig. 2). This peri-urban, IUCN Category II Park is 13 km south-east of Adelaide (Fig. 2). The park was proclaimed in 1891 and underwent a period of use as a semi-formal arboretum and government residence as well as being used for recreation with the creation of tennis courts, grass ovals and picnic areas (South Australian Government 2012). Today it is managed for conservation and recreation with around 250 000 visitors per year (Department of Environment Water and Natural Resources 2003; Mitcham City Council 2015).

Data collection

All recreational trails within the 167 ha of grey box grassy-woodland in Belair National Park were mapped including formal and informal trails. Trails were mapped using a Trimble Yuma GPS tablet (Sunnyvale, CA, USA) with 1–2 m real time accuracy and a recording interval of 1 m.

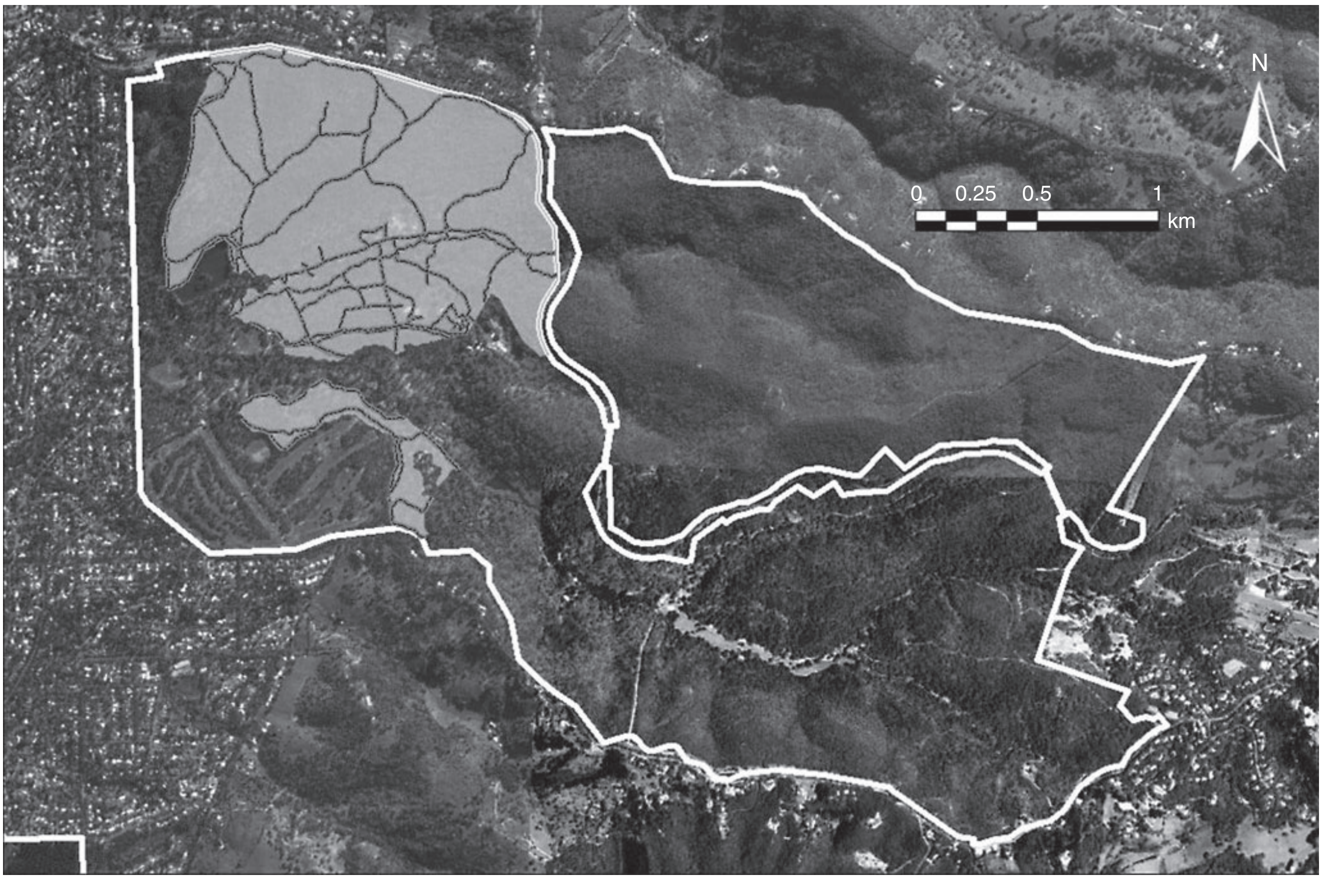


Fig. 2. Map of Belair National Park (thick white boundary line), south-eastern Adelaide, South Australia (35°01′22.70″S, 138°64′84.80″E) showing the location of grey box grassy-woodland remnants (highlighted patches to north-west of Belair National Park) and recreational trail networks within these remnants. All spatial data obtained from the Department of Environment, Water and Natural Resources, South Australia (DEWNR). Map created using ESRI ArcMap ver. 10.1.

The formal and informal trails were further categorised based on their average width and surface using a visual survey method similar to that used in trail condition class assessments (e.g. Farrell and Marion 2001; Ballantyne *et al*. 2014). Average width of the trail tread, defined as the most heavily trafficked area of the trail (Wimpey and Marion 2010), was measured at the start of each new section of trail and at a 100 m intervals along the trail. Average widths were calculated for each section and then all trails grouped into narrow (<1 m), intermediate (1–3 m) and wide (>3 m) trails. Surfaces were visually assessed, with bare soil, gravel and tarmac surfaced trails all found within the woodland. As a result five trail types were identified in total: (1) narrow bare soil, (2) intermediate bare soil, (3) wide bare soil, (4) intermediate gravel and (5) wide tarmac trails. While bare soil and gravel trails are predominantly used for hiking, dog-walking and running, wide tarmac trails are used for recreational vehicles, tourist drives and access to leisure facilities. For each of the five types of trails, 10 stratified random sites were located across the combined length of each type using ET GeoWizards 10.2 (Pretoria, South Africa) in ArcMap (Redlands, CA, USA) ver. 10.1 (see Ballantyne *et al*. 2014) for more detailed sampling of the trail and vegetation.

To assess impacts on the trail surface, soil loss (cross-sectional area), trail slope alignment angle and surface compaction were measured at each of the 10 sites per trail type. Cross-sectional area and trail-slope alignment angle were used to estimate soil loss using the methodology in Olive and Marion (2009). Soil compaction was measured using a penetrometer with a maximum capacity of 4.5 kg cm^{-2} at five equally-spaced points across the span of the trail tread at each site.

To assess the impacts of the five types of trails on vegetation adjacent to the trail, we surveyed all plants (excluding trees) in 1×3 m quadrats located immediately adjacent to the edge of the trail, at 5 and at 10 m away from the trail edge at each site (total of 150 quadrats, e.g. three distances, 10 replicate sites for each of five trail types). In addition, vegetation was assessed in 1×3 m quadrats at each of 10 randomly-selected control sites that were a minimum of 50 m from the edge of trails, roads, mown areas, burnt areas or the woodland edge. Field work was conducted from 20 to 31 October 2014.

In each of the 160 quadrats, total vegetation cover, cover of separate growth forms, species richness and canopy cover were recorded along with abiotic variables (see below). Vegetation cover was measured as the top-cover (vegetation, rock, bare ground, dead wood or moss) and the total overlapping cover of all plant species intercepted by a thin metal wire placed at 100 randomly-located points within the quadrat. The number of 'hits' per species was converted into percentage cover values and used to calculate both overlapping vegetation cover and overlapping cover of separate growth forms. Total species per quadrat was used to calculate total species richness and weed richness. Canopy cover was analysed as a percentage by photographing the quadrangle of airspace directly above each zone and then calculating the number of pixel cross-hair points occupied by canopy, sky or other (Monz *et al*. 2010; Ballantyne and Pickering 2015*b*).

At each site (50 trail sites + 10 controls) abiotic covariates that are likely to affect vegetation were also recorded. These included slope, altitude, aspect, soil type, distance to water and time since last fire. Slope was measured using the Hunter Research and Technology LLC Theodolite HD ver. 3.2 app for iPad (available online at http://hunter.pairsite.com/theodolite/, accessed 3 May 2016) by aligning the top of two poles of equal height 5 m either side of the survey point and measuring the difference in angle. Altitude was measured using a 1 s SRTM-derived 3-s smoothed digital elevation model provided by the Australian Government for the Adelaide region (http://www.ga.gov.au/data-pubs, accessed 3 May 2016). Aspect was measured using an electronic GPS-corrected compass bearing against true north. Distance to water was the Euclidean distance to nearest water body (stream, river, lake) using ArcMap ver. 10.1. Soil type was measured using the soil landscape map units of southern South Australia map (http://www.naturemaps.sa.gov.au/, accessed 3 May 2016). Time since last fire was taken from a fire management dataset recording major fires in protected areas (http://www.naturemaps.sa.gov.au/, accessed 3 May 2016).

Data analyses

The total area of each trail type, and therefore the total area of vegetation lost to the trail surface, was calculated by multiplying the total length (measured in ArcMap ver. 10.1) by the average width of each type of trail. We were then able to calculate total soil loss for each trail type by multiplying average soil loss per site (cm^2) by the total length (cm^3) of that type of trail.

To analyse differences among the trail variables: average soil loss (log10 transformed), trail slope alignment angle (log10 transformed) and surface compaction, a series of one-way ANOVAs were conducted in SPSS ver. 21 (SPSS Inc. Chicago, IL, USA) when the assumptions of normality and variance were satisfied. Least significant differences were used as *post-hoc* tests to determine which pairs of trails differed.

To analyse whether vegetation 10 m from the trails differed from the controls, data for these quadrats was compared with that from control quadrats using a series of one-way ANOVAs. As no significant differences were found, the control quadrats were removed from further analyses, with the 10 m quadrats treated as the 'natural' condition of the vegetation in subsequent analyses.

To assess differences in the impact of the five trail types on understory vegetation, and how far from the trail impacts could be detected, linear mixed models were used to analyse species richness, relative vegetation cover, relative weed cover (arcsine square-root transformed) and weed species richness (log-transformed) in SPSS ver. 21. Trail type, distance and distance × trail type were set as fixed factors, and sites (blocks) used as a random factor. Covariates (abiotic and biotic) were added to the list of fixed factors with aspect and altitude log10-transformed to satisfy assumptions of normality and homoscedasticity. Covariates were added and removed sequentially to determine the most significant models according to Akaike's information criterion (AIC). Finally, to compare any significant effects using the most powerful model, pairwise comparisons using estimated marginal means were computed for trail type and distance using syntax commands. If an interaction was significant, separate one-way ANOVAs were performed for each trail type and distance to understand the nature of the interaction. A P-value of ≤ 0.03 was used to account for the increased chance of falsely assuming a significant difference, i.e. an increased chance of type II errors.

To assess overall similarity of vegetation depending on how far the quadrat was from the trail and the type of trail, values for Sørensen's index were calculated using presence/absence data for all living vascular species per quadrat. A Bray–Curtis similarity matrix was used to compare composition between quadrats (site (distance × trail type)) for all trails with data being extracted and analysed using a one-way ANOVA. Tukey *post-hoc* tests were used to determine the nature of any significant differences.

To analyse differences in species composition and combined relative cover of growth forms between distances and trail types, a permutational multivariate ANOVA (PERMANOVA) was used with trail type, distance and distance × trail type as fixed factors, and site(trail type) as a random factor in PRIMER (Ivybridge, Devon, UK) ver. 6 using a Bray–Curtis dissimilarity matrix on square-root transformed cover data for individual species (divided by total cover) with the cover of litter, bare ground, rock, dead wood and moss removed. To determine if composition between 10 m and control sites was similar, we used a one-way ANOSIM. We found that composition was similar between these two groups of sites and so 10 m was used as the control. To show compositional differences among the five trail types and three distances, multi-dimensional scaling (MDS) plots were used to show the clustering of samples in two-dimensional space with goodness of fit shown as a stress value. This method is useful as it (1) visually portrays patterns by clustering and ordinating samples and (2) using the PERMANOVA function, statistically analyses them with a non-parametric permutation-based test (9999 permutations), similar to that of an ANOVA, but not requiring a normalised data distribution (Clarke 1993). Finally, a SIMPER analysis was used to determine which species contributed to any observed differences between groups.

Additionally, we used separated linear mixed models on total cover of each growth form (arcsine square-root transformed) per quadrat as an independent variable for graminoids, herbs and bulbs to determine the overall strength and direction of any observed effects. Shrubs were removed as a variable as they were absent from many quadrats.

To assess whether the trails had an effect on the cover of any dominant species (occurring in at least 50% of quadrats (75 out of 150)), cover data (arcsine square-root transformed) for that species was included in separate linear mixed models, using *post-hoc* pairwise comparisons to determine the nature of any significant differences. These species included: the native shrub *D. viscosa*, and the two exotic grasses *B. major* and *B. distachyon*.

Results

On-trail impacts

Within the 60 sample sites in the 167 ha of grey box grassy-woodland in Belair National Park, there was a total of 109 plant species including four species of trees, 11 shrubs, 20 bulbs, 49 herbs, 19 grasses, five sedges/rushes and one fern. There was also 20.1 km of trails in these woodlands, with 16.3 km formal and 3.8 km informal tails. There was ~2.2 km of narrow bare, 10.1 km of intermediate bare, 2 km of wide bare, 1.5 km of intermediate gravel and 4.4 km of wide tarmac trails (Table 1). The narrow bare earth trails were all created by users (informal), while the other four trail types were either entirely, or predominantly, trails provided and maintained by the park management.

The five types of trails differed in average width ($F = 52.075$, $P < 0.001$), apart from intermediate bare and intermediate gravel trails ($P = 0.320$). Because of their hardened surface, the wide tarmac trails had no soil loss from the surface. Narrow bare trails had less soil loss (1.4 m^3) than the other three types ($F = 12.991$, $P < 0.001$), which all lost around 60.1 m^3 of soil per metre of trail. Total soil loss was very high for intermediate bare trails, estimated at 68 218.9 m^3 in total, due to the greater length of this type of trail. We noted that total soil loss for intermediate gravel trails was high at 4095.2 m^3 (28.1 m^3 of soil per metre of trail); around 40% greater than narrow bare trails which were longer than the total length of intermediate gravel trails. Compaction ($F = 0.879$, $P = 0.484$) and trail slope

Table 1. Means and standard deviations for different trail variables for each of the five trail types assessed in grey box grassy-woodland in Belair National Park

Significant differences between trail types within variables according to one-way ANOVA are indicated: *, $P < 0.05$ with lettering showing the nature of the difference according to least significant difference *post-hoc* comparisons (e.g. 'a' is significantly different to 'b' but the same as 'a'). Abbreviation: TSAA, trail slope alignment angle

Variable	Narrow bare	Intermediate bare	Wide bare	Intermediate gravel	Wide tarmac	Totals
Length trails (m)	2157	10052	2011	1458	4399	20 079
Area trails (m^2)	1726	25132	7041	3062	23752	60 713
Average vegetation loss (m^2 m^{-1})	0.8	2.5	3.5	2.1	5.4	
Surfacing	Unsurfaced	Unsurfaced	Unsurfaced	Dolomite gravel	Bitumen	
% Formal	0	90	70	100	100	
% Informal	100	10	30	0	0	
Average width (m)	0.8 ± 0.3*a	2.5 ± 0.7*b	3.5 ± 1.2*c	2.1 ± 0.6*b	5.4 ± 0.6*d	
Average soil loss (m^3 m^{-1})	1.35 ± 15.5*a	67.9 ± 58.5*b	84.5 ± 83.9*b	28.1 ± 18.4*b	0*	
Total soil loss (m^3)	2908.4	68218.9	17000.9	4095.2	0	92 223.4
Av. compaction (kg cm^{-2})	4.2 ± 0.5	4.3 ± 0.5	4.4 ± 0.3	4.4 ± 0.2	>4.5	
TSAA (°)	44.5 ± 23.1	37.5 ± 27.8	55.5 ± 29.5	34.5 ± 36.1	39.5 ± 26.8	
Predominant use	Hiking	Hiking, mountain biking	Hiking, mountain biking	Hiking, mountain biking	Driving, hiking	

alignment angle ($F = 0.807$, $P = 0.527$), did not differ among the five types of trails (Table 1).

Because the five types of trails varied in length and width, they differed in the amount of vegetation lost. Intermediate bare trails were associated with the greatest loss (25 132.3 m^2 or 2.5 m^2 lost per metre of trail) of vegetation as they were the most extensive trail type, while wide tarmac trails, which were much shorter accounted for similar levels of vegetation loss (23 751.9 m^2 or 5.4 m^2 lost per metre of trail) because they were wider. Narrow bare trails caused the least loss of vegetation (1726.1 m^2 or 0.8 m^2 lost per metre of trail), in part because they were narrower. Total vegetation loss due to all five trails was 60 712.8 m^2, which is around 2.1% (3.4 ha) of the total remnant of grey box woodland (167 ha).

Was there an effect of the trails on vegetation?

There were no significant differences in any of the dependent variables between the control quadrats (50 m from trails) and those 10 m from the trails except for cover of herbs, indicating that most effects of the trails were limited to vegetation <10 m from the trails. Vegetation in the control and 10 m quadrats was characterised by high cover of the native sub-shrubs *Hibbertia exutiacies* and *Hibbertia australis* (23% cover) and graminoids composed largely of the exotic grasses *B. major* (18%) and *B. distachyon* (11%) along with the native sedge *Lepidosperma curtisiae* (18%) and tussock grass *R. caespitosa* (10%). The native bulbs *Arthropodium strictum* (5%) and *Lomandra multiflora* (5%) and the exotic herb *Plantago lanceolata* (11%) also had comparatively high cover.

Vegetation was affected by the trails, with significant differences in several variables between quadrats on the edge of trails and those 5 and 10 m away from the trails. For all five trail types, there was less vegetation in quadrats on the edge of trails (43% ± 28; $F = 15.534$, $P = <0.001$) compared with quadrats 5 m (62% ± 25, $P = 0.001$) and 10 m (63% ± 23, $P = <0.001$) away, but no differences between the 5 and 10 m quadrats ($P = 0.864$) (Tables 2, 3). Bulb cover was also affected by trails ($F = 11.843$, $P < 0.001$), with reduced cover near the trail (Tables 2, 3). There were no overall effects of trails on the cover of dominant species including the native shrub *D. viscosa*, and the two exotic grasses *B. major* and *B. distachyon*.

Overall composition (based on the ANOSIM) differed with distance from trails, but was also affected by an interaction with trail type (see below). The cover of different growth forms was affected by the trails (Global Rho = 0.26, $P < 0.001$, Stress = 0.18) with an increase in the cover of shrubs and bulbs in quadrats 5 m and 10 m (18.4% and 20.9%) from the trail compared with those on the trail edge (11.8%). There was a slight decrease in the cover of graminoids for the 5 and 10 m quadrats (49.7 and 47.6%) compared with the trail edge (55.6%). These patterns were in part due to decreasing cover of the exotic grasses *B. major* and *B. distachyon* further from the trail and a rise in the cover of native grasses such as *R. caespitosa* and *Microlaena stipoides* var. *stipoides*. Vegetation on the edge of the trails was typically dominated by grasses (average cover 71%) with lower cover of sedges (7%), shrubs (8%) and bulbs (<1%). Vegetation in quadrats 5 m from the trails was also dominated by grasses (60%) but with higher cover of shrubs (13%), sedges (9%) and mostly exotic bulbs (5%). Vegetation in quadrats 10 and 50 m from the trails was dominated by grasses (53%) but >20% of them were native, and there was a higher cover of shrubs (15%), sedges (10%) and native bulbs (7%).

Although there were 39 species of weeds (one shrub, eight bulbs, 11 grasses, 19 herbs) recorded across the 160 quadrats

Table 2. Results from linear mixed models (LMM) and PERMANOVA analyses of vegetation in quadrats at different distances from five types of trails

P-values for significant differences between trail types are provided. Abbreviation: Neg., negative

Dependent variable	Trail type	Distance from trail	Interaction	Significant covariates
Species richness (LMM)				
All	$F = 1.264$ $P = 0.297$	$F = 10.746$ $P = <0.001$	$F = 2.590$ $P = 0.013$	
Weeds	$F = 1.187$ $P = 0.329$	$F = 0.881$ $P = 0.418$	$F = 0.655$ $P = 0.729$	Altitude neg. interaction $F = 5.954$, $P = 0.019$
Total % overlapping cover (LMM)				
Vegetation	$F = 1.140$ $P = 0.350$	$F = 15.675$ $P = <0.001$	$F = 1.195$ $P = 0.311$	
Weeds	$F = 1.204$ $P = 0.323$	$F = 0.561$ $P = 0.573$	$F = 1.030$ $P = 0.422$	Altitude neg. interaction $F = 5.469$, $P = 0.023$
Graminoid	$F = 0.611$ $P = 0.656$	$F = 1.475$ $P = 0.234$	$F = 0.454$ $P = 0.885$	
Herb	$F = 2.558$ $P = 0.049$	$F = 2.902$ $P = 0.060$	$F = 0.715$ $P = 0.678$	
Bulb	$F = 0.529$ $P = 0.715$	$F = 11.843$ $P = <0.001$	$F = 1.341$ $P = 0.233$	
Overlapping cover of dominant species				
Dodonaea viscosa	$F = 0.062$ $P = 0.993$	$F = 3.007$ $P = 0.054$	$F = 0.614$ $P = 0.764$	
Briza major	$F = 2.654$ $P = 0.045$	$F = 2.062$ $P = 0.133$	$F = 2.071$ $P = 0.050$	
Brachypodium distachyon	$F = 0.863$ $P = 0.493$	$F = 0.311$ $P = 0.733$	$F = 0.753$ $P = 0.644$	
Composition (PERMANOVA)				
All species	Pseudo-$F = 1.111$ $P = 0.230$	Pseudo-$F = 2.363$ $P = 0.002$	Pseudo-$F = 1.274$ $P = 0.040$	
Growth forms	Pseudo-$F = 1.054$ $P = 0.398$	Pseudo-$F = 5.824$ $P = <0.001$	Pseudo-$F = 1.112$ $P = 0.314$	
Sørensen similarity index	n/a, n/a	n/a, n/a	$F = 4.564$ $P = 0.004$	

Table 3. Characteristics of vegetation in quadrats for controls vs 10 m, and for quadrats on the edge of the five trail types
Significant differences indicated:*, $P<0.05$ according to linear mixed models with lettering showing the nature of the difference according to least significant difference *post-hoc* comparisons (e.g. 'a' is significantly different to 'b' but the same as 'a'). Abbreviation: Inter., interaction term

	Controls + 10 m	Narrow bare edge	Intermediate bare edge	Wide bare edge	Intermediate gravel edge	Wide tarmac edge	Inter. or main effect?
Species richness	9.6 ± 2.7*a	7.6 ± 2.9*b	8.0 ± 3.3*b	6.3 ± 2.4*c	6.8 ± 3.1*b	4.1 ± 2.3*c	Inter.
				Total % overlapping cover			
Vegetation	80.9 ± 21.3*a	39.3 ± 23.3*b	42.4 ± 27.2*b	34.7 ± 14.9*b	45.4 ± 30.2*b	25.0 ± 14.2*b	Main
Graminoids	36.3 ± 20.9	26.7 ± 20.9	30.5 ± 30.1	17.9 ± 10.4	35.1 ± 33.0	20.1 ± 11.4	None
Herbs	33.2 ± 16.9*a	18.6 ± 17.4*a	9.0 ± 10.1*a	12.5 ± 8.7*a	12.2 ± 12.2*a	3.6 ± 5.0*b	Main
Bulbs	7.1 ± 6.28*a	4.8 ± 5.6*b	6.1 ± 6.4*b	1.9 ± 1.4*b	2.3 ± 2.8*b	1.1 ± 2.2*b	Main
Shrubs	2.7 ± 2.2*a	3.4 ± 3.6*a	3.9 ± 5.3*a	2.5 ± 4.1*a	4.5 ± 12.1*a	0.2 ± 0.48*b	Inter.
				Overlapping cover of dominant species			
Dodonaea viscosa	0.6 ± 1.3	2.8 ± 3.6	3.3 ± 5.3	2.0 ± 4.1	1.7 ± 4.1	0.2 ± 0.4	None
Brachypodium distachyon	5.5 ± 8.3	4.1 ± 5.6	8.4 ± 19.7	3.9 ± 7.3	6.1 ± 9.7	11.8 ± 9.8	None
Briza major	Controls	Narrow bare edge	Intermediate bare edge	Wide bare edge	Intermediate gravel edge	Wide tarmac edge	Inter.
	16.9 ± 17.5*a	10.6 ± 16.6*a	9.4 ± 15.1*a	4.1 ± 5.3*b	5.4 ± 7.7*b	0.8 ± 1.5*c	
		Narrow bare 5 m	Intermediate bare 5 m	Wide bare 5 m	Intermediate gravel 5 m	Wide tarmac 5 m	
		11.5 ± 18.1*ab	14.4 ± 15.8*a	2.7 ± 3.8*c	5.4 ± 7.7*b	2.5 ± 4.9*c	
		Narrow bare 10 m	Intermediate bare 10 m	Wide bare 10 m	Intermediate gravel 10 m	Wide tarmac 10 m	
		15.7 ± 19.1*a	7.9 ± 10.0*ab	0.4 ± 0.9*b	15.9 ± 18.28ac	5.5 ± 9.4*abc	
Sørensen similarity index (%)	n/a	Narrow bare	Intermediate bare	Wide bare	Intermediate gravel	Wide tarmac	Inter.
Edge vs 5 m composition		58.2 ± 16.3	44.9 ± 11.5	43.7 ± 22.5	50.2 ± 25.9	29.4 ± 25.7	
Edge vs 10 m composition		49.9 ± 25.4*a	37.9 ± 12.7*ab	39.4 ± 17.8*ab	49.9 ± 11.6*a	20.7 ± 17.6*b	
5 m vs 10 m composition		55.8 ± 13	54.3 ± 15.1	47.1 ± 13.8	57.7 ± 11.2	52.9 ± 12.5	

surveyed, with average weed cover of 29% per quadrat, there was no effect of the trails on weed cover *per se* (Tables 2, 3).

Was there a difference in impacts among trail types?

Some trail impacts were not simply a result of the presence of any type of trail, but varied among the trails. This included impacts on species richness and composition (Tables 2, 3; Fig. 3). Species richness was affected by an interaction between trail type and distance (linear mixed model, $F=2.207$, $P=0.034$), as was composition (PERMANOVA, Pseudo-$F=1.27$, $P=0.04$, stress = 0.23).

For the wider trails, species richness and composition were affected by distance from the trail, but there was no effect of distance for the narrow and intermediate bare earth trails (Table 2; Fig. 3). For the wide tarmac trail, for example, quadrats on the edge of the trail had less than half the number of species as those further away (edge = 4.1 ± 2.3; 5 m = 6.8 ± 1.5; 10 m = 9.1 ± 2.6; $P<0.021$). The vegetation in quadrats on the edge of the wide tarmac trail also differed in composition to those further away with more exotic grasses such as *B. distachyon*, *B. major* and *A. barbata* close to the trail, but fewer native grasses such as *M. stipoides* var. *stipoides*, *Anthosachne scabra* and *P. labillardieri*, which were mainly found in quadrats 5 m and 10 m away from the trail.

For the wide bare trail there were also fewer species in quadrats on the edge of the trail (6.3 ± 2.4), than in quadrats 5 and 10 m away (8.9 ± 3.1; 8.1 ± 2.1, respectively; $P<0.001$). Composition, however, was more similar (Fig. 3) with a greater abundance of native sedges and graminoids (*L. curtisiae* and *R. caespitosa*) and moderate abundance of exotic grasses (*B. distachyon* and *B. major*) across all distances, but with more herb species further from the trails. For intermediate gravel trails, composition steadily changed from quadrats on the trail edge to those 10 m away with a steady decrease in exotic grasses (*B. distachyon* and *B. major*) and an increase in sub-shrubs (e.g. *H. australis*) and bulbs (e.g. *A. strictum*).

Average similarity in composition based on Sørensen's indices between pairs of quadrats was affected by an interaction between trail type and distance with less similarity between quadrats on the edge of trails and those 10 m away (39.6% ± 20.1 similarity; $F=4.564$, $P=0.004$). The greatest differences in composition were along wide tarmac trails where quadrats on the edge of the trail were very different to those 10 m away (20.7% ± 17.6 similarity). Along intermediate and wide bare trails, similarity was greater between edge and 10 m distances (38.6% ± 15.2) and along narrow bare and intermediate gravel trails, similarity was even higher (49.5% ± 18.5). Along all trails, composition 5 m from the trail was similar to edge quadrats (average 45.3% ± 22.4 similarity; $F=2.490$, $P=0.056$) and even more similar to 10 m quadrats (average 53.6% ± 13.1 similarity; $F=0.925$, $P=0.458$), and therefore acted as a transition zone.

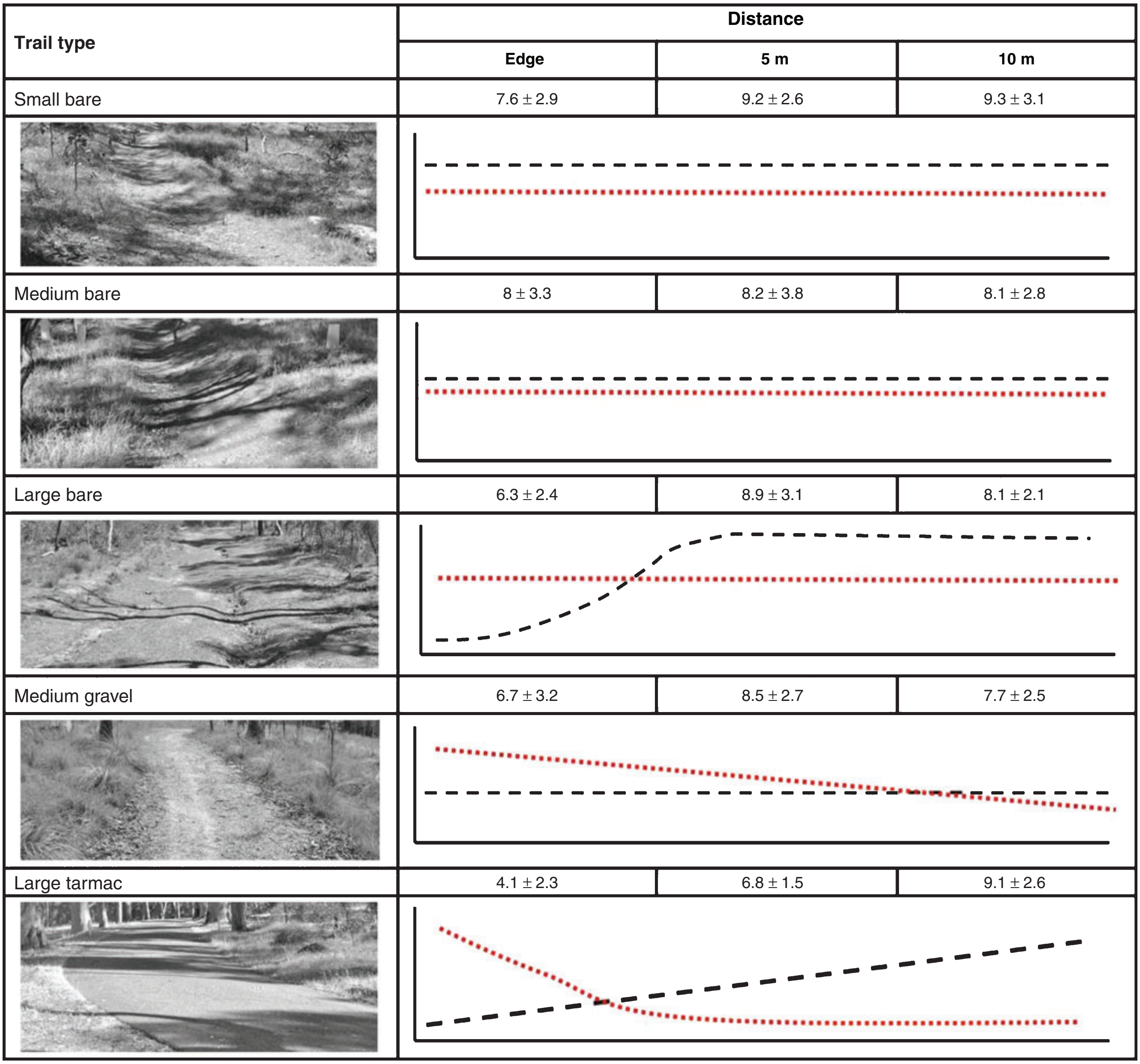


Fig. 3. Effect of five different types of trails on species richness and vegetation composition in grey box grassy-woodland. Lines signify changes in species richness (large dashed line) and species composition (small dotted line) adjacent the edge, 5 and 10 m from the edge of the trail and among the five trail types. Species richness values for each distance are given above each graph while the line for species composition symbolises the nature of the change without respect to denominative changes in variables.

Herb cover was lower along wide tarmac trails, but this effect occurred irrespective of distance from the trail ($F = 2.558$, $P = 0.049$). Also, the 10 m quadrats for the wide tarmac trails differed from the control quadrats indicating either site-specific co-variation or that the impact of this trail type on herbs extends beyond 10 m from the edge of the trail.

The cover of one dominant, *B. major*, was affected by an interaction between trail type and distance, with a slight reduction in cover for quadrats along the edge of wide bare, intermediate gravel and wide tarmac trails ($F = 2.071$, $P = 0.05$) compared with other trails and controls (16.9% ± 17.5). Along the edge of the wide bare trails, this species had low cover (4.1% ± 5.3), but it was even lower in quadrats further from the trail (5 m = 2.7% ± 3.8; 10 m = 0.4% ± 0.9). In quadrats along intermediate gravel and wide tarmac trails, cover of *B. major* increased with increasing distance from the trail (edge = 3.1% ± 4.6; 5 m = 3.9% ± 6.3; 10 m = 10.7% ± 13.8; Table 3).

Abiotic covariates had limited effect on vegetation with altitude having a negative linear interaction with weed species richness ($F = 5.954$, $P = 0.019$) and with weed cover ($F = 5.469$, $P = 0.023$). Slope, aspect, distance to water, soil type, time since last fire, and canopy cover had no significant effect on vegetation.

Discussion

The five types of recreational trails differed in their impacts on vegetation in the endangered grey box grassy-woodland in Belair National Park. Remnants such as this one are important for recreational opportunities, but are also of high conservation value. A common problem for those managing this and other parks, is which type of trails are best to minimise environmental impacts while also providing appropriate recreational opportunities. Part of this challenge has been the limited research comparing environmental impacts among different types of trails.

There is an extensive network of recreational trails in grey box woodland

There were over 20 km of recreational trails in the grey box grassy-woodland of which nearly 90% were formal, management-created trails. The most common of these were intermediate bare earth trails (60%), which collectively resulted in the loss of >25 000 m^2 of vegetation. This was partly due to the total length of this type of trail in the woodland, but also due to the tendency of unsurfaced trails to widen over time (Olive and Marion 2009; Monz *et al*. 2010; Wimpey and Marion 2010). Although less common (25%) in the woodland, other trails such as the wide tarmac trails also resulted in the loss of >23 000 m^2 of vegetation as they were wider (often >7 m) to accommodate recreational vehicles. The width of these trails exposes vegetation on the trail edge to a range of changes in abiotic conditions such as increased wind-blow, which may further reduce tree numbers and canopy cover.

We observed that in Belair National Park, only 11% of trails were informal, user-created trails. This may be due in part to the closure of several informal trails in the park in the past, as part of upgrades to the trail network (Taylor Cullity Lethlean 2009). In other remnants of the same woodland in the region, there were more narrow informal trails (Ballantyne and Treby 2015). Narrow informal trails can be important components of trail systems included in protected areas (Leung *et al*. 2011; Ballantyne *et al*. 2014; Ballantyne and Treby 2015). They can result in extensive loss of habitat (up to 20%), habitat fragmentation and complex internal networks of disturbance disrupting the once-contiguous ecosystems (Ballantyne *et al*. 2014). The resulting impacts from uncontrolled trail networks are complex and hard to manage (Leung *et al*. 2011, 2012; Pickering *et al*. 2012) highlighting the importance of limiting the spread of informal trail networks particularly in areas of high conservation value (Leung *et al*. 2012; Wimpey and Marion 2011).

All five trails affected vegetation adjacent to the trails

All five trails had some impacts on vegetation in the grey box grassy-woodland. This included reduced vegetation cover along the edge of the trails and up to 5 m away, but also the reduced cover of bulbs and shrubs and increased cover of graminoids. These changes are likely to be the result of (1) trampling damage along the edge of the trail mostly by hikers and mountain-bikers, (2) trail maintenance activities (e.g. herbicide spraying, slashing and digging), and (3) changes in abiotic conditions caused by the presence of the trail itself. Abiotic changes could include drier soils compared with vegetated areas, greater convection and conduction of heat, and reductions in local atmospheric moisture (Delgado *et al.* 2007). Additionally, the open nature of the trail corridors could lead to increased wind-blow damaging vegetation along the edge of the trail, especially trees and shrubs (Laurance *et al*. 1998; Harper and MacDonald 2002; Delgado *et al*. 2007; Ballantyne and Pickering 2015*b*).

For the grey box woodland it appears there are already many species with traits able to deal with conditions common along the edges of trails, particularly exotic grasses that are native to the European-Mediterranean region. These species often accumulate dead matter with age, have vertical, recurved leaves and late-spring abscisic acid-accumulation which increases their capacity to deal with exposed hot conditions (Körner 2003; Volaire *et al*. 2009). Species native to the grey box grassy-woodland, such as sclerophyllous shrubs and sub-shrubs (e.g. *Hibbertia* spp.) may also be well adapted to hotter, drier and windier conditions, but like many other sclerophyllous species, are less tolerant of trampling damage (Rickard *et al*. 1994; Bernhardt-Römermann *et al*. 2011; Pescott and Stewart 2014).

Trails differ in their impacts with more on-trail issues from unhardened informal trails

When impacts were compared among trails, we found strong contrasts. Conditions on the trail were generally poorest for bare trails while hardened trails were in good condition, as would be expected. Bare trails varied in width ranging from 0.8 to 4 m, and for the wider trails, soil loss was high. Cumulatively, bare earth trails accounted for 95.6% of total soil loss from trails in the woodland (>88 128.2 m^3 or 6.2 m^3 lost per metre of trail). This again illustrates how unhardened trails can be prone to on-trail degradation (e.g. soil erosion) (Olive and Marion 2009; Wimpey and Marion 2010; Pickering *et al*. 2010*b*) including over time (Marion and Leung 2004).

In contrast, bare trails were narrow (e.g. informal narrow bare earth trails <1 m) with limited soil loss and on-trail conditions were as good as, or even better than, some hardened trails. Indeed, the average soil loss from intermediate gravel trails was 52% higher than from narrow bare earth trails despite there being more narrow bare trails in the woodland. Maintaining such narrow trails through use of education, low gradients, grade reversals and drainage features, (e.g. water bars, drainage dips), strategically placed pass-points could be an efficient and cost-effective way of maintaining on-trail conditions. These methods, however, have had limited success in the past (Olive and Marion 2009) and the apparently 'good' condition of the narrow bare earth trails in the grey box woodland may be because these trails are relatively new (Taylor Cullity Lethlean 2009) or they receive very low levels of traffic.

For vegetation adjacent to trails, there were more impacts from wider hardened trails

When comparing impacts on vegetation adjacent to the trails, we found that wider and hardened trails had more severe impacts that extended further from the trails than narrower bare trails, although narrower bare trails did have poor surface conditions and these proliferate out as trails widen over time. While trail-hardening can be a popular management option to improve on-trail conditions (Pickering *et al*. 2010*b*; Wimpey and Marion 2010; Leung *et al*. 2011), and can maintain sustainable trail networks (Hill and Pickering 2006; Cahill *et al*. 2008; Zhang *et al*. 2012), they often dramatically alter vegetation composition along the trail edge (Godefroid and Koedam 2004; Hill and Pickering 2006; Müllerová *et al*. 2011; Ballantyne and Pickering 2015*b*) and, in some cases, negatively affect visitor experiences (Cahill *et al*. 2008). One factor we were unable to measure accurately was the age of the trails, which is likely to have had some effect on the composition of trail-side vegetation. It is likely that some of the earlier hardened trails will have exerted greater abiotic changes and have been maintained rigorously for a long time, thus leading to more marked edge effects (Godefroid and Koedam 2004).

Vegetation along the edge of wide tarmac trails contained far fewer species, was dominated by exotic grasses (*B. distachyon* and *A. barbata*) and had less herb cover than control quadrats in the woodland. The construction and maintenance of these trails often damages vegetation along the edge of the trail, including during the use of equipment, in the introduction of materials, grading, levelling and laying of heated tarmac and the use of multiple chemical additives and finishers (Buckley 2004; Delgado *et al*. 2007). This can damage shrubs and trees in, and close to, the work zone which often exceeds 20 m either side of the actual trail tread (M Ballantyne, pers. obs.). Once laid, wide tarmac trails often have low albedo and so heat up rapidly, creating localised wind vortices resulting in drier conditions near the trails (Delgado *et al*. 2007; Pohlman *et al*. 2007). These effects may account for why much of the vegetation close to the tarmac trails in the grey box woodland consisted of disturbance-tolerant species close (<5 m) to the trail.

For intermediate gravel trails, there was a more gradual decline in the cover of exotic grasses and increases in native bulbs and shrubs away from the trail. This wider, but less severe, edge effect may be due to the substrates on the trails. The base-rich dolomite gravel used for these types of gravel trails can spread into adjacent vegetation and may leach alkalis into an otherwise base-poor environment through surface-flow (J Quarmby, pers. obs.). In other regions with base-poor soils, the use of foreign alkali gravel has had a dramatic effect on species richness and composition that extends beyond 10 m from the edge of the trail, particularly down-slope of the trail due to changes in soil nutrients (Godefroid and Koedam 2004; Müllerová *et al.* 2011). Reduced disturbance during installation, but gradual leaching from gravel trails may explain why edge effect from these trails on composition was wider, but less severe in the grey box woodland.

Wide bare trails (average width 3.5 m) also resulted in fewer species close to the trail in the grey box woodland. Wider trails that require the removal of canopy vegetation expose the ground surface along and adjacent to the trail to greater insolation, convection and conduction resulting in drier conditions (Pohlman *et al*. 2007). The composition of vegetation along the edge of the wide bare trails in the grey box woodland; however, was generally similar to controls with a more even composition of exotic *v.* native graminoids and more herbs. The cover of *B. major* was particularly low on the edge of wide bare trails and continued to decrease with distance in contrast to tarmac and gravel trails were it increased positively with distance from those trails. There may have been less disturbance in the creation of the wide bare trails (largely just clearing) compared with the wider hardened trails, allowing the retention of complex native edge structure, nutrient levels and soil condition, reducing the dominance of weeds close to the trail edge (Ballantyne and Pickering 2015*b*).

Management recommendations

This research demonstrates that recreational trails differ in their severity and types of impacts caused to vegetation, and so future management of natural areas must account for this variation when developing sustainable trail networks. The results support the use of hardened trails along the most intensely-used areas, on steep slopes or in areas inappropriate for unhardened trails (e.g. riparian areas) where there is a risk of deteriorating on-trail conditions and trail widening. Hardening may also help restrict the creation of informal trails networks (Marion and Leung 2011; Wimpey and Marion 2011; Leung *et al*. 2012). Hardened trails, however, do have local impacts on edge vegetation and so their use should be limited, where possible, in areas of high conservation value, such as in grey box grassy-woodlands. The use of wide tarmac trails for vehicle access and base-rich substrates such as dolomite gravel should be avoided because they appear to have had the greatest effects on plant composition appearing to favour novel disturbance-tolerant species $\geq$10 m from the trail edges. Minimising damage during construction, using locally-sourced surfacing substrates and actively rehabilitating edge vegetation may help mitigate some of these impacts (Ballantyne and Pickering 2015*b*).

The results indicate that where unhardened trails are strategically-located and maintained, they can sustain good on-trail conditions and have limited effect on edge vegetation. This is especially important in areas of high conservation value containing disturbance-sensitive species. As long as unhardened trails are not developed in topographically-sensitive areas such as over soils prone to water-logging, deep or Aeolian soils, steep slopes and riparian zones, trail gradients are kept to a minimum and trails are placed parallel to contour lines with adequate drainage systems, they can remain in good condition over time (Marion and Leung 2004). Maintaining the trail in good condition can be difficult with high use, if it results in vegetation loss, soil erosion and trail widening. These type of issues with unhardened trails can be minimised by using natural anchor points (rocks and logs), strategically-placed pass points and by actively managing for different types and intensities of use. Finally, monitoring how trails affect vegetation and soils, as well as fauna and aquatic systems, and how their use changes over time is important for long-term trail sustainability.

Conclusion

We compared the types and severity of impacts by five types of recreational trails on vegetation in an endangered woodland. These types of ecosystems are undergoing increasing loss and degradation globally due to urbanisation and clearing with the development of recreational trails adding to these threats. We found trails varied in their impacts. Wider unhardened bare earth trails caused the most soil and vegetation loss, whereas hardened trails were less common and so resulted in less overall damage. However, hardened trails had the greatest effect on vegetation along the edge of the trail including decreased species richness, altered composition and decreases in sensitive plant species. It is likely that these changes in vegetation are a result of disturbance during the construction and maintenance of the trails. Based on the results of the current study, and the few others comparing different types of trails, we suggest a mixed management approach may be best, that involves hardening trails in popular areas, but with minimal disturbance during construction, the use of local materials where possible and vegetation rehabilitation once built. In less popular areas and/or those of particularly high biodiversity and conservation value such as grey box grassy-woodland, leaving trails unhardened but investing effort into more strategic trail placement and sustainable trail design may help maintain on-trail conditions and limit the spread of informal trails. These broad recommendations will help assist natural area managers to more effectively account for both recreation and conservation demands, which is of particular importance in already threatened ecosystems such as grey box grassy-woodland.

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