Journal of Environmental Management 207 (2018) 292–302

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Comparison of trailside degradation across a gradient of trail use in the Sonoran Desert

Helen Ivy Rowe [a, *], Melanie Tluczek [a], Jennifer Broatch [b], Dan Gruber [a], Steve Jones [a], Debbie Langenfeld [a], Peggy McNamara [a], Leona Weinstein [a]

[a] McDowell Sonoran Conservancy, Scottsdale, AZ, USA
[b] School of Mathematical and Natural Sciences, Arizona State University at the West Campus, Glendale, AZ, USA

ARTICLE INFO

Article history:
Received 16 June 2017
Received in revised form
13 October 2017
Accepted 10 November 2017

Keywords:
Trail impacts
Native plant species
Nonnatives
Soil crust
Recreation
Sonoran Desert

ABSTRACT

As recreational visitation to the Sonoran Desert increases, the concern of scientists, managers and advocates who manage its natural resources deepens. Although many studies have been conducted on trampling of undisturbed vegetation and the effects of trails on adjacent plant and soil communities, little such research has been conducted in the arid southwest. We sampled nine 450-m trail segments with different visitation levels in Scottsdale's McDowell Sonoran Preserve over three years to understand the effects of visitation on soil erosion, trailside soil crusts and plant communities. Soil crust was reduced by 27–34% near medium and high use trails (an estimated peak rate of 13–70 visitors per hour) compared with control plots, but there was less than 1% reduction near low use trails (peak rate of two to four visitors per hour). We did not detect soil erosion in the center 80% of the trampled area of any of the trails. The number of perennial plant species dropped by less than one plant species on average, but perennial plant cover decreased by 7.5% in trailside plots compared with control plots 6 m off-trail. At the current levels of visitation, the primary management focus should be keeping people on the originally constructed trail tread surface to reduce impact to adjacent soil crusts.

1. Introduction

An estimated 78.3 million adults visited trails in the United States in 2008 and that number is predicted to increase 30% by 2030 (White et al., 2014). One early comprehensive review estimated that trails and campsites together disturb only 1% of the total area of wilderness (Cole, 1987), but this was based on estimates of area rather than direct experiments on the effects of disturbance on ecological function (Adkinson and Jackson, 1996). The concern over visitor disturbance in natural areas has motivated extensive research on trail impacts, as seen by multiple review papers over the years (Ballantyne and Pickering, 2015; Cole, 1987; Hammitt and Cole, 1998; Kuss et al., 1990; Leung and Marion, 2000; Monz et al., 2013), and more recently, in research studying the effects of trails on surrounding ecological communities (as reviewed by Monz et al., 2013). Trail impact studies generally focus either on: a) examining the changes along established trails and around campsites, or b) testing resistance and resilience of undisturbed areas through controlled trampling in undisturbed areas (Monz et al., 2013).

On established trails, disturbance occurs initially at the time of trail construction as a result of opening canopies by vegetation removal, compaction of soil and alteration of drainage patterns by removal of upper soil horizons, and modification of micro topography, affecting microclimate (Cole, 1987). Subsequently, ongoing trail visitation has direct (trampling) and indirect effects (compacted soils, reduced organic matter and reduced soil nutrient changes) on trailside vegetation (Monz et al., 2010). As demand for recreational trails and trail use continues to grow, understanding anthropogenic environmental impacts will become increasingly important to inform sustainable resource management.

The Sonoran Desert is a prime destination for outdoor recreation. For example, Scottsdale's 13,000 ha McDowell Sonoran Preserve (MSP), which is the largest urban preserve in the United States, received 750,000 visits in 2016, yet it is only one of many natural areas surrounding the Phoenix metropolitan area. Since environmental factors such as climate and geology, and the intermediate elements of topography, soil, and vegetation type

* Corresponding author.
E-mail address: helen@mcdowellsonoran.org (H.I. Rowe).

https://doi.org/10.1016/j.jenvman.2017.11.028
0301-4797/

significantly affect the degree and type of trail degradation (Leung and Marion, 1996), each ecological system responds differently to trail visitation and associated impacts. In a recent review of 59 original research papers on trail impacts, over 50% of the papers focused on just three habitat types: temperate forests, alpine and montane grasslands and shrublands, and Mediterranean forests, woodlands and sclerophyll scrub, while only 2 papers (3%) focused on deserts and xeric shrublands (Ballantyne and Pickering, 2015), and these latter papers were either focused on the impacts of roads (Brooks and Lair, 2005), or on the effects of vehicles or pedestrian use on untrampled dune systems (Rickard et al., 1994).

Soil crust is a key biological indicator of disturbance in southwest arid environments (Allen, 2009; Belnap, 1998). In the southwest and similar environments, biological soil crust plays an important role in increasing soil stability, water infiltration, and soil fertility in otherwise erodible, dry, and infertile soils (Belnap, 1994; Belnap and Gardner, 1993; Harper and Marble, 1988; Johansen, 1993; Metting, 1991; Williams et al., 1995). Consequently, soil crust loss can result in soil erosion and loss of soil nutrients (Belnap and Gillette, 1997; Harper and Marble, 1988; Schimel et al., 1985). Soil crusts are likely susceptible to trail impacts because they are brittle when dry and crush easily with trampling (Belnap and Gardner, 1993).

In our informal review of over 75 related papers, none of the trail impact studies in arid regions studied effects across a gradient of use levels. Studies in other regions which did measure impact as a function of use levels produced a variety of results (Ballantyne and Pickering, 2015). Most trampling studies reported a definite, often curvilinear or asymptotic, positive relationship between increased use intensity and increased physical and biological impacts (Andrés-Abellán et al., 2006; Ballantyne and Pickering, 2015; Boucher et al., 1991; Wimpey and Marion, 2010), but others found no clear relationship with use levels (Nepal and Way, 2007) or that plant cover increased closer to trails, regardless of use intensity (Bright, 1986; Hall and Kuss, 1989). Some studies found that other factors—visitor behavior, type of use (equestrian, bicycle, pedestrian), floristic community, topography, and others— appeared to be more important than use levels in causing physical and biological impacts (Adkinson and Jackson, 1996; D'Antonio et al., 2016; Dixon et al., 2004).

Recreation activities can cause impacts to soil, vegetation, wildlife, and water (Leung and Marion, 1996), yet principles of sustainable natural resource management include preserving biological diversity and providing safe, enjoyable experiences for visitors. In order to balance these objectives in a given ecological system, it is critical to understand how increased use affects biodiversity. We investigated the resilience of trailside vegetation and soil crusts to different visitation levels in the Sonoran Desert. Thus, we had two objectives for our study: 1) to evaluate trail impacts on vegetation and soil crusts, and 2) investigate whether these impacts are affected by different levels of trail visitation in the Sonoran Desert.

2. Methods

2.1. Sites

The MSP is comprised of Sonoran Desert Upland habitat (Brown et al., 1979) which lies at the northeastern edge of the Phoenix metropolitan area in central Arizona (33.59 N, 111.76 W). Due to the proximity to the Phoenix urban core, the MSP receives heavy visitation by hikers, bikers, and equestrians. The City of Scottsdale estimates that there were approximately 750,000 individual visits in 2016. Motorized vehicles are not permitted in the MSP.

Annual temperatures ranged from 20 °C to 46.7 °C at the two nearest weather stations during the study years 2014–2016 (Flood Control District of Maricopa County, 2017). Precipitation means from the four nearest precipitation gauges indicated rainfall was above average in the years preceding the first two sampling periods of the study (27.7 cm in 2013, 26.9 cm in 2014; Table 1), and slightly below average rainfall in the year preceding the final sampling season (22.6 cm in 2015; Table 1). The Sonoran Desert climate includes two distinct rainy seasons: one in the winter (December–March), and one in the summer (June–September).

The MSP is topographically and biologically diverse, ranging in elevation from 515 to 1237 m above sea level. A biological inventory conducted between 2011 and 2013 found 368 plant species and 188 vertebrate animal species (Jones and Hull, 2014; McDowell Sonoran Conservancy, 2014). There are 14 distinct plant associations distributed across the MSP (Jones and Hull, 2014). Bedrock geology and soil types differ across the MSP, with predominantly metamorphic rock in the south and decomposed granite in the north (Skotnicki, 2016).

Three blocks were identified which contained similar attributes within each block but were distinct between blocks (Table 2). Within each block, 3 trail segments were selected to represent a gradient of trail visitation levels and to minimize differences in plant association, soils, geology, slope, and elevation within block (Table 2). A fourth control transect that had no visitation was established within each block at least 100 m from the trails.

Plant communities differed by block (Table 2). The common associated perennial plant species at the Gateway block are brittlebush (*Encelia farinosa),* barrel cactus (*Ferocactus cylindraceus*), buckhorn cholla (*Cylindropuntia acanthocarpa*), catclaw acacia (*Acacia greggii*), chain fruit cholla (*Cylindropuntia fulgida*), creosote bush (*Larrea tridentata*), and saguaro cactus (*Carnegiea gigantea*). The Tom's Thumb block shares many of the common perennial plant species with the Gateway block, but being higher elevation also contains Arizona desert –thorn (*Lycium exsertum*), Wright's buckwheat (*Eriogonum wrightii*), fairy duster (*Calliandra eriophylla*), globe mallow (*Sphaeralcea ambigua*), goldeneye (*Bahiopsis parishii*), and Mormon tea (*Ephedra aspera*). Creosote bush (*Larrea tridentata*), and saguaro cactus (*Carnegiea gigantea*) are present in lower densities than they are in the Gateway block. Trail segments within the Brown's Ranch block share the same common perennial plants as Tom's Thumb block, however, saguaros are present in greater density (Jones, 2015).

2.2. Trail visitation levels

The research objectives were met by placing paired plots adjacent and 6 m away from trails of contrasting visitation levels. Trail segments were chosen to capture low, medium and high visitation levels within the same biotic community and soil type according to estimates informed by preliminary data from mechanical and volunteer counters.

We used two approaches to quantify visitation rates during the study. First, mechanical counters (Diamond Traffic Products, Model TTC-4420) were placed at the beginning of each trail segment transect (Fig. 1) and collected data for 24 months (2014–2015). Secondly, volunteers seated next to each mechanical counter counted trail visitors for 2 h during peak visitation (7–9 a.m. in summer, 8–10 a.m. in spring and fall, and 9–11 a.m. in winter) on the third Saturday of each month for January through November 2014 and 2015 (hereafter designated as “volunteer count”).

The mechanical counters provide hourly counts of visitors throughout the year. Unfortunately, the mechanical counters appeared to have numerous data inconsistencies, including 1) daytime and nighttime hours sometimes were reversed, as evidenced by visitation occurring during the night (when the MSP is

Table 1
Mean annual precipitation (mm) data from the Flood Control District of Maricopa County (2017) weather stations closest to study sites.

Weather station #	Weather stations	Total 2013	Dev.[a]	Total 2014	Dev.	Total 2015	Dev.	Long term mean[b]
4585	Reata pass wash	274.07	66.04	290.32	82.30	210.57	2.54	208.03
4595	Pinnacle pk vista	228.60	26.92	266.19	64.52	228.35	26.67	201.68
4935	Reata pass dam	335.28	80.77	287.02	32.51	248.41	−6.10	254.51
5930	Fraesfield Mtn	316.23	15.24	278.89	−22.10	267.97	−33.02	300.99
	Means	288.54	47.24	280.61	39.31	238.82	−2.48	241.30

[a] Dev indicates deviation from the long term average.
[b] Long term averages for rainfall guages began in different years: 4585 in 2001; 4595 in 1998; 4935 in 1993; 5930 in 1989.

closed) on popular trails instead of during the day; 2) data were missing from some counters due to mechanical failure; 3) days of the week were not accurate as evidenced by charts of visitation showing peak visitation during the week (rather than on weekends) and finally, 4) erratic data sometimes were recorded with unusually high numbers inconsistent with the trend for that trail.

Rather than attempting to correct the data, we reported the mechanical counter results for the same 2 h blocks that were counted by volunteers one Saturday per month, using only months for which the mechanical counter data were complete during the same times as the volunteer counts. In doing so, we were able to avoid many of the data errors described above. The mechanical counts and the volunteer counts were divided by two to show an hourly average.

The mechanical block counts were consistent with the volunteer block counts. We used the volunteer block data as the basis for visitation rate of the trails because we were most confident of the quality of this data, and because it had a more complete data set than the mechanical block.

2.3. Sampling design

We designed the experiment to compare vegetation adjacent to the trail (15 "trailside plots") with control plots 6 m from the trail (15 "6 m plots") (3 trail segments x 30 plots × 3 blocks = 270 plots, Fig. 1) on trail segments across a gradient of visitation levels. By using this design, the differences between trail and off-trail plots can be attributed to the construction and use of the trail (Leung and Marion, 1996). Trail segments were selected based on three criteria: capturing 450 m between trail junctions, in the same floristic community, with generally consistent cross-trail slope for drainage. Two marker posts were placed approximately 3 m from trail center on either side of the trail at the beginning and end of the 450 m transect. For trailside plots, 1 m × 1 m quadrats were placed every 30 m so that one side of the quadrat frame was lined up with the most visually obvious perimeter of the trail area which receives the majority of the traffic (95%). The boundary is defined by areas of apparent trampling, disturbance to organic litter, and where vegetation cover is reduced or absent (Marion and Leung, 2001; Olive and Marion, 2009; Wimpey and Marion, 2010). On the same side of the trail, a paired 1 m × 1 m control plot was placed 6 m from the trail (5 m from the trailside plot), perpendicular to the trail.

We measured trail depth to evaluate trail soil erosion at the beginning and end of each transect between the two marker posts. We created notches on each marker post at about 1 m height. To establish the three sampling points, we tied a static 2 mm twisted nylon string (>100# tensile strength) between the notches at each post so that the string was taut and level across the transect. The trail center point and points near each edge of the trail were selected, so that all three points fell within the center 80% of the treaded area of the trail. The distances of these three points from the upslope marker post were recorded and the same points were re-sampled each year. Sampling was conducted by measuring vertically downward from the three sample points to the trail tread surface and recording the distances.

In the second year, we added new site control plots 100 m from a trail (no visitation) to test if the 6 m was a sufficient distance to control for trail disturbance effects on the trailside plots. If 100 m plots were similar to 6 m plots compared to the trailside plots, this would indicate that the 6 m distance was sufficient. One site control transect without paired plots was randomly placed in each block at least 100 m from the nearest trail. As in trail segments, 1 m × 1 m quadrats were placed every 30 m for a total of 15 plots per site control (3 blocks × 15 plots = 45 plots total). Permanent markers were placed in all four corners of each quadrat to ensure the same area would be sampled each year.

2.4. Sampling

Visual estimates of coverage for each plant species and for visible soil crust were made in each 1 m × 1 m plot using six cover classes modified from the Braun-Blanquet cover-abundance scale (Mueller-Dombois and Ellenberg, 1974) as follows (cover class = range = midpoint): 1= <1% = 0.5; 2 = 1–5% = 3; 3 = 5–25% = 15; 4 = 25–50% = 37.5; 5 = 50–75% = 62.5; 6 = 75–100% = 87.5. Plants were considered in the plot if any part of the plant was in or hanging over the edge of the quadrat. Plant and soil crust sampling was conducted at peak spring annual biomass (February–April) each year from 2014 to 2016.

Table 2
Physical attributes of blocks within Scottsdale's McDowell Sonoran Preserve.

Block	Plant community[a]	Soil type[b]	Bedrock[c]	Slope range	Elevation range (m)
Gateway	*Ambrosia deltoidea – Parkinsonia microphylla* mixed scrub association (154.121)	Extremely Sandy Loam	Metamorphic	20–29% (but 5% along 100 m site control)	605–772
Tom's Thumb	*Simmondsia chinensis* – mixed scrub association (154.123)	Very gravely sandy loam	Granite	5–11%	774–901
Brown's Ranch	*Simmondsia chinensis* – mixed scrub association (154.123)	Very gravely sandy loam, very gravely clay loam	Granite	0–3%	804–835

[a] (Jones, 2015).
[b] (USDA Natural Resource Conservation Service, 2017).
[c] (Skotnicki, 2016).

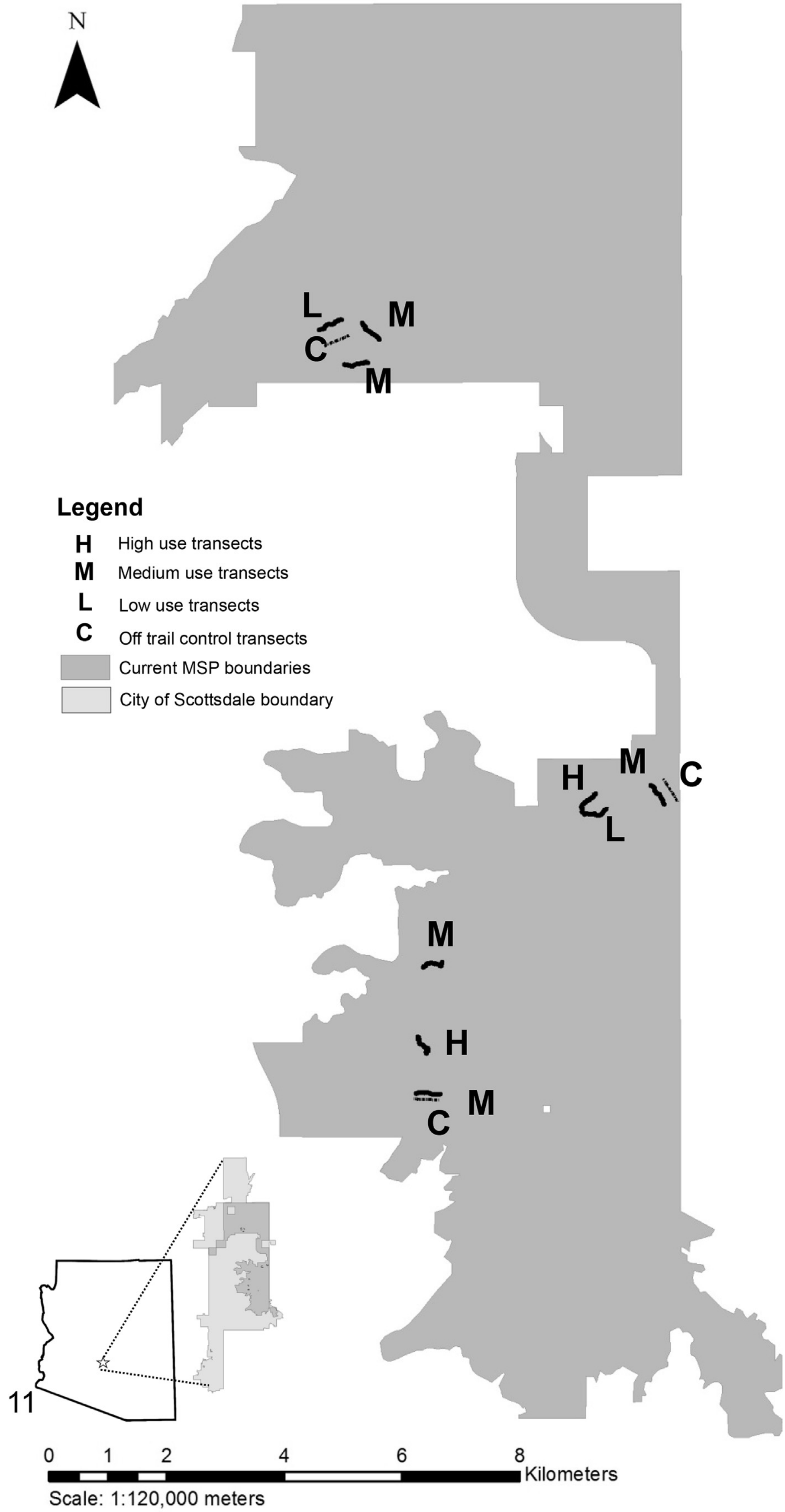

Fig. 1. Locations of transects in Scottsdale's McDowell Sonoran Preserve.

2.5. Data analyses

Volunteer count data was used to establish trail visitation levels. This data was log transformed to meet assumptions of a normal distribution and analyzed using a mixed model with trail segment as a fixed effect. Tukey's HSD was used to compare post hoc multiple comparisons.

Richness and percent cover for native and nonnative plants and functional groups (native perennial and native annual plants) were calculated. The three soil depth points in each trail segment end were averaged, thus there were two averaged data points per year per transect ($n = 6$ per transect). Differences between subsequent years (2015–2014 and 2016–2015) were calculated using the averaged point for each transect end (resulting in two averaged differences for each transect and year pair, $n = 4$ per transect). Residual plots were used to evaluate the distributional properties of the data. We transformed the data to best meet assumptions of normality and homogeneity of variance and to minimize outlier effects. Native species richness and cover, native perennial cover and richness, and native annual richness were square root transformed; native annual cover and soil depth differences were log transformed. Data from three trailside −6 m plot pairs along a medium trail segment for 2015 and one 100 m site control for 2016 were missing. Two plots with atypical rock cover were detected in the native plant richness data as outliers. We ran the analyses with and without these plots and although the resulting significance of the tests did not change, the outliers were highly influential in the model and thus removed. These plots were not removed from other analyses as they had no effect.

A repeated measures mixed model design with an autoregressive covariance error structure (AR (1)) with year, trail visitation level, plot position (trailside, 6 m), and the interaction between trail visitation and plot position as fixed effects, and block as a random effect was used to evaluate native plant cover and richness, including native annual and perennial functional groups. Soil depth means were analyzed using the same model without (AR (1)) with year as a fixed effect and block as a random effect to test whether soil depth changed over time. Soil depth differences were also analyzed using the same model with (AR (1)), with year as 2014–2015 and 2015–2016 and trail visitation levels as fixed effects, and block as a random effect to test whether soil depth changed with trail visitation. Tukey's HSD was used to compare post hoc multiple comparisons. We also ran the same analyses with the mean deviation from average rainfall for each year as a covariate instead of time. The model output was very similar between the two approaches, indicating that year is largely a proxy for rainfall. We report analyses with time as a covariate because it includes more factors associated with year other than total annual rainfall.

Due to the lack of normality of nonnative cover, nonnative richness and soil crust, alternative modeling techniques were employed. These three variables had a high number of measured zero values and few unique percentages recorded. Thus, methods typically used for count data, like Poisson regression, provided more appropriate models. The values were rounded to the nearest integer for modeling. Similar to the standard mixed models, a random effect for block that explicitly models the relationship among the transects within a block was used. However, an autoregressive covariance error structure (AR (1)) term could not be included. All models evaluated the impact of plot position and trail visitation level and their interaction controlling for year. Nonnative richness was modelled using a Poisson mixed model, the square root of soil crust was modelled using negative binomial mixed model, and nonnative cover was modelled using a generalized Poisson mixed model to account for overdispersion in the standard Poisson Mixed Model (SAS Institute Inc., 2012). Tukey-Kramer was used for multiple comparisons tests.

The 100 m site controls were analyzed with 2015 and 2016 data to test whether the 6 m distance from trails was a sufficient distance to control for trail effects. The response variables were tested using the same models described above. In each model plot position (trailside, 6 m, 100 m) and time as the covariate were included as fixed effects and block as a random effect.

All data analyses were generated using SAS/STAT software. Copyright © 2012 SAS Institute Inc. SAS and all other SAS Institute Inc. product or service names are registered trademarks or trademarks of SAS Institute Inc., Cary, NC, USA. Plots were generated in R (R Core Team, 2014) using GGPLOT2 (Wickham, 2009).

3. Results

3.1. Trail visitation

Trail visitation was significantly different ($p < 0.0001$, $F_{(8,148)} = 59.38$) and separated cleanly into three clear levels of use based on Tukeys HSD (Table 3). Relative trail visitation levels within block were in agreement across the different estimation methods (mechanical and human counters; Table 3).

3.2. Soil crusts

The loss of visible soil crusts on trailside plots compared with 6 m plots was mediated by the level of visitation along trails (significant interaction, Fig. 2, Table 4). Along low visitation trail segments, there was no difference between soil crusts trailside and at 6 m, but along medium and high visitation trails, this difference was significant ($P < 0.0001$, $P = 0.0002$, respectively; Fig. 2). In the comparison with the 100 m site controls, both controls (6 m and 100 m) supported significantly more soil crusts than the trailside plots ($p < 0.0001$).

3.3. Plant community response

Native plant cover and richness was reduced on trailside plots compared to the 6 m controls (Table 4). Responses of native plant cover and richness were most pronounced at the level of functional group (Tables 4 and 5), therefore in the figures we present plant cover and richness by functional group. Year as a covariate was significant for every plant response variable.

Native perennial richness and cover was higher on trailside plots compared with 6 m plots, averaged over trail visitation level at the (Table 4, Figs. 2 and 3). Native annual cover had a marginally significant interaction (Table 4), whereby annual plant cover was higher in the trailside plots at low visitation levels, but in the higher use plots cover was the same in trailside and 6 m plots (Fig. 2).

Nonnative cover had a significant interaction in which trailside plots with low use had more nonnative cover than the 6 m plots, but the pattern switched at medium and high visitation levels (Table 4, Fig. 2). Nonnative richness was highest on low use trails, and reduced on medium and high use trails averaged over distance from trail. (Table 4, Figs. 2 and 3).

In the comparisons including the 100 m site controls and only 2015 and 2016 data, the only plant parameters with significant plot position effects (trailside, 6 m, 100 m) were native plant richness, native perennial richness, and nonnative cover (Table 5). No differences in native plant richness were detected on plots 100 m from trails compared with either the 6 m or trailside plots. But trailside plots had lower perennial plant cover than 6 m (Tukey's adjusted $P = 0.03$), consistent with the mixed model results without the site controls (Fig. 2). However native perennial richness on the 100 m

Table 3
Mean visits per hour estimated by mechanical counter and volunteers for peak visitation. Capital letters denote difference between trail use based on volunteer counts using Tukey's HSD multiple means comparisons.

Block	Trail	Mechanical Count			Volunteer Count			Visitation level
		N	Mean	S.E.	N	Mean	S.E.	
Tom's Thumb	Tom's Thumb	9	59.61	7.23	19	64.37	6.22	$High^A$
Gateway	Gateway Saddle	7	46.07	9.00	15	61.13	6.80	$High^A$
Gateway	Windgate	6	15.83	1.35	17	23.82	2.86	$Medium^B$
Brown's Ranch	Upper Ranch	2	26.00	0.50	18	23.06	2.60	$Medium^B$
Gateway	Bell Pass	6	21.83	2.27	18	22.78	2.61	$Medium^B$
Tom's Thumb	Marcus Landslide	9	19.89	4.99	19	19.42	3.14	$Medium^B$
Brown's Ranch	Hackamore	3	14.50	3.40	18	13.69	1.43	$Medium^B$
Tom's Thumb	Feldspar	1	3.00		19	3.32	0.79	Low^C
Brown's Ranch	Rustler	9	0.61	0.40	17	2.44	0.39	Low^C

plots was lower than the 6 m plots (Tukey's adjusted $P < 0.01$) and similar to the trailside plots. Trailside plots had reduced perennial cover compared with the 6 m plots (Tukey's adjusted $P = 0.01$), again consistent with the mixed model results without the site controls (Fig. 2). As expected from the mixed model results without the site controls, trailside and 6 m nonnative plant cover did not

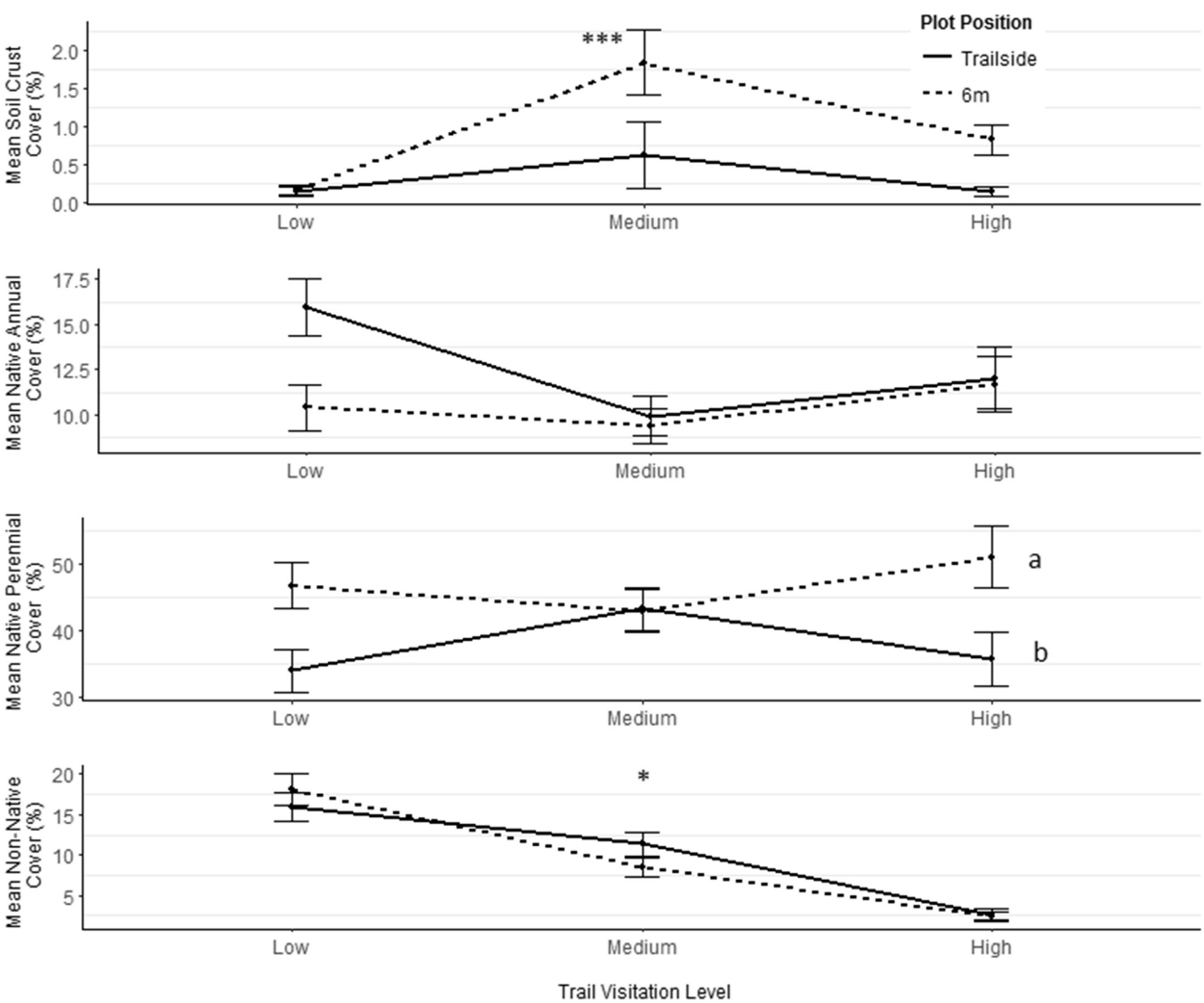

Fig. 2. Mean soil crust, native annual, native perennial, and nonnative percent cover by trail visitation level and plot position, averaged over year. Each error bar is constructed by using one standard error from the mean. Highest level significant differences are indicated by an asterisk for an interaction effect (*p < 0.05, **p < 0.01, ***p < 0.001). If no interaction, differences between treatments using Tukey's HSD multiple means comparisons are indicated by upper case letters for a trail visitation level effect (averaged over distance from trail) and lower case letters for differences between trailside and 6 m from trail plots (averaged over trail visitation).

Table 4
Full factorial mixed model analysis results for soil crust cover and percent plant cover and species richness of each functional group.

	Plot position (trailside, 6 m)			Trail visitation level (high, medium, low)			Year (2014–2016)			Position * Visitatio		
	df	F	P	df	F	P	df	F	P	df	F	P
Soil crust cover	1757	31.60	<0.0001	2757	2.62	0.074	2757	0.99	0.374	2757	3.76	0.024
Native plant cover												
Richness	1791	6.12	0.014	2791	0.69	0.502	2791	72.97	<0.0001	2791	2.44	0.088
Cover	1793	4.81	0.029	2793	1.85	0.158	2793	9.84	<0.0001	2793	1.94	0.145
Native perennials												
Richness	1800	13.09	0.000	2800	0.05	0.956	2800	14.05	<0.0001	2800	1.13	0.322
Cover	1800	5.69	0.017	2800	0.11	0.892	2800	6.73	0.001	2800	1.78	0.169
Native annuals												
Richness	1800	0.12	0.733	2800	1.53	0.218	2800	25.52	<0.0001	2800	0.84	0.434
Cover	1763	0.76	0.384	2763	1.74	0.176	2763	13.10	<0.0001	2763	2.58	0.076
Nonnatives												
Richness	1793	0.11	0.735	2793	12.44	<0.0001	2793	0.62	0.538	2793	0.73	0.483
Cover	1793	0.00	0.992	2793	22.02	<0.0001	2793	21.25	<0.0001	2793	3.08	0.047

* sig p</ = 0.05.
**p < 0.01.
***p < 0.001.
*df = numerator degrees of freedom, denominator degrees of freedom.

differ, but the 100 m plots (adjusted means, 9.64 ± 6.29) were different from the 6 m plots (adjusted means, 6.58 ± 4.27, Tukey's adjusted P < 0.01) and the trailside plots (adjusted means, 7.21 ± 4.67, Tukey's adjusted P = 0.04).

3.4. Soil erosion

There was no detectable change in soil depth within the trail corridor over the three years of the study (P = 0.97, $F_{2,43} = 0.03$). When compared across trail visitation level, using 1-year differences, there were no differences detected by visitation levels, (P = 0.48; $F_{2,11} = 0.48$) or time (2014–2015, 2015–2016, P = 0.56, $F_{1,11} = 0.35$).

4. Discussion

4.1. Soil crusts

Visible soil crusts had the strongest response to the different trail visitation levels studied. At the low visitation rate, soil crusts persisted along the trails, but trailside soil crusts were highly reduced at higher usage levels (an estimated peak rate of 13–70 visitors per hour). Likely trampling next to the trail caused these reductions. In a study conducted in the arid southwest at Grand Canyon National Park, Arizona, just 15 trampling passes caused structural damage and 250 passes destroyed the crusts (Cole, 1990). In contrast to our findings, a study in the Canadian Rockies, Alberta, found no difference in soil crusts or vegetation cover or richness between trailside plots and their 50 m reference plots, even though the trail received over 2000 visits during the summer 2008 when the study was conducted (Crisfield et al., 2012). The difference may be that these trails had a visible rock boundary (Crisfield et al., 2012), which may have discouraged visitors from trampling areas adjacent to the trail, whereas McDowell Sonoran Preserve trails do not have marked trail boundaries. Soil crusts have a critical function in arid lands to stabilize soils and thereby maintain soil fertility. Soil crusts may take at least 20 years to recolonize and provide soil erosion protection (Belnap and Gillette, 1997) and soils take 5000 to 10,000 years to form in arid areas (Webb, 1983). Thus these impacts are consequential and important indicators of disturbance (Allen, 2009; Belnap, 1998).

Table 5
Mixed model analysis results for soil crust cover and percent plant cover and species richness of each functional group for the comparisons with the 100 m site controls.

	Plot position (trailside, 6 m, 100 m)			Year (2014–2016)		
	df	F	P	df	F	P
Soil crust cover	2617	23.71	<0.0001	1617	0.00	0.990
Native plants						
Richness	2616	3.67	0.026	1616	52.3	<0.0001
Cover	2617	0.82	0.440	1617	11.23	0.001
Native perennials						
Richness	2625	6.48	0.002	1625	1.45	0.229
Cover	2625	1.45	0.235	1625	1.20	0.273
Native annuals						
Richness	2625	0.4	0.668	1625	25.57	<0.0001
Cover	2590	2.23	0.109	1590	15.32	0.000
Nonnatives						
Richness	2617	0.73	0.480	1617	0.08	0.773
Cover	2617	5.15	0.006	1617	28.86	<0.0001

* sig p</ = 0.05.
**p < 0.01.
***p < 0.001.
*df = numerator degrees of freedom, denominator degrees of freedom.

4.2. Native plant community

In areas next to trails perennial plant richness and cover were reduced compared to the 6 m plots. Generally, trail construction of established trails has a large initial effect (vegetation removal, soil disturbance) followed by lesser direct and indirect impacts (Cole, 1987; Leung and Marion, 1996). The reduction of perennial plants alongside trails in our study may be attributable to either initial trail building, subsequent trampling effects next to trails, or trail widening, but the effect was not driven by the different visitation levels studied. Nepal and Way (2007) used a similar experimental design comparing trailside and control plots by one high and one low use trail in a Provincial Park, British Columbia, Canada. Similar to our results, they found that the high use trailside plots had less vegetation and moss, lichen, and fungi cover, and less species richness than control plots, and also that trailside plots on both high and low use trails had more exposed soil than the controls (Nepal and Way, 2007). In contrast to our results, others found that species richness increased in high use trailside plots (Bright, 1986; Hall and Kuss, 1989; Roovers et al., 2004a). This result is likely due to a release from competition when the dominant or canopy species open up light or other resources to allow other species to compete in the environment. Given these findings, we might have expected

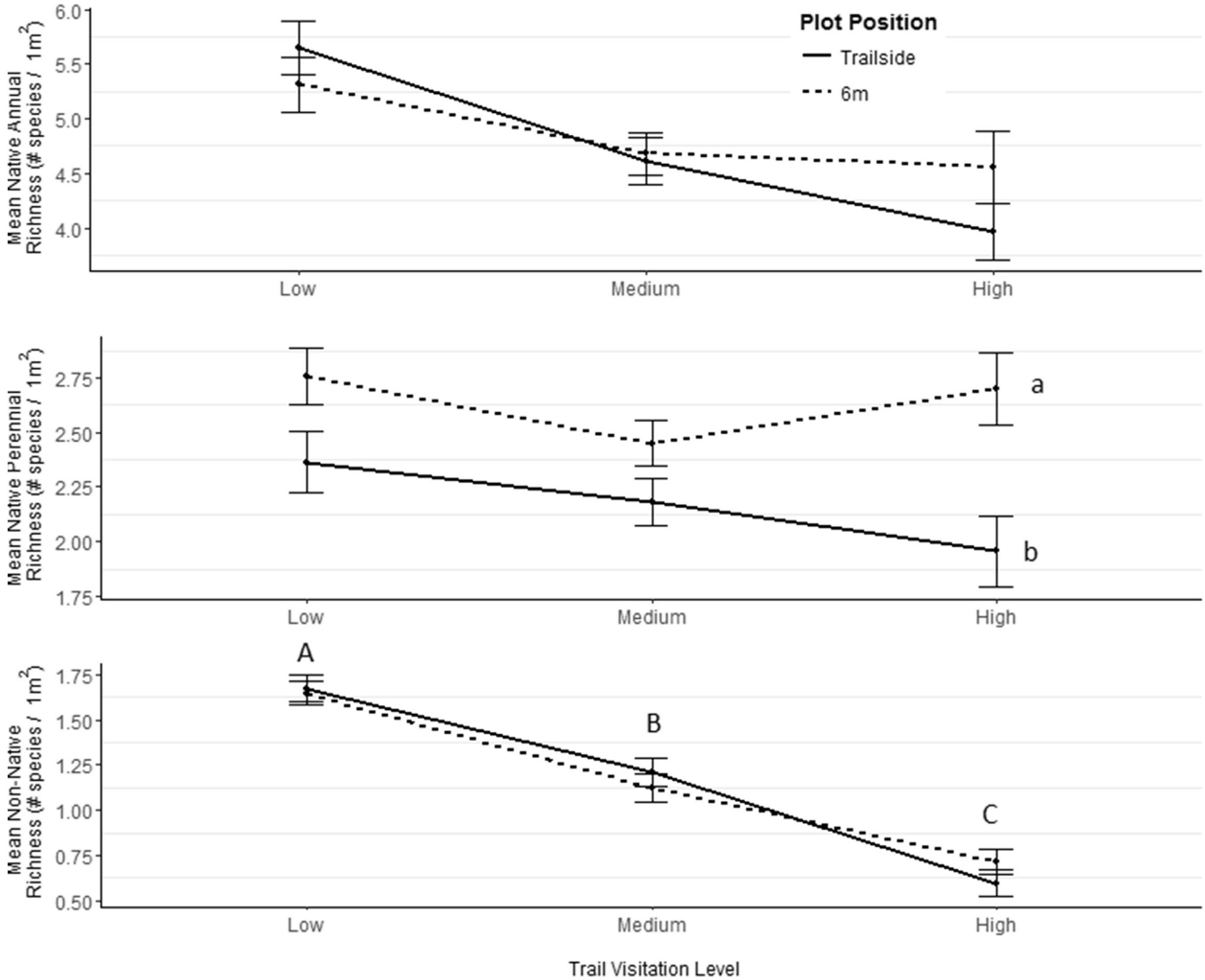

Fig. 3. Percent plant richness of functional groups by trail visitation and plot position. Each error bar is constructed by using one standard error from the mean. Highest level significant differences are indicated by an asterisk for an interaction effect (*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$). If no interaction, differences between treatments using Tukey's HSD multiple means comparisons are indicated by upper case letters for a trail visitation level effect (averaged over distance from trail) and lower case letters for differences between trailside and 6 m from trail plots (averaged over trail visitation).

an increase in annual plant richness or cover given the reduction in perennial cover in trailside plots, but annual plant cover was not different between the trailside and 6 m control plots. In the Sonoran Desert, annuals' limiting resource is moisture rather than light; winter annuals germinate in response to at least one 2.5 cm soaking rain event between October and January (Dimmitt, 2000), and can benefit from moisture retention by perennial plants. In all plant analyses, year, as a covariate which we found to be a good proxy for rainfall, was a strong predictor of plant parameters. This indicates that there is strong interannual variability explained largely by rainfall (Table 1).

Several studies have tried to make generalizations about which systems are most susceptible to trampling or trails, but the results range from densely forested vegetation in Oregon compared with open meadows or open forests (Cole, 1978); understory vegetation of mesophilous forests compared with heath and dry forests in Belgium (Roovers et al., 2004b); floodplains and hemlock communities compared with mesic and dry-mesic upland communities in Indiana (Adkinson and Jackson, 1996); and shrublands compared with alpine grassland, fen and bolster heath sites in Tasmania (Whinam and Chilcott, 1999). Cole (1995a) results suggest that instead of focusing on ecological systems, plant morphological characteristics may better explain response to trampling (Cole, 1995a). Specifically, vegetation stature and erectness increases resistance, and matted grasses and sedges have the most tolerance to trampling, and broad leaved forbs the least (Cole, 1995a, 1995b; Monz et al., 2010). However, others found that upright woody shrubs and tall grasses were most vulnerable to trampling (Whinam and Chilcott, 2003). In the arid southwest system studied here, the annual plants which appear for only a short few months after winter rains appear to be most resilient to trail effects.

4.3. Nonnative plants

We had expected that nonnative plants would be more abundant along the trail compared with the 6 m control plots and would increase with visitation. Although an interaction response was detected in nonnative cover, the differences were very small between trailside and the 6 m plots, and richness decreased for both trailside and 6 m plots nearby higher use trails, contrary to the expected response. In the site control plots located 100 m from the nearest trail, nonnative cover was slightly higher than trailside and 6 m plots indicating that the nonnative species found in the trail corridor do not appear to be distributed by trails. There were only six nonnative species found in our plots over the three years of the study, and two species were only found in one plot each and another in nine plots. The three main nonnatives were red brome (*Bromus rubens),* common Mediterranean grass (*Schismus barbatus),* and redstem stork's bill (*Erodium cicutarium),* all of which are common throughout the MSP and other Phoenix area parks. Our results, that nonnatives do not follow a pattern that indicate that trails function as a primary conduit for nonnative plant spread into natural areas, are supported by studies comparing trailside and interior plots in the Rocky Mountains and foothills (Potito and Beatty, 2005; Tyser and Worleya, 1992). In contrast, however, a similar study in Ontario, Canada showed a pattern consistent with spread from the trail outwards in which nonnative species were highest along trails and decreased at three and 15 m from the trail (Patel, 2000). Other studies compare nonnative prevalence from trailheads into the natural areas along trails finding a positive correlation with visitation use and nonnative plant species. These studies found that sites closer to the trailhead with higher use had an increase of nonnative species cover (Potito and Beatty, 2005) and richness (Bella, 2011) compared with less travelled trail sections farther into the recreation area. Bella (2011) also found that the nonnative richness spread a longer distance from the trailhead on high use trails. Although it's not clear from these studies to what extent trails and trail visitation level contribute to the spread of nonnative plants, in our system the two perennial plant species of highest concern, buffelgrass (*Pennisetum ciliare*) and fountain grass (*Pennisetum setaceum*), were not found in any of the study plots. This, together with the similarity between the trailside and 6 m control plots, suggest that trails are not a strong conduit for nonnatives in the McDowell Sonoran Preserve at the time of the study.

4.4. Soil erosion

Surprisingly, there was no detectable change in soil erosion in the trails as determined by repeated trail-depth measurements, regardless of trail visitation level. This was unexpected since the trails included in the study had markedly different geological and tread characteristics, from erosion-prone granitic grus (sand and small gravel) in the Brown's Ranch and Tom's Thumb blocks to erosion-resistant exposed metamorphic rock and compacted fine dirt in the Gateway block (Skotnicki, 2016), as well as different usage levels within and between blocks. Our results are contrasted by another study conducted in the arid southwest (Guadalupe Mountains National Park, Texas, USA) in which most measured sites showed active soil movement over a 17-month period in an area composed primarily of limestone, especially on major trails compared with control transects (Fish and Brothers, 1981). These changes appeared to be due primarily to sediment deposition rather than erosion, as the trails intercepted waterborne sediment (Fish and Brothers, 1981). Several studies suggest that trail incision depth and rates are highly influenced by trail design (trail grade and trail-slope angle) as shown in South Carolina, USA (Beeco et al., 2013), western Australia (Gager and Conacher, 2001), and western Tasmania (Dixon et al., 2004), and only moderately influenced by trail use (Beeco et al., 2013; Dixon et al., 2004). Beeco et al. (2013) found that horse traffic had similar influence over trail erosion as trail design. In a study of erosional loss in relation to trail design, the highest levels of erosion were associated with either very steep trails or steep trails that also were aligned close to the natural fall line (the route leading down any particular slope) (Marion and Wimpey, 2017), suggesting that the design of trails in our study may have been effective at minimizing erosion.

4.5. Site controls

For the most part, the site control plots that were placed in each block 100 m away were similar to the 6 m plots, suggesting that 6 m was a sufficient distance from the trail to serve as a control from disturbance associated with the trail. Native perennial richness deviated from this pattern in which 100 m plots were more similar to trailside than the 6 m plots. It is possible that because there was only one 100 m site control per block, the site controls were too distant and different from the trailside plots to compare plant community characteristics. However, the pattern found with nonnative plant cover using the site controls helps elucidate patterns of nonnative plant dispersal as discussed above.

5. Management implications

Our results indicate that in this system trails concentrate high levels of use into a small area with little direct impact on surrounding protected natural areas as they are designed to do as long as visitors remain "on-trail". Careful design planning at the time of new trail construction can incorporate natural barriers that discourage off-trail use such as running trail segments alongside boulders or established growth whenever possible or using existing corridors like abandoned roads that already are incised into the terrain. To reduce the localized degradation of soil crusts on established trails, managers may consider site management actions such as physical barriers in conjunction with education campaigns, which have been found to be effective in reducing off-trail use (Hockett et al., 2017; Littlefair and Buckley, 2008; Park et al., 2008). In keeping with the natural aesthetic of the Preserve, we recommend planting vegetation that is both consistent with the surrounding plant community and also of sufficient size or with characteristics that will discourage off-trail use. For example, in the Sonoran Desert buckhorn cholla (*Cylindropuntia acanthocarpa*) or teddy bear cholla (*Cylindropuntia bigelovii*) plants or segments are common local species that can be easily transplanted and create a prickly barrier for visitors.

Trail maintenance can also reduce or eliminate problems within a trail that may be causing visitors to go off-trail to avoid obstacles. Signage to stay on the trail can be effective in combination with revegetation (Hockett et al., 2017). New interpretive displays at trailheads can explain the effects of trail widening on soil crusts and encourage visitors to stay within the trail boundary and "don't bust the crust". Because these trails are multi use, recommendations for horse riders could include choosing wider trails and for mountain bikers to be especially vigilant about keeping their turning radius within the established trail. Etiquette tips for allowing other users to pass could include trying to stay on the trail if possible or carefully choosing a rock or soil crust free area off trail to stand. Since personal contacts may be more effective than interpretive signs in changing off trail use (Hockett et al., 2017; Littlefair and Buckley, 2008), managers and volunteers could be trained and encouraged to promote the message.

6. Conclusions

Monz et al. (2013) reviewed the relationship between recreation use and impact on vegetation and soil. The predominant model is asymptotic and curvilinear, whereby in an undisturbed area with low use levels, small differences in the amount of visitation can lead to substantial impact, whereas in already disturbed areas with high use (trails or campsites), additional impact has proportionally less effect (Monz et al., 2013). An alternative model suggests a sigmoidal response in which at low levels of use, there is little impact, but at medium use, there is a steep impact curve that flattens again after a secondary threshold has been reached (Monz et al., 2013). In our system, only soil crust followed the sigmoidal response; plant cover and richness in trailside plots did not show significant trends according to use level indicating that either additional impact beyond initial trail building has little effect, or that these trails have not experienced high enough use or disturbance to reach a first threshold. The shape of the curve has important management implications. If trail visitation continues to grow in the MSP, it would be useful to know whether to expect a second threshold or continued minimal response. As discussed earlier, trail impacts are expected to differ by ecological system because environmental factors such as climate and geology and the intermediate elements of topography, soil, and vegetation type significantly affect the degree and type of trail degradation (Leung and Marion, 1996). For the first time, we were able test how plants and soil crusts respond to trail proximity and different trail visitation levels, as well as test soil erosion on trails in a Sonoran Desert setting. We found that soil crusts are an early indicator of disturbance, and that perennial and annual plants and soil erosion appear to be resilient to change at least at current levels of use. Trails did not appear to be an important conduit for nonnative species dispersal into the MSP.

Acknowledgements

We thank the McDowell Sonoran Conservancy and an anonymous donor for financial support of this project. A special thanks goes to L. Miller for developing the citizen science training and structure for this project. Citizen scientists K. Fieldstad, B. Pringle, J. Millavec, J. McEnroe, B. Sowle, C. Holt and L. Rivera provided essential project leadership and K. Barron and S. Doan shared their botanical expertise. Thank you to the additional 60 citizen scientists who helped with field data collection and data entry. The City of Scottsdale provided permission for conducting this study in the McDowell Sonoran Preserve.

References

Adkinson, G.P., Jackson, M.T., 1996. Changes in ground-layer vegetation near trails in midwestern US forests. Nat. Area J. 16, 10–23.

Allen, C.D., 2009. Monitoring environmental impact in the upper sonoran lifestyle: a new tool for rapid ecological assessment. Environ. Mutagen. 43, 346–356. https://doi.org/10.1007/s00267-008-9212-5.

Andrés-Abellán, M., López-Serrano, F.R., Morote, F.A.G., Del Cerro-Barja, A., 2006. Assessment of trampling simulation impacts on native vegetation in Mediterranean sclerophyllous forest. Environ. Monit. Assess. 120, 93–107. https://doi.org/10.1007/s10661-005-9051-2.

Ballantyne, M., Pickering, C.M., 2015. The impacts of trail infrastructure on vegetation and soils: current literature and future directions. J. Environ. Manag. https://doi.org/10.1016/j.jenvman.2015.08.032.

Beeco, J.A., Hallo, J.C., English, W.R., Giumetti, G.W., 2013. The importance of spatial nested data in understanding the relationship between visitor use and landscape impacts. Appl. Geogr. 45, 147–157. https://doi.org/10.1016/j.apgeog.2013.09.001.

Bella, E.M., 2011. Invasion prediction on Alaska trails: distribution, habitat, and trail use. Invasive Plant Sci. Manag. 4, 296–305. https://doi.org/10.1614/IPSM-D-10-00083.1.

Belnap, J., 1998. Choosing indicators of natural resource condition: a case study in Arches National Park, Utah, USA. Environ. Mutagen. 22, 635–642. https://doi.org/10.1007/s002679900135.

Belnap, J., 1994. Potential value of cyanobacterial inoculation in revegetation efforts. In: Monsen, S.B., Kitchen, S.G., P.-E., M., A.R (Eds.), Ecology and Management of Annual Rangelands. U.S. Department of Agriculture, Forest Service, Ogden, Utah, pp. 179–185. Technical Report INT-GTR-313.

Belnap, J., Gardner, J., 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium Microcoleus vaginatus. West. North Am. Nat. 53, 40–47.

Belnap, J., Gillette, D.A., 1997. Disturbance of biological soil crusts: impacts on potential wind erodibility of sandy desert soils in SE. Land Degrad. Dev. 8, 335–362.

Boucher, D.H., Aviles, J., Chepote, R., Domínguez Gil, O.E., Vilchez, B., 1991. Recovery of trailside vegetation from trampling in a tropical rain forest. Environ. Mutagen. 15, 257–262. https://doi.org/10.1007/BF02393857.

Bright, J.A., 1986. Hiker impact on herbaceous vegetation along trails in an evergreen woodland of Central Texas. Biol. Conserv. 36, 53–69. https://doi.org/10.1016/0006-3207(86)90101-1.

Brooks, M.L., Lair, B., 2005. Ecological Effects of Vehicular Routes in a Desert Ecosystem. USA US Dep, Las Vegas, NV. Inter. … 23.

Brown, D.E., Lowe, C.H., Pase, C.P., 1979. A digitized classification system for the biotic communities of North America, with community (series) and associated examples for the Southwest. J. Arizona-Nevada Acad. Sci.

Cole, D., 1978. Estimating the susceptibility of wildland vegetation to trailside alteration. J. Appl. Ecol. 15, 281–286.

Cole, D., 1987. Research on soil and vegetation in wilderness: a state-of-knowledge review. General Technical Report INT-220. In: Lucas, R.C. (Ed.), Proceedings-National Wilderness Research Conference: Issues, State-of- Knowledge, Future Directions. Forest Service, Intermountain Research Station, Fort Collins, CO, pp. 135–177.

Cole, D.N., 1995a. Experimental trampling of vegetation. II. Predictors of resistance and resilience. J. Appl. Anim. Res. 32, 215–224.

Cole, D.N., 1995b. Experimental trampling of vegetation. I. Relationship betweeen trampling intensity and vegetation response. J. Appl. Ecol. 32, 203–204.

Cole, D.N., 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. Gt. Basin Nat. Nat. 50, 321–325.

Crisfield, V.E., Macdonald, S.E., Gould, A.J., 2012. Effects of recreational traffic on alpine plant communities in the northern Canadian Rockies. Arctic Antarct. Alpine Res. 44, 277–287. https://doi.org/10.1657/1938-4246-44.3.277.

D'Antonio, A., Monz, C., Larson, N., Rohman, A., 2016. An application of recreation resource assessment techniques to inform management action in an urban-proximate natural area. J. Outdoor Recreat. Tour. 14 https://doi.org/10.1016/j.jort.2016.04.002.

Dimmitt, M.A., 2000. Plant ecology of the sonoran desert region. In: Phillips, S.J.W.C.P. (Ed.), A Natural History of the Sonoran Desert. Arizona-Sonora Desert Museum Press, Tucson, AZ, pp. 129–152.

Dixon, G., Hawes, M., McPherson, G., 2004. Monitoring and modelling walking track impacts in the Tasmanian wilderness world heritage area, Australia. J. Environ. Manag. 71, 305–320. https://doi.org/10.1016/j.jenvman.2004.02.006.

Fish, B., Brothers, L., 1981. NATIONAL which is largely responsible for the uniqueness of this site. The driest from November through April (70 % of the annual precipitation occurs from May through October). Annual precipitation ranges from 340 mm 8.

Flood Control District of Maricopa County, 2017. Climate Data [WWW Document]. http://alert.fcd.maricopa.gov/alert/Google/v2/raininfo.html. (Accessed 5 February 2017).

Gager, P., Conacher, A., 2001. Erosion of access tracks in Kalamunda National Park, Western Australia: causes and management implications. Aust. Geogr. 32, 343–357. https://doi.org/10.1080/00049180120100068.

Hall, C.N., Kuss, F.R., 1989. Vegetation alteration along trails in Shenandoah National Park, Virginia. Biol. Conserv. 48, 211–227. https://doi.org/10.1016/0006-3207(89)90119-5.

Hammitt, W.E., Cole, D.N., 1998. Wildland Recreation: Ecology and Management, second ed. John Wiley and Sons, New York.

Harper, K.T., Marble, J.R., 1988. A role for nonvascular plants in management of arid and semiarid rangelands. In: Tueller, P.T. (Ed.), Vegetation Science Applications for Rangeland Analysis and Management. Kluwer Academic Publisher, Dordrecht, Germany, pp. 135–169.

Hockett, K.S., Marion, J.L., Leung, Y.F., 2017. The efficacy of combined educational and site management actions in reducing off-trail hiking in an urban-proximate protected area. J. Environ. Manag. 203, 17–28. https://doi.org/10.1016/j.jenvman.2017.06.073.

Johansen, J.R., 1993. Cryptogamic crusts of semiarid and arid lands of North America. J. Phycol. 29, 140–147.

Jones, S., 2015. Plant associations of Scottsdale's McDowell sonoran preserve: a report to the city of Scottsdale doi:MSCFI-MSP-008-R.

Jones, S., Hull, C., 2014. Vegetation and flora of the McDowell sonoran preserve. Canoma 10, 1–34.

Kuss, F.R., Graefe, A.R., Vaske, J.J., 1990. Visitor impact management: a review of research. Natl. Park. Conserv. Assoc. 256.

Leung, Y., Marion, J.L., 2000. Recreation impacts and management in Wilderness: a state-of-knowledge review the field of recreation ecology. In: Cole, David N., McCool, Stephen F., Borrie, W.T.O., Jennifer, C. (Eds.), Wilderness Science in a Time of Change Conference— Volume 5: Wilderness Ecosystems, Threats, and Management. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, pp. 23–48. Yu-Fai Leung is Assistant Professor, Department of Parks, Recreation and, Missoula, MT.

Leung, Y.F., Marion, J.L., 1996. Trail degradation as influenced by environmental factors: a state-of-the-knowledge review. J. Soil Water Conserv. 130–136.

Littlefair, C., Buckley, R., 2008. Interpretation reduces ecological impacts of visitors to world heritage site. AMBIO A J. Hum. Environ. 37, 338–341. https://doi.org/10.1579/07-R-393.1.

Marion, J.L., Leung, Y.-F., 2001. Trail resource impacts and an examination of alternative assessment techniques. J. Park Recreat. Adm. 19, 17–37.

Marion, J.L., Wimpey, J., 2017. Assessing the influence of sustainable trail design and maintenance on soil loss. J. Environ. Manag. 189, 46–57. https://doi.org/10.1016/

j.jenvman.2016.11.074.
McDowell Sonoran Conservancy, 2014. The Flora and Fauna of Scottsdale's McDowell Sonoran Preserve.
Metting, B., 1991. Biological Surface Features of Semiarid Lands and Deserts. Marcel Dekker, Inc, New York.
Monz, C.A., Cole, D.N., Leung, Y.F., Marion, J.L., 2010. Sustaining visitor use in protected areas: future opportunities in recreation ecology research based on the USA experience. Environ. Mutagen. 45, 551–562. https://doi.org/10.1007/s00267-009-9406-5.
Monz, C.A., Pickering, C.M., Hadwen, W.L., 2013. Recent advances in recreation ecology and the implications of different relationships between recreation use and ecological impacts. Front. Ecol. Environ. 11, 441–446. https://doi.org/10.1890/120358.
Mueller-Dombois, D., Ellenberg, H., 1974. Aims and Methods of Vegetation Ecology. John Willey, New York.
Nepal, S.K., Way, P., 2007. Comparison of vegetation conditions along two backcountry trails in mount Robson Provincial Park, British Columbia (Canada). J. Environ. Manag. 82, 240–249. https://doi.org/10.1016/j.jenvman.2005.12.016.
Olive, N.D., Marion, J.L., 2009. The influence of use-related, environmental, and managerial factors on soil loss from recreational trails. J. Environ. Manag. 90, 1483–1493. https://doi.org/10.1016/j.jenvman.2008.10.004.
Park, L.O., Manning, R.E., Marion, J.L., Lawson, S.R., Jacobi, C., 2008. Managing visitor impacts in Parks: a multi-method study of the effectiveness of alternative management Practices. J. Park Recreat. Adm. 26, 97–121.
Patel, A., 2000. Assessing the impacts of deer browsing, prescribed burns, visitor use, and trails on an oak-pine forest: Pinery Provincial Park, Ontario. Canada. Nat. Areas J. 20, 250–260.
Potito, A.P., Beatty, S.W., 2005. Impacts of recreation trails on exotic and ruderal species distribution in grassland areas along the Colorado Front Range. Environ. Mutagen. 36, 230–236. https://doi.org/10.1007/s00267-003-0177-0.
R Core Team, 2014. R: a Language and Environment for Statistical Computing.
Rickard, C.A., McLachlan, A., Kerley, G.I.H., 1994. The effects of vehicular and pedestrian traffic on dune vegetation in South Africa. Ocean Coast Manag. 23, 225–247. https://doi.org/10.1016/0964-5691(94)90021-3.
Roovers, P., Baeten, S., Hermy, M., 2004a. Plant species variation across path ecotones in a variety of common vegetation types. Plant Ecol. 170, 107–119. https://doi.org/10.1023/B: VEGE.0000019028.89623.26.
Roovers, P., Verheyen, K., Hermy, M., Gulinck, H., 2004b. Experimental trampling and vegetation recovery in some forest and heathland communities. Appl. Veg. Sci. 7, 111–118. https://doi.org/10.1658/1402-2001(2004)007[]0111:ETAVRI]2.0.CO;2.
SAS Institute Inc, 2012. SAS/STAT® 9.3 User's Guide [WWW Document]. SAS/STAT® 9.3 User's Guid. https://support.sas.com/documentation/cdl/en/statug/63962/HTML/default/viewer.htm#statug_glimmix_sect028.htm. (Accessed 6 June 2017).
Schimel, D.S., Kelly, E.F., Yonker, C., Aguilar, R., Heil, R.D., 1985. Effects of erosional processes on nutrient cycling in semiarid landscapes. In: Caldwell, D.E., Brierley, J.A., Brierley, C.L. (Eds.), Planetary Ecology. Van Nostrand Reinhold, New York.
Skotnicki, S.J., 2016. Geology of the McDowell Mountains. Arizona Geological Survey Contributed Map CM-16-A, 1 map plate. http://tinyurl.com/GeoMap-McdowellMtns.
Tyser, R.W., Worley, C.A., 1992. Alien flora in grasslands to road and trail adjacent in glacier National corridors (U. S. A.). Conserv. Biol. 6, 253–262. https://doi.org/10.1007/BF00410220.
USDA Natural Resource Conservation Service, 2017. Web Soil Survey [WWW Document]. https://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm. (Accessed 1 January 2014).
Webb, R.H., 1983. Compaction of desert soils by off-Road ve. Environ. Eff. off-road Veh. Impacts Manag. Arid Reg. 51–79.
Whinam, J., Chilcott, N., 1999. Impacts of trampling on alpine environments in central Tasmania. J. Environ. Manag. 57, 205–220. https://doi.org/10.1006/jema.1999.0302.
Whinam, J., Chilcott, N.M., 2003. Impacts after four years of experimental trampling on alpine/sub-alpine environments in western Tasmania. J. Environ. Manag. 67, 339–351. https://doi.org/10.1016/S0301-4797(02)00218-9.
White, E.M., Bowker, J.M., Askew, A.E., Langner, L.L., Arnold, J.R., English, B.K., 2014. Federal Outdoor Recreation Trends: Effects on Economic Opportunities October 2014.
Wickham, H., 2009. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag, New York.
Williams, J.D., Dobrowolski, J.P., West, N.E., Gillette, D.A., 1995. Microphytic crust influences on wind erosion. Trans. Am. Soc. Agric. Eng. 38, 131–137.
Wimpey, J.F., Marion, J.L., 2010. The influence of use, environmental and managerial factors on the width of recreational trails. J. Environ. Manag. 91, 2028–2037. https://doi.org/10.1016/j.jenvman.2010.05.017.