Plant Ecology & Diversity, 2015
Vol. 8, Issue 2, 265–276, http://dx.doi.org/10.1080/17550874.2014.893592

Impacts of experimental trampling by hikers and pack animals on a high-altitude alpine sedge meadow in the Andes

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(*Received 1 June 2012; final version received 8 February 2014*)

Background: Damage to alpine plant communities is likely to occur when hikers and pack animals trample vegetation. Currently, there is limited research that quantifies and compares impacts from these activities.
Aims: A manipulative experimental protocol was used to assess damage to alpine meadows by pack animals and hikers in the Aconcagua Provincial Park, Andes, Argentina.
Methods: Vegetation height, overall cover, cover of dominant species and species richness were measured immediately after, and 2 weeks after different numbers of passes (0, 25, 100 and 300) by hikers or pack animals in an experiment, using a randomised block design.
Results: Pack animals had two to three times the impact of hiking on the meadows, with greater reductions in plant height, the cover of one of the dominant sedges and declines in overall vegetation cover after 300 passes. Impacts of pack animals were also apparent at lower levels of use than for hikers. These differences occurred despite the meadow community having relatively high resistance to trampling due to the traits of one of the dominant sedges (*Carex gayana*).
Conclusions: Pack animals caused more damage than hikers to the alpine meadow, but the scale of the difference in short-term impacts depends on the characteristics of the plant community, the amount of use and the vegetation parameters measured. Use of the meadows by hikers and pack animals should be minimised as these meadows are scarce, and have high conservation values.

Keywords: Aconcagua; Andes; alpine sedge meadow; horses; mules; recreation ecology; trampling

Introduction

Mountain protected areas are often iconic destinations for adventure tourists when they include the highest summits of continents, countries and regions (Buckley 2006). To access destinations most tourists hike, but many use commercial operators to transport their equipment on pack animals (McClaran and Cole 1993; Geneletti and Dawa 2009). Pack animals are often used where road access is limited, including in mountains in the Himalayas (Gurung and Seeland 2008; Geneletti and Dawa 2009), the Rocky Mountains (Cole et al. 2004) and the Andes (Byers 2009; Barros et al. 2013).

Both hikers and pack animals can damage vegetation, some with high conservation value and limited distribution. This includes alpine meadows, bogs and shrub-dominated communities which can be very sensitive to trampling and are often slow to recover once damaged (McClaran and Cole 1993; McDougall and Wright 2004; Leung et al. 2011). Impacts of hikers and pack animals include soil erosion and compaction, loss of sensitive species, increases in resistant species, and overall reductions in species richness and total vegetation cover (Whinam et al. 1994; Monz 2002; Cole 2004; Hill and Pickering 2006; Scherrer and Pickering 2006; Nepal and Way 2007; Pickering et al. 2010). The severity of impacts varies among plant communities and among different species depending on their traits (Liddle 1997; Yorks et al. 1997; Whinam and Chilcott 2003; Talora et al. 2007; Hill and Pickering 2009; Törn et al. 2009). For example, plant traits, such as vegetation stature and growth form affect resistance and resilience to trampling. Some graminoids, such as sedges, are more resistant than many shrubs, forbs and ferns because they have below-ground reproductive structures, often narrow leaves and flexible stems (Kuss 1986; Sun and Liddle 1993; Cole 1995; Liddle 1997; Yorks et al. 1997; Hill and Pickering 2009).

Impacts differ in severity among recreational activities, but few studies have quantified the scale of the differences, including comparing those of pack animals, such as horses and mules, with hiking (Törn et al. 2009; Pickering et al. 2010). To date there appear to be only four manipulative experimental studies and all were conducted in the Rocky Mountains of the western United States. They assessed the impacts of pack animals on the height and cover of natural vegetation (Weaver and Dale 1978; Cole and Spildie 1998), or trail impacts including soil erosion and sediment yield (Weaver and Dale 1978; Wilson and Seney 1994; DeLuca et al. 1998). None assessed the impacts of trampling on plant composition or the cover of dominant species. All four found that horses did more harm at lower levels of use than hiking (Weaver and Dale 1978; Wilson and Seney 1994; Cole and Spildie 1998; DeLuca et al. 1998).

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Although these results are expected, management decisions regarding the appropriateness of different recreational activities remain contentious (Phillips and Newsome 2002; Newsome et al. 2008). Issues raised include the narrow geographic spread of much of the research in recreation ecology (Buckley 2005; Pickering and Hill 2007; Hill and Pickering 2009; Monz et al. 2010), including research on the impacts of horses (Newsome et al. 2008; Pickering et al. 2010). Therefore, research in other regions less studied, with different climatic conditions, evolutionary histories and patterns of human use, are important as they may indicate when generalisations based on research from one region applies more broadly, or when local factors maybe more important.

There is limited recreation ecology research from the high Andes of South America (Buckley 2005). Research on visitor impacts in this region is important due to its high conservation value and popularity as a tourist destination (Price 1992; Byers 2009, 2011; Barros et al. 2013). It is also important in terms of testing the generality of assumptions based on existing research in recreation ecology, including the impacts of different types of trampling on vegetation. The Andean flora evolved in the absence of large hard-hoofed animals, such as horses and bovids, and may, therefore, be more vulnerable to intensive trampling by these types of animals compared with native grazing mammals, such as the camelids *Lama guanicoe* and *Vicugna vicugna* (Molinillo and Monasterio 2006; Villalobos and Zalba 2010). Currently there appear to be only five recreational ecology scientific papers from the whole of the Andes, all of which assessed trail impacts and none of which compared impacts between hikers and pack animals, either on or off trails (Hoffman and Alliende 1982; Marion and Linville 2000; Farrell and Marion 2002; Byers 2009; Barros et al. 2013).

We quantified the relative impacts of hikers and pack animals using a manipulative experiment. This experiment was conducted on an alpine sedge meadow in the Aconcagua Provincial Park in Argentina. The Park protects the region around the highest mountain in the Andes, Mt. Aconcagua (6962 m a.s.l.), which is an increasingly popular destination for mountaineers and trekkers from Europe and North America (Dirección de Recursos Naturales 2009; Barros et al. 2013). Although these alpine meadows account for less than 0.4% of area of the Park (Zalazar et al. 2007), they play key role in sustaining rare and endemic flora and fauna including ground-nesting birds (Squeo et al. 2006; Olivera and Lardelli 2009; Barros et al. in press).

The specific aims of the research were to quantify the impacts of trampling by (1) pack animals and (2) hikers on an alpine sedge meadow, and (3) to compare the effects of these two activities for slight (25 passes), moderate (100 passes), and high usage (300 passes).We hypothesised that (1) pack animals would do more damage than hikers at an equivalent number of passes; (2) the sensitivity of the vegetation to trampling would vary based on the vegetation parameter measured; and (3) the impacts of trampling would vary among plant species within the meadow.

Materials and methods

Study area

Aconcagua Provincial Park (69°50’ W, 32°39’ S) in the dry central Andes of Argentina conserves 710 km^2 of glaciers, watersheds, alpine ecosystems and archaeological sites around Mt Aconcagua. Vegetation cover is scarce (30%) due to the harsh climate conditions and geomorphology, with alpine meadows restricted to valley floors between 2400–3800 m a.s.l. (Mendez et al. 2006; Barros et al. 2013). The meadows are local biodiversity hotspots, providing habitat for over 40 bird species including ground-nesting birds (Olivera and Lardelli 2009) and clean water for lowland areas (Dirección de Recursos Naturales 2009).

Prior to being designated as a Category II International Union for the Conservation of Nature (IUCN) Park in 1983, there was limited human use of the region, including transient use by indigenous communities for ceremonies (Barcena 1998), some military training in the mid-1900s (Quiroga 1996) and a few mountain expeditions (Dirección de Recursos Naturales 2009). Since 1980 the Park has been used for sightseeing, mountaineering and trekking (Dirección de Recursos Naturales 2009). In the 2010–2011 season around 27,000 sightseers, 6000 hikers and mountaineers and 5000 pack animals (mules and horses) traversed alpine meadows when accessing view points and campsites (Dirección de Recursos Naturales 2011). No other human use is permitted in the Park other than pack animals (Dirección de Recursos Naturales 2009).

Study site

Experimental trampling by hikers and pack animals with riders was undertaken on a fenced alpine sedge meadow 2 km from the end of the main access road to the Park in the Horcones Valley (69°56’30”S, 32°48’40.7”W, 2949 m a.s.l.) (Figure 1). This meadow was dominated by a *Carex gayana* plant community, growing on sandy loam soils with high organic matter content (16 ± 3%) (Barros 2014). The sedge *C. gayana* is associated with the regional endemic sedge *Eleocharis pseudoalbibracteata*, forming dense stands on depressed riverine landforms in alpine meadows (Mendez et al. 2006). It is characteristic of alpine meadows in the Southern Steppe Ecoregion in South America (Gandullo and Faggi 2005; Squeo et al. 2006; Mendez 2007). Both species are perennial rhizomatous geophytes (Kiesling 2009). *Carex gayana* is characterised by tough, broad leaves with plants up to 65 cm in height, while *E. pseudoalbibracteata* has erect thin leaves with plants up to 17 cm in height (Kiesling 2009).

The fence that protects this meadow (1 ha) was established in 2003, and since then the meadow has only experienced minor grazing and trampling due to occasional gaps in the fence. The experiment was conducted during the flowering and growing season for plants in the warmest time of year in February and March 2011 (Arroyo et al. 1981; Departamento General de Irrigación 2011)

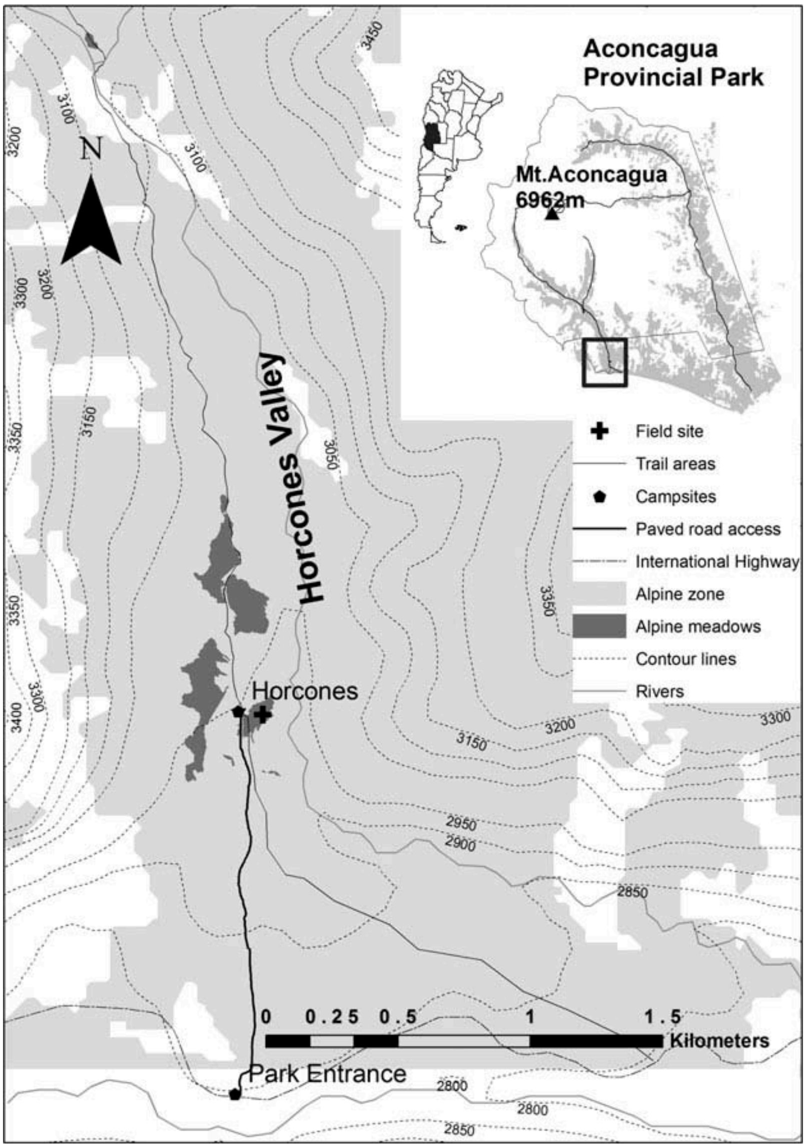

Figure 1. Location of the field site (69° 56′ 30″ W, 32° 48′ 40″ S) where the experimental trampling was conducted on an alpine sedge meadow in Aconcagua Provincial Park, Mendoza, Argentina.

when tourism use of the area was high (Dirección de Recursos Naturales 2009). When the experiment was conducted soils were dry, possibly due to low snowfall over the previous winter (Departamento General de Irrigación 2011).

Experimental design and sampling

Experimental trampling by hikers and pack animals (a horse and a mule each with a rider) was applied to the alpine meadow using a modification of the Cole and Bayfield (1993) methodology to quantify impacts of different recreation activities. The basic design was a randomised block with seven treatments applied to each of six replicate blocks (transects). At the site, six 40 m long and 0.50 m wide parallel replicate transects (e.g. blocks) were marked out in a flat area of meadow with each transect 3 m apart. Transects were each divided into seven lanes each 4 m long and separated by 2 m. The seven experimental treatments were randomly assigned to the lanes. These were: control (no trampling), 25, 100 and 300 passes by horse/mule riders and 25, 100 and 300 passes by hikers (Figure 2). Each pass consisted of a one-way ride/walk at a natural gait. The highest number of passes applied represents the number of passes by pack animals and hikers per day during peak usage of this area of the Park (Dirección de Recursos Naturales 2009). Previous studies on alpine plant communities have found that this level of use can cause significant damage, including to vegetation cover to a range of plant communities (Bell and Bliss 1973; Cole 1995; Monz 2002). There were three hikers and two riders. The hikers and riders averaged 70 kg, the mule weighed 260 kg and the horse weighed around 300 kg (including saddle). Mules and horses had shoes. All trampling and riding was undertaken at the same time.

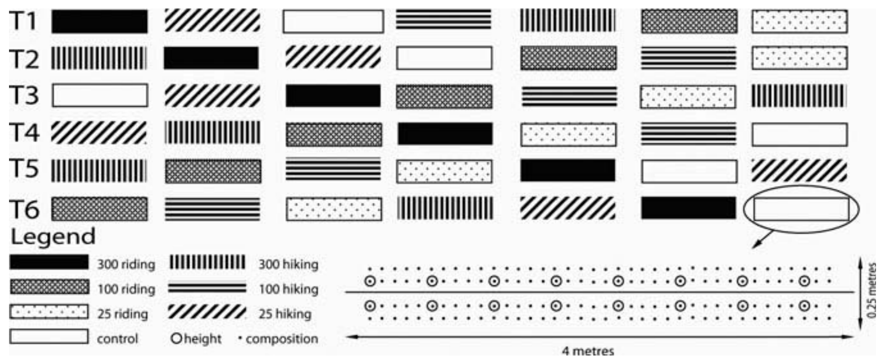

Figure 2. Experimental layout used to evaluate the relative impacts of hiking and riding pack animals on an alpine meadow in Aconcagua Provincial Park, Argentina. There were six parallel 40 m transects (T1 to T6) separated by 3 m. Seven experimental treatments (control, 25, 100 and 300 passes by a rider on a mule/horse, and 25, 100 and 300 passes by a hiker) were applied to lanes using a stratified random design. The expanded lane shown in the bottom right hand side of the Figure indicates the location of the 16 points used to measure vegetation height, and the 152 points used to record the cover of each species.

At the start of the experiment, vegetation height, cover, species richness and composition were recorded in the seven control lanes in a rectangular quadrat of 4 m × 0.25 m (e.g. providing species richness per 1 m^2). Vegetation height was measured just after trampling and 2 weeks later in all lanes. Vegetation height was measured at 16 evenly spaced points along the middle section of each lane (e.g. 5 cm either side of the mid line of the lane, every 50 cm) (Figure 2). Height was the maximum height of vegetation at each point in centimetres. Two weeks after the treatments were applied species richness, composition and cover were recorded in all lanes.

The cover of each species was assessed in the lanes using 152 evenly spaced points (5 cm and 10 cm either side of the centre of the lane every 10 cm, starting at 20 cm) (Figure 2) using a 2 mm diameter metal rod held vertically. The number of hits touched by each species and litter (dead plant material) underneath living vegetation was recorded. Multiple hits per point were possible where different species touched the rod. These data were used to calculate the cover of each species by dividing the number of hits for that species by the 152 points. The same 152 points were used to assess projected cover, but in this case, only one record was made per point: that is, if the tallest strata to touch the rod was vegetation or it was uncovered litter. No rock or bare soil was found in any lane. To obtain a complete species list for the lane, the rectangular quadrat was searched for any additional species present but not 'hit' by one of the points. These species were given an arbitrary low cover value of 0.5%. From these data and the species cover data, the total number of species per 1 m^2 area was calculated for each lane.

Plant biomass was measured using dry weights obtained by cutting all living vegetation (still green) at ground level in one 40 cm × 10 cm quadrat positioned in the centre of each lane at the end of the experiment. The biomass of litter (dead plant material) was measured using the dry weight of all litter that was collected in each lane prior to cutting the vegetation in the quadrat of each lane. Samples were bagged and dried in an oven at 75 °C for 48 h and weighed.

Statistical analysis

The effects of riding and hiking were compared using a series of one-way randomised complete block ANOVAs in the SPSS 17.0 statistical program. Dependent variables analysed included: vegetation height immediately after trampling and two weeks later; vegetation cover, cover of *E. pseudoalbibracteata, C. gayana* and litter, species richness and plant and litter biomass 2 weeks after treatment. Treatment was included as a fixed factor and transect as a random factor. There were no significant differences among transects for any of the variables tested. The assumptions of homogeneity of variance were tested using Levene's test.

To compare differences between treatments on vegetation height immediately after trampling and 2 weeks later, the non-parametric Kruskal–Wallis test was used with the Mann–Whitney U test used to compare pairs of variables. All cover values and the proportion of biomass from living plants were arcsine square root transformed prior to analysis to satisfy the assumptions of the tests. Post hoc tests were carried out by using Tukey's test. This is a widely accepted conservative test for comparing multiple pairwise comparisons on data collected from the same experiment (Underwood 1997). In addition to the ANOVAs, paired t-tests were used to compare the impacts of pack animals and hiking for the same number of passes.

A resistance index was calculated for vegetation height to determine the number of passes required to cause 50% reduction of its original value (Liddle 1997). This was done by visually assessing the graph of mean height of vegetation vs. number of passes. For vegetation cover it was not possible to calculate the resistance index as even the maximum number of passes applied in this study resulted in less than a 50% reduction in vegetation cover.

Vegetation parameters were not converted into relative values as has often been done when using the Cole and

Bayfield (1993) methodology. This was because absolute rather than relative values for this type of data provide more statistically reliable results when directly testing for differences among activities and intensities of use using ANOVA (Pickering et al. 2011). If relative values are used in the ANOVA, the values for the control lanes are all 100%. As a result, comparing controls with treatment lanes using ANOVAs violates two assumptions of the statistical test: homogeneity of variance and normality of the data (Underwood 1997). We confirmed that there were no significant changes in the control lanes over the 2 weeks between the initial treatments and when changes in vegetation were measured 2 weeks later, using a paired *t*-test. Finally, treatments had been randomly allocated to lanes to ensure that any differences found using statistical tests could be attributed to the effect of the treatments and not to the inherent differences among lanes (Underwood 1997; Quinn and Keough 2003).

Results

The untrampled alpine meadow had complete vegetation cover (100%), which consisted almost entirely of the two native sedges: *C. gayana* (99%) and *E. pseudoalbibracteata* (78%) (Table 1). These sedges species were found in all control and trampled lanes. Nine other species were recorded across the control and trampled lanes, but they were not common and had low cover (Table 1). They were a mixture of geophytes, hemicryptophytes and one chamaephyte (Table 1). The average dry biomass of vegetation in the untrampled meadow was 677 ± 56 g m^{-2} with 61± 32 g m^{-2} of litter (Table 2).

Riding and hiking had a range of impacts on the alpine sedge meadow, with significant changes in vegetation height, vegetation cover and the cover of the sedge *E. pseudoalbibracteata*. In addition, riding had significant impacts on the cover of *C. gayana*, litter and the proportion of living plant biomass. As could be expected, plant species richness was the only parameter unaffected by either activity, with an average of four species per m^2 (Tables 2 and 3).

Vegetation height was the most sensitive parameter to trampling, being affected by both activities immediately after 25 passes and 2 weeks later (Table 3). For example, after 2 weeks the reductions in vegetation height were significantly greater in lanes ridden on (16 cm) than lanes trampled by hikers (23 cm) compared with the control lanes (29 cm) after 25 passes (Figure 3, Appendix 1). The resistance index for vegetation height (e.g. the number of passes required to reduce initial vegetation height by 50%) 2 weeks after trampling by a horse/mule was 100 passes and 300 passes by hikers (Figure 3).

Vegetation cover was affected by both activities, but only after 300 passes. It reduced from 100% in control lanes to 84% in lanes ridden on and 94% in lanes trampled by hikers (Table 3 and Figure 4, Appendix 2). Trampling by both activities also resulted in declines in the cover of both dominant species. The sedge *C. gayana* had a cover of 99% in control lanes, but only 84% in lanes after 300 passes by pack animals (Table 3, Appendix 2) and 94% after 300 passes by a hiker (Figure 5, Appendix 1). The other dominant sedge, *E. pseudoalbibracteata*, was less resistant to trampling, with decreased cover after 100 passes by pack animals and 300 passes by hikers (Table 3, Appendix 2). Its cover was 78% in control lanes, but only 31% after 100 passes by pack animals and 27% after 300 passes by a hiker (Figure 5, Appendix 2).

Trampling damage to the vegetation from pack animals resulted in significant increases in the amount of litter 2 weeks post use. The cover of litter went from 41% in controls to 90% after 300 passes by riders, while the biomass of litter went from 61 g m^{-2} in controls to 258 g m^{-2} (Table 2 and Figure 5, Appendix 2). Consequently, the amount and the proportion of living plant biomass was lower after trampling by riders compared with controls (Table 3, Appendix 2). In contrast, hiking did not affect the cover of litter or the proportion of living plant biomass (Figure 5, Appendix 2).

Discussion

This study found that, as predicted, pack animals did more damage to the alpine sedge meadow than hiking. Trampling by pack animals resulted in around two to three times as much damage to vegetation as that caused by hikers, depending on the vegetation parameter measured after 300 passes. Pack animals also caused damage at fewer passes than hikers, again, reflecting their greater impact on the meadow. These differences between activities occurred despite the apparent relatively resistance of the meadow to trampling when assessed using changes in the overall cover of vegetation or the dominant sedge *Carex gayana*.

Differences in the severity of impacts concur with the results from the only other two studies (Weaver and Dale 1978; Cole and Spildie 1998) that compared the relative impacts of hikers and horses on natural vegetation. All three studies found that pack animals did more damage than hikers after the same number of passes, but that differences in the size of their effects on vegetation cover were only apparent after high use for some plant communities. For example, differences in vegetation cover between the two activities were only apparent after 150 passes in vegetation dominated by the shrub *Vaccinium scoparium*, while differences were apparent after only 25 passes in a forb-dominated understorey in the montane zone of the Rocky Mountains of the USA (Cole and Spildie 1998). It appears that the slope and shape of the relationship between increasing use (number of passes) and damage to vegetation can vary between these two activities (Weaver and Dale 1978). Consequently, the size of any difference in impacts between hiking and horse riding depends on the type of vegetation, the vegetation parameter measured and the intensity of use.

The resistance to trampling of the alpine sedge meadow in Aconcagua is likely to reflect the morphological traits of

Table 1. Mean (± SE) cover of plants and litter 2 weeks after experimental trampling in an alpine meadow in the Aconcagua Provincial Park, Argentina.

Species	Family	GF	LF	Control	FC	25h	100h	300h	25r	100r	300r	FT
Carex gayana	Cyperaceae	S	G	98.8 ± 0.5	6	99.3 ± 0.3	97.9 ± 0.8	94.2 ± 2.2	89.4 ± 7.3	97.1 ± 0.9	83.8 ± 2.0	36
Eleocharis pseudoalbibracteata	Cyperaceae	S	G	77.8 ± 8.7	6	47.9 ± 10.8	48.1 ± 10.0	27.4 ± 11.4	73.7 ± 7.3	31.0 ± 7.5	9.2 ± 2.8	36
Hypsela reniformis	Campanulaceae	H	G	3.5 ± 2.0	4	0.5 ± 0.4	2.7 ± 2.4	0.2 ± 0.1	0.6 ± 0.3	0.7 ± 0.5		12
Plantago barbata	Plantaginaceae	H	H	1.6 ± 1.3	3	0.6 ± 0.5		1.7 ± 1.7	9.1 ± 8.1	1.4 ± 1.4		6
Poa subenervis var. *spegazziniana*	Poaceae	G	H	1.3 ± 0.7	3	0.7 ± 0.6	1.5 ± 1.0		5.8 ± 1.7	0.9 ± 0.6	0.8 ± 0.6	16
Taraxacum officinale	Asteraceae	H	H	0.3 ± 0.1	4	0.2 ± 0.1		0.2 ± 0.2			0.1 ± 0.1	4
Werneria pygmaea	Asteraceae	H	C						1.3 ± 1.3	<1	<1	3
Hordeum comosum	Poaceae	G	H	0.4 ± 0.3	2	<1		0.6 ± 0.6	1.2 ± 1.1			4
Deschampsia caespitose var. *caespitose*	Poaceae	G	H			<1			<1			2
Juncus balticus var. *andicola*	Juncaceae	R	G						0.6 ± 0.6			1
Triglochin palustris	Juncaginaceae	H	H						<1			1
Unknown bryophyte	Bryophyte	B									<1	1
Litter				41.6 ± 12.6	6	63.7 ± 9.0	63.5 ± 8.5	74.2 ± 7.7	42.1 ± 11.7	73.1 ± 3.2	90.9 ± 1.3	36

Control, no trampling; 25, 25 passes; 100, 100 passes; 300, 300 passes; h, hikers; r, riders in a horse or a mule. GF, growth form; S, sedge; H, herb; G, grass; R, rush; B, bryophyte; LF, life form (C, chamaephyte; G, geophyte; H, hemicryptophyte) (Raunkiaer 1934). FC, frequency in control lanes; FT, frequency in treated lanes. Six replicates were conducted per treatment.

Table 2. Mean (± SE) plant and litter biomass and species richness in control lanes and lanes subject to different number of passes by hikers and riders 2 weeks after experimental trampling in an alpine meadow in the Aconcagua Provincial Park, Argentina.

	Control	25h	100h	300h	25r	100r	300r
Biomass (g m^{-2})							
Plant	677 ± 56	566 ± 91	540 ± 38	437 ± 96	620 ± 127	650 ± 90	588 ± 113
Litter	61 ± 32	86 ± 27	82 ± 28	115 ± 30	41 ± 13	105 ± 27	258 ± 66
No. species m^{-2}	4.7 ± 1	3.7 ± 1	2.8 ± 0.4	2.8 ± 0.5	4.8 ± 0.7	3.2 ± 0.7	2.8 ± 0.5

Control, no trampling; 25, 25 passes; 100, 100 passes; 300, 300 passes; h, hikers; r, riders in a horse or a mule. Six replicates were conducted per treatment.

Table 3. Results from one-way ANOVAs (complete randomised block) comparing treatments (controls and different intensities of riding and hiking) and transects for vegetation parameters.

	Treatment		Transect	
	H	*P*	H	*P*
Immediately after trampling				
Vegetation height*	25.014	**<0.001**	2.728	0.742
	F	*P*	F	*P*
Two weeks after trampling				
Vegetation height	28.935	**<0.001**	0.805	0.550
Vegetation cover	17.558	**<0.001**	0.331	0.890
Carex gayana	5.667	**<0.001**	0.627	0.681
Eleocharis pseudoalbibracteata	6.678	**<0.001**	0.069	0.996
Litter	5.237	**0.001**	1.994	0.108
Number of species per m^2	1.667	0.164	1.024	0.421
Proportion of biomass from plants	3.765	**0.007**	0.160	0.975

*For vegetation height after experimental treatments, the non-parametric Kruskal–Wallis test was used. Values in bold are significant at α = 0.05.

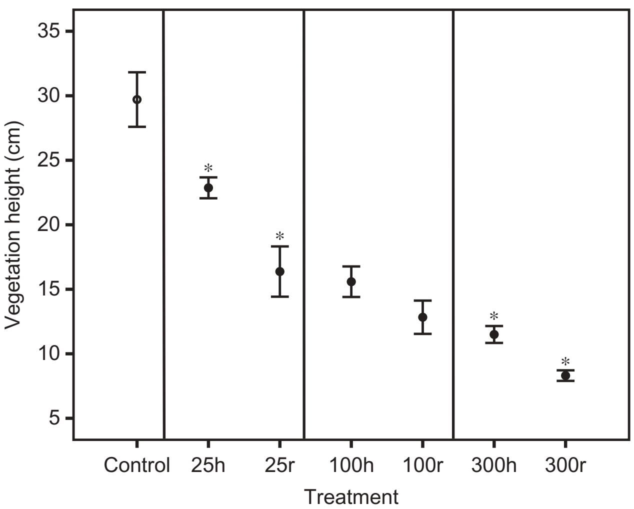

Figure 3. Mean (± SE) vegetation height in cm 2 weeks after experimental trampling of controls, hiking (h), riding a horse or a mule (r) for 25, 100 and 300 passes on an alpine meadow in the Aconcagua Provincial Park, Argentina. Significant differences ($P < 0.05$) between hikers and riders at equivalent number of passes are indicated with an asterisk.

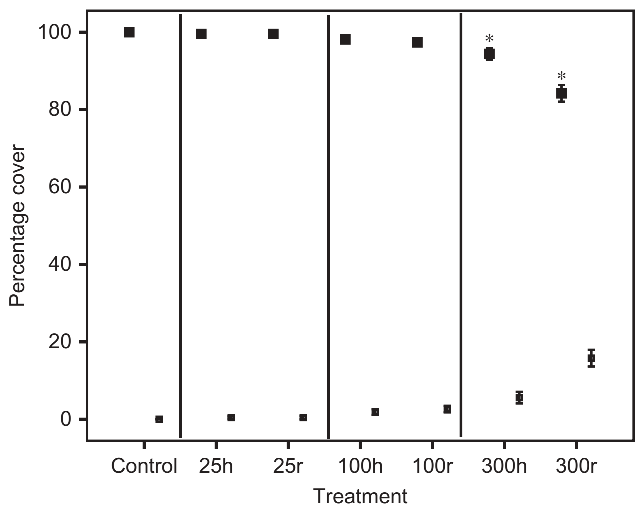

Figure 4. Mean (± SE) percentage cover of vegetation (solid squares) and litter (clear squares) 2 weeks after experimental trampling of controls, hiking (h), riding a horse or a mule (r) for 25, 100 and 300 passes on an alpine meadow in Aconcagua Provincial Park, Argentina. Significant differences ($P < 0.05$) between hikers and riders at equivalent number of passes for vegetation parameters are indicated with an asterisk.

the dominant sedge *C. gayana.* It is taller, and has broad tough leaves that can more easily be flattened rather than broken. In contrast, the other dominant sedge, *E. pseudoalbibracteata,* has erect thin leaves and was less resistant to trampling. Variation in resistance has also been found in other plant communities where the traits of dominant species, including their leaf structure, strongly influence overall

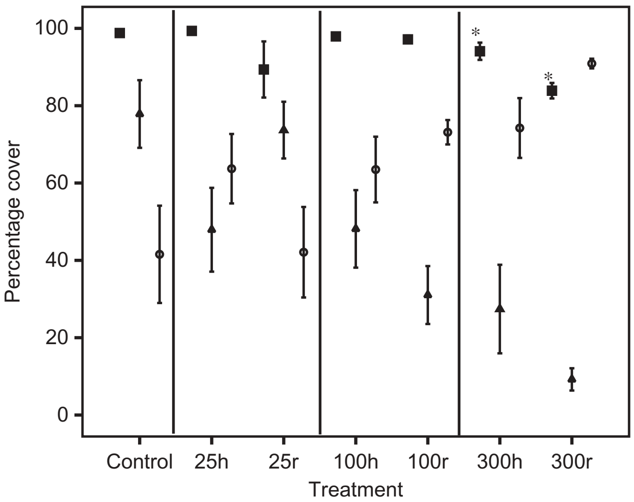

Figure 5. Mean (± SE) percentage cover of (a) *Carex gayana* (solid squares), (b) *Eleocharis pseudoalbibracteata* (solid triangles) and (c) litter (clear circles) 2 weeks after experimental trampling of controls, hiking (h), riding a horse or a mule (r) for 25, 100 and 300 passes on an alpine meadow in Aconcagua Provincial Park, Argentina. Significant differences (P < 0.05) between hikers and riders at equivalent number of passes for vegetation parameters are indicated with an asterisk.

trampling resistance (Bernhardt-Romermann et al. 2011). For example, graminoid-dominated communities are often more resistant to trampling than herb-dominated communities, mainly due to differences in morphological characteristics (Weaver and Dale 1978; Kuss 1986; Cole and Bayfield 1993; Cole 1995; Hill and Pickering 2009; Striker et al. 2011), and some species of *Carex* appear to be particularly resistant to trampling (Bayfield 1979; Grabherr 1982; Cole 1995).

The high resistance of the meadow in Aconcagua to trampling may also have been due to the drier soils during the experiment, as the moisture content of soils can affect the severity of disturbance (Kuss 1986; Liddle 1997; Yorks et al. 1997). Drier soils can be less susceptible to penetration by hikers' boots and horses' and mules' hooves, resulting in less damage to underground plant organs when conditions are drier (Willard and Marr 1970; Hamza and Anderson 2005). For example, experimental studies have found that some species of plants on drier soils were more resistant to trampling than the same species on wetter soils (Leney 1974). It is possible that in wetter years even the apparently quite resistant sedge *C. gayana* in the alpine meadow could be more easily damaged by trampling by hikers and pack animals.

The variation among vegetation parameters in the severity of impacts in Aconcagua highlights the importance of using a range of parameters when assessing impacts of visitor activities. Parameters, such as vegetation height, biomass, cover and composition vary in their sensitivity to trampling, and so it is possible to miss ecologically important changes in plant communities if only less sensitive parameters are used (Growcock 2005; Monz et al. 2013). For example, reductions in vegetation height from trampling are likely to be ecologically important in alpine meadows in Aconcagua as they are used for ground-nesting birds (Olivera and Lardelli 2009; Barros et al. 2013).

As the study only looked at resistance, and not at resilience (recovery), it is not possible to assess the tolerance of this Andean alpine sedge meadow to trampling. High resistance does not automatically indicate high resilience, as has been found in several studies (Cole 1995; Liddle 1997). For example, some studies have indicated that productive and wet sites, such as many alpine meadows that have low resistances, can recover relatively quickly from disturbance (Yorks et al. 1997; Byers 2011). In contrast, alpine sedge meadows with slow-growing underground rhizomes might have high resistance, but recovery could be very slow and even take decades (Grabherr 1982; Ortuño et al. 2006).

Given that this study found that even relatively low levels of use by hikers and pack animals resulted in damage to alpine meadows, it is likely that the current number of visitors (>33,000 visitors) and pack animals (5000 pack animals) traversing these meadows annually are above the disturbance threshold for this plant community. Due to the often unregulated use of many of the meadows in the Park, an extensive network of 'informal' trails has already been created by hikers and pack animals as they select their own routes through the meadows (Barros et al. 2013; Barros 2014). Damage to alpine meadows in this and other Parks in the dry Andes is of particular concern, because while the meadows are of limited distribution (ca. 5% of the total vegetation in the region, Buono et al. 2010), they provide key habitat for wildlife (Squeo et al. 2006), and play an important role in carbon sequestration and water regulation in this dry environments (Otto et al. 2011). Human use of the alpine meadows in this region should only be in ways that maintain their conservation value and the ecosystem services they provide. For the Aconcagua Park, management strategies to minimise damage could include the provision of clearly defined official/hardened trails that avoid meadows whenever possible, and improved regulations controlling pack animals in the Park. Where trails need to traverse alpine meadows, they should be designed so that they are elevated from the ground to avoid trampling damage. The results also highlight the importance of managing increased visitation to all high-altitude and remote parks, especially where pack animals are increasingly used to transport equipment.

Conclusions

Even low trampling rates by hikers and pack animals produced measurable damage in meadow vegetation, with impacts two to three times greater for pack animals than for hikers. Thus, trails should avoid meadows and pack animals must be completely excluded from alpine meadows. When necessary for scientific, management or exploratory reasons, hiking through meadows may be tolerated but only at very low rates.

Acknowledgments

This research was funded by Griffith University. We thank the park rangers and officers at Dirección de Recursos Naturales of Mendoza province who assisted one of the authors in the field and provided internal reports. We thank Sebastián Rossi, Jesus Lucero, Ruben Massarelli, Santiago Echevarría, Diego Marquez, Juan Perez, Juan Emilio Gamboa, Laura Sorli, Pablo Sampano, Rudy Parra and Carlos Salas for their help in the field, Roberto Kiesling and Jorge Gonnet for identifying plant samples, and the consultant statistician Michael Arthur for his advice regarding the analysis of the data. Thanks to Wendy Hill, Clare Morrison and two anonymous reviewers for their valuable contributions to improve the manuscript.

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Appendices

Appendix 1. Results from paired *t*-tests comparing hiking and riding at 25, 100 and 300 passes. Values in bold are significant at α = 0.05.

	25 hike vs 25 ride		100 hike vs 100 ride		300 hike vs 300 ride	
	t	*P*	t	*P*	t	*P*
Height 2 weeks after trampling	2.911	**0.033**	1.268	0.261	4.174	**0.009**
# of species	−0.914	0.402	−0.042	0.695	0	1.000
Biomass						
Litter	1.334	0.240	−0.578	0.588	−2.474	0.056
Plant	−0.317	0.764	−1.341	0.237	−0.765	0.479
Proportion plants	−0.939	0.391	0.454	0.669	0.939	0.391
Cover						
Vegetation	−0.236	0.823	0.595	0.578	2.974	**0.031**
Litter	1.356	0.233	−1.142	0.305	−2.446	0.058
Carex gayana	2.066	0.940	0.653	0.542	2.640	**0.046**
Eleocharis pseudoalbibracteata	−1.472	0.213	2.001	0.102	1.417	0.216

Appendix 2. Tukey post hoc test from One-way Randomised Block ANOVA for vegetation parameters in control lanes and lanes with different number of passes by hikers (h) and riders (r). * For vegetation height immediately after trampling the non-parametric Mann–Whitney *U* test was used. Values in bold are significant at α = 0.05 as this test is conservative and already adjusts for multiple pair-wise comparisons.

	Control	25h	100h	300h	25r	100r	300r
Height immediately after* (top diagonal) and 2 weeks after trampling (bottom diagonal)							
Control		**0.041**	**0.002**	**0.004**	**0.015**	**0.004**	**0.002**
25h	**0.019**		**0.009**	**0.009**	0.065	**0.015**	**0.002**
100h	**<0.001**	**0.011**		0.394	0.818	0.937	**0.015**
300h	**<0.001**	**<0.001**	0.364		0.485	0.749	0.093
25r	**<0.001**	**0.030**	1.000	0.968		0.589	0.180
100r	**<0.001**	**<0.001**	0.781	0.999	0.532		0.065
300r	**<0.001**	**<0.001**	**0.011**	0.648	0.004	0.253	
Vegetation cover (top diagonal) and cover of litter (bottom diagonal)							
Control		0.985	0.274	**0.002**	0.997	0.066	**<0.001**
25h	0.499		0.727	**0.013**	1.000	0.298	**<0.001**
100h	0.527	1.000		0.324	0.600	0.990	**<0.001**
300h	0.098	0.960	0.950		**0.007**	0.757	**0.006**
25r	1.000	0.444	0.471	0.081		0.209	**<0.001**
100r	0.154	0.989	0.985	1.000	0.128		**<0.001**
300r	**0.002**	0.163	0.149	0.656	**0.001**	0.517	
Cover of *Carex* (top diagonal) and *Eleocharis* (bottom diagonal)							
Control		1.000	0.998	0.584	0.167	0.953	**0.002**
25h	0.404		0.977	0.385	0.086	0.838	**0.001**
100h	0.349	1.000		0.875	0.401	0.999	**0.007**
300h	**0.012**	0.627	0.688		0.980	0.986	0.123
25r	1.000	0.621	0.558	0.029		0.687	0.487
100r	**0.027**	0.814	0.861	1.000	0.061		**0.022**
300r	**<0.001**	0.071	0.088	0.842	**0.001**	0.663	
Proportion of biomass plants (not litter)							
Control		0.861	0.913	0.217	1.000	0.777	**0.011**
25h			1.000	0.898	0.950	1.000	0.195
100h				0.843	0.975	1.000	0.152
300h					0.337	0.949	0.835
25r						0.899	**0.022**
100r							0.264