Indicators for assessing tropical alpine rehabilitation practices

Duchicela, S.A.; Cuesta, F.; Pinto, E.; Gosling, W.D.; Young, K.R.

DOI
10.1002/ecs2.2595

Publication date
2019

Document Version
Final published version

Published in
Ecosphere

esa

ECOSPHERE

Indicators for assessing tropical alpine rehabilitation practices

Sisimac A. Duchicela [1,2,†] Francisco Cuesta [1,3] Esteban Pinto,[1] William D. Gosling [3] and Kenneth R. Young [2]

[1]*Biodiversity Department, Consortium for the Sustainable Development of the Andean Ecoregion (CONDESAN), Germán Alemán E12-123, 593 2 224 8491, Quito, Ecuador*
[2]*Department of Geography and the Environment, University of Texas at Austin, Austin, Texas 78712 USA*
[3]*Department of Ecosystem and Landscape Dynamics, University of Amsterdam, Amsterdam, The Netherlands*

Citation: Duchicela, S. A., F. Cuesta, E. Pinto, W. D. Gosling, and K. R. Young. 2019. Indicators for assessing tropical alpine rehabilitation practices. Ecosphere 10(2):e02595. 10.1002/ecs2.2595

Abstract. In this study, we evaluate the effectiveness of indicators for rehabilitation practices in high mountain landscapes that were aimed at increasing grassland palatability and biomass accumulation. Focusing on the department of Huancavelica in Peru, the importance of rehabilitation practiced in this area involves the relationship of alpaca pastoralists and their need to produce wool. Overgrazing in this area has decreased the carrying capacity of the system, which may be problematic for continuing their present levels of grazing. Therefore, rehabilitation practices, including herbivory exclusion, exclusion with added irrigation, and exclusion with water collecting ditches, were installed to increase vegetation biomass and palatability of the vegetation. The effects of the rehabilitation practices were assessed using six indicators: vegetation coverage, species richness, Shannon-Weiner Diversity Index, below and aboveground biomass, and soil organic matter, which were analyzed using mixed-effects models. The indicators show that some practices, such as exclusion and ditches, are positively affecting vegetation coverage while negatively affecting species richness. Additionally, biomass showed lower accumulation in areas not excluded from grazing. Therefore, although some of the treatments were initiated as recently as 2013, we can already observe changes in the indicators involving vegetation composition and structure. In the long term, these indicators may allow us to fully understand the effect of the rehabilitation practices on maintaining the carrying capacity of the system. Furthermore, the general approach should be widely applicable in other utilized landscapes.

Key words: ecological indicators; grassland; overgrazing; pastoralism; rehabilitation; restoration.

Received 6 April 2018; accepted 4 December 2018. Corresponding Editor: Karen A. Haubensak.

† **E-mail:** sisaduch@utexas.edu

Introduction

In arid or low-resource environments, pastoralism is an important production system for human populations (Fratkin 1997), although it is under current stress from environmental and socioeconomic changes. Climate change, land degradation, loss of common property, and outmigration are among the main issues that are affecting the pastoral way of life (Fernandez-Gimenez and Le Febre 2006). Pastoralism's extensive use of land provides subsistence or marketable products, and for some locales, this production is required year-round, due to the lack of growing periods suitable for arable agriculture. Sustainability of such systems has ecological, economic, and social axes; the rates of change in these coupled systems require appropriate adaptive strategies (Folke et al. 2005). Camelid pastoralism is an historically important land use system in the Andes, providing products year-round, and is apparently a successful

adaptation to mountainous environments given its antiquity (Postigo et al. 2008). The dates and location of camelid domestication and the beginnings of alpaca pastoralism suggest it began as early as 8500–8000 yr. BP within the Central Andes (Lynch 1983, Mengoni Goñalons 2008). Subsequently, camelid domestication and pastoralism are thought to have spread across the puna (tropical alpine zone) by horizontal regional exchange over the following 4000 yr with a peak during the Inca Empire between 1430 and 1532 AD (Lynch 1983, Baied and Wheeler 1993).

Present-day evidence of the intimate relationship between mountain-dwellers and alpaca pastoralism in the Andes can be seen through changes in land cover, including vegetation stature and the extent of bare surfaces, which can be used as indicators of overgrazing. Although shifts in the pattern of highland vegetation are typically attributed to climatic and fire processes (Gosling et al. 2009), grazing is an important factor in the vegetation community composition in mountain grasslands (Postigo 2012, Kessler et al. 2014, Sylvester et al. 2017). Furthermore, because grazing intensities are variable even within the same area, the resulting vegetation patterns are heterogeneous and respond to site-specific environmental factors (Stohlgren et al. 1999). Given that the puna ecosystem's vegetation dynamics have been modified for thousands of years, existence of an undisturbed reference of this vegetation is controversial or according to some studies only found in inaccessible places (Villota and Behling 2013, White 2013, Heitkamp et al. 2014, Sylvester et al. 2014).

Here, we examine ecosystem rehabilitation methods, taking advantage of a situation where the social and economic dimensions of alpaca pastoralism were otherwise accounted for through collaborating with highland communities whose members are committed to the long-term sustainable use of their lands. The goals of the rehabilitation done in this area were to increase the biomass of palatable species to increase the carrying capacity and maintain alpaca fiber production. While it is necessary to have a holistic view of a system in order to make sound recommendations for successful rehabilitation, here we analyze the effectiveness of the ecological indicators selected to assess the biophysical variables set in the rehabilitation goals. Additionally, while in this paper we are referring to this study as an assessment of a rehabilitation project, it was originally conceived by the regional and national governments, as well as by the collaborating international agency, as an ecological restoration project. However, because this project focuses on improving ecosystem functioning instead of reverting back to an original state, it will be referred to as "rehabilitation" in this paper (Bradshaw 1997). To set appropriate goals, we followed the framework proposed by Hobbs and Harris (2001), in which the priority of the restoration project is defined by the goals fit for a specific ecosystem and not by setting a comparison to a reference site. In sum, the aim for this rehabilitation project was to repair damage to key ecological functions or ecosystem services, and the findings of this study refer to the assessment of the effects the rehabilitation practices have had on the system.

In general, restoration of grazed grasslands has been done using exclusions (Daubenmire 1940), although active irrigation systems and ditches are also used in areas with poor water distribution (Herzon and Helenius 2008). However, meeting goals in alpine grassland ecosystems has been challenging given the long periods of time involved for ecosystem recovery and past management practices that have modified land cover (Young 2009, Sarmiento 2010). As a result, we hypothesized that to assess the effectiveness of these practices, the results could best be evaluated by examining above and belowground biomass, species richness and diversity, and the amount of organic matter in the soil. These variables capture the relationship between vegetation coverage and water regulation capacity (Saxton and Rawls 2006). Additionally, grassland restoration initiatives in the Andes have included goals for sustainable livelihoods, where local communities could continue to raise alpacas for fiber production while adopting sustainable land management practices. However, these goals, which would be considered socioeconomic variables, go beyond the scope of this study.

In this paper, we report on the effects of seven years of long-term rehabilitation activities organized by the Ministry of the Environment of Peru (MINAM) and the Belgian Development Agency through their PRODERN (Program for Sustainable Economic Development and Strategic

Management of Natural Resources) project. Additionally, we evaluate the effectiveness of these rehabilitation practices by assessing the following indicators: grassland palatability, biomass accretion, and species composition and diversity. The long-term goals of the rehabilitation project are to increase and sustain the system's carrying capacity for alpaca grazing. Three management practices were incorporated: exclusions, which are aimed at keeping alpacas out of the closed area and hence preventing grazing and trampling; trenching of hillsides (ditches), which were installed under the assumption that they would help increase rain and groundwater infiltration and prevent runoff; and irrigation, which uses wells fed by glacial meltwater and is used only in the dry season (Table 1). The irrigation and ditch treatments were installed with the purpose not only of increasing cover of palatable species, but also of increasing the rate at which palatable species dominate, as this would be an expected long-term benefit from these management practices.

Methods

Study site

The Huancavelica department in Peru maintains the highest alpaca wool production in Peru but is also among the poorest economically in the country (Poma et al. 2009; Appendix S1: Fig. S1). Concern derives from multiple factors: First, there is a possible over-use of the land that has decreased the carrying capacity of the region for more alpaca raising (Postigo et al. 2008). Second, maintaining current alpaca concentrations in highland peatlands, which remain wet during the dry seasons due to glacial melt, may be unsustainable (Verzijl and Quispe 2013, López-I-Gelats et al. 2015). The third driver of concern comes from changes in climate requiring adaptability of the rural population to new conditions (Postigo 2014, López-I-Gelats et al. 2015). Therefore, the restoration practices in this area center on promoting the capture and conservation of water for use during the dry season. Although this area has had a long history of grazing, overgrazing has recently begun to be perceived as an environmental threat by local people, including those in the regional government. A decrease in vegetation coverage has been observed to decrease the alpaca's weight and wool quantity (West 1981, Raggi et al. 1994). These symptoms have been noticed by community members in Huancavelica.

The department of Huancavelica is located in the south-central Peruvian highlands ~495 km from the country's capital, Lima. This area, with an altitudinal range of 3800–5200 meters above sea level (m.a.s.l.), lies within the Peruvian Central Andean puna, mostly covered by tussock grasslands combined with occasional poorly

Table 1. Description of treatments installed by the regional government (GORE) and/or PRODERN in the four experimental sites in Huancavelica (HVCA), Peru.

Treatment	Goal	Sites with treatment	Year of installation	Total area excluded (Ha)	Funded By
Herbivory exclusion	Deter alpacas from delaying plant recovery	Pilpichaca	2011–2015	195.5	GORE HVCA-PRADERAS
		Pichccahuasi	2011–2012	70.5	GORE HVCA-PRODERN
		Ingahuasi	2012–2015	139	GORE HVCA-PRADERAS
		Ccarhuancho	2012–2014	108	GORE HVCA-PRADERAS
Exclusion plus ditches	Increase water filtration to improve water accumulation during the dry season	Pilpichaca	2013	89.5	GORE HVCA-PRADERAS
		Ingahuasi	No Data	No Data	No Data
		Ccarhuancho	2012	67	GORE HVCA-PRADERAS
Exclusion plus irrigation	Active irrigation system	Pichccahuasi	2012	10.44	PRODERN
		Ingahuasi	2012	19.75	GORE HVCA-PRADERAS & PRODERN
Control (no exclusion)	Baseline	Pichccahuasi	2015	n/a	CONDESAN
		Ingahuasi	2015	n/a	CONDESAN
		Ccarhuancho	2015	n/a	CONDESAN

drained peatlands dominated by cushion plants of the genus *Distichia* (Postigo et al. 2008). Within Huancavelica, the four sites selected for the study, Pilpichaca (4100 m.a.s.l.), Pichccahuasi (4564 m.a.s.l.), Ingahuasi (4480 m.a.s.l.), and Ccarhuancho (4450 m.a.s.l.), have similar climatic conditions and land use histories (Fig. 1).

A puna is a tropical alpine ecosystem found in the Central Andes, between 7° and 27° South from the equator (Baied and Wheeler 1993), and found at an altitudinal range of 3500 to 5500 m.a.s.l. (Veblen et al. 2007). This ecosystem is characterized by sparse, low-growing vegetation adapted to extreme temperature changes.

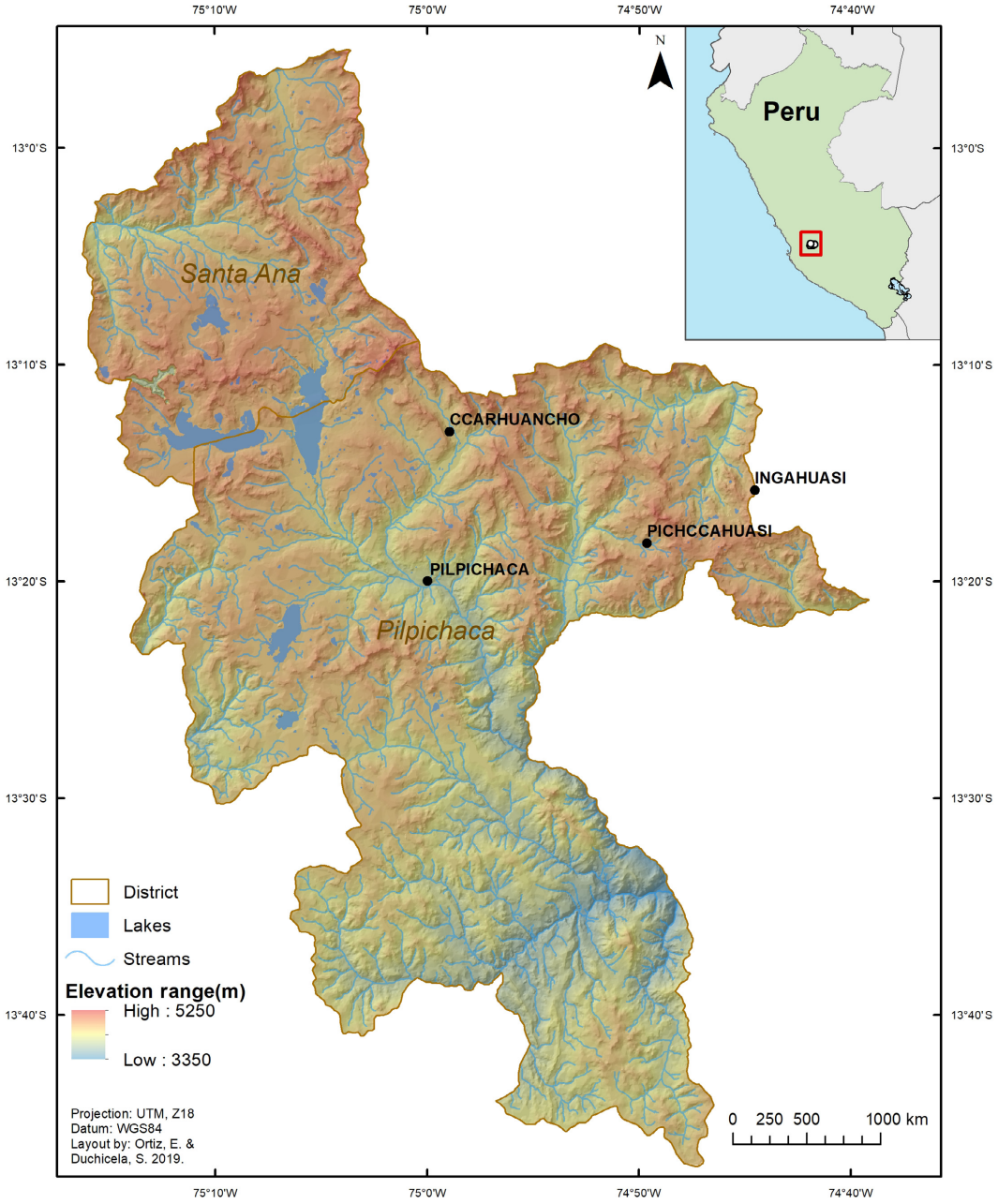

Fig. 1. Map of experimental sites in the Department of Huancavelica in Peru.

One important characteristic of the puna is the low precipitation and humidity that decreases from north to south along the Andes. Additionally, the distinction between wet and dry puna is marked by a decrease in annual precipitation from 500 to 1000 mm for the wet puna to 300–500 mm for the dry puna (Baied and Wheeler 1993). Geographically, the wet puna is found in the north of Peru to the central portion of the eastern cordillera of Bolivia, and the dry puna occurs in western Bolivia, northwestern Argentina, and some parts of southwestern Peru.

The puna has a long history of human intervention, where it has not only been home to ancient civilizations, including pre-Incan cultures, but also to present-day indigenous and peasant communities. There is evidence of human occupation in this area from about 15,000 yr ago (Baied and Wheeler 1993). Therefore, it can be a matter of debate whether there are any natural or undisturbed areas in this region and how recognizable or accessible these areas are. As is mentioned by Josse et al. (2011:158–159), although the puna ecosystem is mainly composed of native species, "these are cultural landscapes that have been managed for centuries."

Seasonality in this area includes a dry and wet season; the wet season starts in late November and finishes in March. However, water availability is a matter of concern and formal irrigation systems are scarce in this area. For example, land use/land cover analyses for Pilpichaca estimate that only 11.4% of the region has irrigation systems in place while 88.6% of the area depends on rainwater (Postigo et al. 2008). Water source is a matter of regional importance given that it varies seasonally, with the wet season which is approximately five months, and the dry season when most water comes from glacially fed streams. Therefore, during much of the year, the base stream discharge is low in relation to the wet season, which in combination with overgrazing significantly affects the functioning of the ecosystem (Dangles et al. 2017). For this reason, combined with strong winds and with recent effects of climate change, all these factors may be contributing to the barren conditions found in Pilpichaca (Postigo 2014) and in landscapes in the rest of the Huancavelica region. In recent decades, trenching of hillsides, also known as drainage ditches, has become a common tool used by local communities (Somers et al. 2018) and has been a widespread national tool proposed by political campaigns as the best way to increase water availability in highland Peru. However, there is scarce literature referring to the ecological effectiveness of these ditches.

Pastoralism is the main economic activity in the Huancavelica region. Typically, for Andean herders, the animals produced are used to barter for other necessary goods (Inamura 1986, Postigo 2012). Herders in this area have mostly focused on raising alpacas (*Lama pacos* L.), llamas (*Lama glama* L.), or sheep (*Ovis aries* L.). In the case of Huancavelica, over-pasturing and excessive trampling are evidenced by the loss in vegetation coverage and by situations wherein one or two plant species, those more resistant to trampling such as *Aciachne acicularis*, become dominant. Other forb-like species may become uncommon or even locally extinct (Salvador et al. 2014).

Alpaca diets change according to seasonal availability of plants (Bryant and Farfan 1984, Reiner 1985), as they prefer grasses and grass-like forbs, that is, *Trichophorum rigidum*, during the dry season, and forbs, that is, *Lachemilla pinnata*, during the wet season, when plant growth increases (Bryant and Farfan 1984). Data collection for the present study was done during the dry season, when grasses and grass-like forbs were well developed. Our results showed that our treatment sites had high coverage of palatable vegetation appropriate for this season.

Through a combination of efforts by the Peruvian Ministry of Economy and Finance (MEF) and MINAM, management practices novel to Huancavelica have been implemented by these agencies since 2009 and 2011, respectively (Table 1). MEF's projects for public investment (PIP) were developed to aid in the production capacity of public goods and services (MEF-DGIP 2014); the PIPs PRADERAS project was developed as the main mechanism to mobilize government funding toward rural communities. In Huancavelica, the PRADERAS project has focused mostly on supporting and aiding communities with their current agricultural practices. In 2011, MINAM formed PRODERN in collaboration with the Belgian Development Agency. PRODERN's main objectives are to aid in the

reduction in current poverty levels using local natural resources and the region's biological diversity (PRODERN 2015*a*). In this collaboration and for these restoration projects, PRODERN supplied the technical support for the funding provided by the PRADERAS project. While this project was not initiated by the communities, current management of the restoration is mainly done by the communities involved (PRODERN 2015*b*). Finally, in 2015, the non-governmental organization Consortium for the Sustainable Development of the Andean Ecoregion (CONDESAN) was asked to create a monitoring plan for these practices, which resulted in the installment of monitoring plots and annual to biennially monitoring.

Methodology

Field sampling.—Sampling and monitoring of the rehabilitation practices were done in 2015, when the practices had been in place for several years. Survey plots were arranged in a block design similar to the experimental design used by Báez et al. (2013; Fig. 2). For each of the three main treatments and their controls, monitoring blocks were installed. Treatments included the following: herbivore exclusion, exclusion plus drainage ditches, and exclusion plus irrigation (Table 1). The sites with no exclusion were used as local controls for the sites, and, therefore, refer to areas that are still subjected to trampling and foraging; they were considered baselines for the monitoring.

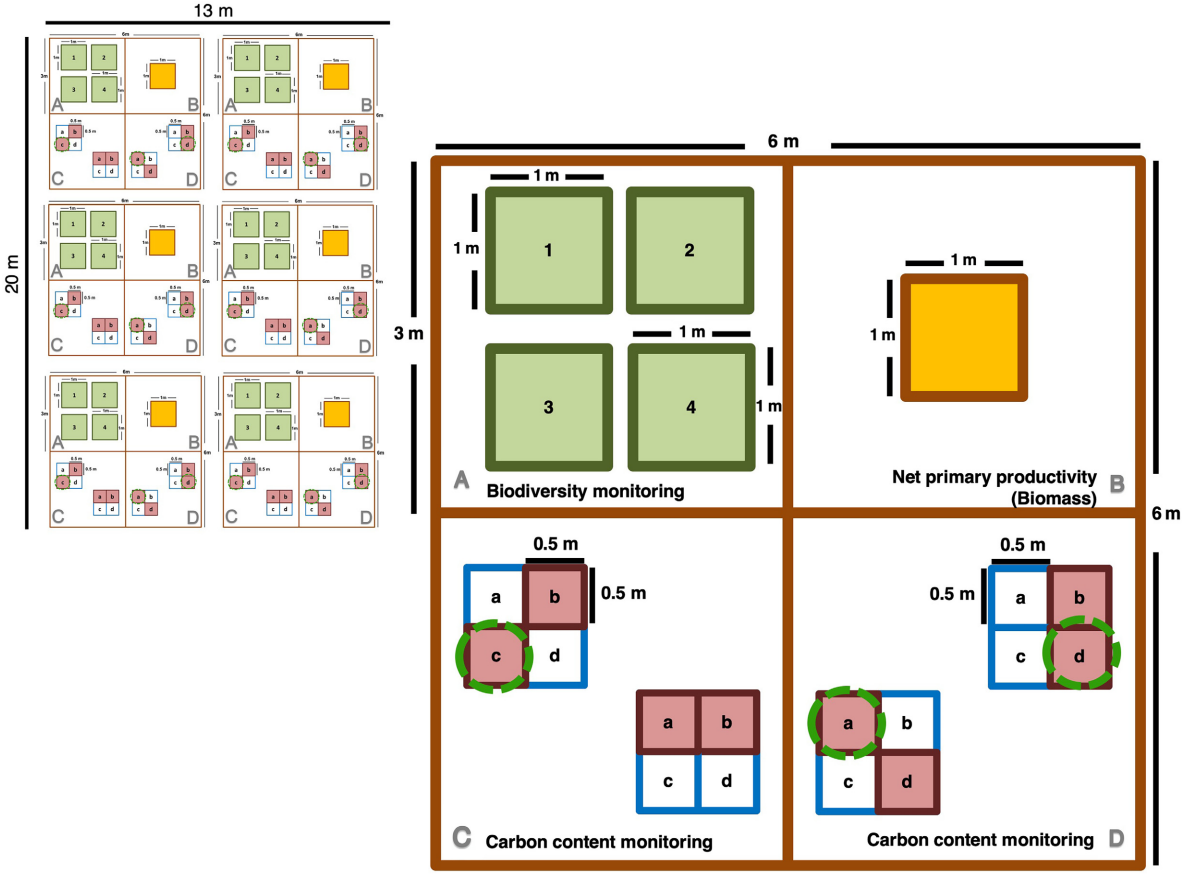

Fig. 2. Block design showing detail of the four experimental plots (sub-units) labeled (A–D). Organization of design consists of blocks containing six to eight units; units contain four sub-units; sub-units contain quadrats, which may contain sub-quadrats as is the case for sub-units (C) and (D). Sub-units (C) and (D) are used for carbon content monitoring. Here, the sub-quadrats sampled are selected at random, symbolized by green circle. Sub-quadrats a–d are used for collecting belowground and aboveground biomass together with soil organic carbon.

A single treatment was assigned per block, and blocks were placed at least 500 m apart, ensuring treatments were spatially separated enough to prevent cross-effects. Two sizes of blocks were installed: blocks with six units had a total area of 20 × 13 m and blocks with eight units had a total area of 27 × 13 m. Four experimental plots (sub-units) were installed within the units labeled with the letters A through D. Plot A was used to monitor biodiversity shifts using four permanent 1 × 1 m quadrats (10 × 10 cm sub-divided 1 × 1 m frames were used for this sampling) to determine frequency and species composition. Plot B was used for primary productivity measurements in a 1 × 1 m quadrat, where regrowth after harvesting will be measured in the next monitoring. Quadrats in plots C and D contained four sub-quadrats used for carbon content monitoring (soil organic matter, necromass, and belowground biomass).

The following indicators were used to assess the effects of the treatments over time: aboveground biomass, soil organic matter (SOM), dried root content of first soil core (belowground biomass), vegetation composition and structure, and litter and vegetation coverage (Table 2).

Laboratory protocol.—Materials obtained from soil coring were weighed upon collection from the field. The samples obtained for biomass were sorted between live matter and decomposing material. After sorting, samples were dried for two days and then weighed again. Plant samples that were not identified in the field were collected, dried, and identified in an herbarium. Samples were processed, and vouchers were deposited in the Augusto Weberbauer Herbarium (MOL) at the Universidad Agraria La Molina in Lima, Peru.

Data analysis.—Data analyses were done in R Studio, version 3.2.2 for Mac (R Core Team 2016). Six metrics were used to determine the success, measured as the deviation from "no exclusion" plots: vegetation coverage, species richness, Shannon-Weiner Diversity Index (hereafter Shannon index), aboveground biomass, belowground biomass, and soil organic matter (Table 2).

Species richness was calculated by counting the number of species per quadrat (in all subunits A) and obtaining the mean of all the biodiversity quadrats per treatment (block) in each site. For the Shannon index, the percent coverage of each species from the plant community composition plots was used, and the index was calculated using the vegan package in R (Oksanen et al. 2016). Additionally, a Games-Howell post hoc test was applied to determine groupings

Table 2. Description and methodology of response variables.

Variable	Metrics	Plot size	Methods	Description	Units
Aboveground biomass	Aboveground biomass (ABG)	1 × 1 m plots and 50 × 50 cm plots	Vegetation was pruned within the two sized plots	Shows the productivity of the system using foliage, bark, trunks, stems	g/m^2
Belowground biomass	Belowground biomass (BGB)	50 × 50 cm plots using 19 × 9.1 cm core	Roots were obtained from soil cores. They were sieved, dried, and weighed	Shows productivity of living root mass	g/m^2
Soil organic matter	Soil organic matter (SOM)	50 × 50 cm plots using 19 × 9.1 cm core	Soil cores were obtained from previously pruned plots. Two cores were extracted, one was used in analysis. Roots were removed from soil and then dried and weighed	Variable shows non-decomposed matter	Percentage
Vegetation composition and structure	Species richness, Shannon Diversity Index	1 × 1 m plots	Count of occurrence of species	Variable shows vegetation diversity	Count, Shannon Index
Vegetation composition and structure	Vegetation coverage	1 × 1 m plots	Percentage coverage of vegetation in 10 × 10 cm subplot	Variable shows land cover	Percentage

Note: Collections followed the IPCC's 2006 good measures guidelines (Aalde et al. 2006).

between treatment means; these were done without taking site or time into consideration.

Mixed-effects models.—We used generalized linear mixed-effects models and general linear mixed-effects models to analyze the effects of the restoration treatments on these six response variables using the lme and glmer functions from the lme4 package in R (Bates et al. 2015). Treatments of all variables were tested against the no exclusion treatment (hereafter control). Specifically, the comparison first assessed whether there was a difference from the control and then whether there was positive (increase in the variable) or negative (decrease in the variable) change.

We used a generalized linear mixed-effects model for species richness, modeling the data with a Poisson distribution (glmer from the lme4 package). Additionally, we modeled vegetation coverage and soil organic matter using the glmmADMB (automatic differentiation model builder) package, modeling the data as a beta distribution. Aboveground biomass, belowground biomass, and the Shannon index were log-transformed, and normality was tested for all variables. The treatments and time of installation were defined as the fixed factors and sites as the random effects. We tested all full models against an empty or null model that did not contain treatments to assess whether the fixed variable was causing an effect on the response variable. This process was repeated for the time variable, where models with time were tested against models without time. Additionally, to determine significant effects of treatment effect on the response variable, we used the lmerTest package in R (Kuznetsova et al. 2016).

Similarity percentage analysis.—We analyzed plant community composition patterns (species abundance patterns) for the different treatments. The objective was to compare which species were contributing the most for each treatment and to compare this to the control. For this purpose, we constructed a matrix of the average cover of each vascular plant species, averaged over the 24 or 32 1 × 1 m quadrats in the 6-unit blocks or 8-unit blocks, respectively. We standardized and square root-transformed species cover, to increase the weight of low-abundance species in the analysis (8% of the species had a mean coverage higher than 3%). We then constructed a similarity matrix using Bray-Curtis as floristic similarity metric and performed a similarity percentage procedure (SIMPER, Primer v6) to determine which plant species characterized the plant community in each of the treatments (up to a 50% contribution to the within-group cumulative similarity). The analysis was carried out in Primer v6 (Clarke and Gorley 2006).

Results

Comparisons against null models and between treatments

Vegetation coverage, aboveground biomass, and belowground biomass were affected by both time and the treatments. Soil organic matter was not affected by the treatments but was affected by time, while species diversity did not consistently have a change with time or treatments. Specifically, results from the ANOVAs comparing the full models and null hypothesis model (Table 3) showed that treatments and time are causing statistically significant differences from the control in most biomass variables, except for soil organic matter. Biodiversity variables, on the other hand, were less affected by either time or treatment except for vegetation cover. These findings were corroborated by the significance tests within the mixed-effects models, where time was also shown to cause significant differences (Table 4). Using the ANOVA results allowed us to distinguish the variables for which both time and treatments were causing differences from the control. Furthermore, the results from the mixed-effects models, using lmerTest, glmer, and glmmADMB, provided insight into the differences due to the specific treatments for each variable. Additionally, the results from the mixed-effects models showed that time of installation is causing most variables to be significantly different from the control except for species richness, which corresponds with the results from the ANOVAs. Therefore, for species

Table 3. Significance of treatment and time effects from full model versus null model ANOVAs.

Variable	Treatment	Time
Vegetation coverage	$P < 0.001$	$P < 0.001$
Species richness	$P = 0.23$	$P = 0.49$
Shannon-Weiner Diversity Index	$P < 0.001$	$P = 0.07$
Aboveground biomass	$P < 0.001$	$P < 0.001$
Belowground biomass	$P < 0.001$	$P < 0.001$
Soil organic matter	$P = 0.28$	$P < 0.05$

Table 4. Coefficients and standard errors of mixed-effects models per response variable tested against control variable.

Variable/Statistic	Belowground biomass	Aboveground biomass	Soil Organic matter	Shannon Diversity Index	Species richness	Vegetation coverage
Intercept	4.69***/(0.91)	6.09***/(1.07)	2.95*/(1.18)	1.96***/(0.47)	1.55***/(0.10)	7.08***/(1.43)
Exclusion	−0.58***/(0.15)	−0.56***/(0.16)	−0.35/(0.20)	−0.12/(0.08)	−0.07/(0.08)	−0.34/(0.24)
Exclusion plus Irrigation	−0.49*/(0.23)	−1.03***/(0.26)	−0.49/(0.30)	−0.22/(0.12)	0.04/(0.08)	0.23/(0.35)
Exclusion plus Ditches	−1.06***/(0.23)	−0.42/(0.24)	−0.48/(0.26)	−0.39***/(0.10)	−0.13/(0.08)	0.88**/(0.30)
Time from installation	−0.06***/(0.02)	−0.09***/(0.02)	−0.05*/(0.02)	−0.02*/(0.01)		−0.09***/(0.02)
Log Likelihood	−162.38	−120.99	55.01	−103.68	−570.06	877.21
Num. obs.	179	151	70	280	280	280
Num. groups: Site	4	4	4	4	4	4
Var: Site (Intercept)	0.01	0.24		0.02	0.03	
Var: Residual	0.35	0.27		0.12		
Variance: Site			0.20			0.78
Dispersion: parameter			21.62			10.03
Dispersion: SD			3.69			1.18

Notes: Standard error is in parenthesis.
$^{***}P < 0.001$; $^{**}P < 0.01$; $^{*}P < 0.05$; $P < 0.1$.

richness, models did not include time of installation as a covariable. Finally, for the Shannon index, time was only marginally significant in the ANOVAs, but when analyzed in the mixed-effects model, time of installation was significant.

Biodiversity

To analyze directly the effects of the treatments on biodiversity, we considered the Shannon index, vegetation coverage, and species richness. Results showed that while species richness was not affected by the treatments, the Shannon index and vegetation coverage were specifically by the exclusion plus ditch treatment. The ANOVAs showed that treatments were associated with significant differences in the Shannon index (Table 3). Therefore, since the Shannon index accounts for evenness as well as richness, the same amount of species might be present but the evenness of the species within the plant community could be changing as a response to the restoration treatments. The Games-Howell results for the Shannon index showed a different grouping for the exclusion plus ditch treatment, which corresponds to the results from the mixed-effects model (Fig. 3), where it shows a significant difference. Additionally, both the model coefficients (Fig. 4, Table 3) and the Games-Howell results (Fig. 3) show a negative change in comparison with the control for this specific treatment. Furthermore, the Games-Howell results group the exclusion plus ditches treatment with the exclusion treatment (Fig. 3), which according to the model coefficients also shows a decrease from the control. However, the exclusion treatment did not significantly affect the Shannon index.

When analyzing the effects of other treatments on the biodiversity variables, the exclusion plus irrigation treatment is shown to be increasing species richness, albeit not with a statistically significant effect. Additionally, for the vegetation coverage variable, results from the null time models against the full models showed that differences from the control are due to time ($P < 0.001$) and the treatments ($P < 0.001$; Table 3). The mixed-effects model showed that exclusion plus ditches was the only treatment causing vegetation coverage to be significantly different from the control (Table 4). This difference was positive, given there was an increase in vegetation coverage. The exclusion plus irrigation treatment also showed an increase in the percentage of vegetation coverage, yet was not found statistically different; however, it was grouped separately from the control in the Games-Howell post hoc (Fig. 4). Finally, the exclusion treatment had a negative effect on all biodiversity variables (Table 4, Fig. 4).

Biomass

The results from the biomass variables show that when livestock are present, plant biomass

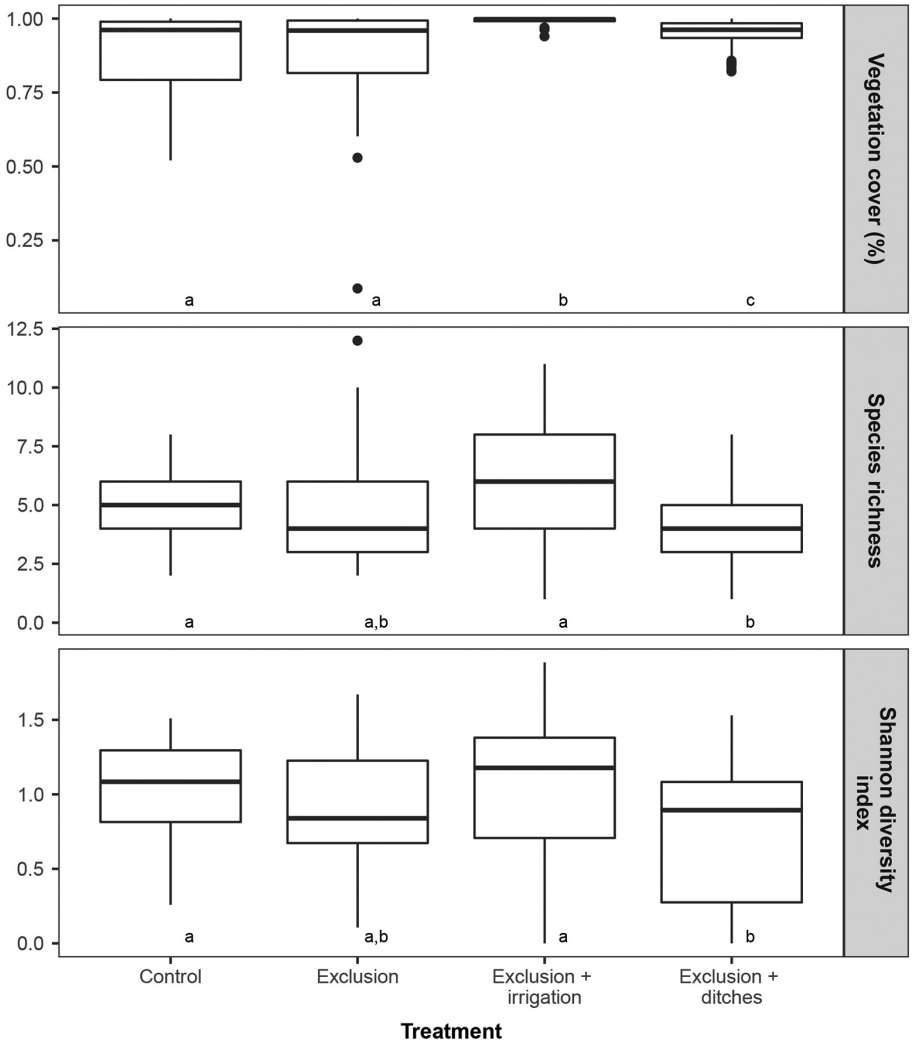

Fig. 3. Boxplots of biodiversity variables including species richness, Shannon Index, and vegetation coverage by treatment. Letters represent groupings resulting from the Games-Howell post hoc.

results from a combination of plant growth and the effects of grazing/trampling and that there is a relationship between belowground processes and vegetation cover. To analyze the effects of the treatments on biomass, we considered the following variables: aboveground biomass, belowground biomass, and soil organic matter. Results for these variables showed that all treatments significantly decreased belowground biomass. Exclusion and exclusion plus irrigation decreased aboveground biomass, and exclusion and exclusion plus ditches decreased soil organic matter. Additionally, soil organic matter had the opposite response from vegetation cover, where if it decreased due to the effect of one treatment, vegetation cover increased for that same treatment.

The ANOVAs comparing full versus null models for above and belowground biomass showed significant differences for time and treatment (Table 3); however, for soil organic matter, no treatments were significant. These results were corroborated by the mixed-effects models, where time of installation was significant for all variables. The three biomass variables responded differently to the treatments. First, the mixed-effects model for aboveground biomass showed a significant decrease in the exclusion and exclusion plus irrigation treatments (Table 4, Fig. 4).

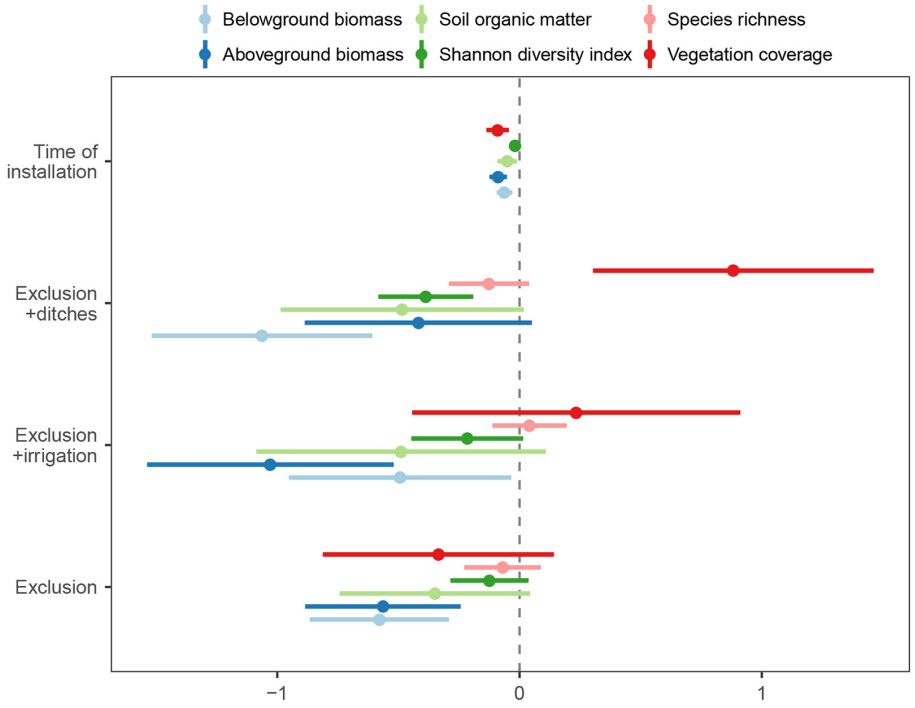

Fig. 4. Comparison of mixed-effects models' coefficients by treatments. Line shows 95% confidence interval.

Second, all treatments caused a significant decrease in belowground biomass (Table 4, Fig. 4). Finally, the exclusion and exclusion plus ditch treatments had a marginally significant decrease in soil organic matter (Table 4, Fig. 4). Additionally, the Games-Howell post hoc (Fig. 5) showed the following similarities: For aboveground biomass, exclusion and exclusion plus irrigation were grouped separately; and for soil organic matter, exclusion plus ditches was grouped separately and was lower than the control. Therefore, although the exclusion treatment was significant (or marginally significant) for all three variables, it was not grouped separately from the control. This difference may be attributed to the Games-Howell post hoc being done for the entire dataset, taking overall differences into account without consideration of the effect of time or site (Appendix S1: Fig. S2), whereas the mixed-effects model accounts for the effects time and site are contributing to any differences in the variables. Similar patterns within the exclusion plus irrigation treatment and the exclusion plus ditch treatment can be seen in the boxplots (Fig. 5); for example, all variables in the exclusion plus irrigation show an increase. Inverse patterns can be seen for the exclusion plus ditch treatment, where all the variables show a decrease from the control.

Similarity percentage analysis

The SIMPER analysis showed which species were dominant in percent cover in each treatment and compared the dissimilarity between treatments (Fig. 6). These differences show a potential shift of the flora composition toward dominance by more palatable species, as an effect of the treatments. The main contributing species in the control was *Aciachne acicularis*, a species highly resistant to trampling and an indicator of degradation; in contrast, *A. acicularis* was the second or third highest contributor to the other treatments, and more palatable species were more dominant. For example, the highest contributor to the exclusion plus ditch group was *Calamagrostis vicunarum* comprising 48.68% of coverage, followed by *Lachemilla pinnata* and then *A. acicularis*. Similarly, *C. vicunarum* contributed 52.12% in the exclusion treatment and 47.79% in the exclusion plus irrigation treatment,

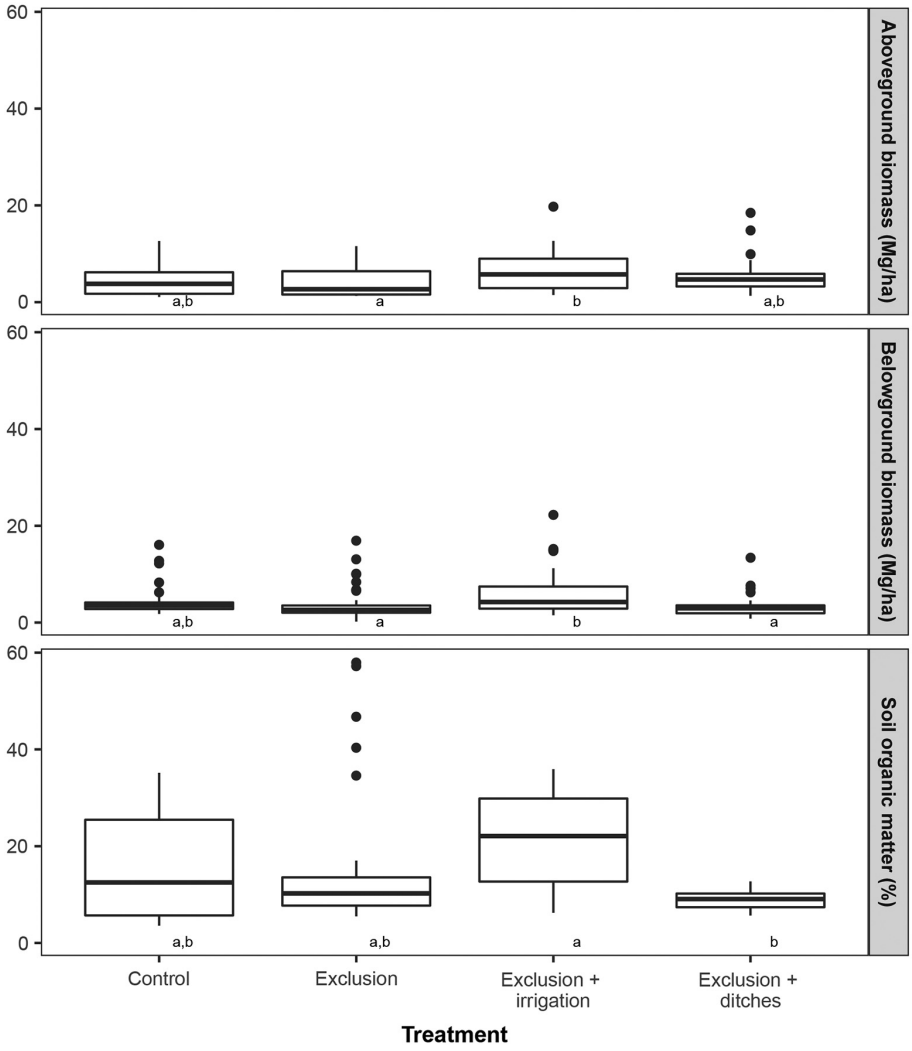

Fig. 5. Boxplots of aboveground biomass, belowground biomass, and soil organic matter in Mg/ha and percent organic matter per treatment.

followed by *A. acicularis* in both cases. The control site is explained largely by *A. acicularis* (43.99%), followed by *L. pinnata* (19.94%), *Azorella diapensioides* (17.76%), and *C. vicunarum* (17.03%). Differences between treatments ranged from 49.58% of dissimilarity (between the exclusion and the exclusion plus irrigation groups) and 56.85% of dissimilarity (between exclusion plus ditches and exclusion plus irrigation). Treatments compared to the control groups were 56.36% dissimilar (against exclusion plus irrigation), 56.42% dissimilar (against exclusion), and 53.14% dissimilar (against exclusion plus ditches).

Discussion

Research on the ecological rehabilitation of Andean landscapes has been scarce, especially those targeting specific issues of treatment selection and goal setting. In the past decade, there has been an increase in this type of project globally due to the increased awareness of two key stressors, climate change and water shortage due to unsustainable land management practices (Suding 2011). Additionally, important economic motives have led to restoration meant to improve grazing quality through the conservation of biodiversity and the favoring of specific native

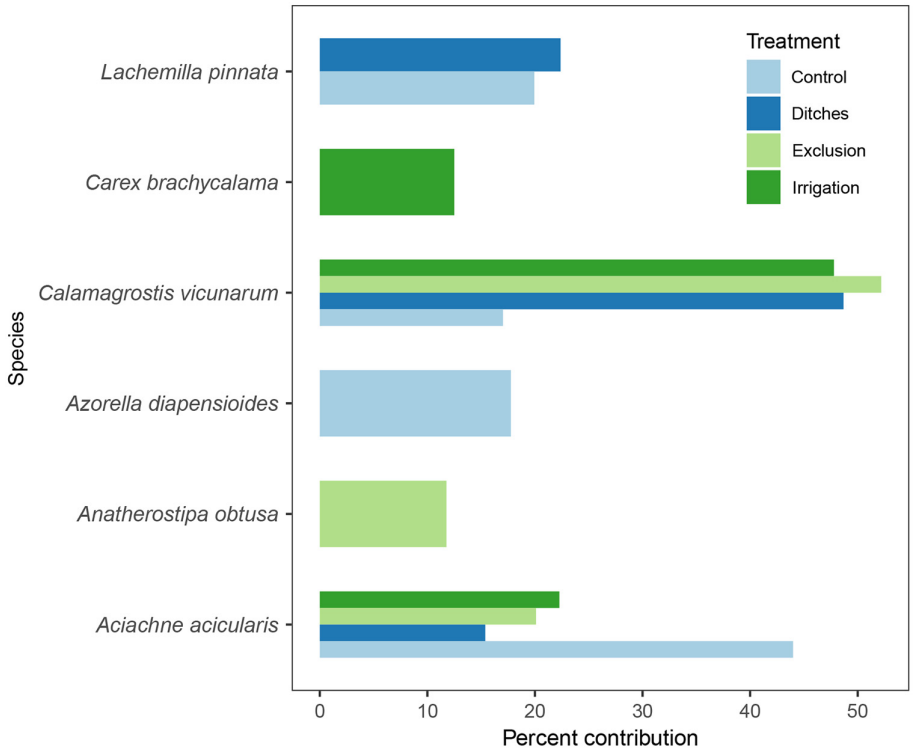

Fig. 6. Percent of most contributing species for each treatment.

species (Posada et al. 2000, Papanastasis 2009). Ecosystem rehabilitation in the Peruvian highlands is relatively uncommon but is needed to sustain the large population that lives in the highlands, which is becoming increasingly marginalized by rapid land use change induced by people and glacial melt (Postigo et al. 2008). It is likely that the methods explored here would be useful in many marginal but utilized landscapes worldwide.

We found that some rehabilitation practices in Huancavelica are making important changes on the respective landscapes. These effects are indicated by the increase in dominance of palatable species and the increase in vegetation cover over bare soil. However, the three treatments in the four study sites showed that ecosystem responses were not linear for some variables and that this variation depended on the treatment and the time period involved. For example, for the irrigation treatment, an increase in vegetation coverage was accompanied by a decrease in species richness and Shannon index (when controlling for time and site). On the other hand, there was an inverse response of aboveground and belowground biomass from vegetation cover; when vegetation cover increased, the other two variables decreased. We demonstrated that the indicators used were sensitive enough to detect that (1) palatable vegetation for alpaca grazing can be increased through rehabilitation methods, (2) there were different factors that affect plant community composition versus biomass accumulation, and (3) that specific indicators provided different kinds of insights in regard to the timing and success of rehabilitation processes.

Indicator species and an increase in vegetation palatability

The dominance of *Aciachne acicularis* in the no exclusion sites is an important finding because it is an indicator species for soil degradation (Salvador et al. 2014). Additionally, *A. acicularis* is not palatable for alpacas, so the high abundance of this species indicates that the areas that were not excluded have a reduced palatability and usefulness for the alpacas. In terms of the

increased species richness in the non-excluded areas, trampling is an intermediate frequency disturbance that promotes competitive exclusion and colonization by less competitive species (Hobbs and Huenneke 1992, Roxburgh et al. 2004). Although the dominance of *Calamagrostis vicunarum* demonstrates an increase in palatability, since it has been shown to be among the plants preferred by alpacas (Bryant and Farfan 1984, Reiner 1985), it also shows that *C. vicunarum* is outcompeting other lower-statured species found in the no exclusion zones, like *A. acicularis* and *Lachemilla pinnata*.

Through our study, the increased abundance inside exclusion areas of species preferred by alpacas shows that the objectives of the restoration were being met. Namely, there are low-technology methods available that increase pasture value for the local people with fencing. However, seasonal variation in the availability of species favored by the alpacas may lead to different results during the wet season. Therefore, a rotation system linked to the wet and dry season is a fundamental need for herd management at the landscape scale (Jacobo et al. 2006). Additionally, because *A. acicularis* is the second or third most common species found in the exclusion treatments (Appendix S1: Table S1), we may be observing a replacement process that is occurring over time. From these results, we can see that biodiversity and species evenness are both useful indicators of palatability for alpaca grazing.

Factors affecting plant community composition and biomass accumulation

Overgrazing can cause a decrease in vegetation cover, which increases the exposure of the soil to climatic elements, in turn inducing loss of organic soil through wind erosion and decreasing soil moisture. To the plant community, this process provokes a replacement toward less-palatable short carpet grass vegetation, like *A. acicularis*, and an increase in bare substrate (Podwojewski et al. 2002, Poulenard et al. 2004). Because certain species increase with the elimination of the continued disturbance of alpaca browsing and trampling, the exclusion treatment provides a set of conditions that trigger new paths for plant communities to assemble. It appears that the recovery process in this system will eventually lead to a colonization by more sensitive species that will enrich the plant community.

A second step in the rehabilitation process is the removal of barriers associated with dominant non-palatable species. Through the treatments implemented at the four sites in Huancavelica, non-palatable species were targeted by providing an appropriate environment allowing less trampling-resistant species to compete. Additionally, the treatments providing extra water targeted an important limitation for plant species with higher water needs. However, factors affecting plant biodiversity seem to be causing trade-offs with biomass accumulation processes. For example, the exclusion plus ditch treatment was most effective in the increase in vegetation cover and was significantly different from the control, but it decreased species richness. On the other hand, exclusion plus irrigation was the only treatment to increase species richness, but it had negative consequences for belowground and aboveground biomass. Important differences in these treatments are related to water provision. In addition, there were specific distinctions between treatments that just targeted trampling as a disturbance versus those treatments that combine the elimination of grazing and trampling, with water as a limitation. For example, the negative effect on all biodiversity variables of the exclusion treatment may be showing a reaction when disturbances such as trampling are eliminated but water stress continues to be high. None of these interactions would have been apparent without this experimental design.

Adaptation of plants to their environment raises the question of whether plants in high arid environments are suffering from water stress (Leuschner 2000). In most subtropical Andean regions, a uniform level of moisture present below the top soil horizons provides the appropriate amount of water for plant uptake. However, during the dry season, plant spacing is a crucial strategy of balancing water–plant relations (Körner 2003). In the case of the Peruvian Andes, changes in water availability are a matter of concern due to glacial melt and more erratic precipitation, and therefore, the loss of water availability must be considered for restoration efforts (Rangwala and Miller 2012). This reduced availability potentially affects spacing between plants, resulting in a sparser landscape.

Therefore, when water limitation is removed, increased productivity results, with either an increased vegetation coverage or higher species richness. These are potentially useful insights for local pastoralists interested in improving their grazing systems.

The objectives and mechanism of drainage ditches may explain the differences in soil organic matter and belowground biomass observed with the ditch versus irrigation treatments (Fig. 5). First, drainage ditches are typically considered part of a water transfer network for agricultural landscapes (Herzon and Helenius 2008), since they accelerate the infiltration of water and ensure its retention. Second, ditches are considered a modifier for erosion since in times of excess, rapid water infiltration prevents sediment loss (Herzon and Helenius 2008, Somers et al. 2018). Finally, water is intercepted by vegetation located directly downslope of the ditch, which also increases infiltration and recharges groundwater (Somers et al. 2018). While few studies show the relationship between vegetation cover and drainage ditches, studies in Finland have shown increases in plant species richness near ditches placed in grasslands (Tarmi et al. 2002). In Huancavelica, however, ditches decreased species richness while increasing vegetation cover. Additionally, there were trade-offs between belowground biomass and vegetation cover that seemed to be more drastic for the drainage ditches than in any of the other treatments, followed by the irrigation treatment. Therefore, the mechanism by which water is available to plants in these two treatments determines this distinct trade-off. In addition, a decrease in belowground biomass for the irrigation and ditch treatments may also be due to an investment of resources in vegetation cover due to the elimination of two stressors, alpaca effects and water shortages. Thus, these two treatments seem to favor vegetation cover at the expense of belowground biomass and soil organic matter.

We showed that for our study site, the use of exclusionary methods is important for increasing vegetation coverage while lessening soil erosion. The difference is notable, and visible to observers, even over just a few years. Therefore, by itself, this practice is already causing discernible changes in the plant community composition compared to areas that have not been excluded from alpaca grazing. However, the prevention of grazing and trampling is also affecting the composition of species. In other words, the species that are favored with the decrease in trampling have specific characteristics, that is, in this case bunch grasses and basal rosettes are favored versus carpet-like grasses.

Indicators for rehabilitation off andean ecosystems

Although water is already a limiting factor in this system, where average precipitation is around 500–600 mL/yr (Lagos et al. 2008), local people have recently been perceiving a decrease in water availability. This further decrease in water availability may be linked to a changing climate, but it may also be due to more exposed soils due to increased grazing, and therefore a decrease in vegetation coverage, which promotes lower water storage. Because the rehabilitation practices aim to eliminate water stress, the higher vegetation coverage and more belowground biomass shown in our results can be indicators of increased productivity (Figs. 3 and 5). Additionally, increased irrigation seemed to be decreasing the soil organic matter, although this was not evident in our results, possibly due to sampling restraints and/or the limited amount of time since treatment installation.

Determining positive indicators of a successful rehabilitation practice has been discussed among restoration scientists (Ruiz-Jaen and Aide 2005). Choosing appropriate indicators is important because it makes monitoring of the restoration more efficient and may be useful for involving local land users in the evaluations. Which indicators exhibit a higher sensitivity for mountainous ecosystems is a debated topic. According to Grabherr and Pauli (2004), an appropriate indicator responds quickly and is sensitive to stress. Soil characteristics, such as soil organic matter and soil microbial biomass, may be among the first variables to change following restoration and therefore may be adequate indicators for mountain ecosystems (Sparling 1992, Carter 2002); additionally, they account for measurable change making them useful for quantifying differences.

Is soil organic matter a good indicator in the Andes and for assessing ecological rehabilitation? According to Abreu et al. (2009) and

Aguilera et al. (2013), the variations of soil components such as soil organic matter are not good indicators for soil fertility restoration in the Andean páramo ecosystem (a wetter tropical alpine environment found to the north of our study area). The heterogeneity of our study landscape caused any variation in soil fertility to be related to an array of other factors and hence not necessarily changing because of the restoration practices. However, it is important to consider the history of land use in these study areas, where their previous use as agricultural fields may have left a legacy of changes in the physical and chemical structure of the soil. In the case of Huancavelica, agriculture is seldom performed, and grazing is done mostly by alpacas and in less proportion by sheep and rarely cattle. In this case, while certain physical characteristics change, important chemical attributes remain the same, making rehabilitation of these areas possible.

In the case of the puna ecosystem of Huancavelica soil organic matter as an indicator for soil quality seems to show measurable changes due to the restoration practices, specifically for the exclusion plus irrigation treatment. Additionally, the sensitivity of this indicator seemed to increase over time. Therefore, soil organic matter is an important indicator to monitor for these rehabilitation practices in puna ecosystems, but results may be time dependent. Important changes may be noticed years after the installation of the practices. Soil organic matter was important here because vegetation diversity and composition are dependent on the availability of nutrients and favorable conditions provided by the soil organic matter. Additionally, this indicator can be easily considered an ecosystem service (Farley et al. 2011); for example, carbon storage could inform policy and socioeconomic goals. In this case, soil organic matter is a dynamic indicator of restoration effects in the puna ecosystem.

Further research

Although the implemented rehabilitation practices seem to be fulfilling their purpose, it is still premature to make specific recommendations to local governments and development agencies regarding replicating or scaling up such practices. This research focused exclusively on assessing the ecological impact of rehabilitation practices but does not try to provide social dimensions of pastoralism in its analysis, which should be incorporated in the analyses prior to informing public policy and international cooperation agendas. Major information gaps still remain regarding an estimate of the costs (e.g., money, time, and area excluded) of implementing such practices and the social benefits derived. Moreover, a key question that remains unanswered is related to the ecosystem regaining its original carrying capacity and the social benefits derived from it. Future research should focus on the following interrelated questions: (1) How many alpacas can graze in the rehabilitated grasslands (and for how long) before degrading the system again? (2) What type of households (e.g., large/small; old/young; and number of grazing areas) would be able to rehabilitate their grasslands (what capacity at the household level is needed to engage in rehabilitation)? (3) What are the implications of excluding grazing areas for households? and (4) Does it impact their short-term economic well-being or security?

Conclusions

1. The rehabilitation treatments assessed in this study targeted two forms of disturbance on the vegetation: exclusion from herbivory and the provision of water. Ecosystem responses to these treatments varied and the indicators used allowed for the assessment of these variations. For example, vegetation coverage seemed to be increasing at a faster rate for the treatments that provided water, yet, this was accompanied by a decrease in species richness.
2. Given that the rehabilitation practices we assessed in this study were aimed at increasing the vegetation palatability for alpaca grazing, the use of indicator species (and their abundances) provided key information in assessing impacts of the different treatments. Areas with treatments concerning water availability combined with exclusion had a reduced percent coverage of *Aciachne acicularis*, a species not palatable for alpacas. Likewise, *Calamagrostis vicunarum*, a palatable species for alpacas, increased in dominance in these same areas.

3. The reduced availability of runoff water in the Peruvian Andes is a matter of concern and must be considered in rehabilitation efforts. The relationship between vegetation cover and the infiltration ditches has been very poorly documented. Here, we were able to see less species richness but a turnover of species toward those more palatable. Yet more time is needed to assess a longer effect of such practices on the system.
4. Choosing impact indicators for assessing rehabilitation practices in high Andean systems is challenging due to the heterogeneity of the landscape. The indicators assessed in this study showed important insights on differences in rate of change, on the direction of change, and on the success of the treatments.

Acknowledgments

We would like to thank the local participants of the four study sites in Huancavelica, Peru, especially Hugo De La Cruz for his invaluable work in the field. Ricardo Jaramillo, Javier Irazabal, and Cecilia Sandoval were indispensable to data collection and field site installation. Edwin Ortiz collaborated with GIS data and map layout. Chase Rakowski and consultants of the Department of Statistics and Data Science at the University of Texas at Austin provided guidance for the statistical analysis and interpretation. Financial support came from the EcoAndes Project conducted by Consortium for the Sustainable Development of the Andean Ecoregion (CONDESAN) and United Nations Environment Programme (UNEP), funded by the Global Environment Facility (GEF), cooperation agreement N° 4750. We thank the Swiss Agency for Development and Cooperation through its Sustainable Mountain Development for Global Change programme (Contract number: 81025761) for covering the Open Access license and making this research open for the general community. Kenneth Young acknowledges funding from the National Science Foundation (DEB-1617429). Finally, we would like to thank two anonymous reviewers who offered comments and suggestions that improved the framework of this study.

Literature Cited

Aalde, H., et al. 2006. Metodologías genéricas aplicables a múltiples categorías de uso de la tierra. *In* H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, editors. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme. IGES, Hayama, Japan.

Abreu, Z., L. D. Llambí, and L. Sarmiento. 2009. Sensitivity of soil restoration indicators during páramo succession in the high tropical andes: chronosequence and permanent plot approaches. Restoration Ecology 17:619–627.

Aguilera, J., P. Motavalli, C. Valdivia, and M. A. Gonzales. 2013. Impacts of cultivation and fallow length on soil carbon and nitrogen availability in the Bolivian Andean Highland Region. Mountain Research and Development 33:391–403.

Báez, S., S. L. Collins, W. T. Pockman, J. E. Johnson, and E. E. Small. 2013. Effects of experimental rainfall manipulations on Chihuahuan Desert grassland and shrubland plant communities. Oecologia 172:1117–1127.

Baied, C. A., and J. C. Wheeler. 1993. Evolution of high Andean Puna ecosystems: environment, climate, and culture change over the last 12,000 years in the Central Andes. Mountain Research and Development 13:145–156.

Bates, D., M. Mächler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using {lme4}. Journal of Statistical Software 67:1–48.

Bradshaw, A. D. 1997. What do we mean by restoration? *in* K. M. Urbanska, N. R. Webb, and P. J. Edwards, editors. Restoration ecology and sustainable development. Cambridge University Press, Cambridge, UK.

Bryant, F. C., and R. D. Farfan. 1984. Dry season forage selection by Alpaca [Lama pacos] in Southern Peru. Journal of Range Management 37:330–333.

Carter, M. R. 2002. Soil quality for sustainable land management. Agronomy Journal 94:38–47.

Clarke, K. R., and R. N. Gorley. 2006. Primer V6: user Manual – Tutorial. PRIMER-E, Plymouth, Maine, USA.

Dangles, O., A. Rabatel, M. Kraemer, G. Zeballos, A. Soruco, D. Jacobsen, and F. Anthelme. 2017. Ecosystem sentinels for climate change? Evidence of wetland cover changes over the last 30 years in the tropical Andes. PLoS ONE 12:e0175814.

Daubenmire, R. F. 1940. Exclusure technique in ecology. Ecology 21:514–515.

Farley, K. A., W. G. Anderson, L. L. Bremer, and C. P. Harden. 2011. Compensation for ecosystem services: an evaluation of efforts to achieve conservation and development in Ecuadorian páramo grasslands. Environmental Conservation 38:393–405.

Fernandez-Gimenez, M. E., and S. Le Febre. 2006. Mobility in pastoral systems: Dynamic flux or downward trend? International Journal of Sustainable Development & World Ecology 13:341–362.

Folke, C., T. Hahn, P. Olsson, and J. Norberg. 2005. Adaptive governance of social-ecological systems. Annual Review of Environment and Resources 30:441–473.

Fratkin, E. 1997. Pastoralism: governance and development issues. Annual Review of Anthropology 26:235–261.

Gosling, W. D., J. A. Hanselman, C. Knox, B. G. Valencia, and M. B. Bush. 2009. Long-term drivers of change in Polylepis woodland distribution in the central Andes. Journal of Vegetation Science 20:1041–1052.

Grabherr, G., and H. Pauli. 2004. Terrestrial ecosystems and global change monitoring: an overview. Pages 46–49Global environmental and social monitoring. Proceedings of the 1st International Thematic Workshop. UNESCO, Vienne, France.

Heitkamp, F., S. P. Sylvester, M. Kessler, M. D. P. V. Sylvester, and H. F. Jungkunst. 2014. Inaccessible Andean sites reveal human-induced weathering in grazed soils. Progress in Physical Geography: earth and Environment 38:576–601.

Herzon, I., and J. Helenius. 2008. Agricultural drainage ditches, their biological importance and functioning. Biological Conservation 141:1171–1183.

Hobbs, R. J., and J. A. Harris. 2001. Restoration Ecology: repairing the Earth's ecosystems in the new millennium. Restoration Ecology 9:239–246.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology 6:324–337.

Inamura, T. 1986. Relaciones estructurales entre pastores y agricultores de un distrito altoandino en el sur del Perú. Pages 141–190 *in* S. Masuda, editor. Etnografia e Historia del mundo andino: Continuidad y cambio. University of Tokyo Press, Tokyo, Japan.

Jacobo, E. J., A. M. Rodríguez, N. Bartoloni, and V. A. Deregibus. 2006. Rotational grazing effects on rangeland vegetation at a farm scale. Rangeland Ecology & Management 59:249–257.

Josse, C., et al. 2011. Physical geography and ecosystems in the tropical Andes. Pages 152–169 *in* S. K. Herzog, R. Martínez, P. M. Jørgensen, and H. Tiessen, editors. Climate change and biodiversity in the tropical Andes. Inter-American Institute for Global Change Research (IAI) and Scientific Committee on Problems of the Environment (SCOPE), Paris, France.

Kessler, M., J. M. Toivonen, S. P. Sylvester, J. R. Kluge, and D. Hertel. 2014. Elevational patterns of Polylepis tree height (Rosaceae) in the high Andes of Peru: role of human impact and climatic conditions. Frontiers in Plant Science 5:194.

Körner, C. 2003. Alpine plant life: functional plant ecology of high mountain ecosystems. Springer Science and Business Media, Berlin, Germany.

Kuznetsova, A., P. Bruun Brockhoff, and R. Haubo Bojesen Christensen. 2016. lmerTest: tests in linear mixed effects models. Journal of Statistical Software 82:1–26.

Lagos, P., Y. Silva, E. Nickl, and K. Mosquera. 2008. El Niño? Related precipitation variability in Perú. Advances in Geosciences 14:231–237.

Leuschner, C. 2000. Are high elevations in tropical mountains arid environments for plants? Ecology 81:1425–1436.

López-I-Gelats, F., J. L. Contreras Paco, R. Huilcas Huayra, O. D. Siguas Robles, E. C. Quispe Peña, and J. Bartolomé Filella. 2015. Adaptation strategies of Andean pastoralist households to both climate and non-climate changes. Human Ecology 43:267–282.

Lynch, T. F. 1983. Camelid pastoralism and the emergence of Tiwanaku civilization in the South-Central Andes. World Archaeology 15:1–14.

Mengoni Goñalons, G. L. 2008. Camelids in ancient Andean societies: a review of the zooarchaeological evidence. Quaternary International 185:59–68.

Ministerio de Economia y Finanzas-Dirección General de Inversión Pública (MEF-DGIP). 2014. Guía general para identificación, formulación y evaluación social de proyectos de inversión pública, a nivel de perfil. MEF-DGIP, Lima, Perú.

Oksanen, J., et al. 2016. vegan: community ecology package. R package version 2.3-5. https://CRAN.R-project.org/package=vegan

Papanastasis, V. P. 2009. Restoration of degraded grazing lands through grazing management: Can it work? Restoration Ecology 17:441–445.

Podwojewski, P., J. Poulenard, T. Zambrana, and R. Hofstede. 2002. Overgrazing effects on vegetation cover and properties of volcanic ash soil in the páramo of Llangahua and La Esperanza (Tungurahua, Ecuador). Soil Use and Management 18:45–55.

Poma, A., C. Ventura, and E. C. Quispe. 2009. Caracterización del perfil de diámetro de fibra de alpaca Huacaya de color blanco. Thesis. Universidad Nacional de Huancavelica, Huancavelica, Peru.

Posada, J. M., T. M. Aide, and J. Cavelier. 2000. Cattle and weedy shrubs as restoration tools of tropical montane rainforest. Restoration Ecology 8:370–379.

Postigo, J. 2012. Responses of plants, pastoralists, and governments to social environmental changes in the Peruvian Southern Andes. Thesis. University of Texas at Austin, Austin, Texas, USA.

Postigo, J. 2014. Perception and resilience of Andean populations facing climate change. Journal of Ethnobiology 34:383–400.

Postigo, J., K. Young, and K. Crews. 2008. Change and continuity in a pastoralist community in the high Peruvian Andes. Human Ecology 36:535–551.

Poulenard, J., J. C. Michel, F. Bartoli, J. M. Portal, and P. Podwojewski. 2004. Water repellency of volcanic ash soils from Ecuadorian páramo: effect of water content and characteristics of hydrophobic organic matter. European Journal of Soil Science 55:487–496.

PRODERN. 2015*a*. Ministra del Ambiente Elsa Galarza destaca logros del PRODERN en Huancavelica. http://prodern.minam.gob.pe/blog/ministra-del-ambiente-elsa-galarza-destaca-logros-del-prodern-en-huancavelica

PRODERN. 2015*b*. Conoce cómo trabajamos en PRODERN. http://prodern.minam.gob.pe/quienes-somos

R Core Team. 2016. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Raggi, L. A., E. Jiliberto, and B. Urquieta. 1994. Feeding and foraging behaviour of alpaca in northern Chile. Journal of Arid Environments 26:73–77.

Rangwala, I., and J. R. Miller. 2012. Climate change in mountains: a review of elevation-dependent warming and its possible causes. Climatic Change 114:527–547.

Reiner, R. J. 1985. Nutrition of alpacas grazing high altitude rangeland in southern Peru. Dissertation. Texas Tech University, Lubbock, Texas, USA.

Roxburgh, S. H., K. Shea, and J. B. Wilson. 2004. The intermediate disturbance hypothesis: patch dynamics and mechanisms of species coexistence. Ecology 85:359–371.

Ruiz-Jaen, M. C., and M. T. Aide. 2005. Restoration success: How is it being measured? Restoration Ecology 13:569–577.

Salvador, F., J. Monerris, and L. Rochefort. 2014. Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. Mires and Peat 15:1–17.

Sarmiento, F. O.. 2010. Anthropogenic change in the landscapes of highland Ecuador. Geographical Review 92:213–234.

Saxton, K. E., and W. J. Rawls. 2006. Soil water characteristic estimates by texture and organic matter for hydrologic solutions. Soil Science Society of America Journal 70:1569–1578.

Somers, L. D., J. M. McKenzie, S. C. Zipper, B. G. Mark, P. Lagos, and M. Baraer. 2018. Does hillslope trenching enhance groundwater recharge and baseflow in the Peruvian Andes? Hydrological Processes 32:318–331.

Sparling, G. 1992. Ratio of microbial biomass carbon to soil organic carbon as a sensitive indicator of changes in soil organic matter. Australian Journal of Soil Research 30:195.

Stohlgren, T. J., L. D. Schell, and B. Vanden Heuvel. 1999. How grazing and soil quality affect native and exotic plant diversity in rocky mountain grasslands. Ecological Applications 9:45–64.

Suding, K. N. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics 42:465–487.

Sylvester, S. P., F. Heitkamp, M. D. P. V. Sylvester, H. F. Jungkunst, H. J. M. Sipman, J. M. Toivonen, C. A. Gonzales Inca, J. C. Ospina, and M. Kessler. 2017. Relict high-Andean ecosystems challenge our concepts of naturalness and human impact. Scientific Reports 7:3334.

Sylvester, S. P., M. D. P. V. Sylvester, and M. Kessler. 2014. Inaccessible ledges as refuges for the natural vegetation of the high Andes. Journal of Vegetation Science 25:1225–1234.

Tarmi, S., H. Tuuri, and J. Helenius. 2002. Plant communities of field boundaries in Finnish farmland. Agricultural and Food Science in Finland 11:121–135.

Veblen, T. T., K. R. Young, and A. R. Orme. 2007. The physical geography of South America. Oxford University Press, New York, New York, USA.

Verzijl, A., and S. G. Quispe. 2013. The system nobody sees: irrigated wetland management and alpaca herding in the peruvian andes. Mountain Research and Development 33:280–293.

Villota, A., and H. Behling. 2013. Late Quaternary vegetation, climate, and fire dynamics: human impact and evidence of past Polylepis populations in the northern Andean depression inferred from the El Cristal record in southeastern Ecuador. Ecotropica 19:39–58.

West, T. L. 1981. Alpaca production in Puno, Peru. Small Ruminant Collaborative Research Support Program (SR-CRSP), Department of Rural Sociology, University of Missouri, Columbia. Publication 3. Macmillan, New York, New York, USA.

White, S. 2013. Grass páramo as hunter-gatherer landscape. Holocene 23:898–915.

Young, K. R. 2009. Andean land use and biodiversity: humanized landscapes in a time of change. Annals of the Missouri Botanical Garden 96:492–507.

Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2595/full