

Anthropogenic fire, vegetation structure and ethnobotanical uses in an alpine shrubland of Nepal's Himalaya

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Abstract. Alpine vegetation of the Himalaya is used as food, medicine or fodder, and is commonly managed with fire by agropastoralists. Prescribed fire can have positive effects on rangeland biodiversity, but studies evaluating its effects in alpine shrublands are scarce. Our objective was to examine the effects of anthropogenic fire on biophysical characteristics, species richness, abundance and composition in an alpine shrubland with socioeconomic value to local peoples in Langtang National Park in central Nepal. We surveyed biophysical variables, vascular plant species richness and composition along three transects at ascending elevations, and conducted interviews with local people and park officials on the use of fire in the region. We found 69 species of vascular plants in 89 plots; species richness was greater in burned plots and with increasing elevation, with 13 species unique to burned plots. We identified 14 indicator species in both burned and unburned plots; eight of them were Himalayan endemics. In burned plots, the indicator species were predominantly grasses and perennial forbs with ethnobotanical uses. This is the first detailed study on alpine shrubland anthropogenic fire in the Nepalese Himalaya. Burning may, at least temporarily, replace woody with more palatable herbaceous species, and weaken the elevational gradient of the shrubland.

Additional keywords: alpine pasture, endemic taxa, indicator species, species richness, transhumance.

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Introduction

Fire has historically imposed intense human-induced alterations in landscapes (Thomas and McAlpine 2010), but can also increase plant species richness (Fox 1981; Peterson and Reich 2007) and support livestock grazing by initiating the regeneration of tender and palatable grasses (Mark and Holdsworth 1979; Mark 1994). Fire is also a natural disturbance process that can have a positive role in rangeland and ecosystem management and biodiversity conservation (Carlson *et al.* 1993; Sheuyange *et al.* 2005; Brandt *et al.* 2013; Davies *et al.* 2014), influence the maintenance of community structure and function, and suppress as well as foster successional processes (Chapin and Van Cleve 1981; Chapin 1983; Shang *et al.* 2007; Barros *et al.* 2017). Variable fire severities and frequencies occur across landscapes depending on several biotic and abiotic factors, such as topography, wind speed, temperature, precipitation and fuel load, as well as combustion type (smouldering v. flaming), stand composition and developmental stage of plant species (White *et al.*

1996; Bigler *et al.* 2005; Bond and Keeley 2005; Collins *et al.* 2007; Harris and Taylor 2015). As a function of these factors, the burn patterns vary regionally or within one landscape, resulting in patches of burned and unburned areas with different shapes, sizes and severities (White 1979; Bond and Keeley 2005). Forest gaps created by fires naturally go through a dynamic process of shifting floristic composition initiated by competition for resources, mostly light, moisture and nutrients (Huston and Smith 1987). It can alter plant species composition and richness by exposing and changing soil properties and providing space for the establishment of pioneer and r-strategist type species, with shorter life span, such as, herbs and shrubs in the case of plants (Wesche 2006; Binelli *et al.* 2008; de Villiers and O'Connor 2011).

Absence of fire for many years in fire-dependent ecosystem leads to change in species composition. For example, suppression of fire for many decades in dry conifer forest in California has led to a dramatic increase in shade-tolerant and fire-intolerant

species, outcompeting shade-intolerant species that are also fire-tolerant woody species (Parsons and DeBenedetti 1979; Habeck 1994). Similarly, woody plant encroachment in semiarid and arid rangeland of Australia and Africa in the absence of fire is a common problem that has resulted in loss of biodiversity and affected livestock production (Watkinson and Ormerod 2001; Price and Morgan 2008; Archer 2010).

The alpine shrubland of Nepal is a region with rapidly changing climate that is likely to experience shifting vegetation structure (Gaur et al. 2003; Telwala et al. 2013; Salick et al. 2019), and is also managed with the use of anthropogenic fire to provide social and economic value to local peoples. Wildfires are common in Nepal in all physiographic zones during the dry period from March to May every year (Matin et al. 2017). Out of 30 220 fire hotspots recorded in Nepal between 2000 and 2013, 7283 (24%) occurred in alpine pastures during the hot and dry season (Parajuli et al. 2015). Approximately 50–58% of wildfires in Nepal are set deliberately by locals to enhance regeneration of grasses for pasture and hunting, clearing land for cultivation, and for firewood and non-timber forest product (NTFP) collection (Fig. 1a, b) (Karkee 1991; Bajracharya 2002; Matin et al. 2017). Along with their essential ecological roles, alpine shrublands contribute socioeconomic benefits to the local people. For example, people in the Himalaya use alpine *Rhododendron* and *Juniperus* species for incense, medicine and fuel, and for shelters for migratory pastoralists grazing their herds (Schmidt-Vogt 1990; Lama et al. 2001; Bhattacharyya 2011). The combined effects of human activities such as grazing, cutting (logging/looping) and trampling decrease the ecosystem's natural resiliency after fire (Folke et al. 2004). Thus, the ecosystem may become more vulnerable to subsequent impacts and the previous dominant vegetation communities may not return with similar diversity or composition.

Beginning approximately 50 years ago, the residents of Langtang National Park observed changes in the composition of vegetation due to fire (personal interviews with local agropastoralists, July 2011). Elders recall that the south-facing forests above Chandanbari and below Lauribina Hill were dominated by *Abies spectabilis* (D. Don) Spach (east Himalayan fir). Intense fire and the felling of trees for fuel wood and timber has since nearly cleared the forest stands on southern slopes and allowed the spread of shrub species such as *Piptanthus nepalensis* (Hook.) Sweet (evergreen laburnum) and *Berberis aristata* DC. (Indian barberry), and herbaceous species such as *Euphorbia wallichii* Hook. f. (Wallich spurge) and *Sambucus adnata* Wall. ex DC. (east Himalayan elderberry) (Fig. 1c, d). In high-elevation areas, *Rhododendron anthopogon* D. Don (dwarf rhododendron) and *R. setosum* D. Don (bristly rhododendron) shrubs form several dominant stands with lush mosses with occasional large *R. campanulatum* D. Don (bell rhododendron) and *Sorbus microphylla* (Wall. ex Hook. f.) Wenz. (small-leaf rowan) shrubs overtopping the smaller shrub stands. However, fire can fragment the continuous vegetation starting from the tree line up to the alpine rhododendron shrubland (Fig. 1e, f).

Globally, the body of research on the effects of fire in forest ecosystems of commercial value is extensive (Risser 1990; Williams et al. 1994; Bigler et al. 2005; Bond and Keeley 2005; Collins et al. 2007), but studies evaluating the effects of fire in alpine shrublands are relatively meagre (Knox and Clarke

2006) and non-existent in Nepal. Most fires above timberline in mountainous regions are severe (Wesche 2006; Williams et al. 2008), which tends to hold true in Himalayan shrublands dominated by aromatic plant species such as *Rhododendron* and *Juniperus*. Sclerophyllous shrublands are particularly susceptible to fire because they are typically dry, and may secrete flammable secondary chemicals (Christensen 1985). The low stature and single physiognomic type of shrublands commonly leads to intense crown fires (Christensen 1985). The alpine zone in the Nepal Himalaya belongs to the Western and Eastern Himalayan alpine shrub and meadow ecoregion (Olson et al. 2001). This region is known for having high species richness and supporting a large number of rare, endemic and threatened species (Shrestha and Joshi 1996; Basnet 2006), the majority of which are important from socioeconomic and cultural perspectives in addition to their conservation significance (Olsen and Larsen 2003; Ghimire et al. 2006, 2008; Salick et al. 2014). Patches of shrubs in alpine meadows are found facilitating the growth of grasses, forbs and many other important herbaceous species, either by providing suitable habitats or by protecting them from herbivores (Jacquez and Patten 1996; Körner 2003; Li et al. 2011). The mosaic – composed of sub-alpine and alpine meadows, shrublands, high-elevation agropastoral fields, forests and a large range of other habitats – is high-elevation Himalayan landscapes inscribed by human activities (Ghimire et al. 2006). Their biodiversity is shaped by the interaction among geological, climatic and topographical factors, cultural traditions and modern land-use impacts.

Studies in the Himalaya are important for understanding how traditional ecological knowledge and practices related to pastoralism are influencing and interacting with alpine shrublands, and how social and climatic developments will impact alpine zone ecology and the socioeconomic futures of the local people. The biodiverse alpine environments in the Himalaya are among the habitats experiencing the most drastic global climate change, with increasing temperature, a heavier and more unpredictable rainfall pattern, and rapidly melting permanent snows and glaciers. Research in the Himalaya suggests that the warming climate has already caused alpine plants and their habitats to shift upslope towards higher elevations (Gaur et al. 2003; Telwala et al. 2013; Salick et al. 2019). Such shifts are continually changing the communities of high-elevation regions, outcompeting threatened and endemic plants and eventually pushing the vital alpine life zones to extinction as they reach ridge tops. Climate change that threatens alpine plants also affects the traditional practices and livelihood of both indigenous peoples and massive downstream populations (Salick and Byg 2007; Salick et al. 2014).

The objective of the present study was to examine the effects of anthropogenic fire on biophysical variables, species richness, abundance and composition in an alpine shrubland with social and economic value to local peoples in Langtang National Park in the northern region of central Nepal. Specific research questions included: (1) do biophysical factors vary between burned and unburned plots; (2) do plant diversity, frequency and composition vary among burned and unburned plots; and (3) are species used by locals as NTFPs enhanced by anthropogenic burning? To address our objective and research questions, we sampled plots across a narrow elevational gradient of alpine



Fig. 1. In Lauribina Danda's alpine zone (>3900 m), besides livestock grazing, local people use these shrublands for extracting different resources, including harvesting of medicinal plants (a), and firewood collection (b). Intense fire and the felling of trees for fuel wood and timber in the Chandanbari area (3200 m above sea level (a.s.l.)), below Lauribina Danda in Langtang National Park has nearly cleared the forest stands on southern slopes (c), and allowed the spread of woody and herbaceous species such as *Piptanthus nepalensis* (Hook.) Sweet (d). Fire has fragmented the continuous vegetation starting from the forest line up to the alpine rhododendron shrubland (e), leaving a mosaic of fire-affected and unaffected shrub patches (f).

shrubland that burned in 2009 from anthropogenic causes, and interviewed local agropastoralists with first-hand knowledge of the ethnobotanical importance of species within the alpine zone.

Methods

Study area

The study area (28°05.371' to 28°05.660'N longitude and 85°23.337' to 85°23.517'E latitude) is located on Lauribina

Danda (danda is hill in Nepali) in the lower alpine zone of Langtang National Park (LNP) in the northern part of central Nepal (Fig. 2). LNP covers subtropical to alpine climatic conditions owing to high elevational variation. The northern aspect is cool and moist while the southern aspect is warmer and drier. LNP receives an annual precipitation of 650 mm (Langtang station, 3920 m above sea level (a.s.l.)) to 1800 mm (Dhunchu station, 1950 m a.s.l.). Most of the precipitation occurs during the summer monsoon season, which lasts from June until the

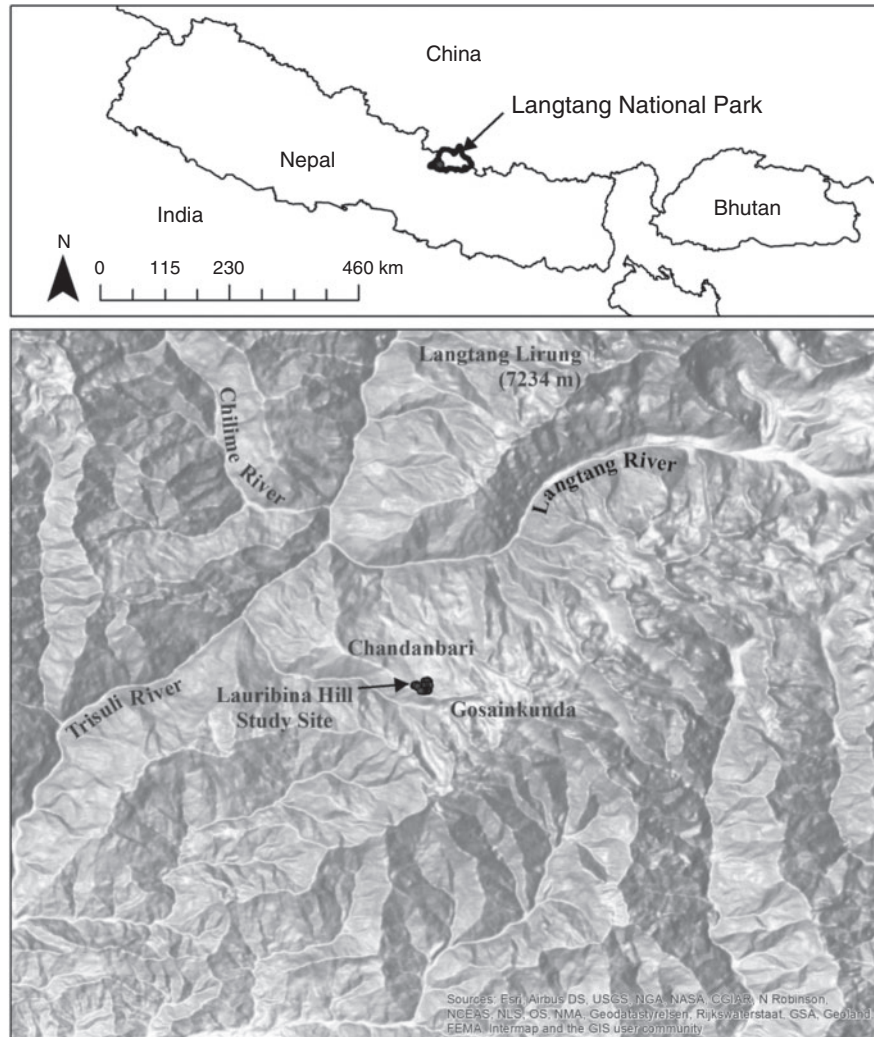


Fig. 2. Map of Nepal showing the study area on Lauribina Danda, which is located in the lower alpine zone of Langtang National Park (LNP) in the northern part of central Nepal at $28^{\circ}05.371' - 28^{\circ}05.660'N$ longitude and $85^{\circ}23.337' - 85^{\circ}23.517'E$ latitude.

beginning of October. Average maximum temperature was $23^{\circ}C$ in June and average minimum temperature was $\sim 2^{\circ}C$ in January.

The study area has $20-35^{\circ}$ slopes with aspects ranging from 310° north-west to 30° north-east. Because sampling covered a small geographical area with plots only in the northern aspect with almost constant slope to minimise the effect of topographic variation, it is unlikely there was a difference in the amount of radiation received in burned and unburned patches. The study area was above the tree line, with a mosaic of habitats dominated by grassland and shrubland. The tree line varies within 3900–4000 m a.s.l., with *Abies spectabilis* and *Betula utilis* D. Don (Himalayan birch) in the overstorey and *Rhododendron campanulatum* and *Sorbus microphylla* in the understorey. The latter two species reach up to 4200 m a.s.l. Above forest line in lower reaches of the alpine zone ($>3900-4200$ m a.s.l.), the vegetation comprises vast stretches of shrubland dominated by *Rhododendron anthopogon*, *R. lepidotum* Wall. ex G. Don (pink scaly

rhododendron) and *R. setosum* on the northern aspects. Other woody species in the lower alpine belt include *Rhododendron campanulatum*, *Juniperus recurva* Buch.-Ham. ex D. Don (Himalayan juniper), *Berberis* spp., *Ephedra gerardiana* Wall. ex Stapf (Gerard jointfir), *Spiraea arcuata* Hook. f. (arching spirea), *Salix* spp. and *Potentilla fruticosa* L. (shrubby cinquefoil) (Chaudhary 1998; SK Ghimire, S Thapa-Magar, MR Shrestha, B Devkota, MR Gubhaju 2008, unpubl. data).

The alpine grassland and shrubland in the study area are used by local agropastoralists for grazing livestock and collecting plants for local use. There were altogether three alpine shelters of stone construction (*goth* in Nepali) in the Lauribina area, which are used during the summer grazing season for shelter and rest for pastoralists, and keeping livestock temporarily. The herders traditionally managed the pastoral land through rotational grazing of livestock, and seasonal burning of vegetation (Karki and McVeigh 2000). As the study area was included under the protected area system in 1976, any unauthorised

anthropogenic fire and illegal or haphazard harvesting of natural resources are subject to a certain degree of regulation. However, the national park acknowledges the traditional practice of subsistence use of natural resources. In interviews, local agropastoralists stated that the Lauribina area has received random and infrequent fire to enhance the growth of palatable species since the establishment of the national park.

Vegetation sampling and data collection

Sampling took place within the alpine rhododendron shrubland of Lauribina Danda, in June 2011, 2 years after an ~ 100 ha fire in 2009. Within the perimeter of the fire, a patchy mosaic was created with high-severity fire killing the majority of above-ground biomass in the burned matrix, but with numerous unburned patches completely escaping fire and absent of fire scars (mean unburned patch size 2893 m^2) (see burn mosaic Fig. 1f). The timing of the fire was determined via field observations in 2009, through subsequent interviews with local agropastoralists ($n = 10$) from the surrounding villages, and was verified on NASA satellite imagery (NASA 2009).

Three parallel transects (each 500–700 m in length) were established paralleling the slope contour: lower summit transect (T1) at 3900 m a.s.l., mid-summit transect (T2) at 4000 m a.s.l., and upper summit transect (T3) at 4100 m a.s.l. In addition to positioning perpendicular to the elevational gradient, the transects were concomitantly perpendicular to the dominant shrub biomass gradient, with the girth (cm) of burned stumps declining with increasing elevation (girth mean \pm s.d.: T1 = 25.80 ± 19.65 , T2 = 6.37 ± 1.08 , T3 = 3.42 ± 0.65 cm).

Each of the three transects was divided into three segments in burned matrix and three segments in unburned shrub patches. Within each segment, plots were systematically placed starting ~ 5 m from the matrix or patch margin and extending to ~ 5 m from the next matrix or patch margin, maintaining a 20–30-m distance between successive plots. A total of 89 plots were sampled, 49 burned and 40 unburned (T1 = 17 burned, 14 unburned plots; T2 = 16 burned, 14 unburned plots; T3 = 16 burned, 12 unburned plots). At each plot, we recorded geographical position (latitude and longitude) using a global positioning system, soil pH using a pH meter, and soil moisture with a moisture-reading electrode (Takemura Electric Works DM-15 soil tester).

In each plot, we sampled vegetation in three 1-m^2 quadrats oriented diagonally, which were further divided into four (0.5×0.5 m) subquadrats. In total, 267 quadrats (1 m^2) and 1068 subquadrats (0.5 m^2) were sampled across all 89 plots. Within each subquadrat, the team recorded presence or absence of all vascular plant species. If a species was present in all four subquadrats, it was assigned a categorical abundance value of 4. If a species was present in three out of four subquadrats, it was assigned a categorical abundance value of 3, and so on. Vegetation data from quadrats was pooled by plot for analysis purposes. We identified as many species in the field as possible following published resources (Polunin and Stainton 1984; Ghimire *et al.* 2008) and authors' personal expertise on alpine plants. Botanical vouchers were deposited at Tribhuvan University Central Herbarium, Nepal, where field identifications were later confirmed. In each quadrat, we also recorded the percentage cover of each of the following biophysical variables: exposed soil,

rocks, dead wood, litter, graminoids (including grasses, sedges and rushes), forbs, shrubs, trees, mosses and lichens, as well as the total number and girth of burned stumps.

We interviewed local agropastoralists ($n = 10$) about the use of plant species recorded in the study area and their palatability for herbivores. We specifically asked whether specific sampled plants are used by local residents, and if used, for what purpose. We also reviewed ethnobotanical literature (for example, Lama *et al.* 2001; Manandhar 2002; SK Ghimire, S Thapa-Magar, MR Shrestha, B Devkota, MR Gubhaju 2008, unpubl. data) to verify the use of the plant from other regions. In the case of species consumed by livestock, we asked respondents to rank the palatability. Following Daalkhaijav (2005), we categorised the palatability into preferred, desirable, consumed but less desirable, not consumable and toxic.

Both verbal and written consents were obtained for the study. Written permission for fieldwork was obtained from the authorities at LNP and Buffer Zone, and at the Department of National Park and Wildlife Conservation, Government of Nepal. Prior verbal informed consents were obtained from the local communities in Chandabari and Lauribina within LNP before establishing participation and consultation of local agropastoralists.

Data analysis

Mann–Whitney U Tests (non-parametric) were conducted using SPSS v.17 (SPSS Inc. Released 2008. SPSS Statistics for Windows, Version 17.0) to compare biophysical data between unburned and burned plots, because the biophysical data did not meet the assumptions of parametric tests even after transformation.

Species richness, α -diversity or the number of vascular plant species per unit area, was calculated for each plot. A second measure, γ -diversity or the number of vascular plant species in unburned patches and the burned matrix, was also calculated. We used two-way ANOVA in Stata v.15 to compare vascular plant species richness at the plot scale between unburned and burned plots and among the three transects (i.e. upper, mid and lower elevations) and their interactions. Post-hoc Tukey Honest Significant Difference tests were conducted to compare pairwise differences among burned and unburned plots within and among transects.

Prior to running both the multiresponse permutation procedure (MRPP) and two-way cluster analyses, species frequency in each plot was square-root-transformed and species found in less than 5% of the plots were discarded. We used MRPP (Biondini *et al.* 1988; McCune *et al.* 2002; Cai 2006) to examine species compositional differences among burned and unburned plots and transects. We performed MRPP on species frequency in each plot for combined datasets, and burned and unburned plots separately and transects separately. MRPP was performed using PC-ORD v.7 (McCune and Mefford 1999) using the Sorensen distance measure.

Two-way cluster analysis was run using PC-ORD v.7 (McCune and Mefford 1999). For the cluster analysis, species frequency was relativised by the column maximum, the distance measure was Sorensen, and a flexible β linkage method with a value of -0.25 was selected.

The presence and abundance of key indicator plant species (Dai *et al.* 2006) are biological characters of groups of sites

Table 1. Biophysical variables recorded in unburned and burned plots
Reporting mean \pm s.e. P -value is based on Mann–Whitney U test, indicating that the medians are significantly different between unburned and burned plots

| Variables | Unburned | Burned | P -value |
|----------------------------|------------------|------------------|------------|
| Soil pH | 5.65 \pm 0.03 | 6.84 \pm 0.01 | <0.001 |
| Soil moisture ^A | 7.97 \pm 0.03 | 4.77 \pm 0.14 | <0.001 |
| Exposed soil cover (%) | 0.82 \pm 0.21 | 2.06 \pm 0.37 | 0.005 |
| Rock cover (%) | 1.96 \pm 0.57 | 3.05 \pm 0.60 | 0.029 |
| Litter cover (%) | 21.04 \pm 2.08 | 23.77 \pm 1.70 | 0.179 |
| Dead wood cover (%) | 2.47 \pm 0.63 | 15.95 \pm 1.00 | <0.001 |
| Number of burned stumps | 0.08 \pm 0.08 | 75.84 \pm 3.55 | <0.001 |
| Moss + lichen cover (%) | 56.24 \pm 3.69 | 32.69 \pm 1.37 | <0.001 |
| Tree cover (%) | 8.24 \pm 2.80 | 0.10 \pm 0.10 | <0.001 |
| Shrub cover (%) | 40.56 \pm 4.39 | 2.54 \pm 0.91 | <0.001 |
| Forb cover (%) | 9.50 \pm 1.67 | 7.84 \pm 0.78 | 0.438 |
| Graminoid cover (%) | 7.97 \pm 1.89 | 14.65 \pm 1.22 | <0.001 |

^ASoil moisture was recorded as a categorical variable, the value of which ranged from 0 (dry) to 8 (moist).

representing habitat types or combinations of habitat types and are of prime interest for ecosystem conservation and management (Legendre and De Cáceres 2013). Thus, we used the indicator value method (IVM) (Dufrene and Legendre 1997) to determine the indicator species for burned and unburned plots within the three transects. Those species with high indicator values are the indicator species (McCune et al. 2002; Dai et al. 2006). We used PC-ORD v. 7 (McCune and Mefford 1999) and a Monte Carlo test with 4999 permutations (McCune and Mefford 1999) to test the statistical significance of indicator values (IV).

Results

Biophysical variables

The unburned and burned plots significantly differed for 10 out of 12 variables sampled (Table 1). Unburned plots showed significantly greater cover values for tree (8.24 v. 0.10%), shrub (40.56 v. 2.54%) and moss–lichen (56.24 v. 32.69%), with *Rhododendron campanulatum*, *R. anthopogon*, *R. setosum* and *R. lepidotum* contributing the main shrub cover. Burned plots showed significantly greater cover values for graminoids (14.65 v. 7.97%), dead wood (15.95 v. 2.47%) and exposed soil (2.06 v. 0.82%) compared with unburned plots. Unburned and burned plots also differed in terms of edaphic properties. Soil in burned plots had higher pH (6.84 v. 5.65) and lower moisture (4.77 v. 7.97) compared with unburned plots (Table 1).

Species richness

Total vascular plant species richness (γ -diversity) varied from 55 species in unburned patches to 62 species in the burned matrix (altogether, 69 species; see Supplementary material Table S1 for the list of species recorded, their elevation range and local use). At the plot level, the two-way ANOVA resulted in an $R^2 = 0.585$, and fire (F -value = 8.97, $P = 0.004$) and transect (F -value = 55.98, $P < 0.001$) both had significant effects on species richness, while the interaction effect was not significant (F -value = 2.49, $P = 0.089$). The mean vascular plant species richness was significantly greater in the burned

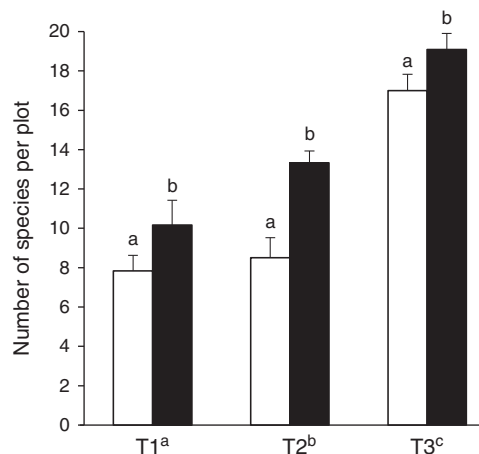


Fig. 3. ANOVA tests for mean vascular plant species richness variation among three transects in burned (black bars) and unburned (white bars) plots and among the three transects: T1 = lower transect (3900 m above sea level (a.s.l.)), T2 = mid transect (4000 m a.s.l.), and T3 = upper transect (4200 m a.s.l.). Different letters at the tops of bars represent a significant difference in vascular plant species richness between burned and unburned plots within transects, and different letter superscripts on transect labels on the x-axis represent significant differences among transects, at the $P = 0.05$ level.

than in the unburned plots at all three transect elevations ($P < 0.05$, Fig. 3). The species richness significantly increased along the elevational gradient from lower to upper transect, indicating that elevation was a statistically significant variable affecting vascular plant species richness (Fig. 3).

Species composition

The MRPP showed statistically significant compositional differences between burned and unburned plots ($P < 0.001$) and also between transects ($P < 0.001$) (Table 2). Multiple pairwise comparisons showed that burned plots in all three transects and unburned plots in Transects 1 and 3 and Transects 2 and 3 had significant compositional differences ($P < 0.001$ for all pairwise comparisons). In unburned plots, Transects 1 and 2 were broadly overlapping and thus the hypothesis of no difference between groups could not be rejected. In addition, within each of the three transects, burned and unburned plots had significantly differing compositions ($P < 0.001$ in T1 and T2, $P < 0.05$ in T3).

The two-way cluster analysis exhibited distinct clustering by transect (Fig. 4). There were two major groups of plots, one completely composed of plots from Transects 1 and 2 (regular and inverted triangles), and one that included all plots from Transect 3 (boxes), 10 plots from Transect 2, and 3 plots from Transect 1. Within the group composed of Transects 1 and 2, burned and unburned plots were largely separated into different subgroups, with the majority of burned plots coming from Transect 1. In addition, the 13 plots from Transects 1 and 2 that were grouped with the plots from Transect 3 were predominantly burned plots from Transect 2. These burned plots from Transect 2 continued to support *Rhododendron anthopogon*, as root sprouts, and had an equivalent number of burned stumps (mean 74.7) to all burned plots together (Table 1). Thus, before fire, plots in Transect 2 were dominated by large shrubs-like plots in Transect 1.

Table 2. Results of multiresponse permutation procedure

Showing the composition difference between burned and unburned plots and transects. The statistics shown are delta (weighted mean within group distance), test statistic T (which describes the separation between the groups), P -value associated with T (determined by numerical integration of the Pearson type III distribution) and agreement statistic A (chance-corrected within group agreement)

| Plot comparisons | Observed delta (δ) | δ under null hypothesis | | | T^A | P^A | A^A |
|------------------------|-----------------------------|--------------------------------|----------|----------|---------|--------|-------|
| | | Expected | Variance | Skewness | | | |
| Burned v. unburned | 0.602 | 0.627 | 0.0001 | -1.536 | -11.249 | <0.001 | 0.039 |
| Transects (T1–T3) | 0.546 | 0.627 | 0.0001 | -1.076 | -25.845 | <0.001 | 0.129 |
| Unburned transects | 0.546 | 0.648 | 0.0006 | -1.000 | -12.892 | <0.001 | 0.157 |
| T1 v. T2 | | | | | -0.555 | 0.233 | 0.006 |
| T1 v. T3 | | | | | -13.243 | <0.001 | 0.201 |
| T2 v. T3 | | | | | -12.098 | <0.001 | 0.169 |
| Burned transects | 0.470 | 0.565 | 0.0003 | -1.159 | -16.780 | <0.001 | 0.167 |
| T1 v. T2 | | | | | -10.671 | <0.001 | 0.109 |
| T1 v. T3 | | | | | -15.158 | <0.001 | 0.186 |
| T2 v. T3 | | | | | -8.933 | <0.001 | 0.087 |
| T1: Burned v. unburned | 0.536 | 0.582 | 0.0003 | -1.031 | -7.941 | <0.001 | 0.079 |
| T2: Burned v. unburned | 0.528 | 0.589 | 0.0004 | -1.421 | -9.488 | <0.001 | 0.105 |
| T3: Burned v. unburned | 0.445 | 0.459 | 0.0003 | -1.677 | -2.585 | 0.026 | 0.032 |

^AThe more negative T , the stronger the separation. The P -value evaluates the probability that an observed difference is due to chance. A is the effect size that is independent of the sample size. A describes within-group homogeneity, compared with the random expectation. When all items are identical within groups, then the observed delta = 0 and $A = 1$, the highest possible value for A . If heterogeneity within groups equals expectation by chance, then $A = 0$ (McCune *et al.* 2002).

Indicator species

Out of 69 vascular plant species recorded in this study, 46 species were common for both burned matrix and unburned patches. The number of unique species (i.e. the species recorded only either in burned or unburned plots) was higher in burned ($n = 13$) than in unburned ($n = 7$) plots (Table 3). Several highly palatable species: *Astragalus donianus* DC. (dark-red milkvetch), *Poa* sp., *Heracleum nepalense* D. Don (Nepal cowparsnip), *Lloydia serotina* (L.) Rchb (mountain spiderwort), *Maianthemum purpureum* (Wall.) LaFrankie (purple mayflower), *Rubus nepalensis* (Hook. f.) Kuntze (Himalayan creeping bramble), and some important medicinal plant species, e.g. *Gentiana depressa* D. Don, *Nardostachys jatamansi* (D. Don) DC. (jatamansi) and *Rheum acuminatum* Hook. f. and Thomson (Himalayan rhubarb), were found only in the open burned matrix created by fire (Table 3). Similarly, unburned plots supported unique species like *Anemone demissa* Hook. f. and Thomson (floppy Himalayan anemone), *Geranium donianum* Sweet (Don's geranium), and *Roscoeia alpina* Royle (mountain roscoe lily), which were absent from the burned habitat and also had important medicinal value (Table 3). *Sorbus microphylla* was the only palatable species unique to unburned plots, while the other unique species were undesirable or not consumed. Himalayan and Pan-Himalayan endemics represented 15 of the 20 identified unique species (Table 3), and the majority of indicator species were also dominated by these endemics (Table 4). In total, 14 plant species were identified as indicators for burned and 14 species for unburned plots ($P < 0.05$, based on Monte Carlo test; Table 4).

The most substantial variation among burned and unburned plots was found with Transect 2. Large shrubs *Rhododendron campanulatum* and *Sorbus microphylla* were indicators of unburned plots in Transect 2, but a forb and a grass were indicators of Transect 2 in burned plots. Transects 1 and 3 were

generally represented by indicators that were forbs, ferns and dwarf woody species in both burned and unburned plots. Transect 3 indicators of burned plots included several species that are recognised indicators of disturbance due to overgrazing, e.g. *Anemone rupestris* Wall. ex Hook. f. and Thomson (blue rock anemone), *Anemone smithiana* Lauener & Panigrahi, and *Euphorbia stracheyi* Boiss. (Himalayan prostrate spurge) (Bauer 1990; Ghimire *et al.* 2006).

Discussion

Anthropogenic burning in alpine shrublands may be sustainable as a temporally and spatially dynamic process, providing fodder, food, fuel, medicine and religiously significant resources to practitioners of burning. Agropastoralists of the Nepalese Himalaya have used fire in alpine pastures purposefully for social and economic reasons, i.e. for the regeneration of grasses and for the promotion of grazing, for an unknown number of generations. Although prescribed fire used as a management tool has been studied in Nepal's subtropical grasslands, where grazing is prohibited (e.g. Peet *et al.* 1999), this is the first detailed study of anthropogenic fire in Himalayan alpine shrublands, where burning and grazing are common disturbances linked to the livelihood of the local people.

The number of unique species was greater in burned than in unburned plots, and most of the unique species were Himalayan or Pan-Himalayan endemics. Some of the species unique to burned plots were rare and threatened forbs, such as *Nardostachys jatamansi* (International Union for Conservation of Nature, IUCN Redlist critically endangered) and *Rheum acuminatum*, which have substantial local use and medicinal values (Manandhar 2002; SK Ghimire, S Thapa-Magar, MR Shrestha, B Devkota, MR Gubhaju 2008, unpubl. data). Some wider-ranging species, such as *Juncus thomsonii* found across central Asia, were also present only in burned habitat. Earlier studies

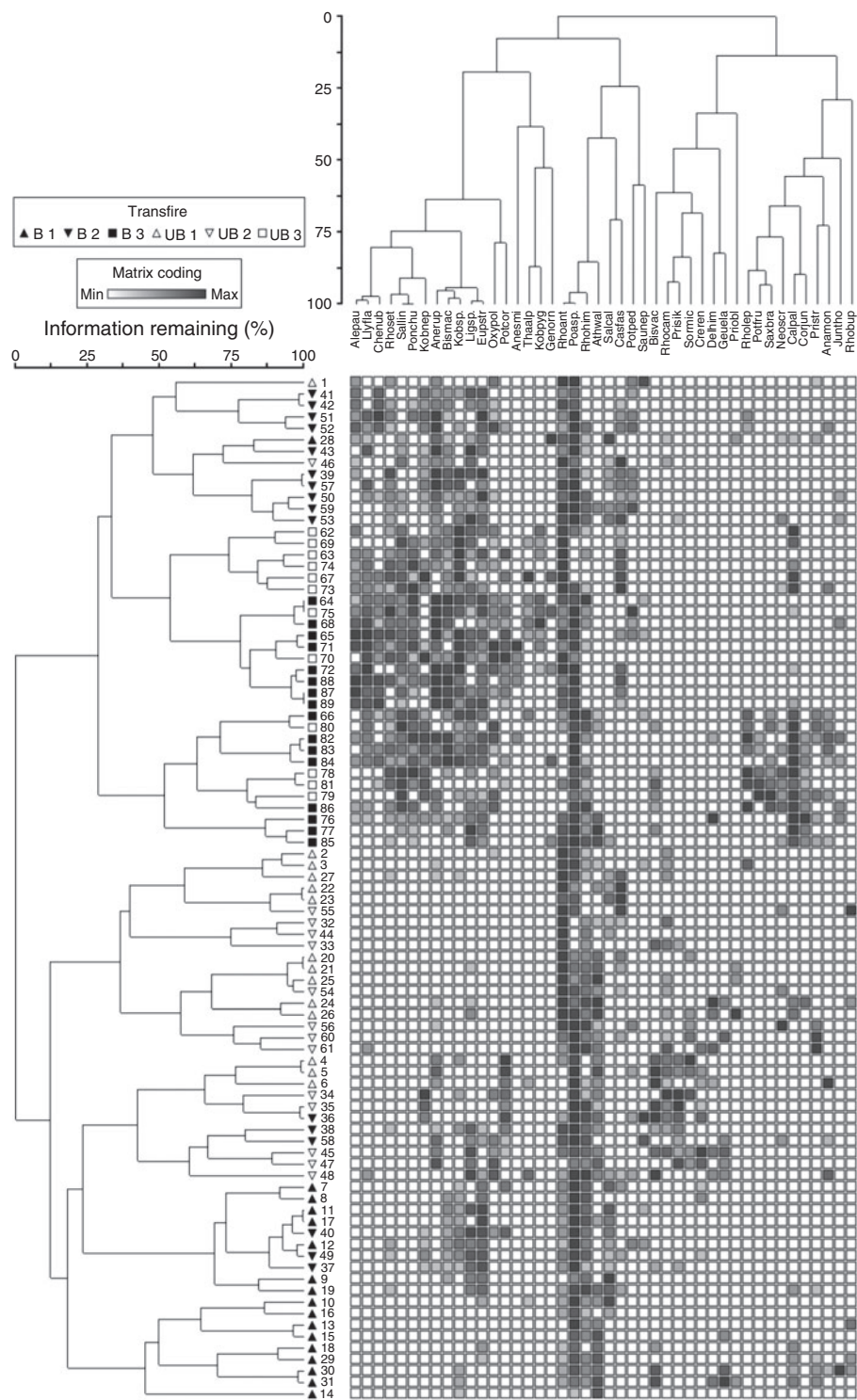


Fig. 4. Two-way cluster analysis of plots (vertical dendrogram) and species (horizontal dendrogram). Matrix shading is based on relative frequency of each species in each plot, with darker shading representing increasing frequency. Numbers on the plot dendrogram are plot numbers and symbols signify burn status (burned, filled symbols; unburned, open symbols) and transect (T1, triangle; T2, inverted triangle; T3, box). Species abbreviations are provided in Table S2.

Table 3. Plant species unique to unburned and burned plots

Including their palatability, major ethnobotanical use and chorotype. P, preferred; U, consumed but undesirable; N, not consumable; CA, central Asiatic; HE, Himalayan endemic; HO, Holarctic; PH, Pan-Himalayan; –, none

| Species | Palatability ^A | Major ethnobotanical use ^B | Chorotype ^C |
|---|---------------------------|---------------------------------------|------------------------|
| Burned | | | |
| <i>Arenaria bryophylla</i> Fernald (Caryophyllaceae) | N | Medicinal | PH |
| <i>Astragalus donianus</i> DC. (Fabaceae) | P | Medicinal | HE |
| <i>Epilobium wallichianum</i> Hausskn. (Onagraceae) | U | Medicinal | PH |
| <i>Gentiana depressa</i> D. Don (Gentianaceae) | U | Medicinal | HE |
| <i>Heracleum nepalense</i> D. Don (Apiaceae) | P | Fodder, food and medicinal | PH |
| <i>Juncus thomsonii</i> Buchenau (Juncaceae) | U | – | CA |
| <i>Lloydia serotina</i> (L.) Rchb. (Liliaceae) | P | Medicinal | HO |
| <i>Maianthemum purpureum</i> (Wall.) LaFrankie (Asparagaceae) | P | Food and medicinal | PH |
| <i>Nardostachys jatamansi</i> (D. Don) DC. (Caprifoliaceae) | N | Medicinal and religious | PH |
| <i>Pedicularis elwesii</i> Hook. F. (Orobanchaceae) | U | – | PH |
| <i>Poa</i> sp. (Poaceae) | P | – | – |
| <i>Rheum acuminatum</i> Hook. F. and Thomson (Polygonaceae) | U | Food and medicinal | PH |
| <i>Rubus nepalensis</i> (Hook. F.) Kuntze (Rosaceae) | P | Food | HE |
| Unburned | | | |
| <i>Anemone demissa</i> Hook. F. and Thomson (Ranunculaceae) | N | Medicinal | PH |
| <i>Eritrichium</i> sp. (Boraginaceae) | N | – | – |
| <i>Geranium donianum</i> Sweet (Geraniaceae) | U | Medicinal | PH |
| <i>Roscoea alpina</i> Royle (Zingiberaceae) | U | Medicinal | HE |
| <i>Sorbus microphylla</i> (Wall. Ex Hook.f.) Wenz. (Rosaceae) | P | Fodder and fuel | PH |
| <i>Thalictrum cultratum</i> Wall. (Ranunculaceae) | U | Medicinal | PH |
| <i>Viola biflora</i> L. (Violaceae) | U | Medicinal | HO |

^APalatability for each species presented here is based on the interviews with herders and local people (details provided in Table S1).

^BMajor ethnobotanical uses: the uses presented in the table are based on present study interviews with herders and local people and references consulted are provided in Tables S1 and S2.

^CChorotype determination is based on species geographical distribution as detailed in Press *et al.* (2000) and Wu *et al.* (1994).

have shown that rare and threatened plant species may be favoured, to some extent, by intermediate burning (Dudley and Lajtha 1993; Van Lear *et al.* 2005). A study evaluating the detailed demographic effects of fire on rare and threatened plant species should be completed in the Himalayan alpine ecosystem to inform best management practices for conservation of species and continued socioeconomic benefit to local agropastoralists.

Many studies have shown that fire positively influences species diversity by maximising the coexistence of numerous species owing to the removal of competing woody biomass and making habitat suitable for forbs, grasses and fire-adapted perennial species (Reilly *et al.* 2006; Twidwell *et al.* 2012; Bowles and Jones 2013). Fire is one of the dominant ecological factors in ericaceous shrublands (Allen *et al.* 1996), and in *Rhododendron* shrublands, fire is favoured by its flammable secondary metabolites, the presence of dense woolly indumentum on leaves (Ng and Corlett 2003; Innocenti *et al.* 2010; Paul *et al.* 2010; Guleria *et al.* 2011), a lush thicket of mature stumps and good-quality fuels. The results of the MRPP and two-way cluster analysis demonstrate that the effects of burning can, at least temporarily, foster a change in habitat composition from one dominated by shrubland species to one where grassland species may thrive. The Lauribina Danda fire created a mosaic of high-severity matrix and unburned patches, converting a homogeneous shrubland to a more heterogeneous community of mixed grassland and shrubland.

There appears to be an interaction between elevation and fire in changes in community composition. Stronger variation in composition and species richness was observed between burned and unburned plots in the mid-elevation Transect 2 compared with the lowest-elevation Transect 1 and highest-elevation Transect 3. The IV analysis demonstrated a change at mid-elevation, at least temporarily, in vegetation composition from shrubland to a grassland similar to higher-elevation alpine grasslands, i.e. large *R. campanulatum* and *Sorbus microphylla* shrub dominance in unburned areas compared with grass, forb and dwarf shrub dominance in burned areas of Transect 2. In addition, in the cluster analysis, unburned plots from Transect 2 grouped nearly uniformly with plots from Transect 1 (also unburned plots did not differ between Transects 1 and 2 with the MRPP analysis), but burned plots from Transect 2 were much more likely to be grouped with plots from Transect 3. The pattern in Transect 2 is similar to results reported from other regions (Walker 2001; Sheuyange *et al.* 2005), where herbaceous and graminoid species are favoured by burning and shrub cover is temporarily reduced (Sheuyange *et al.* 2005).

The dominant shrubland species were not extirpated, and a longer monitoring period is required, but our data indicate that the community likely possesses the capacity to respond resiliently to disturbance and may not permanently shift away from the pre-burn vegetation structure. Although their aboveground biomass was either mineralised or harvested, the shrubs readily sprouted from the remaining rootstock and will likely regain canopy dominance in a cyclical pattern shifting across the alpine shrubland zone as mature patches are burned, utilised and left fallow. Because *Rhododendron* shrubs are highly valued as medicinal and aromatic plants, their resprouting and regrowth

Table 4. Indicator plant species

Data shown are Monte Carlo test of significance of observed maximum indicator value (IV) for vascular plant species in burned and unburned plots; the means and standard deviations of IV from the permutations; and P -values for the hypothesis of no difference between groups. The P -value is based on the proportion of randomised trials (with 4999 permutations) with indicator value equal to or exceeding the observed indicator value. Also including species palatability, major ethnobotanical use and chorotype. P, preferred; D, desirable; U, consumed but undesirable; N, not consumable; CA, central Asiatic; EA, eastern Asiatic; HE, Himalayan endemic; PH, Pan-Himalayan; —, none

| Burn status, transect | Indicator species | Species abbreviation | Observed indicator value (IV) | Mean | s.d. | P^A | Palatability ^B | Major ethnobotanical use ^C | Chorotype ^D |
|-----------------------|-------------------|--|-------------------------------|------|------|--------|---------------------------|---------------------------------------|------------------------|
| Burned | T1 | <i>Salix calyculata</i> Hook. f. ex Andersson (Salicaceae) | 25 | 11.4 | 3.58 | 0.0058 | N | Medicinal | HE |
| | T2 | <i>Poa</i> sp. (Poaceae) | 19.9 | 18.3 | 0.79 | 0.0012 | P | — | — |
| | | <i>Potentilla peduncularis</i> D. Don (Rosaceae) | 23.6 | 9.6 | 3.65 | 0.006 | N | Medicinal | PH |
| | T3 | <i>Aletris pauciflora</i> (Klotzsch) Hand.-Mazz. (Nartheciaceae) | 36.4 | 12 | 3.65 | 0.0004 | D | Medicinal | PH |
| | | <i>Anemone rupestris</i> Wall. ex Hook. f. and Thomson (Ranunculaceae) | 26.7 | 15.3 | 3.18 | 0.0042 | P | — | PH |
| | | <i>Anemone smithiana</i> Lauener and Panigrahi (Ranunculaceae) | 19.5 | 6.7 | 3.61 | 0.0102 | — | — | PH |
| | | <i>Bistorta macrophylla</i> (D. Don) Soják (Polygonaceae) | 33.3 | 14.4 | 3.31 | 0.0004 | U | Medicinal | PH |
| | | <i>Chesneya nubigena</i> (D. Don) Ali (Fabaceae) | 30.9 | 10.6 | 3.66 | 0.0006 | U | Food | PH |
| | | <i>Corydalis juncea</i> Wall. (Papaveraceae) | 27.7 | 8.4 | 3.65 | 0.002 | U | — | HE |
| | | <i>Euphorbia stracheyi</i> Boiss. (Euphorbiaceae) | 32 | 15.9 | 2.83 | 0.0002 | U | Medicinal | PH |
| | | <i>Juncus thomsonii</i> Buchenau (Juncaceae) | 19.3 | 7.6 | 3.71 | 0.0164 | U | — | CA |
| | | <i>Ligusticum</i> sp.L. (Apiaceae) | 23.6 | 15.5 | 3.02 | 0.0168 | — | — | — |
| | | <i>Lloydia flavonutans</i> H. Hara (Liliaceae) | 41.5 | 12.4 | 3.51 | 0.0002 | U | — | HE |
| | | <i>Ponerorchis chusua</i> (D. Don) Soó (Orchidaceae) | 45.3 | 12.1 | 3.5 | 0.0002 | P | Medicinal | EA |
| Unburned | T1 | <i>Athyrium wallichianum</i> Ching (Athuriaceae) | 22.4 | 14.1 | 3.34 | 0.0218 | U | Food and medicinal | PH |
| | | <i>Primula obliqua</i> W.W. Sm. (Primulaceae) | 15.2 | 6.6 | 3.65 | 0.0254 | N | — | HE |
| | T2 | <i>Primula sikkimensis</i> Hook. (Primulaceae) | 18.3 | 8.8 | 3.75 | 0.025 | N | Medicinal | PH |
| | | <i>Rhododendron campanulatum</i> D. Don (Ericaceae) | 33.8 | 10.6 | 3.68 | 0.0004 | N | Medicinal | HE |
| | | <i>Sorbus microphylla</i> (Wall. ex Hook.f.) Wenz. (Rosaceae) | 19 | 7.9 | 3.74 | 0.014 | P | Fodder and Fuel | PH |
| | T3 | <i>Caltha palustris</i> var. <i>himalica</i> Tamura (Ranunculaceae) | 39 | 12.2 | 3.47 | 0.0002 | U | Medicinal | HE |
| | | <i>Kobresia nepalensis</i> (Nees) Kük. (Cyperaceae) | 30.2 | 13.2 | 3.51 | 0.0018 | P | — | PH |
| | | <i>Kobresia pygmaea</i> (C.B. Clarke) C.B. Clarke (Cyperaceae) | 16.9 | 7.9 | 3.79 | 0.0352 | P | — | EA |
| | | <i>Kobresia</i> sp. (Cyperaceae) | 29.3 | 15.1 | 3.17 | 0.0006 | P | — | — |
| | | <i>Oxygraphis polypetala</i> (Raf.) Hook. f. and Thomson (Ranunculaceae) | 23.4 | 12.3 | 3.58 | 0.011 | P | — | HE |
| | | <i>Potentilla fruticosa</i> var. <i>arbuscula</i> (D. Don) Maxim. (Rosaceae) | 20.3 | 6.6 | 3.63 | 0.0074 | N | Medicinal; Religious | PH |
| | | <i>Rhododendron setosum</i> D. Don (Ericaceae) | 33.3 | 13.8 | 3.41 | 0.0002 | N | Food; Religious | HE |
| | | <i>Salix lindleyana</i> Wall. ex Andersson (Salicaceae) | 52.8 | 12.9 | 3.38 | 0.0002 | U | Medicinal | PH |
| | | <i>Saxifraga brachypoda</i> D. Don (Saxifragaceae) | 22.5 | 7.2 | 3.62 | 0.0052 | U | — | PH |

^AProportion of randomised trials with indicator value equal to or exceeding the observed indicator value. $P = (1 + \text{number of runs} \geq \text{observed}) / (1 + \text{number of randomised runs})$.

^BPalatability for each species presented here is based on the interviews with herders and local people (details provided in Table S1).

^CMajor ethnobotanical uses: the uses presented in the table are based on present study interviews with herders and local people and references consulted are provided in S1 and S2 Tables.

^DChorotype determination is based on species geographical distribution as detailed in Press et al. (2000) and Wu et al. (1994).

is an important step in the process that maintains their future availability for ethnobotanical use.

Changes in species range sizes due to climate change are especially likely in montane regions (Myers *et al.* 2000). Climate change is not only a threat to the plant species and ecology of montane regions, but to the social and economic sustainability of local people who utilise vegetation on the slopes of mountain ranges like the Himalaya. Annual temperature is projected to increase in the Himalaya by 4–5°C by the end of the 21st century (Kumar *et al.* 2006). Feeley and Silman (2010) calculated that similar temperature changes are expected on the eastern slopes of the Andes, and that plants will need to migrate upslope >900 m by the end of the century to remain at climate equilibrium. Thus, on some ridges, such as Lauribina Hill, alpine plants will be squeezed between the ridge top and the upward-migrating shrubs and trees. The only remaining habitat options for persistence of herbaceous alpine species may become burned patches where woody cover is consumed. Our results suggest that local agropastoralist's use of fire opens gaps in the shrub canopy and favours herbaceous species at least temporarily. Their use of fire in the future may be the only mechanism for maintaining adequate grazing fodder and other species of ethnobotanical significance in proximity to their current settlements. However, the interaction of increasing temperature with fire may create more dangerous burning conditions over time.

In accordance with the results of earlier studies (Xiang *et al.* 2014), we found that burned and unburned shrub habitat patches differed considerably in edaphic properties; burned soils had higher, approximately neutral, pH and lower soil moisture compared with unburned soils. Generally, fire increases soil pH and nutrient availability, but severe wildfire can have deleterious and lasting effects on soil nutrients, structure, porosity and microbial activity (Certini 2005; Xiang *et al.* 2014), and hydrophobicity of burned soil reduces water absorption capacity and makes soil more prone to erosion (Certini 2005). The present study was completed 2 years after fire, and the observed differences in vegetation composition may have been influenced to an unknown extent by changes in soils. Further, a detailed account of soil property changes due to fire is beyond the scope of this research and requires separate and detailed study. As observed in LNP, colonisation and growth of grasses can benefit from burning with exposed soils, both of which were significantly greater in burned areas on Lauribina Danda, and reduced dominance of competitive woody species (Walker 2001), which covered a significantly smaller area in the burned matrix in our study area.

Conclusions

Fire at LNP in central Nepal was found to open the shrub canopy of *Rhododendron* species and increase species richness of herbaceous plants, including grasses and forbs of ethnobotanical value 2 years after fire. Our study area showed early signs of secondary succession in fire-affected shrublands of the subalpine zone. We also found that the IV analysis demonstrated a change at mid-elevation, at least temporarily, in vegetation composition from shrubland to a grassland similar to higher-elevation alpine grasslands. Burning to some extent also favoured Himalaya and Pan-Himalaya endemics and some rare

and threatened species. Climate change and the potential for increasing anthropogenic impacts from alteration of historic fire regimes, medicinal plant harvesting, logging and grazing may place alpine shrublands at greater risk. Reduction and potential elimination of those shrublands in central Nepal not only threatens biodiversity but the local livelihoods of the people of LNP. Additional data may expose relationships and responses not apparent in the earliest years after high-elevation fires (DellaSala *et al.* 2015). We recommend a landscape-scale analysis of the alpine shrubland and pasture zone to examine the potential existence of a shifting mosaic steady state (Bormann and Likens 1979) created by anthropogenic fire and local management. We also suggest the establishment of long-term monitoring on the effects of anthropogenic fire on alpine shrublands and a plan that includes locals for sustainable management of this valuable habitat.

Conflicts of interest

The authors declare no conflicts of interest.

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