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; Bhattacharuya 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 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 anthropogon* 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 (Olsen 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
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 Langtung 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 (Dhunche station, 1950 m a.s.l.). Most of the precipitation occurs during the summer monsoon season, which lasts from June until the...
beginning of October. Average maximum temperature was 23°C in June and average minimum temperature was ~2°C in January.

The study area has 20–35° 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 arbuscula 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
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soil tester).

system, soil pH using a pH meter, and soil moisture with a

ical position (latitude and longitude) using a global positioning

burned, 12 unburned plots). At each plot, we recorded geograph-

unburned plots; T2

25.80

f

19.65, T2

6.37

T3

3.42

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

from the next matrix or patch margin, maintaining a 20–30-m

oriented diagonally, which were further divided into four

0.5 m) subquadrats. In total, 267 quadrats (1 m²) and

1068 subquadrats (0.5 m²) 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. Vegeta-
tion 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. Botani-
cal vouchers were deposited at Tribhuvan University Central

Herbarium, Nepal, where field identifications were later con-
firmed. 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 Daalkhajav (2005), we categorised the

palatability into preferred, desirable, consumed but less desir-
able, not consumable and toxic.

Both verbal and written consents were obtained for the study.

Written permission for fieldwork was obtained from the autho-

rities 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 commu-
nities in Chandabari and Lauribina within LNP before establish-
ing 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 Win-

dows, 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 pair-
wise differences among burned and unburned plots within and

among transects.

Prior to running both the multiresponse permutation proce-
dure (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
representing habitat types or combinations of habitat types and are of prime interest for ecosystem conservation and management (Legende 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. 0.01%), 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 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 1. Biophysical variables recorded in unburned and burned plots

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

aSoil moisture was recorded as a categorical variable, the value of which ranged from 0 (dry) to 8 (moist).
Table 2. Results of multiresponse permutation procedure

<table>
<thead>
<tr>
<th>Plot comparisons</th>
<th>Observed delta ($\delta$)</th>
<th>$\delta$ under null hypothesis</th>
<th>$T^A$</th>
<th>$p^A$</th>
<th>$A^A$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burned v. unburned</td>
<td>0.602</td>
<td>0.627</td>
<td>0.0001</td>
<td>-1.536</td>
<td>-11.249</td>
</tr>
<tr>
<td>Transects (T1–T3)</td>
<td>0.546</td>
<td>0.627</td>
<td>0.0001</td>
<td>-1.076</td>
<td>-25.845</td>
</tr>
<tr>
<td>Unburned transects</td>
<td>0.546</td>
<td>0.648</td>
<td>0.0006</td>
<td>-1.000</td>
<td>-12.892</td>
</tr>
<tr>
<td>T1 v. T2</td>
<td>-0.555</td>
<td>0.233</td>
<td>0.006</td>
<td></td>
<td></td>
</tr>
<tr>
<td>T1 v. T3</td>
<td>-13.243</td>
<td>&lt;0.001</td>
<td>0.201</td>
<td></td>
<td></td>
</tr>
<tr>
<td>T2 v. T3</td>
<td>-12.098</td>
<td>&lt;0.001</td>
<td>0.169</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Burned transects</td>
<td>0.470</td>
<td>0.565</td>
<td>0.0003</td>
<td>-1.159</td>
<td>-16.780</td>
</tr>
<tr>
<td>T1 v. T2</td>
<td>-10.671</td>
<td>&lt;0.001</td>
<td>0.109</td>
<td></td>
<td></td>
</tr>
<tr>
<td>T1 v. T3</td>
<td>-15.158</td>
<td>&lt;0.001</td>
<td>0.186</td>
<td></td>
<td></td>
</tr>
<tr>
<td>T2 v. T3</td>
<td>-8.933</td>
<td>&lt;0.001</td>
<td>0.087</td>
<td></td>
<td></td>
</tr>
<tr>
<td>T1: Burned v. unburned</td>
<td>0.536</td>
<td>0.582</td>
<td>0.0003</td>
<td>-1.031</td>
<td>-7.941</td>
</tr>
<tr>
<td>T2: Burned v. unburned</td>
<td>0.528</td>
<td>0.589</td>
<td>0.0004</td>
<td>-1.421</td>
<td>-9.488</td>
</tr>
<tr>
<td>T3: Burned v. unburned</td>
<td>0.445</td>
<td>0.459</td>
<td>0.0003</td>
<td>-1.677</td>
<td>-2.585</td>
</tr>
</tbody>
</table>

$A^A$The more negative $T$, the stronger the separation. $p^A$-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., Geranium donianum DC. (dark-red milkvetch), Poa sp., Heracleum nepalense D. Don (Nepal cowparsnip), Lloydia serotina (L.) Rehb (mountain spiderwort), Maianthemum purpureum (Wall.) LaFrankie (purple meadowflower), 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 Roscoea alpina Royle (mountain roscoee 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 widdering range 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.
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 conserva-
tion 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 indumen-
tum 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 an
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 Trans-
sects 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 resil-
iently 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>
<thead>
<tr>
<th>Species</th>
<th>Palatability^A</th>
<th>Major ethnobotanical use^B</th>
<th>Chorotype^C</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burned</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Arenaria bryophylla Fernald (Caryophyllaceae)</td>
<td>P</td>
<td>Medicinal</td>
<td>HE</td>
</tr>
<tr>
<td>Astragalus dominans DC. (Fabaceae)</td>
<td>P</td>
<td>Medicinal</td>
<td>HE</td>
</tr>
<tr>
<td>Epilobium wallachianum Hausskn. (Onagraceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Gentiana depressa D. Don (Gentianaceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>HE</td>
</tr>
<tr>
<td>Heracleum nepalense D. Don (Apiaceae)</td>
<td>P</td>
<td>Fodder, food and medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Juncus thomsonii Buchenau (Juncaceae)</td>
<td>U</td>
<td>–</td>
<td>CA</td>
</tr>
<tr>
<td>Lloydia serotina (L.) Rchb. (Liliaceae)</td>
<td>P</td>
<td>Medicinal</td>
<td>HO</td>
</tr>
<tr>
<td>Maianthemum purpureum (Wall.) LaFrankie</td>
<td>P</td>
<td>Food and medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Nardostachys jatamansi (D. Don) DC. (Caprifoliaceae)</td>
<td>N</td>
<td>Medicinal and religious</td>
<td>PH</td>
</tr>
<tr>
<td>Pedicularis elwesi Hook. F. (Orobanchaceae)</td>
<td>U</td>
<td>–</td>
<td>PH</td>
</tr>
<tr>
<td>Poa sp. (Poaceae)</td>
<td>P</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Rheum acuminatum Hook. F. and Thomson (Polygonaceae)</td>
<td>U</td>
<td>Food and medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Rubus nepalensis (Hook. F.) Kuntze (Rosaceae)</td>
<td>P</td>
<td>Food</td>
<td>HE</td>
</tr>
<tr>
<td>Unburned</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anemone demissa Hook. F. and Thomson (Ranunculaceae)</td>
<td>N</td>
<td>Medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Eritrichium sp. (Boraginaceae)</td>
<td>N</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Geranium donianum Sweet (Geraniaceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Roscoea alpina Royle (Zingiberaceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>HE</td>
</tr>
<tr>
<td>Sorbus microphylla (Wall. Ex Hook. f.) Wenz. (Rosaceae)</td>
<td>P</td>
<td>Fodder and fuel</td>
<td>PH</td>
</tr>
<tr>
<td>Thalictrum cultratum Wall. (Ranunculaceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>PH</td>
</tr>
<tr>
<td>Viola biflora L. (Violaceae)</td>
<td>U</td>
<td>Medicinal</td>
<td>HO</td>
</tr>
</tbody>
</table>

^A Palatability for each species presented here is based on the interviews with
hersders and local people (details provided in Table S1).
^B Major ethnobotanical uses: the uses presented in the table are based on
present study interviews with hersders and local people and references con-
sulted are provided in Tables S1 and S2.
^C Chorotype determination is based on species geographical distribution as
detailed in Press et al. (2000) and Wu et al. (1994).
**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

<table>
<thead>
<tr>
<th>Burn status, transect</th>
<th>Indicator species</th>
<th>Species abbreviation</th>
<th>Observed indicator value (IV)</th>
<th>IV from randomised group Mean s.d.</th>
<th>Palatability$^B$</th>
<th>Major ethnobotanical use$^C$</th>
<th>Chorotype$^D$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burned</td>
<td>Salix calyculata Hook. f. ex Andersson (Salicaceae)</td>
<td>Salcal</td>
<td>25</td>
<td>11.4</td>
<td>3.58</td>
<td>0.0058</td>
<td>N</td>
</tr>
<tr>
<td>T2</td>
<td>Poa sp. (Poaceae)</td>
<td>Poasp</td>
<td>19.9</td>
<td>18.3</td>
<td>0.79</td>
<td>0.0012</td>
<td>P</td>
</tr>
<tr>
<td>T3</td>
<td>Potentilla peduncularis D. Don (Rosaceae)</td>
<td>Potped</td>
<td>23.6</td>
<td>9.6</td>
<td>3.65</td>
<td>0.006</td>
<td>N</td>
</tr>
<tr>
<td>T3</td>
<td>Aletris paniculata (Klotz) Hand.-Mazz. (Nartheciaceae)</td>
<td>Alepau</td>
<td>36.4</td>
<td>12</td>
<td>3.65</td>
<td>0.0004</td>
<td>D</td>
</tr>
<tr>
<td>T3</td>
<td>Anemone rupestris Wall. ex Hook. f. and Thomson (Ranunculaceae)</td>
<td>Anerup</td>
<td>26.7</td>
<td>15.3</td>
<td>3.18</td>
<td>0.0042</td>
<td>P</td>
</tr>
<tr>
<td>Anemone smithiana Lauener and Panigrahi (Ranunculaceae)</td>
<td>Anesmi</td>
<td>19.5</td>
<td>6.7</td>
<td>3.61</td>
<td>0.0102</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Bistorta macrophylla (D. Don) Soják (Polygonaceae)</td>
<td>Bismac</td>
<td>33.3</td>
<td>14.4</td>
<td>3.31</td>
<td>0.0004</td>
<td>U</td>
<td>Medicinal</td>
</tr>
<tr>
<td>Chesneya mbigena (D. Don) Ali (Fabaceae)</td>
<td>Cheneb</td>
<td>30.9</td>
<td>10.6</td>
<td>3.66</td>
<td>0.0006</td>
<td>U</td>
<td>Food</td>
</tr>
<tr>
<td>Corydalis juncea Wall. (Papaveraceae)</td>
<td>Corjun</td>
<td>27.7</td>
<td>8.4</td>
<td>3.65</td>
<td>0.002</td>
<td>U</td>
<td>–</td>
</tr>
<tr>
<td>Euphorbia stracheyi Boiss. (Euphorbiaceae)</td>
<td>Eupstr</td>
<td>32</td>
<td>15.9</td>
<td>2.83</td>
<td>0.0002</td>
<td>U</td>
<td>Medicinal</td>
</tr>
<tr>
<td>Juncus thomsonii Buchenau (Juncaceae)</td>
<td>Juntho</td>
<td>19.3</td>
<td>7.6</td>
<td>3.71</td>
<td>0.0164</td>
<td>U</td>
<td>–</td>
</tr>
<tr>
<td>Ligusticum sp. L. (Apiaceae)</td>
<td>Ligs</td>
<td>23.6</td>
<td>15.5</td>
<td>3.02</td>
<td>0.0168</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Lloydia flavonutans H. Harms (Liliaceae)</td>
<td>Lylita</td>
<td>41.5</td>
<td>12.4</td>
<td>3.51</td>
<td>0.0002</td>
<td>U</td>
<td>–</td>
</tr>
<tr>
<td>Ponerorhiza chaua (D. Don) Soó (Orchidaceae)</td>
<td>Ponchu</td>
<td>45.3</td>
<td>12.1</td>
<td>3.5</td>
<td>0.0002</td>
<td>P</td>
<td>Medicinal</td>
</tr>
<tr>
<td>Unburned</td>
<td>Athyrium wallichianum Ching (Athyriaceae)</td>
<td>Athwal</td>
<td>22.4</td>
<td>14.1</td>
<td>3.34</td>
<td>0.0218</td>
<td>U</td>
</tr>
<tr>
<td>T1</td>
<td>Primula obliqua W. W. Sm. (Primulaceae)</td>
<td>Priobl</td>
<td>15.2</td>
<td>6.6</td>
<td>3.65</td>
<td>0.0254</td>
<td>N</td>
</tr>
<tr>
<td>T2</td>
<td>Primula sikkimensis Hook. (Primulaceae)</td>
<td>Prisi</td>
<td>18.3</td>
<td>8.8</td>
<td>3.75</td>
<td>0.025</td>
<td>N</td>
</tr>
<tr>
<td>Ryhodendron campanulatum D. Don (Ericaceae)</td>
<td>Rhocam</td>
<td>33.8</td>
<td>10.6</td>
<td>3.68</td>
<td>0.0004</td>
<td>N</td>
<td>Medicinal</td>
</tr>
<tr>
<td>Sorhus microphylla (Wall. ex Hook.f.) Wenz. (Rosaceae)</td>
<td>Sormic</td>
<td>19</td>
<td>7.9</td>
<td>3.74</td>
<td>0.014</td>
<td>P</td>
<td>Fodder and Fuel</td>
</tr>
<tr>
<td>T3</td>
<td>Caltha palustris var. himalaica Tamura (Ranunculaceae)</td>
<td>Calpal</td>
<td>39</td>
<td>12.2</td>
<td>3.47</td>
<td>0.0002</td>
<td>U</td>
</tr>
<tr>
<td>Kobresia nepalensis (Nees) Kük. (Cyperaceae)</td>
<td>KobnP</td>
<td>30.2</td>
<td>13.2</td>
<td>3.51</td>
<td>0.0018</td>
<td>P</td>
<td>–</td>
</tr>
<tr>
<td>Kobresia pygmaea (C.B. Clarke) C.B. Clarke (Cyperaceae)</td>
<td>Kobpyg</td>
<td>16.9</td>
<td>7.9</td>
<td>3.79</td>
<td>0.0352</td>
<td>P</td>
<td>–</td>
</tr>
<tr>
<td>Kobresia sp. (Cyperaceae)</td>
<td>Kobsp</td>
<td>29.3</td>
<td>15.1</td>
<td>3.17</td>
<td>0.0006</td>
<td>P</td>
<td>–</td>
</tr>
<tr>
<td>Oxygraphis polyptera (Raf.) Hook. f. and Thomson (Ranunculaceae)</td>
<td>Oxypol</td>
<td>23.4</td>
<td>12.3</td>
<td>3.58</td>
<td>0.011</td>
<td>P</td>
<td>–</td>
</tr>
<tr>
<td>Potentilla fruticosa var. arbutula (D. Don) Maxim. (Rosaceae)</td>
<td>Potfru</td>
<td>20.3</td>
<td>6.6</td>
<td>3.63</td>
<td>0.0074</td>
<td>N</td>
<td>Medicinal; Religious</td>
</tr>
<tr>
<td>Rhodendron setosum D. Don (Ericaceae)</td>
<td>Rhotset</td>
<td>33.3</td>
<td>13.8</td>
<td>3.41</td>
<td>0.0002</td>
<td>N</td>
<td>Food; Religious</td>
</tr>
<tr>
<td>Salix lindleyana Wall. ex Andersson (Salicaceae)</td>
<td>Salinn</td>
<td>52.8</td>
<td>12.9</td>
<td>3.38</td>
<td>0.0002</td>
<td>U</td>
<td>Medicinal</td>
</tr>
<tr>
<td>Saxifraga brachyoda D. Don (Saxifragaceae)</td>
<td>Saxbra</td>
<td>22.5</td>
<td>7.2</td>
<td>3.62</td>
<td>0.0052</td>
<td>U</td>
<td>–</td>
</tr>
</tbody>
</table>

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

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

$^C$Major 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.

$^D$Chorotype determination is based on species geographical distribution as detailed in Press et al. (2000) and Wu et al. (1994).
Anthropogenic fire in an alpine shrubland

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 ethno botanical 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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