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Regime shift on the roof of the world: Alpine meadows converting to shrublands in the southern Himalayas

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ABSTRACT

Worldwide, changing climates and land use practices are escalating woody-plants encroachment into grasslands, reducing biodiversity and altering ecosystem functions. The loss of alpine grasslands is a major conservation concern as they harbor many rare and endemic species. Alpine meadows in Northwest Yunnan, China, represent a global biodiversity hotspot with high species richness, beta diversity, and endemism. Shrubs have expanded greatly in the region and threaten alpine meadow biodiversity. To measure rates of meadow loss due to shrub encroachment and identify its mechanisms, we reconstructed alpine land cover, climate, and land use change from 1950 to 2009 across Northwest Yunnan using satellite data, ground surveys, and interviews. Between 1990 and 2009, at least 39% of the alpine meadows converted to woody shrubs. The patterns of change suggest that a regime shift is occurring. Despite multiple perturbations to the climate and land use systems starting in the 1950s, alpine meadows remained resilient to shrub expansion until the late 1980s. Shrublands rapidly expanded then due to feedback mechanisms involving climate, woody cover, and grazing. Fire may no longer be an effective tool for controlling shrub expansion. This regime shift threatens both endemic meadow biodiversity and local livelihoods. More generally, these trends serve as a warning sign for the greater Himalayan region where similar vegetation changes could greatly affect livelihoods, hydrology, and climate.

1. Introduction

Worldwide, shrubs are encroaching in grasslands, affecting biodiversity, ecosystem function, and carbon storage (Knapp et al., 2008). Climate change, overgrazing, and land use all contribute to shrub encroachment (Walker and Wahren, 2006). When multiple interacting factors reach a critical threshold, a relatively small perturbation can trigger a regime shift, i.e., a sudden, and potentially irreversible, transition to an alternate state (Scheffer, 2009). To conserve grassland biodiversity, the drivers of shrub encroachment need to be identified and their critical thresholds determined. Untangling these factors, however, is challenging in coupled natural and human systems (Brock and Carpenter, 2010).

Shrub encroachment into alpine meadows represents a particular grave conservation threat (Montane et al., 2007; Xu and Wilkes, 2004). Alpine meadows often harbor many endemic plants adapted to harsh conditions, and orogenies have resulted in high levels of both paleo- and neo-endemism (Wu et al., 2007). With the largest alpine ecosystems in the world, alpine meadows in the Greater Himalayan region are of particular concern. Northwest Yunnan Province in southwest China represents a global biodiversity hotspot (Myers et al., 2000), and meadows here have higher species richness, beta diversity, and endemism than elsewhere in the Himalayas (Salick et al., 2009; Xu et al., 2009).

Historically, Himalayan meadows have been used by indigenous agro-pastoralists whose rangeland management practices sustained both local livelihoods and biodiversity (Klein et al., 2011). The highly nutritious meadows are critical to Tibetan transhumance herding, and the replacement of meadows by unpalatable shrubs threatens their livelihoods (Yi et al., 2007). Alpine meadow species can persist in the low-intensity land use systems typical for montane regions, but even subtle environmental changes may threaten them (Galop et al., 2011; Thuiller et al., 2005). For example, when climate change shifts woody vegetation to higher elevations, species of alpine mountaintop meadows may not be able to migrate to other suitable environments (Parmesan, 2006). Likewise, the abandonment of traditional land use practices can result in shrub encroachment (Baur et al., 2006; Laiolo et al., 2004). Indeed, climate change and land use change interact in alpine ecosystems making it difficult to identify the main driver(s) of shrub expansion.

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Since the 1950s, numerous environmental changes affected Himalayan ecosystems. Climate warming rates here are higher than the global average, altering ecosystem productivity (Peng et al., 2010), phenology (Yu et al., 2010), and permafrost (Xu et al., 2009). Changing political and social structures have led to economic development and abandonment of traditional land use practices (Klein et al., 2011; Yeh and Gaerrang, 2010). However, despite grassland degradation throughout the Himalayas (Harris, 2010), rates of shrub encroachment have not been quantified, and its drivers are not well understood.

Our goals were (a) to measure rates of shrub expansion into alpine meadows in Northwest Yunnan and (b) to identify the likely mechanisms driving these changes. We used Landsat TM/ETM + satellite image analysis and dendrochronology to map land-cover distribution and change in the forest/shrub/meadow ecotone of the alpine zone (3800–4500 m) in 1974, 1990, and 2009. In addition, we reconstructed changes since the 1950s using interviews, livestock records, and climate data. We then tested the following hypotheses:

H1. *Climate change*: We hypothesized that temporal patterns of shrub establishment and expansion were caused by changes in temperature and precipitation. Warming alters phenology (Sturm et al., 2005; Yu et al., 2010) and caused shrub encroachment in other alpine (Harte and Shaw, 1995; Klein et al., 2007) and arctic (Elmendorf et al., 2012a,b; Molau, 2010) environments. Snowpack variability can alter productivity of high-elevation systems (Peng et al., 2010) and cause shrub encroachment (Bjork and Molau, 2007; Wipf and Rixen, 2010). In the Himalayas, temperature, precipitation, snowpack, and phenology changed rapidly since the 1950s (Gao et al., 2012; Xu et al., 2009), and NW Yunnan represents a hotspot of change even within the Greater Himalayas (Haynes, 2011), resulting in receding glaciers and advancing tree lines (Baker and Moseley, 2007; Wong et al., 2010).

H2. *Burning cessation*: We hypothesized that the 1988 region-wide burning ban triggered shrub establishment, and that areas where burning continued irrespectively suffered less shrub encroachment. Burning cessation has triggered shrub encroachment world-wide (Roques et al., 2001). In NW Yunnan, herders historically burned shrubs to improve meadows, but the 1988 burning ban may have fostered shrub encroachment (Sherman et al., 2008).

H3. *Shrub autocatalysis*: We hypothesized that shrub autocatalysis, where existing shrubs facilitate further shrub expansion due to feedbacks (D'Odorico et al., 2010; Ratajczak et al., 2011; Sturm et al., 2005), was an important driver of shrub expansion. Feedbacks are critical components of regime shifts, because they push the original ecosystem towards an alternate stable state from which reversion to the original state is difficult or impossible (Scheffer, 2009).

H4. *Overgrazing*: We hypothesized that areas of high grazing intensity had more shrub encroachment, because shrubs establish more easily when other vegetation is removed (Eldridge et al., 2011; Schlesinger et al., 1990).

H5. *Abandonment*: We hypothesized that territories where alpine yak herding was abandoned have particularly high shrub encroachment rates because activities associated with humans (e.g., clearing of woody vegetation) may facilitate encroachment. Abandonment was the major driver of 20th century shrub encroachment in the European Alps (Galop et al., 2011; Montane et al., 2007). In our study area, economic development integrated rural areas into the regional economy of western China since the 1980s (Yeh and Gaerrang, 2010), leading to abandonment of yak herding in some villages (Yi et al., 2007).

2. Materials and methods

2.1. Study area

Northwest Yunnan Province is at the southeast edge of the Qinghai–Tibetan Plateau (QTP), bordering Tibet and Sichuan Province (Fig. 1a), and an IUCN biodiversity hotspot (Myers et al., 2000). Three major rivers (the Yangtze, Mekong, and Salween) create steep gorges, with elevations ranging from 1800 to 6740 m. The alpine zone occurs between 4000 and 5000 m. Climate in the alpine zone is cold with near-constant rain during the monsoon season (June–September) and sporadic snowfall from October through May. The mean annual temperature above 4100 m is <0 °C (Sherman et al., 2008).

The alpine zone consists of shrub, meadow, and scree ecosystems. Meadows exhibit the highest species richness and support the most rare and endemic plant species (Salick et al., 2009; Sherman et al., 2008). Alpine meadow communities are dominated by *Polygonum viviparum*, *Potentilla fulgens*, *Fragaria nilgeerensis*, *Anemone narcissiflorioides*, *Cyananthus flavus*, and *Festuca ovina* and include *Pedicularis dolichoglossa*, *Viola szetchuanensis*, *Agrostis limprichtii*, and *Primula* sp. (Li and Walker, 1986), and beta diversity is high (Salick et al., 2009; Sherman et al., 2008).

Alpine meadows are crucial to local Tibetans' yak husbandry because meadows provide the highest nutritional value (Yi et al., 2007). When interviewed, herders confirmed the increase in shrubs, and cited it as a major threat to livelihoods (Haynes, 2011). The shrub invaders include *Rhododendron* spp., *Juniperus* spp., *Berberis* spp., *Potentilla fruticosa* and *P. glabra*, all of which are either noxious or poor forage for grazers (Haynes et al., in press).

2.2. Satellite image analysis

2.2.1. General approach

We used Landsat TM/ETM + images and a multi-temporal change classification technique to quantify land use and land cover change in the alpine zone (>3800 m) in two time periods: 1974–1990 and 1990–2009. We then summarized land cover patterns and encroachment rates to determine temporal and spatial patterns of shrub encroachment, and test hypothesized drivers of the encroachment. A detailed description of the satellite image analysis is included in Brandt et al., 2012.

2.2.2. Satellite imagery and image pre-processing

The study area is covered by two Landsat footprints (path/row 132/040 and 132/041). We acquired Landsat MSS images from 1974, and Landsat TM/ETM + images from ca. 1990, 2000, and 2009, and co-registered all images to the orthorectified Landsat 2000 GeoCover Dataset. Positional uncertainty was <0.4 pixels (<12 m). Clouds and cloud shadows were masked manually.

2.2.3. Ground-truth data collection

We collected field data to train and validate the satellite classifications in July–November 2008, August–October 2009, July–September 2010, and May–July 2011. Due to the ruggedness of the study area, a random sampling design was not feasible. We collected field data in nine alpine territories prior to image analysis, and nine different alpine territories thereafter (Table 1). We hired local guides, conducted informal interviews about land-use, and recorded land use for at least 40 locations within each land cover type and took photographs of the surrounding 1 ha (Justice and Townshend, 1981). To supplement the field data, we gathered additional ground-truth data from high resolution imagery in GoogleEarth™ (approximately 20% of the ground truth dataset).

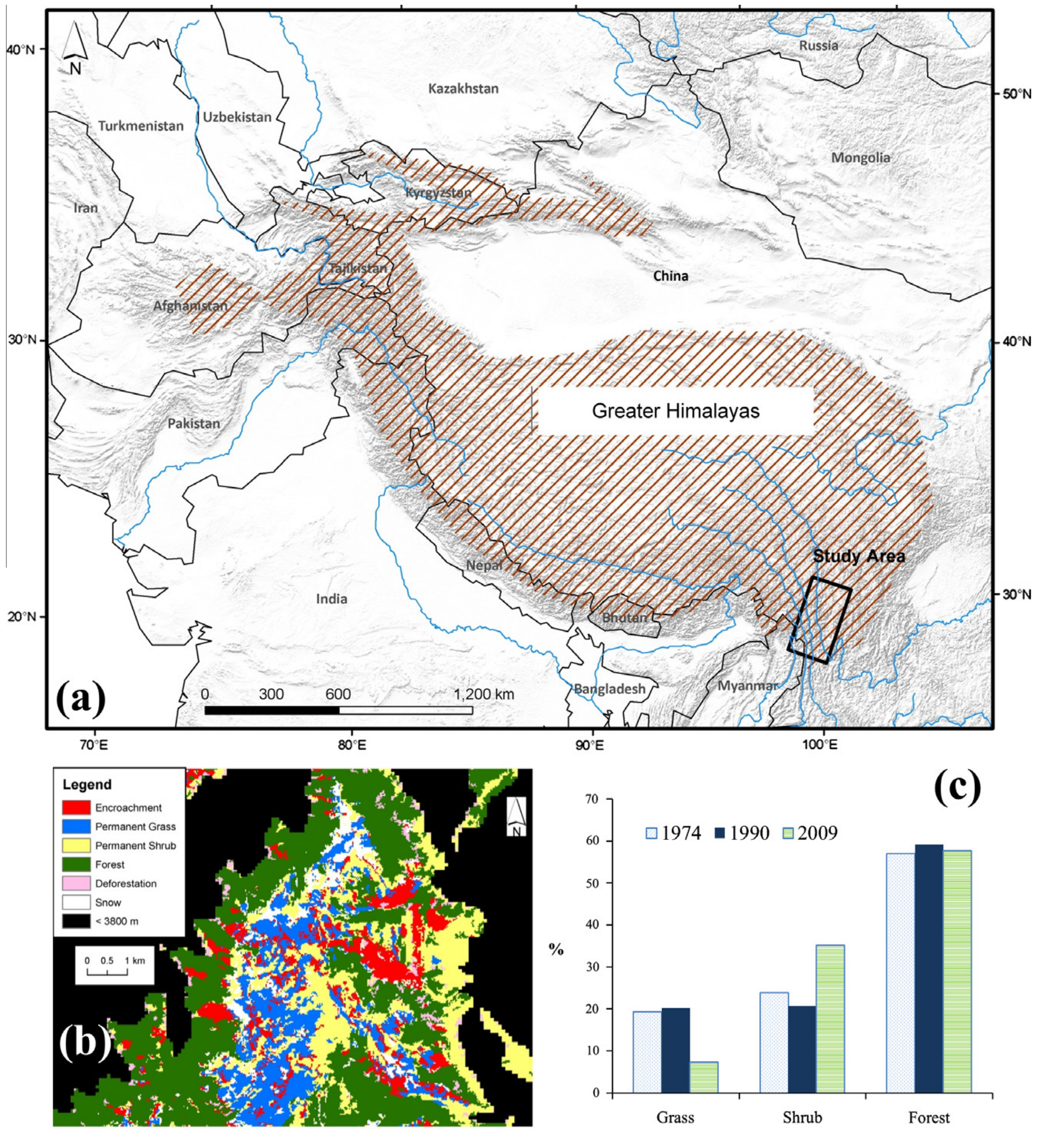

Fig. 1. (a) The location of the study area. (b) Close-up of land cover/land use change from 1990 to 2009 in one alpine territory. (c) Land cover was stable from 1974 to 1990, but grasslands shrank dramatically from 1990 to 2009 while shrublands expanded.

In total, our ground truth dataset included 1573 polygons (93,300 pixels).

2.2.4. Image classification

To classify land cover change, we used a composite analysis approach. Composite analysis uses multi-date imagery, i.e. satellite images from multiple seasons and different years are combined into a single image stack. The added phenological and temporal information of the multi-date imagery allows more detailed land cover classification (Wolter et al., 1995) and more accurate classification of land cover change (Brandt et al., 2012; Kuemmerle et al., 2006). To separate the classes, we used Support Vector Machines (SVM). SVM is a non-parametric classifier that places hyperplanes to separate different classes (Foody and Mathur, 2004). SVM are well-suited for composite analysis in complex environments because they can accommodate the non-normal and multi-modal class distributions of multi-date imagery (Brandt et al., 2012; Huang et al., 2002).

We assessed two time periods, 1974–1990 (historic period) and 1990–2009 (recent period). For the historic period, we combined 1974 MSS images, ca. 1990 TM images, and elevation and hillshade (corresponding to the illumination of the November 19, 1990 Landsat TM image) and separated four classes: Grassland, Shrubland, Forest, and Snow. Similarly, for the recent period, we combined

Table 1
Alpine territories visited from 2007–2011. Types of data collected at each site include: G = Ground truth, D = Dendrochronology, FI = Formal Interview, II = Informal Interview.

Site	Number of visits	Timing of visit	Data collected
AD	5	Pre-classification	G, D, FI, II
BM	4	Pre-classification	G, D, FI, II
ML	3	Pre-classification	G, FI, II
XD	1	Pre-classification	FI, II
SS	1	Pre-classification	G, FI, II
YR	1	Pre-classification	G, FI, II
HP	1	Pre-classification	G, II
GZ	1	Pre-classification	G, II
YB	1	Pre-classification	G, II
MY	1	Pre-classification	G, II
HS	1	Pre and post-classification	G, II
QH	2	Pre and post-classification	G, II
SK	5	Post-classification	G, D, II
AB	1	Post-classification	G, II
TB	1	Post-classification	G, D, II
ZH	1	Post-classification	G, D, II
TC	1	Post-classification	G, II
DX	1	Post-classification	G
LE	1	Post-classification	G
NP	1	Post-classification	G, II
NL	1	Post-classification	G, II

TM/ETM + imagery from 1990, 2000 and 2009, along with elevation and hillshade data (October 28, 1999 image) to classify the same land covers, plus the change class "shrub encroachment", i.e., those areas that converted from grassland in 1990 to shrubland in 2009. To fill cloud and snow gaps, we carried out a classification excluding the spring images (where snow is more extensive). To train and validate our SVM, we randomly selected approximately 1000 points from each class in our ground truth dataset and used 10-fold cross-validation (Brandt et al., 2012). We used a minimum mapping unit of 6 pixels (~0.65 ha).

2.2.5. Accuracy assessment

To validate our change maps, we calculated a confusion matrix, overall, user's and producer's accuracies, plus area-adjusted estimates of land cover proportions and their confidence intervals (Card, 1982). For the accuracy assessment, we used a total of 231 independent ground truth points, including 153 field points from 11 *post hoc* field visits in 2010 and 2011. We also randomly sampled 78 points using GoogleEarth high-resolution imagery. For the shrub encroachment class, we relied solely on field data. Acquiring independent data for the historic shrub encroachment class was not possible retrospectively and we report the cross-validation accuracies only.

2.2.6. Temporal and spatial patterns of encroachment

We estimated shrub encroachment first as the encroachment rate (ER), i.e., the proportional (%) area of grassland in 1990 that converted to shrubland by 2009.

$$ER = 100 * E/(E + PG) \quad (1)$$

Here, E is the number of pixels in the encroachment class, and PG is the number of pixels in the Permanent Grassland class. This measure is a slight overestimate, because most snow-covered areas in the imagery were grasslands. Therefore, we also calculated a snow adjusted encroachment rate (ERA) which assumes that all of the snow-covered areas were grassland:

$$ERA = 100 * E/(E + PG + S) \quad (2)$$

where S is the number of snowy pixels. This represents a conservative rate of shrub encroachment, because a small percentage of the snow-covered areas are shrub, bare ground, scree, or bedrock.

We summarized encroachment rates and land cover for 1990–2009 at several different spatial scales. First, we summarized alpine land cover and change for the entire study area, and inside and outside of the Baimaxueshan National nature reserve, where burning prohibitions are most strictly enforced. Second, we summarized land covers for zones of 100-m elevation, 5°slope, and 45° aspect. Third, we summarized land cover change at the township level. Finally, we summarized encroachment and land cover within the alpine "territory" of each village where we had conducted interviews with herders (see below).

2.3. Dendrochronology

2.3.1. Data collection

To provide greater insight into the timing of shrub establishment we collected shrub cuttings from 125 shrubs in 13 sites and five territories. Prior to satellite image classification, we collected shrub cuttings in areas of mixed meadow and shrub land cover in the summer of 2008, for use in training the image classification. Following image analysis, we sampled areas mapped as encroachment in the summers of 2009 and 2010 for the accuracy assessment.

2.3.2. Analysis

Alpine shrubs of the southeastern Tibetan Plateau produce latewood, have clear growth rings, and can be used for detailed dendrochronological analysis (Liang and Eckstein, 2009; Liang et al., 2012). It was beyond the scope of our study to perform detailed dendrochronological analysis, but we used standardized growth ring counting techniques to obtain estimates of the oldest and average ages of the shrubs at each site (Cook and Kairlukstis, 1990; Montane et al., 2007). At each site, we cut discs from the base of at least 3 of the largest individuals of the dominant shrub species. Samples were sanded and digitally photographed on a ruler to enable thorough ring counting. A ring was indicated by a section of earlywood followed by a section of latewood. Proper training and magnification of microsections enabled false rings to be identified.

2.4. Climate data

2.4.1. Data collection

We acquired meteorological station data from 1960 to 2008 from the two National Meteorological Administration of China stations in our study area, Zhongdian and Deqin counties.

2.4.2. Analysis

We calculated local variation in the rates and seasonality of change over the last half century (1960–2008) using the cumulative method (Lozowski et al., 1989) to filter out noise. We calculated 24- and 48-year rates of change for precipitation, annual daily low, annual daily high, growing season daily low, growing season daily high, winter daily low, and winter daily high (Haynes, 2011).

Alpine vegetation is vulnerable to snowpack and snowfall variability. Ideally, our climate data would have included long-term data on snowpack and snowfall change in the alpine zone. Lacking such data, we estimated alpine zone temperature changes from 1960 to 2009. We were especially interested in periods when average daily low temperatures changed from sub-freezing to above-freezing temperatures.

To estimate temperature changes at alpine elevations, we used 48-year temperature data at the Deqin County station (3320 m), and data from 13 stations ranging from 1646 to 4292 m within Deqin County based on at least 20-year means (Le'anwangdui, 2001). We calculated the linear lapse rate adjustment for each month

(Dodson and Marks, 1997) since this technique provides more detail than the global mean lapse rate of -6.5 °C km^{-1} (Barry and Chorley, 1987), does not require interpolation which is often inaccurate for the Himalayas (Leemans and Cramer, 1991; Rolland, 2003), and was spatially detailed enough for our analysis. The linear lapse rates fit well within global estimates and followed the expected trend from months of lower moisture to higher moisture (Barry and Chorley, 1987; Dodson and Marks, 1997; Rolland, 2003) (February, −6.65 °C km^{-1}; March, −6.35 °C km^{-1}; April, −6.25 °C km^{-1}; May, −5.91 °C km^{-1}; June, −5.59 °C km^{-1}, all $r^2 > 0.97$). Average daily low and high temperatures, as recorded at Deqin climate station, were then used to calculate low and high temperatures at all elevations in the alpine zone (>3800 m) for February through June using our DEM and the lapse rate adjustments.

2.5. Formal interviews and livestock records

2.5.1. Data collection

From 2006 to 2010, we performed informal and formal interviews to determine alpine land use history and land cover change since the 1950s (Haynes, 2011). In the summer of 2006 we held open-ended interviews with local officials and with villagers during house stays. In the summers of 2007 and 2008 we conducted preliminary interviews at the alpine rangelands. Ethnographic investigations were conducted in 2009 and 2010, taking extended trips with Tibetan herders to alpine rangelands to discuss ecological and livelihood changes and to collect ground-truth data for the satellite image analysis. Formal, semi-structured interviews were conducted with 37 herders in six villages during the winter of 2010, before the spring festival holiday during a time when villagers had a break from harvesting and had returned from herding in the distant summer pastures. The survey instrument contained seven sections: background of the interviewee, animal husbandry, burn history, village history, livestock health, ecology, and future expectations. In response to initial interviews, we modified the survey instrument to include additional sections on village history and detailed questions about traditional burning practices. We collected data on general burning practices in five time periods: feudal (prior to 1958), collectivization (1959–1982), decollectivization (1983–1987), post burning restrictions (1988–1999), and present time (2000–2010). Using the snowball method, we sampled households with experienced herders to gather information on the depth of knowledge from those with the longest history in rangeland management in the area. To assess additional village level characteristics and the history of administrative decisions, we also interviewed officials from the grassland monitoring station and animal husbandry bureau.

Livestock data were collected from published literature, including the Deqin County and Diqing Prefecture Almanacs, and annual statistical almanacs (Deqin, 1997; Le'anwangdui, 2001).

2.5.2. Analysis

From the formal interviews, we calculated the percent of respondents who participated in, or had heard of, burning during each of the five time periods included in the survey instrument.

3. Results

3.1. Changes in alpine land cover

Land use and land cover in the alpine zone (Fig. 1b) were mapped with an overall accuracy of 94% and 93% in the recent and historic time period, respectively (Table 2). The overall accuracy of the shrub encroachment class was 91%, and the minimum accuracy of any class was 80%.

Table 2
Alpine land cover/land use classification accuracy assessment for the historic (1974–1990) and recent (1990–2009) classifications.

Class	PA (%)	UA (%)	Adj. area (ha)	±CI (ha)	±CI
Historic classification (1974–1990)					
Grassland	94%	94%	94,566	2245	2%
Shrubland	92%	86%	77,296	2962	4%
Forest	99%	95%	257,203	3949	2%
Snow	94%	93%	107,379	2425	2%
Overall accuracy =	93%				
Khat =	0.93				
Recent classification (1990–2009)					
Encroachment	93%	90%	69,999	9624	14%
Grassland	80%	90%	43,217	9091	21%
Shrubland	90%	86%	109,362	14,373	13%
Forest	97%	99%	311,401	11,290	4%
Overall accuracy =	94%				
Khat =	0.94				

Changes in the proportions of grass and shrub between 1974 and 1990 were negligible, and forest cover remained largely unchanged from 1974 to 2009 (Fig. 1c). However, from 1990 to 2009, meadows shrank from 20% to 7%, representing a shrub encroachment rate of 65%. Even the conservative adjusted encroachment rate (which assumes snow-covered areas to be unchanged grassland) was 39%, representing an annual encroachment rate of 2.1%. All rates reported in the following are adjusted encroachment rates.

3.2. Spatial patterns of shrub encroachment

Shrub encroachment rates varied across the study area, with low rates in the western and southern townships (∼5%) and high rates in the northern and eastern townships (>65%) (Fig. 2a). Encroachment rates, summarized for each of the village territories where we conducted interviews, were negatively associated with the proportion of spring snow cover from the image classification (Pearson's $r = -0.687$) and positively correlated with the proportion of woody cover in 1990 (Pearson's $r = 0.692$, Fig. 2b).

Meadows at the lower elevations of the alpine zone (between 4000 m and 4200 m) were encroached most rapidly (>40%), whereas encroachment rates in the sub-alpine (3800–4000 m) and upper alpine meadows (4300–4500 m) were lower (Fig. 2c). Meadows on the sunnier south, SE, and SW facing slopes were encroached twice as fast (40–50%) as those on north-facing slopes (18–23%) (Fig. 2d). Encroachment rates did not vary among different slope zones.

Encroachment rates near the national highway were highest (∼60%) and declined further away (to about 40%) (Fig. 3a). Encroachment rates within the Baima Reserve (26%), where the burning ban was strictly enforced, were lower than outside the reserve (42%) (Fig. 3b). Remote villages where patchy burning was still practiced had some of the highest rates of encroachment (>50%) compared to <41% in villages where burning was no longer practiced. The two territories that were abandoned exhibited both one of the lowest (12%) and the highest (72%) encroachment rates (Fig. 3b).

3.3. Temporal trends in shrub establishment

According to the satellite classification, eight of the 13 sites sampled for the dendrochronological analysis converted from grass to shrub from 1990 to 2009, three sites were stable grasslands, and two sites were already shrub-dominated before 1990 (Table 3). Seven of the eight converted sites had >50% shrub cover according to

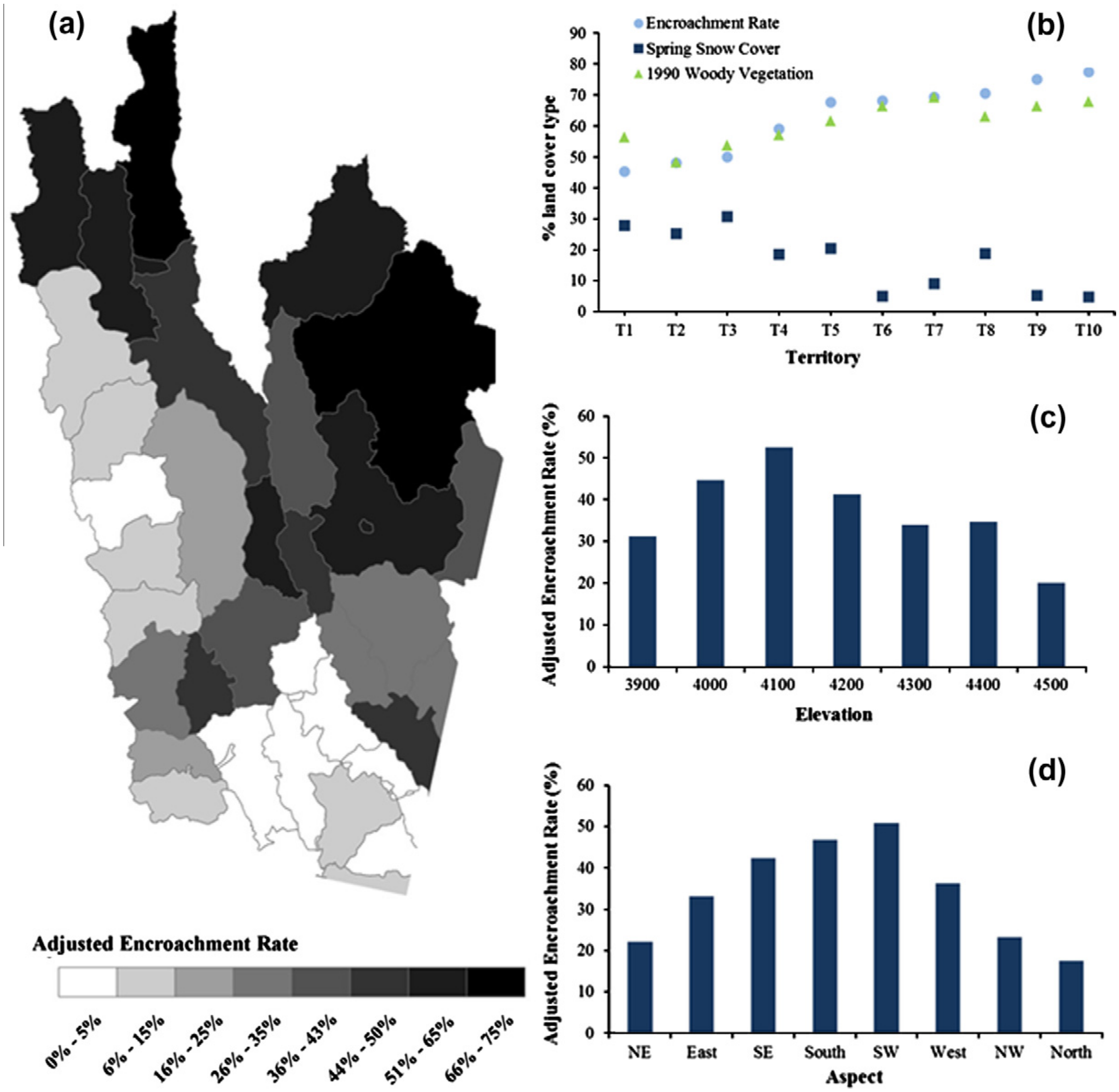

Fig. 2. (a) Encroachment rates were highly variable throughout the study area, with the northern and eastern regions suffering the highest encroachment rates. (b) Shrub encroachment rates were negatively correlated with the proportion of spring snow cover and positively correlated with proportions of 1990 woody vegetation cover. Encroachment was related to (c) elevation and (d) aspect, with meadows in lower elevations of the alpine zone (4000–4200 m) and drier south facing slopes more subject to encroachment.

field visits. All converted sites contained at least three different shrub genera, but dwarf rhododendron was dominant in seven of the eight converted sites. The oldest shrub in each site (typically *Juniperus spp.*) established around 1980 (age = 29 years (SD 4.2)) and the oldest rhododendron shrubs established in 1988 (age = 21 years (SD 8.1)).

3.4. Climate change

Deqin and Shangri-la counties experienced substantial changes in temperature and precipitation from 1960 to 2008. At the Deqin County station the rate of daily high temperature change over the past 24 years was 7.2 °C/100 years and the rate of change in daily low temperatures was 1.4 °C/100 years. At the climate station and at higher elevations, average estimated daily high and low temperatures generally increased from 1960 to 2008 (Fig. 4a and b). At 4000 m, the most dynamic oscillation around the 0° mark occurred in May between 1980 and 1990, with minimum temperatures rising consistently above freezing by the mid 1980s (Fig. 4b). In the alpine zone overall, average daily low May temperatures were generally below freezing at the beginning of our study period (1959–1969) (Fig. 4e), but above freezing for the lower elevations of the alpine zone (i.e. 3800–4200 m) at the end of our study period (1999–2008) (Fig. 4f). Precipitation changes were especially pronounced in spring (February through May) after 1980 (Fig. 4c and d). The overall amount of precipitation during these months did not change considerably, but there was increased rainfall (Fig. 4c) and no snowfall (Fig. 4d) after 1980 at the climate stations.

3.5. Changes in land-use practices

Our interviews and official livestock records highlighted that the period from 1950 to 2009 was delineated into three major eras of policy shift and land tenure: Feudal, Collectivization and Decollectivization (Fig. 5). Alpine herding practices, including yak herd size and burning practices, varied greatly among eras. During the Feudal

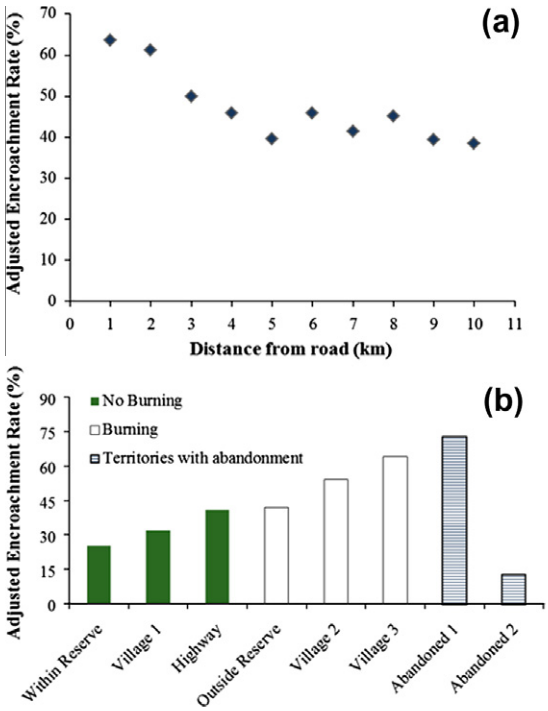

Fig. 3. (a) Higher grazing intensities close to the national highway increased shrub encroachment rates; (b) Relationships between encroachment and burning, and encroachment and abandonment indicate that recent local-scale burning and pasture abandonment did not influence regional patterns of shrub encroachment.

era (pre-1958), village chiefs or monasteries owned and managed the livestock, and yak herds were small (<50,000 total head of yak in Diqing prefecture in 1952). Grasslands were abundant and burning was limited (only 11% of respondents reported burning). During the Collectivization era (1958–1983), livestock and land were collectivized, with incentive to increase production. Total herd sizes grew to >150,000 yaks by 1983, and alpine burning was at its peak (54% of respondents reported burning). Following decollectivization (1983-present), families for the first time owned and managed livestock and land. Total head of yak grew to a peak of almost 250,000 in 2001. Burning diminished (to 40%) immediately after decollectivization, and decreased further after the 1998 burning restrictions (to 22% from 1988 to 1999 and then to only 11% from 2000 to 2009).

3.6. Synthesis of land use, climate and shrub establishment from 1950 to 2009

The synthesis of our satellite, dendrochronology, climate, and land use data (Fig. 5) indicated that alpine ecosystems were subject to mounting perturbations in both the land use and the climate systems from 1950 to 2009 (Fig. 6). During the feudal era (Fig. 6a), alpine land use, climate, and land cover were stable. Following collectivization (Fig. 6b), climate became slightly warmer and wetter, and land use practices shifted due to incentives for increased production under the communal system. Alpine land cover remained stable. Following decollectivization, burning prohibitions were enacted and yak herds grew. Grazing pressure increased throughout the alpine zones, and especially along major transportation routes. Perturbations in the climate system, including rising temperatures and decreasing snowfall, also mounted. The first dwarf rhododendron shrubs established in 1988, and expanded rapidly (Fig. 5).

4. Discussion

4.1. Implications of shrub encroachment for biodiversity

Alpine meadows remained unchanged from 1974 to1990 despite changes in climate and land use, but an abrupt shift occurred from 1990 to 2009 when at least 39% of all meadows were encroached by shrubs. The annual encroachment rate of 2.1% is among the highest reported in the literature (0.1–2.3% per year) for other areas of broad-scale shrub expansion (Briggs et al., 2005; Coop and Givnish, 2007; Sankey and Germino, 2008; Thompson, 2007).

Given the exceptionally rich flora with many endemics, this shrub encroachment is likely to seriously deplete plant diversity in the region. The highest plant diversity of this region is found in alpine meadows (Sherman et al., 2008) between 4200 and 4500 m (Salick et al., 2004). Our results showed that meadows in this elevation zone disappeared at a rapid rate, and the shrinking habitat area represents a major conservation threat (MacArthur and Wilson, 1967). Each territory has distinct species assemblages with little similarity of species among sites (Sherman et al., 2008). With shrub expansion rates approaching 70% in some townships, it is likely that such meadow contraction is leading to a reduction in

Table 3
Summary of dendrochronological shrub analysis taken from 13 different sites in five different alpine territories.

Territory	Site	Land cover classification	Dominant shrub type	Total % shrub cover	Age of all shrubs	
					Average	SD
Visited pre-classification						
BM	B1	Permanent Grass	Rhododendron	50	16	2
	B2	Encroachment	Juniper	60	18	6
	B3	Permanent Shrub	Rhododendron	50	21	4
AD	A1	Permanent Grass	Juniper	6	24	14
	A2	Encroachment	Rhododendron	35	22	8
	A3	Permanent Grass	Juniper	3	18	14
Visited post-classification						
SK	S1	Encroachment	Rhododendron	65	4.7	2
	S2	Permanent Shrub	Rhododendron	100	23	8
	S3	Encroachment	Rhododendron	70	17	8
TB	H1	Encroachment	Rhododendron	100	15	7
	H2	Encroachment	Rhododendron	90	20	9
	H3	Encroachment	Rhododendron	90	14	2
ZH	T1	Encroachment	Rhododendron	75	14	6
Average age of all shrubs					**19**	**11**

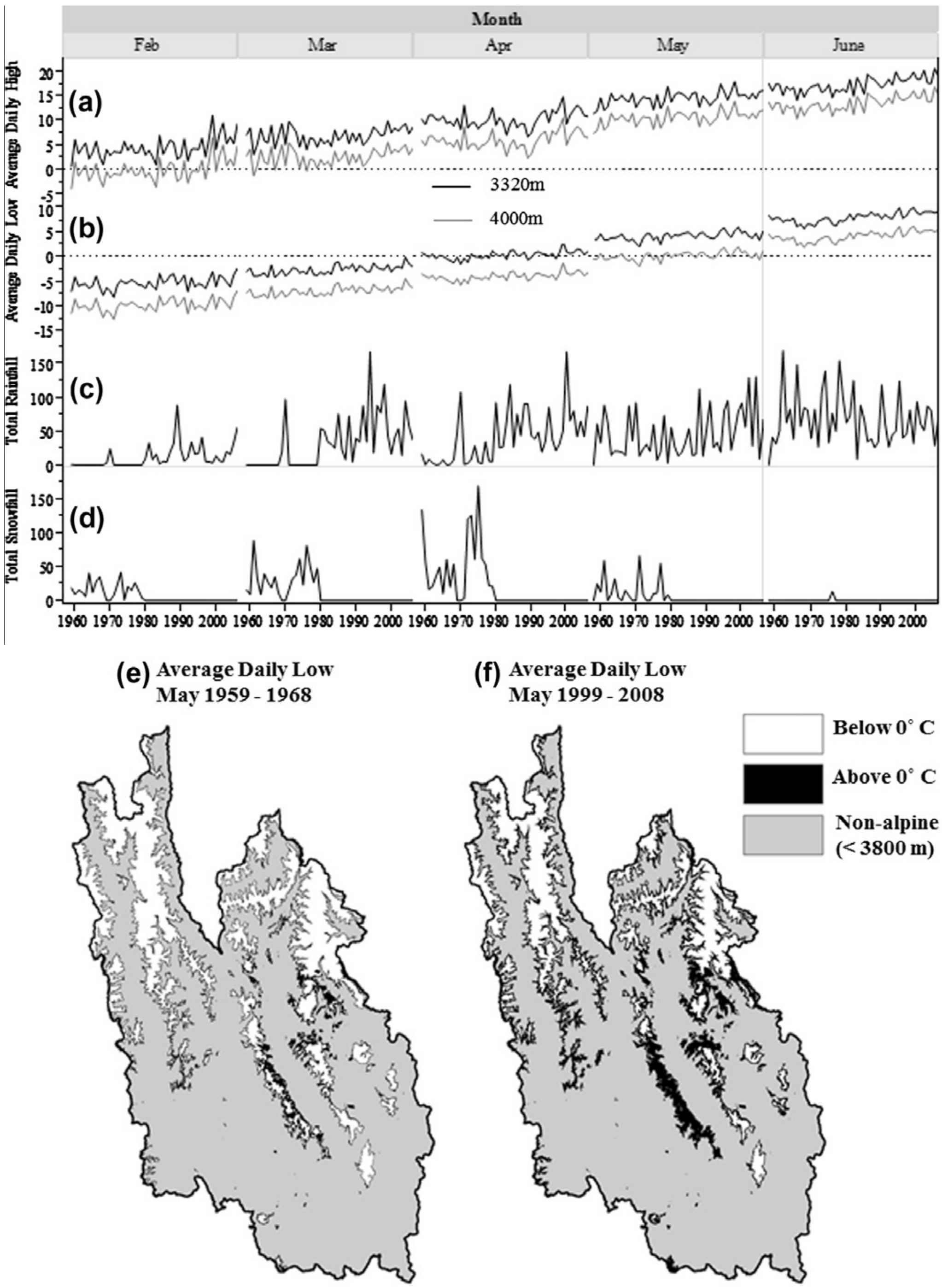

Fig. 4. Average (a) daily low and (b) daily high temperatures (°C) at 3320 m and 4000 m. Total (c) rainfall (mm) and (d) snowfall (mm) measured at Deqin climate station. May average daily low temperatures in the alpine zone from (e) early in the study period (1959–1968) and (f) late in the study period (1999–2008).

endemic plant biodiversity locally (Rejmanek and Rosen, 1992) and regionally (Eldridge et al., 2011; Ratajczak et al., 2012).

4.2. Evidence for a regime shift

The observed patterns of shrub encroachment suggested that the alpine zones of Northwest Yunnan are shifting from an herbaceous to a shrub-dominated ecosystem. Regime shifts occur when an ecosystem shifts suddenly from one state to the other, and the shift cannot easily be reversed even when the original environmental conditions are restored (Brock and Carpenter, 2010; Groffman et al., 2006; Scheffer, 2009).

There were three lines of evidence suggesting that a regime shift occurred. First, hypothesis testing revealed that shrub encroachment was influenced by multiple factors.

H1. *Climate change*: Our results indicated that increasing temperatures since the 1970s in our study area contributed to shrub encroachment. This was not surprising, because climatic change has been linked to shrub encroachment in alpine and arctic ecosystems around the world (Elmendorf et al., 2012a,b; Harte and

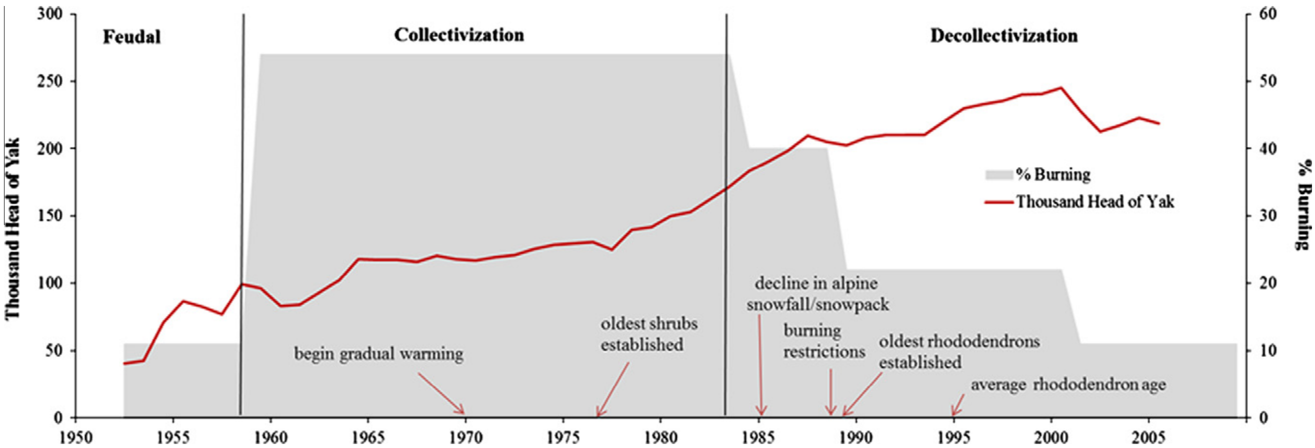

Fig. 5. Synthesis of land cover, land use, and climate change from 1951 to 2009.

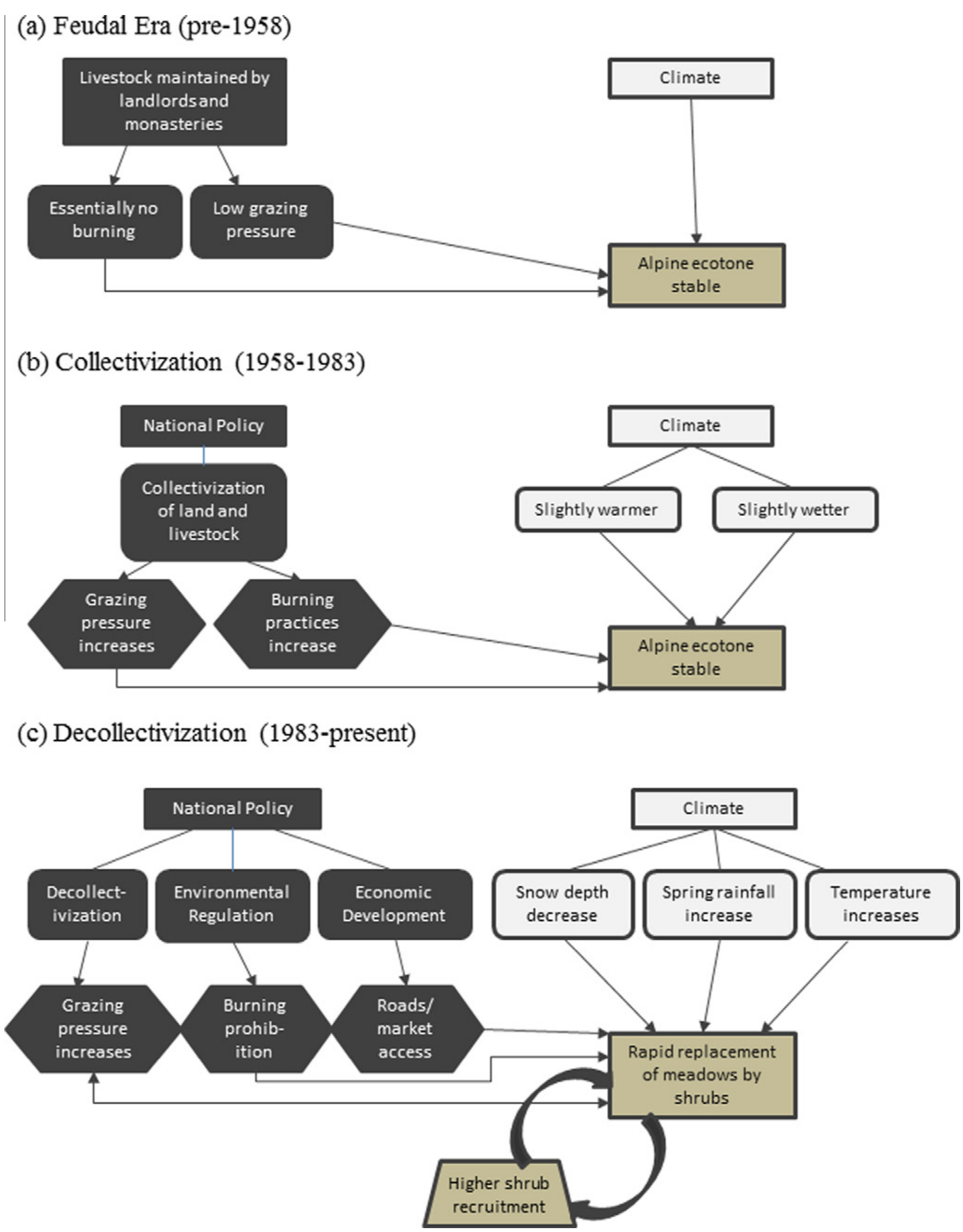

Fig. 6. Interactions between social, climate and alpine sub-systems during (a) the feudal era, (b) the collectivization period and (c) following decollectivization.

Shaw, 1995; Klein et al., 2007). In particular, decreasing snow cover gives a competitive advantage to shrubs over herbaceous species in alpine and arctic environments (Bjork and Molau, 2007; Wipf and Rixen, 2010), and our results indicated that snow cover and shrub encroachment were highly correlated. Furthermore, our alpine temperature estimates indicated that the most dynamic temperature changes occurred during late spring (May) in the mid 1980s when average daily low temperatures rose above freezing, which may have triggered the initial establishment of dwarf rhododendrons in 1988.

H2. *Burning cessation*: The results of our dendrochronological analysis suggested that the region-wide burning ban in 1988 may have fostered initial establishment of dwarf rhododendrons that then spread rapidly. However, contrary to our expectations, burning does not appear to have driven patterns of shrub encroachment. Rates of shrub expansion were higher in areas where burning continued relative to those that did not burn. Studies in other regions have demonstrated that following a shift from grassland to shrubland, shrub cover increases regardless of fire frequency, and some shrub species actually spread faster with burning (Briggs et al., 2005), likely due to fundamental changes in ecosystem structure and function that accompany such an ecosystem shift (Bond and Keeley, 2005).

H3. *Shrub autocatalysis*: Our results suggested that a strong feedback mechanism influenced shrub encroachment rates and patterns, because encroachment rates were higher in areas where woody vegetation was already prevalent. Shrub autocatalysis is a result of positive feedbacks, and common in shrub expansions worldwide (Sturm et al., 2005). It is not possible to determine the exact processes at play in our study area, but spatially-structured (Coop and Givnish, 2007), micro-climatic (D'Odorico et al., 2010) and biotic (Dullinger et al., 2003) feedback mechanisms seem all plausible (D'Odorico et al., 2011).

H4. *Overgrazing*: Overgrazing appears to have accelerated shrub expansion. However, the temporal and spatial relationships between grazing intensity and shrub expansion suggest that grazing effects were not linear, because although grazing intensity increased gradually since the 1950s, rapid shrub encroachment did not occur until after 1990.

H5. *Abandonment*: We found no evidence that land abandonment influenced shrub encroachment. However, land abandonment has, as of yet, been a local and recent phenomenon, unlike the widespread rural exodus that resulted in shrub encroachment in the European Alps.

In summary, our results indicated that 4 out of 5 of our hypothesized drivers influenced shrub establishment and encroachment rates. Multiple causality is the rule in regime shifts, with grassland ecosystems notoriously vulnerable to state changes when both the natural and human systems that affects them change. The presence of multiple drivers of shrub encroachment in our study area is the first line of evidence suggesting a regime shift.

The second line of evidence for a regime shift is the presence of a critical threshold. Typically, prior to a regime shift, perturbations accumulate and start to interact, gradually reducing system resilience before changes arise. After a critical threshold is reached, even a small perturbation can trigger a rapid shift to an alternative stable state. In NW Yunnan, despite increasing stress, we found no record of change in the alpine grasslands from 1950 to the 1980s. However, in the mid- to late 1980s dwarf rhododendron shrubs suddenly established and then spread rapidly.

Initial rhododendron establishment across several alpine territories appears to correspond to two potential triggers. First, May daily minimum temperatures at 4000 m shifted from sub-zero to above-freezing levels in the mid 1980s, likely reducing snowpack and snowfall. Increases in late-spring soil temperature and moisture can provide a competitive advantage for shrubs (Bjork and Molau, 2007; Wipf and Rixen, 2010). Second, the 1988 burning ban immediately reduced burning by about 20%, providing another potential trigger.

A final indicator of a regime shift is evidence for a strong feedback mechanism. A regime shift requires a strong feedback to push a shifting system towards the alternative stable state from which reversion to the original state is difficult. Our results indicated that shrubs spread most rapidly in areas with some prior woody vegetation (shrub autocatalysis). Although burning apparently served historically to effectively control shrubs, burning may no longer do so. In fact, dwarf rhododendrons quickly established and colonized many recently burned (<3 years) plots regardless of the original vegetation, supporting the idea that a regime shift occurred.

4.3. Broader impacts

Shrub encroachment has major implications for ecosystem processes, structure and function, including reduced herbaceous plant biomass and species richness (Ratajczak et al., 2012), altered patterns of soil resource availability and distribution (D'Odorico et al., 2011; Schlesinger et al., 1990) and changes in ecosystem net primary productivity and nutrient balances (Barger et al., 2011; Knapp et al., 2008; Montane et al., 2007). Shifts from one ecosystem to another often start in ecotones, making ecotones important sentinels of change (Hufkens et al., 2009). Our study area straddles the forest-shrub-grassland ecotone at the southern edge of the Himalayas, and may represent a sentinel for the rest of the region. Vegetation shifts at the scale of the Greater Himalayas would have major implications. The high mountains and plateaus of this region regulate the region's climate while serving as the headwaters of three major rivers in Asia. Shrub encroachment alters regional climate systems (Swann et al., 2010) and has significant effects on ecosystem hydrology and biogeochemistry (Knapp et al., 2008). This region is a center of diversity for endemic plants and mammals (Xu et al., 2009), and shrub invasion often leads to a decrease in diversity (Ratajczak et al., 2012). In addition, shrub encroachment forces grazers to rely more on forests, with negative consequences for forest ecosystem diversity (Yi et al., 2007). Therefore, although the exact consequences of shrub encroachment on ecosystem functioning in NW Yunnan alpine ecosystems is not well understood, it is likely that the observed changes have far-reaching implications at multiple spatial scales.

4.4. Management recommendations

Traditional herding systems in NW Yunnan sustained both local livelihoods and endemic biodiversity. However, the socio-economic and climatic conditions have changed dramatically. In the past, grazing and burning both appeared to be compatible with sustaining alpine ecosystems. Presently, burning and grazing may be less sustainable, given climate change and heavy shrub encroachment (Bond and Keeley, 2005). For example, although fire was historically the primary tool for shrub control, our results indicated that it no longer slows shrub expansion. In addition, although our results indicated that areas with the highest grazing pressure had accelerated shrub encroachment rates, experimental studies on the Tibetan Plateau show that moderate levels of grazing can mitigate warming-induced shrub expansion (Klein et al., 2007). This leaves it unclear how yak herding is affecting current alpine environments. Herders, officials, and scientists should together perform *in situ* experimental research to understand grazing thresholds, post-fire regeneration processes, and alternative shrub control strategies. Furthermore, seed collection and seed bank storage, coupled with *ex situ* propagation in botanical gardens throughout the region (Sang et al., 2011), may be needed to preserve threatened endemic plants for future alpine meadow restoration.

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References

Baker, B.B., Moseley, R.K., 2007. Advancing treeline and retreating glaciers: implications for conservation in Yunnan, P.R. China. Arct. Antarct. Alp. Res. 39, 200–209.
Barger, N.N., Archer, S.R., Campbell, J.L., Huang, C.Y., Morton, J.A., Knapp, A.K., 2011. Woody plant proliferation in North American drylands: a synthesis of impacts on ecosystem carbon balance. J. Geophys. Res.-Biogeosci. 116.
Barry, R., Chorley, R., 1987. Atmosphere, Weather, and Climate, fifth ed. Routledge.
Baur, B., Cremene, C., Groza, G., Rakosy, L., Schileyko, A.A., Baur, A., Stoll, P., Erhardt, A., 2006. Effects of abandonment of subalpine hay meadows on plant and invertebrate diversity in Transylvania, Romania. Biol. Conserv. 132, 261–273.
Bjork, R., Molau, U., 2007. Ecology of alpine snowbeds and the impact of global change. Arct. Antarct. Alp. Res. 39, 34–43.
Bond, W.J., Keeley, J.E., 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. Trends Ecol. Evol. 20, 387–394.
Brandt, J.S., Kuemmerle, T., Li, H., Ren, G., Zhu, J., Radeloff, V.C., 2012. Using Landsat imagery to map forest change in southwest China in response to the national logging ban and ecotourism development. Rem. Sens. Environ. 121, 358–369.
Briggs, J.M., Knapp, A.K., Blair, J.M., Heisler, J.L., Hoch, G.A., Lett, M.S., McCarron, J.K., 2005. An ecosystem in transition. Causes and consequences of the conversion of mesic grassland to shrubland. Bioscience 55, 243–254.
Brock, W.A., Carpenter, S.R., 2010. Interacting regime shifts in ecosystems: implication for early warnings. Ecol. Monogr. 80, 353–367.
Card, D., 1982. Using known map category marginal frequencies to improve estimates of thematic map accuracy. Photogramm. Eng. Rem. Sens. 48, 431–439.
Cook, E., Kairlukstis, L. (Eds.), 1990. Methods of Dendrochronology: Applications in the Environmental Sciences. Kluwer, Dordrecht.
Coop, J.D., Givnish, T.J., 2007. Spatial and temporal patterns of recent forest encroachment in montane grasslands of the Valles Caldera, New Mexico, USA. J. Biogeogr. 34, 914–927.
Deqin, 1997. Deqin County Committee Compiled Records, People's Republic of China Chronicles Series, Kunming, China.
D'Odorico, P., Fuentes, J.D., Pockman, W.T., Collins, S.L., He, Y.F., Medeiros, J.S., DeWekker, S., Litvak, M.E., 2010. Positive feedback between microclimate and shrub encroachment in the northern Chihuahuan desert. Ecosphere 1, art17.
D'Odorico, P., Okin, G.S., Bestelmeyer, B.T., 2011. A synthetic review of feedbacks and drivers of shrub encroachment in arid grasslands. Ecohydrology.
Dodson, R., Marks, D., 1997. Daily air temperature iinterpolated at high spatial resolution over a large mountainous region. Clim. Res. 8.
Dullinger, S., Dirnbock, T., Grabherr, G., 2003. Patterns of shrub invasion into high mountain grasslands of the Northern Calcareous Alps, Austria. Arct. Antarct. Alp. Res. 35, 434–441.
Eldridge, D.J., Bowker, M.A., Maestre, F.T., Roger, E., Reynolds, J.F., Whitford, W.G., 2011. Impacts of shrub encroachment on ecosystem structure and functioning: towards a global synthesis. Ecol. Lett. 14, 709–722.
Elmendorf, S.C., Henry, G.H.R., Hollister, R.D., Bjork, R.G., Bjorkman, A.D., Callaghan, T.V., Collier, L.S., Cooper, E.J., Cornelissen, J.H.C., Day, T.A., Fosaa, A.M., Gould, W.A., Gretarsdottir, J., Harte, J., Hermanutz, L., Hik, D.S., Hofgaard, A., Jarrad, F., Jonsdottir, I.S., Keuper, F., Klanderud, K., Klein, J.A., Koh, S., Kudo, G., Lang, S.I., Loewen, V., May, J.L., Mercado, J., Michelsen, A., Molau, U., Myers-Smith, I.H., Oberbauer, S.F., Pieper, S., Post, E., Rixen, C., Robinson, C.H., Schmidt, N.M., Shaver, G.R., Stenstrom, A., Tolvanen, A., Totland, O., Troxler, T., Wahren, C.H., Webber, P.J., Welker, J.M., Wookey, P.A., 2012a. Global assessment of experimental climate warming on tundra vegetation: heterogeneity over space and time. Ecol. Lett. 15, 164–175.
Elmendorf, S.C., Henry, G.H.R., Hollister, R.D., Bjork, R.G., Boulanger-Lapointe, N., Cooper, E.J., Cornelissen, J.H.C., Day, T.A., Dorrepaal, E., Elumeeva, T.G., Gill, M., Gould, W.A., Harte, J., Hik, D.S., Hofgaard, A., Johnson, D.R., Johnstone, J.F., Jonsdottir, I.S., Jorgenson, J.C., Klanderud, K., Klein, J.A., Koh, S., Kudo, G., Lara, M., Levesque, E., Magnusson, B., May, J.L., Mercado-Diaz, J.A., Michelsen, A., Molau, U., Myers-Smith, I.H., Oberbauer, S.F., Onipchenko, V.G., Rixen, C., Martin Schmidt, N., Shaver, G.R., Spasojevic, M.J., Orhallsdottir, O.E., Tolvanen, A., Troxler, T., Tweedie, C.E., Villareal, S., Wahren, C.-H., Walker, X., Webber, P.J., Welker, J.M., Wipf, S., 2012b. Plot-scale evidence of tundra vegetation change and links to recent summer warming. Nat. Clim. Change (advance online publication).
Foody, G.M., Mathur, A., 2004. Toward intelligent training of supervised image classifications: directing training data acquisition for SVM classification. Rem. Sens. Environ. 93, 107–117.
Galop, D., Houet, T., Mazier, F., Leroux, G., Rius, D., 2011. Grazing activities and biodiversity history in the Pyrenees: new insights on high altitude ecosystems in the framework of a human–environment observatory. PAGES News 19, 53–55.
Gao, J., Williams, M.W., Fu, X., Wang, G., Gong, T., 2012. Spatiotemporal distribution of snow in eastern Tibet and the response to climate change. Rem. Sens. Environ. 121, 1–9.
Groffman, P., Baron, J., Blett, T., Gold, A., Goodman, I., Gunderson, L., Levinson, B., Palmer, M., Paerl, H., Peterson, G., Poff, N., Rejeski, D., Reynolds, J., Turner, M., Weathers, K., Wiens, J., 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1–13.
Harris, R.B., 2010. Rangeland degradation on the Qinghai–Tibetan plateau: a review of the evidence of its magnitude and causes. J. Arid Environ. 74, 1–12.
Harte, J., Shaw, R., 1995. Shifting dominance within a montane vegetation community: results of a climate-warming experiment. Science 267, 876–880.
Haynes, M., 2011. Impacts of a Changing Climate and Yak Herding Practices on alpIne Rangelands and Tibetan Livelihoods in Southwest China. PhD Thesis in Botany. University of Wisconsin – Madison, Madison, WI.
Haynes, M., Fang, Z., Waller, D., in press. Grazing Impacts on the Diversity and Composition of Alpine Rangelands in Northwest Yunnan. J. Plant Ecol. http://dx.doi.org/10.1093/jpe/rts021.
Huang, C., Davis, L., Townshend, J., 2002. An assessment of support vector machines for land cover classification. Int. J. Remote Sens. 23, 725–749.
Hufkens, K., Scheunders, P., Ceulemans, R., 2009. Ecotones: methodology and definitions revisited. Ecol. Res. 24, 977–986.
Justice, C., Townshend, J., 1981. Integrating ground data with remote sensing. In: Townshend, J. (Ed.), Terrain Analysis and Remote Sensing. Allen and Unwin, Boston, USA, pp. 38–58.
Klein, J.A., Harte, J., Zhao, X.Q., 2007. Experimental warming, not grazing, decreases rangeland quality on the Tibetan Plateau. Ecol. Appl. 17, 541–557.
Klein, J., Yeh, E., Bump, J., Nyima, Y., Hopping, K., 2011. Coordinating environmental protection and climate change adaptation policy in resource-dependent communities: a case study from the Tibetan Plateau. In: Ford, J.D.F.a.L.B. (Ed.), Climate Change Adaptation in Developed Nations. Springer.
Knapp, A.K., Briggs, J.M., Collins, S.L., Archer, S.R., Bret-Harte, M.S., Ewers, B.E., Peters, D.P., Young, D.R., Shaver, G.R., Pendall, E., Cleary, M.B., 2008. Shrub encroachment in North American grasslands: shifts in growth form dominance rapidly alters control of ecosystem carbon inputs. Glob. Change Biol. 14, 615–623.
Kuemmerle, T., Radeloff, V., Perzanowski, K., Hostert, P., 2006. Cross-border comparison of land cover and landscape pattern in Eastern Europe using a hybrid classification technique. Remote Sens. Environ. 103, 449–464.
Laiolo, P., Dondero, F., Ciliento, E., Rolando, A., 2004. Consequences of pastoral abandonment for the structure and diversity of the alpine avifauna. J. Appl. Ecol. 41, 294–304.
Le'anwangdui, 2001. Diqing Tibetan Autonomous Prefecture Records, People's Republic of China chronicles Series. Yunnan Minorities Press, Kunming, China.
Leemans, R., Cramer, W., 1991. The IIASA Database for Mean Monthly Values of Temperature, Precipitation and Cloudiness on a Global Terrestrial Grid. IIASA Research, Report RR-91-18.
Li, X., Walker, D., 1986. The plant geography of Yunnan Province, Southwest China. J. Biogeogr. 13, 367–397.
Liang, E., Eckstein, D., 2009. Dendrochronological potential of the alpine shrub Rhododendron nivale on the south-eastern Tibetan Plateau. Ann. Bot. 104, 665–670.
Liang, E., Lu, X.M., Ren, P., Li, X.X., Zhu, L.P., Eckstein, D., 2012. Annual increments of juniper dwarf shrubs above the tree line on the central Tibetan Plateau: a useful climatic proxy. Ann. Bot. 109, 721–728.
Lozowski, E.P., Charlton, R.B., Nguyen, C.D., Wilson, J.D., 1989. The use of cumulative monthly mean temperature anomalies in the analysis of local interannual climate variability. J. Clim. 2.
MacArthur, R., Wilson, E., 1967. The Theory of Island Biogeography. Princeton University Press, Princeton, NJ.
Molau, U., 2010. Long-term impacts of observed and induced climate change on tussock tundra near its southern limit in northern Sweden. Plant Ecol. Divers. 3, 29–34.
Montane, F., Rovira, P., Casals, P., 2007. Shrub encroachment into mesic mountain grasslands in the Iberian peninsula: effects of plant quality and temperature on soil C and N stocks. Global Biogeochem. Cycl. 21.
Myers, N., Mittermeier, R., Mittermeier, C., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.
Parmesan, C., 2006. Ecological and evolutionary responses to recent climate change. Annu. Rev. Ecol. Evol. Syst. 37, 637–669.
Peng, S.S., Piao, S.L., Ciais, P., Fang, J.Y., Wang, X.H., 2010. Change in winter snow depth and its impacts on vegetation in China. Glob. Change Biol. 16, 3004–3013.
Ratajczak, Z., Nippert, J.B., Hartman, J.C., Ocheltree, T.W., 2011. Positive feedbacks amplify rates of woody encroachment in mesic tallgrass prairie. Ecosphere 2, art121.

Ratajczak, Z., Nippert, J.B., Collins, S.L., 2012. Woody encroachment decreases diversity across North American grasslands and savannas. Ecology 93, 697–703.
Rejmanek, M., Rosen, E., 1992. Influence of colonizing shrubs on species-area relationships in Alvar plant-communities. J. Veg. Sci. 3, 625–630.
Rolland, C., 2003. Spatial and seasonal variations of air temperature lapse rates in alpine regions. J. Clim. 16.
Roques, K.G., O'Connor, T.G., Watkinson, A.R., 2001. Dynamics of shrub encroachment in an African savanna: relative influences of fire, herbivory, rainfall and density dependence. J. Appl. Ecol. 38, 268–280.
Salick, J., Anderson, D., Woo, J., Sherman, R., Cili, N., Na, A., Dorje, S., 2004. Tibetan ethnobotany and gradient analyses: Menri (Medicine Mountains), Eastern Himalayas. In: Millennium Ecosystem Assessment: Bridging Scales and Epistemologies, Alexandria, Egypt.
Salick, J., Zhendong, F., Byg, A., 2009. Eastern Himalayan alpine plant ecology, Tibetan ethnobotany, and climate change. Global Environ. Change 19, 147–155.
Sang, W., Ma, K., Axmacher, J., 2011. Securing a future for China's wild plant resources. Bioscience 61, 720–725.
Sankey, T., Germino, M., 2008. Assessment of juniper encroachment with the use of satellite imagery and geospatial data. Rangeland Ecol. Manage. 61, 412–418.
Scheffer, M., 2009. Critical Transitions in Nature and Society. Princeton University Press, Princeton, NJ.
Schlesinger, W.H., Reynolds, J.F., Cunningham, G.L., Huenneke, L.F., Jarrell, W.M., Virginia, R.A., Whitford, W.G., 1990. Biological feedbacks in global desertification. Science 247, 1043–1048.
Sherman, R., Mullen, R., Li, H., Fang, Z., Wang, Y., 2008. Spatial patterns of plant diversity and communities in alpine ecosystems of the Hengduan Mountains, northwest Yunnan, China. J. Plant Ecol. 1, 117–136.
Sturm, M., Schimel, J., Michaelson, G., Welker, J.M., Oberbauer, S.F., Liston, G.E., Fahnestock, J., Romanovsky, V.E., 2005. Winter biological processes could help convert arctic tundra to shrubland. Bioscience 55, 17–26.
Swann, A.L., Fung, I.Y., Levis, S., Bonan, G.B., Doney, S.C., 2010. Changes in Arctic vegetation amplify high-latitude warming through the greenhouse effect. Proc. Nat. Acad. Sci. U.S.A. 107, 1295–1300.
Thompson, J., 2007. Mountain Meadows – Here Today, Gone Tomorrow? Meadow Science And restoration. Science Findings – Pacific Northwest Research Station, USDA Forest Service, 5pp.
Thuiller, W., Lavorel, S., Araujo, M.B., Sykes, M.T., Prentice, I.C., 2005. Climate change threats to plant diversity in Europe. Proc. Nat. Acad. Sci. U.S.A. 102, 8245–8250.
Walker, M., Wahren, C., 2006. Plant community responses to experimental warming across the tundra biome. Proc. Nat. Acad. Sci. U.S.A. 103, 1342–1346.
Wipf, S., Rixen, C., 2010. A review of snow manipulation experiments in Arctic and alpine tundra ecosystems. Polar Res. 29, 95–109.
Wolter, P., Mladenoff, D., Host, G., Crow, T., 1995. Improved forest classification in the Northern Lake States using multi-temporal landsat imagery. Photogramm. Eng. Rem. Sens. 61, 1129–1143.
Wong, M., Duan, C., Long, Y., Luo, Y., Xie, G., 2010. How will the distribution and size of subalpine Abies Georgei forest respond to climate change? A study in Nowrthwest Yunnnan, China. Phys. Geogr. 31, 319–335.
Wu, Z., Hang, S., Zhou, K., Hua, P., Li, D., 2007. Origin and differentiation of endemism in the flora of China. Fron. Biol. China 2, 125–143.
Xu, J., Wilkes, A., 2004. Biodiversity impact analysis in northwest Yunnan, southwest China. Biodivers. Conserv. 13, 959–983.
Xu, J., Grumbine, R.E., Shrestha, A., Eriksson, M., Yang, X., Wang, Y., Wilkes, A., 2009. The melting Himalayas: cascading effects of climate change on water, biodiversity, and livelihoods. Conserv. Biol. 23, 520–530.
Yeh, E., Gaerrang, 2010. Tibetan pastoralism in neoliberalising China: continuity and change in Gouli. Area 43, 165–172.
Yi, S., Wu, N., Luo, P., Wang, Q., Shi, F., Sun, G., Ma, J., 2007. Changes in livestock migration patterns in a Tibetan-style agropastoral system. Mount. Res. Dev. 27, 138–145.
Yu, H., Luedeling, E., Xu, J., 2010. Winter and spring warming result in delayed phenology on the Tibetan Plateau. In: Proceedings of the National Academy of Sciences of the United States of America Early Edition.