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Human Impacts on Forest Biodiversity in Protected Walnut-Fruit Forests in Kyrgyzstan

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We used a spatially explicit model of forest dynamics, supported by empirical field data and socioeconomic data, to examine the impacts of human disturbances on a protected forest landscape in Kyrgyzstan. Local use of 27 fruit and nut species was recorded and modeled. Results indicated that in the presence of fuelwood cutting with or without grazing, species of high socioeconomic importance such as Juglans regia, Malus *spp., and* Armeniaca vulgaris *were largely eliminated from the landscape after 50–150 yr. In the absence of disturbance or in the presence of grazing only, decline of these species occurred at a much lower rate, owing to competitive interactions between tree species. This suggests that the current intensity of fuelwood harvesting is not sustainable. Conversely, current grazing intensities were found to have relatively little impact on forest structure and composition, and could potentially play a positive role in supporting regeneration of tree species. These results indicate that both positive and negative impacts on biodiversity can arise from human populations living within a protected area. Potentially, these could be reconciled through the development of participatory approaches to conservation management within this*

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reserve, to ensure the maintenance of its high conservation value while meeting human needs.

KEYWORDS anthropogenic disturbance, biodiversity conservation, forest landscape model, Kyrgyzstan, Landis-II, protected area, threatened species, walnut-fruit forests

INTRODUCTION

Native forests are believed to contain more than half of terrestrial animal and plant species (Secretariat of the Convention on Biological Diversity [SCBD], 2010), and are therefore considered to be of high conservation value worldwide (Newton, 2007a). Worldwide, the area of forest where the conservation of biodiversity is designated as the primary function has increased by 30%since 1990, and now accounts for 12% of the total forest area or more than 460 million ha (Food and Agriculture Organization of the United Nations [FAO], 2010). However, in many areas, biological diverse forests are being subjected to intensive human disturbance, through activities such as the cutting and burning of vegetation and browsing by livestock. Such processes can result in forest clearance, degradation and fragmentation, and consequent loss of biodiversity (Newton, 2007b; Newton et al., 2009). In addition there is a high risk that these ecosystems will continue to degrade as human population increases (Cincotta, Wisnewski, & Engelman, 2000). The establishment of protected areas is widely considered to be the most important approach for preventing such biodiversity loss. Most of the 460 million ha of forest that has been designated for biodiversity conservation are located inside legally established protected areas (FAO, 2010). Given the strong dependence of conservation strategies on protected areas, and the substantial investments made in implementing them, it is important to understand the factors influencing their effectiveness (Gaston, Jackson, Cantú-Salazar, & Cruz-Piñón, 2008). The need for this understanding is urgent, given that many protected areas are currently under threat from human disturbance (Carey, Dudley, & Stolton, 2000; Chape, Harrison, Spalding, & Lysenko, 2005).

If protected areas are to be effective, they will need to be capable of absorbing disturbance while maintaining their function and value (Gunderson, 2000). Approaches to protected area management are therefore required that enable conservation objectives to be achieved while ensuring that human needs are met. This can potentially be achieved by viewing protected areas as dynamic landscapes, in which human activities are an integral element (Bengtsson et al., 2003). This is consistent with a recently developed paradigm for protected area management, in which meeting the needs of local people is viewed as a central component (Phillips, 2003). Features of this new paradigm include management for socioeconomic objectives as well as biodiversity conservation, as illustrated by the development of community-based, participatory, and collaborative approaches to protected area management (Lockwood, Worboys, & Kothari, 2006; Kassa et al., 2009).

In order for such approaches to be successful, they need to be based on an understanding of the impacts of human disturbance on biodiversity. From the perspective of practical conservation management, tools are required that can be used to forecast the impacts of human disturbance on forest communities (Newton et al., 2009). Such tools would enable appropriate management interventions to be identified, and inform the development of effective conservation plans. While understanding of the impacts of anthropogenic disturbance on forest biodiversity is still limited (Newton & Echeverría, 2014), disturbance impacts can potentially be forecast using appropriate modeling approaches. In this investigation, we employ a spatially explicit model (LANDIS-II), which is designed to simulate forests dynamics through the incorporation of spatial processes including succession, seed dispersal, and multiple types of disturbances (Mladenoff, 2004; Scheller et al., 2007). LANDIS-II has been widely applied in different parts of the world (Scheller et al., 2007), increasingly in a conservation context. For example, Newton, Echeverría, Cantarello, and Bolados, (2011) explored the application of LANDIS-II to support systematic conservation planning in a dryland environment in Chile, and Cantarello et al. (2011) employed this model to assess the potential for forest restoration in two Mexican landscapes under different disturbance regimes.

This research was undertaken in a protected walnut-fruit forest in the Tien Shan region of Kyrgyzstan, an area of high conservation value, as illustrated by its designation as a Global 200 ecoregion (Olson et al., 2001) and a global biodiversity hotspot (Myers, Mittermeier, Mittermeier, da Fonseca, & Kent, 2000). Walnut-fruits forests in this region have played a significant role in human history and culture, being the source of domesticated tree crops that are now widely cultivated, and are therefore of exceptional importance as a genetic resource (Hemery & Popov, 1998; Harris, Robinson, & Juniper, 2002; Juniper & Mabberley, 2006; Orozumbekov, 2011). These forests also play a major role in soil protection and in regulating water flow, which is crucial for the irrigation of the fertile and densely populated Fergana Valley (Musuraliev, 1998; Orozumbekov et al., 2009). Intensive use of the walnut-fruit forests has led to concerns that unsustainable patterns of land use have been a major cause of forest loss and degradation. Evidence suggests that timber harvesting in the early 20th century may have caused widespread deforestation as well as the degradation of forest stands (Hemery & Popov, 1998; Venglovsky, 1998). It has also been suggested that intensive collection of walnuts for consumption or sale in local markets may account for observed regeneration failures of walnuts (Herold, 2005); other potential factors include haymaking activities in forest clearings and overgrazing by forest livestock, which are estimated to number around 49,000 animals

(Hemery & Popov, 1998). By comparison, some 50,000 people live within the walnut-fruit forests (Musuraliev, 1998).

In this investigation we used LANDIS-II, supported by empirical field data, to examine the impact of human activities on the dynamics of walnut-fruit forests and its associated biodiversity, with the aim of informing conservation management plans. Model simulations included fuelwood cutting and grazing by livestock, as these disturbances are believed to have caused the most serious degradation of walnut-fruit forests in Kyrgyzstan (Sherbinina, 1998). Specifically, the research addressed the following questions: (a) Do current levels of grazing and fuelwood cutting cause negative impacts on forest biodiversity? (b) Do grazing and fuelwood cutting interact? (c) Can analyses of disturbance regimes, using the modeling approach adopted here, usefully inform the development of conservation management plans?

METHODS

Study Site

Research was conducted in the Sary-Chelek Biosphere Reserve, in the western Tien Shan mountains of Kyrgyzstan (Figure 1a). The majority of the reserve is surrounded by mountains to the North, East and West, with an altitude ranging between 1,200–4,200 m a.s.l. (Figure 1b). The mountains give the region a unique microclimate with relatively mild but snow-intense winters and warm and wet summers. Habitats in the reserve include forest, meadow, steppe, rock escarpments, and some aquatic systems (i.e., Lake Sary-Chelek). Slopes are typically steep and have frequent rocky outcrops. Almost all inhabitants (approximately 5,000 people; Abdymomunov, 2001) live around the village of Arkyt (Figure 1c).

Research focused on mixed forest dominated by walnut (*Juglans regia*) and associated fruit and nut species (*Malus* spp., *Pyrus* spp., and *Prunus* spp.) that occur at altitudes 1,200–2,100 m a.s.l. in the reserve (Kolov, 1998). This area extends for 7,167 ha, covering 30% of the Biosphere Reserve, from 41° 45′ 51″ to 41° 55′ 45″ N and 71° 53′ 55″ to 72° 00′ 46″ E (Figure 1c). Plant diversity is particularly high as the reserve is situated where two distinctive floral communities converge (Hemery & Popov, 1998), and is reportedly the only place where mixed stands of walnut and juniper can be found (Sherbinina, 1998). Tien-Shan spruce (*Picea shrenkiana*) and Semenov's fir (*Abies semenovii*) also grow at lower altitudes than usual in the reserve owing to lower temperatures and higher rainfall than other walnut-fruit forests (Sherbinina, 1998). Fauna in the reserve includes 35 species of mammals and 157 species of birds—including snow leopard (*Panthera uncia*), brown bear (*Ursus arctos*), Eurasian lynx (*Lynx lynx*), golden eagle (*Aquila chrysaetos*), lammergeier (*Gypaetus barbatus*), and peregrine falcon (*Falco peregrinus*; Farrington, 2005).

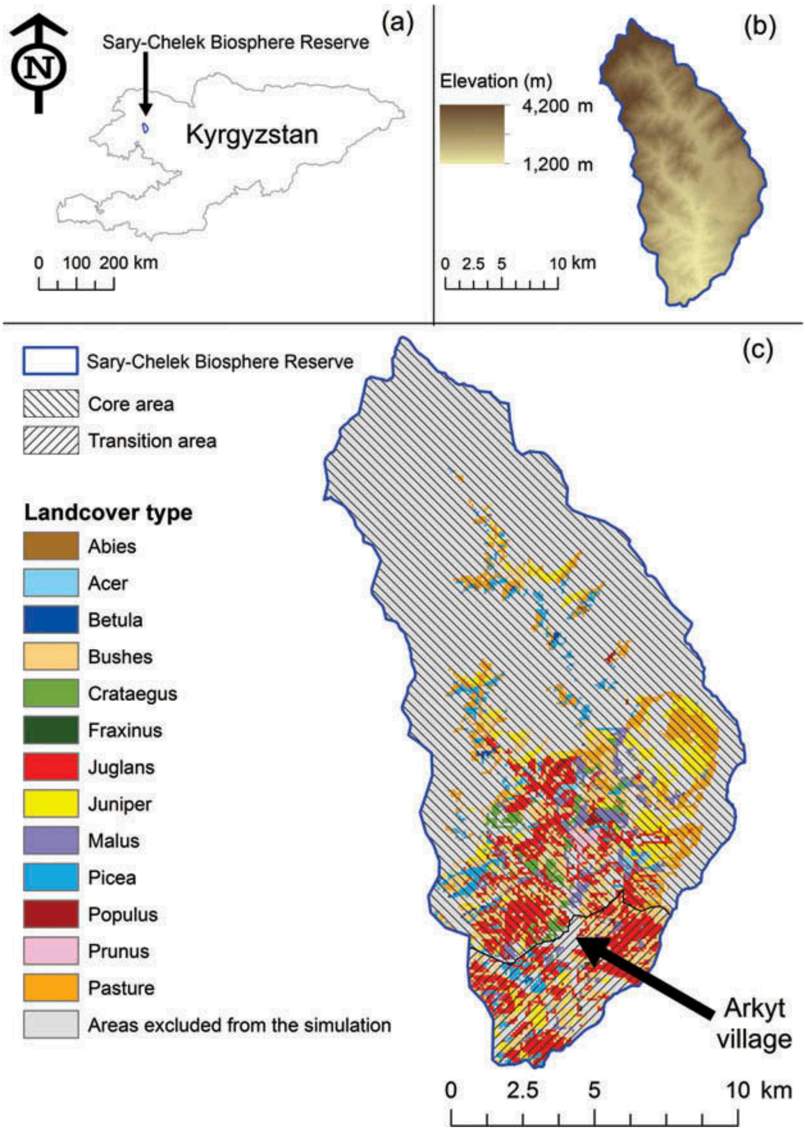

FIGURE 1 (a) Map showing the location of the Sary-Chelek Biosphere Reserve within Kyrgyzstan. (b) Elevation map derived from 50 m contour lines produced by the Kyrgyz Forest Service. (c) Land cover map for the study area simulated; this was derived from the land cover map produced by the Kyrgyz Forest Service and elevation map shown in (b). Grey areas were excluded from the simulation and are represented by sites above 2,100 m a.s.l.—human settlements, lakes and rivers, ice and rocks—where walnut-fruit species cannot establish. Core and transition areas were mapped based on the *Sary-Chelek Forest Management Plan 2003–2013* (see text for more details).

Sary-Chelek was made a National Park in 1945 (Hemery & Popov, 1998), but before the designation of the Biosphere Reserve in 1978, selective logging, grass cutting, and cattle grazing were extensively practiced (United

Nations Educational, Scientific and Cultural Organization [UNESCO], 2000). In efforts to control the impacts of human disturbance, a transition area was designated around the village of Arkyt in 1978, encompassing a total area of 2,214 ha (Figure 1c). The boundary of the transition area is partly delimited by a wire fence with wooden posts running for 4.1 km east to northwest of the village of Arkyt. Tree cutting for fuelwood and timber and grazing livestock (principally cattle, but also horses, donkeys, sheep, and goats; Herold, 2005) are supposed to be restricted to the transition area. However, these activities also occur within the core area owing to limited manpower to enforce rules, and breaches of the buffer fence-line (Sherbinina, 1998). According to UNESCO (2001), cultivation of crops is also prohibited in the core area (Figure 1c), although harvesting of grasses for hay throughout the area was observed in this study. Collection of nuts and fruits, both for consumption and sale, is widespread. These products have played a significant role in the transition to a market economy, following the independence of Kyrgyzstan in 1991. Sale of walnuts is one of the main sources of income to many rural communities in the South Kyrgyzstan's mountain forests (Schmidt, 2005; Schmidt, 2006).

Field Survey Data

The current forest structure and composition was assessed by establishing 32 square plots (0.25 ha) orientated on a northern axis throughout the study area. The location of the southwest corner of each plot was randomly generated within each forest type in the land cover map (Figure 1c) using the ESRI Hawth's Analysis Tools ©2007 Version 3.27 extension within ArcGIS (ESRI Inc., Redlands, CA, USA). The points were located in the field using a Global Position System (GPS) device (Garmin GPS VII, GarminLtd., Southampton, UK). Within each plot, the diameter at breast height (1.3 m dbh) was recorded for all trees ($\geq$10 cm dbh) using a diameter tape (Yamaho Million 12, Yamayo Measuring Tools Co., Ltd., Tokyo, Japan). Trees with multiple stems were recorded individually and counted as separate stems. Nested plots (0.0025–0.25 ha) at the southeast corner were used to measure densities of seedlings, saplings, and shrubs. Seedlings were defined as trees $\leq$1.5 m in height, and saplings as trees $>$1.5 m in height but $<$10 cm dbh, following Newton (2007a). The Flora of China database (n.d.) was used to distinguish shrubs from trees. Wood cores of each species were taken using an increment borer (Suunto, Vantaa, Finland) to provide a representative sample for subsequent determination of age–diameter relationships in the laboratory. Measurements of human disturbance were made by recording the number and species of seedlings and saplings browsed, and the number and species of cut stumps and stems. Species identification for the stumps was established based on bark identification by a local botanist who assisted with the fieldwork.

Socioeconomic Survey

A standardized questionnaire was developed in conjunction with researchers of the Kyrgyz National Agrarian University and Biosphere Reserve staff to determine the extent of harvesting, grazing practices, and use of forest products. The questionnaire included a series of preestablished questions with preset response categories, following Bajracharya, Furley, and Newton (2006). In the majority of cases, questions invited respondents to score the intensity of use of different products using a 4-point Likert scale, where 1 = Never, 2 = Rarely, 3 = Occasionally, and 4 = Often; and to express how the availability of different resources had changed over the previous 10 yr, on a 3-point Likert scale, with 1 = Increase, 2 = Decrease, and 3 = Stayed the same (Appendix A, in the online supplementary information [SI]).

The questionnaire was used as basis for semi-structured interviews in the main village of Arkyt (Figure 1c). Forty-five households were selected randomly, and one individual per household was interviewed. Questions were posed in Russian by local members of the project team. At the start of each interview, the respondent was informed that participation was voluntary, and he or she was asked to answer on behalf of all the people living in the same household. Sixty percent of the people interviewed were men, reflecting the fact that men are the dominant members of the household and were most likely to engage in the interview on behalf of the family. Each household had an average of two adults and four children. The age of respondents displayed a normal distribution with 40% of the respondents being in the 45–54 yr category. All of the respondents had lived in the area for most of their life and were actively involved in using the forest resources.

Model Parameterization and Scenarios

The LANDIS-II model is designed to use a cell-based data format (raster data) as a spatially explicit input to simulate forest dynamics. Within each cell it tracks the presence/absence of tree species age classes (or cohorts) at a time step specified by the user. Vegetation patches can aggregate and disaggregate in response to spatial patterns of stochastic rules of succession and multiple disturbances (including seed dispersal, harvesting, and other management). Forest succession is a competitive process governed by species life history characteristics, and the probability of species establishment on different ecoregions. A detailed description of the LANDIS-II model is provided elsewhere (Scheller et al., 2007; http://www.landis-ii.org).

In the present investigation, raster data included a Digital Elevation Model (DEM), a land cover map including 12 forest types (Figure 1c) and a management areas map. The DEM (90 m cell-size) was derived from the 50 m contour lines provided by the Forestry Planning Office in Bishkek, by using the 3D Analyst Tools in ArcGIS 10.0© (ESRI Inc.). The land cover map (90 m cell-size) was obtained from the rasterization of the vectorial land cover map produced by the Kyrgyz Forest Service in 2002, derived from satellite remote sensing data. The management areas map included three management areas (core, transition, and buffer areas) defined in the Sary-Chelek forest management plan 2003–2013 (Kyrgyz Forest Service, 2003), and the results of the socioeconomic survey, which indicated a buffer zone of 5 km around the Arkyt village as the zone within the core area where human disturbance is present.

LANDIS-II requires an ecoregion map identifying land-types with similar ecological conditions that influence species' ability to establish. In Sary-Chelek, the ecoregions map was derived from the elevation map and the land cover map. Sites above the upper altitudinal limit of *Juglans regia* (i.e., 2,100 m a.s.l.) were excluded from the simulation, as were urban areas, lakes, and steep rocky areas often covered by ice (Figure 1c). Active ecoregions represented ranged between 1,200–2,100 m a.s.l. (Appendix B, in the SI). The species establishment probabilities for each active ecoregion were estimated from a review of the scientific literature (Appendix C, in the SI).

LANDIS-II also requires an initial communities map that specifies the trees species and age classes present at each location at Yr 0 of the simulations. This map was produced by combining the land cover map with the field survey data describing species composition and age distributions. The life history characteristics (longevity, age of sexual maturity, shade tolerance class, effective and maximum seed dispersal distance, vegetative reproduction probability, minimum and maximum age of vegetative reproduction) of the species encountered in the field were extracted from the scientific literature and by consulting local experts (Table 1).

To explore differences in disturbance regimes, four different scenarios were simulated: (a) no disturbance (NO-DIST), (b) grazing (GRAZ), (c) fuelwood cutting (WOODCUT), and (d) grazing and fuelwood cutting combined (GRAZ-WOODCUT). The Base Harvest (v1.2) extension of LANDIS-II (Gustafson, Shifley, Mladenoff, Nimerfro, & He, 2000) was used to generate the harvesting scenarios (i.e., GRAZ, WOODCUT and GRAZ-WOODCUT). A different subset of species was included in each of these scenarios, based on the list of species that had been recorded as being either cut or grazed in the field survey (Table 1). Specific cohorts were removed during each harvest event, with grazing removing any cohorts <10 yr old, and cutting removing a variety of different cohorts, based on typical diameter of stems harvested, obtained from the results of the socioeconomic survey (Appendix D, in the SI). Harvesting impacts were distributed according to the three management areas (i.e., core, transition, and buffer areas) described above. Information on the grazing and cutting pressure (i.e., target area to be grazed/cut in each management area) was obtained by calculating the ratio between the number of saplings and cut stumps per hectare in the transition and buffer

TABLE 1 Details of the Species Characteristics in the Sary-Chelek Study Area, Which Were Included in the LANDIS-II Simulations

Species groups	Species	Long	Mat	ShT	EffD	MaxD	VRP	MinVRP	MaxVRP	Disturb
Abelia corymbosa	Abelcory	30	3	3	400	700	1	2	30	
Abies semenovii[a]	Abieseme	213	30	5	40	120	0	0	0	
Acer semenovii, A. turkestanicum	Acer_spp	157	15	3	87	223	1	50	157	C
Armeniaca vulgaris[a]	Armevulg	138	4	2	400	700	1	10	138	C,G
Berberis nummularia	Berbnumm	60	5	1	400	700	1	2	60	G
Betula pendula, B. tianschanica[a]	Betu_spp	186	22	2	362	2000	1	19	70	G
Caragana spp.	Cara_spp	50	6	3	40	120	0	0	0	
Celtis caucasica	Celtcauc	588	30	2	400	700	1	5	50	C,G
Cerasus mahaleb[b]	Ceramaha	90	4	3	400	700	1	5	50	C,G
Cotoneaster multiflorus, C. oliganthus	Coto_spp	60	3	1	400	700	1	2	60	G
Crataegus turkestanica,[b] *C. pontica*[b]	Crat_spp	250	4	2	217	700	1	5	50	C,G
Euonymus spp.	Euon_spp	50	5	3	400	700	1	2	50	
Exochorda spp.	Exoc_spp.	15	3	2	400	700	1	2	15	G
Fraxinus sogdiana	Fraxsogd	230	33	3	214	560	1	50	70	
Juglans regia[b]	Juglregi	256	12	2	217	400	1	60	100	C,G
Juniperus semiglobosa	Junisemi	650	15	2	400	600	0	0	0	C
Lonicera karelinii	Lonikare	25	3	3	400	700	1	2	25	G
Malus sieversii,[ab] *M. niedzwetzkyana*[ab]	Malu_spp	170	9	1	400	700	1	5	50	C,G
Picea schrenkiana[ab]	Piceschr	250	30	3	40	120	0	0	0	C
Populus alba	Popualba	350	8	2	255	4550	1	16	190	C
Prunus sogdiana,[b] *P. communis*[b]	Prun_spp	138	4	2	400	700	1	5	50	C,G
Pyrus communis,[b] *P. turkomanica,*[b] *P. korshinskyi*[ab]	Pyru_spp	100	4	1	400	700	0	0	0	C,G
Rhamnus cathartica	Rhamcath	44	9	4	400	700	1	2	44	

Rosa spp.	Rosacani	15	3	2	400	700	1	2	15	C,G
Rubus spp.	Rubuidae	10	2	2	400	700	1	2	10	G
Salix alatavica	Salialat	109	14	1	628	25800	1	2	109	
Sorbus persica, S. tianschanica	Sorb_spp.	250	15	2	217	700	0	0	0	G

Note. Long, longevity (years)—Ref. 2–5, 7–9, 12, 14, 16, 18, 21, 23, 29–31, 33–35, 39–40, 43; Mat, age of maturity (years)—Ref. 2, 4–5, 7–8, 15, 27–28, 30, 32–33, 35–36, 43; ShT, shade tolerance class (1 to 5, with 1 for the most shade intolerant and 5 for the most shade tolerant)—Ref. 2, 4, 7–8, 10, 19, 23, 30, 35, 43; EffSD, effective seeding distance (m)—Ref. 1–2, 4, 7, 11, 13, 20, 22, 24–27, 30, 37, 38, 41, 43; MaxSD, maximum seeding distance (m)—Ref. 2, 6, 13, 20, 25, 30, 35, 38, 42–43; VRP, vegetative reproduction probability—Ref. 2, 4, 7-8, 23, 27, 30, 35, 37, 40, 43; MinVRP, minimum age of vegetative reproduction (years)—Ref. 2, 35, 40, 43; MaxVRP, maximum age of vegetative reproduction (years)—Ref. 17, 35, 43. Disturb refers to disturbance impacts; those species denoted "C" were harvested in the cutting scenario, and those denoted "G" were harvested in the grazing scenario. Those species with [a] and/or [b] superscript were selected to produce outputs of their presence in each pixel, under each of the four scenarios (Figure 4). References: 1. (Atkinson, 1992); 2. (Bu et al., 2008); 3. (Burns, 2004); 4. (CABI, 2005); 5. (Core, 1974); 6. (Cremer, 1999); 7. (Dzhangaliev, 2003); 8. (Dzhangaliev et al., 2003); 9. (Earle, 1997); 10. (Ellenberg, 1988); 11. (Goudzwaard, 2006); 12. (Grashof-Bokdam et al., 1998); 13. (Greene & Calogeropoulos, 2002); 14. (Grime et al., 1988); 15. (Grubb et al., 1999); 16. (Gucker, 2007); 17. (He & Mladenoff, 1999); 18. (Hemery & Popov, 1998); 19. (Jahn, 1991); 20. (Martínez & González-Taboada, 2009); 21. (Mattock, 1994); 22. (McVean, 1955); 23. (Morris & Money, 1996); 24. (Müller-Schneider, 1983); 25. (Müller-Schneider, 1986); 26. (Munger, 2002b); 27. (Munger, 2002a); 28. (Orwa et al., 2009); 29. (Paganová, 2001); 30. (Pennanen & Kuuluvainen, 2002); 31. (Peter, 2008); 32. (Raspe et al., 2000); 33. (Schumacher et al., 2004); 34. (Silander & Klepeis, 1999); 35. (Spencer et al., 2008); 36. (Thomas, 2000); 37. (Tirmenstein, 1990); 38. (Vittoz & Engler, 2007); 39. (T. Wang et al., 2004); 40. (Whitney, 1986); 41. (Willems & Bik, 1998); 42. (Wyckoff & Zasada, 2008); 43. (Xu et al., 2007).

[a] Species identified by the IUCN Red List as being vulnerable, endangered or criticallyendangered (Eastwood et al., 2009).

[b] Species identified as important for the local economy in the socioeconomic survey conducted in Sary-Chelek.

areas and the number of saplings and cut stumps per hectare in the core area, as recorded in the field survey. In this way, the disturbance scenarios were designed to simulate current intensities of human disturbance within the reserve.

LANDIS-II simulations were conducted for 300 yr. Five replicated simulations (with varying random number seed) were performed for each disturbance scenario to explore the variability of model predictions. The time steps were set at 10 yr for tree succession, and 1 yr for grazing and fuelwood cutting. The list of ages for each species was therefore grouped into cohorts as follow: ages 1 to 10 (10), 11 to 20 (20), 21 to 30 (30), etc.

Data Analyses

The Age Cohort Statistics v1.0 extension of LANDIS-II was used to produce outputs of (a) maximum age across all species in each pixel; (b) total number of species in each pixel; and (c) presence of selected species in each pixel, under each of the four scenarios. The following species were selected for detailed analysis based on the results of the socioeconomic survey indicating their importance for the local community: *Cerasus mahaleb, Crataegus turkestanica, C. pontica, Juglans regia, Malus sieversii, M. niedzwetzkyana, Picea schrenkiana, Prunus sogdiana, Prunus communis, Pyrus communis, P. turkomanica, P. korshinskyi, Abies semenovii, Armeniaca vulgaris*, and *Betula tianschanica* were also selected on the basis of their IUCN Red List status, indicating that they are threatened with extinction (Table 1).

The LANDIS-II outputs consist of spatially explicit raster maps, each corresponding to a time step specified by the user (10 yr in this study). To facilitate interpretation of results along the 300-yr simulation, species number data were grouped into three classes: (a) 1–4 species, (b) 5–11 species, and (c) >11 species. Statistical analyses were performed using IBM SPSS Statistics v19© (SPSS Inc., Chicago, IL, USA). Descriptive statistics were used to calculate the mean ($\bar{x}$), standard deviations (*SD*) and standard errors (*SE*) of percent cover of trees and selected species, and to assess the normality of the data (Shapiro-Wilk test). Mann-Whitney and Kruskal-Wallis tests were used to compare forest cover values.

RESULTS

A total of 27 tree species and 9 shrub species were encountered in the field survey, although taxonomic uncertainty over some genera (e.g., *Malus*; Juniper & Mabberley, 2006), led to some congeneric species being grouped under the same species attributes in the model. Therefore, a total of 27 species were entered into LANDIS-II (Table 1). *Malus* spp., *Crataegus* spp., *Prunus* spp., and *Juglans regia* were the most widely distributed tree species, occurring in 81, 78, 63, and 59% of the plots, respectively. *Juglans regia, Malus* spp., and *Crataegus* spp. were characterized by high tree density, but only *Juglans regia* contributed significantly to the basal area. Many of the other species were found in low densities across the forest and in a relatively low percentage (<32%) of plots (Table 2). The mean value of tree species per plot was 4, with values ranging from 1 to 7.

Of the 20 tree species with stems >10 cm dbh, 12 were also recorded as saplings and 15 as seedlings. *Rosa* spp. and *Cotoneaster* spp. were the most frequent shrub species (Table 2). Of the dominant species, *Crataegus* spp. was found in high densities for both seedlings (21%) and saplings (30%), suggesting its ability to grow under current grazing pressure. Conversely, *Malus* spp. and *Juglans regia* displayed low densities of seedlings (0.4 and 4.6%, respectively), and saplings (0.5 and 12%, respectively) compared with the number of mature trees, indicating that recruitment is currently limited.

In all LANDIS-II simulations, the 27 species were initially distributed over an area of 6,148 ha, accounting for 86% of the total study area. Forest cover (defined here as percent cover of tree and shrub species >10 yr age) increased rapidly during the first 50 yr, reaching over 93% of the total study area under all scenarios. Higher values of forest area were subsequently maintained under the NO-DIST and GRAZ scenarios compared to WOODCUT and GRAZ-WOODCUT (Figure 2). The median forest cover under NO-DIST did not statistically differ from the median forest cover under GRAZ ($p > .05$, Mann-Whitney test). Similarly the median forest cover under WOODCUT did not statistically differ from the median forest cover under GRAZ-WOODCUT ($p > .05$, Mann-Whitney test). Variation between the five LANDIS-II runs for each scenario was low and was accounted for ±1.6% in estimates of total forest area only under GRAZ-WOODCUT after 50 yr (Figure 2).

Scenarios differed slightly in their impact on the species richness. The majority of the initial landscape (63%) was occupied by more than 11 species. Over the simulation period, species richness decreased significantly under all scenarios and no forest cover was associated with 11 or more species after 300 yr (Figure 3). Under the WOODCUT and GRAZ-WOODCUT scenarios the forest cover occupied by ≥ 5 species decreased from 64% to less than 5% after 150 yr, whereas this occurred 100 yr later under the NO-DIST and GRAZ scenarios. However, the median percentage cover with ≥ 5 species did not differ significantly between scenarios ($p > .05$ Kruskal-Wallis test), indicating a similar pattern in species richness decline over 300 yr. Simulation results also indicated that by the final timestep the different disturbance regimes had little impact on the abundance of the selected species (as listed in the Data Analyses section). The dominant species *Crataegus* spp., *Juglans regia, Malus* spp., and *Prunus* spp. were largely eliminated from the landscape in all scenarios after 300 yr. The relative cover of simulated landscape occupied by *Crataegus* spp., *Juglans regia, Malus* spp., and *Prunus* spp. were only

TABLE 2 Stand Structure of the Sary-Chelek Walnut-Fruit Forests, Represented as the Density and Basal Area (Means ± *SE*) of Tree Species, and Density (Means ± *SE*) of Shrub Species Encountered in the Field Survey. Number of Plots: 32.

Tree species	Basal area (m^2 ha^{-1})	Trees density (N ha^{-1})	Saplings density (N ha^{-1})	Seedlings density (N ha^{-1})
Abelia corymbosa	0.0 ± 0.0	0.6 ± 0.6	6.2 ± 6.2	—
Abies semenovii	0.1 ± 0.1	3.0 ± 2.9	0.4 ± 0.4	3.1 ± 3.1
Acer semenovii, A. turkestanicum	0.2 ± 0.1	6.2 ± 2.8	1.5 ± 1.5	3.1 ± 3.1
Armeniaca vulgaris	0.0 ± 0.0	0.5 ± 0.3	—	—
Berberis nummularia	0.0 ± 0.0	0.3 ± 0.3	17.6 ± 6.7	170.9 ± 60.2
Betula pendula, B. tianschanica	0.1 ± 0.1	1.5 ± 1.4	—	31.2 ± 29.2
Celtis caucasica	0.0 ± 0.0	1.3 ± 0.7	0.8 ± 0.6	349.7 ± 177.1
Cerasus mahaleb	0.2 ± 0.1	6.0 ± 2.3	55.5 ± 49.9	12.6 ± 12.5
Crataegus turkestanica, C. pontica	1.2 ± 0.3	55.7 ± 12.5	279.4 ± 133.6	440.1 ± 229.6
Fraxinus sogdiana	0.2 ± 0.2	2.4 ± 2.2	—	—
Juglans regia	6.8 ± 1.4	58.9 ± 16.5	110.9 ± 87.7	93.8 ± 44.8
Juniperus semiglobosa	0.5 ± 0.3	2.6 ± 1.8	—	2.8 ± 2.8
Lonicera karelinii	0.0 ± 0.0	3.1 ± 3.1	38.5 ± 22.1	47.2 ± 21.6
Malus sieversii, M. niedzwetzkyana	1.6 ± 0.5	58.5 ± 17.2	4.3 ± 2.0	7.6 ± 5.6
Picea schrenkiana	0.3 ± 0.2	4.3 ± 2.6	—	—
Populus alba	0.6 ± 0.4	6.1 ± 4.5	3.9 ± 3.0	96.9 ± 70.4
Prunus sogdiana, P. communis	0.3 ± 0.1	19.5 ± 4.8	78.9 ± 28.0	297.6 ± 107.7
Pyrus communis, P. turkomanica, P. korshinskyi	0.1 ± 0.0	1.6 ± 0.7	—	3.1 ± 3.1
Rhamnus cathartica	0.0 ± 0.0	0.1 ± 0.1	—	7.0 ± 4.1
Salix alatavica	0.0 ± 0.0	0.1 ± 0.1	—	—
Shrub species				
Caragana spp.	—	—	7.3 ± 5.1	4.7 ± 3.4
Cotoneaster multiflorus, C. oliganthus	—	—	193.9 ± 116.4	130.0 ± 68.6
Euonymus spp.	—	—	—	80.0 ± 33.7
Exochorda spp.	—	—	4.5 ± 3.4	3.1 ± 3.1
Rosa spp.	—	—	113.8 ± 50.1	312.1 ± 102.2
Rubus spp.	—	—	—	0.3 ± 0.3
Sorbus persica, S. tianschanica	—	—	—	0.1 ± 0.1
Total	12.3 ± 1.3	236 ± 22.1	919.9 ± 407.1	2,464.9 ± 584.2

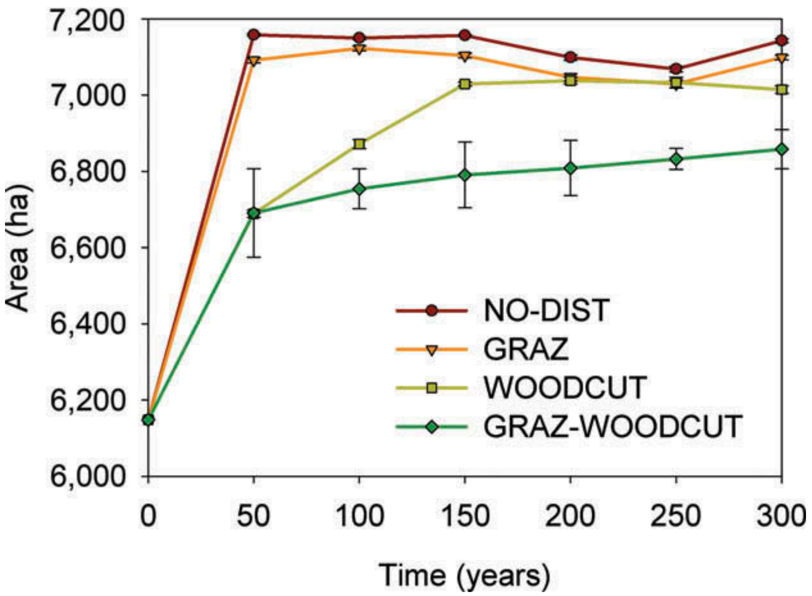

FIGURE 2 Forest extent under different disturbance regimes. Values presented are the area (ha) occupied by up to 27 species, as individuals >10 yr old. Values presented are means of five repeated simulations. Error bars represent the standard deviation. The following symbols are used: circle: no-disturbance scenario; triangle: grazing scenario; square: woodcut scenario; diamond: combined grazing and woodcut scenario. For scenario descriptions see text.

2.6, 2.98, 1.73 and 0%, respectively, under the NO-DIST scenario, which did not differ statistically from the relative covers occupied by the same species under GRAZ, WOODCUT, and GRAZ-WOODCUT scenarios ($p > .05$, Kruskal-Wallis test). The WOODCUT and GRAZ-WOODCUT scenarios, however, influenced the timing of the species' decline, which occurred sooner (after 50–150 yr) compared to the situation under the NO-DIST and GRAZ scenarios (after 150–250 yr). The median percentage cover of the dominant species *Crataegus* spp., *Juglans regia*, and *Malus* spp. under the NO-DIST and GRAZ scenarios was statistically different from the median percentage cover under the WOODCUT and GRAZ-WOODCUT scenarios ($p < .05$, Mann-Whitney test; Figure 4).

Different disturbance regimes also had little influence of the proportion of the landscape occupied by shade-tolerant species such as *Abies semenovii*, the cover of which expanded over the simulation period, reaching over 78% of the landscape after 300 yr in all scenarios (Figure 4). WOODCUT and GRAZ-WOODCUT scenarios favored the expansion of only relatively shade-intolerant species such as *Betula* spp., which the local community does not use for fuelwood. The median percentage cover of *Betula* spp. under WOODCUT and GRAZ-WOODCUT scenarios was statistically higher than the median percentage under the NO-DIST and GRAZ scenarios ($p <$

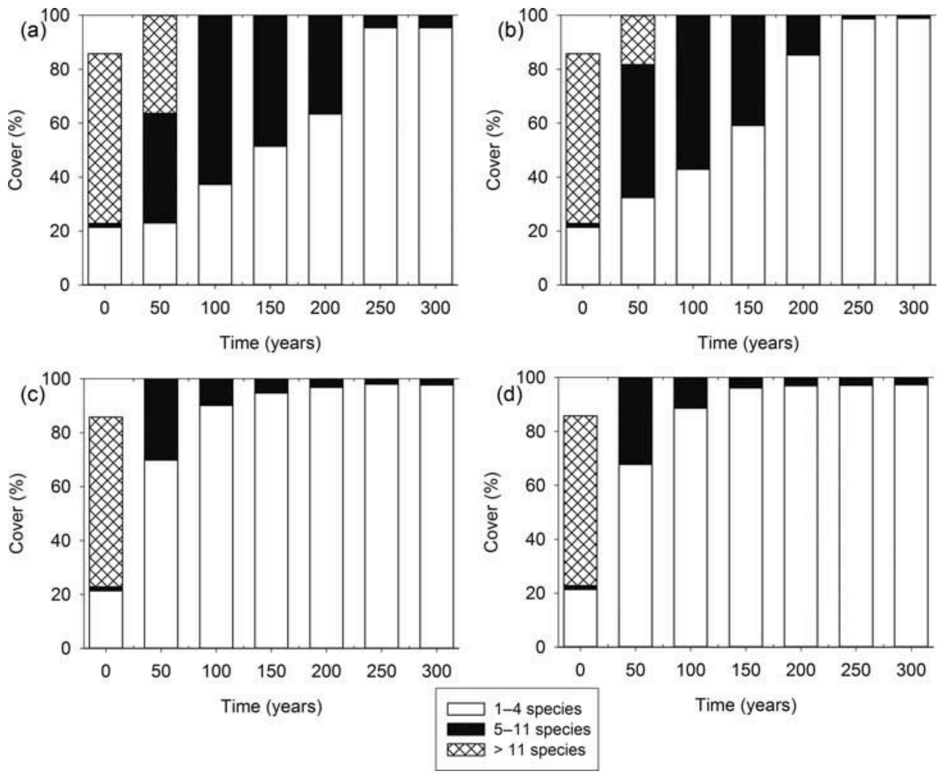

FIGURE 3 Relative cover (%) of simulated landscape occupied by each species richness class under the four scenarios in Sary-Chelek. Species richness classes refer to the 27 species that were modeled: NO-DIST scenario (a), GRAZ scenario (b), WOODCUT scenario (c), and GRAZ-WOODCUT scenario (d). Values presented are means of five repeated simulations. For scenarios, abbreviations, and descriptions see text.

.05, Mann-Whitney test). Variation between the five LANDIS-II runs for each scenario and each of the selected species was low overall. *Crataegus* spp. under GRAZ-WOODCUT was the only species and scenario where this variation accounted for a relatively high percentage of total forest area (23% after 50 and 100 yr).

The socioeconomic study showed that nuts from *Juglans regia* were gathered to the highest extent, with all respondents reporting that they collect walnuts solely for sale. Ninety-six percent of respondents gathered walnuts "occasionally" and 4% gathered walnuts "rarely" ($n = 45$). Fruits from *Malus* spp. and *Prunus* spp. were gathered to the second highest extent (87 and 84% of respondents, respectively) for both subsistence and commercial sale. *Juglans regia* and *Prunus* spp. were the most commonly collected species for fuelwood being harvested "occasionally" by 91 and 88% of respondents, respectively. *Malus* spp. was also reported to be occasionally harvested by 78% of respondents. The harvesting of species for timber was limited, with 71% of respondents stating that they did not harvest any timber from the

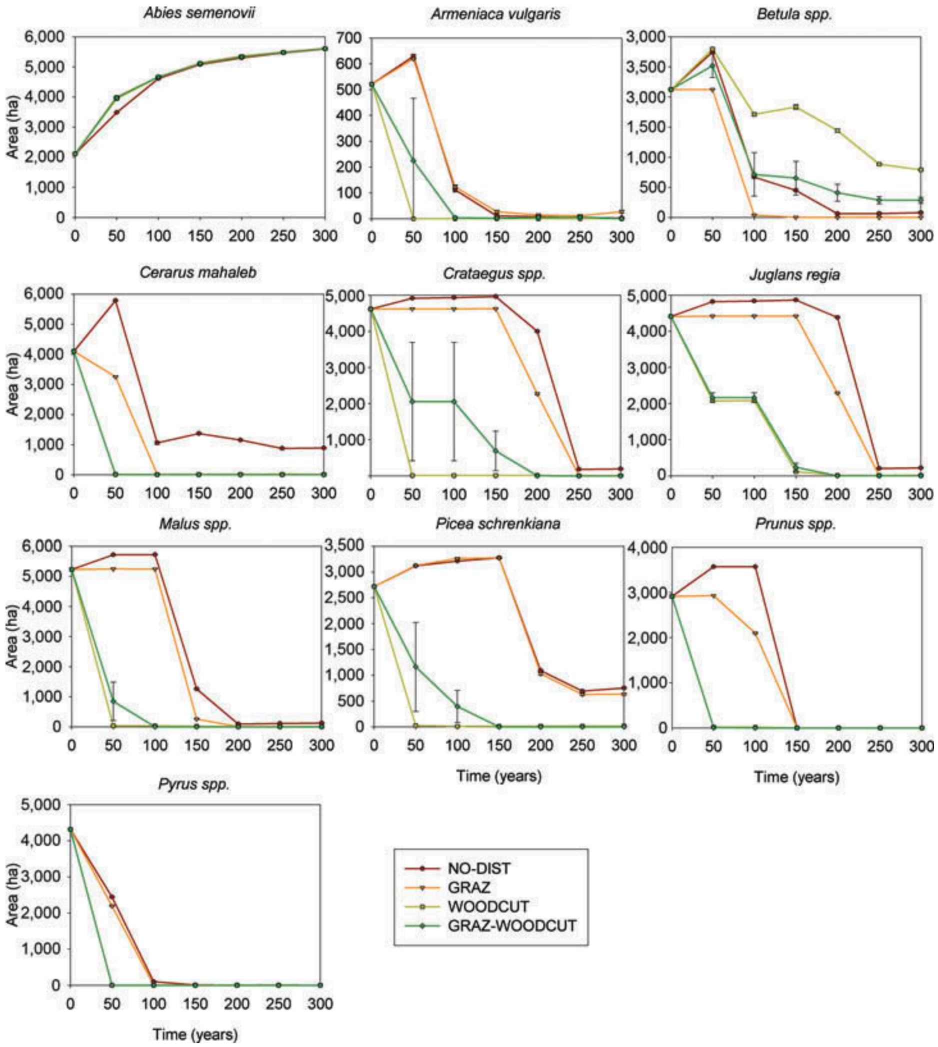

FIGURE 4 Projected extent (ha) of selected species of age >10 yr using the Landis-II model to explore different disturbance regimes. The following symbols are used: circle: no-disturbance scenario; triangle: grazing scenario; square: woodcut scenario; diamond: combined grazing and woodcut scenario. Each 0.90 ha cell could contain up to 27 species. Values presented are means of five repeated simulations. Error bars represent the standard deviation. For scenario descriptions see text. For more details on the selected species see the *Note* below Table 1.

study area. The main species favored for timber use was *Picea schrenkiana*, which was harvested by 29% of respondents. When asked if their household allowed grazing livestock into the core area, every household allowed cattle and almost half (44%) allowed horses into the core area, with a further 37% allowing sheep. The majority (62%) of respondents stated that their livestock traveled less than 5 km away from their farm and 31% indicated

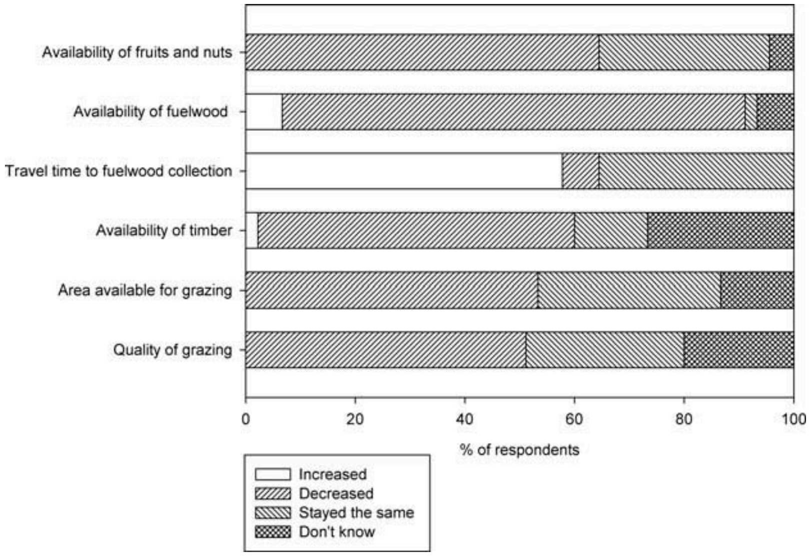

FIGURE 5 Changes in the availability of forest resources reported during the last 10 yr in Sary-Chelek. Bars represent the opinion of 45 local inhabitants that were interviewed in the socioeconomic survey.

that they traveled farther than 5 km. All respondents described pronounced changes in the area during the past 10 yr. The majority of respondents (up to 84%) reported a decreased availability of forest resources (fruits, walnuts, fuelwood, and timber), and a decreased extent and quality of the area used for grazing (Figure 5).

DISCUSSION

Although the walnut-fruit forests of Kyrgyzstan are recognized as a global conservation priority (Myers et al., 2000; Olson et al., 2001), they have been subjected to widespread forest clearance and degradation in recent decades (Sherbinina, 1998). Approaches are therefore urgently required to improve the sustainability of the forest use to ensure that these forests are conserved effectively, and to reduce the risk of extinction of the many threatened species that occur there (Eastwood, Lazkov, & Newton, 2009). Although the conservation importance of the walnut-fruit forests of Kyrgyzstan is well-established (Blaser, Carter, & Gilmour, 1998) and their vegetation history has recently been investigated (Beer et al., 2008), very little information is available concerning their ecology and dynamics. The research presented here appears the first attempt to analyze the dynamics of these forests in response to human disturbance, with the aim of informing their conservation management and sustainable use.

Our investigation indicated that the forest in Sary-Chelek was structurally similar to many other temperate forests, in that it was dominated by relatively few species (Armesto & Figueroa, 1987). The densities of the dominant trees were highly variable, indicating the heterogeneous nature of the forest. This variation is likely attributable to the diverse environmental conditions, associated with the mountainous and steeply sloping terrain. Tree density was relatively low in comparison with some other temperate forests (Gutierrez, Armesto, & Aravena, 2004; Cantarello & Newton, 2008), but the values recorded were consistent with that reported by other studies of walnut-fruit forests of Kyrgyzstan (e.g., Venglovsky, 1998). The low basal area and low regeneration of the dominant species may be attributable to human disturbance, such as increase of fuelwood cutting and uncontrolled grazing that has occurred widely in walnut-fruit forests since the break-up of the Soviet Union (Venglovsky, 1998; Orozumbekov et al., 2009).

Our model simulations suggested that anthropogenic disturbance has relatively little impact on overall forest cover. When subjected to grazing and fuelwood cutting, acting either as a single disturbance or in combination, forest cover always maintained values above 93% during the 300-yr simulations, suggesting that some species resilient to anthropogenic disturbance are able to colonize the landscape. Similarly, species richness was very little impacted by grazing and fuelwood cutting, values following a similar decreasing trend in all scenarios. This can be understood in terms of successional dynamics, with relatively shade-tolerant species such as *Abies semenovii* tending to dominate with time, and to eliminate less shade-tolerant species through competitive exclusion. These results also suggest that the extent of anthropogenic disturbance simulated here, which was based on observations of current human activity, was insufficient to prevent such competitive exclusion taking place. This contrasts with some studies of forests in other regions; for example, Newton et al. (2011) recorded a lower species richness in scenarios without disturbance than those with cutting and grazing in dryland forests of Chile.

The most striking result obtained from the model simulations was the interaction between anthropogenic disturbance and the abundance of different tree species. In the presence of fuelwood cutting with or without grazing, species of high economic importance such as *Juglans regia* and *Malus* spp. and threatened fruit species such as *Armeniaca vulgaris* were largely eliminated from the landscape after 50–150 yr. These results highlight the potentially deleterious impact of fuelwood cutting on the abundance of such fruit- and nut-producing tree species. This supports previous studies that have identified fuelwood cutting to be a serious cause of forest degradation in the region (Sherbinina, 1998; Orozumbekov et al., 2009), and highlights the need to regulate fuelwood cutting, to ensure that this process is sustainable. As noted by Newton (2008), this might be achieved by developing an appropriate monitoring program, so that the impacts of fuelwood collection can regularly be assessed, and the amounts harvested adjusted accordingly. Such monitoring is not currently being conducted in the Sary-Chelek Biosphere Reserve (A. Orozumbekov, personal communication, March 12, 2012).

However, the model simulations also showed that *Juglans regia* still disappeared from the landscape in the absence of disturbance or with grazing only, as a result of competitive interactions, albeit at a much lower rate. This supports recent paleoecological evidence that walnut-fruit forests are of largely anthropogenic origin (Beer et al., 2008), in contrast to the traditional view that these are ancient, relict forests (Sukacev, 1949; Kolov, 1998). Results from pollen analyses by Beer et al. (2008) indicate that the natural forests in this region were originally dominated by *Juniperus, Betula*, and rosaceous trees (presumed to be the fruit-tree genera currently present in the area—namely, *Malus, Pyrus, Prunus*, and *Crataegus*), with little evidence of local presence of walnut prior to 1,000 cal yr BP or even later. Furthermore, the close association of walnut pollen and the pollen of crops and weeds together with the presence of microscopic charcoal particles suggests that the original forests were cleared by fire to support the expansion of agriculture, and potentially to support the establishment of walnut groves (Beer et al., 2008).

Another notable result obtained from the simulations was that grazing did not appear to accelerate the disappearance of the walnut-fruit species. The trend of species decline under grazing was similar to that of no disturbance for the vast majority of the species. This is in contrast to the common belief that animal grazing is a destructive activity for the walnut-fruit forests (Sherbinina, 1998; Orozumbekov et al., 2009). These results are explicable in terms of the relatively low shade-tolerance of the walnut-fruit species, which will require some level of disturbance to the forest canopy in order to establish. Such opportunities for tree regeneration could potentially be maintained by a regular low level of grazing (Yamamoto, 2000). Livestock have been used by both nomadic and sedentary peoples in Central Asia for centuries, and have had a significant impact on the environment. Extensive grazing by cattle is thought to have led to compacted topsoil, damaged root systems, loss of water retention in the soil, and a failure of regeneration in these forests (Sherbinina, 1998). Low levels of grazing, however, could potentially be beneficial by helping to reduce the abundance of competitive shrub or grass species and in increasing forest structural diversity, as has been found in other temperate forests (Homolka & Heroldova, 2003; Reimoser, 2003). Herbivores can also create patches of bare ground through trampling and grazing disturbance, which can create microsites for seedling establishment (McEvoy, McAdam, Mosquera-Losada, & Rigueiro-Rodriguez, 2006). The role of large herbivores in the ecology of forest ecosystems has recently undergone a reappraisal in the light of research by Vera (2000), who identified grazing as a major contributory factor to the maintenance of spatial heterogeneity and diversity in temperate forests. Controlled grazing might therefore make a positive contribution to conservation management of walnut-fruit forests while supporting human livelihoods, a suggestion that does not appear to have been considered previously in the context of these forests.

Major projects—such as USAID Biodiversity Assessment for Kyrgyzstan (Chemonics International Inc., 2001), the Kyrgyz-Swiss Forestry Support Programme implemented by Intercooperation (1995–2009), and the National Forest Assessment of Kyrgyz Republic (Chyngojoev, Surappaeva, & Altrell, 2010)—describe grazing as a significant disturbance restricting regeneration in the walnut-fruit forests of Kyrgyzstan. However, the reports of these projects mainly refer to the effect of overgrazing, which is presented in general terms, and is not supported by quantitative field data. The results of our study suggest that controlled grazing could potentially favor the regeneration of walnut-fruit species while improving local livelihoods. However, this has not been examined to date. Future work should therefore include follow-up research and monitoring to determine both the positive and negative impacts of grazing by livestock on the walnut-fruit forest system.

The current study presents some limitations common to all process-based models of landscape dynamics, including LANDIS-II. Principal among these is the difficulty of obtaining rigorous model validation owing to the lack of long-term data describing the ecological behavior of forests (Newton, 2007a; Shifley, Thompson, Dijak, & Fan, 2008). As very little research has been undertaken into the ecology of walnut-fruit forests (Sherbinina, 1998), the findings presented here should be viewed with caution. Newton (2007a) notes that process-based model outputs should be viewed as hypotheses, which require further testing and the current study is no exception in this regard. In this study, key uncertainties requiring further research include: (a) the dispersal ability of the tree species, to which the LANDIS-II is particularly sensitive; (b) the establishment probability of the tree species across the landscape; (c) the impacts of grazing and fuelwood cutting intensity on different species; and (d) detailed characterization of the initial forest structure. A further key unknown is the extent to which relatively shade tolerant species such as *Abies* and *Picea* can outcompete the currently dominant species *Juglans*, particularly at the lower elevational limits of the conifers. For example, Wang et al. (2004) and Takahashi, Okuhara, Tokumitsu, and Yasue (2011) both indicate that in East Asia mountain areas, the growth of *Abies* and *Picea* spp. at their lower distribution limit is restricted by low precipitation and low soil moisture. Therefore, *Abies* might not be able to become established across the landscape as projected here, particularly in relatively dry years. Further studies on the environmental factors limiting tree species growth along altitudinal gradients are required to address this point, such as those described by Wang Čufar, Eckstein, and Liang (2012). Other areas of uncertainty that might affect the forests dynamics in the future, which are not considered here, are the potential effects of haymaking, and the collection of fruits and nuts. These are common practices that are considered to be threats to the forest, although to a much lesser extent than grazing and fuelwood cutting (Orozumbekov et al., 2009).

Despite these limitations, these results have a number of implications for conservation management. Sary-Chelek is currently managed as a Biosphere Reserve with a core area granted strict protection, and a transition area that includes human settlements where, according to the official sources (UNESCO, 2001), local stakeholders work together to manage the forests resources in a sustainable way. However, the results presented here suggest that appropriate conservation actions should be identified to address the current and predicted forest degradation that is occurring as a result of human activity. As noted above, monitoring and regulation of fuelwood cutting is required to ensure that forest structure and composition is not adversely affected. On the other hand, conservation management approaches might usefully begin to consider the potentially positive impacts that might accrue from controlled grazing. At present, animals are essentially left to roam freely over an extended area, but their impacts are not systematically being monitored. Analysis of the behavior of livestock and their impacts on vegetation would be of value in determining how grazing can best be incorporated into conservation management plans. Conceivably, harvesting of the conifer species for timber might also be consistent with maintaining populations of fruit and nut tree species within the reserve, as well as species richness.

The fact that the fruit and nut tree species appear to be dependent on disturbance for maintenance of populations, coupled with the apparent anthropogenic origin of these forests (Beer et al., 2008), highlights the positive contribution that human activities could potentially make to their future conservation. This could potentially be achieved by adopting a participatory forest management approach, where local people are involved in forestry activities (Arnold, 2001). A key element to the success of participatory forest management is the role of monitoring, which is a valuable tool to help local communities engage in management (Danielsen, Burgess, & Balmford, 2005). The modeling approach presented here could help inform the planning of both management and monitoring actions, to ensure that the main threats are addressed in a spatially explicit manner.

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SUPPLEMENTAL MATERIAL

Supplemental data for this article can be accessed on the publisher's website.

REFERENCES

Abdymomunov, R. A. (2001). *Conclusions of the first national census conducted in 1999: Oblast Jalalabad* (Book III, Series R). Bishkek, Kyrgyzstan: National Statistical Committee of the Kyrgyz Republic.

Armesto, J. J., & Figueroa, J. (1987). Stand structure and dynamics in the temperate rain-rorests of Chiloe Archipelago, Chile. *Journal of Biogeography*, *14*(4), 367–376.

Arnold, J. (2001). *Forests and people: 25 years of community forestry*. Rome, Italy: Food and Agricultural Organization of the United Nations.

Atkinson, M. D. (1992). *Betula pendula* Roth (*B. verrucosa* Ehrh.) and *B. pubescens* Ehrh. *Journal of Ecology*, *80*(4), 837–870.

Bajracharya, S. B., Furley, P. A., & Newton, A. C. (2006). Impacts of community-based conservation on local communities in the Annapurna conservation area, Nepal. *Biodiversity and Conservation*, *15*(8), 2765–2786.

Beer, R., Kaiser, F., Schmidt, K., Ammann, B., Carraro, G., Grisa, E., & Tinner, W. (2008). Vegetation history of the walnut forests in Kyrgyzstan (Central Asia): Natural or anthropogenic origin? *Quaternary Science Reviews*, *27*(5–6), 621–632.

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., . . . Nyström, M. (2003). Reserves, resilience and dynamic landscapes. *Ambio*, *32*(6), 389–396.

Blaser, J., Carter, J., & Gilmour, D. (1998). *Biodiversity and sustainable use of Kyrgyzstan's walnut-fruit forests*. Gland, Switzerland: International Union for Conservation of Nature, Intercooperation.

Bu, R., He, H. S., Hu, Y., Chang, Y., & Larsen, D. R. (2008). Using the LANDIS model to evaluate forest harvesting and planting strategies under possible warming climates in Northeastern China. *Forest Ecology and Management*, *254*(3), 407–419.

Burns, K. C. (2004). Relationships between the demography and distribution of two bird-dispersed plants in an island archipelago. *Journal of Biogeography*, *31*(12), 1935–1943.

CABI. (2005). *Forestry compendium*. Wallingford, United Kingdom: Author.

Cantarello, E., & Newton, A. C. (2008). Identifying cost-effective indicators to assess the conservation status of forested habitats in Natura 2000 sites. *Forest Ecology and Management*, *256*(4), 815–826.

Cantarello, E., Newton, A. C., Hill, R. A., Tejedor-Garavito, N., Williams-Linera, G., Lopez-Barrera, F.,. . . Golicher, D. J. (2011). Simulating the potential for ecological restoration of dryland forests in Mexico under different disturbance regimes. *Ecological Modelling*, *222*(5), 1112–1128.

Carey, C., Dudley, N., & Stolton, S. (2000). *Squandering paradise? The importance and vulnerability of the world's protected areas*. Gland, Switzerland: World Wide Fund for Nature International.

Chape, S., Harrison, J., Spalding, M., & Lysenko, I. (2005). Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *360*(1454), 443–455.

Chemonics International Inc. (2001). *Biodiversity assessment for Kyrgyzstan* [Task Order under the Biodiversity & Sustainable Forestry IQC (BIOFOR), USAID Contract Number: LAG-I-00-99-00014-00]. Washington, DC: U.S. Agency for International Development.

Chyngojoev, A., Surappaeva, B., & Altrell, D. (2010). *Integrated assessment of natural resources of Kyrgyzstan 2008-2010*. Bishkek, Kyrgyzstan: Food and Agriculture Organization of the United Nations.

Cincotta, R. P., Wisnewski, J., & Engelman, R. (2000). Human population in the biodiversity hotspots. *Nature*, *404*(6781), 990–992. doi:10.1038/35010105

Core, E. L. (1974). Brambles. In J. D. Gill & W. M. Healy (Eds.), *Shrubs and vines for northeastern wildlife* (General Technical Report NE-9, pp. 16–19). Broomall, PA: U.S. Department of Agriculture, Forest Service.

Cremer, K. (1999, December). Willow management for Australian rivers. *Natural Resource Management*, 2–22.

Danielsen, F., Burgess, N. D., & Balmford, A. (2005). Monitoring matters: Examining the potential of locally-based approaches. *Biodiversity and Conservation*, *14*(11), 2507–2542.

Dzhangaliev, A. D. (2003). The wild apple tree of Kazakstan. In J. Janick (Ed.), *Wild apple and fruit trees of Central Asia* (Horticultural Reviews, Vol. 29, pp. 63–303). Hoboken, NJ: John Wiley & Sons.

Dzhangaliev, A. D., Salova, T. N., & Turekhanova, P. M. (2003). The wild fruit and nut plants of Kazakstan. In J. Janick (Ed.), *Wild apple and fruit trees of Central Asia* (Horticultural Reviews, Vol. 29, pp. 305–371). Hoboken, NJ: John Wiley & Sons.

Earle, C. J. (1997). *Abies sibirica*. Retrieved from http://www.conifers.org/

Eastwood, A., Lazkov, G., & Newton, A. C. (2009). *The red list of trees of Central Asia*. Gland, Switzerland: International Union for Conservation of Nature.

Ellenberg, H. (1988). *Vegetation ecology of Central Europe* (4th ed.). Cambridge, United Kingdom: Cambridge University Press.

Farrington, J. D. (2005, June). *A report on protected areas, biodiversity, and conservation in the Kyrgyzstan Tien Shan*. Retrieved from http://www.snowleopardnetwork.org/bibliography/Farrington_2005.pdf

Flora of China. (n.d.). Retrieved from http://www.efloras.org

Food and Agriculture Organization of the United Nations. (2010). *Global forest resources assessment 2010: Main report* (FAO Forestry Paper 163). Rome, Italy: Author.

Gaston, K. J., Jackson, S. F., Cantú-Salazar, L., & Cruz-Piñón, G. (2008). The ecological performance of protected areas. *Annual Review of Ecology, Evolution, and Systematics*, *39*(1), 93–113.

Goudzwaard, L. (2006). *Tree database*. Retrieved from http://www.wageningenur.nl/en/Expertise-Services/Chair-groups/Environmental-Sciences/Forest-Ecology-and-Forest-Management-Group/Education/Tree-database.htm

Grashof-Bokdam, C. J., Jansen, J., & Smulders, M. J. M. (1998). Dispersal patterns of *Lonicera periclymenum* determined by genetic analysis. *Molecular Ecology*, *7*(2), 165–174.

Greene, D. F., & Calogeropoulos, C. (2002). Measuring and modelling seed dispersal of terrestrial plants. In J. M. Bullock, R. E. Kenward, & R. S. Hails (Eds.), *Dispersal Ecology: The 42nd Symposium of the British Ecological Society* (pp. 3–23).

Grime, J. P., Hodgson, J. G., & Hunt, R. (1988). *Comparative plant ecology: A functional approach to common British species*. London, United Kingdom: Unwin Hyman.

Grubb, P. J., Kollmann, J., & Lee, W. G. (1999). A garden experiment on susceptibility to rabbit-grazing, sapling growth rates, and age at first reproduction for eleven European woody species. *Plant Biology*, *1*(2), 226–234.

Gucker, C. L. (2007). *Salix discolor*. Retrieved from http://www.fs.fed.us/database/feis/

Gunderson, L. H. (2000). Ecological resilience—In theory and application. *Annual Review of Ecology and Systematics*, *31*(1), 425–439.

Gustafson, E. J., Shifley, S. R., Mladenoff, D. J., Nimerfro, K. K., & He, H. S. (2000). Spatial simulation of forest succession and timber harvesting using LANDIS. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, *30*(1), 32–43.

Gutierrez, A. G., Armesto, J. J., & Aravena, J. C. (2004). Disturbance and regeneration dynamics of an old-growth north Patagonian rain forest in Chiloe Island, Chile. *Journal of Ecology*, *92*(4), 598–608.

Harris, S. A., Robinson, J. P., & Juniper, B. E. (2002). Genetic clues to the origin of the apple. *Trends in Genetics*, *18*(8), 426–430.

He, H. S., & Mladenoff, D. J. (1999). The effects of seed dispersal on the simulation of long-term forest landscape change. *Ecosystems*, *2*(4), 308–319.

Hemery, G. E., & Popov, S. I. (1998). The walnut (*Juglans regia* L.) forests of Kyrgyzstan and their importance as a genetic resource. *Commonwealth Forestry Review*, *77*(4), 272–276.

Herold, B. (2005). *The Avian communities and their habitat associations of the walnut fruit forests of Southern Kyrgyzstan* (Diploma thesis). Ernst Moritz Arndt University of Greifswald, Greifswald, Germany.

Homolka, M., & Heroldova, M. (2003). Impact of large herbivores on mountain forest stands in the Beskydy Mountains. *Forest Ecology and Management*, *181*(1–2), 119–129.

Intercooperation. (1995–2009). *Kyrgyz-Swiss Forestry Support Programme (KIRFOR)*. Retrieved from http://www.swiss-cooperation.admin.ch/centralasia/en/Home/Activities_in_Kyrgyzstan/COMPLETED_PROJECTS/Forestry

Jahn, G. (1991). Temperate deciduous forests of Europe. In E. Röhrig & B. Ulrich (Eds.), *Temperate deciduous forests* (Ecosystems of the World 7, pp. 377–502). Amsterdam, The Netherlands: Elsevier.

Juniper, B. E., & Mabberley, D. J. (2006). *The story of the apple*. Portland, OR: Timber Press.

Kassa, H., Campbell, B., Sandewall, M., Kebede, M., Tesfaye, Y., Dessie, G., . . . Sandewall, K. (2009). Building future scenarios and uncovering persisting challenges of participatory forest management in Chilimo Forest, Central Ethiopia. *Journal of Environmental Management*, *90*(2), 1004–1013.

Kolov, O. (1998). Ecological characteristics of the walnut-fruit forests of southern Kyrgyzstan. In J. Blaser, J. Carter, & D. Gilmour (Eds.), *Biodiversity and sustainable use of Kyrgyzstan's walnut-fruit forests* (pp. 59–61). Gland, Switzerland: International Union for Conservation of Nature, Intercooperation.

Kyrgyz Forest Service. (2002). *Forestry plan of Sary-Chelek Biosphere Reserve (Сарычелекского and Аркытского). Scale 1:25000. Forest inventory 2002*. Bishkek, Kyrgyz: Author.

Kyrgyz Forest Service. (2003). *Sary-Chelek forest management plan 2003–2013*. Bishkek, Kyrgyz: Author. (In Russian)

Lockwood, M., Worboys, G. L., & Kothari, A. (2006). *Managing protected areas: A global guide*. London, United Kingdom: Earthscan.

Martínez, I., & González-Taboada, F. (2009). Seed dispersal patterns in a temperate forest during a mast event: Performance of alternative dispersal kernels. *Oecologia*, *159*(2), 389–400.

Mattock, J. (1994). *The complete book of roses*. London, United Kingdom: Ward Lock.

McEvoy, P. M., McAdam, J. H., Mosquera-Losada, M., & Rigueiro-Rodriguez, A. (2006). Tree regeneration and sapling damage of pedunculate oak *Quercus robur* in a grazed forest in Galicia, NW Spain: A comparison of continuous and rotational grazing systems. *Agroforestry Systems*, *66*(2), 85–92.

McVean, D. N. (1955). Ecology of *Alnus Glutinosa* (L.) Gaertn.: II. Seed distribution and germination. *Journal of Ecology*, *43*(1), 61–71.

Mladenoff, D. J. (2004). LANDIS and forest landscape models. *Ecological Modelling*, *180*(1), 7–19.

Morris, R. J., & Money, S. H. (1996). *Plants For A Future*. Retrieved from http://www.pfaf.org

Müller-Schneider, P. (1983). *Verbreitungsbiologie (Diasporologie) der Blütenpflanzen. 3. Auflage*. Zürich, Schweiz: Geobot. Inst. ETH Stiftung Rübel.

Müller-Schneider, P. (1986). *Verbreitungsbiologie der Blütenpflanzen Graubündens*. Zürich, Schweiz: Geobot. Inst. ETH Stiftung Rübel.

Munger, G. T. (2002a). *Lonicera japonica*. Retrieved from http://www.fs.fed.us/database/feis/

Munger, G. T. (2002b). *Rosa multiflora*. Retrieved from http://www.fs.fed.us/database/feis/

Musuraliev, T. M. (1998). Forest management and policy for the walnut-fruit forests of the Kyrgyz Republic. In J. Blaser, J. Carter, & D. Gilmour (Eds.), *Biodiversity and sustainable use of Kyrgyzstan's walnut-fruit forests* (pp. 3–17). Gland, Switzerland: International Union for Conservation of Nature, Intercooperation.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*(6772), 853–858.

Newton, A. C. (2007a). *Forest ecology and conservation: A handbook of techniques*. Oxford, United Kingdom: Oxford University Press.

Newton, A. C. (2007b). *Biodiversity loss and conservation in fragmented forest landscapes: The forests of montane Mexico and temperate South America*. Wallingford, United Kingdom: CABI.

Newton, A. C. (2008). Conservation of tree species through sustainable use: How can it be achieved in practice? *Oryx*, *42*(2), 195–205.

Newton, A. C., & Echeverría, C. (2014). Analysis of anthropogenic impacts on forest biodiversity as a contribution to empirical theory. In D. Coomes, D. Burslem, & W. D. Simonson (Eds.), *Forests and global change* (pp. 417–446). Cambridge, United Kingdom: Cambridge University Press.

Newton, A. C., Cayuela, L., Echeverría, C., Armesto, J. J., Del Castillo, R. F., Golicher, D., . . . Williams-Linera, G. (2009). Toward integrated analysis of human impacts on forest biodiversity: Lessons from Latin America. *Ecology and Society*, *14*(2), article 2.

Newton, A. C., Echeverría, C., Cantarello, E., & Bolados, G. (2011). Projecting impacts of human disturbances to inform conservation planning and management in a dryland forest landscape. *Biological Conservation*, *144*(7), 1949–1960.

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., . . . Kassem, K. R. (2001). Terrestrial ecoregions of the world: A new map of life on earth. *BioScience*, *51*, 933–938.

Orozumbekov, A. (2011). The natural heritage and conservation of Tien Shan forests. In A. Laletin, J. A. Parrotta, & I. Domashov (Eds.), *Traditional forest related knowledge, biodiversity conservation and sustainable forest management in Eastern Europe, Northern and Central Asia* (IUFRO World Series Vol. 26, pp. 52–55). Vienna, Austria: IUFRO.

Orozumbekov, A., Musuraliev, T., Toktoraliev, B., Kysanov, A., Shamshiev, B., & Sultangaziev, O. (2009). Forest rehabilitation in Kyrgyzstan. In D. K. Lee & M. Kleine (Eds.), *Keep Asia green—Volume IV: West and Central Asia* (IUFRO World Series Volume 20-IV, pp. 131–182). Vienna, Austria: International Union of Forest Research Organizations.

Orwa, C., Mutua, A., Kindt, R., Jamnadass, R., & Simons, A. (2009). *Agroforestree database: A tree reference and selection guide version 4.0*. Retrieved from http://www.worldagroforestry.org/resources/databases/agroforestree

Paganová, V. (2001). The evaluation of height growth of wild pear (*Pyrus pyraster* [L.] Burgsd.) progenies from different regions of Slovak Republic. *Journal of Forest Science*, *47*(10), 464–472.

Pennanen, J., & Kuuluvainen, T. (2002). A spatial simulation approach to natural forest landscape dynamics in boreal Fennoscandia. *Forest Ecology and Management*, *164*(1–3), 157–175.

Peter, D. (2008). *Tree succession planning: Modelling tree longevity in Tuttangga/Park 17, the Adelaide park lands* (Doctoral thesis). University of Adelaide, School of Architecture, Landscape Architecture and Urban Design, Adelaide, Australia.

Phillips, A. (2003). Turning ideas on their head—The new paradigm for protected areas. *The George Wright Forum*, *20*(2), 8–32.

Raspe, O., Findlay, C., & Jacquemart, A. L. (2000). *Sorbus aucuparia* L. *Journal of Ecology*, *8*, 910–930.

Reimoser, F. (2003). Steering the impacts of ungulates on temperate forests. *Journal for Nature Conservation*, *10*(4), 243–252.

Scheller, R. M., Domingo, J. B., Sturtevant, B. R., Williams, J. S., Rudy, A., Gustafson, E. J., & Mladenoff, D. J. (2007). Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. *Ecological Modelling*, *201*(3–4), 409–419.

Schmidt, K. (2006). NTFPs and poverty alleviation in Kyrgyzstan: Potential and critical issues. In J.-L. Pfund & P. Robinson (Eds.), *Non-timber forest products between poverty alleviation and market forces* (pp. 28–29). Bern, Switzerland: Intercooperation.

Schmidt, M. (2005). Utilisation and management changes in South Kyrgyzstan's mountain forests. *Journal of Mountain Science*, *2*(2), 91–104.

Schumacher, S., Bugmann, H., & Mladenoff, D. J. (2004). Improving the formulation of tree growth and succession in a spatially explicit landscape model. *Ecological Modelling*, *180*(1), 175–194.

Secretariat of the Convention on Biological Diversity. (2010). *Global biodiversity outlook 3*. Montréal, QC, Canada: Author.

Sherbinina, E. N. (1998). The problem of conserving the biological diversity of walnut-fruit forests in Kyrgyzstan. In J. Blaser, J. Carter, & D. Gilmour (Eds.), *Biodiversity and sustainable use of Kyrgyzstan's walnut-fruit forests* (pp. 55–58). Gland, Switzerland: International Union for Conservation of Nature, Intercooperation.

Shifley, S. R., Thompson, F. R., III, Dijak, W. D., & Fan, Z. (2008). Forecasting landscape-scale, cumulative effects of forest management on vegetation and wildlife habitat: A case study of issues, limitations, and opportunities. *Forest Ecology and Management*, *254*(3), 474–483.

Silander, J. A., Jr., & Klepeis, D. M. (1999). The invasion ecology of Japanese barberry (*Berberis thunbergii*) in the New England landscape. *Biological Invasions*, *1*(2–3), 189–201.

Spencer, W. D., Rustigian, H. L., Scheller, R. M., Syphard, A., Strittholt, J., & Ward, B. (2008, June). *Baseline evaluation of fisher habitat and population status & effects of fires and fuels management on fishers in the Southern Sierra Nevada*. Unpublished report prepared for USDA Forest Service, Pacific Southwest Region.

Sukacev, V. N. (Ed.). (1949). *Plodovye lesa Yuzhnoy Kirgizii i ikh ispolzovanie* [The fruit-forests of Southern Kyrgyzstan and their utilization] (Vol. 1). Moscow, Russia: AN SSSR. (In Russian)

Takahashi, K., Okuhara, I., Tokumitsu, Y., & Yasue, K. (2011). Responses to climate by tree-ring widths and maximum latewood densities of two *Abies* species at upper and lower altitudinal distribution limits in central Japan. *Trees-Structure and Function*, *25*(4), 745–753.

Thomas, P. (2000). *Trees: Their natural history*. Port Chester, NY: Cambridge University Press.

Tirmenstein, D. (1990). *Rubus idaeus*. Retrieved from http://www.fs.fed.us/database/feis/

United Nations Educational, Scientific and Cultural Organization. (2000). *Sary-Chelek Biosphere Reserve information*. Retrieved from http://www.unesco.org/mabdb/br/brdir/directory/biores.asp?mode=gen&code=KIZ+01

United Nations Educational, Scientific and Cultural Organization. (2001). An introduction to biosphere reserves. *Russian Conservation News, 27*, 2–20. Retrieved from http://www.wild-russia.org/pubs/pdfs/27.pdf

Venglovsky, B. I. (1998). Potentials and constraints for the development of the walnut-fruit forests of Kyrgyzstan. In J. Blaser, J. Carter, & D. Gilmour (Eds.), *Biodiversity and sustainable use of Kyrgyzstan's walnut-fruit forests* (pp. 73–76). Gland, Switzerland: International Union for Conservation of Nature, Intercooperation.

Vera, F. W. M. (2000). *Grazing ecology and forest history*. Wallingford, United Kingdom: CABI.

Vittoz, P., & Engler, R. (2007). Seed dispersal distances: A typology based on dispersal modes and plant traits. *Botanica Helvetica*, *117*(2), 109–124.

Wang, T., Liang, Y., Ren, H., Yu, D., Ni, J., & Ma, K. (2004). Age structure of *Picea schrenkiana* forest along an altitudinal gradient in the central Tianshan Mountains, northwestern China. *Forest Ecology and Management*, *196*(2–3), 267–274.

Wang, Y., Čufar, K., Eckstein, D., & Liang, E. (2012). Variation of maximum tree height and annual shoot growth of Smith fir at various elevations in the Sygera Mountains, southeastern Tibetan Plateau. *PLoS ONE*, *7*(3), e31725.

Whitney, G. G. (1986). A demographic analysis of *Rubus idaeus* and *Rubus pubescens*. *Canadian Journal of Botany*, *64*, 2916–2921.

Willems, J. H., & Bik, L. P. M. (1998). Restoration of high species density in calcareous grassland: The role of seed rain and soil seed bank. *Applied Vegetation Science*, *1*(1), 91–100.

Wyckoff, G. W., & Zasada, J. C. (2008). *Populus* L. In F. T. Bonner & R. P. Karrfalt (Eds.), *The woody plants seed manual* (Agriculture Handbook 727, pp. 856–871). Washington, DC: U.S. Department of Agriculture, Forest Service.

Xu, C. G., Gertner, G. Z., & Scheller, R. M. (2007). Potential effects of interaction between CO_2 and temperature on forest landscape response to global warming. *Global Change Biology*, *13*(7), 1469–1483.

Yamamoto, S.-I. (2000). Forest gap dynamics and tree regeneration. *Journal of Forest Research*, *5*(4), 223–229.