RESEARCH PAPER

POST-FIRE SUCCESSION OF PSEUDO-TAIGA LARCH FOREST IN THE TARVAGATAI MOUNTAIN RANGE, MONGOLIA

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ABSTRACT. Understanding post-fire recovery and succession is crucial for determining the forest's further reestablishment rate and development tendency, facilitating the restoration and protection of degraded forests, and planning post-fire forest management. The main aim of this study was to evaluate forest regeneration and reveal the tendency of plant succession after large-scale fire in the Tarvagatai Mountain range, Central Khangai, Mongolia. The monitoring study on post-fire plant succession and regeneration in the forbs-*Rhytidium* mosses pseudotaiga larch forests was conducted on permanent sample plots from 2007 to 2021 in the forest sites, which were damaged by severe fires in 1996 and 2002. Our results indicated that burned forest was regenerated sufficiently through the several serial stages of post-fire successions as fireweed (*Chamaenerion angustifolium*) community (up to 5 years after fire), fireweed-bonfire moss (*Funaria hygrometrica*) community (from 6 to 10 years), forbs community (11-16 years), grass-forbs young larch forest (17-25 years). Species numbers gradually increased with time in the forest affected by fires, whereas they rose drastically in the forest damaged by fire and livestock browsing due to the increase of ruderal species. In spite of the long recovery period, the post-fire similarity indexes of species composition and coenotic percentage compared with the control forest were relatively low, indicating a slow pre-fire vegetation recovery.

KEYWORDS: forest fire, *Larix sibirica*, forest regeneration, succession, pseudo-taiga forest

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INTRODUCTION

The forest ecosystems of the world, which cover nearly a third of the total land area, provide important forest goods and a wide range of services such as carbon sink and storage, soil and water protection, biodiversity maintenance and conservation (Jenkins and Brian 2018; FAO 2020; FAO and UNEP 2020). Boreal forests cover 27% of the global forest area (FAO 2020). The boreal forests of Mongolia cover 12.6 million hectares or 8.07% of the country's territory (Forest Research and Development Center 2021).

Fire is one of the natural and human-induced forces that have influenced the development of boreal and hemiboreal forests over time. However, it is also a significant source of emitted carbon, contributing to global warming and leading to biodiversity changes (Nasi et al. 2002), altering soil and light conditions, affecting seedbanks, and removing seed trees (Kristi Parro et al. 2015), decreasing the biomass of forest trees (Altrell 2019), transforming the environment of forest existence, and changing the composition and structure of the plant cover (Matveev et al. 2012).

Annual mean air temperature in Mongolia has increased by 2.24°C, whereas annual precipitation has decreased by 7% from 1940 to 2015 (MET 2018). Climate change is expected to increase fire severity and frequency (Hille and Ouden 2004; Jacquelyn et al. 2017; Boucher et al. 2020), as well as increase the burned area in the boreal forest (Ponomarev et al. 2016). Across the country from 1995 to 2020, 179 fires occured annually, covering an average of 357 thousand hectares of forest. Fire in Mongolia damaged 2367 thousand hectares, 2710 thousand hectares, and 583 thousand hectares of forest in 1996, 1997, and 2002, respectively (National Emergency Management Agency 2020). As a result of fires in 1996 and 1997, almost 500 thousand hectares of forests were completely burned and lost their ecological function (Tsogtbaatar 2004).

Monitoring studies on post fire succession are needed, specifically in the Tarvagatai mountain range after the large-scale fires that occurred in 1996 and 2002. Large-scale fires are classified as more than 200 hectares of burnt areas (Valendik et al. 1979). In the Tarvagatai Mountain Range, 114.3 thousand hectares or 52% of forest areas were damaged by fire in 1996, 2002 and 2003 (Jagdag and Gerelbaatar 2016).

Various studies on post-fire succession in the boreal forest exist (Bergeron and Dubue 1988; Bergeron and Dansereau 1993; Johnstone et al. 2004; Greene et al. 2004; Johnstone et al. 2010; Ivanova et al. 2014; Gamova 2014; Tautenhahn et al. 2016). However, only few studies on post-fire regeneration and succession in boreal larch forests are so far available (Uemura et al. 1990; Abaimov et al. 2002; Tsvetkov 2006; Takahashi 2006; Danilin 2009; Lytkina and Mironova 2009; Zyryanova et al. 2010; Matveev et al. 2012; Prokushkin and Zyryanova 2013; Cai et al. 2013, Cai et al. 2018). In particular, few studies on larch forest regeneration after fires have been conducted in Mongolia (Tsogt 1993; Zoyo 2000; Park 2005; Dugarjav 2006; Dorjsuren et al. 2007; Dorjsuren 2009; Park et al. 2009; Undraa et al. 2015; Chu et al. 2017; Undraa et al. 2020).

The monitoring of post-fire forest natural regeneration and succession is crucial to both ecological research on the forest further reestablishment rate and the development tendency and development of scientifically justified policies of forest management and protection (Abaimov and Sofronov 1996; Hille and Ouden 2004; Chen et al. 2014; Chen et al. 2018).

The overall aim of this study was to study natural regeneration and succession after large-scale fires in the Tarvagatai Mountain Range. The objectives of this study were: 1) to investigate the natural regeneration of burnt larch forest; 2) to reveal serial stages of post-fire successions; and 3) to determine the species composition and diversity after fire.

MATERIALS AND METHODS

Study area

The study on post-fire regeneration and succession of pseudo-taiga larch forests (*Larix sibirica Ledeb.*) was carried out in the Tarvagatai Mountain Range of Central Khangai, Tosontsengel Soum, Zavkhan Province, located 890 km northwest of Ulaanbaatar, Mongolia (Fig.1). The Tarvagatai Mountain Range within the Khangai Mountain main range of Mongolia is located in the Central Khangai forest vegetation province, which occupies 1.74 million hectares, or 13.6% of the forest area of Mongolia (Tungalag 2020). The forests of Central Khangai are represented by two altitude-zonal complexes of forest types: pseudo-taiga (83%) and sub-alpine (17%). The most widespread forest type is forbs-Rhytidium mosses pseudo-taiga larch forests (35%) (Dorjsuren 2009). Pseudo-taiga larch forests were first described by Dugarjav et al. (1975) under the name "dry mossy larch forests" in the Tarvagatai Mountain Range. As physiognomy, dry mossy larch forests are similar to green mossy larch taiga; medium-high-density uneven-aged stands, and a well-developed moss layer; however, they are sharply distinguished by many attributes: it has an extreme continental cold and arid climate; the forest soil is mountain forest coarse-humus permafrost soil; the herbaceous layer consists of tundra-alpine, meadow-forest, and forest steppe species; and the moss layer is dominated by the dry mosses *Rhytidium rugosum* and *Abietinella abietina*. Their common feature is that they are very susceptible to anthropogenic influences (Krasnoshekov 1983; Krasnoshekov 1996). Therefore, dry mossy larch forests are named pseudo-taiga (Korotkov 1976).

According to data from 1996-2021, the annual average air temperature is -4.9°C, the annual average precipitation is 223.7 mm (Institute of Meteorology and Hydrology 2021), and the plant growth-intensive period is 141 days (Dorjsuren 2009).

Field data collection

Fieldwork was conducted at the middle of growing seasons (late July and early August), 2007, 2010, 2012, 2015, 2016, 2019, and 2021. Permanent sample plots (PSP) were established to conduct periodic long-term monitoring survey on post-fire regeneration and vegetation. We selected two study areas, Baitsiin Ar and Bayan-Uul in the Tarvagatai Mountain range. The two study areas were nearly 35 km apart from each other. At each study area,

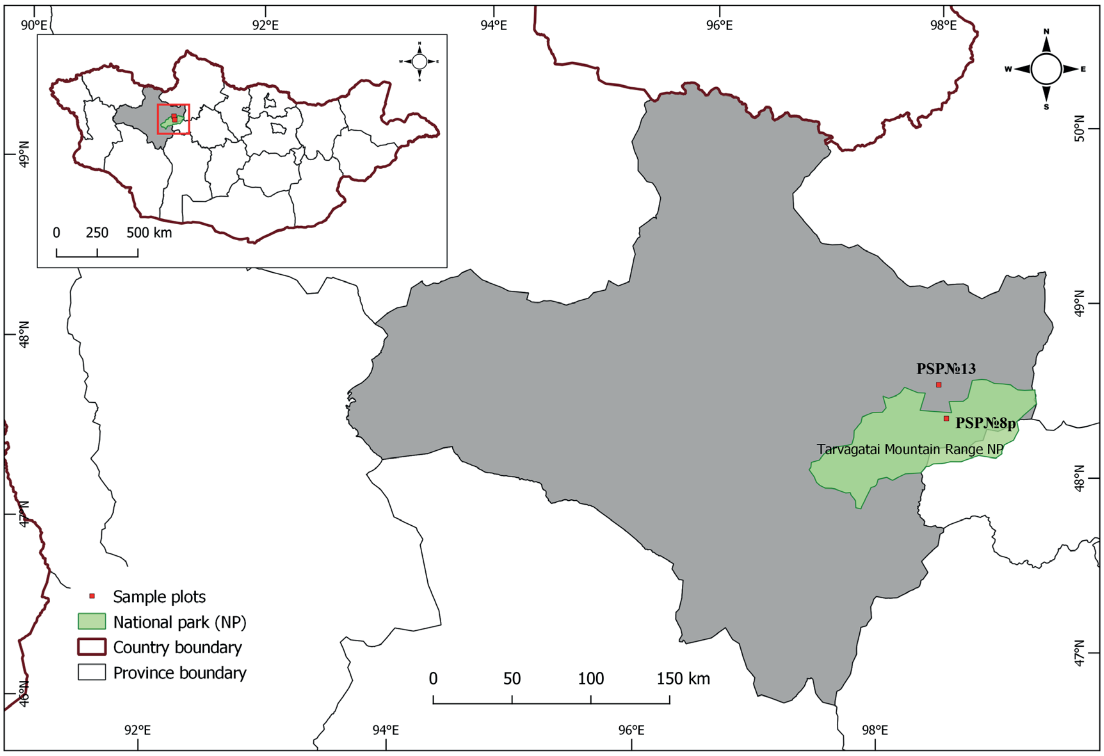


Fig. 1. Location of the study sites

we defined two sites, control forest and burned forest. Site size was provided in Table 1. Two permanent sample plots (PSP №8p and №13) were established in 2007 at 48°16´N, 98°21´E (burned in 2002) and 48°27´N, 98°15´E (burned in 1996) (Table 1). The measurements of number of seedlings and saplings, plant cover, diversity indexes were labelled with year after the fire (AF), which showed the time since the fire. For example, in plot №8p, which burned in 2002, data from 2007 were labelled as AF5 (5 years since the last fire) etc.

The size of permanent sample plot №8p was 20×20 m, which was divided into 16 subplots of 5×5 m. Inventory of seedlings and saplings was conducted each subplots. The size of permanent sample plot №13 was 40×50 m, that was divided into 20 subplots of 10×10 m. The inventory of seedlings and saplings was conducted in 2×2 m quadrates in each subplots. Seedlings and saplings were divided into five height classes: 0-10 cm, 11-50 cm, 51-150 cm, 151-300 cm, and ≥ 300 cm. Plants under 1 year old were considered seedlings, whereas plants ≥ 2 years old were considered saplings. The upper limit of a sapling is a woody plant with up to 8 cm diameter at breast height. We measured the diameter at breast height (DBH; breast height=1.3 m) and height of all standing trees in each subplot of PSP №13.

Shrub, herbaceous, and lichen-moss projective covers were evaluated visually in each quadrate. We conducted a vegetation survey in all quadrates, wherein species composition and cover were recorded in the shrub, herbaceous, and moss-lichen layers. To identify vascular plants and mosses, we used the Key to the Vascular Plants of Mongolia (Grubov 1982) and the Key to the Mosses of Mongolia (Tsegmid 2001).

Data analysis

The post-fire changes in the plant community can be identified by two parameters: 1) a change in the plant species composition, and 2) a change in the plant species projective cover. Based on these two parameters, we calculated the post-fire similarity index in species composition and in coenotic percentage (plant community species projective cover, %) for sample plots.

The species composition similarity index (S_s) was calculated for all pairs of sites, including control and burned stands, using the Sørenson (1948) Eq. 1:

$$S_s = \frac{2c}{a+b} \times 100\% \qquad (1)$$

where a is the number of species in sample A, b is the number of species in sample B, and c is the number of common species in both samples.

The coenotic percentage similarity index (CPS) was calculated using Renkonen's (1938) Eq. 2, which was interpreted as follows:

$$CPS = \sum_{i=1} min\left(p_{1i}, p_{2i}\right) \qquad (2)$$

where p_{1i} the proportion of projective cover (pc_{1i}) multiplied by frequency ($k1i$) of the i-th species in sample A $\left(p_{1i} = \frac{pc_{1i} * k_{1i}}{\Sigma\left(pc_{1i} * k_{1i}\right)}\right)$; p_{2i} the proportion of projective cover (pc_{2i}) multiplied by frequency (k_{2i}) of the i-th species in sample B. $k_{1i} = \frac{n_{1i}}{N_1}$, n_{1i} number of subsamples with i-th species, N_1 total number of subsamples in given sample A

Yearly differences in mean projective covers in shrub and herbaceous, lichen, and moss layers at the study sites were analyzed by the Tukey-Kramer test using the statistic software JMP 4.0.

The Steel Dwass test was performed to detect yearly difference in species richness and species diversity.

In total, 16 quadrates in plot №8p and 20 quadrates in plot №13 were used for species analysis over time post-fire. Shannon-Weaver (Shannon and Weaver 1949), Simpson diversity (Simpson 1949), and Pielou's evenness (Pielou 1975) indices were calculated to identify changes in species diversity after forest fire.

Shannon –Weaver's index (H')

$$H' = -\sum_{i=1}^{S} p_i * In\left(p_1\right) \qquad (3)$$

where s is the number of species, p_i is the proportion of individuals or the abundance of the species expressed as a proportion of total cover

$$D = 1 - \sum_{i=1}^{S} \left(p_1\right)^2 \qquad (4)$$

Simpson's diversity index (D)

$$J = H'/In(S) \qquad (5)$$

Pielou's evenness (J)

where H' is Shannon-Weaver's diversity index, S is the total number of species within a plot and $In(S)$ is denotes the maximum value of H'.

Detrended correspondence analysis (DCA) was performed to identify differences in species composition among sub-quadrats in the sample plots. All statistical analyses were performed using R version 4.1.1[1].

Table 1. Description of the permanent sample plots (PSP) and degradation factors

№	Location	Size of plot, m^2	Elevation a.s.l, m	Coordinate, slope, exposure	Degradation factor
1	Baitsiin Ar (PSP № 8p)	20 × 20 m	2072	N 48^{0}16'366 E 98^{0}21'219 15-180, N	Fire (2002)
2	Bayan-Uul (PSP № 13)	40× 50 m	1963	N 48^{0}27'691 E 98^{0}15'776 5-70, N	Fire (1996)
3	Mukhar valley	40 × 40 m	1876	N 48^{0}37'588 E 98^{0}15'411 5-70, NW	Control forest

[1] R Development Core Team. R: a language and environment for statistical computing, reference index version 4.1.1. Vienna:R Foundation for Statistical Computing, Available at: http://www.R-project.org/ 2021

RESULTS

Tree and regeneration dynamics

No living standing trees remained in PSP №8p. All standing dead trees had already been logged by 5 years post-fire. In plot №13, we compared forest structures of AF20 and AF23 time categories (Table 2). Living tree density per hectare didn't change, and dead tree density decreased 35.0 to 25.0 trees per hectare.

Fig. 2a and Fig. 2b illustrate the numbers of larch saplings in each height class in the permanent sample plots burned in 2002 and 1996. Sapling ages ranged from 2 to 6 years for 1-10 cm saplings, from 8 to 11 years for 51-150 cm saplings, and from 14 to 15 years for 151-300 cm saplings, and from 16 to 22 years for > 300 cm saplings. Larch seedlings have appeared intensively on fresh burnt area. Eight years after the fire, larch seedlings with a height class of 11-50 cm and 51-150 cm were predominated. Ten years after the fire, seedlings with a height class of 51-150 cm have prevailed and saplings with a height class of 151-300 cm have begun to establish. Moreover, on the 14-year-old burnt area, saplings with a height class of 151-300 cm were dominated and saplings with a height class more than 300 cm were formed; and saplings with a height class of 151-300 cm and more than 300 cm have predominated on the 17-year-old burnt area (Fig. 2a). On the PSP №8p, 5 years after the fire, 41.9 thousand two-four-year-old larch seedlings were counted per 1 hectare. Salvage cuttings were carried out on this sample plot in 2007, as a result of which some seedling and saplings were damaged and destroyed. Therefore, in 2010, 8 years after the fire, the number of seedling and saplings was decreased to 26.9 thousand per hectare. In 2012, the number of seedling and saplings was increased to 39.3 thousand per 1 hectare due to sprouting of a seedling from a seed in 2010 and 2011. The number of seedling and saplings from 2015 to 2021, 13-19 years after the fire, was decreased to 25.7 thousand per hectare due to natural thinning of seedlings and saplings (Fig. 2a).

Table 2. Change in forest structure in plot №13 from AF20 to AF23

Fire history	AF20	AF23
Living tree density (individuals ha^{-1})	90.0	90.0
Dead tree density (individuals ha^{-1})	35.0	25.0
Average diameter of living trees (cm)	22.3	23.6
Maximum DBH size (cm)	45.7	47.3
Basal area (m^2 ha^{-1})	3.52	4.16

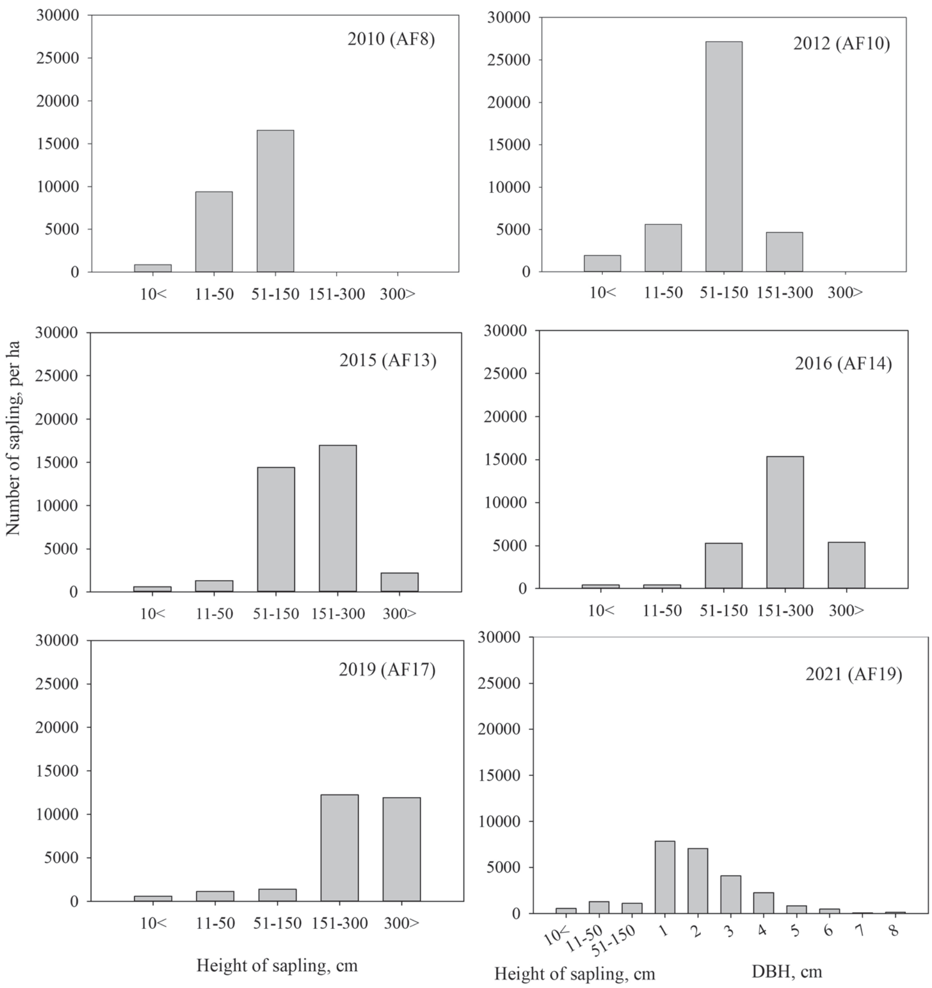


Fig. 2a. Change in the number of larch saplings in PSP №8p height class. DBH - woody plant with up to 8 cm diameter at breast height

PSP №13 is located 4 km south of the Rashaant brigade, and it was observed that livestock of herders' households who reside along the Khojuul river, such as cattle, horses, sheep, and goats graze daily. According to a local herder, 6000 heads of livestock from 30 households graze along the Khojuul river near the Rashaant brigade. The top part of all seedlings and saplings counted were browsed by livestock such as goats and sheep, and their height increment was decreased. Eleven and fourteen years after the fire, larch seedlings with a height class of <10 cm and 11-50 cm, 11-50 cm and 51-150 cm were predominated, respectively. On the 16, 19 and 20-year-old burnt areas, seedlings with a height class of <10 cm were dominated due to sprouting of a seedling from a seed in previous years (Fig. 2b). Eleven years after the fire, 26.9 thousand two-eleven years old larch seedlings were counted per 1 hectare on the PSP №13. In 2012, 16 years after the fire, the number of seedling and saplings was increased to 96.0 thousand per hectare due to sprouting of a seedling from a seed in 2011. The number of seedling and saplings from 2015 to 2021 or in 19-23 years since the fire, was decreased to 46.2 thousand per hectare. Most of the saplings were grazed and damaged by livestock (Fig. 2b).

Understory plant community succession

The shrub layer is not well developed on the burned area. Individual shrubs such as *Lonicera altaica* and *Ribes nigrum*, *Salix spp.*, *Dasiphora fruticosa*, have been occurred in all years since fire in PSP №8p. The projective cover of shrub layer ranged from 1.5 to 4.6% in PSP №8p. The projective cover of shrub layer was not significantly different between 8, 10, 13, 14, 17, and 19 years after the fire (Tukey-Kramer test, $P > 0.05$). However, significant difference was found only between 8 and 14 years after the fire (Tukey-Kramer test, $P < 0.05$) due to increase of *Lonicera altaica* (Table 3).

In PSP №8p, herbaceous cover was 55.5% in AF5 data, where *Chamaenerion angustifolium* dominated. Herbaceous cover was 17.4±1.61 in AF13, where *Poa pratensis* (4.4±1.14) and *C. angustifolium* (3.5±0.54), *Poa attenuate* (3.5±1.3) dominated. In AF19 data, herbaceous cover was 11.8±1.6%, where *P. attenuate* (3.38±1.02), *P. botryoides* (3.13±0.9), *C. angustifolium* (1.8±0.3) dominated (Fig. 3a). The Tukey-Kramer test revealed no significant difference in herbaceous cover between the 6 years after the fire ($P > 0.05$). By fifth year after the fire, ground vegetation develops into **fireweed community** dominated

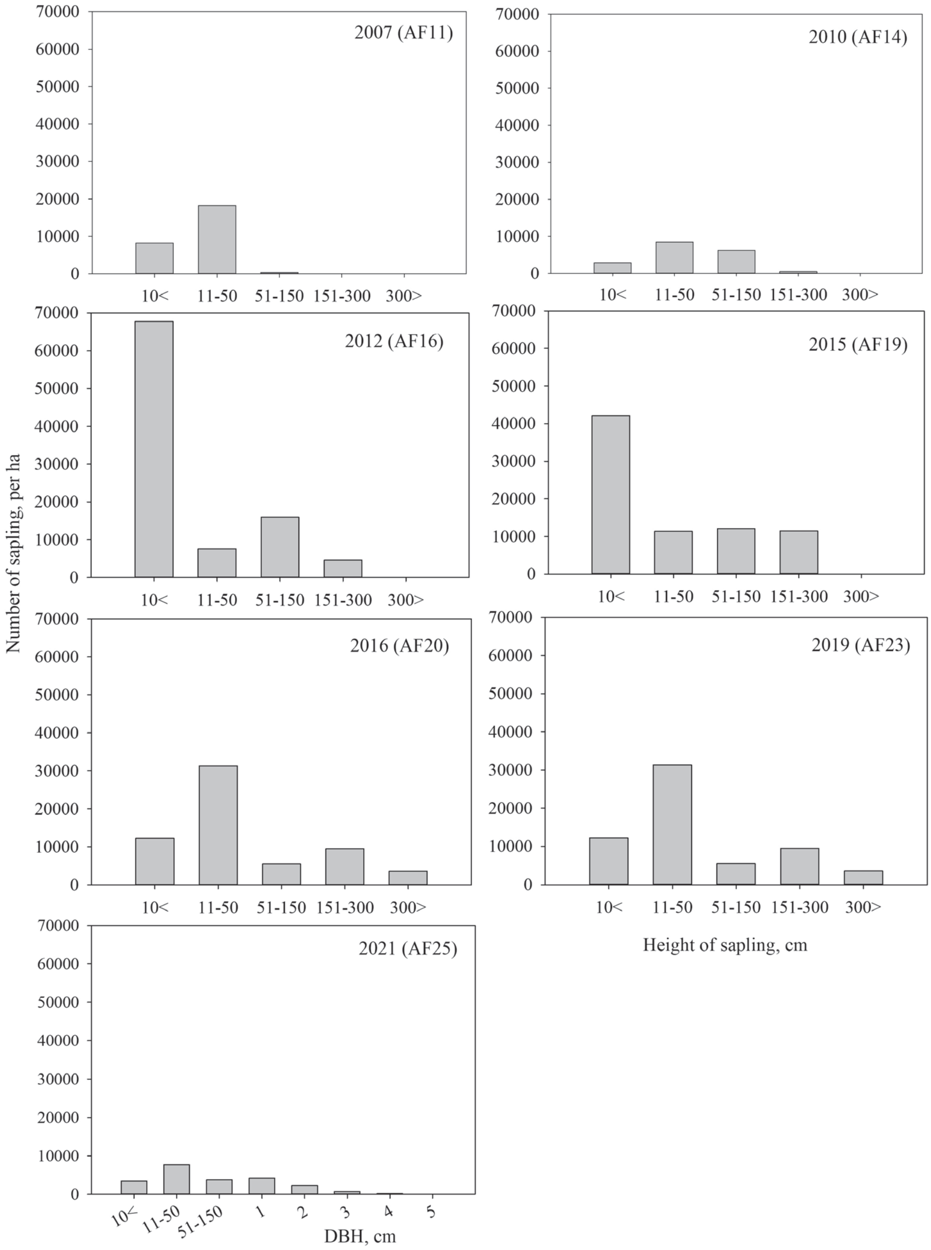


Fig. 2b. Change in the number of larch saplings in PSP №13 height class. DBH - woody plant with up to 5 cm diameter at breast height

Table 3. Change of plant coverage, and species diversity indices for years after the fire of 2002 (PSP №8p)

Variables	Sample plots					
Years after the fire	AF8	AF10	AF13	AF14	AF17	AF19
Coverage of shrub	1.5[b]	3.75[ab]	4.6[ab]	5.25[a]	3.6[ab]	3.8[ab]
Coverage of herb	12.4[a]	15.8[a]	17.5[a]	17.3[a]	12.75[a]	11.8[a]
Coverage of moss and lichen	17.8[a]	16.6[a]	6.3[b]	4.9[b]	3[b]	1.5[b]
Shannon index (H')	1.3[a]	1.6[ab]	1.84[b]	1.83[b]	1.84[b]	1.6a[b]
Simpson index (D)	0.62[a]	0.71[a]	0.79[b]	0.78[b]	0.79[b]	0.76[b]
Species richness (number of species)	7[a] (22)	9.9[b] (31)	9.6[b] (30)	9.31[b] (28)	8.81[ab] (31)	6.63[a] (24)
Evenness (J)	0.7[a]	0.7[a]	0.8[b]	0.8[b]	0.8[b]	0.86[b]

[a, b] Different alphabet denote significantly different among the years after the fire (Steel Dwass test in BZ, $p < 0.05$).

by *Chamaenerion angustifolium*. Species such as *Achillea millefolium, Cerastium pauciflorum, Saussurea stubendorfii, Taraxacum officinale, Koeleria macrantha, Poa attenuate, Poa pratensis, Sedum aizoon, Veronica longifolia*, annuals and biennials such as *Androsace septentrionalis, Corydalis sibirica, Erigeron acer*, fire mosses such as *Funaria hygrometrica, Marchantia polymorpha* newly appeared during years after the fire. In 10 years after the fire, **fireweed-forbs community** dominated by forbs such as *C.angustifolium, Taraxacum officinale* was formed. At the same time, new species such as *Artemisia sericea, Draba nemorosa, Rumex acetosa, Agropyron cristatum* emerged. Within 13-19 years after the fire, a **grasses-forbs** community was formed with the dominance of grasses, including *Poa pratensis, Poa botryoides, Poa attenuata*, and forbs as *C. angustifolium, Taraxacum officinale* (Fig. 3a).

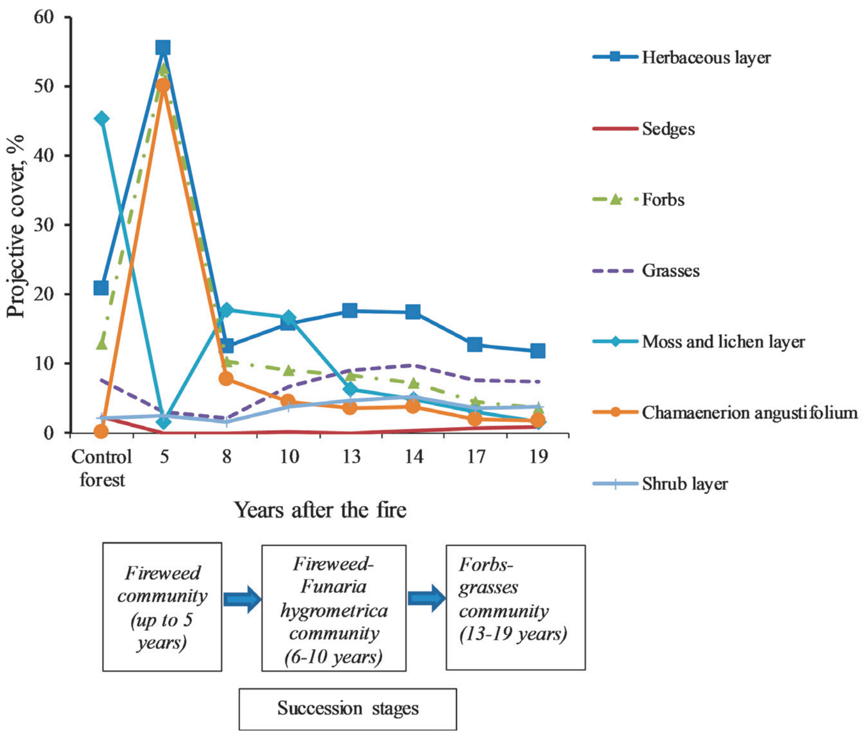
At the early stage of the post-fire succession, the first colonizers as fireweed, *Funaria hygrometrica, and Marchantia polymorpha*, appeared from seeds dispersed over the burned area, and a fireweed community dominated by *Chamaenerion angustifolium* was established by the fifth year after the fire.

The projective cover of lichen and moss layer in control forest was 45.3%, where *Rhytidium rugosum* (27.13%) and *Abietinella abietina* (17.78%) dominated. There was a sharp decline in moss cover due to decrease of *Rhytidium rugosum* immediately after the fire. The cover rose significantly 8 years after the fire due to growth of *Funaria hygrometrica* as fire moss. It decreased significantly 10 years after the fire (Tukey-Kramer test, $P < 0.05$) (Fig. 3a).

The species composition and coenotic percentage similarity indexes of ground vegetation between the control stand and the burned forest (PSP №8p) show a significant difference, indicating a noticeable change in the plant community of the burned area (Fig. 4). Sorenson species composition similarity index ranged from 16.13 to 36.36, with the value from 17 years after the fire being the highest. Coenotic percentage similarity index ranged from 0.42 to 6.23, with the value from 17 years after the fire being the highest and 5 years after the fire as the lowest (Fig. 4a).

In PSP №13, very few shrubs were present in burned forest, and we found only two species (*L.altaica and Cotoneaster melanocarpa*). The projective cover of the shrub layer varied between 0.07 and 0.25% but did not significantly differ among the 7 years (Tukey-Kramer test, $P>0.05$) (Table 4).

(a)



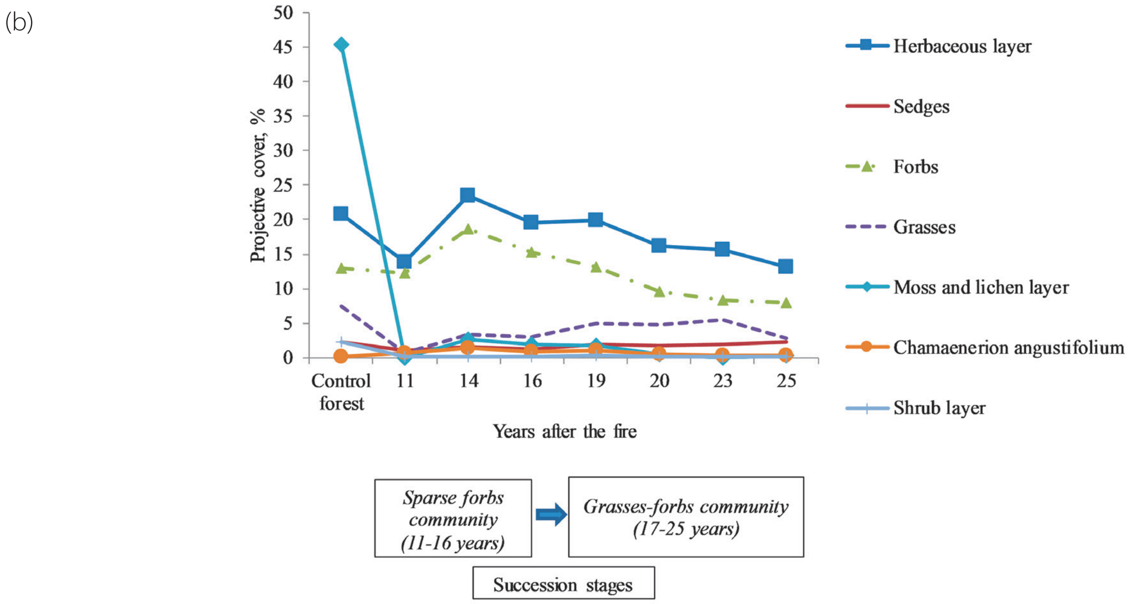


Fig. 3. Dynamic in projective cover of burned forbs-Rhytidium mosses pseudotaiga larch forest (a-in PSP №8p, b-in PSP №13)

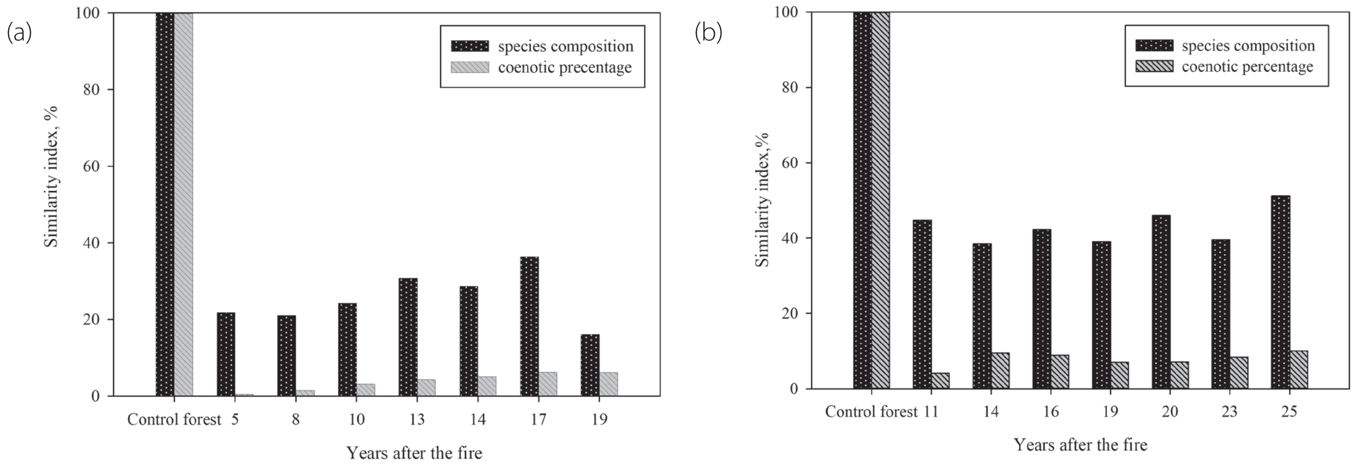


Fig. 4. Similarity indexes of plant communities in burned and unburned stands of forbs-Rhytidium mosses larch forest (a-in PSP №8p, b-in PSP №13)

Table 4. Change of plant coverage, and species diversity indices for years after the fire of 1996 (PSP13)

Variables	Sample plots						
Years after the fire	AF11	AF14	AF16	AF19	AF20	AF23	AF25
Coverage of shrub	0.07^{a}	0.16^{a}	0.15^{a}	0.25^{a}	0.2^{a}	0.08^{a}	0.18^{a}
Coverage of herb	13.9^{a}	23.5^{a}	19.6^{a}	19.9^{a}	16.1^{a}	15.7^{a}	13.1^{a}
Coverage of moss and lichen	0	2.68^{a}	2.03^{a}	1.75^{a}	0.43^{a}	0	0.3^{a}
Shannon index (H')	1.26^{a}	1.85^{b}	2.2^{c}	2.17^{c}	2^{bc}	2^{bc}	1.9^{bc}
Simpson index (D)	0.7^{a}	0.8^{b}	0.9^{b}	0.8^{b}	0.8^{b}	0.8^{b}	0.78^{b}
Species richness (number of species)	6.3^{a} (32)	10.3^{b} (43)	14.5^{c}(55)	13.4^{bc}(52)	12.15^{bc}(52)	12.6^{bc}(51)	10.2^{ab}(47)
Evenness (J)	0.8^{a}	0.8^{a}	0.85^{a}	0.8^{a}	0.8^{a}	0.8^{a}	0.8^{a}

$^{a, b}$ Different alphabet denote significantly different among the years after the fire (Steel Dwass test in BZ, $p < 0.05$).

The projective cover of herbaceous layer was not significantly different between the 7 years (Tukey-Kramer test, P > 0.05). The plant community succession trend in the burned area (plot13) was found to be similar to that of PSP №8p (Fig. 4b).

From 11 to16 years after the fire, sparse forbs community was developed by the dominance of forbs, including *Artemisia gmelinii, Potentilla bifurca, Taraxacum officinale, Artemisia macrocephala*. In 19-25 years after the fire, grasses-forbs community was found by dominant species such as *Poa attenuate, Poa botryoides, Artemisia gmelinii* (Fig. 4b).

On the burned sample plot (PSP №13), the ground vegetation of which was grazed severely by the livestock, newly appeared ruderal species such as *Chenopodium album, Urtica cannabina, Potentilla conferta, Veronica incana, Plantago major, Schizonepeta multifida, Leontopodium ochroleucum, Artemisia Adamsii, A.dracunculus, A.frigida, A.scoparia, Heteropapus hispidus, Elymus dahuricus, E. sibiricus, and K. macrantha, annuals and biennials as well as C. sibirica, A. septentrionalis, D. nemorosa, Lappula myosotis, Artemisia macrocephala, E. acer,* and fire mosses such as *Ceratodon purpureus, F. hygrometrica*.

The projective cover of lichen and moss layers in the control forest was 45.3%, where *Rhytidium rugosum* (27.13%) and *Abietinella abietina* (17.78%) dominated. It dropped dramatically post-fire (Fig. 3b). The projective cover of lichen and moss layers was not significantly different between the 7 years after the fire (Tukey-Kramer test, P > 0.05) (Table 4).

Species composition and coenotic percentage similarity indexes of plan communities between the burned forest and unburned larch stand was shown in Fig. 4. Sorenson species composition similarity indexes of plan communities between the burned forest and unburned larch stand were 38.46-51.22 during 11-25 years after the fire. There was the highest similarity (51.22) for 25 years after the fire and the lowest similarity for (38.46) for 14 years after the fire. Coenotic percentage similarity indexes were 4.12-10.04. There was the highest similarity in 25 years after the fire. However, the lowest similarity was found in 11 years after the fire (Fig. 4b).

Species richness and species diversity change in burned forest

For PSP №8p, the Shannon index was 1.3 by 8 years after the fire (AF8). Then it increased significantly from 1.83 to 1.84 (Steel Dwass test, p < 0.05), which indicates a slight increase in species diversity in old-burned areas (AF13, AF14, AF17) and the diversity index value can be considered very low (Fernando 1998). In the burned area, which was overgrazed (PSP 13), the species diversity increased gradually (Steel Dwass test, p < 0.05), ranged from 1.2 to 2.2 and the values of the Shannon index belong to the class from very low to low (Tables 3, 4).

Species composition

DCA for PSP №8p indicated that the eigen values of axis 1 and 2 were negligible (0.23). However, isolated subquadrates post fire years may be grouped along the second floristic axis into three groups with same species composition: 1) AF8 and AF10; 2) AF13 and AF14; 3) AF17 and AF19 (Fig.6a).

The differences of species composition between groups can be explained by change of the shrub, moss, and

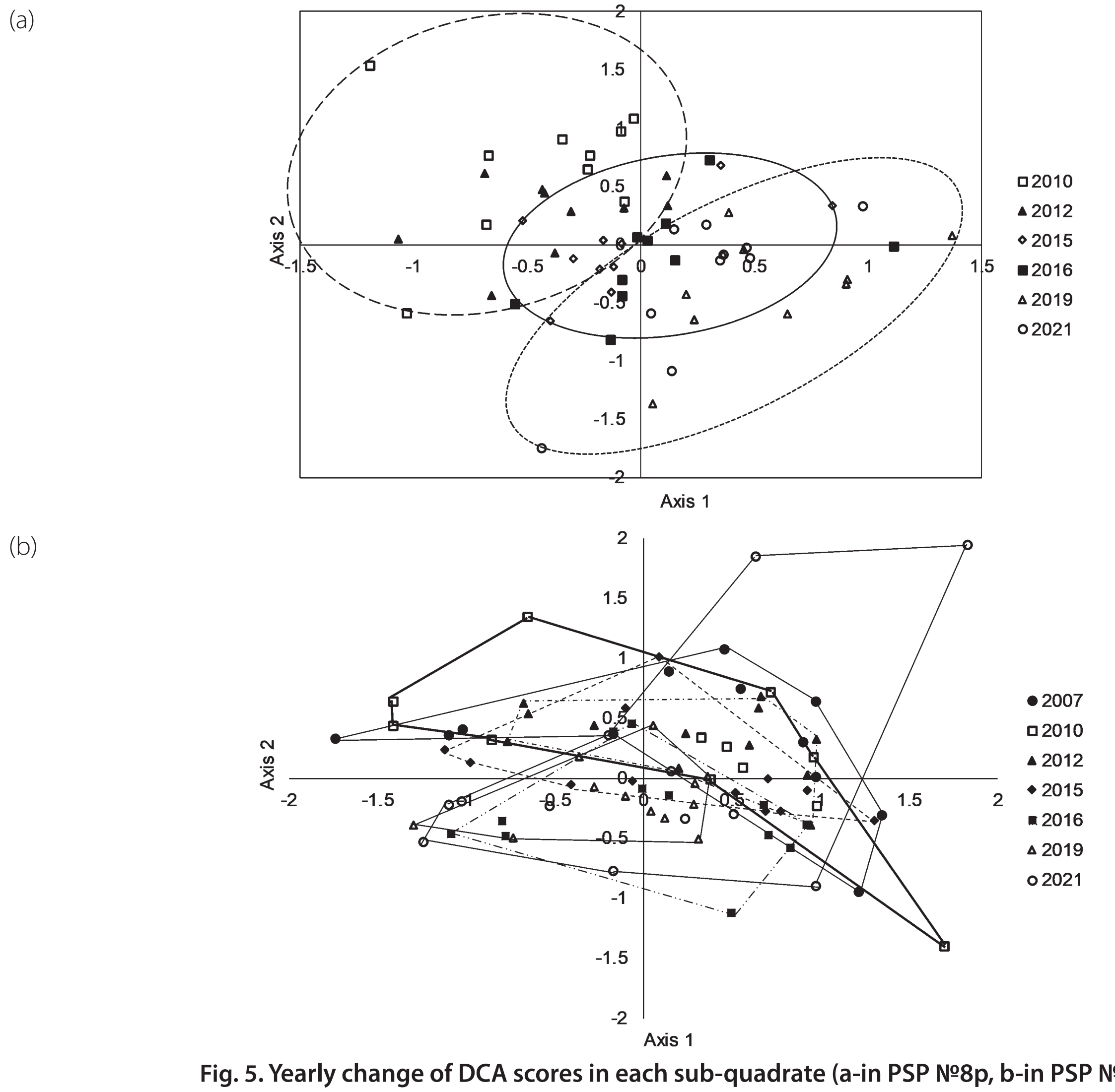


Fig. 5. Yearly change of DCA scores in each sub-quadrate (a-in PSP №8p, b-in PSP №3)

lichen layers. Moss and lichen layers in AF8 and AF10 were significantly higher than AF14, AF17, and AF19. The shrub layer in AF14 was significantly higher than AF8 (Table 3). In PSP №13, the eigen value of axis 1 and 2 were 0.40 and 0.21, respectively. The variance of species composition decreased annually. On the overgrazed burnt area (PSP 13), subquadrates were not grouped by species composition during post-fire years due to grazing of livestock on the ground vegetation and undergrowth (Fig. 5b).

DISCUSSION

Our findings suggested that the pseudo-taiga larch forests damaged by large-scale high-intensity fires in our study area were successfully regenerated by Siberian larch through the following serial succession stages: fireweed (*Chamaenerion angustifolium*) community (up to 5 years after fire), fireweed-bonfire moss (*Funaria hygrometrica*) community (from 6 to 10 years), forbs community (11-16 years), and grass-forbs young larch forest (17-25 years). The expansion of *Funaria hygrometrica, Chamaenerion angustifolium*, and *Corydalis sibirica* on the burnt areas is common in the early stage of the pyrogenic succession of larch forests in the permafrost zone of Central Siberia and Central Yakutia, Mongolia (Takahashi 2006; Dorjsuren 2009; Lytkina and Mironova 2009; Zyryanova et al. 2010).

The primary characteristic of post-fire succession in the larch forest of the Central Khangai region, including the Tarbagatai ridge, is regeneration without tree species replacement. It is generally associated with the cold and dry climate of the Central Khangai region, where Siberian larch trees are distributed and deciduous tree species such as birch and aspen are absent. In contrast, in the Khentey region and North-Eastern Khangai province in Mongolia, as well as in Siberia, Russia, the burned larch forest areas can be regenerated by birch, and aspen tree species or mixed with larch trees (Abaimov and Sofronov 1996; Abaimov et al. 2002; Lytkina and Mironova 2009; Dorjsuren 2009; Zyryanova et al. 2010; Otoda et al. 2013; Undraa and Dorjsuren 2017).

Fires burn thick moss layers in *forbs-Rhytidium* mosses-larch forest and by this create favorable conditions for the regeneration of larch seedlings. In forest stands with a dense moss cover, the emergence and peak of young larch regeneration mostly occur after high-severity fires due to reducing stand competition for moisture and nutrients. Additionally, they destroy shrub and herbaceous layers, moss cushion, slash and litter, which hinder the emergence and development of larch seedlings. Fires enhance soil mineralization, enrich nutrients through ash deposition, and raise temperatures, all of which contribute to favorable conditions for seed germination, seedling rooting, and the development of a self-sown crop (Matveev and Usoltsev 1996; Babintseva and Titova 1996; Sofronov and Volokitina 2010). Moreover, post-fire successful larch tree regrowth depends on soil hydrology such as high plant-available field capacity and hydraulic conductivity in the uppermost soil horizons, which reduces the evaporation loss and the competition of larch saplings with grasses and herbs for water in pseudo-taiga larch forests under semi-arid conditions of the Tarvagatai Mountain range (Schneider et al. 2021).

In the Tarvagatai Mountain Range, 114,3 thousand hectares or 52% of forest areas were damaged by fire in 1996, 2002 and 2003 (Jagdag and Gerelbaatar 2016). In Central Khangai, 70 % of burned forests are regenerating successfully by larch trees (Tungalag and Dorjsuren 2017). According to a long-term study, 70-75 % of the forest area burned by high-intensity fires regenerates naturally (Dugarjav 2006). Our study reveals that the number of seedling and saplings 19-23 years after the fire ranged from 25.7 thousand to 46.2 thousand per hectare. In some areas affected by the 2002 fire, 71,100 young trees were counted per hectare in 2021. Therefore, we suggest the recommendation to conduct thinning in dense young stands originated post-fire. Thinning reduces tree density, crown closure, and fire intensity and increase forest productivity (Agee et al. 2006; Khongor and Tsogt 2019, Banerjee 2020).

CONCLUSION

In the Tarvagatai Mountain range, Siberian larch trees are predominated; however, deciduous tree species such as birch and aspen are absent due to the cold and dry climate of the Central Khangai region. *Forbs-Rhytidium* mosses pseudo-taiga larch forests disturbed by the large-scale forest fires in the Tarvagatai Mountain range were sufficiently recovered by Siberian larch without tree species replacement. Post-fire regenerative succession proceeds through several serial stages and establishes grass-forbs young larch forest in 17-25 years after fire. Species numbers gradually increased with time in the forest affected by fires, whereas they rose drastically in the forest damaged by fire and livestock browsing due to the increase of ruderal species. Despite the long recovery period, the post-fire similarity indexes of species composition and coenotic percentage compared with the control forest were relatively low, indicating a slow pre-fire vegetation recovery.

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