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After Fire Regenerative Successions in Larch (*Larix sibirica* Ledeb.) Forests of the Central Khangai in Mongolia

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The changes in the composition and the projective cover of understory species and regeneration after fires in 1996 and 2002 in pseudotaiga larch forests of the Central Khangai, Mongolia have been investigated. Descriptions of vegetation and inventory of natural regeneration were carried out in 2007, 2010 and 2012 on permanent sample plots that were established in each of two sites. According to the Detrended Correspondence Analysis (DCA), no drastic changes were observed in species composition of the plant community during 11, 14, 16 years after fire of 1996 and 8, 10 years after fire of 2002. Regarding the plant cover, remarkable change was not observed in vegetation, except in 5 year-old stand. Grasses-herbs community with young larch saplings 10 years after fire of 2002 was formed. Loose herbs community with young larch saplings 16 years after fire of 1996 was established. Moreover plant life forms were identified during the succession years post-fire. By Christen C. Raunkiær life form classification (1937), life forms such as cryptophytes and hemicryptophytes either kept or shared their dominant position following fire. These plants seem to be fire-resistant and have a high ability to recover post-fire. After the fires, natural regeneration was successful. However, grazing may affect negatively to larch regeneration. It can be explained by difference of large saplings in two plots. The number of large saplings per hectare was low despite the long recovery period in one plot.

Keywords: *forest fire, succession, regeneration, sapling, seedling, pseudotaiga, forest conservation, detrended correspondence analysis, C. C. Raunkiær life form classification, Tarbagatai Mountain Range, Central Khangai, Mongolia*.

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INTRODUCTION

Understanding succession is important to predicting the changes that forest vegetation undergoes after fire as it recovers. Fuller (1991) stated that succession refers to the order in which plants return to an area after fire or other disturbance. Previous studies have suggested that boreal forests have been shaped and controlled by fire for several hundred years (Gromtsev, 2002; Shorohova et al., 2009). Likewise, Mongolian forests have been formed by fire, which determines the spatial and temporal dynamics of forest ecosystems (Valendik et al., 1998; Goldammer, 2007). Productivity of Siberian larch has decreased and composition of the boreal forest in northern Mongolia may be altered by changes in the fire regime due to climate warming (Dulamsuren et al., 2011) and intensive human impact (Otoda et al., 2013).

Limited studies on the succession and recovery in larch stands after fire were conducted. After fire, natural regeneration of larch stands in Siberian forest, progressed in 4 different directions depending of fire intensity: 1) secondary *L. sibirica* stand, 2) deciduous stand, 3) bush stand and, 4) grassland (Park, 2005; Takahashi, 2006). Zyryanova et al. (2010) studied recovery of larch forests vegetation after fire disturbance in Central Siberia. They identified stages of post fire progressive succession. According to their study, mature larch forests with a continuous lichen-moss mat develop in about 90–100 years after strong running ground fire. Moreover, final restoration of the prefire plant species composition and the cover percentage of the species are completed during a period of 50 to 90 years. The larch forests have developed without replacement of dominant tree species. Although birch trees (*Betula pendula*) occur with larch trees following the fire, larch trees keep their dominant position in the successional years (Zyryanova et al., 2010). Moreover by study in Central Yakutia, Eastern Siberia, Russia the recovery of the plant community after low intensity fire takes 50–60 years (Lytkina, Mironova, 2009). There have formed larch forests. However birch trees (*Betula platyphylla*) were found during the successional stages and they disappeared in later successional stages.

A. P. Abaimov et al. (2002) have determined general scheme of postfire forest dynamics in the uneven-aged sparse and closed larch forests, cryolithic zone of Siberia. After weak and middle fires progressive successions can develop either without larch forest change or by birch and even by vegetation type replacement. In the latter case they are replaced by secondary shrub tundras (Abaimov et al., 2002). Pyrogenic factor affect positively on post-fire regeneration of larch forest in the Central Siberia and its success depends on fire intensity, relief, seed rich year (Tsvetkov, 2006; Matveev et al., 2012).

Long-term monitoring survey in pseudotaiga larch forests, in Central Khangai, Mongolia for 38 years (1972–2010) was conducted (Dorjsuren, 2008; Dorjsuren et al., 2013). He found that after high intensity fire larch forests have formed and the reestablishment of vegetation cover of diverse herbs-rhytidium larch forest is slow (25–38 years). Scale of fire is small (about 2 ha). The Khangai region is considered a region of low fire risk because of a relatively humid cool climate at high altitude. However, due to fires in 1996, 2002 and 2003, significant areas (65 751 ha) of larch forest were destroyed by fire in the Tarbagatai Mountain Range as one of the range Khangai mountains (Dorjsuren et al., 2013). The scale of the fire was large and destructive to forest resources. In the Central Khangai, surface fires are dominant in pseudotaiga larch forests, which grow on permafrost soils. They account for 96 % of the total forest fires that occur there (Dorjsuren, 2009). Studies by P. M. Matveev and E. N. Savina (1977) have shown that in forests of the Central Khangai the fire season is divided into spring from late March to mid-June, and autumn from late August to mid-October. High forest fire risk occurs in these periods due to limited precipitation in fall and spring, the presence of strong winds, the prevalence of low relative humidity, abundant dead litter and, lack of vegetative herbaceous vegetation. Herbs-Rhytidium mosses forest type is the most prone-fire type in Central Khangai. The area damaged by fire has increased particularly in 1996–1998 in Mongolia due to climate change and weak management of the government in transition from a centrally planned economy to a market economy (Goldammer, 2007) and inappropriate human activities (Chuluunbaatar, 2008). The result of tree ring study in Tarbagatay Mountains indicated that a warming trend was observed much since 1900s. Tree growth at high-elevation is limited by temperature, not precipitation (Jacoby et al., 1996). In addition, the climate has become warmer in the territory of Zavkhan province under the climate change caused by the effect of green house gas due to destructive human activities. For instance, the number of hot days was increased by 25 days. The plant growth period was increased by 23 days (Gomboluudev, 2012).

To undertake any activities in post-fire stands, it is important to clarify regeneration and vegetation recovery following fire. We tested: 1) change in plant community and recovery in larch stands after large-scale fires; 2) correlation between herbaceous plant coverage and some environmental factors (soil electricity conductivity, soil moisture); 3) impact of grazing to the forest recovery in terms of grazing pressure.

MATERIALS AND METHODS

Study site. Fieldwork was carried out in the Tarbagatai Mountain Range, Tarbagatai Nuruu National Park, located 890 km northwest of Ulaanbaatar, Mongolia (Fig. 1).

The study sites were located in common pseudo-taiga larch forests, which are composed

Fig. 1. Location of the study sites.

of pure Siberian larch (*Larix sibirica* Ledeb.); the herb stratum of these stands was dominated by *Rhytidium* mosses (Ogorodnikov et al., 1983). The sites were assigned two management levels: restricted zone and buffer zone. The annual average air temperature is –4.8 °C, the annual average precipitation is 209.5 mm (Institute …, 2012), and the plant growth-intensive period is 141 days long (Dorjsuren, 2009).

Field survey. Fieldwork was conducted in late July and early August of 2007, 2010, and 2012. Two permanent study plots were established in each of the restricted zone (RZ; 48°16′N, 98°21′E, burned in 2002) and the buffer zone (BZ; 48°27′N, 98°15′E, burned in 1996) (Table 1).

Study plots were 30 × 50 m in the BZ and 20 × 20 m in the RZ, divided into 15 and 16 subplots, respectively. Subplots were 10 × 10 m in the BZ and 5 × 5 m in the RZ. Study quadrats (2 × 2 m) were then established in each subplot. We measured the diameter at breast height (DBH; breast height = 1.3 m) and height of all standing trees and counted the number of saplings in each subplot. Saplings were divided into five height classes: 0–10, 10–50, 50–150, 150–300, and ≥ 300 cm. Plants under 1 year old were considered seedlings, whereas plants ≥ 2 years old were considered saplings. Upper limit of a saplings is a woody plant with up to 5.0 cm diameter at breast height. We conducted a vegetation survey in all quadrats, wherein species composition and cover were recorded in the shrub, herbaceous, and moss-lichen layers. The cover of dominant species was measured in 2007 in the RZ. Cover was estimated visually according to the Drude Scale (Schennikov, 1964). 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). We used a WET-2 sensor (Delta-T Devices Ltd., UK) to measure soil electricity conductivity and soil moisture. We only measured environmental variables in 2012.

Table 1

Information of study plots

Site name	RZ	BZ
Year of the fire	2002	1996
Plot size, ha	0.04	0.15
Altitude, m	2072	1963
Slope angle, °	15–18	5–7

Data analysis. The measurement data were labeled with time since fire. In the RZ, data from 2007 were labeled as AF5 (5 years since the last fire), data from 2010 were labeled as AF8, and data from 2012 were labeled as AF10. Likewise, in the BZ, data from 2007, 2010, and 2012 were labeled as AF11, AF14, and AF16, respectively. In total, 12 quadrats in the BZ and 10 in the RZ were used for species analyses over time post-fire. To examine fire resistance and restoration of plants after fire, shrubs, herbs, mosses, and lichens were classified using Christen C. Raunkiær life form classification (1937). Shannon-Weaver (Shannon, Weaver, 1949), Simpson diversity (Smith, Wilson, 1996), and Pielou’s evenness (Pielou, 1975) indices were calculated to identify changes in species diversity following fire. In addition, plant species occurring in burned areas were classified by their natural habitats (Grubov, 1982; Dorjsuren, 2009). Detrended correspondence analysis (DCA) was performed to identify differences in species composition among sub-quadrats in plots. All statistical analyses were performed using R version 2.15.2 (R Development Core Team, 2012).

RESULTS

Tree dynamics and regeneration. No living standing trees were left in the RZ. All standing dead trees had already been felled by 5 years post-fire (AF5). In the BZ, we compared forest structures of the AF11 and AF16 time categories (Table 2).

Living tree density decreased from 80.0 to 66.6 trees per ha, and dead tree density decreased from 193.0 to 160.0 trees per ha. At

Table 2

Change in forest structure in BZ from AF11 to AF16

Year of the fire	AF11	AF16
Living tree density, trees per ha	80.0	66.6
Dead tree density, trees per ha	193.0	160.0
Maximum DBH size, cm	40.0	43.6
Basal, m^2/ ha	4.0	4.5

AF11, the average diameter of living trees was (22.3 ± 11.9) cm and the average diameter of dead trees was (11.0 ± 3.4) cm; living trees had a larger diameter than dead trees ($P < 0.05$). At AF16, the average diameter of living trees was (27.5 ± 10.9) cm and the average diameter of dead trees was (12.5 ± 3.9) cm; living trees had a larger diameter than dead trees ($P < 0.05$).

Sapling age varied from 1 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. The numbers of larch saplings in each height class in the RZ and BZ are shown in Fig. 2.

Seedlings were not recorded in either AF8 or AF10 data. The total density of saplings in subplots was 41 900 trees per ha in AF5, 26 875 per ha in AF8 and 39 375 per ha in AF10.

In total, 1 125 seedlings per ha and 26 875 saplings per ha were found in AF11 and 250 seedlings per ha and 18 125 saplings per ha were found in AF14 data. In total, 6000 seedlings per ha and 96 000 saplings per ha were found in AF16 data. However, more than 70 % (67 750 saplings per ha) of saplings were less than 10 cm in height.

Understory plant community dynamics. Understory changed annually cover, with shrub cover ranging from 1.5 to 4.2 % in the RZ. Five species such as *Salix* spp., *Potentilla fruticosa*, *Spiraea media*, *Lonicera altaica*, and *Ribes nigrum* were recorded in the RZ. In other words, *Potentilla fruticosa*, *Salix* spp., and *S. media* occurred in AF5. In AF8 *L. altaica* and *R. nigrum* were observed in the shrub layer, whereas *S. media* was absent. *Salix* spp. occurred in all years since fire in this study. No difference was observed between AF8 and AF10 data with respect to shrub layer cover ($P = 0.15$). In the BZ, very few shrubs were present in the forests, and we found only two species (*L. altaica* and *Cotoneaster melanocarpa*). Total shrub layer cover varied between 0.08 and 0.4 %, but did not significantly differ among the 3 years ($P = 0.41$) (Table 3).

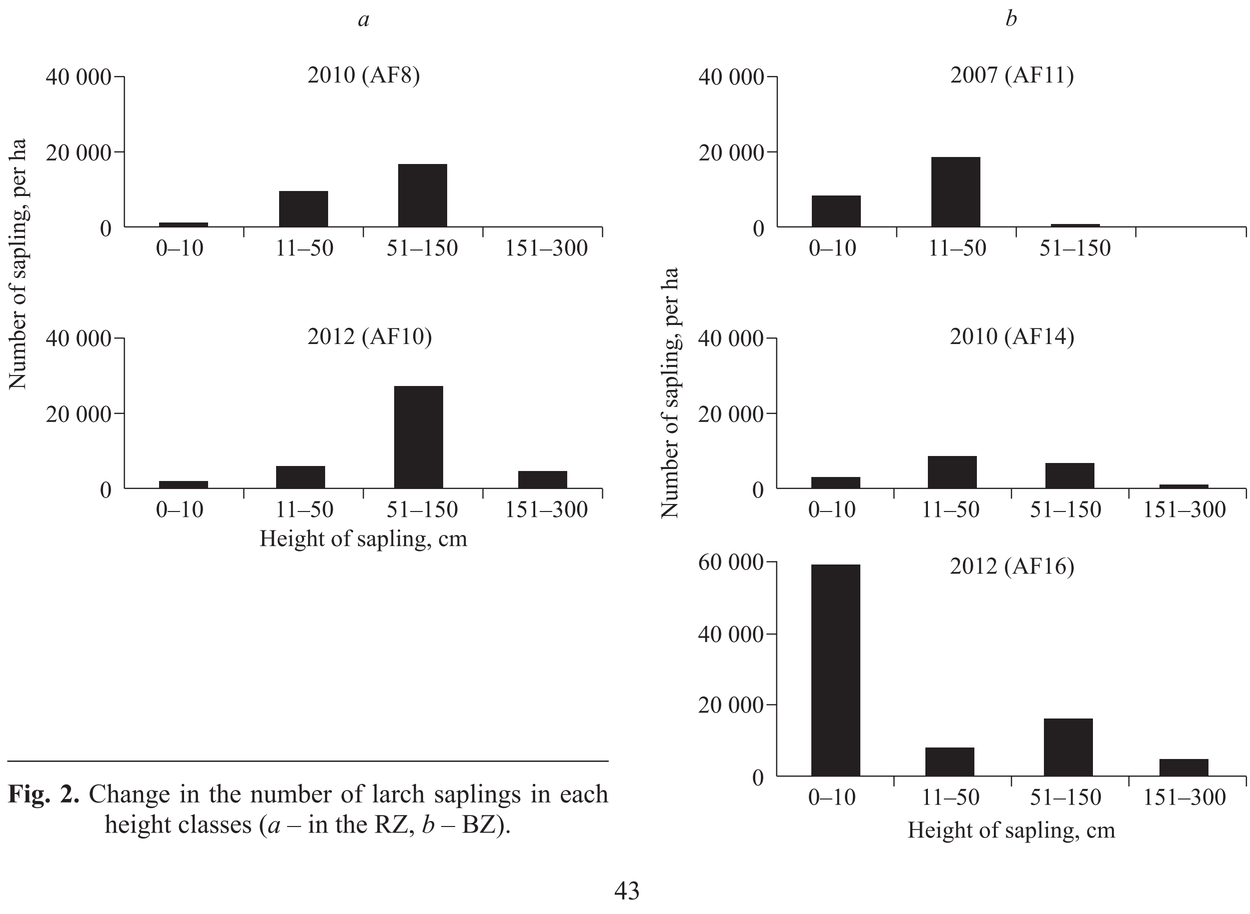

Fig. 2. Change in the number of larch saplings in each height classes (*a* – in the RZ, *b* – BZ).

Table 3

Change of plant coverage, species diversity indices for years after the fire

Location	RZ	RZ	BZ	BZ	BZ
Years after the fire	AF8	AF10	AF11	AF14	AF16
Coverage of shrub	1.5 ± 1.4	4.2 ± 3.2	0.1 ± 0.0	0.2 ± 0.0	0.2 ± 0.2
Coverage of herb	13.1 ± 6.0	17.0 ± 8.3	13.2 ± 11	27.2 ± 19	20.5 ± 16
Coverage of moss	18.6 ± 11.0	17.6 ± 8.0	0	3.2 ± 1.4	2.0 ± 1.7
Shannon index (H´)	1.3 ± 0.4[a]	1.65 ± 0.3[b]	1.3 ± 0.5[a]	1.9 ± 0.4[b]	2.2 ± 0.2[b]
Simpson index (D)	0.6 ± 0.13	0.7 ± 0.1	0.6 ± 0.2[a]	0.8 ± 0.1[a]	0.8 ± 0.1[b]
Species richness	6.6 ± 1.9	10.6 ± 2.8	6.2 ± 3.4	11.2 ± 3.5	15.3 ± 4.7
Evenness (J)	0.7 ± 0.1	0.7 ± 0.1	0.8 ± 0.2[a]	0.8 ± 0.1[a]	0.8 ± 0.1[a]

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

In the RZ herbaceous cover was 55.5 % in AF5 data, where *Chamanerion angustifolium* (50 %) and *Festuca ovina* (3 %) dominated. In AF8 data herbaceous cover was 13.1 ± 6.0, where *C. angustifolium* (8.5 ± 5.5) and *Taraxacum officinale* (2.1 ± 1.1) dominated. In AF10 data herbaceous cover was 17.0 ± 8.3, where *C. angustifolium* (5.3 ± 3.8), and *Poa attenuata* (2.9 ± 2.4), *T. officinale* (2.8 ± 1.5) dominated. Dominance was mainly affected by a decrease in *C. angustifolium* cover, after which ruderal species (*T. officinalis*) and dense tussock grass (*P. attenuata*) appeared. No difference existed between AF8 and AF10 data ($P = 0.40$). In the BZ, herbaceous cover was 13.2 ± 11.0 in AF11, 27.2 ± 19.0 in AF14, and 20.5 ± 16.0 in AF16 data. Although herbaceous cover increased by 14 % in AF14, it decreased by 7 % in AF16 data.

However, total herbaceous cover was not significantly different among the 3 years ($P = 0.08$). The dominant species were *Artemisia gmelinii* and *Potentilla bifurca* in AF11, *T. officinale*, *A. gmelinii*, and *P. bifurca* in AF14, and *A. gmelinii*, *T. officinale*, and *Artemisia macrocephala* in AF16 data.

In the RZ the moss *Funaria hygrometrica* comprised 1 % of the moss layer in the AF5 stand and the lichen *Peltigera canina* made up 0.5 %. *F. hygrometrica* was dominant, with a cover of 17.4 ± 11.3 in AF8 data. *Marchantia polymorpha* was present but its cover was only 0.1 ± 0.3. *F. hygrometrica* cover was 16.2 ± 2.4 in AF10 data. With respect to moss cover, no difference existed between AF8 and AF10 data ($P = 0.91$). In AF8 and AF10 data moss and lichen cover seemed to increase compared to AF5. In the BZ moss was absent in AF11, one moss species (*F. hygrometrica*) was present in AF14, and two moss species (*F. hygrometrica* and *Ceratodon purpureus*) were present in AF16 data. However, total moss cover was not significantly different among the 3 years ($P = 0.13$).

For the RZ Shannon index showed that there was significant difference between 8 and 10 years (U test, W = 19, $p < 0.05$). Regarding to the BZ, Shannon index showed that there was significant difference between 11 and 14 years (Steel-Dwass test, $t = 2.42$, $p < 0.05$), between 11 and 16 years (Steel-Dwass test, $t = 3.69$, $p < 0.001$). However, there was no significant difference between 14 and 16 years (Steel Dwass test, $t = 2.19$). The Simpson diversity index implied that there was significant difference between 11 and 16 years (Steel Dwass test, $t = 3.17$, $p < 0.05$) (Table 3).

Life-forms. The yearly changes in C. C. Raunkiær (1937) life-forms in the RZ are shown in Fig. 3, *a*.

Hemicryptophytes were the major life-form in AF5 and AF10 data, but they were codominant with cryptophytes in AF8. Cryptophytes were the second major life-form in AF5 and AF10 data and mosses tended to increase. Therophytes were present in AF8 and AF10 data. According to natural habitat classifications, forest meadow species made up 61 % of species, forest steppe species 14 %, forest species 11 %, fire mosses 7 %, alpine species 4 %, and lichen 4 %. The changes in life-forms year to year in the BZ are shown in Fig. 3, *b*.

Cryptophytes kept their dominant position in succession in the years following fire, hemicryptophytes maintained their position as the second major life-form, and therophytes increased gradually as the third major life-form.

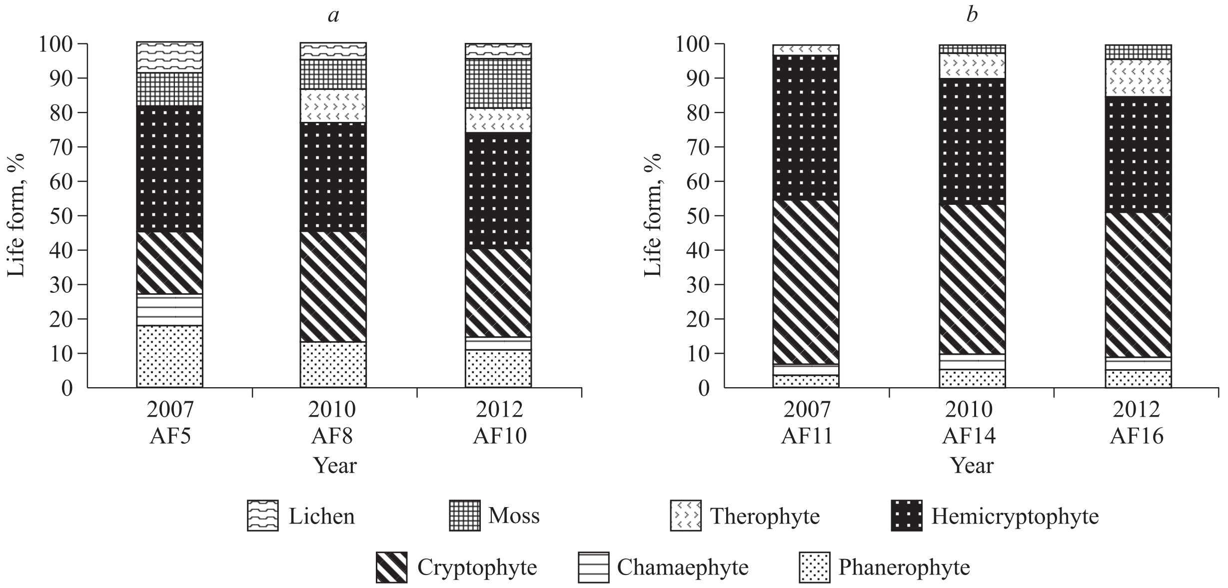

Fig. 3. The yearly change of life form (*a* – in BZ, *b* – RZ).

Forest meadow and forest steppe species were dominant, making up 46 and 34 % of species, respectively. Forest specialists comprised 13 % of species, alpine species 4 %, and fire mosses 4 %.

Species richness and species diversity. Mean species richness was not significantly different between the RZ and BZ, but their Shannon indices differed ($P < 0.05$). In the BZ, a significant difference in the Shannon index was observed between AF11 and AF14 data ($P < 0.05$) and between AF11 and AF16 ($P < 0.01$), but not between AF14 and AF16. Finally, a significant difference was detected between AF11 and AF16 data with respect to the Simpson diversity index ($P < 0.05$).

Species composition. In the RZ, species composition was not different between AF8 and AF10 data (Fig. 4).

The eigen values of axis 1 and 2 were 0.25 and 0.18, respectively. In the BZ, the variance of species composition decreased annually (Fig. 4, *b*). The eigen values of axis 1 and 2 were 0.51 and 0.20, respectively. Electrical conductivity was positively correlated with axis 1 ($r = 0.62$, $P < 0.05$). Soil moisture was not significantly correlated with axis 1 ($P = 0.31$). No significant correlation was observed between axis 2 and environmental factors (EC: $P = 0.57$; VWC: $P = 0.41$).

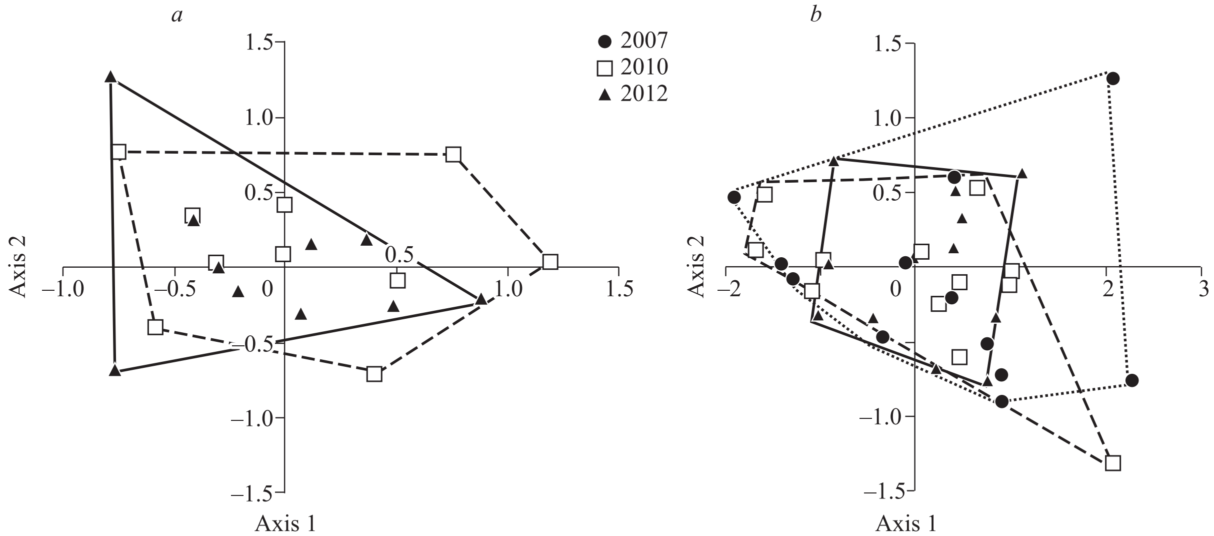

Fig. 4. Yearly change of DCA scores in each sub-quadrate (*a* – in RZ, *b* – in BZ).

DISCUSSION

After fire, the pseudo-taiga larch forests in our study area recovered without tree replacement. Zyryanova et al. (2007) suggested that forest regenerative processes appear to have similar trajectories in the global permafrost area. Permafrost zones occupy almost two thirds of Mongolia. It is considered that Mongolia is a southern fringe of the Siberian permafrost zone. According to the schematic map of permafrost distribution in Mongolia (Sharkhuu, 2003), Khangai belongs to the continuous and discontinuous (50–100 %) permafrost areas. Previous studies have also found that larch forests develop without tree replacement after fire (Takahashi, 2006; Dorjsuren, 2008; Lytkina, Mironova, 2009).

Seedlings and saplings may be restored via adult trees that survived a fire. However, large trees died in the RZ. During a high-intensity surface fire, lateral and surface roots of trees are badly damaged (Dorjsuren, 2008). Therefore, such a fire often leads to full or nearly complete destruction of forest stands. Some large diameter trees remained after fire in the BZ. In general, young trees are not well protected and are therefore easily damaged by fire (Archibold, 1995); thus, young trees were likely very flammable in the BZ.

The amount of saplings was sufficient for the successful tree regeneration at our study sites. Indeed, large saplings increased consistently year to year. However, the number of large saplings (51–150 cm) in the BZ was lower than in the RZ, which may be explained by grazing pressure. Livestock has increased over time in the BZ, according to an assessment report of protected areas (Management…, 2012). Grazing in Mongolia interrupts the natural regeneration of forests and damages young seedlings and saplings (Hilbig, 1995; Gunin et al., 1999; Dugarjav, 2006); currently, 3000 head graze in winter throughout BZ, Tarbagatai National Park.

Pseudo-taiga larch forests have a high sapling density post-fire. For example, the number of saplings and seedlings was 41 900 trees per ha in AF5 data. According to natural forest regeneration criterion in pseudo-taiga larch forest by Dugarjav (2006) , it is considered that while the amount of larch big (up to 101 cm) saplings is more than 2500 per ha post-fire regeneration is sufficient. Saplings are depressed by big trees and low light conditions (Korotkov, Dorjsuren, 1983). Sofronov and Volokitina (2010) demonstrated that the main barrier to larch regeneration in northern open woodlands is the moss and duff layer. Fire destroy this thick moss and litter (up to 7–8 cm) (Dorjsuren, 2009), allowing larch regeneration to occur in a new, thin organic layer.

However, sapling establishment was fluctuating in the BZ. Increases in grasses and sedges, and increased root competition with herbaceous plants after fire (Dorjsuren, 2009), cause a reduction in seedling establishment and death of self-seeded larch saplings. This may also be affected by variations in climate, year, grazing pressure, and human impact. In our study, sapling establishment is also closely related to seed-rich years of residual forest trees.

Recovery of the pre-fire plant species progressed slowly at our sites, but species richness and plant cover did not change post-fire. Moreover, a DCA also showed no large change in species composition. Cryptophytes and heterophyte were dominant in the successional stage after fire, in agreement with Rowe (1983), who stated that hemicryptophytes are moderately protected and cryptophytes are well protected from fire. Forest meadow and forest steppe species were common in AF10 and AF16 data. Forest steppe and meadow species make up a considerable part of the pseudo-taiga plant community (Dugarjav et al., 1975). One of the important criteria for determining plant restoration post-fire is the regrowth of moss, and moss cover was not yet established at our study plots. However, we observed fire bryophytes, such as *M. polymorpha*, *C. purpureus*, and *F. hygrometrica*, which appeared after fire. Many authors have observed that pyrophyte, ruderal species (*C. angustifolium*, *Corydalis sibirica*), and moss species favored post-fire (*M. polymorpha*, *C. purpureus*, *F. hygrometrica*) appear at the earliest stages of succession in Siberian larch forests (Lytkina, Mironova, 2009; Zyryanova et al., 2010). *Marchantia polymorpha*, *C. purpureus*, and *F. hygrometrica* are dominant after a high-intensity fire during early succession, and decrease progressively with time (Matthews, 1993; Esposito et al., 1999). Soil electrical conductivity was the only factor significantly correlated with vegetation data in the BZ. Kutiel and Naveh (1987) noted that electrical conductivity and pH increase significantly immediately after fire. Although fire decreases total nitrogen, available nitrogen is increased and is perhaps a major factor promoting the rapid growth of herbaceous plants.

During high-intensity surface fires, complete mineralization occurs in soil (Krasnoshchekov, Dorjsuren, 2007) and in mountain forests, the protective ground vegetation layer and litterare destroyed, which intensifies erosion (Krasnoshchekov, Dorjsuren, 2007). In the BZ, some trees were left alive post-fire. Here, our plot was located on a relatively gentle slope compared to the RZ, which along with altitude may have been an influence.

CONCLUSIONS

Pseudo-taiga larch forests developed after fire without tree replacement and the natural regeneration post-fire was satisfactory. However, the number of large saplings (50–150 cm) was lower in the BZ than in the RZ, being perhaps less protected from grazing. After fire, plant species diversity increased significantly with time due to an increase in early successional plants, although the recovery of common forest plants was impaired. No drastic differences were observed in plant cover among years, as the recovery of pre-fire plant species is relatively slow. Occasional fires stimulate regeneration of *L. sibirica*, but tree canopy recovery occurs only over the long term. In other words, frequent, intense fires will cause retrogression of the pseudo-taiga forest ecosystem. In addition, *L. sibirica* seems to regenerate under relatively low grazing pressure, and heavy grazing will slow the regeneration of trees and herbaceous plants considerably. Therefore, burned areas should be protected from overgrazing and future fires.

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ПОСЛЕПОЖАРНЫЕ ВОССТАНОВИТЕЛЬНЫЕ СУКЦЕССИИ В ЛИСТВЕННИЧНЫХ (*Larix sibirica* Ledeb.) ЛЕСАХ ЦЕНТРАЛЬНОГО ХАНГАЯ В МОНГОЛИИ

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Изучены изменения состава и проективного покрытия видов растительного покрова и лесовосстановления после пожаров 1996 и 2002 гг. в псевдотаежных лиственничных лесах в Центральном Хангае Монголии. Описания растительного покрова и учет естественного возобновления проведены в 2007, 2010 и 2012 гг. на постоянных пробных площадях, заложенных на двух участках. Проведенный бестрендовый анализ соответствий (Detrended Correspondence Analysis – DCA) показал, что на 11, 14, 16-летних гарях 1996 г. и 8, 10-летних гарях 2002 г. не произошло существенного изменения видового состава растительных сообществ. Что касается растительного покрова, существенных отличий между покрытиями разных годов не обнаружено, за исключением гари 5-летней давности. Через 10 лет после пожара 2002 г. сформировалось злаково-разнотравное сообщество с подростом лиственницы, а через 16 лет после пожара 1996 г. – разнотравное сообщество с подростом лиственницы. Изучение жизненных форм растений с использованием классификации C. C. Raunkiær (1937) в годы после пожарной сукцессии показало, что жизненные формы, такие как крипто- и гемикриптофиты, сохраняли свое доминирующее положение после пожарного периода. Установлено, что растения этих жизненных форм являются более огнестойкими и имеют высокую способность к самовосстановлению после пожара. После пожаров естественное возобновление происходит успешно, но, например, выпас скота может негативно повлиять на возобновление лиственницы, о чем свидетельствует разная высота крупного подроста на двух пробных площадях. Количество крупного подроста на 1 га было низким, несмотря на длительный период восстановления на пробной площади.

Ключевые слова: *лесной пожар, сукцессия, возобновление, всходы, подрост, псевдотайга, сохранение лесов, бестрендовый анализ соответствий, жизненные формы растений, классификация C. C. Raunkiær, хребет Тарбагатай, Центральный Хангай, Монголия*.

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