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Conservation values in set-aside black alder forests adjacent to managed stands: short-term changes

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Forests dominated by black alder (*Alnus glutinosa)* potentially support habitats with high biological diversity, including rare and endangered species with specific demands for the habitat. However, the knowledge on the response of set-aside black alder forests to adjacent stand management is still insufficient for planning sustainable forest management and requires additional studies on the dynamics of complex organism groups. We conducted repeated inventories of ground vegetation, epiphytic lichens, polypores, and mollusks in the periphery and interior of 10 set-aside black alder-dominated forest stands in Latvia over seven years after adjacent forest management, to determine the response of these organism groups. Our results showed that the diversity of the studied organism groups either remained unchanged or increased from 2004 to 2011. The volume of dead wood increased significantly in all studied plots and correlated positively with polypore abundance and species diversity. We observed an increase in mollusk species number in the stand interior, but not in the periphery plots. No significant spatial differences in forest stand parameters or species diversity were found between stand interior and periphery plots in either the first or the second survey. The obtained results suggested that the 60-meter periphery zone was able to maintain species richness and diversity similar to the interior, highlighting the importance of black alder-dominated forests in supporting species diversity across the studied organism groups.

Keywords: Vegetation, Mollusks, Polypores, Species Diversity, Stand Periphery, Stand Interior

Introduction

Ecosystem fragmentation has a profound ecological impact, including habitat loss, isolation of populations, and altered species interactions (Mullu 2016). It magnifies the negative aspects of the edge effect by altering the microclimate and increasing exposure to invasive species and predators, thereby significantly affecting species diversity and composition, community dynamics, and ecosystem functioning (Chen et al. 1999). In forested ecosystems, human activity such as deforestation and forest management contributes to the increase of edge influence (Harper et al. 2005). As fragmentation is expected to continue and have a great impact on future forest functioning, it is crucial to consider the role of edges for sustainable forest management and biodiversity conservation planning (Vanneste et al. 2024). Attention should especially be focused on the felling areas adjacent to forests of high conservation value to minimize the edge effect into the adjacent forests (Pöpperl & Rupert 2021). Although the entire forest ecosystem has an important role in the forest landscape structure (Vanneste et al. 2024), most studies focus on the forest interior and exclude forest edges to avoid the complex and varying environmental conditions of the ecotone (Pöpperl & Rupert 2021).

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Forests dominated by black alder (*Alnus glutinosa* [L.] Gaertn.) potentially support habitats with high biological diversity (Hrivnák et al. 2024). Part of these stands was considered woodland key habitats (WKH) at the end of the 20th century. The concept was created in the Nordic countries to promote sustainable forest management that conserves biodiversity in production forest landscapes (Timonen et al. 2011), by representing high-quality small habitat patches with voluntary set-aside status due to the presence of rare (specialist) species and habitat structures according to a set of defined criteria (Timonen et al. 2011). Nowadays, some of these set-aside stands are considered European Union (EU) importance habitats in the boreal forest region (Auninš 2013), and black alder-dominated alluvial and swamp forests have especially high importance for conservation objectives (Council Directive 1992). In addition, black alder is an important component of floodplain and swamp forests in Central Europe (Hrivnák et al. 2024). While several studies in the boreal forests conclude that these high-quality set-aside forest stands are a cost-efficient tool to preserve biodiversity values in production forest landscapes (Timonen et al. 2011), at the same time, they are also criticized for their small size, lack of connectivity, and the strong influence of the edge (Hottola & Siitonen 2008). The rich diversity in black alder for-

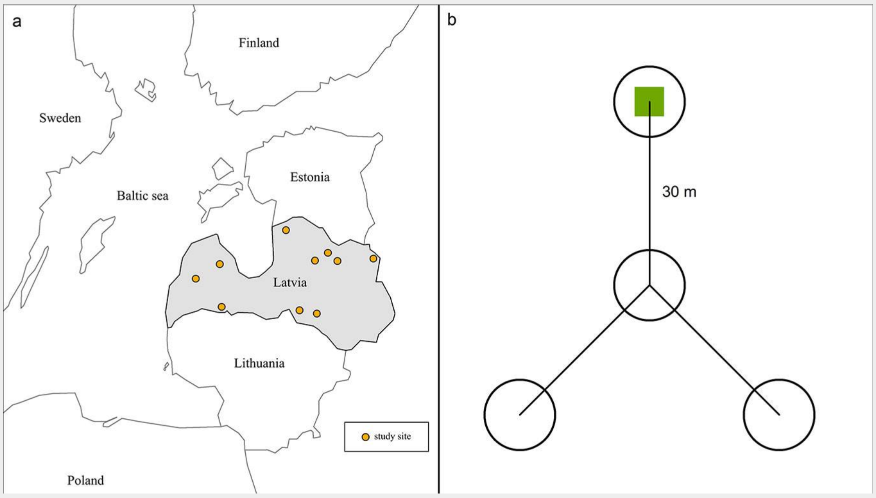

Fig. 1 - Location of the study sites in Latvia (a) and the general layout of a sample plot set (b). Green square: vegetation assessment subplot.

ests is determined by several factors, such as the structural diversity of living trees and dead wood, the mosaic structure with raised hummocks, and the specific hydrological regime that ensures increased air and soil humidity (Prieditis 1997). Live and dead trees provide habitats for vascular plants, lichens, bryophytes, invertebrates, fungi, and other species (Alexander et al. 2006). Tree-related biodiversity aspects have been studied more deeply compared to other organism groups. Previous studies, while highlighting the rich diversity of flora (Prieditis 1997, Douda et al. 2016) and fauna (Pilate 2009) associated with black alder stands, also stress the lack of knowledge on other organism groups. Only a few studies have dealt with the species composition of lichens (Jüriado & Paal 2003) and polypores (Hottola & Siitonen 2008). Since black alder habitats have a very specific microclimate and high diversity of temporal and spatial structure, representing vulnerable and protected habitats throughout Europe, comprehensive studies on the dynamics of complex organism groups in set-aside patches surrounded by managed forest are necessary to determine whether black alder stands can provide hotspots for biodiversity in a managed forest landscape with logging in adjacent stands, and how different species' groups are affected by the forest management.

In this study, we investigated the dynamics of forest structure, ground vegetation, epiphytic lichen, polypore, and mollusk species in the stand interior and periphery of set-aside black alder-dominated forests over seven years after environmental changes caused by different intensity logging activities in adjacent stands. We hypothesized that the measured parameters, such as dead wood volume, epiphytic lichen, polypore, and mollusk diversity, would significantly differ between periphery and interior sample plots.

Tab. 1 - Characteristics of the studied black alder forest stands. Some of the studied stands were considered European Union importance habitats after 2014: 9080* (Fennoscandian deciduous swamp woods), 91D0* (bog woodlands), and 91E0* (alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior*).

Study site	Stand age in 2004	Soil type	Adjacent stand management	Management year	EU importance habitat
Aloja	88	wet mineral soil	Clear-cut	2006	9080*
Benkava	86	wet peat soil	Clear-cut	2005	9080*
Erberge	87	wet peat soil	Clear-cut with transition zone	2005	9080*
Kaive	75	wet peat soil	Selective cutting	2006	9080*
Kurmale	80	wet peat soil	Clear-cut	2004	91D0*
Mežole	70	wet peat soil	Clear-cut	2004	-
Palsmane	85	drained peat soil	Sanitary cutting	2005	91E0*
Talsi	105	wet peat soil	Clear-cut with transition zone	2006	-
Viesite	94	wet peat soil	Clear-cut	2005	-
Ziguri	95	drained peat soil	Clear-cut	2004	-

Material and methods

Study sites

The study was conducted in 10 sites across Latvia (Fig. 1a) in forest stands dominated by black alder (*Alnus glutinosa* Gaertn.), which corresponded or potentially corresponded to WKHs in 2004. The study sites were located near forest stands where different forest management activities, such as clear-cutting, selective cutting and sanitary cutting were carried out from 2004 to 2006; two sites represented clear-cuts with transition zones (Tab. 1). For three study sites the clearfelling of surrounding forests was conducted in 2004 (Tab. 1). Each site was divided into two sections (blocks), *i.e.*, the periphery part and the interior part. The part of the stand further than 60 m from the boundary with the managed forest was considered interior, while the outer stand area (up to 60 m from the boundary) was considered periphery. The width of 60 m was selected considering other studies on edge effect (Murcia 1995).

Field assessment

An initial survey was conducted in 2004 and 2005, and measurements were repeated in 2011 following management activities. Since most of the data was collected in 2004, this year is henceforth referred to as the first assessment year. Two sets of sample plots (one in the stand periphery and the other in the interior section) were established according to a general regular pattern (where possible, considering the compartment configuration). Circular plots with a radius of 12.62 m and an area of 500 m² were placed 30 m apart from each other (Fig. 1b). Sampling of trees, deadwood, and different organism groups was carried out in either the entire set or a subset of the plots as described further.

In each circular sample plot, height, diameter, and species of all living trees exceeding the breast height diameter of 6.0 cm were measured. Additionally, the height and diameter of all ≥ 1 m long standing and lying deadwood pieces with a diameter ≥ 10 cm at the butt end were recorded. For lying deadwood, only the part located within the sample plot was considered.

For ground vegetation inventory, one circular sample plot in each sample plot set (interior and periphery) was selected, subjectively choosing two of the most similar ones. A 20 × 20 m survey subplot was established in each plot, and the percentage of projective cover was recorded for each vascular plant species, according to the nomenclature of vascular plants from Gavrilova & Sulcs (1999).

Epiphytic lichens were recorded in all sample plots of each sample plot set. For the survey, three black alder trees of similar age, located closest to the center of each plot (but from different clusters of trees), were selected. The survey of lichens on the trees was conducted in a clockwise direction (North-East-South-West), using the line method at two heights, *i.e.*, 0.5 m and 1.5 m above the root collar (Donis et al. 2004, Straupe & Donis 2006). The trunks of the trees were girdled with bands where the occurrence of all lichen species touching them was marked with a precision of 0.1 cm in circumference. To ensure accuracy of the repeated measurements, the starting point was marked with a screw in the trunk. Lichen species were identified in the field (where possible) or in the laboratory. The nomenclature for lichens from Piterans (2001) was used.

Mollusk inventory was carried out at the end of summer and in autumn in the vegetation subplots by the sifted litter method and manually. While walking in a zigzag pattern, litter was taken by hand after every second step and sifted into a litter sieve (10 mm). The volume of each sample was 3-5 liters. The samples were then delivered to the laboratory. Three liters of each sample were air-dried and afterward sieved with soil sieves (5 mm, 3 mm, 2.5 mm, 2 mm, 1 mm). The mollusk shells were removed with tweezers, sorted, and identified. Soil species were identified according to Kerney et al. (1983) identification guide. Freshwater species were identified according to the nomenclature by Glöer & Meier-Brook (2003).

Polypore species were inventoried on all measured living trees and deadwood pieces within the sample plots, recording all sporophore bodies and their species. When necessary, the fragments of the sporophore bodies were collected for species identification in the laboratory. Species were identified according to the nomenclature of European polypores (Ryvarden & Gilbertson 1993).

Data analysis

To describe the diversity of the studied organism groups, the Shannon diversity index was calculated based on the mean cover or number of individuals of each species. All statistical analyses were conducted using the R environment (R Core Team 2021). The normality of data distribution was tested with the Shapiro-Wilk test. The parametric *t*-test or the non-parametric Wilcoxon signed-rank test was used to evaluate differences in the taxocoena diversity and forest stand characteristics between the stand periphery and interior in the first and second surveys. To assess the differences in species diversity and forest stand characteristics over time in both stand periphery and interior, the same tests for paired samples (paired=TRUE) were used. To assess the interaction effect between location (periphery, interior) and time, linear mixed-effects models (lmer) and generalized linear mixed-effects models (glmer) were applied, with site identity included as a random factor to account for spatial dependency. For all analyses, a significance level α = 0.05 was used. Bonferroni correction was applied to adjust p-values for multiple comparisons.

Correlations between the Shannon diversity index of polypores, the abundance and number of polypore indicator species, and dead wood volume were tested using Spearman's rank correlation test [function cor.test()]. The relationships were considered significant when p-values < 0.05.

The representation of species' abundance was specific to each recorded organism group. For vascular plant species, it was represented as species cover. For lichens, the projective coverage of each species was obtained by summarizing the line occupied by each lichen species in cm, dividing that by the perimeter of the trunk, and then multiplying it by 100. Then the data obtained on the sample trees were summarised and divided by the number of studied trees. Consequently, the average projective coverage for each lichen species in the sample plot was obtained (Straupe & Donis 2006). The abundance of mollusk and polypore species was related to the number of individuals per species.

In our study, we used indicator species of woodland key habitats that reflect forest longevity and structures important for biological diversity (Ek et al. 2002). In Latvia, all these species are currently considered indicator species of forest biotopes of EU importance (Auninš 2013).

Results

Short-term changes

Forest stand

In total, 13 tree species were recorded in the canopy layer of the studied sites (*Alnus incana, Frangula alnus, Fraxinus excelsior, Pinus sylvestris, Populus tremula, Quercus robur, Salix caprea, Sorbus aucuparia, Tilia cordata, Ulmus glabra*), with *Alnus glutinosa, Picea abies,* and *Betula* spp. dominating both the periphery and the interior area (Fig. 2). The average stand basal area of trees in the stand periphery sites had decreased significantly from 37.32 m^2 ha^{-1} in 2004 to 35.63 m^2 ha^{-1} in 2011 (p-value = 0.0207), but when individual tree species were compared, only *Betula* spp. basal area showed a significant decrease in the stand periphery plots, from 8.13 m^2 ha^{-1} in 2004 to 7.52 m^2 ha^{-1} in 2011 (p-value = 0.0144). In the stand interior plots no significant difference between the average stand basal area in 2004 (37.08 m^2 ha^{-1}) and 2011 (37.42 m^2 ha^{-1}) was found (p-value = 0.5258); however, two tree species showed a significant change: the average basal area of *Alnus glutinosa* had increased from 18.41 m^2 ha^{-1} in 2004 to 19.04 m^2 ha^{-1} in 2011 (p-value = 0.0085), while the average basal area of *Fraxinus excelsior* had decreased from 0.48 to 0.1 m^2 ha^{-1} (p-value = 0.0129).

The volume of dead wood had increased significantly from 2004 to 2011 in both the stand periphery and the interior sites (see Fig. S1 in Supplementary material). The average dead wood volume in the stand periphery plots increased from 41.97 m^3 ha^{-1} in 2004 to 68.47 m^3 ha^{-1} in 2011 (p-value = 0.0002), but in the stand interior sites it increased from 42.07 to 74.53 m^3 ha^{-1} (p-value = 0.003).

Ground vegetation

In total, 125 vascular plant species in the herbaceous layer of the studied sites were recorded in 2004 and 2011 (see Tab. S1 in Supplementary material). The most common vascular plant species (found in at least 70% of all study plots) were *Filipendula ulmaria, Lysimachia vulgaris, Cardamine pratensis, Athyrium filix-femina, Dryopteris carthusiana,* and *Oxalis acetosella*. On average, 28.3 species were found in the stand periphery plots in 2004 and 28.9 species in 2011, while in the stand interior plots, the average species number was 27.7

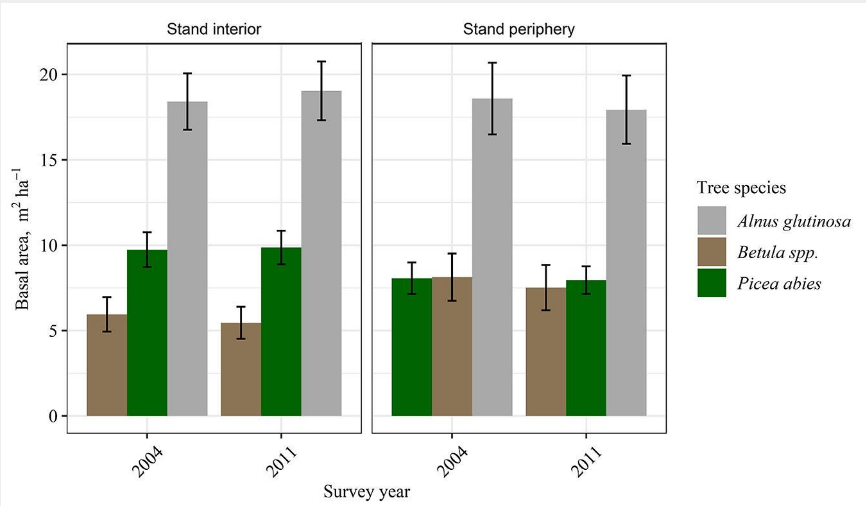

Fig. 2 - Average basal area of *Alnus incana, Betula* spp., and *Picea abies* (m^2 ha^{-1}) in the stand interior and periphery study sites in 2004 and 2011. Error bars represent standard error.

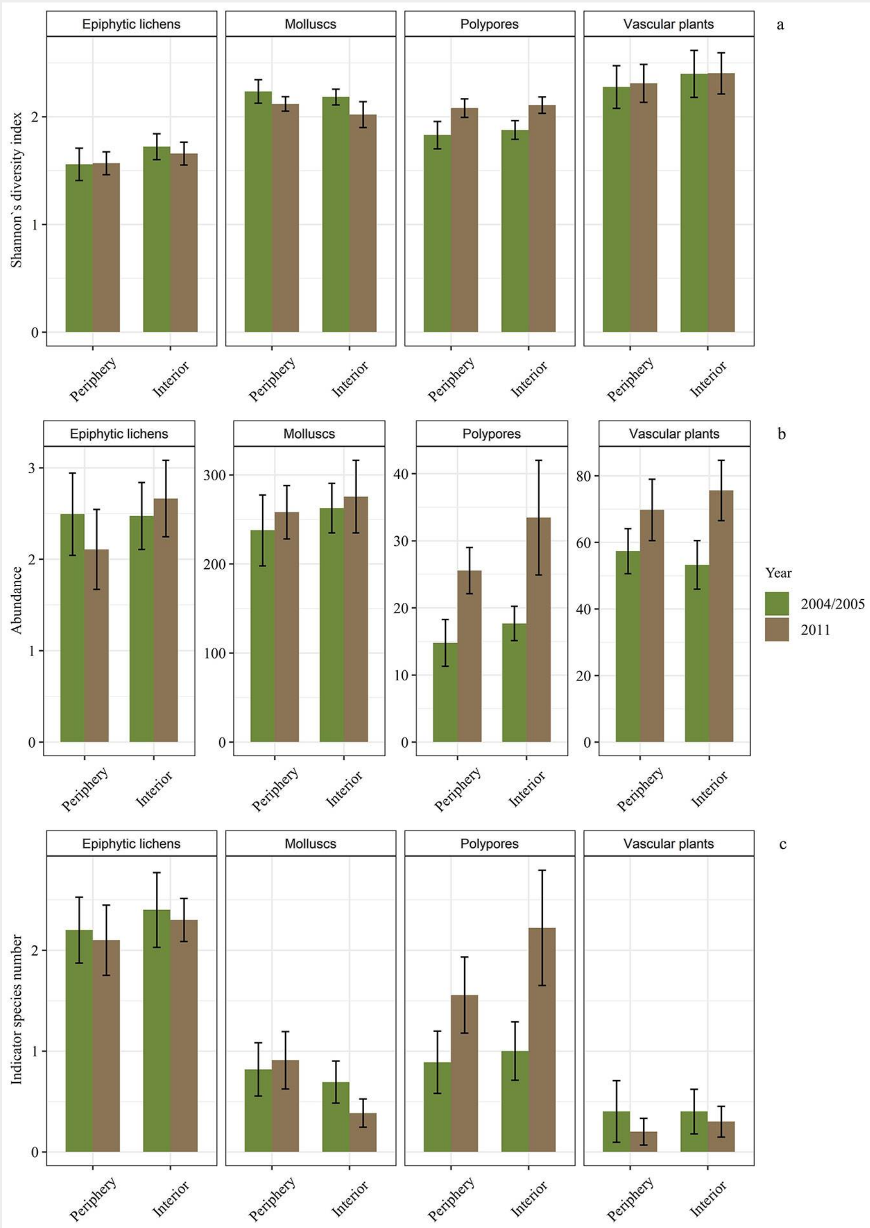

Fig. 3 - The average value of Shannon diversity index (a), abundance (b), and indicator species number (c) of epiphytic lichens, mollusks, polypores, and vascular plants in the stand interior and periphery plots in the first and second inventory. Error bars represent standard error.

and 30.8 in 2024 and 2011, respectively. During the study period, the abundance (Fig. 3b) had increased significantly in both the stand periphery and the interior plots (p-value = 0.0349 and 0.0099, respectively), while the average number of species and the value of the Shannon diversity index did not change significantly (p-values > 0.05). Among the inventories, the abundance of 42 species had increased in the periphery plots. The highest increase was observed for species *Oxalis acetosella* and *Rubus idaeus.* Comparing the data from both inventories in the interior plots, the total abundance had increased for 56 species, mainly influenced by the increase in species such as *Lysimachia vulgaris, Aegopodium podagraria, Carex caespitosa, Filipendula ulmaria, Vaccinium myrtillus,* and *Rubus saxatilis*. The species *Thelypteris palustris* and *Galeobdolon luteum* showed a high increase in both studied site plots (see Tab. S1 in Supplementary material).

From 2004 to 2011, three vascular plant indicator species were recorded in the herbaceous layer of the studied sites (see Tab. S1 in Supplementary material). Indicator species constituted 2.4% of all recorded vascular plant species in the herbaceous layer. No significant changes in the average number (Fig. 3c) or abundance of indicator species in the stand periphery and the interior plots from 2004 to 2011 were found (p-values > 0.05). Two indicator species were not found in the second assessment: *Listeria ovata* and *Sanicula europaea*. The abundance of *Carex remota* had decreased in the stand periphery plots, while in the stand interior plots it had increased.

Epiphytic lichens

In total, 24 epiphytic lichen taxa were recorded in the study sites in both inventories (see Tab. S1 in Supplementary material). On average, 6.2 taxa were found in the stand periphery plots in 2004 and 5.6 species in 2011, while in the interior plots the average species number was 6.8 and 6.2, respectively. Six epiphytic lichen indicator species were recorded in the studied sites (see Tab. S1). Indicator species constituted 29% of all recorded epiphytic lichen species. The most common epiphytic lichen taxa (found in at least 70% of all study plots) were *Cladonia coniocraea, Lepraria* spp., and two lichen indicator species *Arthonia spadicea* and *Graphis scripta*. The average number of species and the value of the Shannon diversity index (Fig. 3a) in the stand periphery and interior plots did not change significantly either for all species or indicator species (p-values > 0.05).

The abundance (Fig. 3b) did not change significantly either in the stand periphery or the interior plots (p>0.05). However, comparing the initial and repeated inventory data obtained in periphery sample plots, it was found that the total projective cover of epiphytic lichens showed a tendency to decrease, mainly influenced by the projective cover decrease of *Pertusaria amara* and *Parmelia sulcata*. In contrast, the projective cover of seven other species had decreased slightly, and five epiphytic lichen species disappeared. *Opegrapha varia* was recorded only in the repeated inventory. For some species, an increase in the projective cover was observed, most notably for *Lecanactis abietina*.

Comparing the data of both inventories from the interior plots, it was found that the total projective cover of epiphytic lichens increased over time. It was mainly influenced by the increase of *Cladonia coniocraea* and *Lepraria spp.*, while for six other species the projective cover had increased slightly (see Tab. S1 in Supplementary material). For nine species a decrease of the projective cover was observed, and eight species were not recorded in the repeated inventory.

Polypores

In total, 56 polypore species were recorded in the study sites in 2004 and 2011 (Tab. S1 in Supplementary material). The most common polypore species (found in at least 70% of all study plots) were *Fomes fomentarius, Fomitopsis pinicola, Inonotus radiatus,* and *Trichaptum abietinum.* On average, 7.8 species were found in the stand periphery plots in 2004 and 11.2 species in 2011, while in the interior plots the average number of species was 8.4 and 11.8 in 2004 and 2011, respectively. From 2004 to 2011, the average number of species in the stand periphery and interior plots increased significantly (p-value = 0.0101 and 0.0167), as well as the abundance of polypores (p-value = 0.0024 and 0.0142 – Fig. 3b). However, the average value of the Shannon diversity index (Fig. 3a) did not increase significantly either in the stand periphery or the interior (p-values > 0.05).

In total, 10 polypore indicator species were recorded in the studied sites from 2004 to 2011 (Tab. S1), six of them considered biotope specialist species (Auninš 2013). Indicator species constituted 18.5% of all recorded polypore species. From 2004 to 2011, the average number of indicator species (Fig. 3c) and their abundance in the stand periphery and the interior plots increased significantly (p-value = 0.0494 and 0.0302).

The Shannon diversity index of polypores had a moderately strong and positive correlation (r_s = 0.4259, p-value = 0.0077) with the dead wood volume. The abundance and number of polypore indicator species also showed a positive correlation with the volume of dead wood (r_s = 0.366, p-value = 0.0238; r_s = 0.4519, p-value = 0.0044).

Mollusks

In total, 40 mollusk species were recorded in the study sites during the first and second inventory (Tab. S1 in Supplementary material). The most common mollusk species (found in at least 70% of all study plots) were *Carychium minimum, Carychium tridentatum, Cochlicopa lubrica, Euconulus fulvus, Vertigo substriata, Zonitoides nitidus, Punctum pygmaeum, Perforatella bidentata*, and *Nesovitrea petronella*. On average, 16.6 species were found in the stand periphery plots in 2004 and 16.5 species in 2011, while in the interior plots, the average number of recorded species was 15.2 and 17.1 in 2004 and 2011, respectively. The average number of species in the interior plots increased significantly during the study period (p-value = 0.0101 and 0.0167), but in the stand periphery, no significant change was detected (p-value = 0.948). From 2004 to 2011, the value of the Shannon diversity index (Fig. 3a) and the abundance (Fig. 3b) in the stand periphery and the interior plots did not change significantly (p-values > 0.05).

In total, six mollusk indicator species were recorded in the studied sites (Tab. S1). Indicator species constituted 10.7% of all recorded mollusk species. No significant changes in the average number (Fig. 3c) or abundance of indicator species were found during the study period, either in the stand periphery or the interior plots (p-values > 0.05).

Spatial differences

There were no statistically significant differences in abundance, average species number, or the Shannon diversity index between the studied organism groups when comparing the stand periphery and interior plots in both inventories. The number and abundance of indicator species, along with the basal area of living trees and the volume of dead wood, did not vary significantly between the stand periphery and the interior in both assessments (p-values > 0.05).

Linear mixed-effects models

No significant interaction between location and time was detected for any of the studied response variables (p > 0.05). Among the studied variables, only the abundance of vascular plants, polypores, and mollusks was significantly affected by time. Their abundance was significantly higher in 2011 than in 2004/2005 (p= 0.0017; p < 0.0001; p = 0.0005). Additionally, polypore abundance was significantly higher in the interior compared to the periphery (p = 0.0007).

Discussion

When studying the impact of adjacent logging on the structure and composition of stands with (potentially) high conservation value, a particular emphasis should be placed on the edge effect, which can cause significant changes in biotic and abiotic conditions (Harper et al. 2005). By conducting repeated inventories of several organism groups occupying different ecological niches, we assessed black alder swamp forest species diversity and composition to understand the potential influence of nearby forest management and to observe short-term changes over a seven-year period.

Short-term changes

Dead wood

Over the period between measurements, the amount of dead wood had significantly increased in the studied plots. While in both surveys the amount of deadwood was high compared with countries such as Sweden (Jönsson & Jonsson 2007), similar quantities of coarse wood debris have been reported in deciduous forests of the Baltic region (Lõhmus & Kraut 2010, Madzule et al. 2012). Additionally, the increase in total volume of downed logs can partly be explained by the influence of the windstorm Erwin (Gudrun) in 2005 (Donis et al. 2005). The amount of dead wood in the forest is seen as a proxy for management intensity (Green & Peterken 1997) because many forest-dwelling species are associated with large-dimension substrates, and therefore, nature conservation aims to maintain these quantities. More generally, for biodiversity aspects, it is important to estimate the dynamics of dead wood decay to predict substrate continuity in the longer term, especially for species dependent on this specific substrate. The decomposition rate for some species might take decades (*Picea abies* or *Pinus sylvestris*), and for deciduous trees, it takes 10-20 years to reach the most advanced decay stage. Importantly, the presence of different organisms such as epixylic species, fungi, invertebrates, reptiles, and mammals strongly correlates with substrate decomposition rate, dimensions, spatial distribution, and tree species richness (Stokland et al. 2012). Our study further indicates that quantities of deadwood are increasing, and additional factors such as natural disturbances (abiotic factors), gap dynamics, and tree mortality (due to the competitiveness and overmature age) might provide favorable conditions for species dependent on dead wood substrate in the black alder forests.

Vascular plants

Previous studies have shown that management could favor those vascular plant species that are early-successional colonizers and mostly reach the peak in the first decade after disturbance (Dovčiak & Brown 2014). It increases plant species richness near the forest edge (Pöpperl & Rupert 2021), primarily due to higher light intensity, which is the main factor influencing plant distribution (Esseen et al. 2016). The studies on the impact of management on understorey plant species richness show inconsistent results, including some that suggest no effect at all (Duguid & Ashton 2013, Dovčiak & Brown 2014). These findings are in accordance with our results – the species richness did not respond to management in the surrounding stands, at least in the 60-meter zone from the disturbance during the seven years. However, we found that plant species abundance had increased significantly in both the periphery and the interior study plots. The vegetation cover is a potentially good indicator for management and for gap sizes, indicating changes in canopy cover (Oettel & Lapin 2021). As the vegetation cover had increased in both the interior and periphery plots of our sites, it could not be attributed solely to management impact, as there was no significant interaction between site location and survey year. However, this is probably explained by the different niche availability and general species competitiveness. Also, some abiotic factors, particularly meteorological conditions in the inventory year, must be considered. For instance, low precipitation and high temperatures cause desiccation of the depression patches, subsequently enabling vegetation development there.

The natural dynamics of the tree stand also play a role. As the projective crown cover decreases over time, better light and substrate availability lead to an increase in projective cover for generalist species. Additionally, factors such as resource availability and heterogeneity, as well as the degree of harvest intensity, site conditions, and successional processes, may define understorey plant diversity more than the management-induced disturbance itself (Duguid & Ashton 2013). In summary, our sampling scheme did not allow for the detection of a clear response of vascular plant species to the edge effect because a

60-meter zone is likely too wide to represent the immediate response to surrounding management. As demonstrated before, these differences are usually reflected at smaller spatial scales (Eldegard et al. 2015, Pöpperl & Rupert 2021). For herbaceous plant species, the edge is rather a transition zone classified as ecoline, and its effect on species richness and diversity could be weak (Vanneste et al. 2024). In addition, the edge effect on plant diversity and richness tends to be greater on the open side than on the forest side of the border (Eldegard et al. 2015). It must also be noted that our inventories did not cover the initial response to the surrounding forest management. Previous studies suggested that for vascular plants, the most profound changes in species number occur during the first part of the decade (Dovčiak & Brown 2014).

Lichens

Contrary to some other studies (Aragón et al. 2015, Koelemeijer et al. 2022), our data on the changes in the epiphytic lichen diversity did not indicate any influence of forest management. In general, the change in lichen species could be explained by several environmental factors that over time are influenced by logging activities in the surrounding forest stands (Esseen & Renhorn 1998), as well as conditions inside the studied stands, such as air humidity (Király et al. 2013), light conditions (Odor et al. 2013). Although no statistically significant differences were observed, there was a tendency for lichen species richness to decrease in both studied sample sites. Still, the total cover of epiphytic lichen taxa tended to increase in the interior plots. Such differences could be explained by the replacement of epiphytic species between the edge and forest interior when light-demanding species are more common near the forest edge and shade-tolerant species' abundance increases toward the forest interior (Nascimbene et al. 2013, Aragón et al. 2015).

According to our results, successional changes between species are still ongoing, especially since the observed trends were similar in both periphery and interior sample plots. The decrease of cover occurred for six species predominantly associated with shaded environments (ecological indicator value of light ≤ 3) in the periphery stands, including four indicator species such as *Arthonia leucopellea, Arthonia vinosa, Arthonia spadicea*, and *Opegrapha vermicellifera* (Tab. S1). Only two of these species showed a decrease in the stand interior plots, and the other indicators increased. The abundance of the more light-demanding lichen species (ecological value of light ≥ 7) also decreased in the second survey, except for two lichens *Hypogymnia physodes* and *Melanelixia glabra* found only in plots closer to the forest edge. The forest stand periphery may create a more suitable environment for indicator species and species that prefer more constant conditions (Nascimbene et al. 2013). This may also be explained by natural disturbances. For example, the significant increase in the deadwood amount between the assessments may have created conditions that could potentially change microclimate patterns (Chen et al. 1999), and, consequently, may be affecting the epiphytic lichen species composition both in the stand interior and the periphery plots. Our results were similar to those of Sitzia et al. (2017), indicating that significant environmental changes caused by management could not be recorded in the lichen species richness over such a short period, though differences in species composition were observable.

Polypores

In this study, polypores were the only organism group with a significant increase in both the abundance and the species diversity over the study period. This is likely explained by the significant increase in the dead wood volume from 2004 to 2011, as polypores are directly dependent on suitable dead wood as substrate (Peltoniemi et al. 2013). Previous research demonstrated that polypore diversity increases with dead wood volume (Ylisirniö et al. 2016). In our study, the Shannon diversity index of polypores, along with the abundance and number of polypore indicator species, showed a significant correlation with dead wood volume. This highlights the importance of dead wood as a structural component of black alder forest habitat.

However, it should be noted that many polypore species can survive on dead wood over the long term (Sippola & Renvall 1999). Ylisirniö et al. (2016) suggested that log-level microclimate measurements and long-term monitoring of polypore species are needed to better understand the impact of microclimate changes on species. The effect of microclimate changes resulting from adjacent stand management may be delayed, as the moisture content of dead wood changes gradually. Therefore, we cannot exclude the possibility of observing some changes in future surveys.

Two of ten recorded polypore indicator species, *Junghuhnia collabens* and *Phellinus populicola*, were not detected in the second survey in the stand periphery plots. In contrast, the occurrence of other indicator species in both relative locations either increased or remained the same. It should be emphasized that the occurrence and abundance of the two mentioned species were already low in 2004; therefore, we cannot rule out the possibility of a natural decline due to competition, as the overall number of polypore species increased significantly. Besides, *Junghuhnia collabens* is an annual polypore: climatic conditions are important for initiating fruiting and consequently for the detection of this species (Boddy et al. 2014).

Mollusks

In our study, the increase in mollusk species number in the stand interior, but not in the periphery plots, possibly indicates slightly more stable conditions in the interior part of the stand, likely related to soil moisture level. Although the differences in species number in the stand periphery were not significant, a clear-cut area directly adjacent to the black alder forest can affect the microclimate at the forest stand edge (Boeraeve et al. 2019). According to Hylander et al. (2004), in forests under intensive management, moist areas of various sizes and types, including buffer zones around clear-cuts, are particularly important for conserving snail diversity and abundance. Hylander et al. (2004) also emphasized the importance of the buffer zone width, concluding that a 10-meter-wide buffer zone does not provide an optimal microclimate for snails due to wind influence.

Boeraeve et al. (2019) demonstrated that soil moisture content in black alder forests increases with distance from the forest edge. Mollusks are suggested as suitable indicators of hydrological conditions (Cejka & Hamerlik 2009), due to the positive relationships between snail species diversity and soil moisture. It should be noted that in our study, the indicator species are mollusk species of moderately moist forests (Rudzite et al. 2010), where they are found more frequently than in excessively wet forests (Pilate 2009). Therefore, the lack of changes in the abundance and number of mollusk indicator species could also indicate the absence of changes in the hydrological regime. However, not only indicator species, but also changes in the general snail species diversity indicate changes in the environmental conditions (Horáčková et al. 2014). In the study of land snails as indicators of soil humidity, Cejka & Hamerlik (2009) concluded that humidity significantly impacts snail species richness and abundance, and changes in the structure of snail communities can reflect moisture variations at sites.

The abundance of dead wood may also positively impact terrestrial mollusk diversity by retaining moisture and providing shelter and food resources (Gheoca et al. 2021). Similar to the pattern observed for polypores, our results show a short-term increase in the mollusk abundance. We assume that an increase in the dead wood volume could be one of the factors positively influencing mollusk abundance in both the stand periphery and interior.

Spatial changes

Our results did not show any significant spatial differences in the forest stand parameters or species diversity between the stand periphery plots, which characterized the zones up to 60 m from the managed adjacent stand, and the stand interior plots located further than 60 m from the edge of the adjacent stand. However, instead of showing the absence of logging impact, our results suggest a similarity in the black alder stands assessed during both surveys in terms of species diversity, as well as a similar response to nearby forest management. The effect of the adjacent forest stand logging may be more localized, possibly affecting only the immediate edge area. Such changes would not manifest themselves in our study design.

Conclusions

Forests dominated by black alder are part of forested wetlands, recognized as vital biodiverse ecosystems. The biodiversity of many forested landscapes can largely depend on the prevalence of wet forest stands within them. Thus, the small set-aside forest patches in managed forest landscapes could be important biodiversity hotspots and even considered as EU importance habitats. Our research showed that black alder forests may preserve the richness of different organism groups, supporting a broad range of biodiversity indicator species such as snails, lichens, mollusks, and vascular plants. However, the adjacent management could impact the environmental conditions, which in turn could affect species associated with specific hydrological conditions in such forests. Despite potential influences from adjacent management practices, both species diversity and abundance have remained relatively stable or even increased, showing that the 60-meter periphery zone can maintain species richness and diversity similar to the interior. However, rather than implying a lack of impact from logging, our results indicate stability in the studied forest ecosystems in terms of species diversity, highlighting the importance of deadwood, especially in the conservation of polypores. At the same time, our study has some limitations. Firstly, different organism groups may require varying time frames to respond to environmental changes, whereas our research only showed short-term changes in species diversity. It would be valuable to monitor the impact of management on species diversity in the long term. In addition, it is important to understand how these forests contribute to maintaining species diversity and their distribution to adjacent stands in a managed forest landscape. Secondly, we have a limited number of studied stands with diverse management intensity in adjacent stands. Nevertheless, our results in the studied sites highlight the importance of black alder forests in supporting the diversity of different organism groups, which is significant in a broader European context.

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Supplementary Material

Tab. S1 - Occurrence and abundance of all recorded taxa in the stand interior and periphery plots in 2004 and 2011.

Fig. S1 - The volume of dead wood (m^3) in the study sites in 2004 and 2011.

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