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Landscape Diversity for Reduced Risk of Insect Damage: A Case Study of Spruce Bud Scale in Latvia

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Abstract: Spruce bud scale (*Physokermes piceae* (Schrnk.)) has gained attention due to recent outbreaks in the eastern Baltic Sea region—Poland, Lithuania, and Latvia. In the spring of 2010, it spread rapidly across Latvia, affecting large areas of Norway spruce stands. Therefore, the aim of our study was to assess the effects of landscape heterogeneity on the damage caused by spruce bud scale in Norway spruce stands. In this study, we evaluated landscape metrics for middle-aged (40 to 70 years old) Norway spruce-dominated stands (>70% of stand's basal area) in four of the most affected forest massifs and two unaffected forest massifs. We used a binary logistic generalized linear mixed effects model (GLMMs) to assess the effect of environmental factors on the abundance of the spruce bud scale. Our results show that increased local diversity within 100 m of a forest patch apparently reduced the probability of spruce bud scale presence. We also found that the diversity within 1000 m of a patch was associated with an increased probability of spruce bud scale damage. A quantitative analysis of landscape metrics in our study indicated that greater landscape-scale diversity of stands may reduce insect damages.

Keywords: landscape diversity; spatial scale; scale insects; forest structure

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1. Introduction

Norway spruce (*Picea abies* (L.) Karst.) is highly susceptible to both abiotic and biotic disturbances such as windthrow, drought and outbreaks of pests due to tree architecture [1–3]. Norway spruce in the hemiboreal and boreal forests could potentially lead to difficulties with growth and particularly regeneration under most scenarios of predicted climate changes [2,4,5]. Consecutive drought events in combination with high temperatures in Norway spruce stands across Europe (from Scandinavia to Western Europe) can affect tree water relations and hence resistance to pests [2,3,6]. In recent decades, the intensity and spatial extent of pest outbreaks in forests throughout the Northern Hemisphere have already increased due to climate change impacts [7].

Previous studies in the hemi-boreal and boreal zones have focused primarily on the damage to Norway spruce caused by spruce bark beetles [8,9], whereas little is known about other insect pests, e.g., bud scale [10,11]. The effects of bud scale have mostly been limited. However, scale insects are known as one of the most successful invaders of new environments [12], and the historical distribution of this species has been rather limited to the north (e.g., a range limit of 48–50° N latitude, [13]). Severe damage to Norway spruce stands by scale insects has previously been reported from Lithuania [14] and eastern Poland [15].

Spruce bud scale (abbreviated as SBS) was quite rare in the hemiboreal zone and the propagation and infestation of this pest has caused problems in this climate zone, primarily in urban areas, e.g., parks and gardens [16,17] until an outbreak occurred in 2010 [11,18]. However, beginning in the spring of 2010, the dispersal of *Physokermes piceae* (Schrnk.) across Latvia [19,20], Lithuania [21,22], and southern Sweden [23] as well as an unspecified *Physokermes* sp. in Germany and Poland [10] and *Physokermes inopinatus* in Sweden [11,18] has resulted in a rapid spread across the hemiboreal zone. Furthermore, considering predicted climate changes, this northward shift of scale insects could potentially accelerate and lead to even more frequent and severe damages, such as major outbreaks [24].

The distribution of pests is influenced by landscape heterogeneity [25]. The spread of pests may be enhanced or retarded by heterogeneity across multiple spatial scales [26,27]. Moreover, pest distribution is affected by individual responses to variation in spatial extent within their environment [28]. This is often reflected in forest structure patterns at the local and landscape scales [29]. Therefore, understanding the patterns of a species' movement at different spatial scales may clarify the drivers of species diversity [30].

The relationship between landscape structure and species distribution is determined by a number of factors, including landscape characteristics, habitat diversity (e.g., configuration and composition), and predator-prey interactions [4,31]. These are major factors that may influence the ability of a species to disperse as well as to survive, reproduce successfully, or establish a population [31]. These factors can be studied in terms of forests stands and their characteristics, such as landscape metrics [32,33].

Landscape structure plays an important role in the risk of forest damage from insects and disease [26,34]. The effect of environmental complexity (spatial heterogeneity) affects both the spread of disturbances over the landscape [26] and resilience at the landscape scale [35]. Various reports have indicated that with insect pests, more serious problems arise in areas with a homogeneous landscape spatial pattern e.g., from silvicultural practices, especially forest even-aged monocultures and pure coniferous stands, especially in areas with unsuitable site conditions [36,37]. Candau and Fleming [38], describing large–scale patterns in defoliation frequency, found that the most important forest characteristics (e.g., stands age, basal area) were reflected in species composition.

The integration of landscape ecology into resource management by quantifying it in terms of landscape metrics has not been easy [24]. Although, the effects of landscape diversity on insect outbreaks in Norway spruce stands have previously been studied extensively in the temperate and boreal climate zones, there is a lack of studies of spatial diversity in the prevailing conditions of the hemiboreal zone [2]. Therefore, the aim of this study was to examine the influence of forest landscape diversity on SBS infested Norway spruce stands following the 2010 outbreak in the hemiboreal forests of Latvia. We hypothesize that insect pest damages will be lower in patches with higher forest structural diversity.

2. Materials and Methods

2.1. Study Area

The study was conducted in forest massifs that were disturbed by spruce bud scale (*Physokermes piceae* (Schrnk.)) infestation in 2010. The study area is located in Latvia between 56°4′ N and 56°28′ N and 25°19′ E and 26°41′ E in the hemiboreal forest zone (Figure 1). The landscape in this region was shaped by traditional agriculture (farmland, abandoned farmland), forests, meadows, peatlands, lakes, rivers and localities that led to a complex mosaic with almost 40% forested area. Forest trees comprise pure evergreen (Scots pine (*Pinus sylvestris)*, Norway spruce (*Picea abies* L. Karst.)) and deciduous (birch (*Betula* spp.), aspen (*Populus tremula* L.), black alder (*Alnus glutinosa*)) stands, as well as mixed stands. Norway spruce represents 13.1% of forest cover.

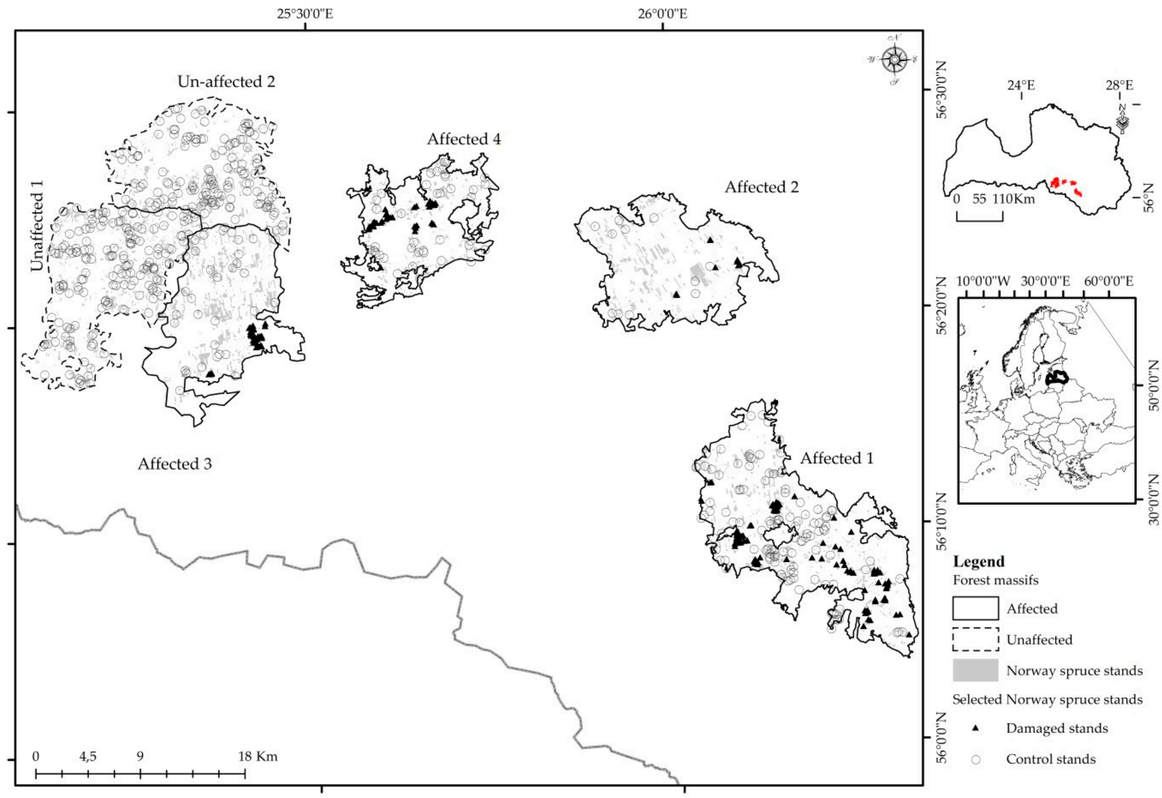

Figure 1. Locations of selected forest massifs in Latvia and spatial distribution of *P. piceae* (Schrnk.) affected stands.

The climate is moist with moderate winters. The mean annual precipitation is 713 mm (min 533 mm to max 854 mm), and the annual mean temperature is 5.9 °C, with monthly mean temperatures in May and June over a 30-year period of 11.74 °C and 15.3 °C, respectively. The active vegetation period (daily mean air temperature > 5 °C) normally lasts for 192 to 200 days [39]. The topography in the study area is relatively flat with an elevation of 160 m above sea level. Stagnic luvisols, gleyic luvisols, hapalic podzols and eutirc gleysols are the prevailing (86.5%) soil types [40].

2.2. Data Source

Study sites were located within six forest massifs, including four affected by the 2010 SBS outbreak (Affected 1–Affected 4) and two control forest massifs (Unaffected 1 and Unaffected 2) where no damages were registered (Figure 1). To provide a systematic and uniform approach, the largest forest massif was split into smaller sub-massifs (similar by size), Affected 3, Unaffected 1 and Unaffected 2, according to natural borders such as highways, rivers or bogse, based on information that spruce bud scale migrates over short distances (from tree to tree, by contact of tree crowns, and in some cases by wind [41]).

We obtained spatial data (polygon shapefile) from the Latvia State Forest Service on the stands felled in sanitary clear-cuts due to the outbreak of SBS in 2010. Additionally, for each polygon within the studied forest massifs, we obtained the available digitized inventory data up to 2010. Next, forest stands were grouped into 16 forest land cover/use categories based on forest inventory data of 2010 (Table 1).

Table 1. List of the forest land cover/use classification categories.

Defined Forest Land Cover/Use Categories	Description
Damaged stands	Norway spruce stands where sanitary clear-cut occurred after SBS infestation in 2010
Scots pine pure stands [1]	Scots pine pure stands
Scots pine mixed woodland [2]	Scots pine mixed with Norway spruce, birch or aspen
Norway spruce control stands [3]	Norway spruce pure stands from 40 to 70 years in age
Norway spruce mixed woodland	Norway spruce mixed with scots pine, birch, black alder, grey alder, aspen or ash
Norway spruce pure stands	Norway spruce pure stands younger than 40 years or older than 70 years
Non-forest land	Agricultural land, farmland, the buffer area outside the boundaries of the forest massif
Soft broadleaves pure stands	Pure birch, black alder, grey alder or aspen stands
Soft broadleaves mixed woodland	Mixed birch, black alder, grey alder or aspen stands
Hard broadleaves stands	Common oak, ash, wych elm, linden or maple stands
Other damages	Any species stands damaged by windthrows, snow damages, other insect outbreaks or browsing damages
Infrastructure	Infrastructure such as roads, forest roads, power lines, buildings or crossrides
Glade	A forest opening of less than 0.2 ha
Wetlands	Includes mangroves, rivers, lakes, ditches or seasonally inundated areas
Clear-cut	Any species stands felled in a clear-cut
Bogs	Low, transition or high swamps

[1] pure stand- the proportion of the dominant species in pure stands was greater than 70% of overstory basal area, [2] mixed stands are a combination of two species where the proportion of one species is not greater than 70% of overstory basal area. [3] the category Norway spruce "*Target stands*" contains those stands that were subsampled for detailed comparison analysis.

Additionally, to deal with edge effect, we generated a buffer with a distance of 1000 m (as a greatest buffer size around patches) outside of the forest massifs to avoid bias that may affect results, as we were missing data on the land cover/use categories outside of the forest massifs. The generated buffer area was added to the vector layer of forest stands with the given land cover/use category defined as "non-forest area".

The damaged stands within the affected forest massifs were subsampled using three main criteria. We selected only those stands where Norway spruce was more than 70% of basal area, the stand age ranged between 40 and 70 years, and a sanitary clear-cut was performed. We analyzed 194 damaged pure Norway spruce stands that met the selection criteria with a total area of 460.3 ha. The area of individual stands ranged from 0.09 to 13.5 ha. To eliminate sampling biases, we used the Sampling Design Tool for ArcGIS 10 to create a subset of undamaged Norway spruce stands (called "control stands") within the affected and unaffected forest massifs that were similar in composition and structure to the damaged stands [42,43]. A minimum distance of 500 m was set between damaged stands and control stands to avoid biased results due to possibly incomplete observation of insect damages in 2010. In total, 194 control stands within affected and 300 control stands within unaffected forest massifs, were randomly selected.

2.3. Data Processing

Landscape metrics were calculated according to the categories described below using the Patch Analyst extension in ArcGIS 10.2 [44]. The landscape metrics were selected based on previous research and their presumed ecological significance because of high multicollinearity with each other [45,46]. In this study, we selected 12 landscape metrics for class-level analysis: Class percentage of landscape—PLAND (%); mean patch size—MPS (ha); core area—CA (ha); core area index—CAI (percentage of the patch more than 20 m from patch edge); mean shape index—MSI (when MSI = 1, the shape of the patch is compact, such as a perfect circle or square); mean perimeter area ratio—MPAR; mean patch edge—MPE (m); edge density—ED (m ha^{-1}); patch density—PD (number of patches per 100 ha); mean fractal dimensions—FRAC MN; interspersion and juxtaposition index—IJI; and mean contiguity index—CONTIG (for a detailed description, see [30]).

To assess the effect of local diversity, initially we tested the effect of abundance of each of land cover/use group. Furthermore, the Shannon diversity index (H) was calculated for each of the damaged stands and control stands (polygons) based on the proportion of land cover/use categories within the outer buffer areas of 100, 250, 500 and 1000 m (Figure 2). To highlight landscape heterogeneity, the land cover/use categories were split by forest type, where applicable. The calculation was performed in two steps. First, to calculate the proportions of the land cover/use category areas, the intersection between the forest stands (class data) and the buffer zones (zone data) was performed using the "*Tabulate intersection*" geoprocessing tool in ArcGIS 10.2.1 (Figure 2). Second, to quantify H for each polygon of the damaged stands and control stands, we used the "diversity" function in the "*vegan*" library in R [47]. H is defined by $= -\sum_{i=1}^{S} p_i log_b p_i$, where p_i is the proportion of the land cover/use category area of the patch within the area of the patch buffer, S is the count of land cover/use categories and b is the base of the logarithm.

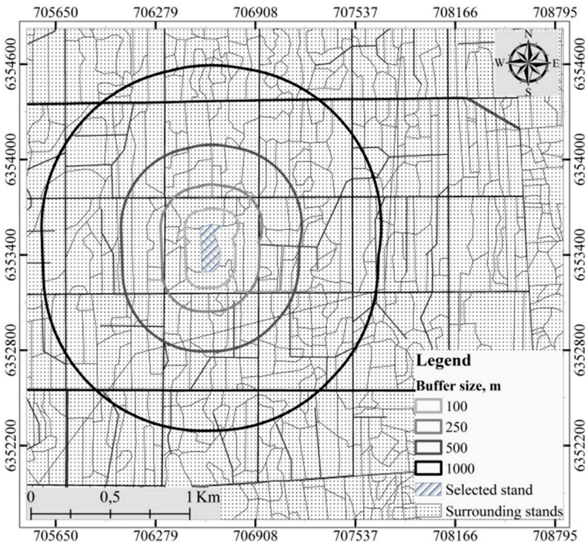

Figure 2. The selected stand surrounded by other stands in the forest massif overlaid with a layer of buffer areas used for calculation of the Shannon diversity index.

2.4. Statistical Analyses

Spatial autocorrelation was tested to assess whether damaged stands or control stands in affected forest massifs had come from a hypothetical random distribution based on both feature locations and attribute values using the Global Moran's I statistic [48]. Spatial statistical analyses were performed using the Spatial Statistics toolbox in ArcGIS 10.2 [43].

The homogeneity of variances among the mean values of landscape metrics for the selected Norway spruce stands (we combined both damaged and control stands in affected forest massifs) between the affected and unaffected forest massifs were analyzed using Kruskal-Wallis test with χ^2 approximation, and the level of significance was $p < 0.05$. Data were first checked for normal distribution using the Shapiro-Wilk test.

The differences in the relative abundance of land cover/use categories within a forest massif were investigated using the χ^2 test. We tested only the most abundant land cover/use categories within a forest massif, such as soft broadleaves mixed woodlands, pure Scots pine stands, pure soft broadleaves stands, pure Norway spruce stands (Non-target stands), and pure Norway spruce stands (including a subset of the selected Norway spruce stands). A comparison of the spatial patterns around patches of damaged stands and control stands (including adjacent patches) created by different land cover/use categories was computed using the χ^2 test. We also investigated the difference between damaged stands and control stands with respect to the length (m) of the shared border with neighboring stands, including Norway spruce, in their composition using the χ^2 test.

We used a Bayesian binary logistic generalized linear mixed effects model (GLMM) as implemented in the software R 3.5.0 [49] library *brms* [50] to assess the effect of landscape metrics on the probability of SBS presence in the stand. The generalized linear mixed-effects model was built based on data on the damaged stands and control stands in affected forest massifs. Forest massif was used as a random effect to account for possible correlation of stands coming from the same massif and to deal with pseudo-replication. To account for spatial dependence between observations, spatial conditional autoregressive correlation structure was added to the model. The number of iterations was set to 12,000 for each of four chains; the thinning rate was set to 8. Convergence of the model was assessed by Rhat values (all values between 1.00 and 1.04). Before constructing the GLMM, multicollinearity amongst all predictor variables was checked based on variance inflation factors (VIF) and by analyzing paired correlations (only factors with VIF < 3 were included in the model). From all landscape metrics, only five variables were included in the model: *H* 100, *H* 1000, MSI, MPAR and core area. The predictor variables were scaled to avoid failure of model convergence.

3. Results

3.1. SBS Distribution in Forest Areas

The spatial patterns of distribution of the land cover/use categories differed significantly (χ^2 test, $p < 0.001$) among forest massifs (Table 2). The patches of damaged stands in Affected forest massifs exhibited clustered distributions in the Affected 1, Affected 2 and Affected 4 forest massifs, while in the Affected 3 forest massif, the distribution of damaged stands was considered random. We found that in all forest massifs, the control stands exhibited a clustered pattern. The IJI indicated that the patch interspersion of selected Norway spruce stands distributed quite similarly among available patch categories in both affected (combining both damaged stands and control stands) and unaffected forest massifs (70.9% and 68.17%, respectively).

Table 2. Proportions of the forest massif area of the different land cover/use categories.

Row Labels	Affected 1	Affected 2	Affected 3	Affected 4	Unaffected 1	Unaffected 2
Other damages	0.0	0.0	0.0	0.0	0.0	0.0
Bogs	0.1	2.4	0.5	11.3	2.0	2.9
Wetlands	0.4	0.2	0.3	0.3	0.3	0.2
Infrastructure	0.4	0.8	0.6	0.6	1.5	2.6
Hard broadleaves	0.4	0.1	0.9	0.5	0.3	0.1
Selected Norway spruce *control stands* *	1.0	0.2	0.5	1.0	1.6	1.6
Damaged stands **	1.4	0.2	0.7	1.3	0.0	0.0
Glade	1.5	1.3	1.1	0.9	0.8	1.2
Non-forest	1.9	5.2	2.6	2.8	1.9	2.2
Scots pine mixed woodland	2.4	3.1	1.8	2.6	4.2	4.9
Clear-cut	4.5	4.3	5.0	2.6	3.0	4.7
Norway spruce mixed woodland	5.1	5.8	4.1	3.7	4.4	3.9
Norway spruce Target stands	7.3	9.1	13.0	6.8	9.9	8.5
Norway spruce stands (Non-target stands)	8.1	9.7	6.4	9.4	4.7	4.5
Soft broadleaves	9.6	9.7	13.9	10.5	10.4	7.1
Scots pine pure stands	18.5	10.6	12.0	16.7	28.4	29.0
Soft broadleaves mixed woodland	37.4	37.4	36.6	28.9	26.6	26.7

* subset of Norway spruce Target stands for detailed analysis. ** subset of Norway spruce Target stands where clear-cut was conducted after SBS infestation.

The area of damaged stands varied among forest massifs. The greatest presence of the SBS damaged stands (112 stands with total class area (PLAND) of 234.0 ha, or 1.36% of the total forest massif area) was reported in the forest massif Affected 1. The lowest presence of the SBS-affected stands (13 stands with total class area of 21.5 ha or 0.18%) was reported within the Affected 2 forest massif. The class area of selected control stands in affected forest massifs varied similarly from 0.16% to 1.04% (67.8 ha to 181.5 ha) of the total forest massif area (Table 2).

3.2. Habitat Characterisation with Landscape Metrics at Class Level

Landscape metrics of the selected Norway spruce stands varied among affected and unaffected forest massifs (Kruskal Wallis test, $p < 0.05$). Patch size (MPS) differed significantly among forest massifs (χ^2 = 36.766, df = 1, p-value < 0.001, Figure 3a). Affected forests have larger patches of selected Norway spruce stands (combining both damaged and control stands) compared to unaffected forest massifs, with average patch sizes of 2.19 ± 0.19 (confidence interval (CI)) and 1.32 ± 0.13 ha, respectively. Patch mean core area also differed among forest massifs (χ^2 = 33.674, df = 1, *p*-value < 0.001). The core area (CA) of selected Norway spruce stands within unaffected forest massif was smaller and more edge-influenced than within affected forest massifs (0.47 ± 0.8 ha and 0.98 ± 0.13 ha, respectively). The percentage of the patch that is comprised of core area (CAI) was greater for affected forest massifs (30.7%) than for unaffected forest massifs (23.8%, $p < 0.001$, Figure 3b). Patch shape (MSI) was relatively simple and did not differ significantly between affected and unaffected forest massifs for selected Norway spruce stands (1.43 for affected and 1.40 for unaffected forest massifs, respectively (Figure 3c).

Patches with significantly higher-edge density were found in affected forest massifs compared to unaffected forest massifs (χ^2 = 32.808, df = 1, *p*-value < 0.001), whereas the mean length of all edge segments relative to the landscape area (ED) for patches of selected Norway spruce stands did not differ among affected and unaffected forest massifs (Figure 3d).

Significantly smaller contiguous patches of selected Norway spruce stands were found in unaffected forest massifs than in affected forest massifs (χ^2 = 33.674, df = 1, *p*-value < 0.001, 0.81 ± 0.01 and 0.84 ± 0.01, respectively).

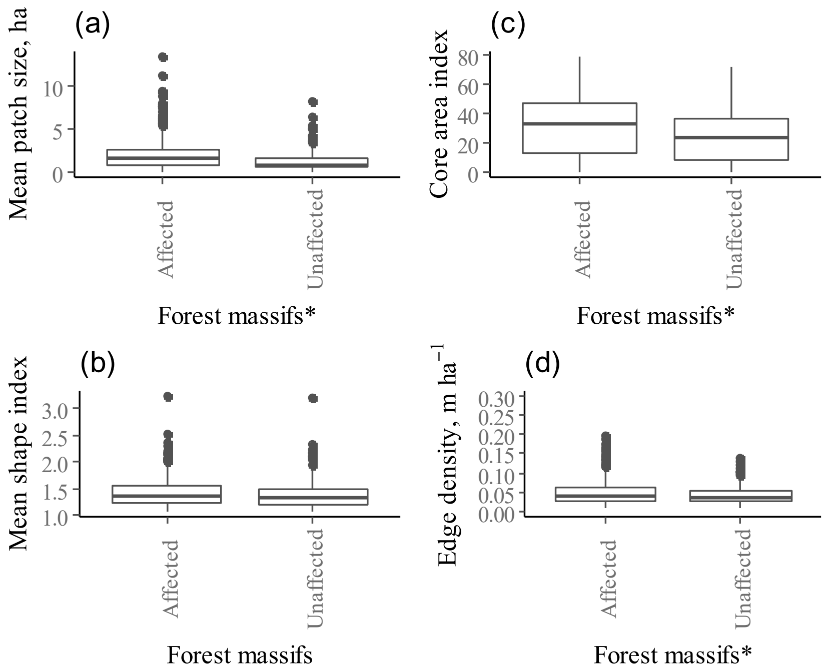

Figure 3. Mean landscape metrics ((**a**) mean patch size, (**b**) core area index, (**c**) mean shape index, (**d**) edge density) for selected Norway spruce stands (we combined both damaged and control stands in affected forest massifs) between the affected and unaffected forest massifs (* indicate significant difference using Kruskal-Wallis test; overall significance level = 0.05).

3.3. *Spatial Heterogeneity of Pest-Damaged Stands*

We found no differences ($p > 0.05$) between damaged and control stands in the length of the shared border with neighboring stands with compositions including Norway spruce. Within forest massifs, the surroundings of damaged stands and control stands (apart in affected and unaffected forest massifs) were significantly different (χ^2 = 2785.9, $p < 0.001$, Figure 4). The damaged stands were embedded in a landscape dominated by birch (41% of total area of surrounding stands), Norway spruce (33%) and Scots pine (16%) stands. In contrast, the neighboring area around control stands in affected forest massiffs was composed of Norway spruce (39%), birch (29%) and Scots pine (11%) dominated stands; additionally, almost 12% of the cover of surrounding stands was formed by black alder and aspen stands. We found that in the unaffected forest massifs, the areas surrounding the selected Norway spruce stands were dominated by Norway spruce (31%), birch (26%), Scots pine (24%) and black alder stands (9%); see Figure 4.

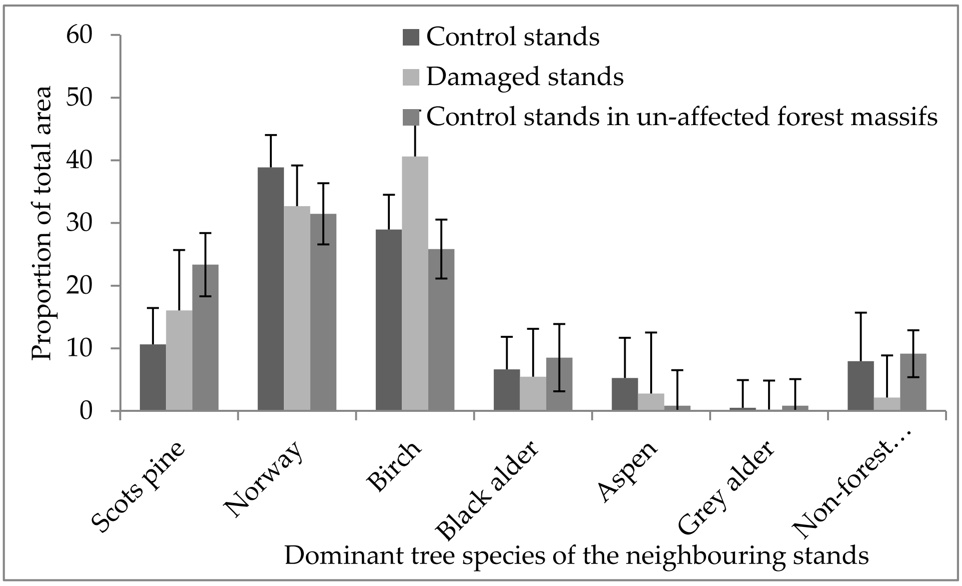

Figure 4. The distribution of dominant tree cover in the areas surrounding damaged stands, control stands in affected forest massifs and control stands in unaffected forest massifs (±95% confidence interval).

Measures of the landscape heterogeneity surrounding the selected Norway spruce stands varied among forest massifs and were not consistent across different buffer radii (Figure 5). At the 100 m scale, the forest diversity (*H* 100) around selected Norway spruce stands in affected forest massifs was lower than in unaffected forest massifs (χ^2 = 88.228, df = 1, *p*-value < 0.001).

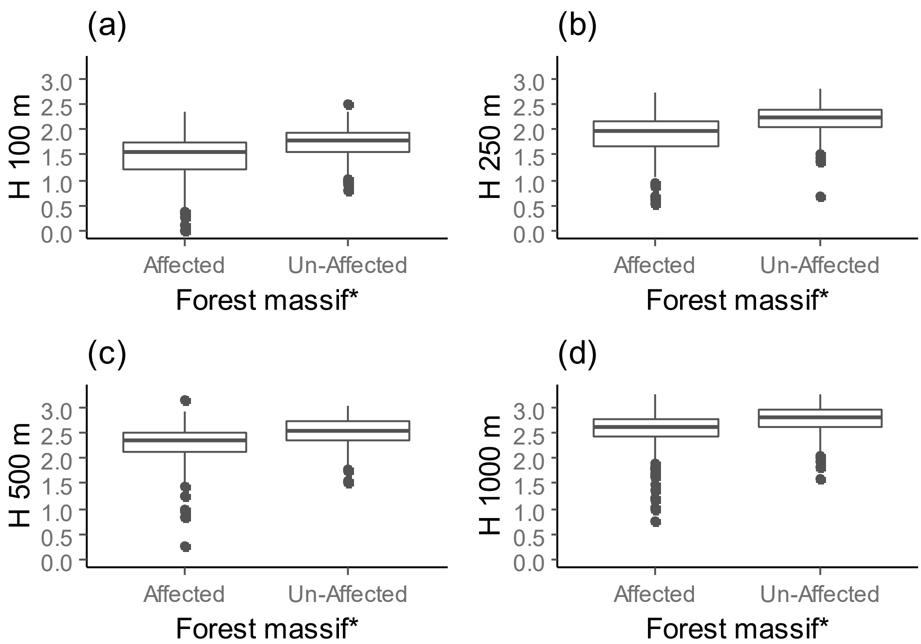

Figure 5. The forest diversity surrounding selected Norway spruce stands within different buffer radii ((**a**) *H* 100, (**b**) *H* 250, (**c**) *H* 500, (**d**) *H* 1000) for affected and unaffected forest massifs (* indicate significant difference using Kruskal-Wallis test; overall significance level = 0.05).

Similar, we found lower spatial diversity within 250 m of selected Norway spruce stands (*H* 250) in affected forest massifs than in unaffected forest massifs (χ^2 = 108.03, df = 1, *p*-value < 0.001). At the 500 m scale (*H* 500), however, the forest diversity around selected stands in affected forest massifs was higher than in unaffected forest massifs (χ^2 = 92.937, df = 1, *p*-value < 0.001). At the 1000 m buffer radius (*H* 1000), the surrounding environment was more diverse in unaffected forest massifs than in affected forest massifs (χ^2 = 56.705, df = 1, *p*-value < 0.001).

Our results highlight the influence of spatial scale on SBS damages within affected forest massifs. We found that the difference in surrounding forest diversity between damaged stands and control stands in affected forest massifs varied with buffer distance. For buffer radii up to 250 m in the affected forest massifs, the forest diversity was higher surrounding control stands than damaged stands. In contrast, for a 500 m or greater buffer size in affected forest massifs, higher forest diversity of surroundings was found for damaged stands than for control stands (Figure 6). However, a high correlation (r = 0.60, $p < 0.001$) with similar distances was estimated, i.e., *H* 100 vs. *H* 250 and *H* 500 vs. *H* 1000.

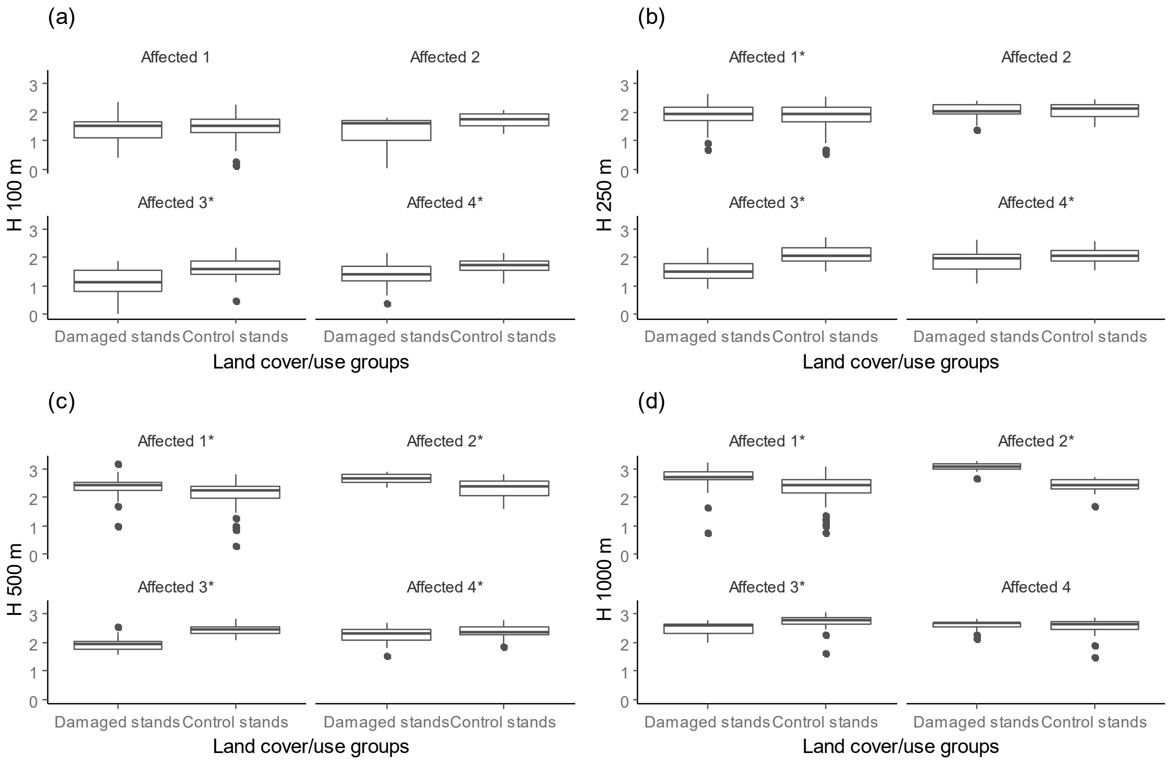

Figure 6. The Shannon diversity index (H) for forest cover types surrounding damaged and control stands within different buffer distances ((**a**) *H* 100, (**b**) *H* 250, (**c**) *H* 500, (**d**) *H* 1000) in affected forest massifs (* indicate significant difference using Kruskal-Wallis test; overall significance level = 0.05).

3.4. Habitat Modelling of Spatial Distribution of Predicted Spruce Stand Damage by Spruce Bud Scale

The binary logistic GLMM analysis indicated that there was no significant effect of mean patch size (MPS) (estimate −0.03, Credibility interval −2.83 to 2.50), MSI (estimate −0.57, CI −2.80 to 1.64) or core area (estimate 2.53, CI −0.24 to 6.19) on the probability of SBS presence (Table 3). The diversity around patches within 1000 m (*H* 1000) (estimate 5.82, CI 2.15 to 10.32) was associated with an increased probability of SBS presence. In contrast, we found that increased local diversity around patches within 100 m (*H* 100) (estimate −4.24, CI −7.86 to −1.72) reduced the probability of SBS presence. Model results showed that variation (standard deviation) for random effect was 5.80 (CI 0.19 to 18.65) and estimate for car (correlation structure) was 0.98 (CI 0.95 to 1.00).

Table 3. The model for predicting the effect of landscape heterogeneity on SBS spread in a forested landscape (l-95% CI—lower 95% credibility interval, u-95% CI—upper 95% credibility interval).

Parameter	Estimate	Est. Error	l-95% CI	u-95% CI
(Intercept)	2.14	5.81	–9.09	14.19
H 100	−4.24	1.58	–7.86	−1.72
H 1000	5.82	2.09	2.15	10.32
MSI	−0.57	1.10	–2.80	1.64
MPS	−0.03	1.33	–2.83	2.50
Core area	2.53	1.63	−0.24	6.19

4. Discussion

The relationships between landscape characteristics and pest distribution could be associated with factors such as habitat diversity, patch size and shape, edge effect, and the connectivity and diversity of stands across the landscape [51,52]. The 2010 outbreak of SBS was evaluated through the interrelated analysis of landscape metrics for patches of selected Norway spruce stands among affected and unaffected forest massifs. The dispersal and movement of SBS indicated the influence of landscape characteristics. The abundance of different insect populations and their movement between habitat patches are affected by patch size and isolation. For example, a population could exist at a higher density in larger-sized patches than in small patches [53]. In our study, the patch size of the selected Norway spruce stands was more relevant in the affected forest massifs (Figure 3a). In contrast, a previous study in Lithuania found no effect of forest stand area on SBS invasion [54].

Previous studies that have investigated the effects of landscape heterogeneity on forest insects have focused on three related aspects of landscape structure: abundance of host species, edge effects and fragmentation [45]. Although, the probability of insect dispersal will increase with favorable conditions and a high abundance of host species [55], we found high clustering between selected Norway spruce stands in affected forest massifs, implying that SBS tends to colonize such/similar habitat islands in the forest massif. The study of *Physokermes inopinatos* [17] in Sweden suggests that in spring 2010, bud scales were able to spread to surrounding spruce stands from those areas that were already affected in 2009. This is supported by findings in other studies that SBS spreads to surrounding stands in 1 to 2 years [56]. Moreover, the SBS migrates mainly short distances via contact of the infested tree's crown with its neighbors [20,41]. Consequently, SBS is generally concentrated in core areas, and subsequent damages originate from these areas [17]. Our findings support this fact, suggesting that core areas of selected Norway spruce stands were greatest in the affected forest massifs. In contrast, an insufficient interior area of Norway spruce forest in un-affected forest massifs indicates that a smaller interior forest area may limit the dispersal of SBS populations.

Insect movement rates can vary markedly based on forest configuration features such as patch edges compared to interiors, due to altered environmental conditions [57]. The SBS reproduce and grow in sunny and dry places [56]. The results suggest that significantly shorter edge lengths for selected Norway spruce stands were more relevant within unaffected forest massifs. Although, caution should be exercised, as forest edge may also be a suitable habitat for predator species; additionally, species richness at patch edges could increase as these areas contain properties of both adjacent patches [55]. In some cases, the edge effects may vary in relation to the nature of the edge. For example, a coniferous surrounded by other coniferous stands contrast to a much-lower extent with its adjacent land covers than a similar coniferous patch surrounded by hardwood species [58]; this also enhanced the dispersal abilities of SBS among patches.

Each component in a landscape could be associated to a certain ecological function. It may also reflect the behavior of species in relation to spatial heterogeneity. For example, irregular patch shape corresponds to higher edge effects [59]. Therefore, the relationship between patch shape complexity and edges can significantly affect the population dynamics of insects [57]. In the study, polygon shapes were simplified, and mean shape index of polygons was relatively low in both landscapes (Figure 3c).

This may be explained by extensive forest management and land use intensification, leading to relative simplification of the shapes of patches [60].

The forest composition at different spatial scales (such as local neighborhood and/or a regional scale) may serve as a key factor in the movement of insect populations [34,59,61]. The movement of species across a landscape is associated with a certain connectivity value [53] and can be described by an index of patch boundary configurations and thus patch shape [62]. In the study, affected and unaffected forest landscapes showed the same level of patch adjacencies (IJI). Specifically, within both affected and unaffected forest massifs, patches of selected Norway spruce stands were well adjacent to each other and distributed approximately 70% of the possible equitable distribution within forest massifs. The IJI quantifies the level of patch isolation by focusing solely on the edges and is maximum when each category represented in a landscape shares a common border with each of the others [63].

The capacity of SBS to disperse apparently was not affected by the neighboring stands if Norway spruce was found in the neighboring stands' composition. This suggests that—apart from "habitat islands"—the observed aggregation of patches of selected Norway spruce stands within both affected and unaffected forest massifs was also related to other factors (see Figure 3). Bergeron et al. [64], in studies of spruce budworm, found that coniferous stands embedded in a matrix dominated by deciduous stands were less vulnerable than large expanses of coniferous forest. Findings of the authors indicate a significant difference in the spatial patterns of the surroundings of damaged and control stands. However, the impact of deciduous tree cover on the probability of the presence of SBS insects in this study still remains unclear due to a high proportion of deciduous tree cover that was present in the surroundings of both damaged and control stands. Moreover, the high proportion of pure Scots pine stands in unaffected forest massifs indicates a predation interaction effect on community dynamics, as it is widely acknowledged that Scots pine is a host tree for *Anthribus nebulosus,* a predator of spruce bud scale. Research in Serbia suggests that the predatory species *Anthribus nebulosus* reduced the population of scales by 68%–80% [16]. In Germany, the corresponding reduction of the population of spruce bud scales was 38%–59% [65], and in Lithuania, *Anthribus nebulosus* was considered to have high efficacy at reducing this pest [54]. The spatial scale of the heterogeneity of forest stands in large fragments might be another factor explaining the presence of a species-area relationship [66]. A complex landscape pattern in combination with effective forest management may reduce insect outbreaks in the future [28]. Consequently, changes in forest structure elements, such as smaller patches, higher edge densities, and decrease in interior area, leading to a more complex patch structure caused by forest management, may lead to a decreased risk of pest presence [30,32]. Thus, in forested landscapes, the role of mixed forest composition or configuration is related to ecological processes affecting the occurrence and spread of insect outbreaks [45]. We found that the Shannon diversity index of the land cover surrounding damaged and control stands in affected forest massifs varied among different habitat scales. This indicates that the SBS infections might be related to effects of a significant scale and high spatial variability. We determined that with an increased buffer size, the H diversity around damaged stands became more diverse than around control stands. This indicates that spreading of this pest could potentially be related to diversity at larger spatial scales, not only to local habitat diversity. The same appears to apply to other relatively mobile invertebrates [67]. Moreover, the model suggests that the probability of the presence of SBS decreases with greater local diversity surrounding a stand, while a higher probability of the presence of SBS is related to increased diversity within 1000 m (H 1000) and a larger patch size.

5. Conclusions

This study indicates that higher forest structural diversity has an impact on spruce bud scale presence at different spatial scales that can be characterized by landscape metrics. In addition, the predicted probability of the presence of spruce bud scale increases with certain landscape characteristics. Two apparently contradictory patterns in the effect of landscape diversity on presence of spruce bud scale were determined. Higher spatial heterogeneity at the local scale correlates to lower probability of spruce bud scale presence and greater resistance to damage by this pest species. This suggests that increased local diversity around patches results in decreased presence of spruce bud scale. However, more complex spatial structure will not guarantee effective pest regulation, especially since the pest's presence in Norway spruce stands may be positively associated with higher diversity at a broader scale (i.e., within a buffer of 1000 m around the patch). These investigations could help identify effective pest-control mechanisms for forest management. Hence, this may facilitate the planning process for forest owners in order to reduce the risk of pest outbreaks in the future. However, there is still incomplete understanding of the different interactions between landscape metrics and the presence of these new pests in hemiboreal forests. Thus, more detailed studies are needed to specify the interrelations between abiotic and biotic factors and dispersal at local and regional scales in forested landscapes.

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