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Effects of undergrowth removal and edge proximity on ground beetles and vascular plants in urban boreal forests

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Abstract

Urban forests are regularly managed for human safety and esthetic reasons, but they are crucial habitat for many species. Removals of undergrowth occur commonly in these forests, yet the ecological consequences of these operations are poorly understood. We sampled ground beetles (Coleoptera, Carabidae) and vascular plants along 20-m edge gradients in Finnish urban forests, in five stands treated 0.5–2.5 years earlier with undergrowth removal and in five untreated stands. We hypothesized that undergrowth removal and edge proximity would benefit opportunistic and open-habitat species, whereas shady-habitat species would be affected negatively. (1) Regarding carabids, diversity and evenness indices, open-habitat species and *Carabus nemoralis* responded positively, and forest species, *Leistus terminatus* and *Pterostichus oblongopunctatus* responded negatively, to the undergrowth removal. Regarding plants, generalists, *Maianthemum bifolium*, *Rubus saxatilis* and *Sorbus aucuparia* responded positively, and forest species, *Geranium sylvaticum*, *Oxalis acetocella* and *Vaccinium myrtillus* responded negatively, to the undergrowth removal. (2) Edge proximity had little effect on both plants and carabids. However, open-habitat carabids were less abundant and less speciose, and the plants *Oxalis acetocella*, *Trientalis europaea* and *Rubus saxatilis* had higher cover, 10–20 m from than right at the edge. (3) Plant (but not carabid) community responded to the undergrowth removal but not to the edge proximity. When managing urban forests, we recommend an avoidance of undergrowth removals at sites that host rare or threatened forest-associated flora and fauna.

Key words: abundance, Carabidae, community, edge effect, management, species richness

Introduction

Urban areas currently host 54% of the Globe's human population, and the proportion may reach 68% by 2050 (United Nations 2018). This trend is made possible by, e.g. increasing intensification of land use and the spread of urban areas (Seto, Guneralp, and Hutyra 2012). These in turn result in loss and fragmentation of earlier contiguous habitat and alterations in biodiversity (McDonnell et al. 1997; Whitford, Handley, and Ennos 2001; Grimm et al. 2008). At a patch level, urban species communities

generally contain fewer species and different dominance structure, compared with communities in rural areas (Niemelä et al. 2011). This pattern results from such stressors as pollutants and land use, characteristic for urban environments (McDonnell et al. 1997; Forman 2008).

Urban forests bear many values. For humans, they provide many ecosystem services, such as plant pollination, ground and surface water protection, particle filtering from the air, and noise reduction (Forman 2008; Salo 2015). Urban people commonly appreciate esthetic and recreational values of these

forests (Tyrväinen, Silvennoinen, and Kolehmainen 2003; Chiesura 2004; Kabisch, Qureshi, and Haase 2015). Moreover, from a biodiversity perspective, these forests provide habitat and dispersal routes for many species (Niemelä et al. 2011). Urban areas can also host rare and threatened species (e.g. Alvey 2006; Jones and Leather 2012; Ives et al. 2016). Biodiversity surveys in cities world-wide support this view; Finnish examples include Kurtto (2002), Savola (2012) and Ellermaa (2018). However, quite often the urban spread also produces 'more of the same': urban areas are particularly favorable for opportunistic generalists, disturbance tolerant species and cosmopolitan species (McKinney 2006, 2008; Sadler et al. 2006).

Even if urban forests can be considered important from many viewpoints, their preservation competes with economics (construction projects, urban spread) and their ecological quality is often compromised by such factors as edge effects or trampling. Generally, these factors tend to negatively impact species that are associated with mature forests, and benefit opportunistic species and others that occupy well-exposed habitats, such as meadows (e.g. Littlemore and Barker 2002; Malmivaara, Löfström, and Vanha-Majamaa 2002; Magura, Tóthmérész, and Molnár 2008; Malmivaara-Lämsä et al. 2008; Kotze et al. 2012). Moreover, management of urban forests is common, not necessarily for economic but for safety or aesthetic reasons (e.g. Hamberg, Löfström and Häkkinen 2012). Undergrowth removal, for instance, is a routine operation in these forests, yet its ecological effects are poorly understood except, perhaps, on soil quality and tree growth. With undergrowth we refer to trees and bushes that have not reached the canopy-tree layer, thus exclude floor- and field-layer vegetation.

In this article, we compare urban boreal forests where undergrowth had recently been removed to similar forests where undergrowth had been retained. Our data consist of ground beetles (Coleoptera, Carabidae; hereafter 'carabids') and vascular plants sampled at different distances from edges of these forests. North European carabids and vascular plants are ecologically and taxonomically well known, and both respond to micro-habitat variation at different spatial scales (e.g. Niemelä 1997; Reinikainen et al. 2000; Mossberg and Stenberg 2003; Koivula 2011). These qualities make them suitable model organisms for studying biological responses to human impact at spatial scales of within and between stands, which are relevant for urban land-use management. Surprisingly, then, in their review on biodiversity of urban parks, Nielsen et al. (2014) concluded that studies on urban environments seldom bridge between fauna and flora, a core feature of this study.

Research concerning boreal forest management suggests that undergrowth removal can impact carabids and plants. For example, clear-cutting changes the assemblage (hereafter 'community' for convenience) structure of carabids (Niemelä, Koivula, and Kotze 2007) and plants (Jalonen and Vanha-Majamaa 2001; Pykälä 2004; Palviainen et al. 2005; Tonteri et al. 2016) through rapid colonization of open-habitat associated species and increase of succession generalists (Niemelä et al. 1988; Spence et al. 1996; Koivula 2002; Pykälä 2004). If living trees are retained, changes in species composition can be considerably smaller (Bergstedt and Milberg 2001; Macdonald and Fenniak 2007; Work et al. 2010; Johnson, Strengbom, and Kouki 2014), perhaps because of smaller changes in the forest-floor shade and micro-climate (Koivula 2012). Hence the undergrowth removal—studied here—may be expected to affect carabids and plants only little, as the sheltering canopy trees are retained. However, undergrowth removal is often accompanied

by removals of tree trunks and branches from the ground, resulting in lower structural variation and altered nutritional conditions in the soil, which in turn potentially impact forest specialists negatively, and succession generalists positively (Palviainen 2005; Nittérus, Åström, and Gunnarsson 2007).

Adjacent habitats exchange energy, species and nutrients across their mutual boundary, and species communities change gradually across a gradient from one habitat to another; these are referred to as edge gradients (Murcia 1995). In this study, the gradients reach from street-side verges toward interior parts of forested patches. These edges are abrupt interfaces between forested patches and artificial surfaces (paved roads, built-up areas etc.), the latter being quite inhospitable for carabids and plants (e.g. Guirado, Pino, and Roda 2006; Prass et al. 2017). In forest-edge gradients, the overall richness and open-habitat associated species may decrease, and shade-demanding species may increase, toward interior forest, as earlier shown for carabids (Heliölä, Koivula, and Niemelä 2001; Magura 2002; Koivula, Hyyryläinen, and Soininen 2004) and plants (Guirado, Pino, and Roda 2006; Hamberg et al. 2008; Vallet et al. 2010).

Here, we propose the following predictions:

1. As communities of carabids and vascular plants in managed boreal and temperate forests largely consist of open-habitat species and succession/canopy-closure generalists (Tonteri 1994; Pykälä 2004; Niemelä et al. 2007; Aavik et al. 2009; Bescond, Fenton, and Bergeron 2011; Duguid and Ashton 2013), we expect the total species richness and diversity of carabids and plants to peak in stands where undergrowth has been removed, and at sampling plots closer to the edge.
2. The abundance and species richness of species requiring shady conditions or closed canopy (hereafter 'forest species' for convenience) should decrease, whereas those of open-habitat associated species (hereafter 'open-habitat species') should increase, following undergrowth removal.
3. The abundance and richness of forest species should increase, and those of open-habitat species should decrease, with increasing distance from forest-patch edge toward interior.
4. If at least some carabids and plants respond to the undergrowth removal and/or edge proximity, then the community structure of these groups should change with these two factors accordingly.

Methods

We selected five study sites, each with two stands 50–200 m apart, distributed over the city of Jyväskylä, Finland (62°14' N, 25°44' E; Fig. 1). Each stand represented Myrtillus or Oxalis-Myrtillus type forest (Cajander 1949). The dominant canopy trees were 60–100 years old Norway spruce (*Picea abies*), Scots pine (*Pinus sylvestris*), aspen (*Populus tremula*) and birches (*Betula pendula* and *Betula pubescens*). The undergrowth—prior to its removal from half of the stands—consisted mostly of birches, aspen, gray alder (*Alnus incana*), rowan (*Sorbus aucuparia*) and alder buckthorn (*Rhamnus frangula*). Forest floor and field vegetation was dominated by *Vaccinium* dwarf shrubs, *Calamagrostis* and *Deschampsia* grasses, and by *Pleurozium*, *Dicranum* and *Hylocomium* mosses.

About 45–70% of undergrowth and occasional canopy trees were removed from the other of the two stands at each site during winters 2006/2007 (two stands) or 2008/2009 (three stands). The other stand at each site was left intact at the time of study (Table 1). With 'treatment' we refer to the undergrowth removal

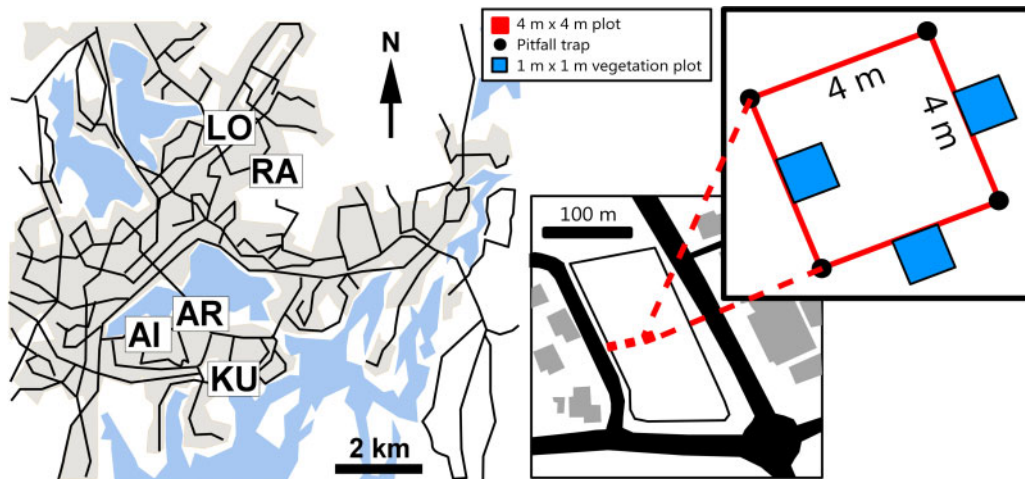


Figure 1: Spatial distribution of five sampling sites, each with two study stands, in the city of Jyväskylä, Finland. Black lines are main roads or streets; light blue patches are lakes; light gray patches show built-up areas (settlement, industrial etc.). The sites are indicated with two-letter codes (compare Table 1). The vignettes show an example study stand (Ainola, untreated) with three sampling stations arranged to an edge-distance gradient from 0 to 20 m toward patch interior, and a sampling station showing the placing of four pitfall traps and three vegetation plots.

Table 1: Structural variables collected from 10 study-forest stands in Jyväskylä, Finland

| Variable | AI-UN | AI-TR | AR-UN | AR-TR | KU-UN | KU-TR | LO-UN | LO-TR | RA-UN | RA-TR |
|--|-----------|-------|-----------|-------|-----------|-------|-----------|-------|-----------|-------|
| Structural variables | | | | | | | | | | |
| Edge compass direction | NE | SW | S | E | SE | NW | W | W | NW | NW |
| Treatment winter* | 2008/2009 | | 2008/2009 | | 2008/2009 | | 2006/2007 | | 2006/2007 | |
| Canopy cover (%) | 83 | 60 | 80 | 50 | 63 | 23 | 57 | 10 | 45 | 35 |
| Canopy-layer trees, age* | 65 | 90 | 100 | 100 | 90 | 90 | 85 | 90 | 60 | 55 |
| Trees, volume (m ³ /ha)* | 104 | 211 | 157 | 218 | 198 | 173 | 224 | 197 | 107 | 183 |
| Trees, n/ha* | 440 | 350 | 370 | 340 | 350 | 320 | 540 | 220 | 440 | 340 |
| Understory, n/ha | 1400 | 600 | 2800 | 1300 | 4100 | 1600 | 12 800 | 3600 | 1500 | 800 |
| Litter, cover (%) | 27.3 | 16.7 | 29.8 | 45.5 | 25.0 | 27.8 | 46.7 | 14.2 | 33.3 | 17.2 |
| Rocks, cover (%) | 0.6 | 0.0 | 24.5 | 0.0 | 15.6 | 16.1 | 0.6 | 2.3 | 4.0 | 2.4 |
| Mosses, cover (%) | 10.2 | 5.9 | 13.3 | 0.0 | 9.0 | 6.9 | 3.1 | 2.5 | 9.6 | 1.3 |
| Bare ground, cover (%) | 3.8 | 7.3 | 2.3 | 1.1 | 1.3 | 0.7 | 0.2 | 0.2 | 0.1 | 0.1 |
| Carabid beetles (mean/sampling station) | | | | | | | | | | |
| Number of species | 8.7 | 8.7 | 3.7 | 9.0 | 10.7 | 7.0 | 4.3 | 5.7 | 7.3 | 6.7 |
| Number of individuals | 60.3 | 29.3 | 17.3 | 72.0 | 98.0 | 28.3 | 11.3 | 12.0 | 42.7 | 17.7 |
| Diversity (Shannon-Wiener H) | 1.6 | 1.8 | 0.9 | 1.8 | 1.8 | 1.6 | 1.2 | 1.5 | 1.4 | 1.4 |
| Evenness (Simpson's J) | 0.7 | 0.8 | 0.5 | 0.8 | 0.8 | 0.8 | 0.7 | 0.7 | 0.7 | 0.7 |
| Vascular plants (mean/sampling station) | | | | | | | | | | |
| Number of species | 13.3 | 15.7 | 10.3 | 13.3 | 14.7 | 15.0 | 16.0 | 11.7 | 13.0 | 18.7 |
| Diversity (Shannon-Wiener H) | 1.8 | 2.1 | 1.8 | 1.8 | 2.0 | 2.1 | 2.2 | 1.6 | 1.7 | 2.1 |
| Evenness (Simpson's J) | 0.7 | 0.8 | 0.8 | 0.7 | 0.7 | 0.8 | 0.8 | 0.7 | 0.7 | 0.7 |

Stand identities are indicated with two-part abbreviations based on site name and treatment. AI, Ainola; AR, Ainolanranta; KU, Kuokkala; LO, Lohikoski; RA, Rasinrinne (compare Fig. 1); UN, untreated; TR, treated (undergrowth removed). The structural variables are based on data from the city of Jyväskylä (indicated with *) and on our own inventories; the richness measures for carabid beetles and plants are based on data used in this study.

below. The treatment was done by city timberjacks using clearing saws and was accompanied by removals of logging residue, which is a normal procedure in urban forest management in Finland.

We sampled carabids and vascular plants during May–September 2009, i.e. 0.5–2.5 years after the treatment. Within each of the 10 stands, we established three sampling stations: at 0 m, i.e. the stand edge, and at distances 10 and 20 m into the forest interior. We acknowledge that edge effects may reach at least 25–30 m into the forest interior (e.g. Chen, Franklin, and Spies 1993) but we consistently applied the 20 m distance because this was the farthest from the nearest edge in some stands. The compass directions of edge facing varied independently in treated and untreated stands (Table 1).

To collect carabids, we used pitfall traps (mouth diameter 70 mm, volume 0.25 l, half filled with 30% propylene glycol and a drop of detergent, covered with a 10 × 10 cm plastic roof set 2–3 cm above the trap rim; e.g. Koivula et al. 2003). We placed four traps into each corner of a 4 × 4 m square of each sampling station (Fig. 1). The traps were operating from late May to early September and were serviced every 2–4 weeks. We sorted the samples and identified the collected beetles in a laboratory. For the purposes of analysis, we pooled the samples of a given station over the whole summer, thus resulting in (10 stands × 3 stations) 30 samples.

We carried out vegetation inventories in June by establishing three 1 × 1 m plots at each sampling station in each stand

(Fig. 1). For each plot, we recorded each plant species along with its percent cover. For analysis purposes we pooled vegetation data by averaging percent covers of each plant species per sampling station, thus resulting in a total of 30 samples. Table 1 shows structural variation in our study stands. These variables were not used in analyses, however, as they represent different spatial scale from our sampling, and our focus was on undergrowth removal and edge proximity.

We analyzed the carabid and plant abundance and richness data using generalized linear mixed-effects models (GLMMs; Zuur et al. 2009). Our response variables were the total number of species, Shannon-Wiener and Simpson diversity indices, pooled samples of three ecological groups (forest, generalist and open-habitat species), and number of captured individuals (for carabids) or percent cover (for plants) of abundant species. Regarding the three ecological groups of species, we based the grouping on Niemelä, Koivula, and Kotze (2007) for carabids and Reinikainen et al. (2000) and Mossberg and Stenberg (2003) for plants (Appendix). We analyzed both the pooled abundance (for carabids) or percent cover (for plants) and the number of species for groups or species found in >50% of samples, and excluded sites with no observations (one site for the beetle *Pterostichus melanarius*, and for the plants *Calamagrostis arundinaceus* and *Rubus saxatilis*). It must be noted that passive collecting methods (such as pitfall trapping) do not collect different species with their true relative proportions but are biased toward larger and more actively moving species (Koivula 2011), which limits the interpretation of indices or community analyses for carabids.

In GLMM, we set site (each with two stands) as a random factor to account for spatial inter-dependence of adjacent stands and their sampling stations, and treatment (undergrowth not removed, undergrowth removed in the winter of 2006/2007, or undergrowth removed in the winter of 2008/2009) and edge proximity (0, 10 or 20 m into the patch interior) as fixed factors. Due to the often low sample sizes, we did not include an interaction between treatment and edge proximity in our models. Regarding carabid data, sample losses were rare and apparently random; preliminary runs with sampling effort as a covariate suggested no effect so we excluded effort from our models. We initially used untransformed data with Gaussian error distribution and checked the normality of residuals using Q-Q plots. If these plots indicated a lack of normality, we square-root transformed the data and reran the models, and if the issue still persisted, we applied quasi-Poisson error distribution (e.g. Ver Hoef and Boveng 2007). The Q-Q plots of final models are shown in Supplementary Figures S1 and S2.

We also explored the possibility for edge responses to differ between treated and untreated stands by calculating separate Spearman rank correlation coefficients between the edge proximity and the number of species, the two indices, the abundance and richness of the three ecological groups of species, and the abundance or percent cover of different species. Due to the high number of comparisons we interpret these results with caution.

To assess community responses to treatment and edge proximity, we used Non-metric Multi-Dimensional Scaling (NMDS; Borg and Groenen 2005) by applying the Bray-Curtis distance matrix. We analyzed beetle and plant data separately. We tested the distinctiveness of samples of untreated stands, and stands treated in the winters of 2006/2007 or 2008/2009, and samples collected at different edge distances (0, 10 or 20 m), using permutational multivariate analysis of variance (ANOVA) (adonis function with strata = site in the R package *vegan*; Oksanen et al. 2018).

We ran the analyses using R 3.4.4 (R Core Team 2018) with *car* (Fox and Weisberg 2011), *MASS* (Venables and Ripley 2002),

vegan (Oksanen et al. 2018) and *lme4* packages (Bates et al. 2015).

Results

We collected 1167 carabid individuals of 26 species and found 61 vascular plant species (canopy trees excluded; Appendix). Six carabid species (*Calathus micropterus*, *Carabus nemoralis*, *Leistus terminatus*, *P.melanarius*, *Pterostichus oblongopunctatus* and *Trechus secalis*) and nine plant species (*C.arundinaceus*, *Deschampsia flexuosa*, *Geranium sylvaticum*, *Maianthemum bifolium*, *Oxalis acetocella*, *R.saxatilis*, *S.aucuparia*, *Trientalis europaea* and *Vaccinium myrtillus*) occupied more than half of the sampling stations and were thus subjected to GLMM, in addition to the total and species-group abundances and richness measures.

Responses to the undergrowth removal

In carabids, the treatment did not affect the number of species, but the two indices peaked in stands treated in the winter of 2008/2009 (Table 2). In terms of carabid abundance, forest carabids had lowest, and open-habitat carabids had highest, catches in stands treated in the winter of 2006/2007 (Fig. 2); generalists did not respond significantly to the treatment. In terms of species richness, forest or generalist carabids did not respond significantly to the treatment, whereas open-habitat carabids were significantly more speciose in stands treated in the winter of 2006/2007 than in untreated stands (Table 2).

Regarding the plant community, the number of species or the two indices did not show a treatment response (Table 2). Furthermore, forest plants had lower cover in treated than in untreated stands, irrespective of time since treatment (Fig. 2), whereas generalist plants had marginally higher cover in stands treated in the winter of 2006/2007 than in untreated stands. Plant species richness measures for the three ecological groups did not show a treatment response (Table 2).

Three carabid species showed a detectable treatment response (Table 3). *Carabus nemoralis* was marginally more abundant in stands treated in the winter of 2008/2009 than in untreated stands. *Leistus terminatus* was marginally less numerous in stands treated in the winter of 2006/2007, and *P.oblongopunctatus* was marginally less numerous in stands treated in the winter of 2008/2009, than in untreated stands (Table 3).

Six plant species responded significantly or marginally significantly to the treatment (Table 3). *Oxalis acetocella* and *V.myrtillus* had lower cover in treated than in untreated stands, irrespective of time since undergrowth removal (Fig. 3), whereas *G.sylvaticum* had lower cover in the 2008/2009, but not in the 2006/2007, than in untreated stands. *Maianthemum bifolium* had higher cover in stands treated in both winters (2006/2007 and 2008/2009; Fig. 3), *R.saxatilis* had higher cover in 2006/2007, and *S.aucuparia* had higher cover in 2008/2009, than in untreated stands (Table 3).

Responses to the edge proximity

Regarding both carabids and plants, edge proximity did not significantly affect the total diversity measures or the ecological groups of species (Table 2). However, the abundance and richness of open-habitat carabids were marginally lower at 10 m from than at the edge.

In terms of individual species, none of the carabids responded to the edge gradient (Table 3). Regarding plant

Table 2: GLMM summary for carabid beetles and vascular plants sampled in 10 forest stands in Jyväskylä, Finland

| Variable | Effect | SE | t | P | Variable | Effect | SE | t | P |
|--|--------|------|-------|-------|--|--------|------|-------|-------|
| Carabids, number of species | | | | | Plants, number of species | | | | |
| (Intercept) | 6.77 | 1.01 | 6.70 | 0.000 | (Intercept) | 14.43 | 1.46 | 9.87 | 0.000 |
| Treat 06/07 | −0.37 | 1.27 | −0.29 | 0.771 | Treat 06/07 | 1.83 | 1.93 | 0.95 | 0.352 |
| Treat 08/09 | 1.03 | 1.08 | 0.95 | 0.350 | Treat 08/09 | 1.33 | 1.69 | 0.79 | 0.437 |
| Prox 10 m | −0.20 | 1.09 | −0.18 | 0.857 | Prox 10 m | −1.40 | 1.79 | −0.78 | 0.442 |
| Prox 20 m | 0.70 | 1.09 | 0.64 | 0.530 | Prox 20 m | −1.90 | 1.79 | −1.06 | 0.299 |
| Carabids, Shannon-Wiener diversity (H) | | | | | Plants, Shannon-Wiener diversity (H) | | | | |
| (Intercept) | 1.41 | 0.13 | 10.77 | 0.000 | (Intercept) | 1.87 | 0.12 | 15.00 | 0.000 |
| Treat 06/07 | 0.08 | 0.17 | 0.46 | 0.653 | Treat 06/07 | −0.06 | 0.17 | −0.34 | 0.735 |
| Treat 08/09 | 0.36 | 0.15 | 2.45 | 0.022 | Treat 08/09 | 0.13 | 0.14 | 0.88 | 0.388 |
| Prox 10 m | 0.00 | 0.15 | 0.00 | 1.000 | Prox 10 m | −0.04 | 0.15 | −0.26 | 0.796 |
| Prox 20 m | −0.04 | 0.15 | −0.26 | 0.796 | Prox 20 m | 0.04 | 0.15 | 0.26 | 0.796 |
| Carabids, Simpson evenness (J) | | | | | Plants, Simpson evenness (J) | | | | |
| (Intercept) | 0.69 | 0.04 | 16.90 | 0.000 | (Intercept) | 0.76 | 0.03 | 26.08 | 0.000 |
| Treat 06/07 | 0.02 | 0.05 | 0.43 | 0.674 | Treat 06/07 | −0.01 | 0.04 | −0.17 | 0.865 |
| Treat 08/09 | 0.13 | 0.05 | 2.78 | 0.010 | Treat 08/09 | 0.03 | 0.03 | 0.79 | 0.438 |
| Prox 10 m | −0.01 | 0.05 | −0.21 | 0.840 | Prox 10 m | 0.02 | 0.04 | 0.56 | 0.582 |
| Prox 20 m | −0.07 | 0.05 | −1.43 | 0.166 | Prox 20 m | 0.01 | 0.04 | 0.28 | 0.783 |
| Forest carabids, abundance (SQRT) | | | | | Forest plants, % cover | | | | |
| (Intercept) | 3.52 | 0.84 | 4.18 | 0.001 | (Intercept) | 0.27 | 0.08 | 3.19 | 0.009 |
| Treat 06/07 | −1.88 | 1.08 | −1.74 | 0.094 | Treat 06/07 | −0.22 | 0.09 | −2.30 | 0.030 |
| Treat 08/09 | −0.89 | 0.92 | −0.97 | 0.340 | Treat 08/09 | −0.19 | 0.08 | −2.40 | 0.025 |
| Prox 10 m | 0.96 | 0.94 | 1.02 | 0.320 | Prox 10 m | 0.13 | 0.08 | 1.60 | 0.123 |
| Prox 20 m | 1.62 | 0.94 | 1.72 | 0.100 | Prox 20 m | 0.04 | 0.08 | 0.56 | 0.583 |
| Generalist carabids, abundance (SQRT) | | | | | Generalist plants, % cover | | | | |
| (Intercept) | 4.01 | 0.84 | 4.79 | 0.003 | (Intercept) | 0.32 | 0.08 | 4.06 | 0.000 |
| Treat 06/07 | −0.19 | 0.84 | −0.23 | 0.822 | Treat 06/07 | 0.20 | 0.11 | 1.88 | 0.071 |
| Treat 08/09 | 0.12 | 0.69 | 0.18 | 0.862 | Treat 08/09 | −0.07 | 0.09 | −0.71 | 0.484 |
| Prox 10 m | −0.30 | 0.68 | −0.45 | 0.658 | Prox 10 m | −0.04 | 0.10 | −0.45 | 0.657 |
| Prox 20 m | 0.15 | 0.68 | 0.22 | 0.832 | Prox 20 m | −0.13 | 0.10 | −1.37 | 0.184 |
| Open-habitat carabids, abundance (SQRT) | | | | | Open-habitat plants, % cover | | | | |
| (Intercept) | 0.37 | 0.21 | 1.78 | 0.092 | (Intercept) | 0.35 | 0.12 | 2.85 | 0.026 |
| Treat 06/07 | 0.81 | 0.27 | 3.04 | 0.006 | Treat 06/07 | 0.10 | 0.11 | 0.86 | 0.399 |
| Treat 08/09 | 0.28 | 0.23 | 1.21 | 0.238 | Treat 08/09 | 0.02 | 0.09 | 0.21 | 0.836 |
| Prox 10 m | −0.41 | 0.24 | −1.75 | 0.094 | Prox 10 m | −0.02 | 0.09 | −0.26 | 0.799 |
| Prox 20 m | −0.29 | 0.24 | −1.23 | 0.232 | Prox 20 m | −0.04 | 0.09 | −0.45 | 0.655 |
| Forest carabids, number of species (QP) | | | | | Forest plants, number of species | | | | |
| (Intercept) | 0.85 | 0.20 | 4.17 | 0.000 | (Intercept) | 2.77 | 0.59 | 4.71 | 0.001 |
| Treat 06/07 | −0.31 | 0.25 | −1.22 | 0.238 | Treat 06/07 | −0.09 | 0.61 | −0.14 | 0.890 |
| Treat 08/09 | −0.06 | 0.21 | −0.29 | 0.773 | Treat 08/09 | −0.50 | 0.51 | −0.98 | 0.337 |
| Prox 10 m | 0.17 | 0.21 | 0.78 | 0.443 | Prox 10 m | 0.20 | 0.50 | 0.40 | 0.692 |
| Prox 20 m | 0.24 | 0.21 | 1.15 | 0.264 | Prox 20 m | −0.10 | 0.50 | −0.20 | 0.843 |
| Generalist carabids, number of species | | | | | Generalist plants, number of speies | | | | |
| (Intercept) | 4.00 | 0.86 | 4.64 | 0.002 | (Intercept) | 7.50 | 1.01 | 7.44 | 0.000 |
| Treat 06/07 | 0.12 | 0.90 | 0.14 | 0.892 | Treat 06/07 | 0.66 | 1.18 | 0.56 | 0.579 |
| Treat 08/09 | 0.58 | 0.75 | 0.79 | 0.441 | Treat 08/09 | 0.89 | 0.99 | 0.91 | 0.374 |
| Prox 10 m | −0.20 | 0.73 | −0.27 | 0.787 | Prox 10 m | −0.90 | 0.98 | −0.92 | 0.369 |
| Prox 20 m | 0.40 | 0.73 | 0.55 | 0.590 | Prox 20 m | −0.40 | 0.98 | −0.41 | 0.688 |
| Open-habitat carabids, number of species | | | | | Open-habitat plants, number of species | | | | |
| (Intercept) | 0.37 | 0.20 | 1.84 | 0.084 | (Intercept) | 3.83 | 0.65 | 5.87 | 0.000 |
| Treat 06/07 | 0.81 | 0.25 | 3.17 | 0.004 | Treat 06/07 | −0.10 | 0.86 | −0.12 | 0.909 |
| Treat 08/09 | 0.24 | 0.22 | 1.11 | 0.277 | Treat 08/09 | 0.29 | 0.75 | 0.38 | 0.705 |
| Prox 10 m | −0.40 | 0.22 | −1.81 | 0.085 | Prox 10 m | −0.60 | 0.80 | −0.75 | 0.460 |
| Prox 20 m | −0.30 | 0.22 | −1.36 | 0.190 | Prox 20 m | −1.10 | 0.80 | −1.38 | 0.181 |

Independent or explanatory variables are model intercept, treatment (Treat; undergrowth not removed, or removed in the winter of 2006/2007 or 2008/2009) and edge proximity (Prox; 0, 10 or 20 m from stand edge toward the patch interior). Effect, estimated effect; SE, standard error for Effect; t shows test statistics for untransformed data unless specified otherwise (SQRT, square-root transformation or QP, quasi-Poisson error distribution, were applied to improve normality); P = probability for no effect. In all analyses site was a random variable to account for spatial autocorrelation. Sample size = 30. For details, see text.

species, *O. acetocella* had significantly higher cover at 20 m from than at the edge, whereas *R. saxatilis* peaked marginally at 10 m and *T. europaea* also marginally at 20 m.

In our exploration of possible interactions between treatment and edge proximity, most correlations in untreated or in treated stands were weak and non-significant (Table 4).

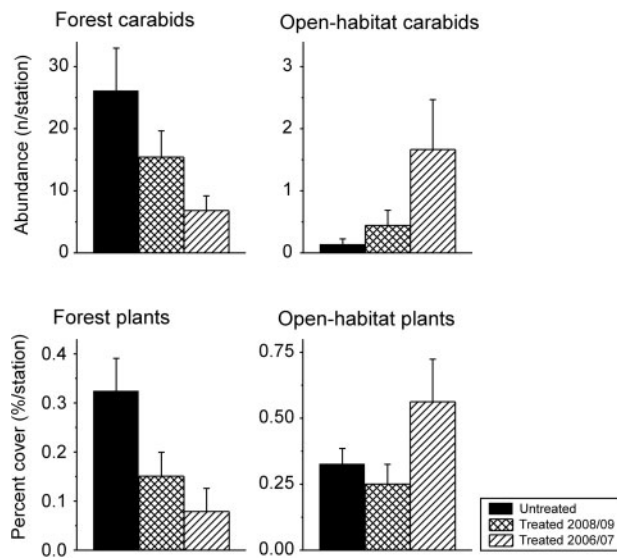


Figure 2: Column plots (mean + SE) for the number of individuals of forest and open-habitat associated carabids (top) and the percent cover of forest and open-habitat plants (bottom). See Table 2 for statistics.

However, *P.oblongopunctatus* increased marginally toward forest interior in treated but not in untreated stands, and *C.micropterus* and *L.terminatus* showed similar trends in untreated but not in treated stands.

Community responses to the undergrowth removal and edge proximity

Two-dimensional NMDS solutions had very high stress for both carabids and plants (>0.24 ; not shown), so we report 3D solutions; final stress for both carabids and plants was 0.15. Carabid samples from untreated stands, and stands treated in the winters of 2006/2007 or 2008/2009, did not differ systematically from each other (permutational multivariate ANOVA; $F = 1.14$, $R^2 = 0.08$, $P = 0.875$) and the samples were also similar with respect to edge proximity ($F = 0.57$, $R^2 = 0.04$, $P = 0.869$). Plant community, however, showed a significant treatment response ($F = 1.82$, $R^2 = 0.12$, $P = 0.029$; Fig. 4) but did not respond to the edge proximity ($F = 0.60$, $R^2 = 0.04$, $P = 0.923$).

Discussion

Strong effects of undergrowth removal on carabids and plants

We found that the carabid diversity—as measured by Shannon-Wiener and Simpson indices—was affected by the understory removal, implying rapid alteration in the community structure. This finding is in line with many species-richness responses to logging (e.g. Kuuluvainen, Tahvonen, and Aakala 2012; Nascimbene, Thor, and Nimis 2013; Fedrowitz et al. 2014) but apparently ephemeral, as the difference was significant for stands treated in the winter of 2008/2009 (treated half a year earlier) but not for 2006/2007 (2.5 years earlier; see discussion below). The lack of a treatment response in plant diversity, on the other hand, may have resulted from the relatively short post-treatment period (Bergstedt and Milberg 2001) and/or urbanization-related, historical landscape-scale change in the plant community (Niemelä et al. 2011). Thus, urban species communities may not consistently host fewer species than

rural communities do (Niemelä and Kotze 2009), but they may host fewer habitat specialists (Magura, Lövei, and Tóthmérész 2010; Vallet et al. 2010). Indeed, our data consisted mostly of succession generalists or open-habitat associated species, and specialists of mature or old-growth forests were missing (Appendix). These patterns suggest community homogenization, specialist loss and generalist/opportunist increase in urban ecosystems (Venn, Kotze, and Niemelä 2003; McKinney 2006, 2008).

We also found that the groups of forest plants and forest carabids were less abundant, and open-habitat carabids (but not open-habitat plants) were more abundant and more speciose, in treated than in untreated stands. These findings are in line with our predictions based on responses of these groups to logging (see Introduction section). Moreover, these responses were strongest in stands treated in the winter of 2006/2007, which suggests a 1–3 years response delay, as earlier reported for several forest plants by, e.g. Bergstedt and Milberg (2001) for logging and Ilsson et al. (2006) for windstorm disturbance. Both positive and negative responses to urban stressors are common (e.g. Jones and Leather 2012), and the opposite responses of forest and open-habitat species groups may have resulted in the lack of a detectable response in the total numbers of species. Contrasting responses between ecological groups of species nevertheless reflect the importance of examining different elements of communities rather than just the overall number of species in management and conservation assessments (e.g. Koivula 2011).

At the species level, the plants *M.bifolium* responded positively, and *O.acetocella* and *V.myrtillus* responded negatively, to the undergrowth removal in both the 2006/2007 and 2008/2009 stands. For *M.bifolium*, the positive response may be considered surprising, as the species is associated with shady conditions (Kujala 1926) but may be explained through competition. Thus, the species may have taken the advantage of undergrowth removal per se and negative responses of some common plants to it, such as *V.myrtillus*. The negative responses of *O.acetocella* and *V.myrtillus* are in line with previous studies on the effects of stand-replacing disturbances, such as clear-cutting and windstorm (Attegrim and Sjöberg 1996; Jalonen and Vanha-Majamaa 2001; Ilsson et al. 2006; Johnson, Strengbom, and Kouki 2014; Tonteri et al. 2016; but see Nybakken, Selås, and Ohlson 2013; Eldegard et al. 2019). All three species may have responded to treatment-caused alterations in shade and/or micro-climate, including variation in wind, temperature and moisture (e.g. Chen et al. 1999; Harper et al. 2015). We did not measure these factors or soil qualities, but further research on these and on inter-specific interactions (competition, pollination, predation and herbivory) would shed light on mechanisms behind the detected responses.

Several other carabid and plant species responded to the undergrowth removal only in stands treated in 2006/2007, suggesting a response delay, or only in stands treated in 2008/2009, suggesting an ephemeral response. We found evidence for a time lag in the treatment response of the carabid *L.terminatus*. Time lags in biological responses to habitat change are common and range from days to tens of years (e.g. Vellend et al. 2006) and, regarding logging, can be at least a few years in carabids and other arthropods (e.g. Spence et al. 1996; Vance and Nol 2003; Matveinen-Huju et al. 2006; Koivula et al. 2019). Moreover, some carabid responses were apparently ephemeral: *C.nemorialis* responded positively, and *P.oblongopunctatus* responded negatively, to the treatment in the 2008/2009 but not in the 2006/2007 stands. The former is considered a generalist of open

Table 3: GLMM summary for the abundance of carabid beetle species and percent cover of vascular plant species sampled in 10 forest stands in Jyväskylä, Finland

| Variable | Effect | SE | t | P | Variable | Effect | SE | t | P |
|--------------------------------|--------|------|-------|-------|-------------------------------|--------|------|-------|-------|
| <i>C.micropterus</i> (SQRT-QP) | | | | | <i>G.sylvaticum</i> (SQRT) | | | | |
| (Intercept) | −0.49 | 0.47 | −1.04 | 0.312 | (Intercept) | 0.11 | 0.04 | 2.53 | 0.027 |
| Treat 06/07 | −0.24 | 0.53 | −0.45 | 0.658 | Treat 06/07 | 0.00 | 0.05 | 0.02 | 0.982 |
| Treat 08/09 | 0.06 | 0.41 | 0.16 | 0.877 | Treat 08/09 | −0.11 | 0.05 | −2.49 | 0.020 |
| Prox 10 m | 0.80 | 0.52 | 1.53 | 0.141 | Prox 10 m | 0.01 | 0.05 | 0.27 | 0.789 |
| Prox 20 m | 0.86 | 0.52 | 1.65 | 0.114 | Prox 20 m | 0.00 | 0.05 | 0.01 | 0.992 |
| <i>C.nemorialis</i> | | | | | <i>M.bifolium</i> (SQRT) | | | | |
| (Intercept) | 5.30 | 1.92 | 2.76 | 0.017 | (Intercept) | 0.10 | 0.04 | 2.83 | 0.011 |
| Treat 06/07 | −0.13 | 2.36 | −0.05 | 0.958 | Treat 06/07 | 0.13 | 0.05 | 2.80 | 0.010 |
| Treat 08/09 | 3.75 | 1.99 | 1.88 | 0.072 | Treat 08/09 | 0.07 | 0.04 | 1.89 | 0.071 |
| Prox 10 m | −2.20 | 2.01 | −1.09 | 0.287 | Prox 10 m | −0.03 | 0.04 | −0.74 | 0.468 |
| Prox 20 m | 1.10 | 2.01 | 0.55 | 0.591 | Prox 20 m | 0.03 | 0.04 | 0.71 | 0.488 |
| <i>L.terminatus</i> (SQRT) | | | | | <i>O.acetocella</i> (SQRT-QP) | | | | |
| (Intercept) | 1.25 | 0.36 | 3.43 | 0.002 | (Intercept) | 0.20 | 0.06 | 3.42 | 0.011 |
| Treat 06/07 | −0.87 | 0.48 | −1.80 | 0.084 | Treat 06/07 | −0.21 | 0.06 | −3.74 | 0.001 |
| Treat 08/09 | −0.26 | 0.42 | −0.61 | 0.548 | Treat 08/09 | −0.10 | 0.05 | −2.11 | 0.046 |
| Prox 10 m | 0.44 | 0.45 | 0.98 | 0.335 | Prox 10 m | 0.03 | 0.04 | 0.63 | 0.537 |
| Prox 20 m | 0.62 | 0.45 | 1.39 | 0.176 | Prox 20 m | 0.10 | 0.04 | 2.32 | 0.031 |
| <i>P.melanarius</i> | | | | | <i>R.saxatilis</i> (SQRT) | | | | |
| (Intercept) | 5.71 | 3.01 | 1.90 | 0.073 | (Intercept) | 0.12 | 0.06 | 2.04 | 0.055 |
| Treat 06/07 | −4.83 | 4.76 | −1.02 | 0.323 | Treat 06/07 | 0.20 | 0.07 | 2.74 | 0.013 |
| Treat 08/09 | 2.39 | 3.25 | 0.73 | 0.472 | Treat 08/09 | −0.06 | 0.07 | −0.84 | 0.409 |
| Prox 10 m | −1.25 | 3.69 | −0.34 | 0.738 | Prox 10 m | 0.14 | 0.07 | 1.95 | 0.066 |
| Prox 20 m | −0.38 | 3.69 | −0.10 | 0.920 | Prox 20 m | −0.08 | 0.07 | −1.17 | 0.256 |
| <i>P.oblongopunctatus</i> | | | | | <i>S.aucuparia</i> (SQRT) | | | | |
| (Intercept) | 9.93 | 5.58 | 1.78 | 0.110 | (Intercept) | 0.08 | 0.06 | 1.30 | 0.232 |
| Treat 06/07 | −3.41 | 6.32 | −0.54 | 0.595 | Treat 06/07 | −0.02 | 0.06 | −0.33 | 0.748 |
| Treat 08/09 | −10.17 | 5.28 | −1.93 | 0.067 | Treat 08/09 | 0.11 | 0.05 | 2.20 | 0.038 |
| Prox 10 m | 3.60 | 5.23 | 0.69 | 0.500 | Prox 10 m | 0.04 | 0.05 | 0.90 | 0.377 |
| Prox 20 m | 9.00 | 5.23 | 1.72 | 0.101 | Prox 20 m | −0.01 | 0.05 | −0.27 | 0.791 |
| <i>T.secalis</i> (SQRT) | | | | | <i>T.europaea</i> (SQRT) | | | | |
| (Intercept) | 0.95 | 0.47 | 2.01 | 0.055 | (Intercept) | 0.06 | 0.02 | 2.79 | 0.022 |
| Treat 06/07 | −0.75 | 0.63 | −1.19 | 0.244 | Treat 06/07 | 0.01 | 0.02 | 0.28 | 0.780 |
| Treat 08/09 | 0.56 | 0.55 | 1.03 | 0.313 | Treat 08/09 | −0.02 | 0.02 | −1.21 | 0.240 |
| Prox 10 m | 0.31 | 0.58 | 0.53 | 0.599 | Prox 10 m | 0.03 | 0.02 | 1.61 | 0.123 |
| Prox 20 m | 0.29 | 0.58 | 0.49 | 0.626 | Prox 20 m | 0.03 | 0.02 | 1.73 | 0.099 |
| <i>C.arundinaceus</i> | | | | | <i>V.myrtillus</i> (SQRT) | | | | |
| (Intercept) | 0.10 | 0.05 | 1.88 | 0.104 | (Intercept) | 0.23 | 0.08 | 2.95 | 0.017 |
| Treat 06/07 | 0.05 | 0.05 | 1.02 | 0.321 | Treat 06/07 | −0.14 | 0.08 | −1.74 | 0.094 |
| Treat 08/09 | −0.02 | 0.05 | −0.40 | 0.692 | Treat 08/09 | −0.15 | 0.07 | −2.18 | 0.040 |
| Prox 10 m | 0.01 | 0.05 | 0.32 | 0.755 | Prox 10 m | 0.06 | 0.07 | 0.93 | 0.365 |
| Prox 20 m | 0.00 | 0.05 | 0.01 | 0.991 | Prox 20 m | 0.02 | 0.07 | 0.23 | 0.818 |
| <i>D.flexuosa</i> (SQRT) | | | | | | | | | |
| (Intercept) | 0.13 | 0.05 | 2.62 | 0.015 | | | | | |
| Treat 06/07 | −0.03 | 0.06 | −0.42 | 0.677 | | | | | |
| Treat 08/09 | 0.00 | 0.06 | −0.03 | 0.978 | | | | | |
| Prox 10 m | −0.03 | 0.06 | −0.52 | 0.606 | | | | | |
| Prox 20 m | −0.04 | 0.06 | −0.72 | 0.477 | | | | | |

Independent or explanatory variables are model intercept, treatment (Treat; undergrowth not removed, or removed in the winter of 2006/2007 or 2008/2009) and edge proximity (Prox; 0, 10 or 20 m from stand edge toward the patch interior). Effect, estimated effect; SE, standard error for Effect; t shows test statistics for untransformed data unless specified otherwise (SQRT, square-root transformation and/or QP, quasi-Poisson error distribution, were applied to improve normality); P, probability for no effect. In all analyses site was a random variable to account for spatial autocorrelation. Sample size = 30 except was 24 for *P.melanarius*, *C.arundinaceus* and *R.saxatilis*. See text for details.

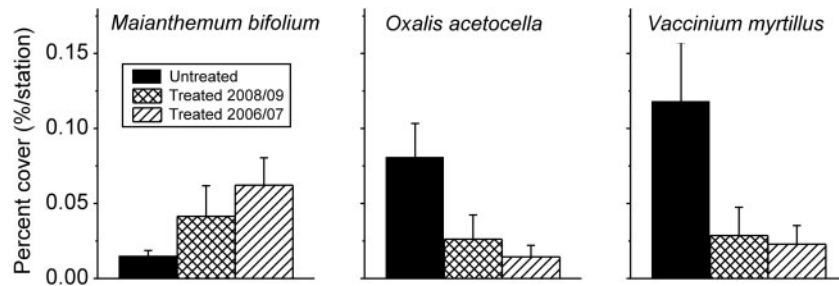
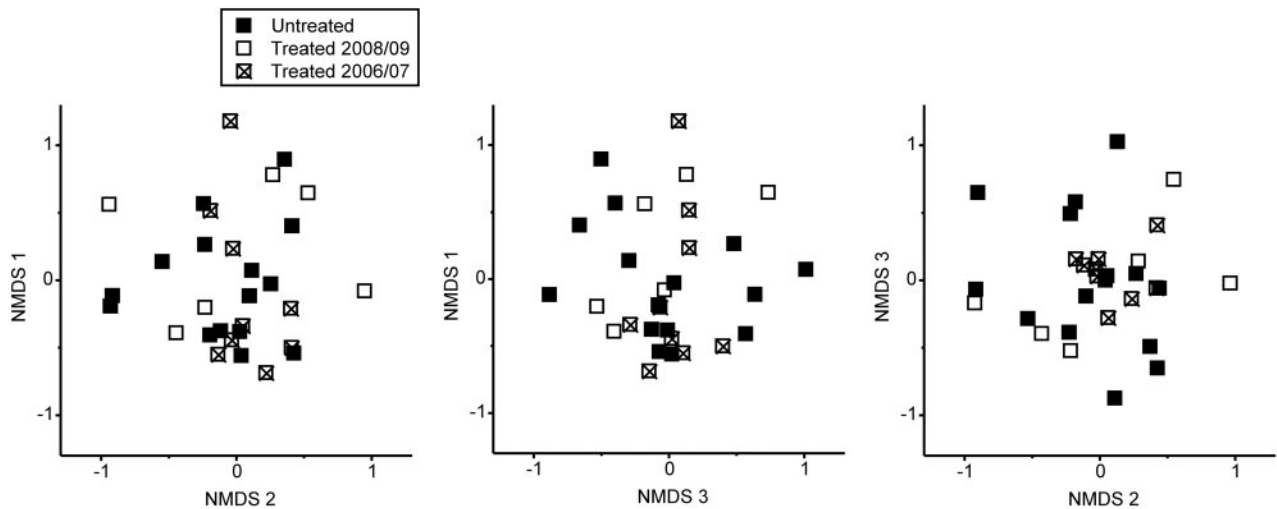
habitats and light forests, whereas the latter occupies different kinds of forest (Lindroth 1985, 1986; Niemelä, Koivula, and Kotze 2007). These responses may result from rapid changes in vegetation (see discussion above), which in turn affects the forest-floor micro-climate, shade and, indirectly, food items of these mostly predatory beetles (Thiele 1977; Kotze et al. 2011). In our study, some changes in vegetation indeed seemed short

lived: the covers of *S.aucuparia* and *G.sylvaticum* were lower only in stands treated in the winter of 2008/2009 than in untreated stands. *Sorbus aucuparia* was targeted by the undergrowth removals but may have quickly recovered due to sprouting (Zywiec and Ledwon 2008), explaining the cover similarity in untreated stands and in stands treated in 2006/2007. *Geranium sylvaticum*, on the other hand, might have temporarily suffered

Table 4: Correlations between ground beetles or vascular plants and edge distance in five untreated and five treated (undergrowth removed) stands in Jyväskylä, Finland, using Spearman rank correlation (rho coefficients shown; for details, see text)

| Group | Untreated | Treated | Group/species | Untreated | Treated |
|--|-----------|---------|---------------------------|-----------|---------|
| Carabids, number of species | -0.10 | 0.42 | <i>C.micropterus</i> | 0.47 | 0.11 |
| Carabids, diversity (H) | -0.15 | 0.20 | <i>C.nemoralis</i> | -0.21 | 0.25 |
| Carabids, evenness (J) | -0.36 | 0.02 | <i>L.terminatus</i> | 0.46 | 0.14 |
| Forest carabids, abundance | 0.23 | 0.40 | <i>P.melanarius</i> | -0.24 | 0.00 |
| Generalist carabids, abundance | -0.18 | 0.26 | <i>P.oblongopunctatus</i> | 0.17 | 0.45 |
| Open-habitat carabids, abundance | -0.24 | -0.27 | <i>T.secalis</i> | 0.07 | 0.18 |
| Forest carabids, number of species | 0.14 | 0.28 | <i>C.arundinaceus</i> | 0.18 | 0.12 |
| Generalist carabids, number of species | -0.20 | 0.31 | <i>D.flexuosa</i> | -0.23 | -0.15 |
| Open-habitat carabids, number of species | -0.24 | -0.27 | <i>G.sylvaticum</i> | 0.04 | -0.02 |
| Vascular plants, number of species | -0.08 | -0.17 | <i>M.bifolium</i> | 0.34 | 0.11 |
| Vascular plants, diversity (H) | 0.09 | -0.05 | <i>O.acetocella</i> | 0.37 | 0.29 |
| Vascular plants, evenness (J) | 0.00 | 0.04 | <i>R.saxatilis</i> | -0.30 | -0.18 |
| Forest plants, % cover | 0.34 | -0.02 | <i>S.aucuparia</i> | -0.07 | -0.14 |
| Generalist plants, % cover | -0.21 | -0.13 | <i>T.europaea</i> | 0.11 | 0.39 |
| Open-habitat plants, % cover | 0.00 | -0.08 | <i>V.myrtillus</i> | -0.09 | 0.19 |
| Forest plants, number of species | 0.06 | -0.05 | | | |
| Generalist plants, number of species | -0.22 | -0.18 | | | |
| Open-habitat plants, number of species | -0.22 | -0.26 | | | |

None were significant, but coefficients >0.44 are marginally significant.

**Figure 3:** The percent cover (mean + SE) of three plant species showing significant treatment responses. See Table 3 for statistics.**Figure 4:** Three-dimensional NMDS plot for vascular plants. Different combinations of the three dimensions of sample scores are shown (NMDS 1–2, 1–3 and 2–3). Solid black symbols indicate sampling stations of untreated stands; cross-center symbols indicate stations of stands treated in the winter of 2008/2009; and hollow symbols indicate stations of stands treated in the winter of 2006/2007. Stations at different edge distances were similar but the three treatment categories differed significantly (see text).

from trampling by personnel doing the undergrowth removals (Hamberg et al. 2008, 2010). Thus, the leaf area but not the number of specimens might have been impacted, allowing a rapid recovery after trampling (Thurston and Reader 2001). Comparisons between counts of specimens and percent covers would confirm this hypothesis; we measured only percent cover.

Small effects of edge proximity on carabids and plants

Edge proximity impacted carabids and plants only little, but open-habitat carabids were less abundant and speciose, and the plants *O.acetocella*, *T.europaea* and *R.saxatilis* had higher cover, at 10–20 m from than right at the edge. Changes within the first few tens of meters from the forest edge toward interior have earlier been reported for many taxa, including plants and carabids (e.g. Heliölä, Koivula, and Niemelä 2001; Honnay, Verheyen, and Hermy 2002; Huggard and Vyse 2002; Gallé and Fehér 2006; Hamberg et al. 2008, 2010; Malmivaara-Lämsä et al. 2008). Micro-climatic conditions, including light, apparently differed sufficiently between the edge and the 10–20 m sampling stations to result in detectable responses (Matlack 1993).

The generally low magnitude of edge effects may have occurred because of a general loss of specialists in urban landscapes (see above), and/or the short edge gradients studied here. Edge effects, as reflected by altered species compositions, may reach 50 m into the interior in boreal and temperate forests (e.g. Esseen and Renhorn 1998; Honnay, Verheyen, and Hermy 2002; Hamberg et al. 2008). Thus, the studied 20 m may have only captured small responses of species that do not avoid open habitats but are just less abundant there—generalists to varying degrees. Yet another explanation is that many opportunistic species of surrounding habitats have evidently invaded the studied forest fragments (Burke and Goulet 1998). In total 14 out of the 23 species of plants classified as open-habitat associated occurred at least once in sampling stations that were 20 m from the edge.

Our edge explorations of untreated or treated stands also suggest minor effects (Table 4). All the correlation coefficients were relatively low, and only three were marginally significant, which could occur by chance alone. Three forest-associated carabid species showed edge responses: *P.oblongopunctatus* increased slightly toward forest interior in treated but not in untreated stands, whereas *C.micropterus* and *L.terminatus* showed similar trends in untreated but not in treated stands. Regarding the former species, the more extensive cover by bushes and small trees could mitigate edge effects in untreated stands (Didham and Lawton 1999; Niemelä, Koivula, and Kotze 2007). The intuitively surprising responses by *C.micropterus* and *L.terminatus*, on the other hand, might represent a ‘panic reaction’ to conditions turned unfavorable. Such reaction could result in higher mobility (Charrier, Petit, and Burel 1997) and consequently elevated catches of these beetles throughout the edge proximity gradient.

Community structure was little affected by undergrowth removal and edge proximity

In terms of the overall community structure, carabids did not respond to the treatment or to the 20-m edge gradient, whereas plants responded to the treatment. Even for plants, the samples of untreated and treated stands may be better described as being different on average rather than being distinctive (Fig. 4). Similarly, the soil microbial community in Finnish urban forests varied only little within 20 m, but was more different 50 m, from edge to interior (Malmivaara-Lämsä et al. 2008; see also Harper et al. 2005).

The ambiguity may have been because most species in our data were open-habitat associated, or generalists of habitat openness (Appendix; see discussion above) or, not mutually exclusively, the communities varied remarkably independent of the treatment or edge proximity. Indeed, 40 out of a total of 87 species occurred in only 1–2 stands so that 18 were found solely in treated and 13 solely in untreated stands. Moreover, 27 species occupied 3–5 stands, but only 3 of these were solely found in treated or in untreated stands. Another explanation for such difficult-to-predict occurrences of plants and carabids may be habitat heterogeneity (Honnay, Hermy, and Coppin 1999; Marchand and Houle 2006; Fuller, Oliver, and Leather 2008; see Table 1). Moreover, our stands could have hosted remnant populations of some species; partly for this reason, community heterogeneity may sometimes be higher in fragmented urban than in contiguous rural habitats (e.g. Klausnitzer and Richter 1983; Niemelä et al. 2002; Venn, Kotze, and Niemelä 2003; Magura, Tóthmérész, and Molnár 2008).

Conclusions

Our study is among the first that assesses the impact of undergrowth removal on ecological communities. From ecological, management and conservation perspectives, our most important finding was that species associated with closed-canopy conditions often suffered, and species thriving in open habitats benefited, from undergrowth removal and, to some degree, from edge proximity. Clearly, maintaining variation in urban forest management may generally support biodiversity of urban areas (Rydbeck and Falck 2000). For instance, variable retention potentially benefits species associated with shady conditions (Gustafsson et al. 2012; Fedrowitz et al. 2014), especially if applied to maintain and add structural heterogeneity, such as variation in tree densities and layers, and very large and dead trees (e.g. Craig and Macdonald 2009; Pinzon, Spence, and Langor 2012; Stokland, Siitonen, and Jonsson 2012; Suominen et al. 2015; Heikkala, Martikainen, and Kouki 2016; Joellsson et al. 2017). Based on present results, we also recommend managers of urban forests to assess the local fauna and flora before harvesting operations, such as undergrowth removal, to avoid potentially negative impacts on rare and threatened species requiring shady conditions.

Data availability

Data are available at Dryad repository. DOI: 10.5061/dryad.h15c4sp

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Supplementary data

Supplementary data are available at JUECOL online.

Conflict of interest statement. None declared.

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Appendix

Table A1: Species of carabid beetles and plants, collected from 10 forest stands in Jyväskylä, Finland, in alphabetical order

| Species | HAB | UN TOT | UN OCC | TR TOT | TR OCC | Species | HAB | UN TOT | UN OCC | TR TOT | TR OCC |
|--------------------------------|-----|-----------|-----------|-----------|-----------|-------------------------------|-----|-----------|-----------|-----------|-----------|
| Carabid beetles (1) | | | | | | Carabid beetles (2) | | | | | |
| <i>Amara communis</i> | o | 1 | 1 | 4 | 3 | <i>Loricera pilicornis</i> | g | 0 | 0 | 3 | 1 |
| <i>Amara lunicollis</i> | o | 0 | 0 | 1 | 1 | <i>Notiophilus biguttatus</i> | g | 1 | 1 | 1 | 1 |
| <i>Anisodactylus binotatus</i> | o | 0 | 0 | 1 | 1 | <i>Patrobus assimilis</i> | g | 12 | 3 | 35 | 3 |
| <i>Badister lacertosus</i> | g | 2 | 2 | 2 | 2 | <i>Patrobus atrorufus</i> | g | 11 | 6 | 17 | 5 |
| <i>Calathus melanocephalus</i> | o | 1 | 1 | 5 | 2 | <i>Pterostichus diligens</i> | g | 1 | 1 | 2 | 2 |
| <i>C.micropterus</i> | f | 42 | 10 | 33 | 10 | <i>P.melanarius</i> | g | 69 | 7 | 62 | 8 |
| <i>Carabus hortensis</i> | f | 82 | 6 | 13 | 2 | <i>Pterostichus niger</i> | g | 10 | 6 | 7 | 4 |
| <i>C.nemoralis</i> | g | 74 | 11 | 107 | 14 | <i>Pterostichus nigrita</i> | g | 1 | 1 | 0 | 0 |
| <i>Cychnus caraboides</i> | f | 3 | 1 | 0 | 0 | <i>P.oblongopunctatus</i> | f | 100 | 12 | 212 | 11 |
| <i>Harpalus laevipes</i> | g | 2 | 1 | 2 | 1 | <i>Pterostichus strenuus</i> | g | 6 | 5 | 6 | 3 |
| <i>Harpalus tardus</i> | o | 0 | 0 | 3 | 2 | <i>Synuchus vivalis</i> | g | 10 | 6 | 15 | 7 |
| <i>Leistus ferrugineus</i> | g | 23 | 5 | 9 | 5 | <i>Trechus rivularis</i> | f | 1 | 1 | 0 | 0 |
| <i>L.terminatus</i> | f | 52 | 13 | 33 | 10 | <i>T.secalis</i> | g | 43 | 9 | 47 | 8 |
| Vascular plants (1) | | | | | | Vascular plants (2) | | | | | |
| <i>Acer platanoides</i> | f | 0.00 | 0 | 0.33 | 4 | <i>Lonicera xylosteum</i> | f | 3.00 | 2 | 0.00 | 0 |
| <i>Actaea spicata</i> | f | 0.00 | 0 | 1.33 | 1 | <i>Luzula luzuloides</i> | g | 4.67 | 1 | 0.00 | 0 |
| <i>Aegopodium podagraria</i> | f | 2.32 | 4 | 3.59 | 4 | <i>Luzula pilosa</i> | g | 0.02 | 1 | 0.13 | 3 |
| <i>Agrostis capillaris</i> | o | 0.47 | 4 | 0.00 | 0 | <i>M.bifolium</i> | g | 1.49 | 12 | 4.98 | 14 |
| <i>Alchemilla</i> sp | o | 0.10 | 3 | 0.00 | 0 | <i>Melampyrum pratense</i> | g | 0.49 | 3 | 0.63 | 3 |
| <i>Anthriscus sylvestris</i> | o | 0.13 | 2 | 0.70 | 4 | <i>Melampyrum sylvaticum</i> | g | 0.86 | 5 | 0.34 | 2 |
| <i>Artemisia vulgaris</i> | o | 0.08 | 2 | 0.11 | 1 | <i>Melica nutans</i> | g | 0.14 | 2 | 0.71 | 6 |
| <i>Athyrium filix-femina</i> | g | 1.17 | 5 | 0.07 | 1 | <i>Milium effusum</i> | f | 0.00 | 0 | 1.11 | 1 |
| <i>C.arundinaceus</i> | o | 9.56 | 11 | 8.31 | 9 | <i>Nardus stricta</i> | o | 0.00 | 0 | 0.22 | 1 |
| <i>Campanula rotundifolia</i> | o | 0.00 | 0 | 0.01 | 1 | <i>O.acetocella</i> | f | 8.07 | 13 | 2.16 | 9 |
| <i>Carex acuta</i> | r | 0.02 | 1 | 0.00 | 0 | <i>Paris quadrifolia</i> | f | 0.57 | 3 | 0.75 | 7 |
| <i>Carex canescens</i> | r | 0.22 | 1 | 0.00 | 0 | <i>Phleum pratense</i> | o | 0.02 | 1 | 0.00 | 0 |
| <i>Cirsium arvense</i> | o | 0.16 | 2 | 0.00 | 0 | <i>Poa chaixii</i> | o | 0.14 | 2 | 0.00 | 0 |
| <i>Convallaria majalis</i> | g | 2.96 | 4 | 1.19 | 2 | <i>Ranunculus acris</i> | o | 0.04 | 1 | 0.00 | 0 |
| <i>Deschampsia cespitosa</i> | o | 0.37 | 3 | 0.00 | 0 | <i>Ranunculus repens</i> | o | 1.00 | 2 | 0.80 | 3 |
| <i>D.flexuosa</i> | o | 2.60 | 7 | 2.36 | 8 | <i>Ribes rubrum</i> | g | 0.16 | 2 | 0.56 | 3 |
| <i>Dryopteris carthusiana</i> | g | 0.00 | 0 | 0.44 | 1 | <i>Rubus idaeus</i> | g | 3.24 | 7 | 2.74 | 5 |
| <i>Dryopteris filix-mas</i> | g | 0.90 | 2 | 1.24 | 3 | <i>R.saxatilis</i> | g | 2.42 | 9 | 7.97 | 9 |
| <i>Epilobium angustifolium</i> | o | 0.00 | 0 | 0.51 | 4 | <i>Scrophularia nodosa</i> | g | 0.13 | 1 | 0.00 | 0 |
| <i>Epilobium montanum</i> | g | 0.09 | 4 | 0.03 | 1 | <i>Solidago virgaurea</i> | g | 0.01 | 1 | 0.00 | 0 |
| <i>Equisetum pratense</i> | f | 2.36 | 4 | 2.09 | 3 | <i>S.aucuparia</i> | g | 2.98 | 6 | 3.21 | 14 |
| <i>Equisetum sylvaticum</i> | g | 0.31 | 3 | 0.33 | 1 | <i>Taraxacum</i> sp | o | 0.00 | 0 | 0.04 | 4 |
| <i>Festuca ovina</i> | o | 0.93 | 1 | 0.00 | 0 | <i>T.europaea</i> | g | 1.07 | 11 | 0.73 | 12 |
| <i>Fragaria vesca</i> | o | 2.17 | 3 | 0.23 | 5 | <i>Urtica dioica</i> | g | 0.04 | 1 | 0.12 | 2 |
| <i>Galium album</i> | o | 0.00 | 0 | 1.31 | 3 | <i>Urtica urens</i> | g | 1.12 | 2 | 0.00 | 0 |
| <i>G.sylvaticum</i> | g | 1.13 | 6 | 2.69 | 9 | <i>V.myrtillus</i> | f | 11.78 | 12 | 2.64 | 11 |
| <i>Geum urbanum</i> | g | 0.00 | 0 | 0.38 | 2 | <i>Vaccinium vitis-idaea</i> | g | 0.32 | 5 | 0.41 | 5 |
| <i>Gymnocarpium dryopteris</i> | g | 0.62 | 3 | 1.49 | 7 | <i>Veronica chamaedrys</i> | o | 0.53 | 3 | 0.29 | 3 |
| <i>Hieracium sylvatica</i> | g | 1.74 | 7 | 0.19 | 4 | <i>Veronica officinalis</i> | o | 0.06 | 2 | 0.08 | 1 |
| <i>Hypericum maculatum</i> | o | 0.05 | 1 | 0.30 | 3 | <i>Viola riviniana</i> | g | 0.24 | 6 | 0.18 | 6 |
| <i>Linnaea borealis</i> | f | 0.00 | 0 | 0.27 | 1 | | | | | | |

Columns show, from left to right, habitat association (HAB; f, closed forest; g, forest-open habitat generalist; and o, open habitat), total catch or average percent cover (TOT) and total number of occupied sampling stations (OCC; max 15) for untreated (UN) and treated (TR) stands.