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The Conservation Potential of Brook-side Key Habitats in Managed Boreal Forests

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Today, maintaining biodiversity is included in the targets of boreal forest management. A widespread approach in northern Europe is to identify and preserve woodland key habitats within managed forests. Woodland key habitats are expected to be patches that host populations of threatened and declining species, and the preservation of these patches is assumed to enable the persistence of the focal species in the landscape. In Finland, the criteria for selecting woodland key habitats are defined in the Finnish Forest Act, and the selection has been done by forest practitioners. Our objective was to determine whether the surroundings of boreal brooks and rivulets qualified as key habitats are truly different from brook-side habitats not granted the key habitat status, and whether the brook-side habitats of the two types differ from the forest matrix managed for timber production. We found that the two brook-side habitats were in most aspects rather alike but there was a difference in the composition of ground vegetation assemblages. In contrast, the control forests were distinct from the brook-sides in terms of dead wood, species richness and assemblages of polypores, species richness of epiphytic mosses, and the composition of beetle assemblages. We conclude that brook-sides in general provide an important habitat clearly diverging from the surrounding matrix but that the conservation value of the brook-sides granted the key habitat status may not be substantially larger than that of the brook-sides without the status.

Keywords dead wood, forest management, saproxylic species, vegetation

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1 Introduction

In boreal landscapes, there is an obvious confrontation between forest biodiversity and commercial forestry. Unfortunately, the status quo with the current forest management policy in boreal forests seems to be a constant increase of the number of threatened species (see Hanski 2000). However, since maintaining biodiversity is included in the targets of boreal forest management (Hansson 1997, Spence 2001), alternative forestry practices must be developed in response to the declining diversity. In northern Europe, one approach is to identify and maintain so called woodland key habitats in forests managed for timber production to enable threatened and declining species to persist in the landscape dominated by forest management. There exists slight variation between Fennoscandian and Baltic countries in details how the woodland key habitats are defined and protected (Nitare and Norén 1992, Aasaaren and Sverdrup-Thygeson 1994, Tenhola and Yrjönen 2000, Sohlberg 2004, Timonen et al. 2010), but the collective aim is to preserve those habitat patches in managed forests that are thought to be of greatest value from the standpoint of forest ecosystem and biodiversity.

In Finland, the current forest legislation, i.e. Finnish Forest Act (1093/1996), was developed and passed in 1996 and introduced in the beginning of 1997. The prime aim of the Act is to allow sustainable management and utilization of forests while safeguarding their biodiversity. In the Act, the primary means of promoting the biodiversity in the Finnish managed forests is the preservation of the woodland key habitats. Woodland key habitats are expected to be biodiversity hotspots, where the occurrence of threatened and demanding species is the most likely. Moreover, these habitats are assumed to be natural or natural-like and to possess permanent structural characteristics that can maintain populations of focal species. Furthermore, to become qualified as a Forest Act habitat by forestry practitioners, the habitat has to be clearly distinguishable from its surroundings and to be small in size. In Finland, 13 different habitat types considered as woodland key habitats have been listed, the most numerous being the immediate surroundings of boreal brooks or rivu-

lets (Yrjönen 2004, Kotiaho and Selonen 2006). The brook adjacent riparian ecotone is considered to be a vulnerable habitat type and the diversity of its species assemblages is often severely influenced by forest management (e.g. Swanson and Franklin 1992, Darveau et al. 1995, Machtans et al. 1996, Naiman and Decamps 1997, Lambert and Hannon 2000, Coroi et al. 2004, Hylander et al. 2005, Muir et al. 2006).

The function of key habitat based conservation has frequently been questioned (Hanski 2002, 2005). First, the Finnish Forest Act defines that all silvicultural practices on the key habitat must be carried out such that the valuable characteristics of the habitat are preserved (e.g. Tenhola and Yrjönen 2000), but this is an unfortunately ambiguous definition and it is lacking in directions and means. Second, rare and threatened species are assumed to occur in these habitats, but evidence is still scarce and controversial. In some studies, key habitats have been shown to host a high number of red-listed and indicator species (Gustafsson et al. 1999, Gustafsson 2002), while in other studies key habitats have not differed from other forest habitats (Sverdrup-Thygeson 2002, Gustafsson et al. 2004). However, when the current evidence is drawn together, there appears to be a slight difference between key habitats and the forest matrix in favour of key habitats (Timonen et al. 2011). Third, it is questionable whether such “precision conservation” can be efficient under the prevailing forest structure and dynamics because in boreal forests of northern Europe the whole forest matrix is altered: in the past disturbances occurred within the undisturbed landscape but today the undisturbed patches occur within the disturbed landscape.

In general, the long term efficiency of key habitat based biodiversity protection remains uncertain and poorly known (Spence 2001). To determine the conservation potential of key habitats, the first step is to determine whether these habitats differ from the surrounding matrix (i.e. managed forests). However, it is equally important to know whether the key habitats differ from a comparable habitat (i.e. structurally similar habitats that have not been granted the key habitat status). In our study, we focused on brook-side key habitats determined by the Finnish Forest Act. The aim was to determine 1) whether the brook-side key

habitats host higher species diversity or different species assemblages than other brook-side habitats, and 2) whether the brook-side key habitats or brook-sides in general differ from typical managed boreal coniferous forests.

2 Methods

The study was conducted in Central Finland on sites located within the south and middle boreal vegetation zones and two separate data sets were collected. The study sites were mature managed Norway spruce (*Picea abies*) dominated coniferous forests with some Scots pine (*Pinus sylvestris*) and occasional deciduous trees. The forests were owned and managed by forest companies. Also the identification and qualification of key habitats had been done by forest companies in years 2001–2002. The study sites had been under intensive management, i.e. classical rotation forestry, for several decades. Within the rotation, thinning of an even-aged stand is carried out 2–3 times and rotation ends up in the final felling or clear cutting.

For the first data set (Dataset 1), we selected 20 study sites in year 2002. The study sites were located within an 80×100 km area around the city of Jyväskylä (62°N, 25°E). Each study site was established along a separate brook, and the sites were separated from each other by several kilometres. On each site, we established three study plots. One plot was located on a brook-side key habitat, the second along the same brook on a brook-side habitat not classified as a key habitat (control brook-side), and the third in the nearby managed forest (control forest). The distance between adjacent plots within one study site was always less than 300 m. The size of each of the study plots was 0.1 ha: on the brook-side habitats a 20×50-m rectangle was established such that the longitudinal axis was centered on the brook (i.e. 10×50 m on each side of the brook), or if that was not possible, a 10 x 100-m rectangle was established along one side of the brook. In control forests, the study plot was always a 20×50-m rectangle.

Dead wood, epiphytic mosses, polyporous fungi and beetles were selected as response variables

in the study. All dead wood units with thick-end diameter over 5 cm were measured in May 2002. For each unit, we determined the thick-end diameter, length, tree species and decay stage (5-stage classification after Renvall 1995). The volume of dead wood was calculated by using simple geometrics and the volume of whole trunks by using the equations provided in Laasasenaho (1982). For each of the sites the volume of dead wood was transformed to cubic metres per hectare. Epiphytic moss species were inventoried in May 2002 from all standing trees from 50 cm above the ground up to 250 cm. Polypore inventories were conducted in Sept–Oct 2002 by checking all dead wood units and live tree trunks. If there were several fruiting bodies of the same polypore species on one piece of dead wood, it was considered as one occurrence. Beetles were captured from mid-May to mid-July 2002 with window traps that were set hanging from a wire between two trees. Three traps were set on each study plot and the data of the three traps were pooled for statistical analyses.

For the second dataset (Dataset 2), we selected 16 study sites in year 2003. The study sites were located within a 120-km radius from the city of Jyväskylä and sites were separate from the sites in dataset 1. Of the 16 study sites, eight of them were brook-side key habitats and eight brook-side habitats without the status (control brook-side). Each study site was established on a separate brook and key habitats and control brook-sides were spatially intermixed within the study area. On each study site, three orthogonal sampling lines from the brook edge were established at intervals of 15 meters. From each 5-meter long and one meter wide sampling line vascular plants and mosses of the ground layer were counted and identified to species, one square meter at a time. The data of the three lines were pooled for statistical analyses.

The differences between habitats in dead wood variables, in the numbers of individuals, and in the numbers and diversities of species were explored with ANOVA. In dataset 1 we used randomized blocks design entering site as a random factor to the analysis, and in dataset 2 each brook-side was treated as an independent sample. LSD test was used in multiple comparisons between habitats. Data distributions were normalized by log₁₀- or

Table 1. The results of between-habitat comparisons (LSD test) of dataset 1. MD =, mean difference = $a - b$.

	Key habitat ^a vs. control brook ^b		Key habitat ^a vs. control forest ^b		Control brook ^a vs. control forest ^b		SE
	MD	p	MD	p	MD	p	
Dead wood							
Number of pieces	0.102	0.242	0.319	0.001	0.217	0.016	0.086
Number of decaying stages	0.013	0.417	0.044	0.011	0.030	0.070	0.016
Total volume	0.094	0.194	0.165	0.026	0.071	0.326	0.071
Diversity of tree species	0.007	0.821	0.095	0.002	0.089	0.004	0.286
Number of tree species	-0.001	0.991	0.120	0.005	0.121	0.005	0.040
Proportion of deciduous trees	0.119	0.321	0.362	0.004	0.242	0.048	0.119
Polypores							
Number of species	0.027	0.649	0.160	0.010	0.133	0.029	0.059
Number of occurrences	0.085	0.316	0.233	0.009	0.148	0.087	0.084
Diversity	0.033	0.824	0.385	0.013	0.352	0.023	0.149
Epiphytic mosses							
Number of species	0.140	0.082	0.300	<0.001	0.160	0.047	0.078
Number of occurrences	0.177	0.074	0.381	<0.001	0.204	0.042	0.097

arcsin-transformation. Shannon-Wiener index was used as a measure of diversity. The analyses were performed with SPSS (version 14).

Analysis of Similarities (ANOSIM) was used to test whether the species composition differed between habitats. ANOSIM is a non-metric test based on distance measure and it uses the rank order of dissimilarity values, thus analogous to non-metric multidimensional scaling (NMDS). The polypore species compositions and tree species compositions of dead wood were further depicted by NMDS to illustrate the differences between habitats and to explore the associations between polypore species and deadwood variables. In addition, an indicator species analysis (Dufrene and Legendre 1997) was used to determine species characteristic of each habitat. This analysis gives an indicator value with statistics (Monte Carlo permutation) for each species. The indicator value (IV) is percent of perfect indication (i.e. $IV = 100\%$) and the value is verified by Monte Carlo permutation. ANOSIM was performed with PAST (version 2.09) (Hammer et al. 2001) and indicator species analysis and NMDS were carried out using PC-ORD (version 4.41).

3 Results

3.1 Dataset 1: Dead Wood

The amount of dead wood was relatively low on all sites, the average volume on the brook-side key habitat, the control brook-side and the control forest being 7.6 ± 5.9 , 5.7 ± 4.3 and 4.9 ± 4.2 m³/ha, respectively (mean \pm SD). Over all, the volume of dead wood tended to differ between habitats ($F_{2,38} = 2.701$, $p = 0.080$), and brook-side key habitat had more dead wood than control forest (Table 1). The average number of dead wood pieces was 42 per site. The number of dead wood pieces differed between habitats ($F_{2,38} = 7.203$, $p = 0.002$) such that both brook-side habitats had a higher number of dead wood pieces than control forest but the brook-side habitats did not differ from each other (Table 1). The dead wood pieces were rather small and the average thick-end diameter of dead wood in all was 8.7 cm. The average diameter did not differ between habitats ($F_{2,38} = 0.245$, $p = 0.784$). On the other hand, the average decay stage differed between habitats ($F_{2,38} = 3.794$, $p = 0.031$). The difference between habitats was due to the decay stage being higher on brook-side key habitat than in control forest (Table 1).

The number of dead wood tree species differed between habitats ($F_{2,38} = 6.003$, $p = 0.005$). Both

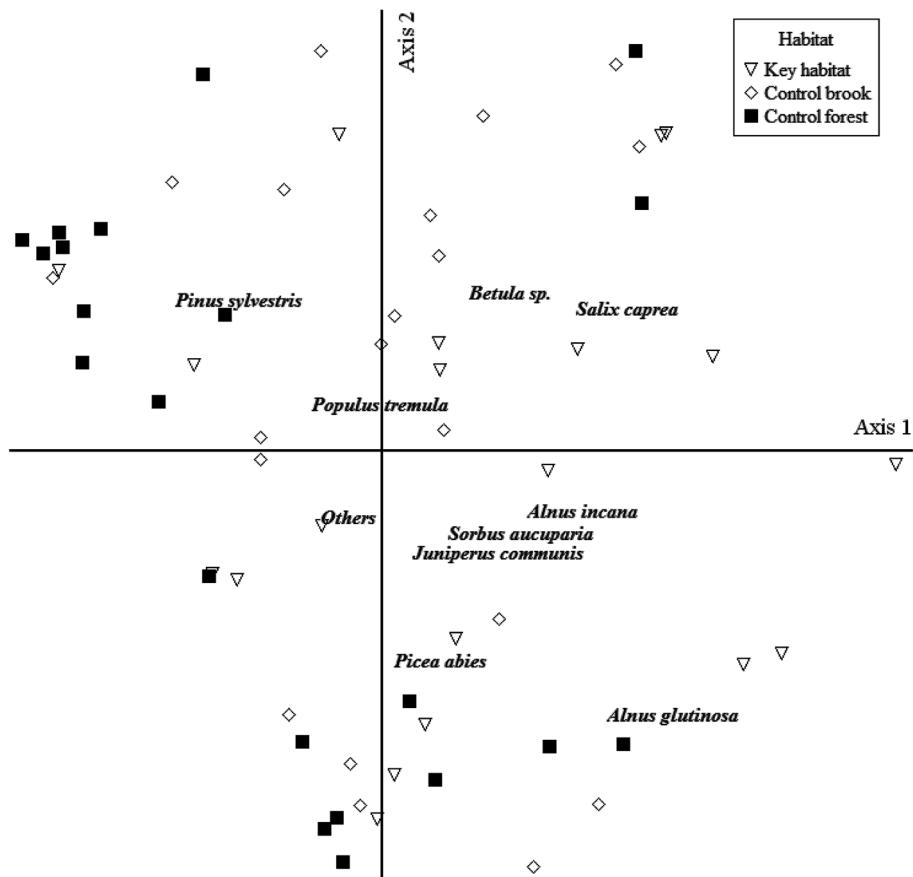


Fig. 1. NMDS ordination of the dead wood species composition.

brook-side habitats had more dead wood species than control forest while there was no difference between the brook-side habitats (Table 1). In addition, the species diversity of dead wood differed between habitats ($F_{2,38}=6.877$, $p=0.003$). Again, both brook-side habitats had higher diversity than control forest, but there was no difference between the brook-side habitats (Table 1). Also, the proportion of deciduous tree species differed between habitats ($F_{2,38}=4.823$, $p=0.014$). Control forest had less deciduous dead wood than both brook-side habitats, while brook-side key habitat and control brook-side did not differ from each other (Table 1).

According to the ANOSIM, the dead wood species composition of brook-side key habitat differed from that of control forest, while control

brook-side habitat tended to differ from control forest but there was no difference between the brook-side habitats (Table 2). In addition, the dead wood composition of brook-side key habitat was characterized by the presence of grey alder (*Alnus incana*) and goat willow (*Salix caprea*), while no specific indicator species were found for other habitats (Table 3). In the NMDS ordination space, control forests formed two distinct groups while there was large variation among the brook-side habitats (Fig. 1). According to the ordination, the main difference between habitats was that the dead wood of most of the control forests consisted almost exclusively of either Norway spruce or Scots pine while the brook-sides were characterized by a diverse deciduous admixture.

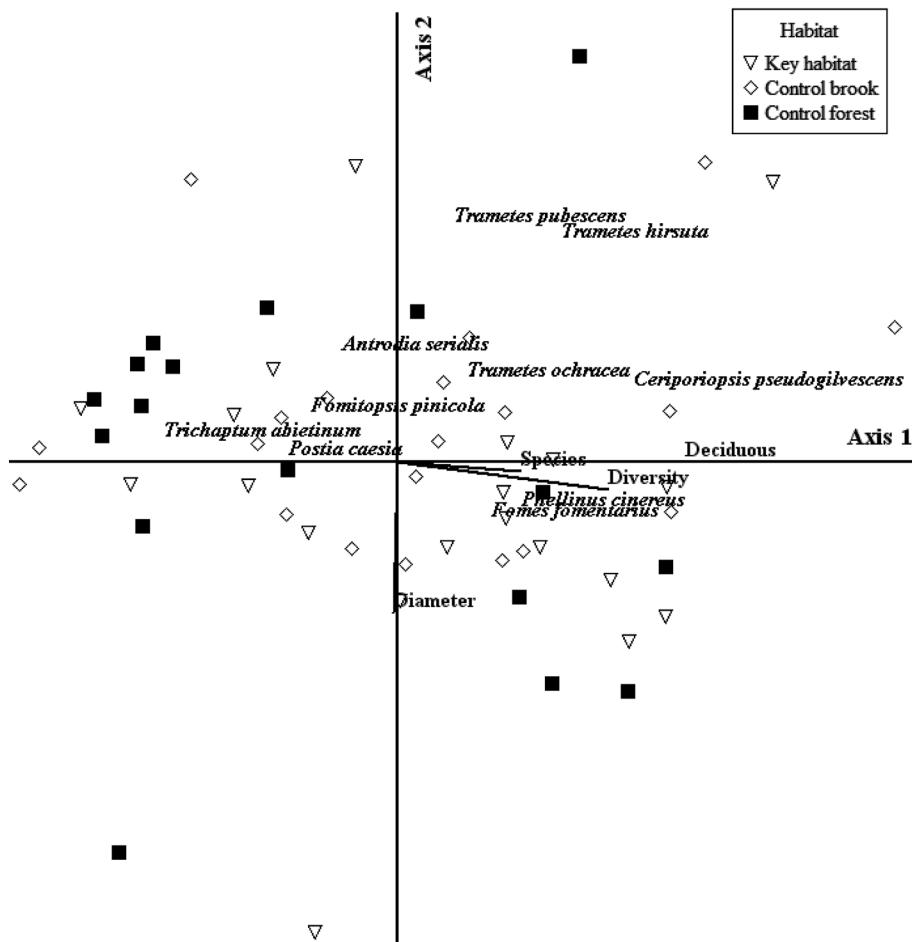


Fig. 2. NMDS ordination of polypore species composition. Dead wood variables have been included in the analysis as environmental variables. Only the polypore species and dead wood variables significantly correlating with either of the axes are presented in the figure. Deciduous = proportion of deciduous dead wood; Diameter = average diameter of dead wood; Diversity = diversity of dead wood; Species = number of dead wood tree species. See also Table 4.

3.2 Dataset 1: Polypores

In total, 53 polypore species were recorded. The number of polypore species differed between habitats ($F_{2,38}=4.277$, $p=0.021$). There were less polypore species in control forest than on both brook-side habitats (Table 1). Similarly, the number of polypore occurrences differed between habitats ($F_{2,38}=3.930$, $p=0.028$). Control forest had fewer occurrences than the brook-side key habitat (Table 1). Also the diversity of polypores differed between habitats ($F_{2,38}=4.137$,

$p=0.024$), the diversity being lower in control forest than on both brook-side habitats (Table 1). According to ANOSIM, the polypore species composition of brook-side key habitat differed from other habitats (Table 2), but no clear indicator species were found.

In the NMDS, a 2-dimensional solution was found (Fig. 2). Three dead wood variables (number and diversity of species and proportion of deciduous trees) were positively correlated with axis 1, and the average diameter of dead wood was negatively correlated with axis 2 (Table 4). Two

Table 2. Analysis of similarities (ANOSIM) of species composition between habitats. Vascular plant and moss comparisons are from dataset 2, thus including only brook-side habitats. The number of Monte Carlo permutations is 10000.

	Key habitat vs. control brook		Key habitat vs. control forest		Control brook vs. control forest	
	R	p	R	p	R	p
Deadwood	0.002	0.415	0.095	0.010	0.049	0.053
Polypores	-0.016	0.701	0.071	0.047	0.025	0.222
Epiphytic mosses	0.008	0.296	-0.028	0.597	-0.070	0.923
Saproxylic beetles	-0.009	0.584	-0.016	0.707	-0.032	0.891
Non-saproxylic beetles	-0.053	0.975	0.079	0.018	0.054	0.054
Vascular plants	0.438	<0.001				
Mosses	0.125	0.038				

polypore species that utilize deciduous dead wood were positively correlated with axis 1 while three species that prefer coniferous dead wood showed a negative correlation (Table 4). Three species that are often found on small diameter deciduous dead wood were positively correlated with axis 2 and one species that prefers large-diameter dead wood showed a negative correlation (Table 4). Control forest formed two separate groups in the NMDS ordination space while the brook-side habitats were intermixed along axis 1 and slightly separated from the control forest groups (Fig. 2).

3.3 Dataset 1: Epiphytic Moss Species

Only 7 species of epiphytic moss species were found. Nevertheless, the number of moss species differed between habitats ($F_{2,38}=7.381$, $p=0.002$). Also the number of occurrences of epiphytic moss species differed between habitats ($F_{2,38}=7.805$, $p=0.001$). The control forest had less species and occurrences than both brook-side habitats but there were no differences between the brook-side habitats (Table 1). However, there were no differences in the diversity of epiphytic moss species between the habitats ($F_{2,38}=0.612$, $p=0.549$) According to ANOSIM, there were no difference in the species composition between habitats (Table 2). No indicator species were found.

3.4 Dataset 1: Beetle Species

In total, 5978 beetle individuals representing 353 species were captured from which 4380 individuals and 165 species were classified as saproxylic. Among saproxylic beetles, the number of species, number of individuals and diversity did not differ between habitats (species, individuals and diversity, $F_{2,38}=0.558$, $p=0.577$, $F_{2,38}=0.950$, $p=0.296$ and $F_{2,38}=1.508$, $p=0.234$, respectively). Similarly, among non-saproxylic beetles, there were no differences between habitats (species, individuals and diversity, $F_{2,38}=1.459$, $p=0.245$, $F_{2,38}=1.418$, $p=0.255$ and $F_{2,38}=1.142$, $p=0.330$, respectively). The saproxylic species composition did not differ between habitats (Table 2), but there was a difference in the non-saproxylic species composition between brook-side key habitat and control forest (Table 2). In addition, a few indicator species were found among both beetle groups. All five saproxylic indicator species were indicating brook-side habitats, while indicator non-saproxylic species were found for all habitat types (Table 3).

3.5 Dataset 2: Vascular Plant and Moss Species

In total, 70 vascular plant species and 53 moss species were identified. The number of vascular plant and moss species did not differ between brook-side key habitat and control brook-side (vascular plants and mosses, $F_{1,16}=1.361$, $p=0.263$, and $F_{1,16}=0.114$, $p=0.740$, respectively). Likewise, there were no differences in

Table 3. The results of indicator species analysis. Species with significant indicator value (IV) and the habitat type they are indicative of are presented. The number of Monte Carlo permutations is 10000.

	Species	Habitat	IV	p
Deadwood	<i>Alnus incana</i>	Key habitat	46.2	0.042
	<i>Sorbus aucuparia</i>	Key habitat	32.4	0.014
Saproxyllic beetles	<i>Denticollis linearis</i>	Key habitat	31.0	0.025
	<i>Malthodes fuscus</i>	Control brook	24.0	0.027
	<i>Phloeotribus spinulosus</i>	Control brook	21.0	0.022
	<i>Pyropterus nigroruber</i>	Control brook	28.6	0.008
	<i>Rhizophagus dispar</i>	Key habitat	20.0	0.042
Non-saproxyllic beetles	<i>Atomaria pulchra</i>	Control forest	23.1	0.023
	<i>Cyphon coarctatus</i>	Control brook	20.0	0.029
	<i>Nicrophorus vespilloides</i>	Control brook	20.0	0.028
	<i>Sepedophilus littoreus</i>	Key habitat	39.8	0.024
Moss species	<i>Sphagnum angustifolium</i>	Key habitat	81.6	0.004
	<i>Rhizomnium punctatum</i>	Key habitat	67.4	0.045
Vascular plants	<i>Athyrium filix-femina</i>	Key habitat	77.2	0.007
	<i>Equisetum sylvaticum</i>	Key habitat	67.5	0.05
	<i>Maianthemum bifolium</i>	Key habitat	71.1	0.001
	<i>Oxalis acetosella</i>	Key habitat	71.6	0.014
	<i>Trientalis europaea</i>	Key habitat	78.5	0.003

Table 4. Pearson correlations between variables and ordination axes of the NMDS of polypore species composition. Only the polypore species significantly correlating with either of the axes are presented.

	Axis 1		Axis 2	
	r	p	R	p
Polypores				
<i>Fomes fomentarius</i>	0.360	0.006	-0.314	0.016
<i>Ceriporiopsis pseudogilvescens</i>	0.323	0.013	0.137	0.305
<i>Phellinus cinereus</i>	0.287	0.029	-0.158	0.236
<i>Trametes hirsuta</i>	0.204	0.125	0.372	0.004
<i>Trametes ochracea</i>	0.187	0.160	0.295	0.025
<i>Trametes pubescens</i>	0.072	0.591	0.406	0.002
<i>Antrodia serialis</i>	-0.100	0.455	0.264	0.045
<i>Postia caesia</i>	-0.283	0.031	0.004	0.976
<i>Fomitopsis pinicola</i>	-0.309	0.018	0.222	0.094
<i>Trichaptum abietinum</i>	-0.459	0.000	0.053	0.693
Dead wood variables				
Number of pieces	-0.066	0.623	0.052	0.698
Average diameter	-0.057	0.671	-0.314	0.016
Volume	-0.121	0.366	-0.215	0.105
Decay stage	0.151	0.258	0.148	0.268
Proportion of deciduous trees	0.435	0.001	0.011	0.935
Number of dead wood species	0.282	0.032	-0.083	0.536
Diversity of dead wood species	0.369	0.004	-0.132	0.323

the diversities of vascular plant and moss species between the brook side habitats (vascular plants and mosses, $F_{1,16}=1.966$, $p=0.183$, and $F_{1,16}=0.112$, $p=0.743$, respectively). However, vascular plant and moss species compositions differed between the two habitats (Table 2). In both cases, all indicator species were indicating key habitat, five species of vascular plants and two species of mosses (Table 3).

4 Discussion

Our objective was to determine whether brook-side key habitats host distinct species assemblages or higher diversity than other brook-side habitats not granted the key habitat status, and whether brook-side habitats of the two types differ from the matrix, i.e. typical managed boreal coniferous forest. The most obvious differences were found between brook-side habitats and control forest. Although there were also some differences between the brook-side habitats, in general they were rather alike. For instance, the dead wood composition of the brook-side habitats was rather similar, and polypores followed predictably the same pattern. However, the ground layer flora of key habitat was distinct from control brook-side.

Compared to control forest, the brook-side habitats were characterized by more diverse and abundant dead wood and wealth of deciduous dead wood. A similar pattern has also been found in other studies comparing brook-side key habitats to managed forests (Hottola and Siitonen 2008). Although the average dead wood volumes per habitat were rather low, they were nevertheless more than reported in the Finnish national forest inventory (NFI) (Finnish statistical yearbook... 2007). According to the NFI, the average volume in managed forests in southern Finland is 2.7 m³/ha (Finnish statistical yearbook... 2007). Therefore, brook-side habitats had double the average volume, but also control forest had more dead wood than an average managed forest. Between the brook-side habitats, the quality, quantity and composition of dead wood were rather alike, but the brook-side key habitat was characterized by particular deciduous dead wood, *Alnus incana*

and *Salix caprea*. Nevertheless, according to dead wood, all brook-side habitats appear to be equally valuable and divergent from the surrounding matrix. In this sense, every single brook-side habitat may have potential to host more dead wood dependent species than the surrounding managed forest and be of conservation value.

The diverse polypore species assemblage of brook-sides is likely to reflect the diversity of dead wood resources. Hottola and Siitonen (2008) found that total species richness of polypores, and in particular that of deciduous wood specialists, was higher on brook-side key habitats than in normal managed forests, and that the species richness was best explained by the volume and diversity of dead wood. We found that species richness, diversity and community composition of both brook-side habitats differed from control forest. The abundance of several species was associated to the abundance of deciduous dead wood or to high diversity of dead wood, both being important characteristics of brook-side habitats. No specific indicator species were found, which was likely due to the brook-side habitats being so similar in terms of species composition. Interestingly, we found that dead wood volume was not a major factor affecting the composition of polypore assemblages. This is noteworthy, since conservation efforts are often focused on volume. However, our results suggest that it would be more important to ensure dead wood diversity in terms of species, sizes, and decay stages, as the volume itself is likely to come along with the sufficient diversity of these qualities.

It is clear that the microclimate and likely also the productivity of a brook-side habitat differs from the surrounding forest. Brook-side habitats are generally moister, more productive and have relatively higher pH than the upland forest which affects the moss assemblages (see e.g. Proctor 1990, Frisvoll and Presto 1997, Hylander et al. 2005). Consequently, a higher number of epiphytic moss species and more occurrences were found on both brook-side habitats than in control forest. However, the number of species per site was generally low due to the sites being managed spruce dominated forests where important substrates such as aspen (*Populus tremula*) are currently scarce. Thus, brook-side habitats of managed forests are not optimal habitats for

epiphytic mosses but they may nevertheless have some potential to maintain more diverse assemblages than the surrounding matrix.

It is important to note that the beetle data was collected with window traps that capture flying beetles and cannot provide an accurate measure for such a small area (see Saint-Germain et al. 2006, Sverdrup-Thygeson and Birkemoe 2009). Therefore, the catches are more or less reflecting the beetle assemblage of the whole surrounding forest, which is likely to hamper observing small between-habitat differences. We found no differences between studied habitats in the number, diversity and community composition of saproxylic beetle species but the communities of non-saproxylic beetle species differed between key habitat and control forest. The difference among non-saproxylic species is likely to reflect the obvious differences in the moisture and vegetation between the habitats. This is supported by the ecology of the indicator non-saproxylic species, of which e.g. *Cyphon coarctatus* is associated to moist habitats while *Atomaria pulchra* is associated to needle litter of coniferous trees. Among saproxylic species, the observed species assemblages consisted of species typical of managed boreal forests, that can obviously cope with low dead wood volume and with a monoculture of tree species thus being relatively unaffected by forest management (see e.g. Martikainen et al. 2000). For more specialized saproxylic species, a small isolated brook-side habitat with relatively low amount of dead wood is probably inadequate. However, some indicator saproxylic species were found for brook-side habitats, of which those associated to key habitat (*Denticollis linearis* and *Rhizophagus dispar*) are species that frequently utilize deciduous dead wood.

The primary criterion for classifying forest habitats, and also identifying key habitats, is the composition of ground layer vegetation. Furthermore, the same environmental factors (e.g. moisture and pH) affect moss (Frisvoll and Presto 1997) and vascular plant assemblages (Zinko et al. 2006). Therefore, it is almost self-evident that the vascular plant and moss assemblages of brook-sides, and key habitats in particular, differ from upland forests. Also the species richness of vascular plants (see Naiman and Decamps 1997, Nilsson and Svedmark 2002) and mosses (e.g. Dynesius

2001) is likely to differ, brook-side habitats in Scandinavia having naturally much more species than upland forests. Thus, concerning these groups, the meaningful question is whether the different brook-side habitats differ from each other. We found that although the species richness of vascular plants and mosses did not differ between the habitats, the assemblages were distinctly different between key habitats and control brook-sides, and that there were several indicator species to the key habitats (see Table 3).

The characteristic vascular plant species of a key habitat include species typical of grass-herb forest type, (*Athyrium filix-femina* and *Equisetum sylvaticum*), species associated to Oxalis-Myrtillus site type (*Maianthemum bifolium* and *Oxalis acetosella*), as well as species indicating moist conditions (*Trientalis europaea*). One moss species (*Rhizomnium punctatum*) characteristic of key habitat is a species typical of brook-sides and has declined due to forestry and forest drainage, while the other indicator moss species (*Sphagnum angustifolium*) is a common species of peatlands. Thus, the key habitats appear to be characterized by a variety of sub-habitats within the brook-side habitat, typically including moist and lush patches with divergent flora.

The classification of key habitats is based on on-site inventory, which is frequently done by foresters, with limited time and competence to identify the species, and the method rests largely on conspicuous plant species. Therefore, it comes as no surprise that the vascular plant, and to lesser extent moss assemblages, are the main differences between key habitats and comparable habitats not granted the status. Sites with eye-catching species, or sites with moist and marshy plots that are of low value for forestry, may be easily classified as key habitats while sites with less striking, but possibly equally valuable characteristics may be easily ignored.

The starting point of the study was to determine whether the Finnish Forest Act provides means to safeguard the diversity in managed boreal forests. It appears that brook-sides are diverse habitats hosting species and resources scarce in typical managed boreal forests. Protecting species-rich habitats does not necessarily mean protecting the species under greatest threat, but it is nevertheless clear that brook-sides substantially con-

tribute to the diversity of managed forests. A more important finding is that in many aspects brook-side key habitats do not differ from the brook-sides without the key habitat status. But are all brook-side habitats worth protecting then? The major problem in “precision conservation” in managed forests is extinction debt: reducing total area of habitat increases the probability of extinctions (see Hanski 2000, Ovaskainen 2002, Hanski 2008). Preserving certain key habitats of small area in managed forests is supposed to be cost-effective, but leads easily to extinctions after a time lag due to small populations or habitat alteration induced by changes in the surrounding matrix (Selonen and Kotiaho in prep.). Therefore, if the aim of conservation is, as it should be, to maintain viable populations of species, the more we preserve the area with obvious conservation value, i.e. diverging from the matrix, the better results we get. The network of key habitats is an important but probably inadequate supplement to forest conservation, which would benefit from the addition of comparable habitat but also from adjusting the management outside the protected area such as maintaining adequate buffer strips (e.g. Darveau et al. 1998, Johnson and Jones 2000, Harper and MacDonald 2001, Selonen and Kotiaho in prep.).

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