Losing your home: temporal changes in forest landscape structure due to timber harvest accelerate Northern goshawk (*Accipiter gentilis*) nest stand losses

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In Finland, forests are both one of the main national economic resources and the main source of biodiversity. Lack of detailed empirical evidence of how forestry affects biodiversity limits the development of sustainable forest management. Previous studies have found that the Northern goshawk (*Accipiter gentilis*) is a keystone species in mature coniferous boreal forests, and that its presence is associated with high local biodiversity. To understand how timber harvest affect goshawk nest-stand constancy and nest occupancy, we analyzed changes in nine landscape classes in two buffer zones (100 m, 250 m) around goshawk nests in Western Finland during 2005–2013. Patterns of nest occupancy and nest stand loss during 2005–2013 was compared with corresponding data from 1999–2005 to investigate possible long-term changes in the same. We found a positive connection between the proportion of mature spruce forest and nest occupancy. Nest stand constancy was at a lower level during 2005–2013 than during 1999–2005. This was mainly because of forest loss due to cutting of nest stands. Given that the loss of goshawk nest stands has accelerated and because timber harvest in the area was found to be specifically targeted towards forest types favoured by goshawks, forestry may have an even larger negative effect on biodiversity locally than previously thought given that goshawk nest stands are associated with high biodiversity. To render the Finnish forestry ecologically sustainable, we propose that protection of goshawk nest stands should be an integrated part of standard forestry practices in the future.

1. Introduction

Worldwide there is an urge for solutions towards a more sustainable use of land and resources, due to threats posed by climate change and the loss of biodiversity (Steffen *et al.* 2015). The main threats to species are loss of habitat and habitat fragmentation, which directly decrease the population size through lowering productivity and survival (Baillie *et al.* 2004, Fischer & Lindenmayer 2007).

In northern Europe, forestry intensified strongly after the 1960s and as a consequence the natural dynamics and succession of managed forests was completely transformed during the following decades (Östlund *et al.* 1997). More recently, there has been an even further increase in the demand for forestry-based products and with that there is a risk of increasingly exaggerated forest resource utilization, which is likely to jeopardize biodiversity and thereby ecosystem functioning (Kotiaho et al. 2017). Although protection areas may serve as refuges for several species, these areas are not sufficient to maintain effective population sizes. Therefore, recent initiatives have underlined the importance of practicing sustainable management also in managed forest environments in order to preserve biodiversity (e.g., Santangeli *et al.* 2012, Kotiaho *et al.* 2017). The sustainability issues of increased logging in boreal forests therefore needs to be considered thoroughly in order to maintain healthy wildlife populations and secure long-term persistence of functional biodiversity.

Especially among more sensitive species, such as extreme habitat specialists and top predators, intense forestry practices may cause the carrying capacity of the environment to be reduced as a consequence of decreasing food resources and suitable breeding habitat (Newton 1998, Byholm *et al.* 2007, Byholm & Kekkonen 2008) eventually leading to population decline (Hyvärinen *et al.* 2019). When present, top predators are assumed to be good indicators of functional biodiversity (Sergio *et al.* 2005). For example, in Finland the diversity of small birds and polypores (wood-decaying fungi) were higher in plots occupied by Northern goshawks *Accipiter gentilis* (from here onward goshawk) than in non-occupied control plots (Burgas *et al.* 2014), and in California, USA, the presence of spotted owls *Strix occidentalis* coincided with high diversity of molluscs and salamanders (Dunk *et al.* 2006). Since the goshawk is a documented keystone species in mature forests and its presence is strongly associated with high biodiversity (Sergio *et al.* 2006, Burgas *et al.* 2014, but see Ozaki *et al.* 2006), sparing goshawk nest stands from cutting would thus be a cost-effective way to preserve biodiversity in managed forest landscapes (Burgas *et al.* 2016).

Our aim is to analyze the temporal patterns in nest occupancy of goshawks breeding in a man-

aged forest landscape in southern Finland during 1999–2013, in particular in relation to timber harvest activities. We also investigated how nest occupancy patterns of goshawks were related to temporal changes in forest landscape configuration at the nest-stand scale (in 100 m and 250 m buffers), as judged from digital land cover data during 2005–2013. Our hypothesis was that goshawk nest occupancy is determined by the proportional area of mature forest in the vicinity of the nest and that timber harvesting at the nest-stand scale will have negative impact on nest-stand constancy (which is whether the forest vegetation in a goshawk nest stand remains intact) and thus on nest occupancy. Our second hypothesis was that goshawk nest stand constancy and nest occupancy is affected by timber harvesting more than by all other factors combined because mature to old-growth spruce dominated forests presently are the main targets for forest cutting in Finland (Muukkonen *et al.* 2012).

2. Materials and methods

2.1. Study species and study area

The goshawk is a medium-sized forest raptor widely distributed over the northern hemisphere (Cramp & Simmons 1980, Squires & Reynolds 1997). It has a diet consisting of small-medium sized birds and mammals (Møller *et al.* 2012) that are typically caught from an ambush. In common with many other larger-sized raptors, goshawks build large stick-nests that are used repeatedly in consecutive years. As nests are repaired continuously, the same nests are typically used for several years, often even during decades by multiple generations if the nest stand remains intact (Kenward 2006, P. Byholm, personal observations).

The study area is located in Southern Ostrobothnia, Western Finland around the small town of Närpes (lat 62°00'–62°55' N, long 21°05'–22°40' E). The area covers roughly 4,000 km^2 and the local landscape is characterized by a mixture of heavily managed boreal forest, bogs and fields. During 1999–2013, the study area was searched annually for potential goshawk nests. Nest occupancy and breeding performance (number of fledged young) of each nesting attempt were re-

Table 1. Landscape classes analyzed, cut-off values used for classifying forest into specific classes regarding using volume of timber (m^3 / ha) and age of trees (years) and their original sources (see text for further details).

Landscape class	Volume of trees (m^3 / ha)	Age of trees (years)	Source
Mature spruce forest (MSF)	≥ 120		MS-NFI
Mature mixed forest (MMF)	≥ 120	≥ 70	MS-NFI
Mature pine forest (MPF)	≥ 120		MS-NFI
Young forests (YF)		$40 \leq x < 69$	MS-NFI
Clear-cuts, plantations (CCP)		$0 < x < 40$	MS-NFI
Agricultural areas (AGR)			CORINE, EEA
Urban areas (UAR)			CORINE, EEA
Water bodies (WB)			CORINE, EEA
Open bogs and marshes (WET)			CORINE, EEA

corded, and the exact nest coordinates of all nests were registered (c.f. Byholm 2005, Byholm *et al.* 2012, Burgas *et al.* 2014).

Here we use data on 46 different nests in 40 territories (*i.e.*, the study unit is a specific nest), which were occupied at least once in 2005 and/or 2013 in order to study the change in nest occupancy during nine years as a function of landscape composition changes at the nest-stand scale. Of the nests, 11 were occupied in both 2005 and 2013, 29 were occupied only in 2005, 6 were occupied only in 2013 (these six nests were newly built nests at six territories active already in 2005). This material was then compared with information collected back to 1999 to investigate possible long-term changes in nest occupancy and nest stand constancy (i.e., to what degree nests occupied in 2005 were used for nesting in 1999 and in 2013, respectively).

2.2. Landscape data

For 2005 and 2013 we constructed land classes by combining Multi-source national forest inventory (MS-NFI) data (Tomppo *et al.* 2009, Mäkisara *et al.* 2016) and Corine Land Cover data (CLC 2006-Finland, 2008; CLC 2012-Finland, 2014). In its original form, the MS-NFI forest data is presented as m^3 / ha separately for different tree species (Norwegian spruce *Picea abies*, Scots pine *Pinus sylvestris*, birch *Betula* spp. and other broad-leaved trees) and with separate stand age information. The MS-NFI and CORINE datasets have dif-

ferent spatial resolution: MS-NFI rasters have 25 m × 25 m (for 2005) and 16 m × 16 m (for 2013) resolution, and CORINE rasters have 25 m × 25 m (for 2006) and 20 m × 20 m (for 2012) resolution. Because of this, we resampled all datasets in ArcGIS to one common resolution (25 m × 25 m). Prior to this, each pixel of forest volume data was – using MS-NFI – classified into five forest classes (cf. Table 1) following a stepwise approach. Spruce and pine dominated pixels were first classified into the categories mature spruce forest (MSF), mature pine forest (MPF) and the temporary class "other" according to volume wood (m^3 / ha).

Forest was classified as mature if the wood volume was ≥ 120 m^3 / ha, to "other" if it was < 120 m^3 / ha. Second, the class "other" and the merged birch and other broad-leaved classes were combined, and pixels were classified into three additional forest classes (mature mixed forest (MMF), young forest (YF) or clear-cuts and plantations (CCP)) using the average age of trees (x ≥ 70 years, 40 < x < 69 years and 0 < x < 39 years) to compensate for wood volume estimation due to summing m^3 / ha values (cf. Table 1). The remaining pixels were classified into four additional classes by the use of CORINE as outlined in Table 1. We pooled power lines data from CORINE with the class urban areas (UAR). We merged the reclassified MS-NFI data of 2005 with CORINE dataset of 2006 and the MS-NFI data of 2013 with CORINE data of 2012. We gave priority to the information on forest habitats from MS-NFI when combining it with CORINE.

Using this material, we analyzed the landscape composition in buffer zones with 100 m and 250 m radii around the nest sites. These areas around the nest sites were chosen because landscape composition in close vicinity of the nest site (the nest stand) is a major determinant of nest occupancy in goshawks (e.g., McClaren *et al.* 2015, Reynolds *et al.* 2008). We used the "isectpolyrst" tool in Geospatial Modeling Environment software (version 0.7.3.0) to get separate values of land class proportions for each buffer. Detailed analysis of habitats was done by FRAGSTAT software using (version 4.2.1.603). For each buffer, we calculated proportions of different landscape classes for the two chosen buffers for each nest. Landscape data were analyzed using ArcGIS software version 10.5.1.

2.3. Statistical modeling

We analyzed the effects of landscape structure on the probability of nest occupancy in forest stands that had occupied nests in 2005 and/or 2013. All explanatory variables used were standardized by subtracting the mean and dividing by the standard deviation to make effect sizes directly comparable (zero mean and SD unit).

We built the nest occupancy model as a generalized linear mixed model fit by Laplace approximation of the likelihood and logit link. Occupancy was entered as a binomial variable (1 = occupied, 0 = empty) and the explanatory variables were the (standardized) proportion of mature spruce forest (MSF), mature mixed forest (MMF), young forest (YF) and mature pine forest (MPF). Clear-cuts and plantations variable (CCP) was left out of the model because it was highly negatively correlated with MSF (250 m buffer: $r_s = -0.78$, $n = 80$, $p < 0.001$; 100 m buffer: $r_s = -0.85$, $n = 80$, $p < 0.001$; n as obtained from 40 territories occupied in 2005 with pairwise values for 2005 and 2013). We also did not include the landscape variables agricultural fields (AGR), water bodies (WB), urban areas (UAR) or wetlands (WET) in the models as their proportional areas were close to zero in nearly all buffers and they are not important in goshawk habitat selection (e.g., Penteriani 2002). Territory ID was entered as a random intercept to account for the repeated measures of the same sites in 2005 and 2013. We used the same model structure for the different buffer sizes (100 m radius and 250 m radius) around the nest.

We calculated the change in habitat structure in the 100 m and 250 m buffers around nests used for nesting in 2005 and/or 2013 ($n = 40$ territories). In six cases, the location of the nest within the territory had changed between the years. In two cases, the movement was > 300 m and these territories were excluded from all analyses. In two additional cases, a new nest had been built less than 100 m (19 m and 84 m) from the 2005 nest location, and for these the nest coordinates of 2005 were used in both 2005 and 2013. Two new nests that were active in 2013 were included in the models on the 250 m-scale as they were located within the radius of the 250 m buffer (145 m and 160 m respectively) of the original nest used for nesting in 2005. Therefore, sample sizes differ between models of the 100 m buffer ($n = 36$) and the 250 m buffer ($n = 38$).

The change in the amount of different forest types was calculated as the difference in the proportions of the given landscape element between 2005 and 2013. We modeled the probability of nest occupancy in 2013 as a function of the change in the (standardized) habitat variables MSF, MMF, MPF and YF using data from the 100 m and 250 m buffers around the nest. We used a GLM with binomial errors and logit link fit with Laplace approximation. All statistical analyses were conducted using R 3.4.2 (R Core Team 2017).

3. Results

3.1. Change in nest occupancy and nest stand constancy during 1999–2013

Of 40 goshawk nests that were occupied in 2005, nine (23%) were in use in 2013 leaving 31 (77%) being unused. Of the unused nests, three (10%) nests had fallen down although the nest supporting platform remained intact, seven (23%) were unused but still available in an unchanged forest stand and 21 (67%) were lost due to timber harvesting. In those cases, the nest stand was either clear-cut completely ($n = 14$) or alternatively ($n = 7$) so heavily modified from timber harvesting activities it was rendered unsuitable for nesting (see discussion). There were four new nests in 2013,
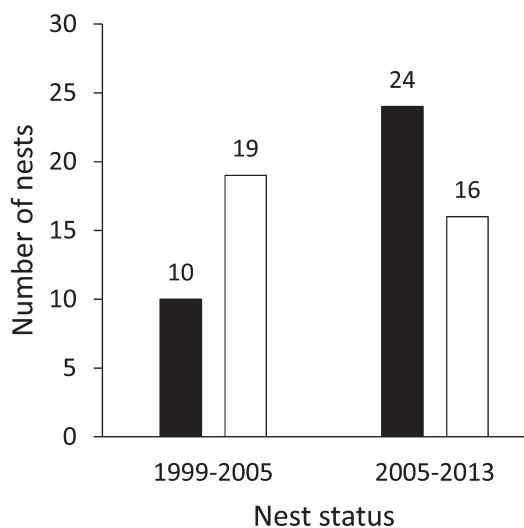
Fig. 1. Change in constancy of goshawk nest stands as measured from the difference in the numbers of lost and intact nest as compared between 1999–2005 (*n* = 29) and 2005–2013 (*n* = 40). Black bars represent lost nests, hollow bars intact nests. The numbers above the bars refer to category-specific sample size.

Table 2. The percentage (%) of different landscape classes (MSF = Mature spruce forest, MMF = Mature mixed forest, MPF = Mature pine forest, YF = young forest, CCP = Clear-cuts and plantations, AGR = Agricultural areas, UAR = Urban areas, WB = Water bodies, WET = Open bogs and marshes) as relative to the area of the whole study area in 2005 and 2013, as well as the temporal change (in%) of specific classes between 2005 and 2013.

Landscape class	2005 (%)	2013 (%)	Change (%)
MSF	5.42	5.52	0.10
MMF	18.07	14.72	−3.35
MPF	2.11	8.23	6.12
YF	25.63	22.68	−2.95
CCP	20.16	22.12	1.96
AGR	18.74	18.28	−0.46
UAR	5.14	4.05	−1.09
WB	0.81	0.79	−0.02
WET	3.91	3.59	−0.32

which were not present on the same territories in 2005. During 2005–2013, an average of 3.0 nests were lost (fallen nests + nests lost as a result of timber harvesting combined) annually. Disregarding cases where nests were unused but still were present in an intact forest stand, this corresponds to an annual nest-loss rate of 9.1%. If excluding natural nest losses, 2.6 nests were on average lost due to timber harvesting practices *per se* annually. This corresponds to an annual nest-loss rate of 7.9%.

Using the nests occupied in 2005 as reference and dating back in time to 1999, 11 (28%) of the 40 nests were built only after 1999. Among the 29 remaining nests that were around already in 1999, 19 (66%) were used for nesting in 2005 while 10 (34%) were lost due to timber harvesting (all by clear cutting) by the summer of 2005. None belonged to the category "available but unused" and none had fallen down. On average 1.7 nests were lost annually during 1999–2005, which corresponds to an annual nest-loss rate of 5.8%.

The probability that a nest stand would remain intact and suitable for breeding differed significantly as compared between 1999–2005 and 2005–2013 (Chi-square test, $\chi^2 = 4.38$, *df* = 1, *p* = 0.036; Fig. 1). The overall annual nest-loss rate in-

creased 57% between the two time periods, and the annual nest-loss as caused by timber harvesting alone increased by 36%. Of the total amount of goshawk nests that were lost due to timber harvesting during 1999–2015, 31 nests (82%) were lost because the whole nest stand was clear-cut, while 7 (18%) were abandoned as a result of harvesting activities in the immediate vicinity of the nest even if the nest tree itself was spared (see discussion).

3.2. Forest habitat configuration in 2005 and 2013

There was practically no change in the percentage of land covered by MSF during 2005–2013 when analyzed over the whole study area. The proportion of MMF and YF decreased, while the proportions of MPF and CCP increased. The proportions of other land-use classes (UAR, AGR, WET and WB) remained largely constant (Table 2).

In the 100 m buffer zone, MSF made up the majority of the land area in both 2005 and 2013 when averaged over both occupied and empty nests (Table 3). MMF and YF were the second most common landscape classes, followed by CCP. Although the proportion of MSF slightly increased as compared between 2005 and 2013 over the whole study area (Table 2), the proportion of MSF (and MMF) significantly decreased (due to

Table 3. Proportion of different forest types (± SD) in the 100 m and 250 m buffer zones around goshawk nests in relation to year and occupancy status and as averaged over all analyzed locations. Wilcoxon signed-ranks tests were used for comparing years, Mann–Whitney U-tests for comparing occupancy status. Significances: ° = $p < 0.1$, * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$. Landscape classes as in Table 1, Other is AGR, UAR, WB and WET combined.

	Proportion ± SD					Direction of change and significance	
	2005	2013	Occupied	Not occupied	Overall	From 2005 to 2013	Occupied and not occupied
100m							
MSF	0.44 ± 0.24	0.35 ± 0.28	0.47 ± 0.23	0.20 ± 0.24	0.39 ± 0.26	–, *	–, **
MMF	0.19 ± 0.12	0.15 ± 0.13	0.18 ± 0.12	0.16 ± 0.15	0.17 ± 0.13	–, *	0, NS
MPF	0.01 ± 0.02	0.04 ± 0.05	0.02 ± 0.03	0.04 ± 0.06	0.02 ± 0.04	+, ***	+, *
YF	0.19 ± 0.11	0.16 ± 0.11	0.18 ± 0.11	0.15 ± 0.13	0.18 ± 0.11	0, NS	0, NS
CCP	0.16 ± 0.16	0.29 ± 0.31	0.14 ± 0.15	0.43 ± 0.33	0.22 ± 0.25	+, **	+, **
Other	0.01 ± 0.05	0.01 ± 0.05	0.01 ± 0.04	0.02 ± 0.03	0.01 ± 0.05	0, NS	0, NS
250m							
MSF	0.25 ± 0.14	0.21 ± 0.15	0.25 ± 0.14	0.16 ± 0.14	0.23 ± 0.14	0, NS	–, *
MMF	0.19 ± 0.06	0.15 ± 0.08	0.18 ± 0.06	0.15 ± 0.10	0.17 ± 0.07	–, *	0, NS
MPF	0.01 ± 0.01	0.06 ±0.05	0.03 ± 0.05	0.04 ± 0.03	0.04 ± 0.04	+, ***	+, *
YF	0.28 ± 0.09	0.22 ± 0.06	0.27 ± 0.08	0.21 ± 0.07	0.25 ± 0.08	–, **	–, **
CCP	0.24 ± 0.14	0.32 ± 0.21	0.24 ± 0.15	0.38 ± 0.21	0.28 ± 0.18	+, °	+, **
Other	0.03 ± 0.10	0.03 ± 0.10	0.03 ± 0.09	0.05 ± 0.13	0.03 ± 0.10	0, NS	0, NS

cutting) in the 100 m buffer zone. In 2005 in the 100 m buffer zone, MSF at goshawk nest sites was characterized by significantly larger trees (as measured from m^3 timber using the MS-NFI data), than in an MSF-fragment on average ($n = 50$ random points) in the study area (nest: 249.3 ± 141.3 m^3, random points: 184.0 ± 121.1 m^3, Two sample t-test, $t = 2.35$, $df = 87$, $p = 0.02$). As in the study area in general, CCP and MPF increased also at the 100 m scale. The proportions of other landscape classes were insignificant (Table 3).

In the 250 m buffer zone, the proportion of MSF was substantial too, although YF and CCP were about equally common (Table 3). In the 250 m buffer zone, when comparing 2005 with 2013, the proportion of MSF remained stable, the proportions of MMF and YF decreased, and the proportion of CCP and MPF increased (Table 3).

3.3. Habitat variables determining goshawk nest occupancy

Within the 100 m buffers, the probability for a nest being occupied increased with increasing amount of MSF ($b = 1.79 ± 0.46$, $z = 3.92$, $p < 0.001$, Fig. 2a). YF and MMF also had positive effects on occupancy (YF: $b = 0.78 ± 0.39$, $z = 2.01$, $p = 0.04$; MMF: $b = 0.89 ± 0.37$, $z = 2.41$, $p = 0.02$), whereas the amount of MPF had a negative effect ($b = -1.03 ± 0.37$, $z = -2.80$, $p = 0.005$). The summed proportion of the remaining landscape variables had no effect on occupancy ($z = 0.86$, $p = 0.39$).

Also at the 250 m buffer radius, nest occupancy was positively associated with increasing amount of MSF ($b = 0.78 ± 0.31$, $z = 2.51$, $p = 0.01$, Fig. 2b) and increasing amount of YF ($b = 0.84 ± 0.35$, $z = 2.41$, $p = 0.02$), whereas MPF ($z = -0.49$, $p = 0.63$), MMF ($z = -0.49$, $p = 0.62$) and the summed proportion of the remaining landscape variables ($z = 0.71$, $p = 0.48$) were unrelated to occupancy.

3.4. The effect of change in forest habitat configuration over time on nest occupancy

The probability of a nest being occupied in 2013 was determined by the change in the proportion of MSF and MMF in the nest vicinity: the proportion
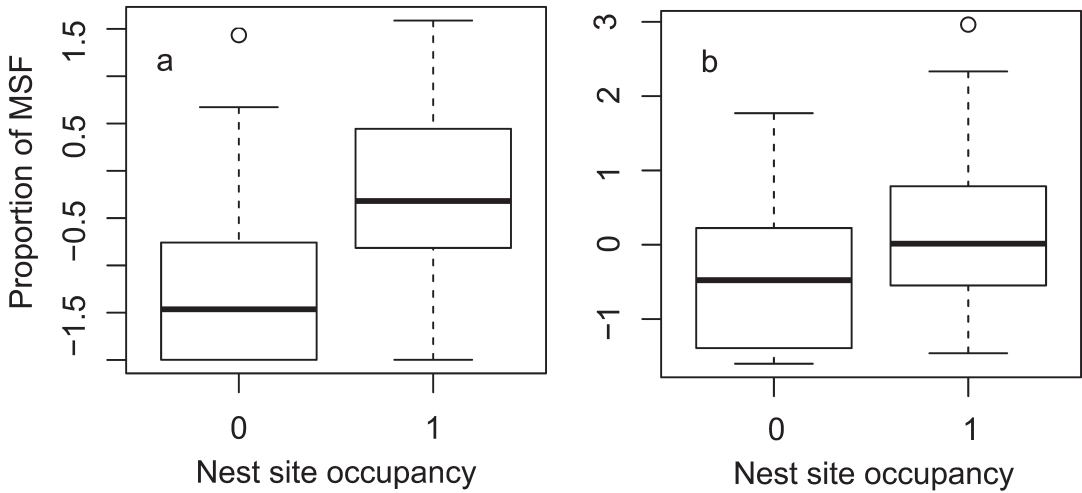
Fig. 2. Box plots showing the median and Q25 / Q75 of the amount of area covered with mature spruce forest (MSF) in empty (0) and occupied (1) goshawk nest sites on (a) 100 m radius and (b) 250 m radius buffers. MSF is standardized to zero mean and unit SD.
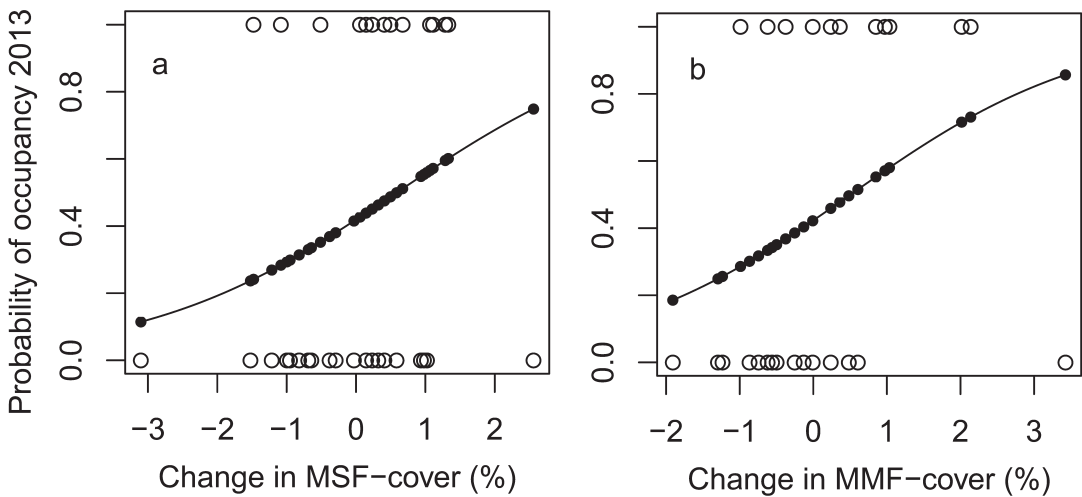

Fig. 3. The probability of nest occupancy as a function of change in (a) mature spruce forest (MSF) cover and (b) mature mixed forest (MMF) cover within 100 m around a nest occupied in 2013 as compared with 2005. MSF and MMF are standardized to zero mean and unit SD. Negative values indicate that the cover has decreased and a positive value that it has increased. Statistics of the full models can be found in the text.

of MSF and MMF in the 100 m buffers decreased at unoccupied nests, whereas the proportion slightly increased around nests that remained occupied (MSF: $b = 0.90 \pm 0.45$, $z = 1.98$, $p = 0.05$; MMF: $b = 1.16 \pm 0.48$, $z = 2.40$, $p = 0.02$, Fig. 3). None of the other habitat variables had significant effects on the probability of occupancy of goshawk nests (YF: $z = 1.02$, $p = 0.31$; MPF: $z = -1.19$,

$p = 0.24$). When analyzing the change in habitat proportion in the 250 m buffer zones, the change in MSF and MMF were no longer of importance for nest occupancy (MSF: $z = 0.43$, $p = 0.67$: MMF: $z = 0.39$, $p = 0.70$). Neither did the other forest types explain the probability of nest occupancy at the 250 m-scale (YF: $z = -0.53$, $p = 0.60$; MPF: $z = -1.18$, $p = 0.24$).

4. Discussion

4.1. Goshawk nest occupancy
and forest change

Our study shows that when a goshawk nest becomes inactive in a typical Finnish forest landscape, this is, in a majority of cases, the result of forest harvesting activities at the nest stand level. The speed at which goshawk nests were lost had accelerated between 1999–2005 and 2005–2013. We use detailed forest inventory data (Tomppo *et al.* 2009, Mäkisara *et al.* 2016) to show that the core area around goshawk nests was dominated by MSF.

We further show that MSF (and MMF) around goshawk nests has decreased significantly as compared between 2005 and 2013. This loss of goshawk nest stands and the following decrease in nest occupancy has occurred despite the fact that there has been no clear decrease in the overall proportion of goshawk prime nest habitat (i.e., MSF) in the study area as a whole. This indicates that recent timber harvesting has specifically targeted forest stands that are used by goshawk for nesting more than would be expected if MSF-stands were cut at random.

It is not within the scope of this study to investigate the details of the exact reason(s) for this result, but for example, given that the nest stands in 2005 were characterized by larger trees than the average (MSF) forest fragments in the study area, the trees in the nest stands would have been an especially attractive source of income for the forest owners and therefore cut.

As judged from the analysis of the effect of landscape elements and forest management on nest occupancy here, as well as from other investigations (reviewed in Penteriani 2002), nest stand characteristics constitute the most important factor influencing nest occupancy patterns in goshawks. Somewhat surprisingly, and in contrast to many earlier studies, YF had a positive effect on nest occupancy probability, whereas the effect of mature pine forest was negative (at the 100 m scale) or neutral (at the 250 m scale). When comparing the 2005 situation with that of 2013, the 2005 nests that remained occupied showed a slight increase of MSF (and MMF) in the 100 m-buffer (but not in the 250 m-buffer), whereas MSF (and MMF) sig-

nificantly decreased around nests that were unoccupied.

While ca. 30% of the 2005 nests that were unoccupied in 2013 were unused due to natural reasons (nest had fallen down or was unused albeit the nest and nest stand was intact), the single most important factor resulting in a nest being unused was harvest of timber. As such, this is in line with the findings from a meta-analysis conducted by Rodriguez *et al.* (2016) showing that timber harvest close to nests of goshawks affect occupancy negatively. In our study area, the loss of nests due to timber harvest activities was usually because the nest stand was cut completely, but in ca. 20% of cases the nest was abandoned even though the nest tree was spared.

Such cases include situations where a) the nest tree and 5–15 additional trees were spared, but the rest of the parcel was cut, b) the nest stand was heavily thinned and / or clear-cuts appeared in the absolute vicinity (neighbor parcels, usually < 50 m from the nest to clear-cut edge) of the nest. Due to the strict habitat requirements of goshawks, all of these situations in practice render the nest no longer a real alternative for the hawk to breed in (Penteriani 2002, Kenward 2006, Rodriguez *et al.* 2016).

The finding that the probability for a goshawk's nest turning unoccupied was significantly higher during 2005–2013 than during 1999–2005 is a result that deserves some more attention. As outlined above, the main reason for this was increased timber harvesting of goshawk nest stands between the two time periods. This acceleration in the rate of nest destruction due to timber harvesting is in deep contrast to the findings reported from a case study conducted in Eastern Finland during the same time period (Santangeli *et al.* 2012). However, in that case nests were spared from cutting through a specific conservation program where landowners were motivated to voluntarily set aside forest buffers around raptor nests. No such program has been adopted in our study region. Since this kind of specific conservation initiatives are not part of any general management recommendation, our result is in fact likely to represent the typical situation over Finland as a whole.

Our results also show that even though the amount of prime habitat of a species (here MSF, *cf.*

Table 2) would not decrease at the level of the landscape, this does not mean that all stands characterized by prime habitat would be freely available for the focal species. While the present study setup does not allow a firm disentanglement of specific causes and consequences, we see two possible, not mutually exclusive, explanations for the present result.

First, some of the 2005 nests that remained unoccupied in 2013 although the nest and nest stand remained intact, likely remained unoccupied due to death of the nest owners. This conclusion is backed up by the observation that in 2009–2010 the size of the Finnish goshawk population severely dropped due to a combination of unusually cold winter weather and low winter prey abundance (Honkala *et al.* 2011). Second, due to the territorial behavior and ideal despotic distribution of central place foraging predators (Zimmerman *et al.* 2002), large-scale variation in prey availability and inter-specific competition acting on the distribution on goshawks in the study population (Byholm *et al.* 2007, Byholm *et al.* 2012), all habitat patches are not likely to be free to be colonized in practice. This is supported by the fact that the goshawk is a prime example of a species that utilizes the same nest stand (and nest) even for decades if only the forest stays in the right (climax) successional stage (Kenward 2006, P. Byholm, personal observations).

Because a (meta)population response towards habitat destruction typically comes with a time lag (Hanski & Ovaskainen 2002), the impact of timber harvesting directed towards nest-stands is thus possibly even more detrimental in the long-term than what the direct numbers reported here may suggest. However, since other factors, such as decreasing food resources (Byholm *et al.* 2007) and a changing climate (Lehikoinen *et al.* 2013), are likely to be of importance for goshawk demography too, the relative importance of different drivers on goshawk population development should be more comprehensively examined in future work.

4.2. Biodiversity and goshawk

Our study presents direct evidence of rapid adverse impacts of increased timber harvest activity on nest-occupancy patterns on the declining Finnish goshawk population. Given the rapid loss of biodiversity in general due to forestry activities in Finland and globally (Kotiaho *et al.* 2017, Hyvärinen *et al.* 2019), new initiatives to conserve biodiversity are needed. Against this background and since Finland ratified the Convention on Biological Diversity (United Nations 1992) to halt biodiversity loss by 2020 already in 1994, the finding that the speed by which goshawk nests and nest stands were lost due to timber harvesting accelerated during the course of the study can be viewed as a policy failure. This is not only because of the negative impact that loss of forest has on the goshawks themselves, but also because goshawk nest stands host more local biodiversity than forest fragments of similar size and type uninhabited by goshawks (Sergio *et al.* 2006, Byholm *et al.* 2012, Burgas *et al.* 2014).

Thus, harvesting a goshawk nest stand will have a disproportionate negative impact on local biodiversity as compared to harvesting similar sized forest stands that lack goshawk nests. Since protecting the immediacies of goshawk nests has been shown to be a more cost-efficient alternative to protect local biodiversity than multiple other alternatives (Burgas *et al.* 2016), we propose that forest stands hosting goshawk nests should be given additional value in the future when grading forest areas which are intended to be protected. Obviously, because of the area-species number relationship (MacArthur & Wilson 1967), the size of the (planned) conservation area is also of central importance. If specifically aiming at securing the long-term occupancy of goshawk nest stands, this would require leaving forest intact within a radius of ideally 100 m from the nest since alterations in the forest habitat at this scale according to our results is of importance for long-term nest occupancy (see also Santangeli *et al.* 2012, Rodriguez *et al.* 2016).

Since even a network including all goshawk nests in a specific region is not likely to be successful in conserving biodiversity in the long run (*cf.* Burgas *et al.* 2016), also additional conservation actions are needed to safeguard regional biodiversity. However, as a first step towards a more ecologically sustainable forestry we suggest sparing mature forest within a buffer zone of ideally 100 m from a goshawk nest should be a normal procedure of future forestry-management practices.

Förlusten av duvhökars (*Accipiter gentilis*) boskogsfigurer har accelererat som ett resultat av förändrade hyggesåtgärder

I Finland är skog en av de viktigaste nationella ekonomiska resurserna samtidigt som skogsnatur också är den främsta källan till biologisk mångfald. Brist på detaljerade empiriska bevis för hur rådande skogsbruk påverkar biologisk mångfald begränsar utvecklingen av hållbar skogsresursförvaltning. Tidigare studier har visat att duvhök (*Accipiter gentilis*) är en grundpelarart i äldre barrskog och att dess närvaro är förknippad med hög lokal biologisk mångfald.

För att förstå i vilken grad hyggesåtgärder slår mot skogsfigurer bebodda av duvhökar, och om dessa inverkar på sannolikheten att ett bo är ockuperat, analyserade vi förändringar i nio landskapsklasser i två buffertzoner (100 m, 250 m) runt duvhöksbon i västra Finland under åren 2005–2013. Duvhöksbons ockupationssannolikhet och beständigheten av skogsfigurer använda av duvhökar för häckning 2005–2013 jämfördes med motsvarande data 1999–2005 för att undersöka möjliga långsiktiga förändringar i desamma. Vi hittade en positiv koppling mellan andelen mogen granskog i landskapet kring ett duvhöksbo och sannolikheten för att ett bo är bebott. Beständigheten av skogsfigurer med duvhöksbo var på en lägre nivå 2005–2013 än 1999–2005. Detta berodde främst på ökad hyggesaktivitet.

Med tanke på att förlusten av duvhöksbon har accelererat, och då hyggesåtgärderna var specifikt riktade mot skogsfigurer som prefereras av duvhökar, kan rådande hyggespraxis ha en ännu större negativ effekt på biologisk mångfald lokalt än man tidigare uppskattat. Detta eftersom de skogsfigurer där duvhökar häckar i snitt uppvisar en högre biologisk mångfald än kontrollområden av samma storlek. För att göra det finländska skogsbruket ekologiskt hållbart föreslår vi att skydd av skog runt duvhöksbon ska vara en integrerad del av normal skogsbrukspraxis i framtiden.

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