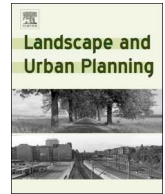




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Sweden does not meet agreed national and international forest biodiversity targets: A call for adaptive landscape planning

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ABSTRACT

Loss of forest naturalness challenges the maintenance of green infrastructure (GI) for biodiversity conservation and delivery of diverse ecosystem services. Using the Convention on Biological Diversity's Aichi target #11 with its quantitative and qualitative criteria as a normative model, we aim at supporting landscape planning through a pioneering assessment of the extent to which existing amounts and spatial distributions of High Conservation Value Forests (HCVFs) meet these criteria. Highly forested and committed to both intensive wood production and evidence-based conservation targets of 17–20% protected areas, Sweden was chosen as a case study. Specifically, we estimated the amount, regional representation, and functional connectivity of HCVF patches using virtual bird species, validated the results using field surveys of focal bird species, and assessed conservation target fulfilment. Finally, we linked these results to the regional distribution of forest land ownership categories, and stress that these provide different opportunities for landscape planning. Even if 31% of forest land in Sweden is officially protected, voluntarily set-aside, or not used for wood production now and in the future, we show that applying the representation and connectivity criteria of Aichi target #11 reduces this figure to an effective GI of 12%. When disaggregating the five ecoregions the effective GI was 54% for the sub-alpine forest ecoregion, which hosts EU's last intact forest landscapes, but only 3–8% in the other four ecoregions where wood production is predominant. This results in an increasing need for forest habitat and landscape restoration from north to south. The large regional variation in the opportunity for landscape planning stresses the need for a portfolio of different approaches. We stress the need to secure funding mechanisms for compensating land owners' investments in GI, and to adapt both the approaches and spatial extents of landscape planning units to land ownership structure.

1. Introduction

Traditionally, forest management has aimed to maximize economic revenue from timber, pulpwood and biomass (Puettmann, Coates, & Messier, 2012). However, current policies about sustainable forest management also target the maintenance of other benefits from forest landscapes, such as biodiversity and social values (e.g., Forest Europe,

2015; The Montréal Process, 2015). Ecosystem services and nature's contribution to people are current concepts capturing the need to sustain all these dimensions of sustainable landscapes (IPBES, 2019). Yet, a long-term focus on high sustained yield wood production has caused loss and simplification of once naturally dynamic forests in many European regions (Sabatini et al., 2018). This has negative consequences for biodiversity conservation and delivery of a diverse range of

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0169-2046/ © 2020 The Authors. Published by Elsevier B.V. This is an open access article under the CC BY license (<http://creativecommons.org/licenses/by/4.0/>).

ecosystem services (e.g., [IPBES, 2019](#); see also: [Eyvindson, Repo, & Mönkkönen, 2018](#); [Svensson, Andersson, Sandström, Mikusiński, & Jonsson, 2019](#); [Triviño et al., 2015](#)). The European continent's northern conifer-dominated forests are highly relevant for comparative studies of this widespread pattern. This is because they simultaneously host the last intact forest landscapes in Europe ([Potapov et al., 2008](#)), and landscapes dominated by forests aimed at wood production with very high forest management intensity ([Angelstam et al., 2018](#); [Kuhmonen, Mikkola, Storränk, & Lindholm, 2017](#); [Naumov et al., 2018](#)).

Sweden hosts only 0.4% of the global productive forest cover ([FAO, 2015](#)), but due to effective sustained high yield forestry provides 10% of the sawn timber, pulp and paper traded on the global market ([Helander, 2015](#)). Sweden is also committed to biodiversity conservation at multiple levels. Internationally, Sweden is a signatory of the Convention on Biological Diversity, the Aichi target #11 ([CBD, 2010](#)) of which states that at least 17% of terrestrial ecosystems should be conserved through an effective system of protected areas covering representative land covers with functional connectivity. Additionally, Aichi target #7 prescribes sustainable management of the forest landscape surrounding protected areas, for example through tree retention (e.g., [Fedrowitz et al., 2014](#)). Sweden has also ratified EU-level policies such as the EU Biodiversity Strategy ([European Commission, 2011](#)) and the EU strategy Green Infrastructure (hereafter GI) for natural and semi-natural land covers ([European Commission, 2013](#)). Finally, national forest and environmental policies aim at maintaining viable populations of all naturally occurring species ([Angelstam et al., 2011](#); [Angelstam & Andersson, 2001](#)), by setting a quantitative target of 20% protected areas on forest land ([Miljödepartementet, 2014](#)). Similarly, Sweden is committed to maintaining GI in support of social values ([Elbakidze et al., 2017](#)). Sweden is thus an appropriate setting for studies evaluating the opportunity to increasing high sustained yield wood production ([Felton et al., 2019](#)), while also meeting other forest policy objectives such as biodiversity conservation ([Lindahl et al., 2017](#)) and human well-being ([MEA, 2005](#); [Tzoulas et al., 2007](#); [Elbakidze et al., 2017](#)).

A long history of forest use to provide raw material for the forest industry ([Lundmark, Josefsson, & Östlund, 2013](#)) has left Sweden with an ecoregionally uneven distribution of forests with high value for biodiversity conservation ([Skogsstyrelsen, 2019a](#); [Angelstam et al., 2011](#); [Jonsson, Svensson, Mikusiński, Manton, & Angelstam, 2019](#); [Svensson et al., 2019](#)). Identification of remaining High Conservation Value Forests (hereafter HCVFs) has therefore been the focus of mapping projects both in the field ([Timonen et al., 2010](#)) and by remote sensing ([Bovin, Elcim, & Wennberg, 2017](#)), as a way for finding potential additions to the protected forest areas network, and thus to strengthen its functionality as GI. The term HCVF was coined by the Forest Stewardship Council in the late 1990s to highlight the forest areas that deserve conservation in a forest certification context. Although there are several different types of HCVFs ([Jennings et al., 2003](#)), the term is commonly used to define forests that are particularly important for biodiversity conservation, such as old-growth, ancient, primary, virgin, intact, primeval or continuity forests ([Buchwald, 2005](#)). In Sweden, databases describing the spatial locations of existing and potential HCVFs (e.g., [Svensson et al., 2019](#)) have only recently become available. However, even if these data are currently being used by local, regional and national authorities for GI planning, there is a lack of harmonised cross-regional and country-wide spatial analyses. Such analyses would help policy makers, regional authorities, planners and managers to improve biodiversity policy implementation and strategic GI planning approaches.

There is currently an intense and polarised debate over different forestry narratives in Sweden (e.g., [Sténs & Mårald, 2020](#)). A key topic is how much forest is “protected” vs. “not available for intensified wood production” ([Anon., 2018](#)). In 1997, the Swedish government and parliament decided to increase the protection of productive forests by 900,000 ha before 2010 ([Angelstam et al., 2011](#)), and listed two groups

of instruments to meet that target ([Statskontoret, 2007:33 ff.](#)). This was achieved by implementing two groups of protection measures. The first group consists of three formal forest protection types (national park, nature reserve and biotope protection areas; [Miljöbalken, 1998:808](#)). The second includes voluntary instruments such as forest certification systems, conservation agreement areas, Sveaskog Co's Ekoparks and conservation management areas focusing on the white-backed woodpecker (*Dendrocopos leucotos*) as a focal species. However, driven by a strong political interest ([Anon., 2018](#)) and pressures from some land owner representatives ([Sténs & Mårald, 2020](#)), there is currently a tendency to shift the focus from “protected areas” to “areas not available for intensified wood production”. This means including in national accounting also (1) areas of variable retention (e.g., buffer strips and consideration areas according to § 30 in the Swedish forestry law ([Skogsvårdslag, 1979:429](#)), and (2) unproductive forest land ([Skogsstyrelsen, 2019a](#)). This has increased the “protected area” proportion in Sweden from 13% ([Skogsstyrelsen, 2019a](#)) to 31% “not available for intensified wood production” both at present and in the future ([Anon., 2018](#)). This change calls for evaluation of the consequences for biodiversity reporting, policy and conservation actions. For example, biodiversity conservation is not only about the total area of forests not used for wood production. Also ecoregional representation and green infrastructure functionality linked to connectivity ([Gaston, Jackson, Cantú-Salazar, & Cruz-Piñón, 2008](#)) are critical. There is thus a need to assess the extent to which the outcome of this debate has consequences for Sweden's ability to meet national and international targets and agreements about biodiversity conservation. The term green infrastructure i.e. “a strategically planned network of high quality natural and semi-natural areas with other environmental features, which is designed and managed to deliver a wide range of ecosystem services and protect biodiversity in both rural and urban settings” ([European Commission, 2013](#)) captures this, and is a current focal spatial concept promoting landscape planning (e.g., [Snäll, Lehtomäki, Arponen, Elith, & Moilanen, 2016](#); [Hermoso, Morán-Ordóñez, Lanzas, & Brotons, 2020](#)) for both conservation and use of different land covers. The broader landscape perspective taken in the green infrastructure concept is in line with socio-ecological system approaches reflected for example in the European Landscape convention (e.g., [Mikusiński et al., 2013](#)) and the IPBES framework's focus on interoperability among disciplines, stakeholders and knowledge systems ([Díaz et al., 2015](#)).

Spatial planning plays a crucial role for the creation and maintenance of a functional GI (e.g., [Botequilha Leitão & Ahern, 2002](#)). To be effective, planning should integrate both the ecological and social aspects of the system, i.e. it should embrace a landscape approach (e.g., [Arts et al., 2017](#); [Angelstam et al., 2019a](#)). Key ecological aspects involve securing the representation of different forest ecosystems and development stages after disturbances, and ensuring that these patches have sufficient quality, size and functional connectivity to maintain species that cannot cope with intensive forest management. Hence, spatial planning requires detailed knowledge about which forest patches are valuable for biodiversity conservation, and which represent different forest biotopes and representative portfolios of natural and anthropogenic disturbance regimes at multiple spatial scales (e.g., [Angelstam & Kuuluvainen, 2004](#); [Jönsson, Fraver, & Jonsson, 2009](#)). Key social system aspects involve landscape stewardship as “place-based, landscape-scale expression of broader ecosystem stewardship” ([Bieling & Plieninger, 2017](#); [Angelstam & Elbakidze, 2017](#); [Primdahl et al., 2018](#)). Landscape stewardship considers the variability of land ownerships, since owners may have different knowledge, will and understanding of biodiversity and ecosystem services (e.g., [Richnau et al., 2013](#)). To stress this study's focus on supporting strategic decision-making concerning GI, hereafter, we use the term landscape planning, which should integrate land-uses across spatial scales. However, few studies so far integrate GI in a planning concept that captures the complexity of landscapes as social-ecological systems ([Hansen & Pauleit, 2014](#)).

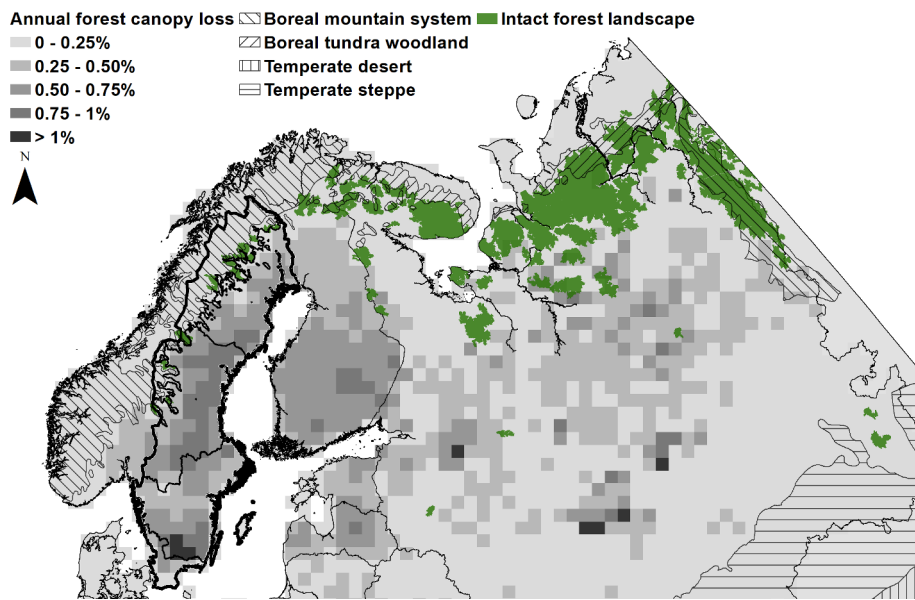


Fig. 1. Map illustrating the location of the last intact forest landscapes in Sweden, marked with bold border, and northern Europe, as well as areas with different forest management intensities. Europe’s last intact forest landscapes (> 50,000 ha; from Potapov et al., 2008; <http://www.intactforests.org/data.ifl.html>) are confined to the most inaccessible parts of northern Europe. Mountain, tundra, steppe and desert do not have forest as natural potential vegetation (Bohn & Gollub, 2006). Forest management intensity was estimated as the proportion of the annually harvested area in the northern part of the European continent based on the annual loss of high forest (i.e. clear-felling systems as opposed to coppice and coppice with standards), see Angelstam et al. (2017). On average 7% of the canopy loss was linked to fire and windfall, and the rest was caused by clear-felling (see Hansen et al., 2013). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

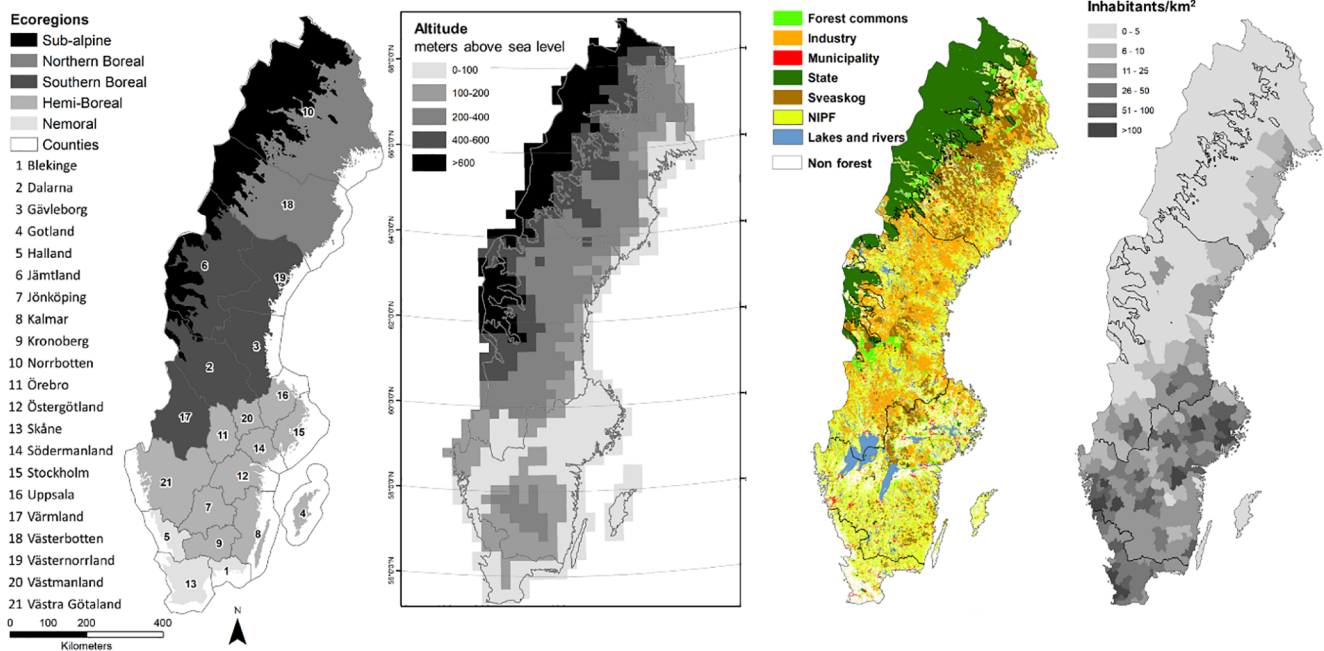


Fig. 2. Maps showing four aspects of Sweden as a social-ecological system. Far left: stratification of Sweden into forest ecoregions based on separation into counties, which are the units for statistical reporting and regional spatial planning (Naturvårdsverket and Statistiska Centralbyrån, 2018), and by the border of sub-alpine forests (SKSFS, 1991). Centre left: latitude and altitude by 25 × 25 km pixels. Centre right: land owner categories (NIPF means non-industrial private forest owners). Far right: Sweden’s 290 municipalities and their human population density (Statistics Sweden 2018). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

With this background the three aims of this study are to assess: (1) the functionality of HCVF networks as green infrastructure among forest ecoregions with different histories of forestry intensification in Sweden; (2) the potential of HCVPs to satisfy CBD’s Aichi target #11 of 17% protected areas with its constituent qualitative criteria; and (3) the opportunity for landscape planning towards functional GI given different types of land ownership patterns in Sweden’s forest ecoregions. Finally, given the current strong global interest in forestry intensification, we discuss what lessons can be learned from Sweden, on the rivalry between forestry intensification and maintenance of functional GI for countries and regions on the same trajectory towards intensified forest management.

2. Methods

2.1. Sweden as a case study

Sweden hosts both regions with high forest management intensity and the European Union’s last intact forest landscapes (Fig. 1; Jonsson et al., 2019). As such, Sweden captures both the consequences of the current expansion of north European frontiers of “wood mining” in the old-growth forests of remote NW Russia, and the subsequent forest management intensification closer to the west European market (e.g., Naumov et al., 2018) (Fig. 1). The process towards effective wood production was initially driven by the Swedish iron industry’s need for charcoal in the Swedish mining region Bergslagen (Obbarius, 1845).

When the international frontier of wood mining reached western Sweden in the 19th century (Wallmo, Andersson, Hesselman, & Pettersson, 1914) the process of transforming once naturally dynamic forests (e.g., Pennanen, 2002) into an effective wood production system commenced. Forestry intensification scaled up progressively at the national level during the 20th century, driven by an increasing demand for raw material (Lundmark et al., 2013). As a result, Sweden has few remnants of intact forest landscapes (Hansen et al., 2013; Potapov et al., 2008), and the majority of forests are managed by clear felling systems with rotations of 45 to 100 years. This corresponds to regional final felling rates of ca. 1–2% per year (Jonsson et al., 2019; Fig. 1). The frontier of forest landscape transformation continues to reduce the remaining remnants of HCVFs in Sweden (Svensson et al., 2019).

In addition to forest landscape history, the regional diversity of forest environments must be considered. In Sweden, 23.6 million ha is productive (annual wood production $> 1\text{m}^3\text{ha}^{-1}$) and 4.6 million ha is unproductive forest land (Naturvårdsverket and Statistiska centralbyrån, 2018:32). Sweden has five different forest ecoregions that range from broad-leaved deciduous nemoral forests and hemi-boreal forests in the south, to south and north boreal as well as sub-alpine forests in the north (Fig. 2). These five forest ecoregions are linked to both latitudinal and altitudinal (Fig. 2) factors affecting the vegetation growing period, forest site production capacity and species distributions. The northern borders of the nemoral and hemi-boreal forest ecoregions broadly parallel the northern contiguous distribution of beech (*Fagus sylvatica*) and oak (*Quercus robur*), respectively. Together with the two boreal ecoregions further north, which are dominated by Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*), these four ecoregions are widely used for intensive wood production. In contrast, the sub-alpine ecoregion hosts the lowest proportion of productive forest among all ecoregions (Fig. 3), is dominated by Norway spruce and mountain birch (*Betula pubescens* ssp. *czerepanovii*), and is confined to the Scandinavian mountain range's eastern edge (SKSFS, 1991). The altitudinal tree-line limit for these forest ranges from ca 900 m a.s.l. in the south and to ca. 500 m a.s.l. in the north.

When it comes to land ownership, Swedish forests are mostly owned by non-industrial private forest owners (49%), private forest industry (23%), and the rest by the National Property Board, the state forest company Sveaskog Co., public bodies such as municipalities and regions, the church and forest commons (28%) (Fig. 2, SLU, 2018).

2.2. HCVF habitat network functionality

As the only available complete compilation of HCVFs, we used the national Swedish database (Anon., 2017). Documented conservation values of HCVFs include naturalness, which is indicated by dead wood in different stages of decay (Ylisirniö, Mönkkönen, Hallikainen, Ranta-

Maunus, & Kouki, 2016; Bütler, Angelstam, & Schlaepfer, 2004), multi-layered old-growth vegetation structure (Edenius, Brodin, & White, 2004) and presence of indicator species (Esseen, Ehnström, Ericson, & Sjöberg, 1997; Norén, Nitare, Larsson, Hultgren, & Bergengren, 2002; Timonen et al., 2010). This database with HCVFs $\geq 10 \times 10$ m covers all five Swedish forest ecoregions (Fig. 2) as of 2015-12-31, and includes several categories of long-term and short-term formally protected, voluntarily set aside areas (for definitions see Table 1), and unprotected areas with documented conservation values.

Site class productivity determines the local and regional distribution of different forest types and disturbance regimes (e.g., Angelstam, 1998). As a proxy of the regional variation in productivity, we stratified protected and unprotected HCVFs by altitude and latitude (Fig. 2), both being strongly related to the length of the vegetation period (Lindgren, Ying, Elfving, & Lindgren, 1994). The altitudinal ranges 0–100, 100–200 m a.s.l. represent the area below the level of the highest post-glacial marine limit in Sweden; i.e. a region dominated by sedimentary parent soil material and thus more productive site types. The ranges 200–400 and 400–600 m a.s.l., on the other hand, represent poorer soils and colder climates. Latitude was mapped in 2-degree steps from 54 to 70.

To identify local areas that contain concentrations of HCVFs, and thus functional habitat networks, we applied mapping of habitat availability (e.g., Manton, Angelstam, & Mikusiński, 2005; Edenius & Mikusiński, 2006) to define areas that satisfy the quantitative habitat requirements of a set of focal forest species. Landscapes with sufficient amounts of habitats for those species (i.e. “hotspots”) were thus spatially contrasted with intensively managed landscapes where those habitats requirements were not satisfied (i.e. “coldspots”). To select species for modelling of habitat availability we applied the focal species approach (Lambeck, 1997). This approach is based on the idea that conservation of specialised and area-demanding species can contribute to the conservation of many other naturally co-occurring species (Roberge & Angelstam, 2006).

Being validated as good biodiversity indicators in old-growth forests (Juutinen & Mönkkönen, 2004; Virkkala & Rajasärkkä, 2006) and well recognised by the public (Mikusiński, Roberge, & Fuller, 2018), we focused on resident forest bird species dependent on the high invertebrate abundance found in old-growth forests (e.g., Pettersson, Ball, Renhorn, Esseen, & Sjöberg, 1995). Ottvall et al. (2009) pointed out the importance of structurally complex forests, i.e. dense multi-storey forests with numerous old trees and often with a high proportion of standing and lying dead wood for several of these birds. Thus, we assumed that HCVFs match the habitat requirements of the selected focal bird species.

Angelstam et al. (2004) compiled data about the habitat requirements of 17 focal forest bird species, most of them listed in the EC Birds Directive Annex I, and some additional specialised species selected to represent land cover types linked to both natural and cultural forest disturbance regimes. Several of these bird species are used as indicators of biodiversity in relation to the national Swedish environmental objectives (Green, Haas, & Lindström, 2020, www.sverigesmiljomal.se). The median required suitable habitat patch size for the focal bird species listed by Angelstam et al. (2004) was 50 ha and the median minimum proportion of suitable habitat at the landscape level was 20%. Given that the average size of HCVF polygons (4.8 ha) is an order of magnitude smaller compared to species' requirements, and that birds have the ability to find resources in adjacent patches, we modelled two alternative virtual focal bird species (e.g., Mikusiński & Edenius, 2006) with 5 and 50 ha area requirements for “hotspots”. These area requirements correspond to those of different resident bird species, such as the guild of forest tits with smaller habitat area requirements (Passeriformes, Paridae), and specialised birds (e.g., three-toed woodpecker (*Picoides tridactylus*) and Siberian jay (*Perisoreus infaustus*)) with larger habitat area requirements.

We used the following habitat modelling approach (Manton et al.,

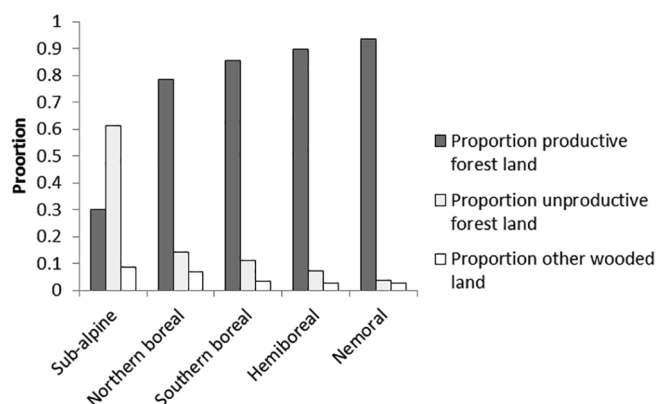


Fig. 3. Proportion of productive and unproductive forests and the proportion of other wooded land in Sweden's forest ecoregions (data from SLU, 2018, and Cornelia Roberge, personal communication, 2018-09-10).

Table 1

Conservation instruments representing the formally protected areas and voluntarily set-asides included in the High Conservation Value Forest database (Bovin et al., 2017:10), and the areas and proportions of these categories in Sweden (Skogsstyrelsen, 2019a). Additionally, the data base contains unprotected forests with high nature values. The total area of forest land in Sweden is 28.1 million ha (SLU, 2019).

Category	Conservation instrument	Area (10 ³ ha)	Area proportion (%)
Formally protected (The Environmental Code (Miljöbalken chapter 7))	National park	1751	6.2
	Nature reserve (including nature conservation areas)		
	Biotope protection	29	0.1
Formalised set-asides(The Land Code (Jordabalken chapter 7))	Natura 2000 (in 2019 a total of 90.4% of those were in national parks and nature reserves)	131	0.5
	Nature conservation agreements with the Swedish Forest Agency (e.g., conservation areas in the state forest company Sveaskog's Ekoparks, Bergvik's white-backed woodpecker areas)	143	0.5
Other types of formal protection	Agreements between state agencies and reserves under establishment	389	1.4
Voluntary set-asides	Swedish Forest Agency's woodland key habitats, Swedish Forest Agency's Nature conservation objects (lower class woodland key habitat), Forest companies' woodland key habitats, State-owned near-natural forests ("SNUS")	1210	4.3

2005). First, all HCVF polygons were converted to a 50 × 50 m raster grid for the less-demanding virtual species, and 100 × 100 m for the more demanding species. Second, we defined functional forest patches by buffering HCVF polygons by 50 m (100 m distance between patches) in the first case, and by 200 m (400 m between patches) in the second. In this way, we linked adjacent HCVF polygons, thus assuming that our virtual species have the ability to disperse across adjacent patches to satisfy their resource needs. Even if inter-patch distances acceptable for all resident boreal forest birds are not available, the parameter values used here correspond to empirical knowledge about resident boreal bird species (e.g., Åberg, Swenson, & Angelstam, 2003; Jansson & Angelstam, 1999). Then sufficiently large (5 and 50 ha, respectively) stands were selected. Third, we used nearest neighbourhood analysis for the two virtual species' habitat stands using moving window sizes of 2x2 km and 5x5 km, i.e. their "landscape level", respectively. For each of the resulting probability surface maps we adopted a minimum habitat availability threshold of 20% to define suitable functional HCVF networks at the landscape level (Angelstam et al., 2004). This threshold value is consistent with a significant volume of research relating to habitat needs across many species (e.g., Hanski, 2011; Svancara, Brannon, Scott, Groves, Noss, & Pressey, 2005).

The habitat models of the two virtual species were first applied on formally protected HCVF areas only, and then on the network composed by both formally protected and unprotected HCVFs. This resulted in four models (two virtual species for two groups of HCVFs) of estimates of functional connectivity. We presented the results numerically as the relative proportions for the five forest ecoregions of (i) all HCVF patches irrespective of their size (per definition 100%), (ii) only for sufficiently large HCVF patches (suitable stands), and (iii) the amount of sufficiently large HCVF patches with functional connectivity (suitable tracts) at the landscape level. We also made maps showing the location of functional tracts ("hotspots") for entire Sweden. Areas that are distant from hotspots (i.e. "coldspots") were located by buffering the functional tracts by 10-km intervals starting at 20 km.

To validate our models we employed data from the systematic long-term biodiversity monitoring programme on birds, i.e. the Swedish Bird Survey (SBS; Green et al., 2020; www.fageltaxering.lu.se). Specifically, we used data from the 716 fixed bird survey routes, 500 of which have been surveyed yearly in early summer since 1996. The fixed routes are placed in a completely systematic 25x25 km grid all over Sweden, and each route consists of an 8 km long line transect following a 2x2 km square. Along the routes, all birds seen and heard are identified to species and counted. Being systematically located, the survey routes sample all major habitats in approximately the same proportion as the habitat occurrence on national and ecoregional levels (for more details see Green et al. (2020) and www.fageltaxering.lu.se).

Based on these data, we calculated the frequency of occurrence (proportion of routes with observations) of three specialised forest bird species on the fixed routes during the period 2008–2017. These were

three-toed woodpecker as a dead wood specialist, and Siberian jay and Siberian tit (*Poecile cinctus*) as a more and a less area-demanding member of the guild of resident boreal bird species, respectively. All three species selected have been assessed as good biodiversity indicators of boreal structurally complex forests old-growth forests (Juutinen & Mönkkönen, 2004; Virkkala & Rajasärkkä, 2006; Ottvall et al., 2009), and are established biodiversity indicators in Sweden. We compared routes completely within HCVF-tracts (i.e. "hotspots") and routes completely outside HCVF-tracts (i.e. in "coldspots", see Green (2019) for more details). We restricted this validation to the sub-alpine, and south and north boreal, ecoregions of Sweden because the distributions of these three focal species are restricted to that part of the country. In total, data from 33 "hotspot" routes and 42 "coldspot" routes were included from the sup-alpine ecoregion. From the north and south boreal ecoregions there were 25 "hotspot" routes and 266 "coldspot" routes.

2.3. Fulfilment of Aichi target #11

Following Hong and Shim (2018), we view CBD's Aichi target #11 with its quantitative and qualitative criteria reflecting conservation science as an appropriate normative model for assessing GI functionality. To evaluate to what extent Sweden satisfies CBD's Aichi target #11 of 17% protected areas we considered the data on the different kinds of formally protected areas, voluntarily set-asides, nature consideration areas and unproductive land officially compiled as potential assets to meet CBD's Aichi target #11 (see Anon. (2018) and Skogsstyrelsen (2019a)). To evaluate the effectiveness of these conservation instruments we also compared these conservation instruments with respect to their size, duration, decision-making, control and method for monitoring.

First we compared the raw area figures for formally protected and voluntarily set-aside forests to the figures when also considering variable retention and unproductive land as protected. We then used the final results from the modelling of habitat network functionality (Fig. 6), to derive the mean proportion for both virtual species of all HCVFs contributing to functional green infrastructure. We suggest that this proportion may be considered as a correction factors to evaluate the effectiveness of the current protected area system network for the different ecoregions. As an estimate of the lower biodiversity value in unproductive forest we used Hämäläinen's et al. (2018) estimate that Swedish productive forests compared to unproductive forests have on average between 1.8 (rocky outcrops) and 3.6 (forested mires) higher volumes of dead wood. As dead wood is a widely used indicator of forest biodiversity and with > 7000 associated species in Fennoscandia (Stokland, Siitonen, & Jonsson, 2012), it represents a relevant proxy for the GI-contribution from unproductive forests. Hence, to estimate GI-functionality we divided the area of unproductive forests by 3 as this category is strongly dominated by mire forests. For comparison we also

Table 2
Basic information about four groups of conservation instruments in Sweden (Anon, 2018; Skogsstyrelsen, 2019a), including two types of formal protection (for details see Table 1), voluntary set-aside areas, nature consideration areas and unproductive forests (from Statskontoret (2007) and Skogsstyrelsen. (2019a)).

	(i.i) Formal; according to the Environmental Code	(i.i.i) Formal; according to the Land Code	(ii) Voluntary set-aside areas	(iii) Nature considerations (§ 30, Forestry law)	(iv) Unproductive (wood production < 1m ³ ha ⁻¹ yr ⁻¹) (§ 13a, Forestry law)
Area and proportion of all forest land in 2019, totalling 28.1 10 ⁶ ha	(i.i and i.ii) 2335 10 ³ ha (8.3%)		(ii) 1210 10 ³ ha (4.3%)	(iii) 426 10 ³ ha (1.5%)	(iv) 3,239 10 ³ ha
Area and proportion of productive forest (wood production > 1m ³ ha ⁻¹ yr ⁻¹) in 2019, totalling 23.6 10 ⁶ ha	(i.i and i.ii) 1382 10 ³ ha (5.8%)		(ii) 1,210 10 ³ ha (5.1%)	(iii) 426 10 ³ ha (1.8%)	(iv) (11.5%) (iv) NA
Aim	National park, nature reserve: conserve and develop nature of high value for plants, animals and people	Biotope protection: conserve terrestrial or aquatic habitat for threatened species	A complement to formal protection	Consideration to biodiversity conservation in managed forest	Wood harvest not recommended
Establishment	1909 and 1964, respectively	1993	1995	1979	1979
Target size	Usually > 20 ha	Variable	> 0.5 ha	< ca 0.5 ha	> 0.1 ha
Duration	Permanent	Variable	Unknown	Unknown	Permanent
Decision by	Parliament, Government, County, Municipality	Agreement between the State or Municipality and owner	Land owner	Parliament, Government, Forest Agency	Parliament, Government
Control	County	State	Forest certification	Forest Agency	Forest Agency
Monitoring	Georeferenced GIS polygons	Georeferenced GIS polygons	GIS data and questionnaires	Random field sampling	National Forest Inventory

present the uncorrected area estimates for all four groups of conservation instruments (Table 2, Fig. 2).

2.4. Implication of forest ownership structure

The type of land ownership is of paramount importance when planning conservation actions (Lockwood, 2010). As a proxy for the opportunity for landscape planning we used forest ownership categories, which reflect different opportunities, objectives and approaches to landscape planning (cf., Andersson, Angelstam, Elbakidze, Axelsson, & Degerman, 2013). We defined three groups. First, public owners with policy mandates to focus on a wide range of material and immaterial forest landscape values, such as municipalities (Richnau et al., 2013), the National Property Board (Eliasson & Törnlund, 2018) and the Board of Fortifications. Second, forest industry, especially where forest holdings form contiguous large blocks, and the state forest company Sveaskog which has both explicit and implicit obligations to consider other forest benefits than wood (SOU, 2002). Third, non-industrial private owners, which normally own small properties (the median class is 6–20 ha, Statistics Sweden (2014)), and whose orientation towards wood production (Nordén, Coria, Jönsson, Lagergren, & Lehsten, 2017) normally increases with property size (e.g., Eggers, Lämås, Lind, & Öhman, 2014). Landowner organizations representing the latter category argue against formal protection and strong legal regulations, which are seen as threat to their ownership rights and freedom to manage their forests (Sténs & Mårald, 2020).

Taken together, the local and regional mix of these three ownership categories is likely to affect the applicability of spatial planning over entire landscapes. Here, we considered these three ownership categories as having three levels of planning complexities from low (public owners) to moderate (forest industry) and high (non-industrial private owners). To understand the potential impact of forest ownership categories, we combined the HCVF data and land ownership data with GIS, and derived the proportions of protected versus unprotected HCVF stands across ownership categories across ecoregions.

3. Results

3.1. HCVF habitat network functionality

The total area of all the 641,095 HCVF polygons in the original database was 3,099,450 ha, with an average size of 4.8 ha. The number and size of HCVF patches varies greatly between formally protected and unprotected HCVFs. While formally protected areas (i.e. national park, nature reserve, biotope protection, conservation agreements) consisted of 231,386 polygons with a mean size of 10.1 ha and a total 2,348,411 ha, unprotected HCVF consisted of 409,709 polygons with a mean size of only 1.8 ha covering a total of 751,039 ha. About 60% of all HCVFs were located in the sub-alpine forest ecoregion, which covers only 11% of the terrestrial land base in Sweden (Fig. 2). In this ecoregion HCVFs constituted 58% of the forest area, and 90% of the HCVFs were formally protected. In stark contrast, the other ecoregions only comprised small area proportions (4–6%) of HCVFs, with about half of them being formally protected. There were two distinctive groups in the size distribution of HCVF patches among ecoregions. First, the sub-alpine forest ecoregion was dominated by patches exceeding 10,000 ha with those > 100,000 ha covering 30% of the total area (Fig. 4). Second, the patch size distributions in the four other ecoregions were similar, with most patch sizes being smaller than 1000 ha (Fig. 4). The proportion of all HCVFs (protected and unprotected) varied substantially along the altitudinal and latitudinal range (Fig. 5). Below 100 m a.s.l. there was no latitudinal trend, and for the altitudinal interval 100–200 m a.s.l. the distribution of HCVFs patches was heavily skewed towards NW Sweden. For the interval 400–600 m a.s.l. the proportion of HCVFs increased rapidly from 60 degrees northern latitude. Due to the regional topography, land at or above 400–600 m a.s.l.

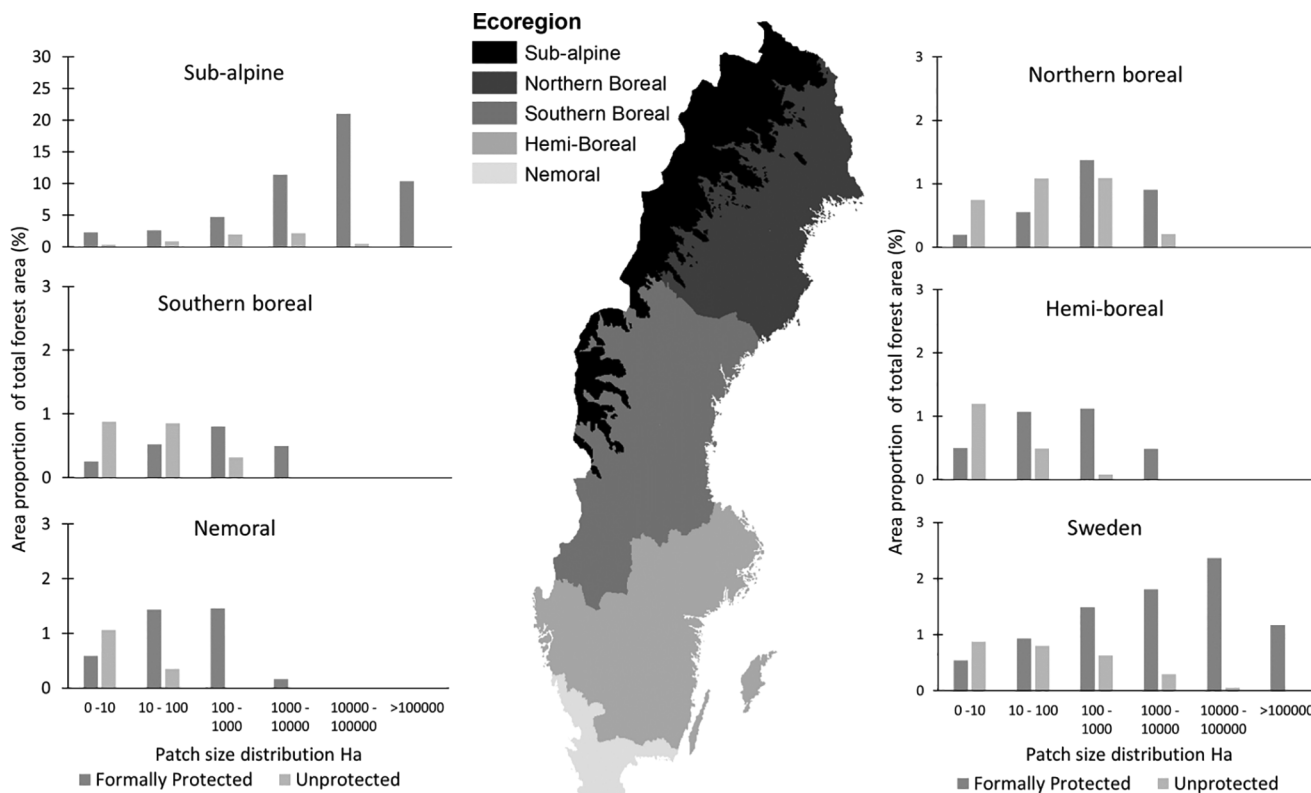


Fig. 4. Size distribution of formally protected HCVF areas (dark grey bars) and unprotected HCVFs (grey bars) according to Anon. (2017) in the five forest ecoregions, and entire Sweden. Note that the range of values on the y-axis is 10 times larger for sub-alpine forests than all other histograms.

was not found below 60 degrees northern latitude; further north the proportion of HCVFs increased steeply with increasing altitude and latitude.

We explored the results from mapping of habitat availability focusing on two aspects of HCVFs as green infrastructure. First, regardless of the criteria used, in the sub-alpine forest ecoregion > 99% of HCVF patches were sufficiently large, and 95–97% constituted functioning habitat networks (Fig. 6). The habitat network functionality in the other forest ecoregions was dramatically different, and deteriorated towards

the south. For the less demanding species (Fig. 6 lower panels), 95–99% of the existing suitable forest stands were large enough, while only 41–83% of these formed functional networks at the landscape level. For the more demanding species (Fig. 6 upper panels), the figures were 71–92% and only 14–55%, respectively. Second, although the total area of HCVFs was larger when including both the protected and unprotected areas, the proportion of functional HCVF patches decreased as the unprotected HCVFs are largely situated in heavily fragmented landscapes. Hence, the relative contribution to habitat network

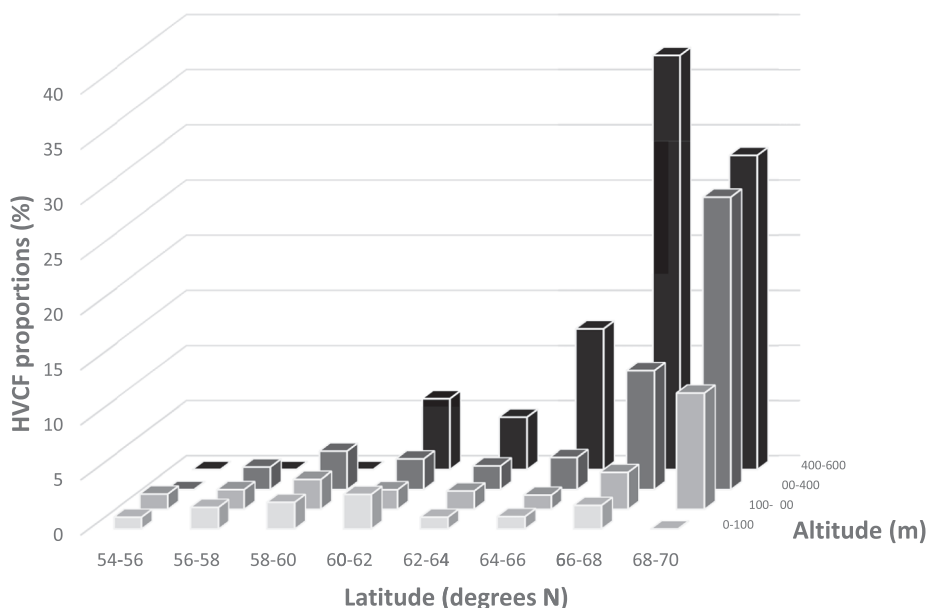


Fig. 5. Proportion of all HCVFs (protected and unprotected) along the altitudinal and latitudinal gradients in Sweden.

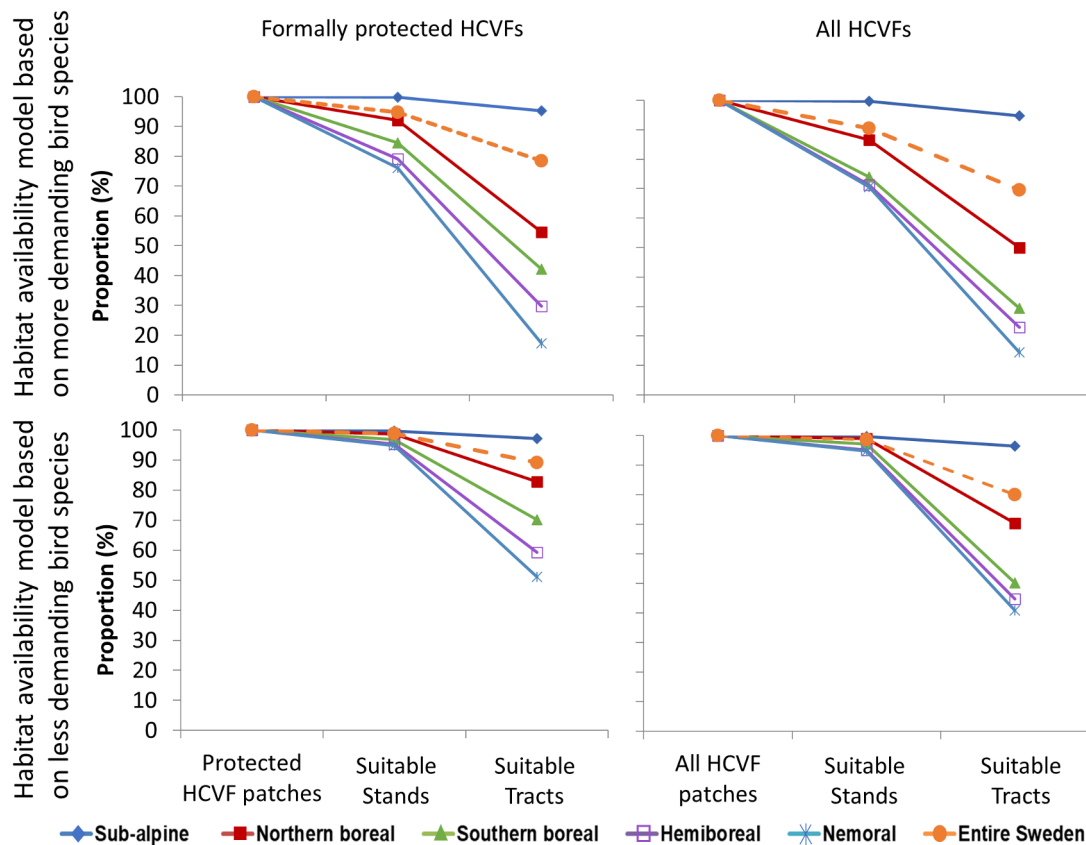


Fig. 6. Proportions of HCVFs contributing to habitat network functionality (i.e. suitable tracts forming “hotspots” in Fig. 7) for a less (bottom) and more (top) demanding virtual focal bird species, and for only formally protected (left) vs. all (right) HCVFs, across Sweden’s five forest ecoregions. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

functionality (Fig. 6, left and right panels, respectively) shows that the proportion of functional HCVF patches for entire Sweden declines from 78% for protected to 70% for all HCVFs for more demanding species, and from 89% vs. 80% for the less demanding virtual species. However, while for sub-alpine forest there was no difference (both 95%), in the four ecoregions focusing on intensive wood production, the mean proportion of functional HCVFs declined from 36 to 29%.

This is visualised using maps showing the distribution of both suitable tracts (“hotspots”), and of areas far from them (“coldspots”) (Fig. 7). For both virtual species “hotspots” were mostly limited to the sub-alpine forest ecoregion and adjacent parts of the northern boreal forest ecoregion. Few hotspots occurred in other forest ecoregions. Conversely, for the more demanding virtual species there were several distinct “coldspots” in both southern and northern Sweden at lower altitudes.

The frequency of occurrence of the three species selected for validation using fixed survey routes and within “hotspot” and “coldspot” areas in northern Sweden corroborated the conclusions of the modelling approach (Fig. 8). In the sub-alpine forest ecoregion where a large part (95–97%) of both protected and unprotected HCVF-patches formed functional habitat networks, the three species had statistically higher frequency of occurrences (df = 1 for all tests) on the fixed routes in “hotspots” compared to “coldspots” (three-toed woodpecker: chi-square 8.7, p = 0.003; Siberian jay: chi-square 15.5, p = 0.0001; Siberian tit: chi-square 8.3, p = 0.004). In the combined southern and northern boreal ecoregions the frequency of occurrences on the fixed routes in “hotspots” compared to “coldspots” was significantly higher for the three-toed woodpecker (chi-square 6.8, p = 0.009), but without any difference for the Siberian jay (chi-square 0.5, p = 0.47) and Siberian tit (chi-square 0.09, p = 0.76). Occurrences within “hotspots” in the sub-alpine forest ecoregion were higher than in “hotspots” in the two

boreal forest ecoregions (three-toed woodpecker: chi-square 3.5, p = 0.06; Siberian jay: chi-square 15.5, p = 0.0005; Siberian tit: chi-square 10.3, p = 0.001).

3.2. Fulfilment of Aichi target #11

The size, duration, decision-making, control and method for monitoring of the four conservation instruments suggest that they differ in conservation effectiveness (Table 2). There was a clear decline in the patch size and duration from formally protected areas (> 20 ha and permanent) via voluntarily set-asides to nature consideration areas (< ca. 0.5 ha and unknown). With respect to decision-making regarding the four conservation instruments, for all but the voluntary set-asides the decisions are made by public bodies. The methods for monitoring ranged from georeferenced GIS data for formally protected areas to questionnaires and random field sampling for the other instruments.

At the national level, Swedish forests with conservation instruments currently cover 26% of the forest area, excluding current nature consideration areas the figure is 24% (Fig. 9). However, further considering the proportion of HCVF patches actually contributing to habitat network functionality (proportion of functional tracts in Fig. 6), and adjusting for the lower biodiversity value of unproductive, reduced this proportion to 12%. Of this, 6% of forest land was formally protected, 3% voluntary set-aside and 3% unproductive. The sub-alpine forest ecoregions stood out with a total of 72% of the total forest area potentially contributing to GI, of which 54% contributed to Aichi target #11, and 44% was formally protected. For the four other forest regions where the focus is on wood production the corresponding numbers were 14–23%, 3–8% and 1–3%, respectively, of all forests.

Fig. 9 summarises two key aspects of the distribution of these four

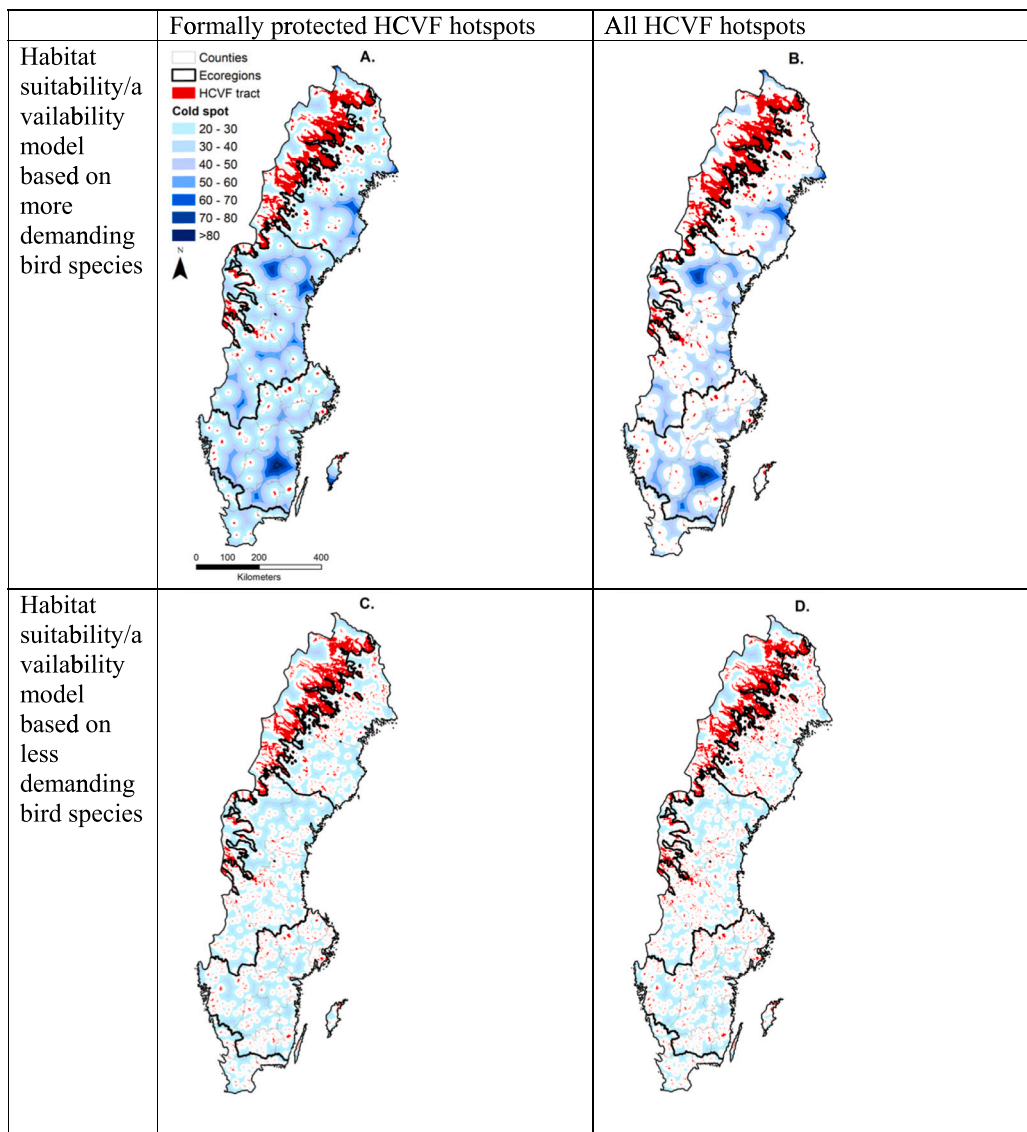


Fig. 7. Location of “hotspots” in northwest Sweden (red) and “coldspots” (blue tone denoting distance in km from hotspot edge) for two habitat availability modelling alternatives representing less and more demanding bird species (different rows) and formally protected HCVFs and all HCVFs (different columns). The five forest ecoregions are marked with thick lines (cf. Fig. 2). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

types of set-asides as components of the GI. The first is the large difference between the sub-alpine forests, which is mostly composed of unproductive forests of little economic interest, versus the four other forest ecoregions focusing on maximum sustained yield forestry. The second is the considerable difference between the total area of different set-aside types and an estimate of the functionality as GI for biodiversity conservation by considering the role of the limited connectivity (Fig. 6) and the lower biodiversity conservation value of unproductive forest.

3.3. Implication of forest ownership structure

The variation in the regional mix of the three land owner categories is strong in Sweden (see map Fig. 10). In the sub-alpine forest ecoregion, the National Property Board is the predominant public owner of HCVFs amounting to > 95% of the protected and > 60% of unprotected HCVFs. For the other four ecoregions, there was a transition in HCVF ownership from the northern boreal forest ecoregion to the nemoral forest ecoregion, with non-industrial private forest owners dominating for protected and especially unprotected HCVF (Fig. 10).

These results suggest that, overall, the opportunity for landscape planning at multiple spatial scales decreases from north to south. Given the different distribution of forest ownership among forest ecoregions, we found decreasing opportunities for spatial planning of green infrastructure at the landscape level from north to south, which parallel the trend in deteriorating habitat network functionality (Figs. 6 and 7).

4. Discussion

4.1. Poor representation and connectivity of HCVFs

This study shows that the long history of forestry intensification in Sweden has clearly affected the ecoregional representation and connectivity of forest habitat network, and therefore its functionality as green infrastructure. High levels of habitat loss and fragmentation have led to small patch sizes and to low densities of remaining HCVF patch fragments throughout most of the country. Although in the sub-alpine ecoregion HCVFs still constitute a functional network of suitable habitat for the focal bird species that cannot cope with intensive forest management, this is generally not the case in the four lower altitude forest

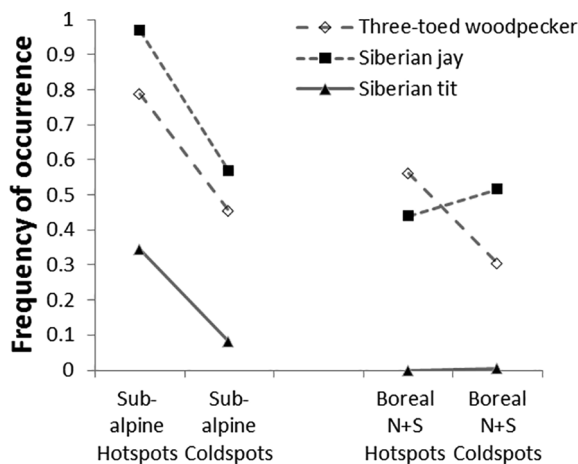


Fig. 8. Validation of spatial models of habitat availability for virtual species (Figs. 6 and 7) using the frequency of occurrence of three resident focal boreal forest bird species in the Swedish sub-alpine forest ecoregion (left) and the boreal (north + south) forest ecoregion (right) based on data from the fixed routes within the Swedish Bird Survey (Green, 2019; Green et al., 2020). (“Hotspots” and “Coldspots” refer to Fig. 7.)

ecoregions that focus on high sustained yield wood production. Even if most HCVF stands are large enough to be suitable for focal bird populations in these ecoregions, only a fraction of these stands were sufficiently connected to form functional green infrastructure, and this fraction decreased from north to south.

This study suggests that the intensively managed matrix surrounding HCVFs provides limited habitat for species typical of old growth forests. This is indicated by the observation that the frequency of occurrence of the three forest indicator bird species was higher inside vs. outside functional tracts of HCVFs in the sub-alpine ecoregion, and also higher in the larger functional tracts in the sub-alpine forest than in the smaller tracts in the northern and southern boreal ecoregions. Consequently, the intensively managed matrix surrounding “hotspots” provides a limited contribution to functional habitat networks. Similarly, Orlikowska, Svensson, Roberge, Blicharska, and Mikusiński (2020) showed that the capacity of smaller protected forest areas to provide habitat for species that require HCVFs depends on the habitat quality of the surrounding matrix. Unfortunately, forests managed for intensive wood production have lower bird species diversity compared to HCVF habitats in Sweden (e.g., Roberge & Angelstam, 2006; Lindbladh et al., 2019). This is linked to the insufficient level of tree

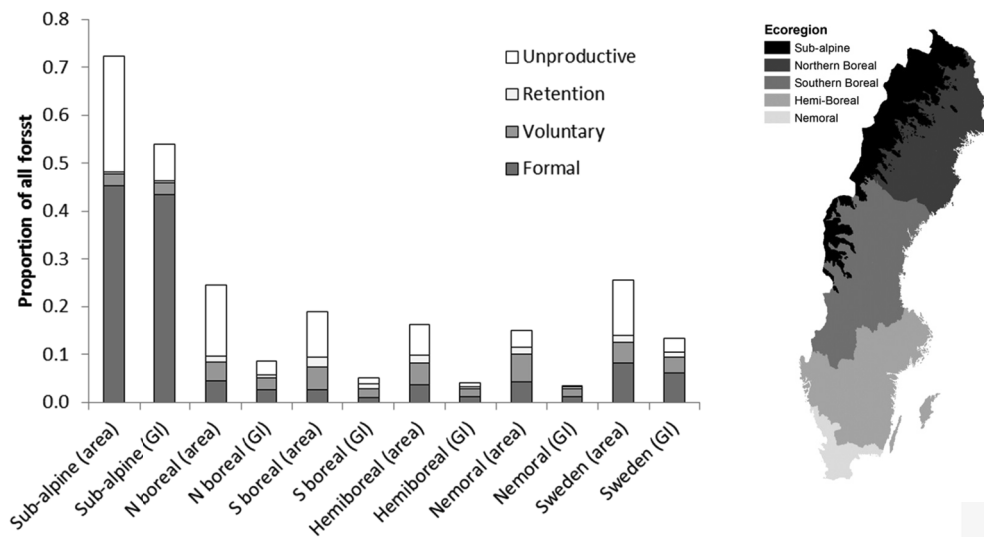


Fig. 9. Focusing on three categories of set-asides with harvesting restrictions on forest and other wooded land (formally protected, voluntary set-aside, unproductive) plus retention trees within harvested stands, this figure summarises the differences between total area and estimates of green infrastructure (GI) functionality as contributions to Aichi target # 11. Note that retention areas refer to Aichi target # 7 about sustainable management of the matrix around protected areas.

retention within current Swedish management practices (Gustafsson et al., 2012; Basile, Mikusiński, & Storch, 2019). Not surprisingly, many forest bird species have thus declined in Sweden in the last four decades (Green et al., 2020; Wirdheim, 2020), and similar trends are also observed in the neighbouring countries (Angelstam et al., 2004; Lehikoinen & Virkkala, 2018). This trend probably goes beyond bird communities, since connectivity at the landscape and regional levels has been reported as particularly critical also for other forest dependent taxa, such as wood-inhabiting fungi (Abrego, Bässler, Christensen, & Heilmann-Clausen, 2015; Nordén et al., 2018).

The validation of our approach to map functional habitat networks using the frequency of occurrence of the three indicator bird species worked well, and needs to be replicated also for other ecoregions and taxa. However, our habitat mapping relies on two optimistic assumptions. First, the only existing Swedish HCVF dataset which we used is composed of a mix of forest types with different habitat qualities. While the two habitat availability modelling variables patch size and connectivity can readily be derived empirically and used in spatial wall-to-wall analyses (e.g., Elbakidze et al., 2016), detailed information on actual forest stand and patch quality is not available for entire Sweden (e.g., Manton et al., 2005). Low habitat patch quality would reduce the functionality of the forest habitat network(s) even further, especially in the four forest ecoregions where the history of intensive forestry is long and dominates, and the amount of old-growth properties such as dead wood are low (e.g., Jonsson et al., 2016). Additionally, small forest patches are subject to micro-climatic edge effects that reduce the effective patch area (e.g., Aune, Jonsson, & Moen, 2005; Ruete, Snäll, & Jonsson, 2016). This has long-term negative effects on old-growth dependent lichens and bracket fungi species (Jönsson, Ruete, Kellner, Gunnarsson, & Snäll, 2017). Moreover, the habitat quality of protected forest areas in Sweden does not reflect that of naturally dynamic forests (Felton et al., 2019; Hedwall & Mikusiński, 2015). In fact, due to the reduced role currently played by natural disturbance in boreal forests (e.g., Angelstam, 1998), HCVFs not directly affected by forestry may nevertheless gradually decrease their levels of naturalness (Uotila, Kouki, Kontkanen, & Pulkkinen, 2002). Moreover, analyses separating different forest types show an even more serious fragmentation of the Swedish forest landscape (e.g., comparisons of Scots pine vs. Norway spruce HCVFs (Angelstam & Andersson, 2013). Second, we assumed that all HCVF patches would survive in the long-term. However, the longevity of different types of protected areas differs among categories. While formal area protection is permanent (national parks, nature reserves and biotope protection) or agreed for up to 49 years (conservation agreements) (Table 2), voluntary set-asides of forest stands have uncertain duration. Changes of forest owner or seized adherence to

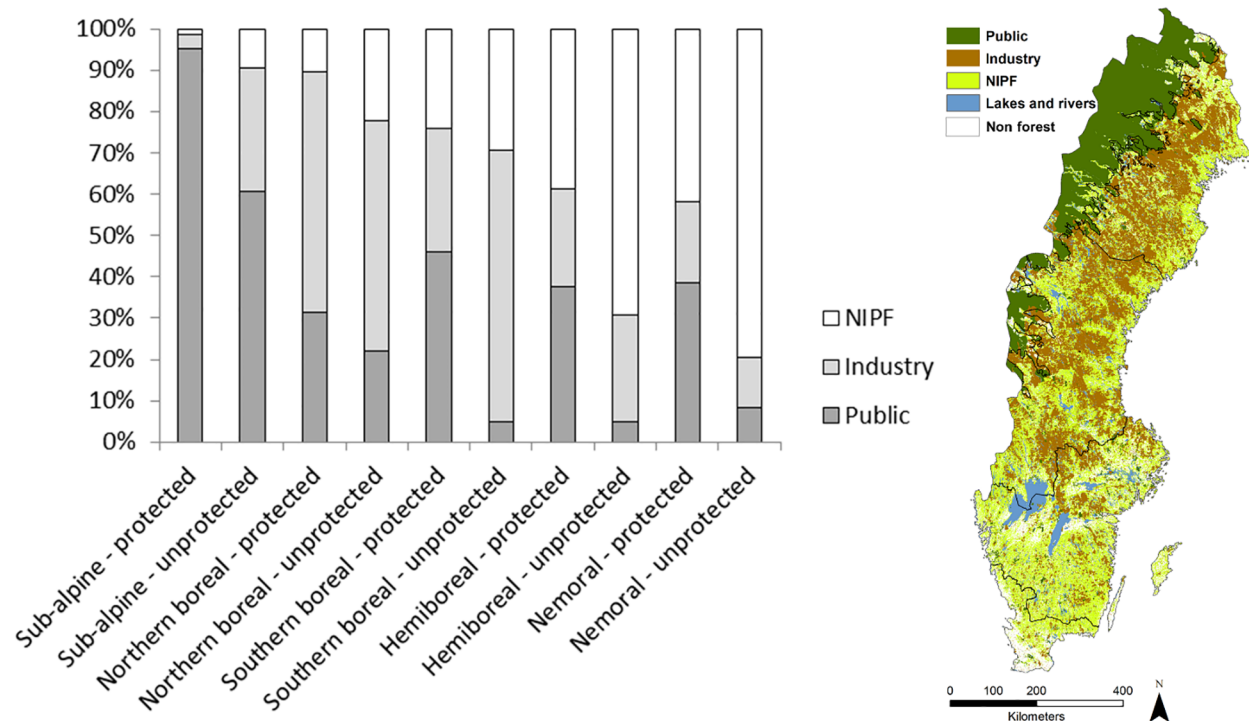


Fig. 10. Histogram showing which group of forest owner category that owns protected versus unprotected HCVF stands in Sweden's five forest ecoregions. Public represents the Environmental Protection Agency, National Property Board and Swedish Fortifications Agency, and municipalities; Industry is represented by the state forest company Sveaskog Co. and the forest companies Bergvik, SCA, Holmen etc.; and Non-Industrial Private Forest owners (NIPF), including commons representing private forest owners' collectively owned land. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

forest certification may, however, jeopardize the longevity of HCVFs with short notice.

To conclude, even if our assumption that all HCVFs represent potential habitat for relevant species is a simplification of reality, our results provide a baseline for assessing the current situation regarding biodiversity, as well as a proof of concept that HCVFs can be used at a strategic level when planning for functional green infrastructure at a landscape scale.

4.2. Conservation targets are not fulfilled

Whether or not Sweden can meet the criteria of Aichi target #11 of 17% of protected areas depends how effective different conservation instruments are (Table 2), and if their representation and connectivity are taken into account. In this study, we showed that 24% of all Swedish forests have some kind of harvesting restrictions (Skogsstyrelsen (2019a)). However, this figure drastically reduces to 12% when considering only functionally connected areas, and the lower conservation value of unproductive forests. We also showed that habitat network functionality was largely dependent on formally protected HCVFs, with unprotected forests playing a marginal role. Adding the much smaller and more fragmented unprotected HCVFs actually led to a smaller overall proportion of functional habitat (see Fig. 6). This illustrates the negative effect of increased fragmentation on connectivity.

We also found that the ecoregional distribution of HCVFs is heavily biased. In the sub-alpine ecoregion 45% of all forest land is formally protected, as opposed to the other four forest ecoregions, where only 3 to 4% of forest is under formal protection. A similar north-south gradient in the size of protected areas is also found in Finland (Virkkala et al., 2018). The creation of new national parks and nature reserves over the past decades, most of which are concentrated in unproductive areas and the sub-alpine forest ecoregion (e.g., Angelstam et al., 2011), has not changed the regionally unbalanced distribution of protected

forests in Sweden (c.f., Nilsson & Götmark, 1992; Svensson et al., 2019). However, while protected areas and voluntary set-asides have indeed increased (Elbakidze et al., 2013), the harvesting of remaining near-natural forest remnants never subject to clear-felling has continued at a faster rate (Svensson et al., 2019). Thus, the net effect of conservation and intensification is negative.

While unproductive forests do contribute to biodiversity conservation for some forest types (e.g., Hämäläinen, Strengbom, & Ranius, 2018; Jönsson & Snäll, 2020), the forest habitat composition is different, the species richness is lower and some demanding (red-listed) species are missing when compared to un-managed productive sites (e.g., Hämäläinen et al., 2018). Hence, unproductive forests only provide relevant habitat for some of the species associated to productive forests. The role of unproductive forests is most pronounced in the sub-alpine ecoregion where this land cover type is twice as common as productive forests (Fig. 3).

When considering representation and connectivity to estimate the proportion of HCVFs forming functional habitat networks, as recommended by CBD's Aichi target #11, the situation becomes even more critical in the four forest ecoregions with effective high wood yield forestry. We show that these ecoregions are clearly below (only 3–8% in total) international and national policy targets of 17–20% functional GI (Fig. 9). Locating and protecting additional HCVFs should therefore be a priority in order to secure a representative protected area network as a functional forest GI. However, because the amount of HCVFs is insufficient, forest habitat and landscape restoration (Chazdon et al., 2016; Mansourian, 2017) is necessary.

In addition to stands with harvesting restrictions (formally protected, voluntary set-aside, unproductive), nature consideration areas, and retention trees, within harvested stands may contribute to biodiversity conservation. However, it should rather be seen as a practice that relates to Aichi target #7 on sustainable management than contributing to target #11 about protection. Unfortunately, as shown in

Estonia, the long-term survival of retention trees is low (Rosenvald et al., 2019). Additionally, for Finland Kuuluvainen, Lindberg, Vanha-Majamaa, Keto-Tokoi, and Punttila (2019:1) concluded that: "The development of retention practices in Finland indicates that the aim has not been to use ecological understanding to attain specific ecological sustainability goals, but rather to define the lowest level of retention that still allows access to the market." Sufficient quality of the forest matrix can, however, also support biodiversity conservation. Applying a wider range of forest management systems which emulate natural disturbance regimes has therefore been proposed as a solution (Angelstam, 1998; Fries, Johansson, Pettersson, & Simonsson, 1997). Wide application of this would have reduced the need for protected areas in Sweden (Angelstam & Andersson, 2001). However, the clear-felling system (Matthews, 1991; Puettmann et al., 2012), with gradually lowered final felling ages, has remained as the standard silvicultural system in Sweden, and with low levels of tree retention at final felling (Gustafsson et al., 2012; Simonsson, Gustafsson, & Östlund, 2015).

4.3. Need for regionally adapted landscape planning approaches

To effectively translate GI policies into action in local landscapes, it is crucial to acknowledge that there are different land ownership categories and landscape histories, and that value chains can be based on wood and biomass as well as other ecosystem services (e.g., Jonsson et al., 2019). Coping with this requires regionally adapted approaches that can engage multiple stakeholders and actors through regionally adapted evidence-based landscape planning (Lazdinis, Angelstam, & Püzl, 2019). Therefore, landscape planning needs to match the perceived benefits of forests and spatial extents of people's sense of place (e.g., Stedman, 1999). We found a high heterogeneity of land owner categories among regions and local landscapes in Sweden (Fig. 10), which tracks the large differences in human population density across Sweden's municipalities (Fig. 2). The considerably higher human population density in the nemoral and hemiboreal forest ecoregions in southern Sweden coincides with > 80% non-industrial private forest ownership. The number of forest properties per unit area increases from the north to the south (SLU, 2018).

Separate legal frameworks for forest management, municipal comprehensive and regional planning complicate integrated land-use planning (Stjernström, Pettersson, & Karlsson, 2018). In Sweden, the mandate for planning for forestry aimed at wood production lies with the owners of forest estates. However, operational planning is commonly not carried out by owners, but by wood buyers within the forest industry. While citizens have positive attitudes towards biodiversity (Nordén et al., 2017), and citizens prefer forests with high levels of naturalness and cultural landscapes (Elbakidze et al., 2017), municipalities are traditionally not planning land use outside built-up areas. Instead planning for and purchase of HCVMs to create formally protected areas for biodiversity conservation and human well-being is mainly made by the regional county administrations (Fig. 2). Availability of spatial data with sufficient thematic resolution, and poorly developed knowledge-policy interface at multiple levels are additional barriers for landscape planning (Elbakidze et al., 2015). Indeed, the collaboration of local stakeholders using a facilitated landscape approach (e.g., Angelstam et al., 2019abc) is crucial to ensure the success of any landscape planning initiative. To cope with this we suggest the following three approaches towards landscape planning supporting functional GI in different contexts.

4.3.1. Publicly owned forest landscapes in the sub-alpine ecoregion

Biodiversity can be maintained with different levels of ambition. Characterised as large intact forest landscapes the sub-alpine forests in Sweden are of Pan-European importance for biodiversity conservation (Sabatini et al., 2018; Jonsson et al., 2019). Sweden could therefore expand the current focus on viable populations by also including ecological integrity and resilience in its forest and environmental policy

and objectives (Angelstam et al., 2011; Miljödepartementet, 2014). The north-south orientation of the Scandinavian mountain range is favourable in terms of providing a > 1000-km green belt with high functional connectivity that provides opportunity for dispersal within a particular envelope of climatic conditions. In the sub-alpine ecoregion, the 15 municipalities of NW Sweden's mountain region have key but difficult role in exercising their landscape planning mandate, but they are also limited by declining populations and economy of rural areas (Bjärstig et al., 2018; Carlsson, Lidestav, Bjärstig, Svensson, & Nordström, 2017; Stjernström et al., 2018; Thellbro, Bjärstig, & Eckerberg, 2018). Jonsson et al. (2019) proposed three coping approaches. First, to provide landowners with economic compensation for investments towards conserving the remaining sub-alpine forest located in functional habitat networks. Second, establish compensation schemes to support land-owners to progressively shift from even-aged to continuous cover forest management systems (Valasiuk et al., 2018). Third, develop value chains based not only on economic values, but also on socio-cultural and ecological benefits at different spatial scales (Sayer, 2009; Valasiuk et al., 2018).

4.3.2. Private industrial forest landscapes

The "land-sharing versus land-sparing" discussion provides a framework for comparing different approaches to handle the rivalry between biodiversity conservation and high yield wood and biomass production at a landscape scale (e.g., Edwards et al., 2014). Both modelling (Hanski, 2011; Manton et al., 2005; Tittler, Messier, & Fall, 2012; Rybicki & Hanski, 2013) and empirical studies (Angelstam, Manton, Pedersen, & Elbakidze, 2017; Angelstam et al., 2018; Edwards et al., 2014; Naumov et al., 2018; Pohjanmies, Triviño, Le Tortorec, Salminen, & Mönkkönen, 2017) suggest that where forest naturalness is the vision, land-sparing strategies that segregate wood production and biodiversity conservation are more effective for habitat network functionality than land-sharing. From a biodiversity conservation point-of-view, different site conditions and forest disturbance regimes imply that there are several distinct types of green infrastructure (e.g., Fries et al., 1997), which are different from those that focus on wood production. Large industrial forest owners are already applying a land-sparing strategy by implementing innovative forms of spatial planning. Sveaskog Co's Ekoparks (Angelstam & Bergman, 2004), SCA Co's Diversity parks (Mångfaldspark), as well as Holmen's multiple use planning (Normark, 2015) are three examples. These landscape planning approaches go in the right direction, since they clearly improve habitat network functionality by handling connectivity. Land-sharing attempts to combine wood and biomass commodities with multiple-use and biodiversity conservation across entire landscapes (Ekroos, Rundlöf, & Smith, 2013) are also strategies in that direction. Examples include the maintenance of traditional cultural woodlands (e.g., Garrido et al., 2017). Yet, these cannot replace the establishment and conservation of a functionally connected green infrastructure.

4.3.3. Non-industrial private forest landscapes

In landscapes dominated by non-industrial private forest owners, active landscape planning is complicated. Due to the extreme fragmentation of forest ownership, and the diverse attitudes concerning the responsibility for biodiversity conservation (e.g., Danley, 2018), this ownership category is limited to indirect approaches to landscape planning. Site conditions in terms of soil moisture and nutrient availability can be used to guide green infrastructure maintenance (e.g., Angelstam, 1998). First, sustaining riparian forests on wet rich sites supports both terrestrial and aquatic biodiversity (Angelstam & Lazdinis, 2017). Second, dry poor soils are suitable for conservation management using low intensity prescribed fire (e.g., Fries et al., 1997). Third, the historic spatial division of traditional village systems into in-field for crops on agricultural land and hay-making, and out-field for forest grazing, provide guidance for identifying biocultural forest legacies, such as old deciduous trees on wooded grassland (e.g.,

Dorresteijn, Hartel, Hanspach, von Wehrden, & Fischer, 2013; Mikusiński, Angelstam, & Sporrang, 2003; Manton & Angelstam, 2018). Remnants of such traditional cultural landscapes not subject to Norway spruce plantation (Lindbladh, Axelsson, Hultberg, Brunet, & Felton, 2014), such as wooded grasslands, have similar structures of importance for biodiversity conservation and human well-being (Garrido et al., 2017).

4.4. Forestry intensification or sustainable landscapes?

As a starting point, our study takes the global conservation targets established by the CBD as representing an ecologically informed and internationally agreed normative framework relevant for sustainable forest management in Sweden. We show that current conservation measures do not fulfil the quantitative and qualitative ambitions of CBD's Aichi target #11. In fact, CBD is facing a revision process, and its Zero 2020 draft (CBD, 2020) indicates a higher protected area target than 17%. The rivalry between conservation and wood production in Sweden is an interesting example on how competing narratives over reality and increased tensions may develop (Mårwald, Sandström, & Nordin, 2017; Sandström et al., 2020). With terms like bio-economy, a new forestry intensification discourse is beginning to dominate the previous sustainable forest management discourse, which simultaneously considers economic benefits, biodiversity conservation and rural development (Pülzl, Kleinschmit, & Arts, 2014; EASAC, 2017). The term sustainable landscape captures this (e.g., Selman, 2008). While this study addresses both the quantitative and the qualitative criteria of Aichi #11, the competing bio-economy narrative focuses on increased forest harvest and forestry intensification aimed at supporting societal transitions towards a bio-based economy. Estimating how effective green infrastructures are is a way to identify the boundaries for forestry intensification.

We interpret the creative book-keeping approach in Sweden's national forest programme (e.g., Anon., 2018) as a way to side-line the true spirit of Aichi target #11 by merely focusing on the numerical target of 17% for the entire country (Anon., 2018), and disregarding the qualitative criteria of target #11 such as representativeness and connectivity (Fig. 9). Politically, this has translated into new forestry-related policy issues that aim to strengthen private forest ownership, provide new flexible approaches to forest protection (including in the sub-alpine forests), and further increase forest production (Skogsstyrelsen, 2019b). This has also led to abandonment of the national inventory of HCVFs using the Woodland Key Habitat approach (Wester, Thomasson, & Claesson, 2019), which is unfortunate because locating and protecting HCVFs contributing to green infrastructure could fill critical gaps.

Although specific to Sweden, this study illustrates a critical situation for biodiversity conservation that is relevant for many other countries, and will become exacerbated if plans to increase biomass production under the auspices of bio-economy are realised (see, EASAC, 2017; Kleinschmit et al., 2014; Pülzl et al., 2014). Already Imbeau, Mönkkönen, and Desrochers (2001) concluded that the northern expansion of commercial forestry in eastern Canada is likely to result in the significant decline of several resident species, as has occurred in Finland, and Sweden. The strong push for forestry intensification is already affecting forestry in other northern forest countries such as Russia (Naumov, Angelstam, & Elbakidze, 2016) and Latvia (Angelstam et al., 2018; Naumov et al., 2018). Simultaneously, the strong Linnéan heritage of biological literacy, forest certification emerging in 1992 and systematic mapping of HCVFs using the woodland key habitat approach (e.g., Timonen et al., 2010) allows Sweden as being portrayed as a role model for biodiversity conservation as one dimension of sustainable forest management. The results from this study are not consistent with this optimistic narrative for successful biodiversity conservation. We stress the need to better understand, for countries and ecoregions with different landscape histories, the implications of shifting the focus away

from representative and functionally connected formally protected areas to the narrative that retention and buffer strips are sufficient to maintain an effective green infrastructure for biodiversity conservation. Sweden's long-term experience with attempting multi-scale conservation and intensive forestry provides insights for countries trying to maintain forest biodiversity on other continents (see Felton et al., 2019).

5. Conclusions

Securing an effective even-aged silvicultural cycle satisfying the needs of the forest industry is the overarching objective of the Swedish forestry model. Simultaneously, representative types of green infrastructure for both biodiversity conservation and supply of ecosystem services for human well-being should be secured. Statements that 26–31% of all Swedish forests are not available to systematic clear felling can give the impression that these can be accounted for as “protected areas”. Ignoring the need for ecoregional representation and connectivity, creates the misleading narrative that international, EU and national conservation targets have been met in Sweden. Using the most recent figures on Swedish conservation instruments, and evidence-based knowledge for assessment habitat network functionality using modelling of habitat availability, and a validation approach, we estimate that only 12% of all Swedish forests meet national targets of 20% CBD's and the Aichi target #11 of 17%. Additionally, the ecoregional representation in Sweden is poor. A high proportion (54%) of functional forest tracts were concentrated in the sub-alpine ecoregion. In the four forest ecoregions where intensive forest management is carried out, which cover 89% of Sweden, the estimated functionality is reduced to 3–8%. This study illustrates a clash between a long-term focus on high sustained yield forestry, and the need to maintain and restore HCVFs as contributions to functional green infrastructure representing many different forest types. The large regional variation in the opportunity for landscape planning stresses the need for a portfolio of different approaches. Where land ownership structure is contiguous (e.g., public and industrial forest owners) landscape planning and restoration can be used. For smallholders, other approaches are needed. Site conditions in terms of soil moisture and nutrient availability, and conservation of cultural forest landscapes are two options to guide forest management bottom-up. Finally, we underline that assigning all “areas not available for intensified wood production” as “protected” is not equivalent to effective biodiversity conservation in the spirit of CBD's Aichi target #11. Therefore, we warn other countries against the risks of such a paradigm shift, which will compromise the ability to effectively meet international conservation targets.

Credit authorship contribution statement

Per Angelstam: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Resources, Writing - original draft, Writing - review & editing, Visualization, Supervision, Project administration, Funding acquisition. **Michael Manton:** Conceptualization, Methodology, Data curation, Formal analysis, Investigation, Resources, Writing - original draft, Writing - review & editing, Funding acquisition. **Martin Green:** Conceptualization, Validation, Formal analysis, Writing - original draft, Writing - review & editing. **Bengt-Gunnar Jonsson:** Conceptualization, Validation, Formal analysis, Writing - review & editing. **Grzegorz Mikusiński:** Conceptualization, Writing - review & editing. **Johan Svensson:** Conceptualization, Writing - review & editing. **Francesco Maria Sabatini:** Conceptualization, Writing - review & editing.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.landurbplan.2020.103838>.

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