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Abstract

Loss of natural forests has been identified as a critical conservation challenge worldwide. This loss impede the establishment of a functional green infrastructure as a spatiotemporally connected landscape-scale network of habitats enhancing biodiversity, favorable conservation status and ecosystem services. In many regions this loss is caused by forest clearcutting. Through retrospective satellite images analysis we assessed a 50-60 year spatiotemporal clearcutting impact trajectory on natural and near-natural boreal forests across a sizable and representative region from the Gulf of Bothnia to the Scandinavian Mountain Range in northern Fennoscandia. Our analysis broadly covers the whole forest clearcutting period and thus our study approach and results can be applied for comprehensive impact assessment of industrial forest management. Our results demonstrate profound disturbance on natural forest landscape configuration. The whole forest landscape is in a late phase in a transition from a natural or near-natural to a land-use modified state. Our results provide evidence of natural forest loss and spatial polarization at the regional scale, with a pre-dominant share of valuable habitats left in the mountain area, whereas the inland area has been more severely impacted. We highlight the importance of interior forest areas as most valuable biodiversity hotspots and the central axis of green infrastructure. Superimposing the effects of edge disturbance on forest fragmentation, the loss of interior forest entities further aggravate the conservation premises. Our results also show a loss of large contiguous forest patches and indicate patch size homogenization. The current forest protection share is low in the region and with geographical imbalance as the absolute majority is located in remote and low productive sites in the mountain area. Our approach provides possibilities to identify forest areas for directed conservation actions in the form of new protection, restoration and nature conservation oriented forest management, for implementing a functional green infrastructure.

Key words

Change detection, clearcutting, continuity forest, forest core areas, forest fragmentation, landscape configuration, satellite image, Sweden

Introduction

Globally, the ongoing loss of natural boreal forests has been assessed as the second largest after the forest loss in the tropics, in both absolute and proportional terms (Hansen et al. 2013). In Fennoscandia, this loss is pre-dominantly caused by clearcutting forestry (Kouki et al. 2001), whereas in North America and parts of Eurasia also the impact of beyond-baseline levels of forest fires is contributing (de Groot et al. 2013). Although intact forests still persist on vast areas in many boreal regions (Potapov et al. 2008; Gauthier et al. 2015), the frontiers of natural forest landscapes are being modified and relocated at critical rates (Potapov et al. 2017). In the Fennoscandian boreal biome, where forestry has had major and widespread impact (Kouki et al. 2001), clearcutting continues also in the remaining fragments of natural and near-natural forests (Forest Europe 2015), notwithstanding policies that advocate increasing conservation rates, landscape-context approaches and awareness that favorable conservation status for many target forest habitats and species is not secured (e.g., Mehtälä & Vuorisalo 2007; van Teeffelen et al. 2012; Sverdrup-Thygeson et al. 2014; Orlikowska et al. 2016). The continued escalation of human footprint (Tucker et al. 2018) and loss of intact forest landscapes impedes conservation of biodiversity and ecosystem services (e.g., Watson et al. 2018). Reaching environmental policy goals in the Fennoscandian forest landscape, such as the Aichi Biodiversity Targets (CBD 2010), demands rigorous efforts.

The broad-scale and long-term forestry impact has raised much concern about the ecological integrity of the remaining natural forest fragments (Jönsson et al. 2009; Kuuluvainen 2009; Moen et al. 2014). Remnant forests with temporal and spatial continuity of key habitat attributes function as hotspots for many species (e.g., Paillet et al. 2010) and thus have a critical role for forest biodiversity (Hanski 1999; Ranius & Kindvall 2006; Nordén et al. 2014). Forest continuity implies old-growth habitat attributes present for several tree generations within a defined patch and an uninterrupted supply of continuity patches in a landscape matrix (e.g., Nordén et al. 2014). Continuity is associated with forest interior core areas that are less influenced by proximity to peripheral and external disturbance factors and thus may provide a refuge for natural structures and processes (Riitters et al. 2016; Pfeifer et al. 2017). The conservation significance of core areas and continuity are undoubtedly very high on both habitat and landscape scale, which consequently is reflected in nature conservation policy and planning (e.g., Angelstam et al. 2011; Aksenov et al. 2014; Müller et al. 2018).

In addressing forest conservation and sustainability in landscapes dominated by managed forests, arguments and knowledge accumulate on the need to increase forest protection but also to expand restoration, retention, multifunctional forestry and other conservation-oriented management (e.g., Gustafsson et al. 2012; Lindenmayer et al. 2012). Both remaining core areas and the surrounding forest matrix need to be regarded for persistence and resilience of ecosystem functions, biodiversity and ecosystem services (Mikusiński et al. 2007; Swift & Hannon 2010; van Teeffelen et al. 2012; Aksenov et al. 2014). Accordingly, also forests in a modified state are needed as pathways for species movement and expansion of habitats (Bengtsson et al. 2003), acknowledging the meta-population capacity of the landscape (Hanski & Ovaskainen 2000).

The concept of green infrastructure (GI) has expanded from promoting ecosystem values and human well-being in urban environments (Tzoulas et al. 2007) to a mainstream EU environmental policy (EC

2013). The EU member states are presently implementing GI (e.g., Snäll et al. 2016). Green infrastructure is a strategic and operation planning network of spatiotemporally connected natural and semi-natural habitats that supports and mobilize ecological connectivity, favorable conservation status, ecosystem services and ecosystem multi-functionality at multiple scales, also under ongoing climate change and forest management (Benedict & MacMahon 2002; Liquete et al. 2015; Mehtälä & Vuorisalo 2007; Johnstone et al. 2016). For managed landscapes, biodiversity and sustainability of ecosystems and their services require approaches that address and mitigate habitat fragmentation. Forest areas suitable for protection, restoration and conservation-oriented management need to be identified (Halme et al. 2013; Rybicki & Hanski 2013; Müller et al. 2018). Hence, accurate mapping of valuable forests provides important input to GI-implementation. For mapped gross data based on remote sensing information, which currently is widely applied for landscape impact and change detection (e.g. Tyukavina et al. 2016; Sverdrup-Thygeson et al. 2016; Potapov et al. 2017), it is of particular value to consider and assess ecologically relevant parameters for defining the most important GI-components. As ecological connectivity may be used as a measure for assessing the ecological performance of forest habitats (Lindenmayer et al. 2006), mapping connectivity of continuity forests thus provides needed input.

In this study we addressed the challenges in establishing a functional GI across a large geographic area extending from the Gulf of Bothnia to the Scandinavian Mountain Range in northern Sweden. The area exemplifies a significant and representative part of the Fennoscandian boreal biome with a pronounced influence of forest management. Only 4% of the productive and 7% of all forest land in Sweden is formally protected at present, with the absolute majority located in a narrow zone in the mountainous area (Anon 2017). This is very far from the 17% in Aichi target # 11. Through retrospective analyses of satellite images we sequentially detected clearcuts during the last 50-60 years, and accordingly mapped the forest landscape change trajectory of lost and remaining forest. This time period broadly covers the industrial forest clearcutting era in the study region (Lundmark et al. 2013). We regard the initiation of widespread clearcutting at the middle of the 20th century, with large harvesting areas, soil scarification and artificial regeneration (Ecke et al. 2013), as an onset in the transition to a managed forest landscape. Our study rationale was that identified remaining forest fragments, with or without traces of earlier management, represents components of a functional GI to which protection, restoration or conservation-oriented management should be directed. Since our analyses are based exclusively on remote sensing data, we denote the identified non-clearcut forest patches as “proxy continuous cover forests” (pCF). In a similar approach in pan-tropical forests, Tyukavina et al. (2016) applied the term “hinterland forest” for remote sensing-identified patches without recent disturbance. The spatiotemporal resolution applied in our study allows for operational approaches that complement earlier mapping of intact boreal landscape at pan-national and larger scales, such as those by Potapov et al. (2008, 2011).

The main research questions concerned the spatiotemporal changes in landscape-level configuration as a consequence of long-term clear-cutting forestry, including: 1) How the remaining pCF are distributed across the gradient from coast to mountain; 2) How the amount and distribution of pCF has changed over time; 3) How the distribution of pCF relate to protected forest and to total forest land area over time; and 4) How spatiotemporal forest core area can be assessed by considering edge influence. Our results are discussed with reference to boreal forest loss and fragmentation and to prospects for establishing a functional GI in a landscape that has been and most likely will continue to be dominated by forest management.

Methods

Study region

The 45,755 km^2 study region represents a forest-landscape transition extending from the coastal mid-boreal to the northern boreal and the birch-dominated (*Betula pubescens* ssp. *czerepanovii*) sub-alpine zones (Gustafsson & Ahlén 1996). The predominant tree species are Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*), 46% and 22% respectively (Anon 2017, on productive forest in the County of Västerbotten which cover a dominant share of the study region. The altitude gradient equals about 900 meters from sea level to the alpine tree line, with associated macro-climatic and forest site productivity changes.

The study region is dominated by a managed forest landscape. The coast to mountain gradient represents a historical progression of the south-north and east-west movement of modern forestry. More evident forest exploitation has occurred since the mid-1800s, including a first wave large diameter saw timber harvesting followed by a period of selective logging with some clearcuts, and since the mid-1900s with dominating clearcutting forestry (Lundmark et al. 2013; Ecke et al. 2013).

Data sources

Across a large landscape with multiple land forms, land cover types and land-owner categories, remote sensing with ancillary data presents an opportunity to compile holistic information (e.g, Kennedy et al. 2014). The Landsat program was launched in 1972 as the first program tailored for global cover (Wulder et al. 2012). Satellite image-based change detection have since then successfully been used to map, e.g., land-use change (Muukkonen et al. 2012), deforestation (Potapov et al. 2017) and minimally disturbed forests (Tyukavina et al. 2016).

We used 24 Landsat images from 1973 to 2014 to identify and define remaining forest (Supporting Information A1, A2). The red spectral band was used as this wavelength is suitable for distinguishing changes in forest cover (Potapov et al. 2008; 2011). Since we aimed for creating a spatiotemporal continuous data set in a gradient from the coast to the mountains, scenes with minimal amount of clouds were pooled into a patchwork of seven satellite scene batches that together determined the extension of the study region and zones. For correcting remaining minor cloud-overlay, we used supplementary images from the same year. However, since not only recent clearcuts but also older clearcuts with or without young regenerating forests can be detected and since the site productivity and thus tree growth capacity is rather poor in the study region, clearcuts prior to the acquisition year of the earliest available images could be detected. This allowed us to interpret one (coastal area) to two (mountain area) decades further back and generate a 50-60 year forest landscape change sequence.

Data on formal protection, were downloaded from the web service “Skyddad natur”, (Swedish Environmental Protection Agency 2017). Data on land cover were downloaded from the GSD-Road Map (National Land Survey 2017), which is a continuously updated database.

Change detection, spatiotemporal stratification and analyses

We applied spectral change detection through maximum likelihood classification for each image pair to identify clearcuts (Supporting Information A1). To make batch pairs compatible for change detection, we stretched and histogram-matched each image. As training samples for the supervised classification we randomly selected six training sites for ‘clearcut’ and ‘uncut’, respectively, per 1,000 km^2. For each time step, a new set of random training sites were selected and new polygons covering either “clearcut” or “uncut” areas were delineated. In the classification of “uncut” in the earliest images, we used new “clearcut” detected in the next later images. Through this procedure we were able to sequentially detect and withdraw clearcuts and map remaining forests. To avoid including very small forest fragments in the analysis, all polygons with an area <2 hectares were withdrawn. In addition, by overlaying information from the GSD-Road Map, data on non-forest areas and non-productive forests (tree growth <1 m^3/ha/year) were withdrawn.

We defined five 10-year time steps from 1973 to 2013 based on satellite image acquisition year (Supporting Information A2). Since acquisition years are not fixed and evenly distributed in time across the study region, we randomly assigned each pCF-patch with a time stamp from within each change detection time interval. This procedure provided each pCF-patch with a specific time stamp and enabled segmentation into continuous time steps, towards which the distribution of remaining pCF could be determined and analyzed.

We based our east-to-west zones on the geographical areas represented by the five largest satellite scene batches (Fig. 2), hereafter denoted “Coastal”, “Eastern inland”, “Western inland”, “Foothill” and “Mountain” (Supporting Information A5). To define the functional patch core area, we assessed the influence of periphery-center estimates on pCF configuration by moving the patch edge 25, 50 and 100 m inwards each patch, following the routine by Ruete et al. (2016). To avoid including very small patches in the core area analysis and thus to avoid skewed patch-area distribution, all pCF-polygons that through edge reduction became <1 hectare were withdrawn.

Results

The study region is strongly dominated by forest land with between 71 and 81% forest cover in the Foothill, Western inland, Eastern inland and Coastal zones. The forest cover in the Mountain zone was lower (55%) (Supporting Information A2). Our results demonstrate a significant variation in natural forest landscape configuration (Figs. 1, 2). A regional polarization, with the Eastern and Western inland and Foothill zones more heavily affected than in particular the Mountain and Coastal zones, has occurred as a consequence of clearcutting (Fig. 3, Supporting Information A4). Figure 3 illustrates the continuous decrease in pCF-area over time across all zones, but with varying rate along the coast to mountain gradient. A pCF-cover of 80% and above dominates in the first two time steps, whereas cover classes below 40% increase substantially from the 1993 time step and onwards.

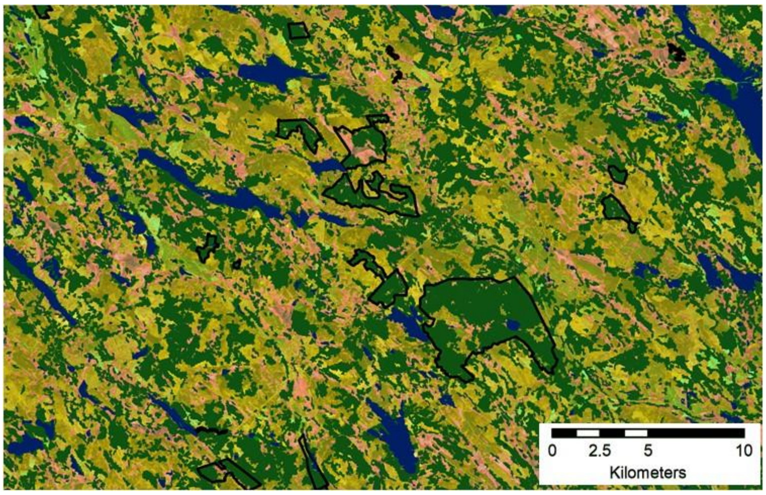

Figure 1. Landsat 8 satellite image (2013) that illustrates the distribution of proxy continuous cover forests (pCF) (green), legally protected areas (black line) and clear cut forest land (light green-yellowish to orange depending on time after clear-cutting with brighter colors cut more recently) within the study area zone Western inland. Dark blue is water bodies and in pink open mires, agriculture land and other non-forest areas. The larger protected are in the low-right center, consisting of three polygons, is the 2,369 ha Björnlandet National Park.

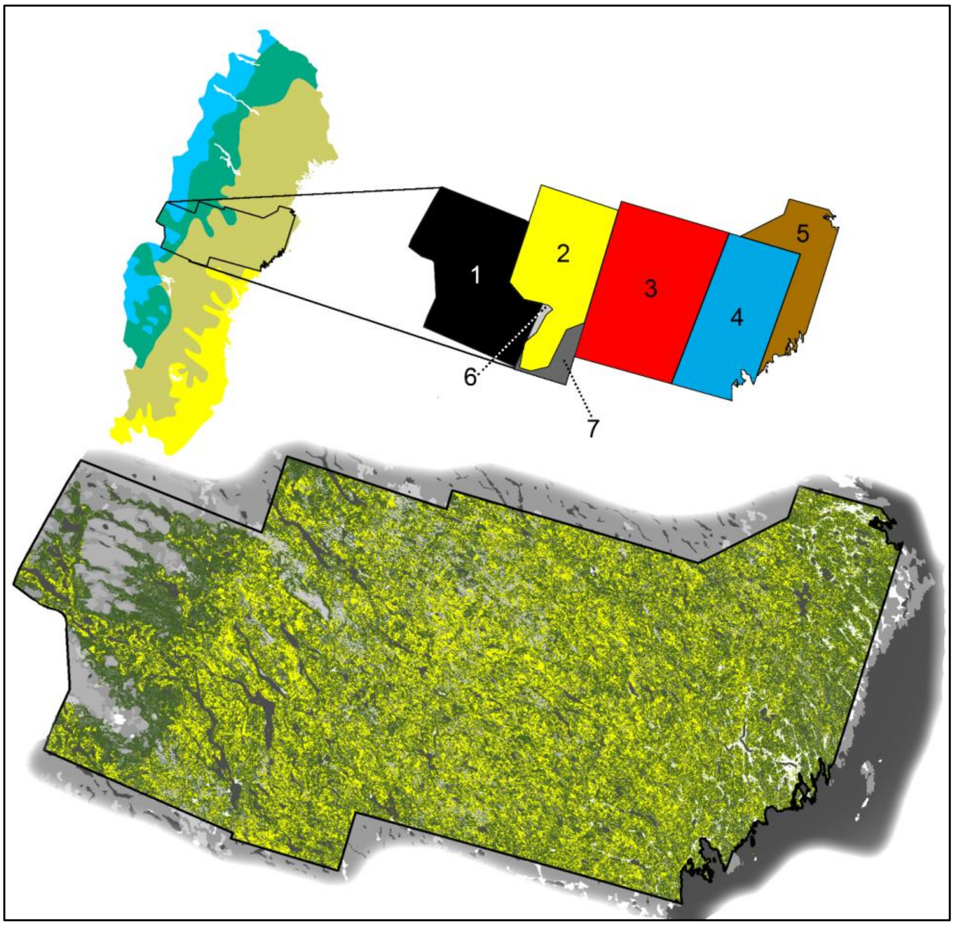

Figure 2. Upper left panel: Northern Sweden with vegetation zones according to Gustafsson & Ahlén (1996); the alpine zone in blue, the northern boreal zone in green, the middle boreal zone in beige, and the southern boreal zone in yellow. The study area is marked with a black line. Upper right panel: The compiled satellite scene batches 1-7 where batches 1-5 build up the zones along the gradient from coast to mountain. Batches no. 6 and 7 are significantly smaller than the other. Bottom map: The situation in 2013 with clearcuts in yellow and remaining proxy continuity forests (pCF) in green projected on top of a grey scale map showing the surrounding area. Grey and white within the study area indicate non-forest areas, i.e. alpine, mires, lakes, agriculture and urban land.

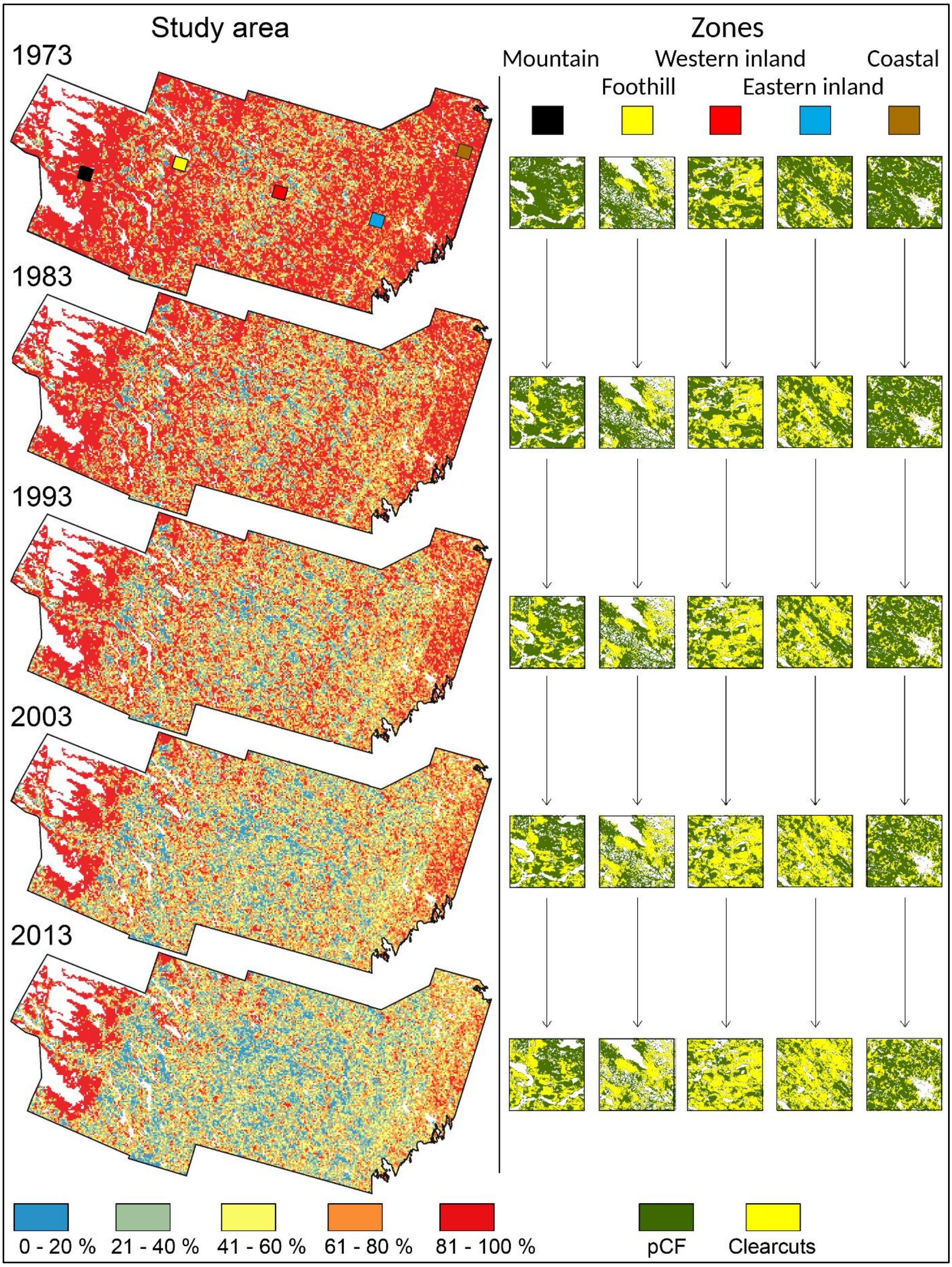

Figure 3. Left panel: Maps showing the proportion in 20%-classes of remaining proxy continuity forests (pCF) relative to the total forest land, calculated for 1km² raster squares (n = 41,734), for each of the five time steps. Right panel: Examples (15x15km) showing the situation in Mountain, Foothill, Western inland, Eastern inland and Coastal zones for each time step. Clear-cut areas are marked in yellow, pCF-areas in green and non-forest areas, i.e. alpine, mires, lakes and agriculture land in white.

In total about 53% of the forest land has been clearcut during the assessed time period, which equals an annual rate of 0.68% (Fig 4A; Supporting Information A3). The observed clearcutting rate over time indicates a higher annual pCF-loss in the Foothill (0.84%), Western (0.80%) and Eastern (0.79%) inland compared with the Mountain (0.65%) and Coastal (0.71%) zones. Only 4.8% of the forest land was protected in 2013 (Fig 4B). Designation of protected areas shows two evident steps in areal increase (Fig. 4B). The first step occurred the latter part of the 1980s with the largest increase in the Foothill zone, and the second in the early part of the 1990s with the largest increase in the Mountain zone and also with marked increase in the Western inland and Foothill zones. The share of protection have increased gradually for all zones but remain at low levels, particularly in the Eastern inland and Coastal zones. The Mountain zone, with 14.3% protection, contributes more than 50% of the protected area in the region.

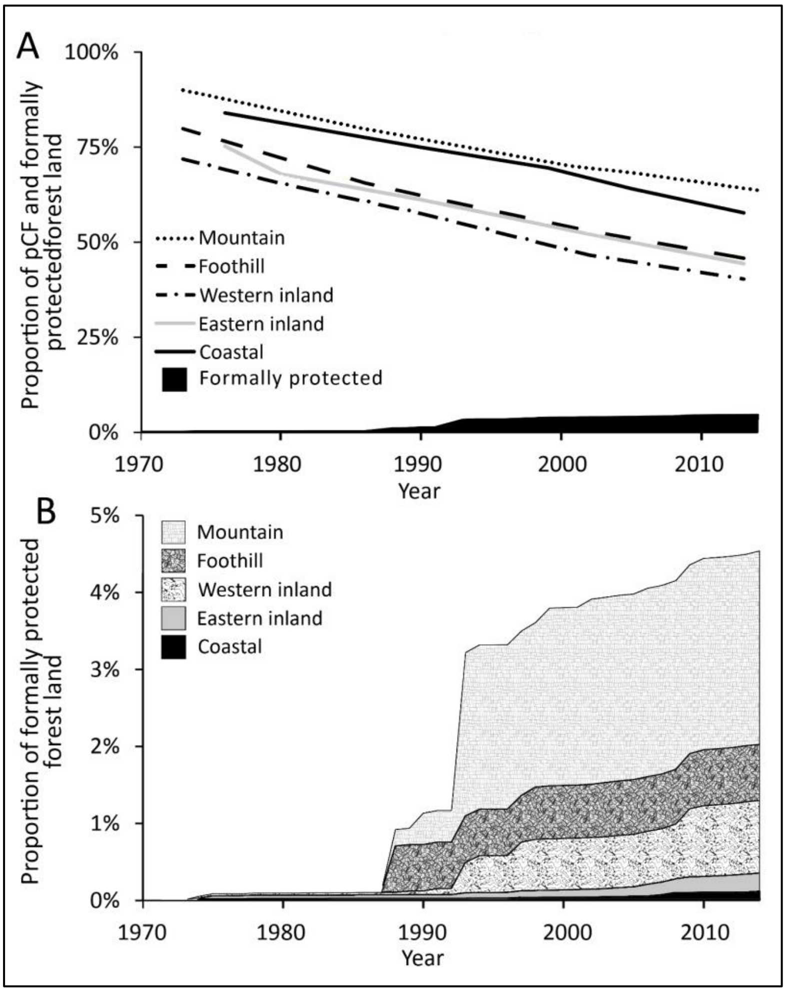

Figure 4. A: The temporal development of the proportion of proxy continuous forest cover (pCF) for Mountain, Foothill, Western inland, 'Eastern inland' and Coastal zones, with the proportion of formally protected forest land at the bottom of the graph. B: The proportion of formally protected forest land (including national parks, nature reserves, biotope protection areas and nature protection agreements) along the time sequence for the Mountain, Foothill, Western inland, Eastern inland and Coastal zones.

The spatial characteristics and thus premises for the ecological functionality of the forest landscape has been altered, with remaining functional pCF-core areas strongly fragmented and reduced in size. Figure 5 illustrates core area distribution, determined for different edge depths. For the pCF-patches, the largest patch, mean patch size and proportion core area become considerable smaller over time for all zones (Table 1, Supporting Information A6). In the 1973 time step, all zones had a largest patch between 225,000 and just below 300,000 ha. The 225,853 ha largest patch in the Mountain zone in 1973 encompassed 40% of all forest land in the zone (Supporting Information A2) whereas the largest patch in 2014 encompassed only 1%, based on 100 m edge depth reduction. With 100m edge depth, the largest patch sizes decreased from 23,468 to 6,102 ha (Mountain) and from 6,114 to 351ha (Coastal). For the Mountain and Foothill zones the results show a considerable size reduction already between the two earliest time steps, whereas for the other zones the area decrease of the largest patch follows a more gradual trend. A salient result is the decrease in largest patch from 257,715 ha to 38,668 ha between 1973 and 1983 in the Foothills zone.

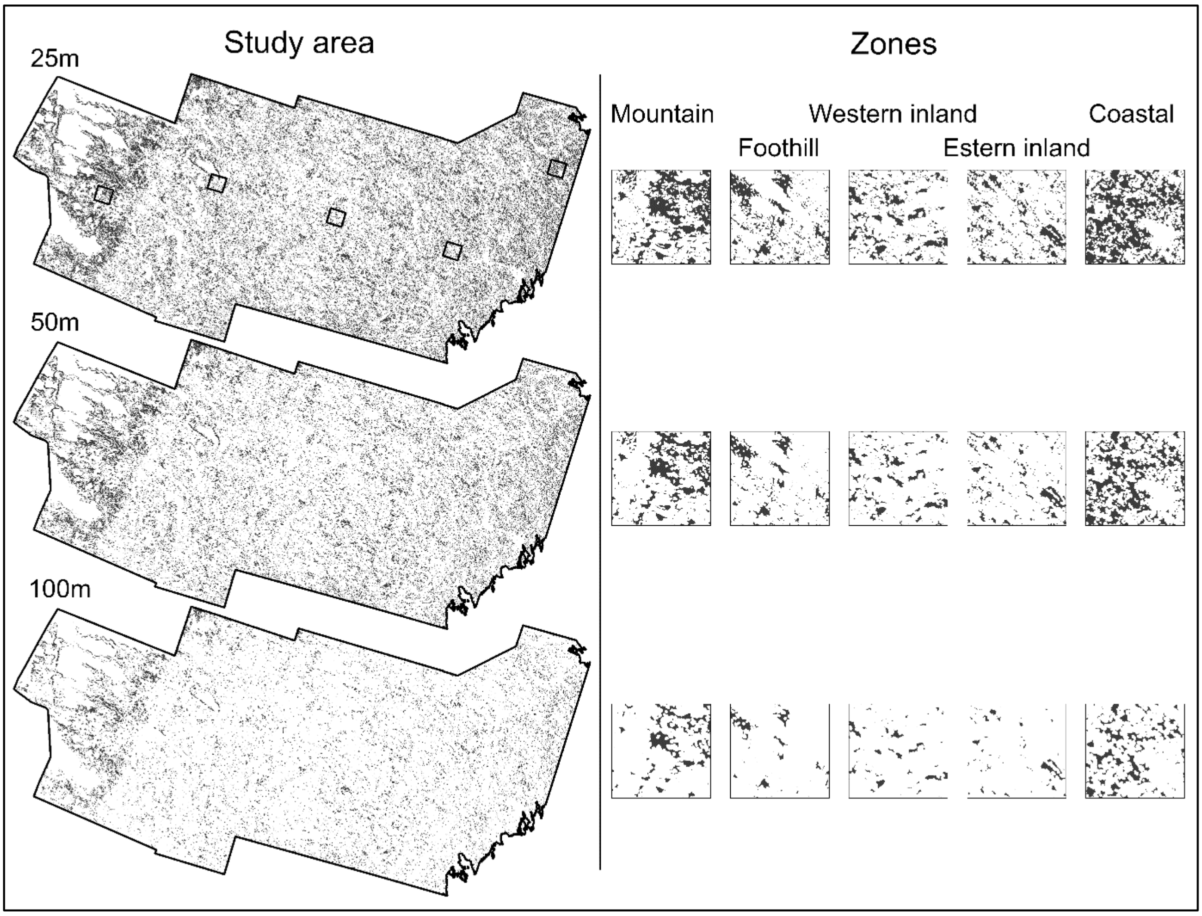

Figure 5. Left panel: Proxy continuity forest (pCF) functional core area determined for the situation in 2013. The functional core area was estimated by assessing edge effects by systematically reducing edge depth by 25m, 50m and 100m towards the center of each pCF-patch. Right panel: Examples (15x15km, same examples as in Fig. 2) showing the pCF-core area situation in Mountain, Foothill, Western inland, Eastern inland and Coastal zones. Remaining pCF-patches <1 ha were withdrawn before mapping.

The mean patch size decreased continually over time for all zones (Table 1). However, in some cases the mean area increased when patch-edge was considered. This is understood as an effect of fragmentation of large patches into several smaller patches, but still of relatively large individual size. For example, for Foothill (1973) we identified 3,668 patches <100 ha compared with 9,160 (2013), and for Western inland (1973) we identified 231 patches compared with 568 (2013) (Supporting Information A2). Assuming a core area edge depth of 100 m, we found between 40% and 32% remaining core area relative to total forest area in the 1973 step compared with between 17% and 6% in 2013. Assuming 50 m and 25 m edge depths, we found slightly higher but constantly decreasing remaining core area over time. The relative decrease from 1973 to 2013 was between 54% (Western inland) and 69% (Mountain) in pCF-share (original pCF-patch; Table 1), between 20% (Eastern

Table 1. Proxy continuity forest (pCF) largest patch, mean patch size and proportional functional core area relative to total forest land for all the original pCF-patches and for the pCF-patches ≥1 ha with 25, 50 and 100 m edge reduction towards the center of each patch, for five points in time following satellite image year for 'Mountain', 'Foothill', 'Western inland', 'Eastern inland' and 'Coastal' zones.

Zone	Year	*Original pCF-patch*			*25 m edge reduction*			*50 m edge reduction*			*100 m edge reduction*		
		Patch size (ha)		*%*	*Patch size (ha)*		*%*	*Patch size (ha)*		*%*	*Patch size (ha)*		*%*
		Largest	*Mean*	*Core area*	*Largest*	*Mean*	*Core area*	*Largest*	*Mean*	*Core area*	*Largest*	*Mean*	*Core area*
Mountain	1973	225,853	139	90	132,763	156	72	60,452	129	59	23,468	92	40
	1986	144,038	89	79	59,989	94	60	38,090	80	47	9,451	57	30
	2001	78,356	55	69	43,418	53	49	17,138	46	36	6,484	35	21
	2005	75,288	50	67	42,223	48	47	16,141	42	35	6,484	33	19
	2014	68,714	39	62	39,615	38	42	13,829	35	30	6,102	30	17
Foothill	1973	257,715	87	79	48,331	112	63	41,945	95	51	15,838	80	35
	1986	38,688	47	64	13,221	55	47	5,548	47	36	3,141	40	22
	1990	30,664	41	61	7,778	46	43	4,433	40	32	3,141	35	19
	2002	18,989	27	51	5,021	27	33	4,074	23	22	2,693	20	11
	2013	7,036	19	44	4,529	17	25	3,695	16	16	2,586	15	7
Western inland	1973	288,553	75	71	93,925	95	56	59,297	85	45	13,838	69	30
	1980	223,686	56	64	21,250	48	40	25,346	65	38	7,155	53	25
	1990	34,917	38	56	8,033	44	40	8,033	44	30	2,474	37	18
	2002	14,367	23	45	2,659	24	28	1,482	22	19	738	19	10
	2013	4,902	16	38	1,506	16	21	920	15	13	714	13	6
Eastern inland	1976	292,748	109	75	188,821	125	60	43,686	99	49	10,085	70	32
	1980	246,633	79	67	60,263	96	52	21,167	72	41	7,903	51	26
	1990	163,174	56	60	15,269	63	45	9,126	48	34	4,244	35	20
	2002	28,812	35	51	3,739	31	34	2,529	24	23	786	18	11
	2013	6,246	22	43	1,323	18	25	865	15	15	457	11	6
Coastal	1976	245,623	159	84	180,962	178	67	97,759	131	54	6,114	68	35
	1990	200,586	101	75	122,615	98	57	26,595	69	43	2,775	36	25
	1999	181,846	77	69	62,102	66	50	12,458	44	36	1,115	25	19
	2005	115,038	57	63	13,396	41	43	4,682	28	29	440	17	14
	2013	58,789	38	57	6,806	26	35	1,420	19	23	351	13	10

inland) and 30% (Mountain) in mean patch size, and between 2-3% (Western, Eastern inland and Foothills) and 28% (Mountain) in largest patch size. The decrease trend line for mean patch size approached the decrease trend line of largest patch (Supporting Information A6, which also shows relative decrease considering 100m edge depth). For mean patch size and largest patch size, the most profound changes were sequentially later in time from the Foothills (1983) to the Western inland (1993) and the Eastern inland (2003) zones.

Thus, in addition to a regional polarization with the central inland areas more heavily affected than the mountain and coastal areas, our results demonstrate that fragmentation, patch size and core area reduction has been extensive and continuous. Our results also indicate an eastward movement of the most profound impact over time, and a pCF-patch size-homogenization on regional scale with the most prominent impact on the large patches.

Discussion

The boreal forest biome has a relatively high proportion of intact forests and low degree of human footprint in comparison to other main forest biomes (Gauthier et al. 2015; Watson et al. 2018). In Fennoscandia, the boreal forest landscape escaped major and widespread forest loss for a long time and has been perceived as Europe's last wilderness area (Kuuluvainen et al. 2017). The continuing impact of forest clearcutting and other land use, however, has generated substantial attention on degradation, decline and fragmentation of forest landscapes and habitats with presumed or actual high nature conservation values (e.g., Moen et al. 2014; Potapov et al. 2017; Watson et al. 2018). As concluded by many (e.g., Aune et al. 2005; Lindenmayer at al. 2006; Sverdrup-Thygeson et al. 2014; Potapov et al. 2017), the ecological qualities and spatial connectivity of remaining valuable forest habitats need to be mapped and assessed for conservation actions such as forest protection and GI-planning and implementation. The spatiotemporal forest landscape change and forest fragmentation trajectory reported here, across a representable and sizable area of the boreal biome, reveals recent and pronounced impacts. Loss of valuable forest habitats continues, which challenges the GI implementation and conservation attainment also in a region where significant levels of valuable forest habitats still are present.

Our results demonstrate a substantial and rapid loss of natural and near-natural forest habitats during the last 50 to 60 years of intensive forest management. The remaining pCF-areas are strongly fragmented, the pCF-patch areas and functional core areas have decreased substantially and the natural landscape configuration has become disrupted. Salient examples are the reduction in the area of the largest patches during the study period and the dramatic effects when considering core area by assessing edge disturbance depths. We expect similar patterns in other boreal regions in northern Europe. The national average harvest rate in Sweden exceeds that of the study region (0.85 %; Anon 2017) and is similar to the rate in Finland (0.7-0.8 %; Luke 2017). Our results further show a polarization with particularly low pCF-share in the inland and higher shares in the coastal and mountainous zones, as well as a general homogenization of patch size distribution and gradual loss of the largest remaining patches. Therefore, the expected biodiversity and ecosystem services loss in the inland could be particularly severe. Following the predictions from species-area derived relationship

(Rybicki & Hanski 2013) and immediate as well as future extinction debts (Hanski & Ovaskainen 2002), increased fragmentation and smaller area of remaining patches may cause local extinctions and overall decline of species diversity with specific impact on forest interior species. The observed loss of pCF-core area, representing a key entity with interior-ecosystem habitat qualities less influenced by edge disturbance, needs directed attention in strategic and operational GI-implementation.

The higher share of pCF-areas in the mountain region was expected due to the later arrival of modern forestry and the advance of nature conservation from the 1970s and onwards, with an emphasis on northwest Sweden. However, our results also indicate that the coastal area have higher pCF-share compared to the inland. Clearcutting was initiated earlier in the coastal area and mature reforested areas may have been detected as pCF. In fact, a recent pilot survey on a similar type of data in the coastal area to the south of the study region indicated that about 40% of detected pCF-areas may be managed stands (Ahlcrona et al. 2017). Hence, extended retrospective temporal sequence and complementary methods are required to identify remaining pCF-areas in the coastal region. Given the normal forest harvesting rotation period of 80 to 100 years in the coastal area, it needs to be evaluated whether mature regenerated forests have developed values suitable for inclusion in a functional GI. Furthermore, the above mentioned pilot survey indicated that open and semi-open forest were a source of error. Hence, such forests were excluded in our study to improve data consistency. However, as non-productive forest land can harbor significant continuity values, data and methods should be improved to allow also assessment of such forest land.

Remaining pCF-patches represent already protected areas, but more importantly also forests that based on conservation value or spatial location can contribute to building a functional GI. Studies show that red-listed forest bryophytes and lichens may survive and possibly also colonize harvested forest areas if adequate conservation measures are applied (Perhans et al. 2014), if dispersal sources exist within close proximity (Hanski 1999, 2011). In managed and fragmented forest landscapes, remaining minimally disturbed continuity forests support species and ecological processes that require more stable old-growth conditions (e.g. Paillet et al. 2010; Dondina et al. 2017). Such areas need to occur in a significant portion of the landscape (e.g., Gustafsson et al. 2012). In line with CBD (2010), threshold levels of 10-30% protected area have been suggested (Hanski 2011).

In this study we highlighted the importance of interior forest areas as most valuable core areas (Aune et al. 2005; Siitonen et al. 2005) and as key components in a functional GI. Edge disturbance sensitivity varies with the species in question and the spatial characteristics of the patches (Murcia 1995). Superimposing effects of forest loss, fragmentation and loss of core area, create aggravating circumstances for conservation values (Riitters et al. 2016; Pfeifer et al. 2017). Even though about half of the forest land has not been subject to clearcutting during the time period studied here, our results show that the net effect on the remaining functional pCF-core areas becomes very pronounced. We stress that acknowledging edge disturbance in conservation planning and design of a functional GI, regardless of selected edge depth, should be a standard procedure in particular in management-dominated forest landscapes where buffering the most valuable forest entities is needed. Forecasting the trends of fragmentation and forest landscape alteration demonstrated in this study into the future, jeopardize achievement of Aichi Biodiversity targets, in particular #7 on sustainable management, biodiversity and conservation, #11 on setting aside a minimum of 17% of terrestrial areas, and #15 on restoring degraded ecosystems (CBD 2010).

The current distribution of formally protected forests is biased towards the mountainous area, thus on generally less productive sites and in more remote locations, while more accessible and productive areas have experienced a larger forest loss. The current share of protection in the eastern and coastal part of our study area is, albeit slow increase over time, at very modest levels. Consequently, we argue that conservation emphasis, including restoration of valuable forests, also need to be placed on the inland and coastal regions to secure connected GI-components across the east to west gradient.

In practical terms, the focus of conservation action should be on field assessment of biodiversity values of the remaining pCF-patches followed by protection with or without restoration management of the high-quality patches, with emphasis on the inland where the detected loss of pCF has been particularly dramatic. Further in depth spatial analyses aiming at identifying the most efficient ways to improve the GI-functionality will be necessary (e.g., Mönkkönen et al. 2014). To minimize the adverse effects of fragmentation and increase the conservation benefits, a general recommendation is that habitat fragments should be protected in clusters rather than randomly scattered (Rybicki & Hanski 2013). In the context of conservation policy it should be noted that our results demonstrate pronounced decrease in pCF-area, in particular on the largest pCF-patches in the foothills and inland region, continues also in this century.

In summary, we have provided evidence for extensive, rapid and recent loss of natural or near-natural forest patches, fragmentation and pronounced forest landscape change across a sizable region of the boreal forest biome. As an effect of clearcutting forest management, the landscape is in a late transitional stage to a land use-modified stage. In addition to climate change that are expected to impact ecology and resilience (Kuuluvainen 2017), this transition needs attention not to jeopardize ecosystem adaptation capacity. To support strategic and operational planning for functional GI in forest landscapes and to fulfil the quantitative and qualitative goals of the EU habitat and species Directives and the CBD Aichi targets, there is an urgent need for identification and directed actions towards those valuable habitats that still exists. Despite increasing overall rate of forest protection, the share of protected forests remain at very low levels compared to the global target of 17 % and display a marked geographical imbalance with the absolute majority of the protection in the remote and low productive mountain zone. Complementary protection is critical, as well as conservation-oriented management and restoration in the surrounding managed landscape. The remaining pCF-patches represents optional target entities for such directed actions. The findings in this study provide input to the implementation of GI as a conceptual approach to address connectivity of forest habitats and landscape scale strategic conservation planning aiming to strengthen and complement current networks of protected forests.

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Supporting Information

Appendix 1 on mapping, change detection and data management procedures, Appendix 2 with supporting study data, and Appendices 3 to 6 on supporting data analyses are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Supporting information

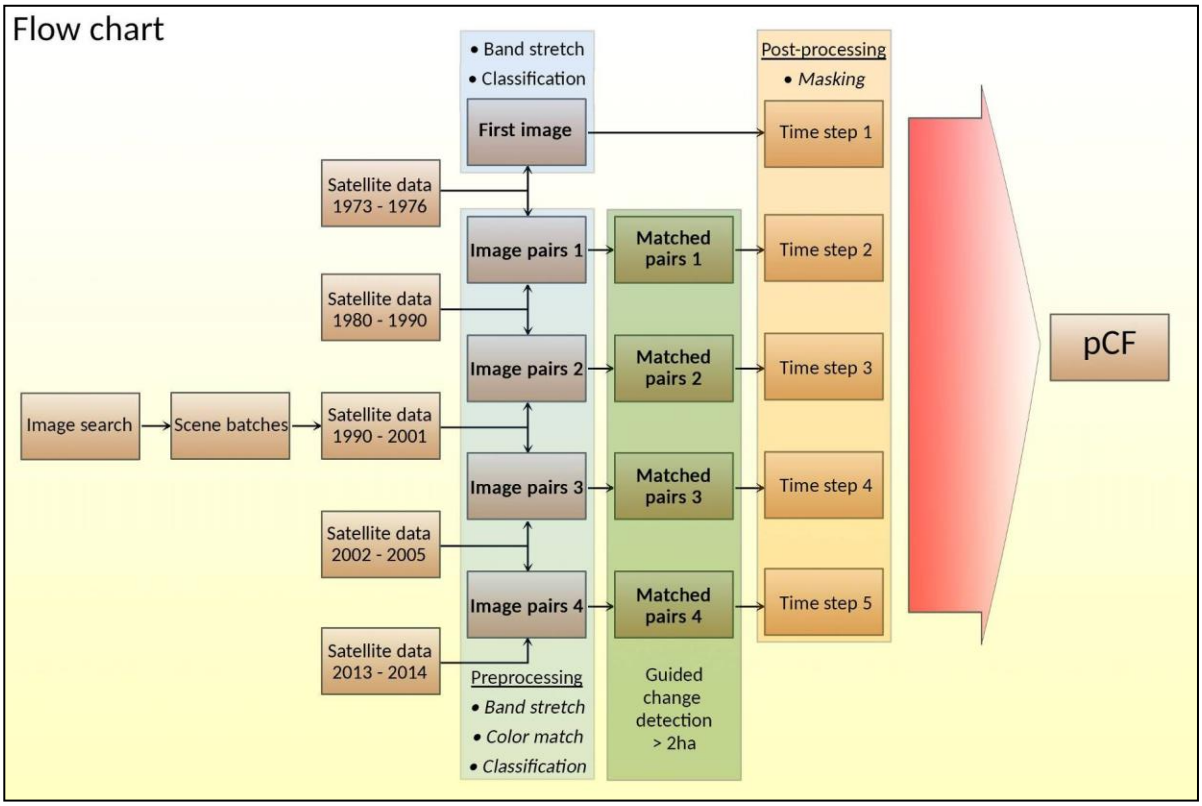
Appendix 1: Mapping, change detection and data management procedures

Initially, cloud free images were searched on United States Geological Survey USGS, Earth Explorer (2014) and Swedish Land Survey Agency, Saccess (2014). Both services provide high quality, multiband products that are commonly used in remote sensing studies. To able long-term detection of landscape change (forest clearcutting), we used data from the Landsat program. After completing the initial image searching, we had gathered 24 partially overlapping and mostly cloud-free satellite images that were later assembled into seven individual scene batches (A2). Still, in a few cases clouds were present. To correct the data we used supplementary satellite images from the same year.

In the following change detection analyses we used the Landsat sensor red spectral band which contains a wavelength suitable for distinguishing changes in forest cover. The change image classification into “clearcut” and “uncut” was performed by using the maximum likelihood classification method, which is the most commonly used supervised classification method for remotely sensed image data (e.g. Goodenough and Shlien 1974; Strahler 1980). The maximum likelihood algorithm assumes that data for each training class in every spectral band are normally distributed.

To classify unknown pixels, the maximum likelihood classification evaluates the variance and covariance of the spectral response patterns and assigns each pixel to the highest probability class (Lillesand et al. 2008). Satellite image-based change detection is a well-established approach to map changes in forest cover and other land-use and natural changes (e.g., Coppin et al 2004; Radke et al. 2005; Potapov et al. 2008; Muukkonen et al. 2012; Margono et al. 2014; Potapov et al. 2017). The classification procedure and change detection was carried out in ESRI ArcMap.

Before analysis we stretched the color scale for each image pair to equal band length and used histogram matching to make images compatible. For detection of clear cuts made prior to the 1970s (the first Landsat images in our batches) we only used the classification part of the procedure. Hence, band stretching, and histogram matching was not done. To withdraw landscape elements other than forest, we used data on non-forest areas and non-productive forests (tree growth <1 m3/ha/year) from the GSD-Road Map (National Land Survey 2017). The image preparation was done in Erdas Imagine.



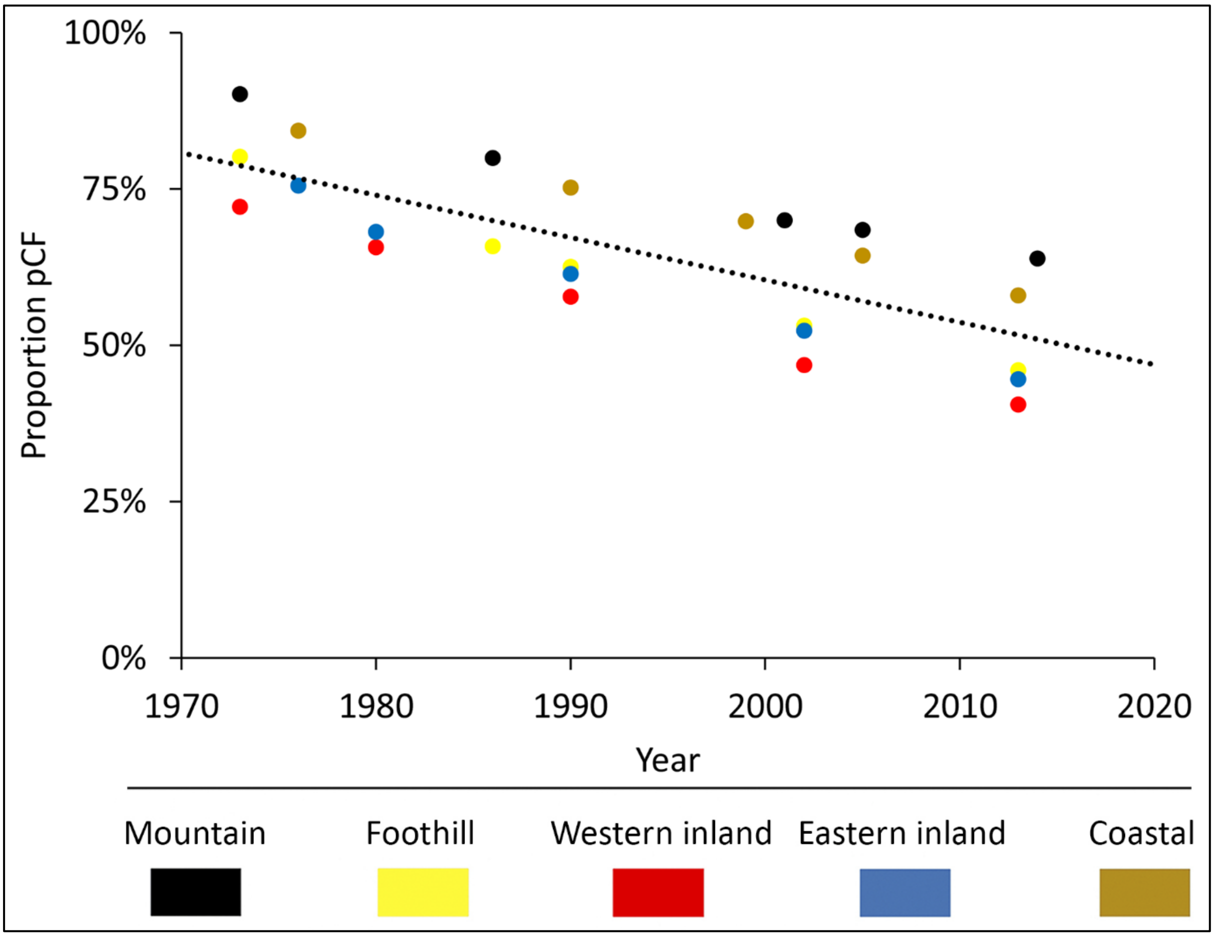
Appendix 1: Flow chart illustrating the main steps in the construction of the time steps and the mapping of proxy continuous cover forest (pCF) areas.

Appendix 2. Supporting study data

Appendix 2: Landsat data on acquisition year, total land area, proportion forest land area, and the proportion of formally protected forest land (national parks, nature reserves and forest conservation areas), for each satellite scene batch and time step. The Landsat satellite images used were multi-spectral scanner (MSS, Landsat 2), thematic mapper (TM, Landsat 4 and 5), enhanced thematic mapper (ETM, Landsat 7) and operational land imager (OLI, Landsat 8).

Scene batch no	Acquisition year	Pixel size (m)	Sensor name	Tot. area (km2)	Forest land (%)	Formally protected area (%)	Number of individual forest fragments ≥ 1ha
1				10,370	55		
	1973	60.0	MSS			0.0	3,668
	1986	30.0	TM			0.1	5,070
	2001	30.0	ETM			13.1	7,126
	2005	30.0	TM			13.7	7,568
	2014	30.0	OLI			14.3	9,160
2				8,254	71		
	1973	57.0	MSS			0.0	5,368
	1986	28.5	TM			0.0	8,043
	1990	30.0	ETM			4.1	8,780
	2002	30.0	TM			4.5	11,051
	2013	30.0	OLI			4.8	13,577
3				12,656	78		
	1973	57.0	MSS			0.0	9,348
	1980	30.0	TM			0.1	11,399
	1990	28.5	TM			0.2	14,571
	2002	30.0	ETM			2.2	19,631
	2013	30.0	OLI			3.1	23,839
4				8,644	81		
	1976	57.0	MSS			0.2	4,830
	1980	28.5	TM			0.2	6,013
	1990	30.0	TM			0.3	7,534
	2002	30.0	ETM			0.5	10,230
	2013	30.0	OLI			1.1	13,742
5				4,680	76		
	1976	57.0	MSS			0.2	1,864
	1990	57.0	MSS			0.3	2,610
	1999	30.0	TM			0.4	3,187
	2005	30.0	ETM			0.5	3,956
	2013	30.0	OLI			1.0	5,279
6				177	76		
	1973	80.0	MSS			0.0	31
	1986	57.0	MSS			0.0	35
	2001	30.0	TM			0.0	44
	2005	30.0	ETM			0.0	255
	2014	30.0	OLI			0.5	361
7				974	80		
	1973	80.0	MSS			0.0	609
	1986	30.0	TM			0.1	879
	1992	30.0	ETM			0.1	1,038
	2002	30.0	TM			0.1	1,270
	2013	30.0	OLI			0.1	1,594

Appendix 3: Supporting data analyses



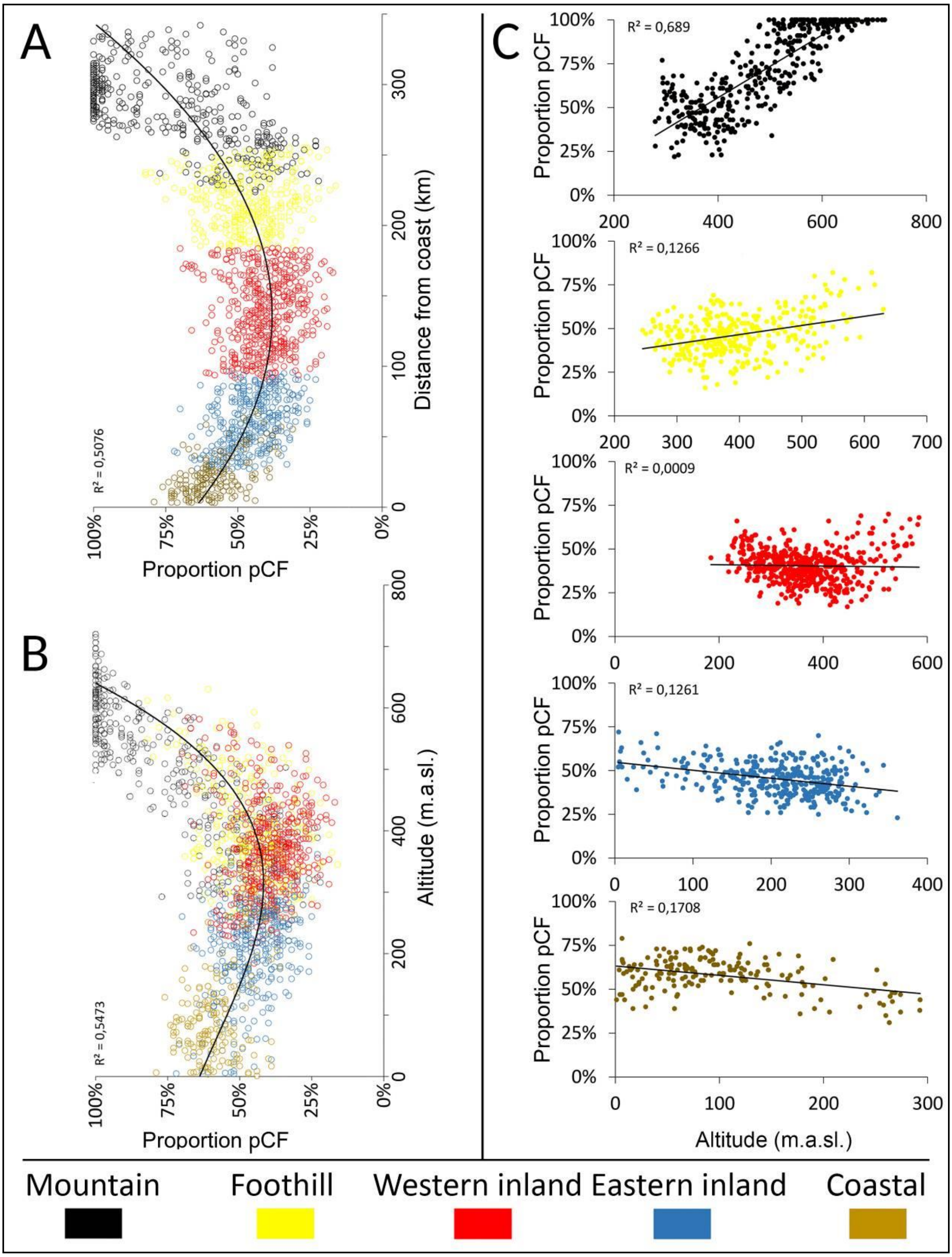
Appendix 3: Clear-cutting rate based on proportion remaining proxy continuous cover forests (pCF) for each of the five zones. The trend line represents the mean cutting rate for the whole study area (R^2 = 0.57).

Appendix 4: Supporting data analyses

Appendix 4: The proportional (per cent) area in each zone and total for the study area covered by remaining proxy continuity forest (pCF), estimated on 5x5 km raster squares (n = 1,750) for the situation in 2013. Presented in fraction classes 0%, 1-50%, 51-75%, 76-99% and 100%.

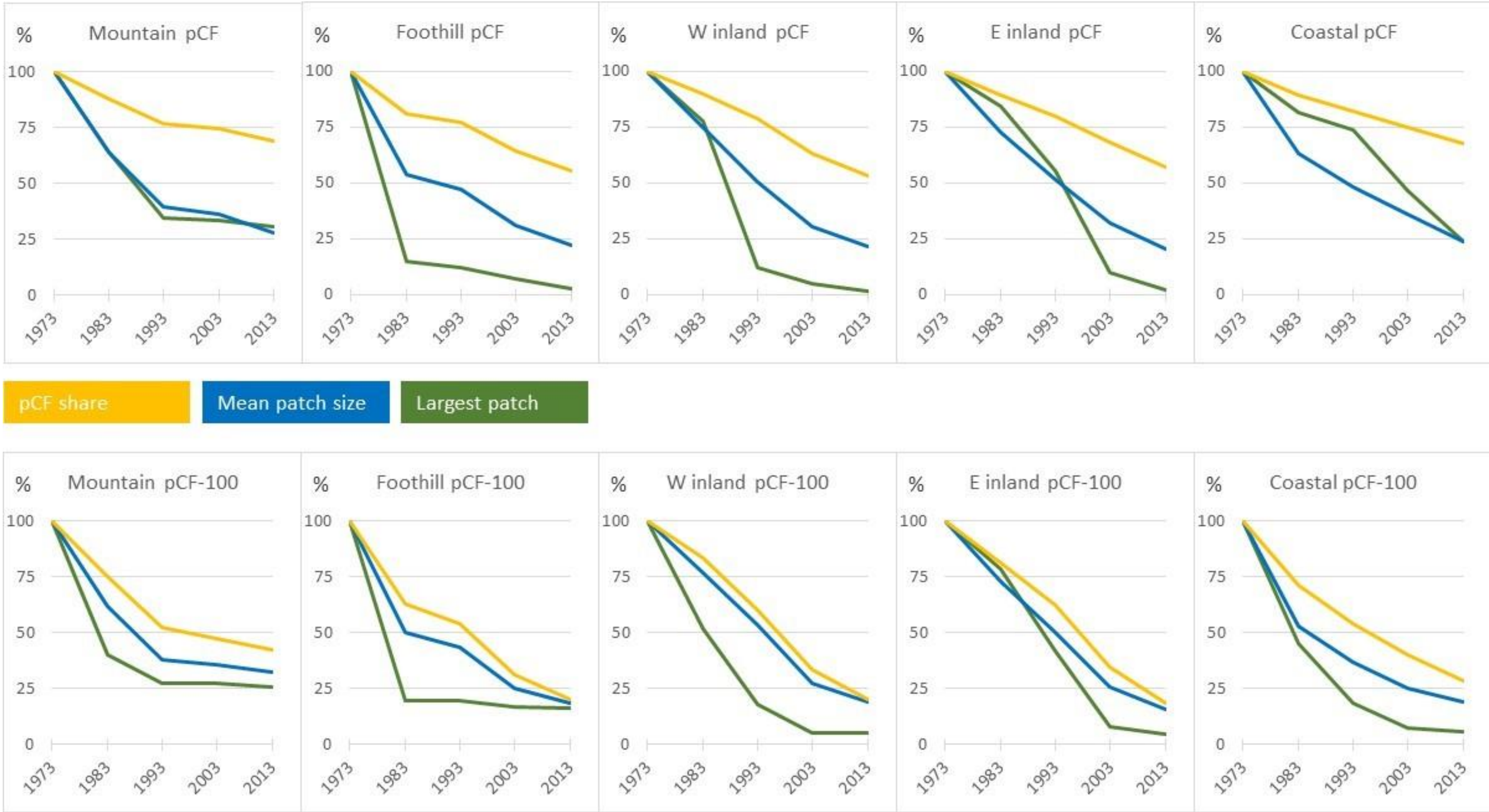
Zone	*0% pCF*	*1-50% pCF*	*51-75% pCF*	*76-99% pCF*	*100% pCF*
Mountain	0.0	33.0	22.4	21.9	22.7
Foothill	0.2	56.0	28.1	14.2	1.6
Western Inland	0.1	68.0	24.9	6.8	0.2
Eastern Inland	0.0	60.7	32.5	6.7	0.1
Coastal	0.0	32.1	49.9	17.7	0.3

Appendix 5: Supporting data analyses



Appendix 5. Left panel: Graphs showing how the proportion of remaining proxy continuity forests (pCF) corresponds to the independent variables distance from coast (A) and altitude (B), based on 5x5 km pixels (n = 1750). Right panel (C): Graphs showing how pCF corresponds to the independent variable altitude for each of the five zones. For data on elevation, we used the 50 m grid, digital elevation model from the Swedish Land Survey Agency (2017)

Appendix 6: Supporting data analyses



Appendix 6: Relative decrease in total pCF-area, mean pCF-patch size and largest patch for the 1973 to 2013 time steps for original pCF-patch and for pCH-patches with 100m edge depth reduction. Data builds on Table 1 in Svensson et al. (2018) that provides proxy continuity forest (pCF) largest patch, mean patch size and proportional functional core area relative to total forest land for all the original pCF-patches and for the pCF-patches ≥1 ha with 25, 50 and 100 m edge reduction towards the center of each patch, for five points in time following satellite image year for 'Mountain', 'Foothill', 'Western inland', 'Eastern inland' and 'Coastal' zones.

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