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Wood production and biodiversity conservation are rival forestry objectives in Europe's Baltic Sea Region

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Citation: Angelstam, P., V. Naumov, M. Elbakidze, M. Manton, J. Priednieks, and Z. Rendenieks. 2018. Wood production and biodiversity conservation are rival forestry objectives in Europe's Baltic Sea Region. Ecosphere 9(3):e02119. 10.1002/ecs2.2119

Abstract. The policy term green infrastructure highlights the need to maintain functional ecosystems as a foundation for sustainable societies. Because forests are the main natural ecosystems in Europe, it is crucial to understand the extent to which forest landscape management delivers functional green infrastructures. We used the steep west–east gradient in forest landscape history, land ownership, and political culture within northern Europe's Baltic Sea Region to assess regional profiles of benefits delivered by forest landscapes. The aim was to support policy-makers and planners with evidence-based knowledge about the current conditions for effective wood production and biodiversity conservation. We developed and modeled four regional-level indicators for sustained yield wood production and four for biodiversity conservation using public spatial data. The western case study regions in Sweden and Latvia had high forest management intensity with balanced forest losses and gains which was spatially correlated, thus indicating an even stand age class distribution at the local scale and therefore long-term sustained yields. In contrast, the eastern case study regions in Belarus and Russia showed spatial segregation of areas with forest losses and gains. Regarding biodiversity conservation indicators, the west–east gradient was reversed. In the Russian, Belarusian, and Latvian case study regions, tree species composition was more natural than in Sweden, and the size of contiguous areas without forest loss was larger. In all four case study regions, 54–85% of the total land base consisted of forest cover, which is above critical fragmentation thresholds for forest landscape fragmentation. The results show that green infrastructures for wood production and biodiversity conservation are inversely related among the four case study regions, and thus rival. While restoration for biodiversity conservation is needed in the west, intensified use of wood and biomass is possible in the east. However, a cautious approach should be applied because intensification of wood production threatens biodiversity. We discuss the barriers and bridges for spatial planning in countries with different types of land ownership and political cultures and stress the need for a landscape approach based on evidence-based collaborative learning processes that include both different academic disciplines and stakeholders that represent different sectors and levels of governance.

Key words: biodiversity; collaborative learning; ecosystem services; governance; green infrastructure; land-sharing; land-sparing; spatial planning; sustained yield forestry.

Received 2 August 2017; revised 27 November 2017; accepted 9 January 2018. Corresponding Editor: Margherita Gioria.

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Introduction

Contemporary policy documents highlight the need to use a development approach that satisfies all sustainability dimensions based on material and immaterial values of landscapes and regions (e.g., European Commission 2009, 2013*a*). Ultimately, functional ecosystems form the supply base for human well-being and the development of sustainable societies (Burkhard et al. 2012). Ecological networks are a solution and have been subject to research, policy, and practice in Europe for decades (Jongman et al. 2011, Čivić and Jones-Walters 2014). The EU's green infrastructure policy (European Commission 2013*a, b*) retains this ambition and aims at a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of benefits, today often referred to as ecosystem services (e.g., de Groot et al. 2002, Lele et al. 2013). It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas, in both rural and urban settings. Development of green infrastructure is a key step toward the success of maintaining and enhancing biodiversity according to the EU 2020 Biodiversity Strategy (http://ec.europa.eu/environment/nature/ecosystems/strategy/index_en.htm). The backbone of EU's green infrastructure policy is the Natura 2000 network of high conservation value areas (Salomaa et al. 2017). The EU's green infrastructure policy thus aims at ensuring that conservation, restoration, and management of green infrastructure will become parts of integrated spatial planning and territorial development. Implementing green infrastructure policy requires maintenance of sufficient amounts of patches of different representative vegetation types, which then form ecological networks. Note, however, that the term green infrastructure evolved more than a century ago along two paths, in the United Kingdom and the United States. The UK approach views green infrastructure as the linking of urban parks and other green space into functional networks to benefit people, while the US approach sees green infrastructure primarily as a biodiversity conservation measure to counteract habitat degradation and fragmentation (e.g., Benedict and McMahon 2006, Allen 2014).

The increased pressure for higher biological production in terrestrial and aquatic systems under the heading of bio-economy (McCormick and Kautto 2013) and the expansion of the human footprint through housing, transport, communication, and energy infrastructures make the management and sustainability of green infrastructures a difficult balancing act (e.g., Popescu et al. 2014). Resolving competing interests through collaborative spatial planning is a key approach for reaching sustainability (e.g., Elbakidze et al. 2015), as well as a challenge for developing inclusive governance processes (Baker 2006) and active adaptive management (Walters 1986, 1997).

Therefore, to understand whether land covers representing a region's ecosystems actually form functional green infrastructures, it is crucial to assess the entire chain of actions from policy to practice, including evidence-based knowledge about the resulting state of sustainability in landscapes and regions. This requires both evaluation of the policy process and the outcomes of this process (Rauschmayer et al. 2009). Evaluation of the former involves assessment of what constitutes good governance (Currie-Alder 2005, Baker 2006), including elements such as more and improved information management and learning, a legitimate process, and the normative aims of transparency and participation. According to Rauschmayer et al. (2009), outcomes of the policy process can be divided into two parts: firstly, the outputs in terms of implementation of practices and rules to be applied by governors at multiple levels, including pronouncements of norms (e.g., Lammerts van Bueren and Blom 1997) such as strategic performance targets for short-term and long-term goals (e.g., Angelstam and Andersson 2001), as well as tactical planning and operational management approaches (e.g., Eriksson and Hammer 2006). Depending on the sector, management involves practices that imply both pressures and responses on sustainability (Butchart et al. 2010). Secondly, the consequences of the operational implementation of strategic and tactical plans by managers on the sustainability of landscapes and regions need to be assessed (e.g., Angelstam et al. 2011*a*, Elbakidze et al. 2011, 2015). This requires monitoring of indicators that measure the effectiveness of policy implementation tools on different aspects of sustainability, such as the functionality of green infrastructures.

There are three approaches to monitoring: (1) implementation monitoring, (2) validation monitoring, and (3) effectiveness monitoring (Busch and Trexler 2003). The development of effectiveness monitoring based on evidence-based knowledge about the states of green infrastructures is a prerequisite for planning toward functional green infrastructures (Müller and Burkhard 2012). While policy-level indicators generally focus on national level reporting about policy implementation, spatially explicit data at the scale of local landscapes in regions are needed to provide knowledge about the state of green infrastructure for effective steering by spatial planning. Consequently, monitoring must also be undertaken at the scale where land management takes place, that is, in a local forest management unit or an administrative unit such as a municipality. A range of studies have proposed indicators and presented different frameworks for monitoring forest values, all of which need to be validated before being used (e.g., Stem et al. 2005).

Successful implementation of green infrastructure policy thus requires that stakeholders and actors coordinate and integrate their monitoring of land covers, landscapes, and regions. However, given that national and business policies, landscape histories and governance contexts among countries in Europe are very diverse, Pan-European and EU policies linked to green infrastructures are likely to be comprehended and implemented differently. This is not made easier by national-, regional-, or local-level rhetoric that only stresses continued pressure on valuable natural systems, or only the responses made in terms of set-asides and management practices. The net effect of pressures and responses on the state of ecosystems is thus confounded to practitioners, policy-makers, and the public, unless comprehensive analyses are made (e.g., Elbakidze et al. 2016). Indicators of green infrastructure functionality need to address specific benefits and be robust and understandable.

The steep west–east gradient in landscape history, ownership, and political culture within Europe's Baltic Sea Region (BSR) provides excellent opportunities to explore the consequences for the profile of benefits delivered by landscapes' green infrastructures (e.g., Kern and Loffelsend 2004, Metzger and Schmitt 2012, Angelstam et al. 2013, 2017*b*). Following the enlargement of the European Union in 2004 by inclusion of the three Baltic States and Poland, the Baltic Sea has become close to an EU-internal sea. The BSR strategy (European Commission, 2009) aims at functional coordination and more efficient use of financial resources and existing cooperation schemes between Sweden, Finland, Estonia, Latvia, Lithuania, Poland, Germany, and Denmark. The European Commission (2014) noted that the involvement of stakeholders needs to be strengthened, including parliaments at different levels, regional governments, and civil society. Consistent with that, EU InterReg and other funding mechanisms for neighborhood collaboration have a broader geographical scope than the EU. Non-EU countries are also participating actively in work with the BSR Cooperation. These include Norway, Russia, and sometimes Iceland and Belarus (Swedish Agency for Economic and Regional Growth 2014). The BSR strategy emerged due to a suite of critical environmental problems as well as severe differences in infrastructural accessibility and economic development among regions (e.g., Bengtsson 2009). Being the first macro-regional cooperation of its kind in the EU, the BSR is a good test case. Thus, it is important to assess the extent to which this new model functions and whether this model may be applied in other macro-regions. This requires evidence-based knowledge about how different countries' governance and management affect environment, infrastructure, and development of natural resources.

With forests being the main natural ecosystems in the BSR, it is thus crucial to understand the extent to which forest landscape management satisfies the different dimensions of sustainable forest management policy by maintaining functional green infrastructure. Effective wood production and habitat for biodiversity conservation are two key benefits received from forest landscapes, which are articulated in policy (e.g., Edwards and Kleinschmit 2013) and practice (e.g., Juutinen and Mönkkönen 2004). Integrative vs. segregative approaches to sustainable forest management are debated as solutions (Bollmann and Braunisch 2013). This dichotomy is analogous to land-sharing, which combines wood production with biodiversity conservation across a landscape, and land-sparing, in which more intense forestry is combined with protected areas (Edwards et al. 2014). Maximum sustained yield wood production and biodiversity conservation with policy

ambitions about viable populations of specialized species and their habitats as well as ecological processes are rival in the same local forest area (e.g., Mönkkönen et al. 2014). Accommodating both production and biodiversity conservation thus requires spatial planning that includes multiple sectors at the scales of landscapes and regions (e.g., Vierikko et al. 2008, Angelstam et al. 2011*a*, Elbakidze et al. 2015). This requires evidence-based knowledge about the state of these benefits in different contexts, such as in the BSR with its diversity of contexts.

The aim of this study was to assess the relative state of sustained yield wood production as a means of delivering provisioning ecosystem services, and biodiversity conservation to supply supporting (or habitat) services, as rival objectives in the west–east gradient of the BSR. Using public spatial databases with proxy data relevant to forest landscapes in different regions, we developed and modeled four effectiveness indicators (sensu Busch and Trexler 2003) of wood production and four indicators for biodiversity conservation at the regional level in Sweden, Latvia, Belarus, and Russia. The resulting parameter values were then compared and the results discussed in the context of land-sharing or land-sparing.

Study Areas

The BSR is diverse regarding both ecological and social systems. In terms of ecosystems, the BSR's most common potential natural vegetation is forest, from the temperate zone in the south with broad-leaved tree species to the boreal biome in the north dominated by conifers (e.g., Laasimer et al. 1993). The different forest biomes form broad longitudinal zones at similar latitudes in both the western and eastern parts of the BSR (Ahti et al. 1968). Naturally, these forest biomes are dominated by site-specific disturbance regimes, which result in a diversity of forest environments ranging from old-growth forests, mixed deciduous–coniferous, and deciduous forests during certain successional stages and to forests disturbed by fire, wind, and flooding (e.g., Angelstam and Kuuluvainen 2004, Shorohova et al. 2011). However, the social systems of these countries and regions differ considerably in landscape history and governance across the west–east gradient. The BSR ranges from western countries with long-term stable rules and democratic institutions such as Sweden to Belarus and the Russian Federation with authoritarian rule (Hague et al. 1992), as well as countries in transition (e.g., Latvia).

Sweden and Russia have always been independent countries, but with different governance traditions (Hauge et al. 1992), as well as approaches to forest management linked to different forest histories and ideologies (Nordberg et al. 2013, Naumov et al. 2016). While Sweden has developed maximum sustained yield principles, Russia is still by and large focusing on wood mining (Knize and Romanyuk 2006, Elbakidze et al. 2013). Before the Soviet occupation in 1940, Latvia was an independent state with developed property structure and German forest management traditions that focused on the principle of even-aged stand age distributions (Dumpe 1999, see also Puetmann et al. 2009). During the Soviet period, forest cover in Latvia increased due to forced abandonment of agricultural land (Vanwambeke et al. 2012). After the collapse of the Soviet Union in 1991, agricultural and forest land was to be returned to its previous owners, but the forest restitution process is gradual involving finding alternative land, implementation by acquiring legal document, and developing forest management plans. Intensification of forest harvesting and management to increase economic benefits by application of maximum sustained yield principles were introduced in the 1990s using Sweden as an example (E. Peterhofs, *personal communication*). In contrast to Latvia, which moved to democratic governance from its independence, Belarus has remained under authoritarian rule being entwined with Russia. These four countries thus represent a clear west–east gradient in forest governance and management methods (Duncker et al. 2012). In each country, we identified one representative case study area with similar sizes in the southern part of the boreal forest ecoregion (Laasimer et al. 1993), including entire Latvia as one of them (Fig. 1).

Strategic spatial planning needs to be addressed within an area that is sufficiently large to satisfy a particular benefit. Regarding wood production, say that an industry requires 2 Mm^3 annually (S. Lundell, *personal communication*). With a wood production of 3–5 m^3/ha per year, the total area of forest planning units needs to encompass

Fig. 1. Map of the Baltic Sea Region countries (in gray color) showing in the four regional case studies Bergslagen–Mälardalen (the five counties: Stockholm, Uppsala, Södermanland, Örebro, and Västmanland) in Sweden (34,400 km^2), entire Latvia (64,600 km^2), Vitebsk oblast (40,100 km^2) in Belarus, and Pskov oblast (55,300 km^2) in NW Russia.

4,000–6,500 km^2. If the forest cover in the region is 50%, the planning region for securing wood for the industry should be in the order of 8000–13,000 km^2. Biodiversity conservation is about maintaining sufficient amounts of habitat suitable to host viable species populations and ecological interactions. For example, Angelstam et al. (2004) estimated that the required size of management units for viable populations of species with large area requirements is ~10,000 km^2. To match this range of necessary area extents of an assessment region for sustained yield wood production and biodiversity conservation, we choose to focus on five counties (Stockholm, Uppsala, Södermanland, Örebro, and Västmanland) in Sweden's Bergslagen–Mälardalen region, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia, respectively (Fig. 1).

Methodology

Spatial data sources and their characteristics

Spatial planning is hierarchical, ranging from larger to smaller spatial extents and longer to shorter time spans. Regional-level planning is thus subsequently sub-divided into plans for smaller units, such as municipalities, and sectorial management units for forest management and biodiversity conservation. With this study's regional focus to support strategic planning in local landscapes, analyses were made based on the spatial resolution of a 10 × 10 km grid. This area (100 km^2) corresponds to the order of magnitude of forest landscape planning units in the BSR, which typically is 50–300 km^2 (Angelstam and Pettersson 1997, Elbakidze et al. 2016).

Given the lack of harmonized national spatial forest management data in the BSR, we based the analyses on three internationally harmonized thematic data sets: (1) the global data set about forest cover change at pixel size 30 × 30 m produced by Hansen et al. (2013) covering the period 2000–2013, (2) the map of coniferous and broad-leaved forest produced by the European Forest Institute (EFI) (Kempeneers et al. 2011), and (3) OpenStreetMap data on road networks, for which Ather (2009) demonstrated 80% accuracy compared to official data sets. We pre-processed some of data the sets to mirror the indicator requirements. According to FAO (2000), forest is defined as >5 m high woody vegetation with a canopy cover of >10% that covers >0.5 ha. We thus included only 30-m pixels in Hansen et al.'s (2013) data set with more than 10% forest cover, but not for the data from the EFI (see Kempeneers et al. 2011) for which we used the forest cover proportions provided in that database. Details about the data sources are summarized in Table 1. To minimize area distortions (Nyerges and Jankowski 1989), raster data were re-projected to metric Albers equal-area conic projection (Snyder 1987) with standard parallels 53° and 61° N and central meridian at 23° E. To make the numerical analyses and maps consistent, the borders of the four study regions were generalized to fit a 10 × 10 km grid. Analyses of raster data were done with GRASS (Neteler and Mitasova 2013) and statistics calculations with R (R Development Core Team 2013).

To achieve the lowest data loss, we used a nearest neighbor algorithm to resample categorical data (forest cover loss and gain); for all other data, we employed bilinear interpolation (Parker et al. 1983). The global change data set (Hansen et al. 2013) already has induced errors because it was created based on 654,178 growing season Landsat 7 Enhanced Thematic Mapper Plus scenes which have Universal Transverse Mercator planar coordinate system. We did not use a grid approach for the road density algorithm, that is, calculation of lengths within rectangular grid cells, due to limitation caused by artificial cell borders. Instead, we applied an algorithm based on kernel density estimation (Cai et al. 2013). We

Table 1. List of provisioning and supporting (habitat) services, indicators, associated variables, and units analyzed in this study, as well as data and algorithm.

Ecosystem service	Indicator	Variable (unit)	Data source (spatial resolution)	Algorithm for 10 × 10 km grid
Provisioning	Forest management intensity (FORINT)	Forest loss per year (proportion)	Hansen et al. (2013) (1 arc second)	SUM loss/SUM forest area during 2000–2013 divided by 14
	Net forest gain (FORGAIN)	Net forest gain ($\log_{10}$)	Hansen et al. (2013) (1 arc second)	$\log_{10}$(SUM gain/SUM loss)
	Accessible coniferous forest (ACC_CON)	Conifer forest with good access (index)	(a) Kempeneers (2011) (1 × 1 km) (b) OpenStreetMap	(a) SUM area (conifer ≥ 70%) (b) SUM road length/SUM forest area (c) index (a × b)
	Economic sustainability (ECONSUST)	Relationship between forest gain and loss (slope of linear regression)	Hansen et al. (2013) (1 arc second)	SUM gain = $b_0 + b_1 \times$ SUM loss; if $b_1 = 1$ then forestry is economically sustainable
Supporting (habitat)	Deciduous forest habitat (DECHAB)	Proportion of local landscape (10 × 10 km) with deciduous forest (proportion)	Kempeneers (2011) (1 × 1 km)	SUM area (decid ≥ 50%)
	Mixed forest habitat (MIXHAB)	Proportion of local landscape (10 × 10 km) with mixed coniferous–deciduous forest (proportion)	Kempeneers (2011) (1 × 1 km)	Mask (conif and decid)
	Intact forest (INTACT)	Size of patches with low forest loss (km^2)	Hansen et al. (2013) (1 arc second)	(1) a = SUM loss <1% (2) clump (a)
	Forest landscape fragmentation (FRAG)	Proportion of local landscapes (10 × 10 km) with forest cover (proportion)	Hansen et al. (2013) (1 arc second)	(1) SUM forest cover >40% (2) classification into three categories

thus argue that it is safe to draw conclusions for the coarse spatial scale chosen to explore differences among regions in the BSR.

Indicators for effectiveness monitoring in management units

Focusing on effectiveness monitoring (see Busch and Trexler 2003), this study takes the perspective of spatial planners in forest landscapes. The principle strategic, tactical, and operational planning for maximum sustained yield wood production is a general paradigm in forest management worldwide (Elbakidze et al. 2013). It implies achieving high continuous wood production, at the earliest practical time and a balance between increment and cutting (Nieuwenhuis 2000), that is, a steady-state balance of gains and losses of forest stands in a local forest management unit. The emergence of systematic conservation planning takes the same hierarchical approach (Angelstam et al. 2011*a*). In the BSR, the size of landscape planning units is typically 50–300 km^2 (Angelstam and Pettersson 1997, Elbakidze et al. 2016). We used 100 km^2 areas as virtual forest management units. To monitor the effectiveness of wood production, we developed indicators for forest management intensity (FORINT) and for the balance between loss and gain (FORGAIN). To mirror the focus on conifers, we developed an indicator for accessible coniferous forest (ACC_CON). Finally, to be economically sustainable, costs and incomes need to be balanced within the forest management units, which led to the indicator ECONSUST (see Table 1).

Biodiversity is about species, habitats, and processes (e.g., Brumelis et al. 2011). Land management affects habitat directly by modifying the quality, amount, and spatial configuration of land cover patches, which thus need to be considered in spatial planning. Maintaining green infrastructures requires that sufficient amounts of representative naturally occurring forest ecosystems are maintained (Angelstam and Andersson 2001). This means that patches of different land covers should be of sufficient quality and size and have sufficient connectivity. Given the focus on coniferous forest as a base for industrial use, we analyzed deciduous forest habitat (DECHAB) and mixed deciduous and coniferous (MIXHAB) as two key natural forest land covers. Additionally, we modeled the amount of forested areas of different size with low levels of stand-replacing disturbance (INTACT), and the level of fragmentation of the forest land cover irrespective of age class by using both natural features, such as lakes and mires, and anthropogenic features, such as agriculture and urban areas (FRAG; see Table 1).

Indicators of sustained yield wood production

Forest management intensity (FORINT).—In the BSR, managed forest rotation times for conifer range from 60 to 90 yr in Sweden (Fries et al. 2015), 81–121 yr in Latvia (Anonymous 2007), 80 yr in Belarus (Zinovskij 2006), and 80–120 yr in Pskov (V. Rezhetov, *personal communication*). This means that in an entire local forest management unit, subject to clear-felling forestry practices in the long term, a little over ~1% of the forest area should be harvested annually. Hence, forest loss values considerably over 1% per year would be unsustainable or be the result of natural disturbances such as fire and wind throw. Conversely, very low forest loss values indicate poor wood resource utilization. As a proxy for annual harvest by clear-felling, we used the forest loss raster maps for 2000–2013 created by Hansen et al. (2013). During this period, there were no catastrophic wind throws such as the storms "Gudrun" (2005) and "Per" (2007) and Västmanland fire (2014) in Sweden on forest land within the case study regions. However, in 2011 there was one large fire event in easternmost Pskov oblast on a bog complex.

Net forest gain (FORGAIN).—The net forest gain was estimated as the sum of gained forest area divided by the sum of the loss forest area for each 10 × 10 km grid cells. Due to the great variance within the raster data (net forest gain), it was log-transformed with a base of 10.

Accessible coniferous forest (ACC_CON).—By and large, the wood trade and forest industrial economy in the BSR focus on the conifers Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*; Tilli and Skutin 2004). Economic benefits in forestry are maximized if conifer stands are concentrated, and can be reached by permanent roads for both (1) harvesting by commercial thinning and final felling and (2) repeated silvicultural treatments by owners with the long-term vision to sustain wood yields over several rotations (Sundberg and Silversides 1988). Hence, the opportunity for economically sustainable production of coniferous wood should be higher in local landscapes

with higher proportion of coniferous forest areas and higher road density. To estimate the available relative amount of coniferous forest resources (ACC_CON), we used the EFI forest database with spatial resolution of 1 km^2 (Kempeneers et al. 2011). Using the scale of a local landscape, we selected 1 km^2 pixels with >70% conifers and then expressed this as the proportion of conifer-dominated pixels within local landscapes (10 × 10 km). In the BSR, the dominating harvesting machine system is based on harvester–forwarder teams practicing cut to length. The economically acceptable distance to forward timber from the harvesting site to a permanent forest road is <500 m (Sundberg and Silversides 1988). Thus, to provide full access to forest resources by road transportation, the road density should be >1 km/km^2 (Sundberg and Silversides 1988, Skogsstyrelsen 1991, Goltsev et al. 2012:126). To eliminate flaws of grid computing (Cai et al. 2013), we calculated road density at a 200-m resolution by using Gaussian kernel function with a radius of 500 m. The kernel density raster was then aggregated to 1 × 1 km resolution. Pixels with values <1 km/km^2 were omitted. Further, we computed proportions of valid pixels within 10 × 10 km grid cells. The resulting indicator of accessible coniferous forest was estimated as the coniferous forest resource (ACC_CON) multiplied by proportions of sufficient road density in 1 km^2 pixels, and summarized for local landscapes (10 × 10 km pixels).

Economic sustainability (ECONSUST).—During a full rotation of sustained yield forest management, the balance between costs and incomes for a forest stand changes in the management cycle from one final felling event to the next. While final felling yields high net revenue, site preparation and tree planting are investments that imply significant costs (Brukas and Weber 2009). Intermediate felling by thinning to improve stand productivity, and to gain additional wood volumes, ranges from being cost-neutral early in the rotation to yielding an increasing profit net revenue. Hence, to allow for intensive forest management based on silvicultural treatments, a forest management unit needs to include both regeneration areas and areas ready for commercial thinning and final harvest. A precondition for economic viability in an area is thus that both forest gain and forest loss should be spatially sufficiently juxtaposed within forest management units. To estimate this, we calculated the regression line between forest loss and forest gain among 100 km^2 virtual forest management planning units within each of the four regions and used the slope (BETA) as an indicator economic sustainability.

Indicators of biodiversity conservation

Deciduous forest habitat (DECHAB).—To estimate the available relative amount of deciduous forest as habitat (DECHAB), we used the EFI forest data with a spatial resolution of 1 km^2 (Kempeneers et al. 2011). This is motivated by the area requirements of specialized focal species (e.g., Angelstam et al. 2004). To estimate DECHAB at the scale of a local landscape (10 × 10 km), we selected all 1 km^2 pixels with >50% deciduous forest and expressed DECHAB as the proportion of local landscapes.

Mixed forest habitat (MIXHAB).—To estimate the relative amount of mixed forest resources as habitat (MIXHAB), we masked the forest cover by the raster themes CON and DECHAB with a spatial resolution of 1 km^2. To estimate MIXHAB at the scale of a local landscape (10 × 10 km), we selected all 1 km^2 pixels not classified as CON or DECHAB. This is motivated by the area requirements of specialized focal species (e.g., Angelstam et al. 2004). MIXHAB was expressed as the proportion within local 10 × 10 km landscapes.

Intact forest areas (INTACT).—Naturally dynamic boreal forests are dominated by late-successional stages after stand-replacing disturbances (Pennanen 2002) as well as gap dynamic forests and multi-cohort forests after low-intensity disturbances (Shorohova et al. 2011). Such forest landscapes should thus have a low proportion of stand-replacing forest loss and thus harvesting would not be detected by satellite-based remote sensing data (Potapov et al. 2015). To calculate the patch size distribution of forest areas with low forest loss at the stand scale, we mapped areas with low forest loss (<0.1% per year) during the period 2000–2013 in individual 1 km^2 cells and compared the size distributions of contiguous 1 km^2 cells with low forest loss among the case study regions. Contiguous patches were defined as 1 km^2 cells that touch adjacent cells in any direction, including diagonally. The results were

presented as histograms showing the area distribution among patch size classes (<1, 1–10, 11–100, >100 km^2).

Forest landscape fragmentation (FRAG).—As a result of forest clearing for agriculture and settlements in the BSR during Millennia, forest cover has been lost and fragmented. Forest loss is higher on more fertile soils and at lower altitudes (e.g., Angelstam and Andersson 2001). This is an obstacle to the maintenance of habitat networks that have sufficient amounts of habitat to be functional (Angelstam et al. 2011*a*). Due to past forest loss, local landscapes may thus have passed thresholds for the proportion of habitat that permits presence of particular species. There are two key thresholds. The first is when contiguous habitat is broken up into patches, thus no longer permitting percolation of individuals of different species through an unfragmented habitat. Individuals of species can usually move across shorter distances outside their habitat patch, a local landscape may be perceived as contiguous as long as the proportion of habitat exceeds 40–50% (With and Crist 1995, Fahrig 2003). We used a threshold of 40% remaining forest land as habitat for the potential occurrence of species that need contiguous forest (see Angelstam et al. 2017*a*). The second key threshold is when fragments begin increasing in inter-patch distance and thus isolation. We applied the Nagoya agreement's 17% target (CBD 2010) for what has been agreed internationally as the minimum proportion that needs to be allocated for protected areas. First, we included only 1 × 1 km pixels (see above) with >40% forest cover, that is, above the percolation threshold for the local forest stand scale. Second, the data were aggregated into coarser local landscapes (10 × 10 km). Finally, we re-classified raster data into three categories based on these two key threshold values, viz. (1) areas that permit percolation (i.e., >40% forest cover), (2) areas with intermediate levels of fragmentation within local landscapes (17–40%), and (3) areas that suffer from forest fragmentation (<17%).

Results

Sustained yield wood production

Forest management intensity (FORINT).—In both the Swedish Bergslagen–Mälardalen region and Latvia, the mean annual forest loss rates among 10 × 10 km pixels were very similar and ranged from 0.75 to 0.82% (Table 2). In contrast, the annual forest loss rates in Vitebsk and Pskov ranged from 0.30 to 0.21%. Additionally, the spatial distributions of different forest management intensities among the local 10 × 10 km landscapes were clearly different when comparing Swedish Bergslagen–Mälardalen and Latvia on the one hand and Belarusian Vitebsk and Russian Pskov on the other. While in the former two countries, FORINT was normally distributed, in the latter two FORINT was clearly skewed toward low forest management intensities (Fig. 2). The spatial distribution of loss rates among the four case study regions confirms the major difference between the western and eastern groups of countries (Fig. 3).

Net forest gain (FORGAIN).—All regions showed negative forest gain during the study period. Latvia had the lowest mean value (−0.73) and then followed Swedish Bergslagen–Mälardalen and Belarusian Vitebsk (−0.46 and −0.44). Russian Pskov had the highest mean value (−0.32; Table 2). The range of variation among local 10 × 10 km landscapes was lower in Bergslagen–Mälardalen and Latvia than in Vitebsk and Pskov (Table 2, Fig. 4). For Vitebsk, and especially Pskov, many of the local 10 × 10 km

Table 2. Indicator values for four indicators of provisioning ecosystem services and four supporting ecosystem services (habitat services).

Region	Provisioning ecosystem services				Supporting ecosystem services			
	FORINT	FORGAIN	ACC_CON	ECONSUST	DECHAB	MIXHAB	INTACT	FRAG
Bergslagen–Mälardalen (Sweden)	0.0075	−0.46	3699	0.64	0	0.55	3.29	3.29
Latvia (entire country)	0.0082	−0.73	677	0.49	0.39	12.38	3.18	3.18
Vitebsk (Belarus)	0.0030	−0.44	611	0.44	2.40	6.88	4.52	4.52
Pskov (Russia)	0.0021	−0.32	6.45	0.19	0.37	58.86	7.54	7.54

Note: For details on the indicator units and how they were calculated, see Table 1.

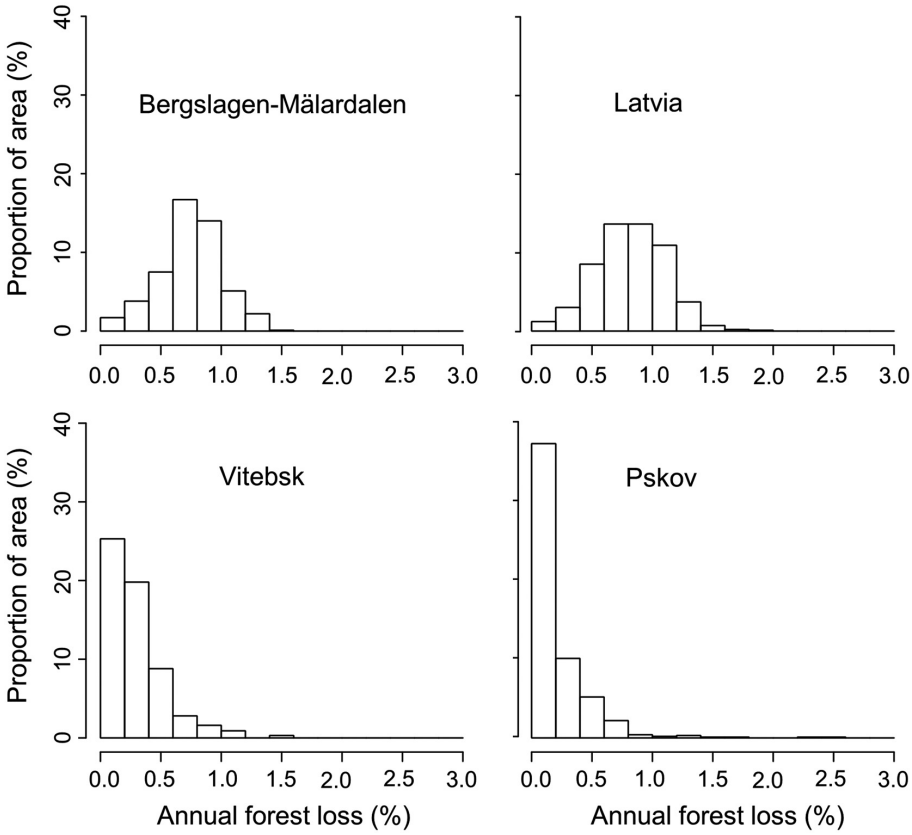

Fig. 2. Distribution of the proportion of the annual forest loss (FORINT) 2000–2013 as an indicator of forest management intensity inside the forest mask among 10 × 10 km grid cells in Bergslagen–Mälardalen counties in Sweden (380 grid cells), Latvia (648 grid cells), Vitebsk oblast in Belarus (400 grid cells), and Pskov in Russia (547 grid cells).

landscapes showed high net forest gain (up to 0.81 and 1.61, respectively; Fig. 3).

Accessible coniferous forest (ACC_CON).—The distribution of contiguous coniferous forest was concentrated to the NW part of the Bergslagen–Mälardalen region (Fig. 5). In Latvia and Vitebsk, coniferous forests had a patchy distribution. Finally, in Pskov, coniferous forest was very rare. Regarding road density, Bergslagen–Mälardalen stands out with the highest values, followed by Belarus, Latvia, and Pskov (Fig. 5, Table 3). The resulting index of accessible coniferous wood (i.e., a × b in Table 3) showed a very steep gradient from Sweden (3698) to Latvia and Vitebsk which were similar (677 and 611) and to Pskov (6; see Table 3). The map showing the ACC_CON index of the accessible coniferous forest also showed considerable variation among landscapes within each region. For example, the NW (i.e., the Bergslagen region; Angelstam et al. 2013) and SE parts of the Bergslagen–Mälardalen region were clearly different (Fig. 5). In Latvia and Vitebsk, the occurrence of higher ACC_CON values was patchily distributed throughout these regions, and in Pskov, there was virtually no ACC_CON.

Economic sustainability (ECONSUST).—Bergslagen–Mälardalen clearly showed the best relationship between forest loss and forest gain (slope 0.41 and R^2 = 0.64; Fig. 6). Next, Latvia and Vitebsk had similar slopes and R^2 values (Table 4). Finally, Pskov showed a very poor relationship with low slope (0.10) and R^2 value of 0.19.

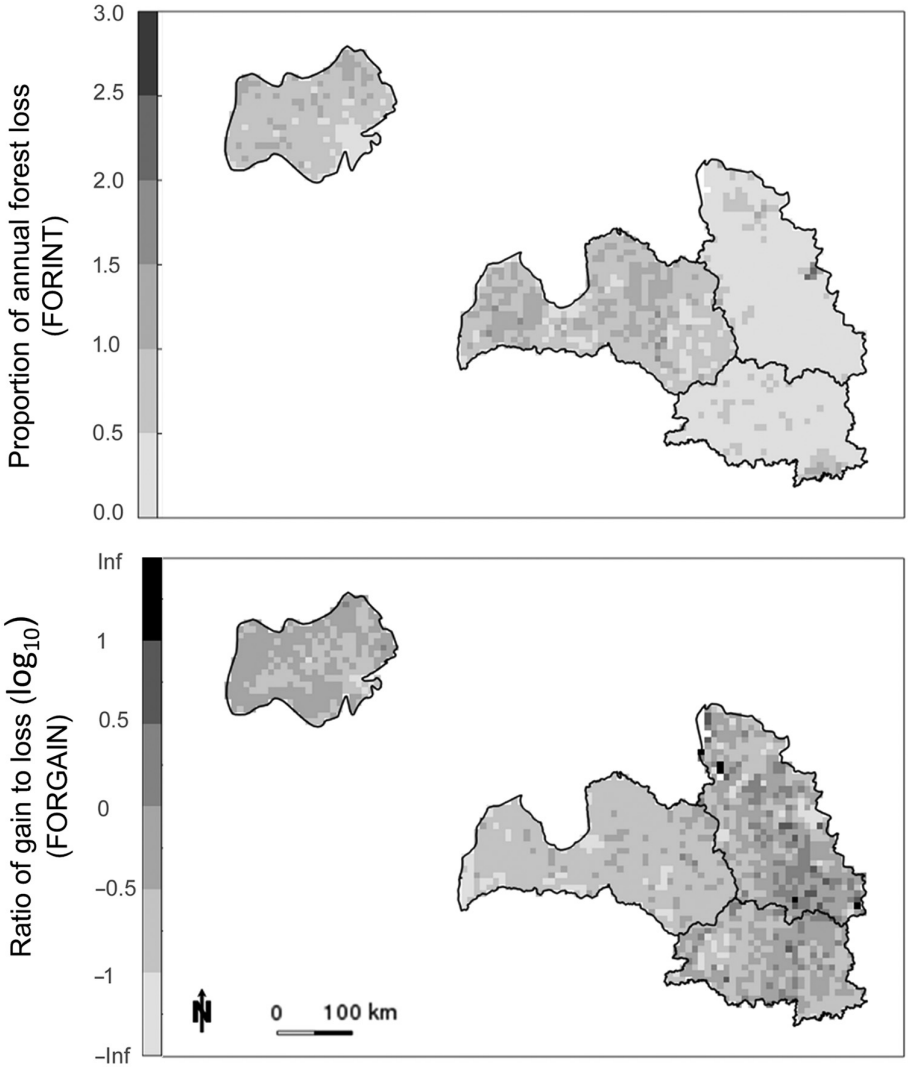

Fig. 3. Maps showing forest management intensity (FORINT; top) and net forest gain (FORGAIN; bottom) in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia. FORINT is expressed as the proportion of the annual forest loss 2000–2013 inside the forest mask among 10 × 10 km grid cells and FORGAIN expressed as the ratio of gain to loss expressed as $\log_{10}$ classes (see also Fig. 4).

Biodiversity conservation

Deciduous forest habitat (DECHAB).—The amount of 100-ha deciduous forest habitat patches was generally low (0–2.4%), but still clearly differed among the four case study regions. Bergslagen–Mälardalen was devoid of deciduous forest concentrations (see map in Fig. 7, and Table 2). In Vitebsk and Pskov, there were some large deciduous forest massifs.

Mixed forest habitat (MIXHAB).—Also, the amount of 100-ha patches with mixed deciduous and coniferous forest was clearly different among the four regions. In Bergslagen–Mälardalen, the amount of mixed forest concentrations was very low (0.55%), except some patches in the east (see map in Fig. 7, and Table 2). Latvia and Vitebsk had similar proportions of MIXHAB (12.4 and 6.9%, respectively) and spatial distributions of

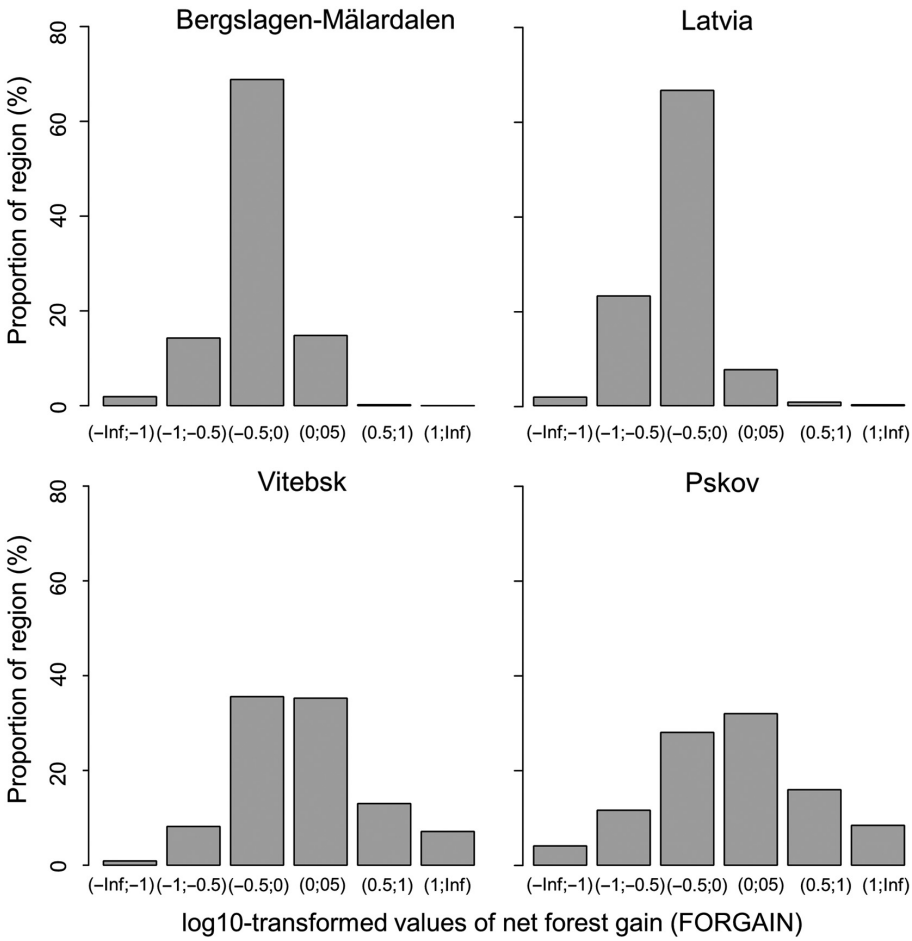

Fig. 4. Net forest gain (FORGAIN) divided into six $\log_{10}$ classes for the four regional case studies Bergslagen–Mälardalen in Sweden (380 grid cells), entire Latvia (648 grid cells), Vitebsk oblast in Belarus (400 grid cells), and Pskov oblast in NW Russia (547 grid cells).

mixed forest. Pskov oblast had one large contiguous mixed forest massif that covered on average 59% of the local 10 × 10 km landscapes.

Intact forest areas (INTACT).—The mean size of contiguous patches created by merging adjacent 1 km^2 pixels with <1% forest loss increased from Sweden (3.3 km^2) and Latvia (3.18 km^2) to Vitebsk (4.5 km^2) and to Pskov (7.5 km^2; see Table 2). The proportion of area located in patches >10 km^2 (i.e., 1000 ha) increased from Sweden (37%) and Latvia (33%) to Vitebsk (50%) and Pskov (70%; Fig. 8). As a result, Pskov and Vitebsk had large contiguous forest patches spread over the regions (Fig. 9).

Forest landscape fragmentation (FRAG).—All four case study regions were similar in terms of contiguous forest areas with a level of forest land fragmentation that exceeds the percolation threshold (Fig. 9). Histograms of the distribution of local landscape into different landscape fragmentation classes (<17%, 17–39%, >40%) also show that all regions have low proportions (3–13%) of local landscapes that suffer severe (<17%) forest landscape fragmentation (Fig. 10).

Discussion

Regional-level comparison reveals major differences

Landscape history gradients and harmonized open-access data.—This macro-scale study focuses on use of open-access remote sensing data as a base

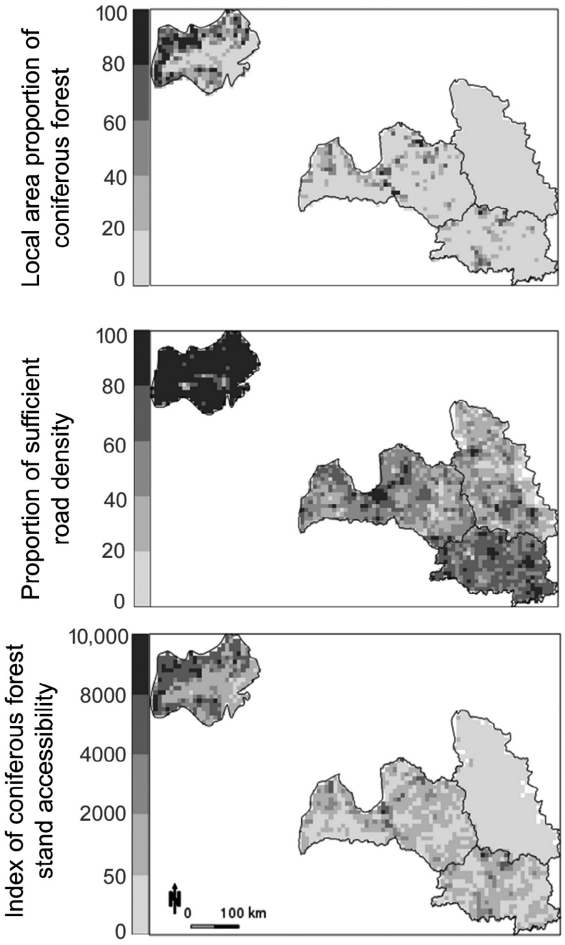

Fig. 5. Maps showing the local area proportion (%) of coniferous forest (top), road density (ACCON; middle), and an index of coniferous forest stands accessible from roads (CONWOOD; bottom) with an estimated forwarder distance of <500 m (c) in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

for comparing forest landscapes' opportunity for wood production and biodiversity conservation in regions with different histories of forest landscapes as coupled human-nature systems. With its large contrasts between forest landscape history and governance legacies in different parts, the BSR is unique as a time machine (Angelstam et al. 2011*b*). This offers the opportunity for both comparative studies of different governance contexts (Elbakidze et al. 2010) and the consequences of different landscape histories for sustaining populations of species that require naturally dynamic habitats (Roberge et al. 2008). We clearly show that different regional units within the BSR demonstrate very different states regarding wood production and biodiversity conservation functions of forest landscapes. Indeed, the results for the eight indicators developed and analyzed in this study demonstrate a clear inverse relationship among the four case study regions (Fig. 11).

International comparisons aimed at landscape and regional planning in the BSR's west and east are hampered by the accessibility of harmonized data. Many databases are produced for EU countries only, and in general, digital land cover data for Russia and Belarus are difficult or expensive to obtain. Implementing green infrastructure policy requires integrated spatial planning using evidence-based knowledge about the states of landscapes in different regions and countries (Angelstam et al. 2017*a*, Valasiuk et al. 2017), and multi-level cross-sectorial integration (Elbakidze et al. 2015). Thus, while coarse remote sensing data are needed to cover large spatial extents in a harmonized manner (e.g., Rose et al. 2015) as a base for strategic planning in the BSR region, for tactical and operational planning at the local level our approach needs to be succeeded by finer thematic resolution that match different forest age classes, tree species, and stand structure (Manton et al. 2005, Naumov 2017).

Assessment of wood production using indicators.— Indicators are approximations, the validity of which depends on the extent to which existing data sources mirror derived verifier variables. For example, the forest loss data used in this study have a minimum mapping unit of 25 × 25 m. This means that forest harvesting made by selective felling that covers spatial units smaller than this pixel size in the remote sensing data would be underestimated. For example, the low proportion of clear-cuts in Vitebsk oblast (8%; Zinovskij 2006) makes it likely that forest loss data derived from remote sensing underestimate the effects of forestry on biodiversity. Additionally, indicators may represent relative rather than absolute numbers. We used the slope for the relationship between gain and loss as an indicator of economic sustainability, with the assumption that the closer the slope is to 1, the higher the probability of economic sustainability. To see whether forest gain has occurred in absolute terms, however, a

Table 3. Indicators of conifer resource density, road density, and an index for the amount of accessible coniferous forest by 10 × 10 km grid cells.

Region	Variables	Mean	Median	Range	Variance	Skewness
Bergslagen–Mälardalen (Sweden) (n = 380)	A. Conifer resource in 1 km^2 pixels dominated by coniferous forest)	40.62	35	0-8	876.58	0.29
	B. Proportions (%) of areas with road density ≥ 1 (km/km^2)	88.52	94	8–100	255.16	−2.86
	Accessible coniferous forest index (A × B) (ACC_CON)	3698.56	3201	0–956	7,642,968	0.29
Latvia (entire country) (n = 648)	A. Conifer resource in 1 km^2 pixels dominated by coniferous forest	12.91	4	0-5	335.07	1.87
	B. Proportions (%) of areas with road density ≥ 1 (km/km^2)	48.68	47	2–100	308.35	0.21
	Accessible coniferous forest index (A × B) (ACC_CON)	677.1	199	0–8360	1,223,367	2.67
Vitebsk (Belarus) (n = 400)	A. Conifer resource in 1 km^2 pixels dominated by coniferous forest	9.65	2	0–73	225.96	2.15
	B. Proportions (%) of areas with road density ≥ 1 (km/km^2)	64.23	66	2–99	205.67	−0.51
	Accessible coniferous forest index (A × B) (ACC_CON)	611.23	150	0–6745	1,029,591	2.78
Pskov (Russia) (n = 547)	A. Conifer resource in 1 km^2 pixels dominated by coniferous forest	0.21	0	0–27	2.19	12.87
	B. Proportions (%) of areas with road density ≥ 1 (km/km^2)	35.4	34	1–97	324.11	0.44
	Accessible coniferous forest index (A × B) (ACC_CON)	6.45	0	0–1377	4214.11	18.56

sufficient amount of time is needed for vegetation to recover. Therefore, if the time lag for forest recovery in terms of regeneration of young forest is 5 yr (Potapov et al. 2015), loss from T_0 to T_10 should be compared with gain from T_5 to T_15. Another caveat regards the estimate of the accessible amount of coniferous stands (ACC_CON). Not only forest road density affects accessibility of wood for road transportation. Additionally, the distance from harvested wood at the forest roadside to value-adding units such as paper and pulp industries and sawmills should be considered (Naumov et al. 2017). In this study, the Swedish Bergslagen–Mälardalen, with access to value-adding industries on good public roads, differs from the case study regions in Latvia, Belarus, and Pskov, all lacking paper and pulp industries.

With forest being the natural potential vegetation in the BSR (Bohn et al. 2000), in all case study regions the dominating reason for loss of forest land, that is, not loss of stands with taller trees, is clearing of forests for agriculture and gray infrastructure. In Sweden, pastures and poor agricultural land were re-forested after WW2, which led to increased forest land. In Latvia, Belarus, and Russia, forest cover is currently increasing due to encroaching forest on abandoned agricultural lands with less fertile soils and on wet soils (Angelstam et al. 2005, Potapov et al. 2015). Further intensification of forest management is currently desired to increase economic benefits (e.g., Naumov et al. 2016). This is further stressed by current policies concerning increasing use of bioenergy.

Assessment of the opportunity for biodiversity conservation using indicators.—Clearly, current opportunities for biodiversity conservation are higher in the Belarusian Vitebsk and Russian Pskov regions, and still to some extent in Latvia, than in the Swedish case study region. This is indicated by lower forest management intensity, higher proportions of mixed deciduous–coniferous forest, and larger patches of forests with low forest loss toward the east. The status of species in western vs. eastern parts of the BSR confirms this. By compiling the breeding status and population trends for 17 specialized bird species that together represent all forest habitats in the seven countries of the BSR, Angelstam et al. (2004) found that the distribution among the categories extinct, declining, no trend, and increase was skewed toward a more negative situation in the three western

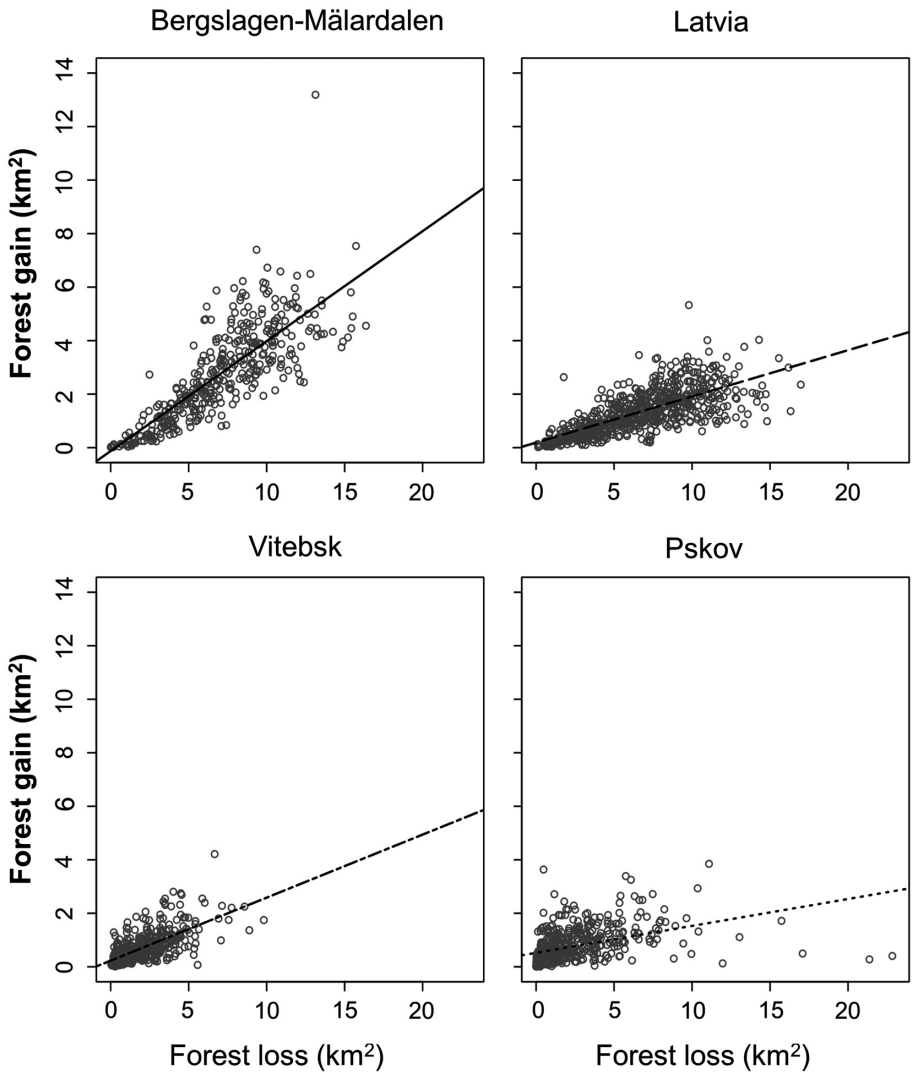

Fig. 6. The ECONSUST indicator based on regressions between forest loss on the x-axis and forest gain on the y-axis among 10 × 10 km pixels in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

Table 4. Linear regression between gain and loss with 10 × 10 km areas within the four study areas.

Region	Intercept	Slope	R^2	n	P
Bergslagen–Mälardalen (Sweden)	−0.13	0.41	0.64	380	<0.0001
Latvia (entire country)	0.19	0.17	0.49	648	<0.0001
Vitebsk (Belarus)	0.23	0.24	0.44	400	<0.0001
Pskov (Russia)	0.52	0.10	0.19	547	<0.0001

Note: R^2 is used as the indicator ECONSUST (see Table 1).

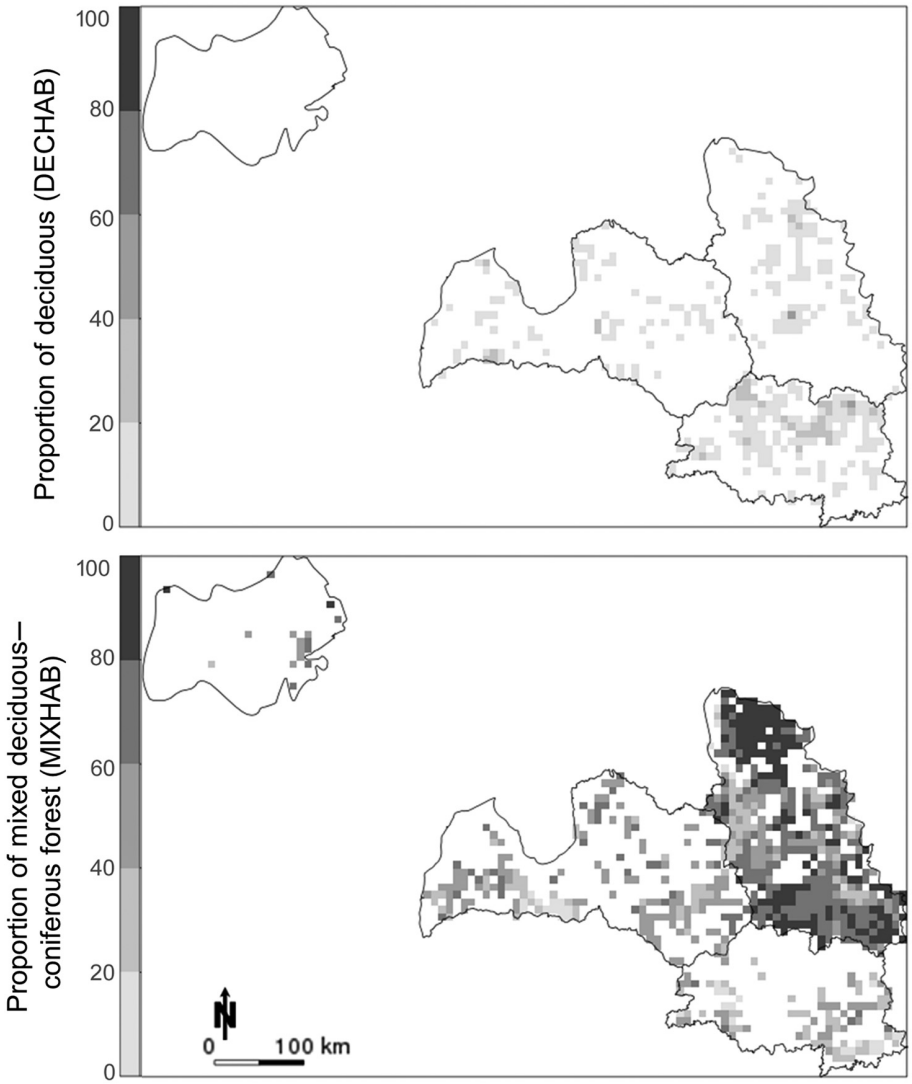

Fig. 7. Maps showing the proportion (%) of deciduous (DECHAB; top) and mixed deciduous–coniferous forest (MIXHAB; bottom) in local 10 × 10 km landscapes in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

countries (Denmark, Sweden, and Finland) with more intensive forest landscape management compared to the four eastern countries (Estonia, Latvia, Lithuania, and Poland). All six extinctions were confined to the three western countries.

However, the current status does not imply that this west–east trend in biodiversity status will remain. The mechanism is that a long-term wood production focus in forest management gradually modifies the tree species composition; removes dead wood in different decay stages, old stands, and trees; and changes the microclimate (e.g., Vellak and Paal 1999). Intensification of forestry in Latvia (Potapov et al. 2015, Rendenieks et al. 2015*a*) is already affecting green infrastructures for wood production positively at the expense of opportunities for biodiversity conservation. This is clearly noted in Latvia's report to EU on unfavorable status of forest habitats (http://bd.eionet.europa.eu/article17/reports2012/habitat/report/?period=3&group=Forests&country=LV&region=). For example, while from a sustained

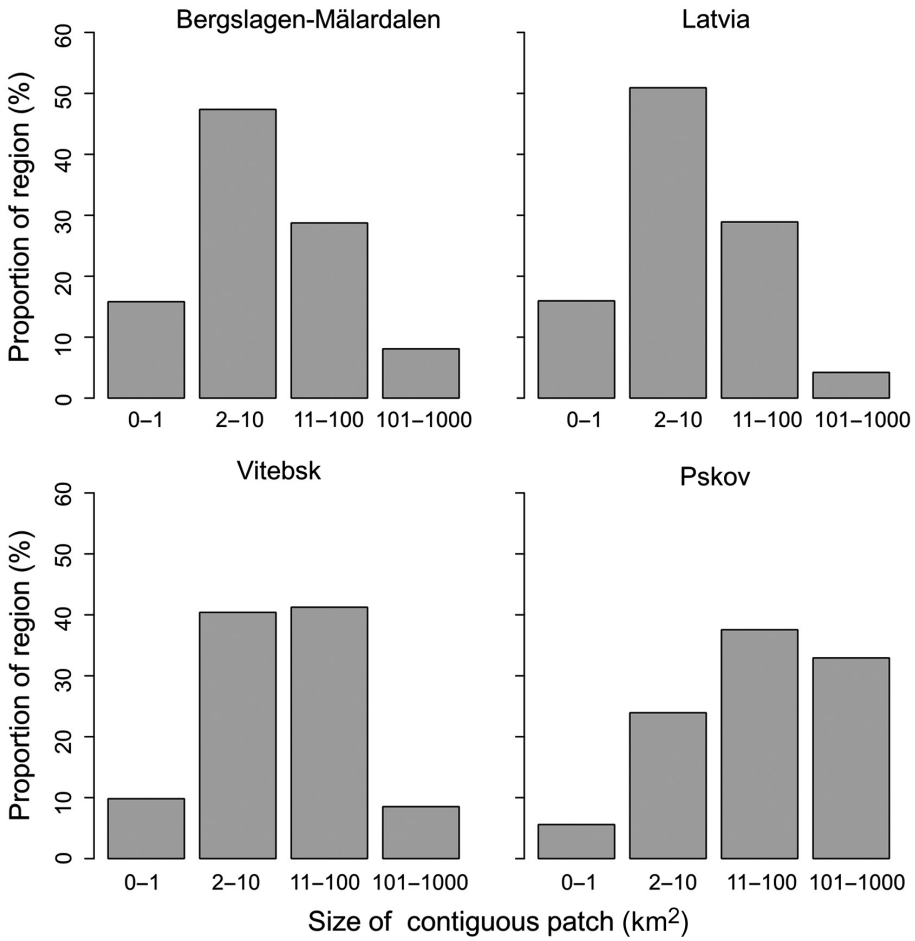

Fig. 8. Histograms showing the proportion of differently sized contiguous areas with low forest loss (the INTACT indicator) in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

yield perspective forest gain means improved opportunities for wood and biomass harvest, neither rapid vegetation regeneration after forest harvesting nor encroaching vegetation on abandoned agricultural land, are factors that improve forest biodiversity conservation in the short and medium term.

Our study, as well as Potapov et al.'s (2015) analysis of Eastern Europe's forest cover dynamics from 1985 to 2012 and Hsu and Zomer (2016), shows that Latvia stands out with high forest loss. Additionally, the fact that there were more deciduous trees in the easternmost case study regions does not mean that the relatively better status for biodiversity in Latvia will remain in the long term. The limiting factors for most focal species of birds, invertebrates, lichens, mosses, and fungi are presence of old (i.e., mature in an ecological sense) stands, stands with mixed tree species, and the amount of the dead wood (preferably large dimensions and in different decay stages; e.g., Siitonen 2001). Increased economic activity in Latvia is likely to cause reduced proportions of biologically old stands (Rendenieks et al. 2015*b*) resulting in reduced habitat quality, patch size, and functional connectivity (Angelstam and Andersson 2013) in the short and medium term. Modeling suggests that due to current tree retention practices during final felling, the conditions for biodiversity conservation could improve in the long term (Roberge et al. 2015). However, the amount of tree

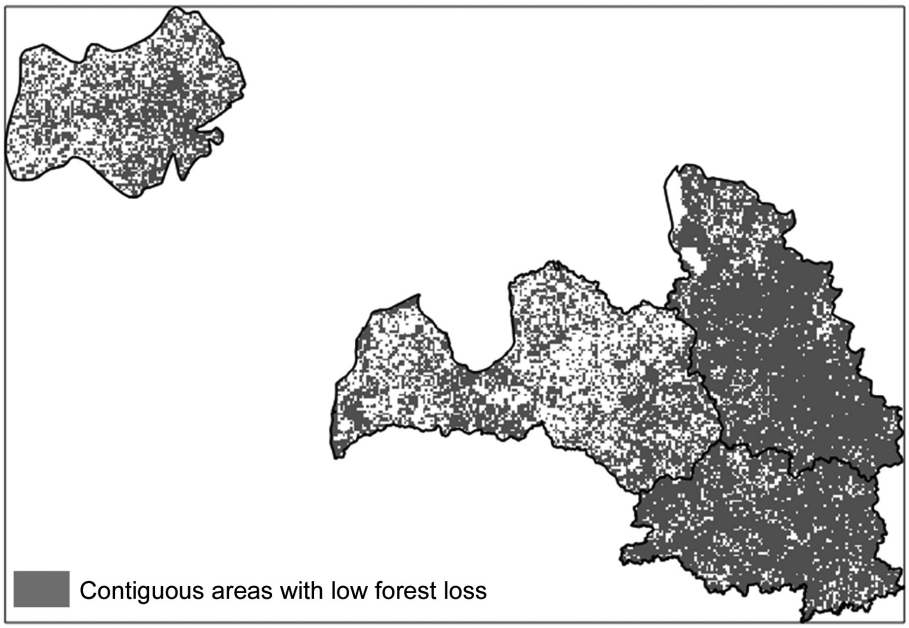

Fig. 9. Map showing the location of contiguous areas with low forest loss (INTACT) in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

retention is low in Sweden (Gustafsson et al. 2012), and the quality and connectivity of set-asides are poor (Elbakidze et al. 2011). Additionally, maintenance of natural disturbances such as fire and gap dynamic is important for biodiversity conservation (Shorohova et al. 2011). Thus, the overall conclusion is that forest landscapes with a long history of effective wood production require landscape restoration in terms of increased amount natural structures at multiple spatial scales (Halme et al. 2013). Countries and regions aiming at forestry intensification should be aware of this.

The analyses of the remaining area proportion of forest showed that there is in principle potential for conserving biodiversity at the local 10 × 10 km landscape scale, except in the parts of the case study regions that are dominated by agricultural land. However, to deal with habitat loss and forest fragmentation of particular forest types and age classes for biodiversity conservation requires the combination of restoration, management, and conservation in the long term by segregation of different forest functions. This includes two distinct aspects. The first one is the gradual transformation of forest land to agricultural lands affecting the representation of different forest types. This forest loss is generally linked to a systematic bias toward loss of more fertile forest site types being transferred to agricultural land (e.g., Angelstam and Andersson 2001). The second aspect is the reduction of older age classes linked to intensified forest management including lower final felling ages. For example, in Latvia, there was a considerable increase in the total final felling volumes and concentration of clear-cuts after the restoration of state independence in 1991. Thus, while from an economic perspective, forest will have re-appeared 3–5 yr after clear-felling (Potapov et al. 2015; NASA Earth Observatory (http://eoimages.gsfc.nasa.gov/images/imagerecords/86000/86221/latvia_etm_2012_lrg.jpg), the regeneration time of forest habitats suitable for natural forest specialist species is much longer than the current rotation time.

Functional green infrastructure requires integrated spatial planning

This study stresses the need to develop regionally adapted solutions for functional green infrastructure to secure both a high sustained yield wood production and biodiversity conservation. This requires both compass and gyroscope (Lee 1994). Compass is about knowing the states and trends of regions, based on evidence-based

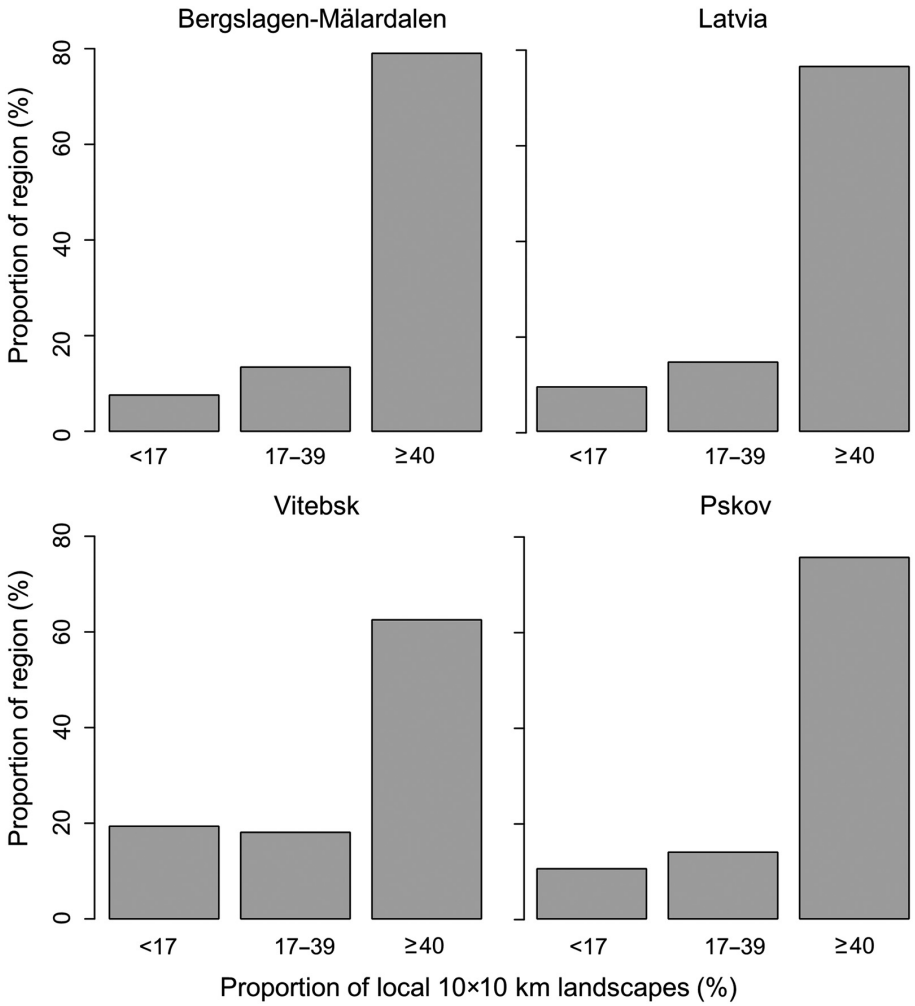

Fig. 10. Histograms of the distribution of local landscapes (10 × 10 km pixels) into different forest landscape proportion classes in percentage in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia. The number 17 corresponds to the lowest possible forest cover that could potentially support the Nagoya target of 17% protected area (CBD 2010), and 40% represents the forest cover above which forest area is contiguous (With and Crist 1995).

knowledge about ecological targets/tipping points and measures, for managing, restoring, and recreating habitats for species, including preferences of humans, habitats, and ecosystem processes and functions. Gyroscope is about developing collaborative learning among researchers, stakeholders, and policy-makers, including managers and users in the field, businesses, policy actors, local administrations, and citizens. Together, an integrated focus on ecological and social systems forms the base for transparent sustainable development processes toward sustainability through functional green infrastructures. The terms landscape approach (Axelsson et al. 2011, Sayer et al. 2013) and landscape restoration (Laestadius et al. 2015) capture this. Concepts such as Biosphere Reserve, Model Forest, and Long-Term Socio-Ecological Research platforms are solutions being explored in Sweden and Russia (e.g., Elbakidze et al. 2010, 2013, Angelstam and Elbakidze 2017) and in Latvia (Melecis et al. 2014). However, their success hinges on their ability to critically assess landscape trajectories by monitoring the outcomes of attempts toward collaborative spatial planning.

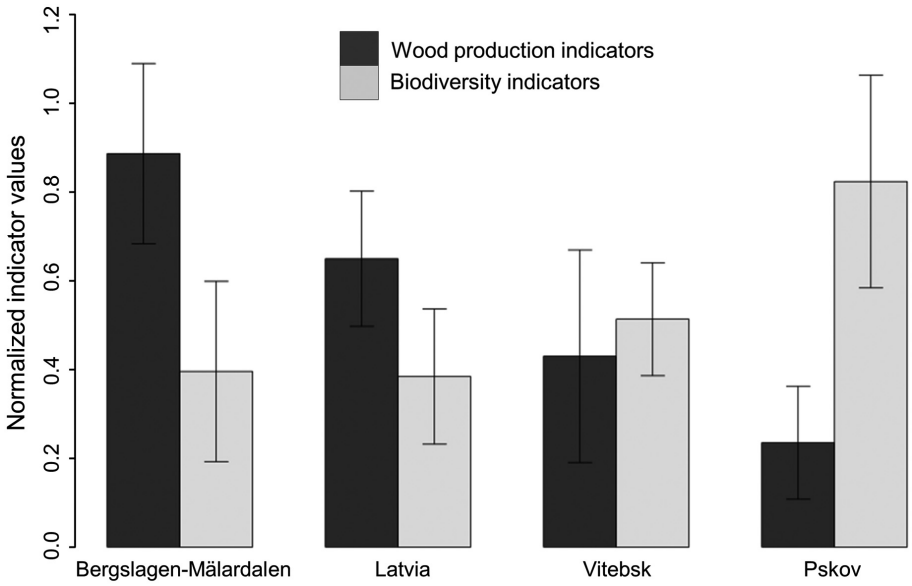

Fig. 11. Means (depictured as bars) and SE (as lines) of normalized values for four indicators of wood production (dark gray) and four of biodiversity conservation (light gray; Table 2), respectively, among the in the four regional case studies Bergslagen–Mälardalen in Sweden, entire Latvia, Vitebsk oblast in Belarus, and Pskov oblast in Russia.

Our comparative study confirms the work of Mönkkönen et al. (2014), who concluded that maximizing sustained yield wood production and effective biodiversity conservation cannot be combined in the same area. This is primarily due to the fact that the habitat suitability for a number of key biodiversity indicators is strongly linked to old-growth forest with high quantities of diverse types of dead wood (Tikkanen et al. 2007), but also undisturbed early post-disturbance and middle-aged successional stages with deciduous trees (Angelstam et al. 2004). Intensification of sustained yield forestry leads to a decrease in the proportion of forest development stages important for biodiversity conservation. These factors thus require policy-makers to choose between conflicting priorities and for planners to make trade-offs between economic and biodiversity values at different spatial scales (European Commission 2013*a, b*). This dilemma thus requires application of strategic regional and tactical spatial planning and management approaches with a landscape and regional perspective (Angelstam 1998, Puettmann et al. 2009).

Regional gap analysis is a method that strategically assesses the extent to which networks of areas set-aside for conservation represent the different representative land covers of a region (e.g., Scott et al. 1993, Angelstam et al. 2017*c*). Angelstam and Andersson (2001) and Lõhmus et al. (2004) used the emerging empirical knowledge about how much habitat is needed to maintain functional habitat networks for biodiversity conservation, and assessed the extent to which this can be satisfied through formally protected areas, voluntary set-asides and sustainable use of the matrix surrounding protected areas and set-asides. To secure green infrastructure functionality, tactical spatial planning based on, for example, habitat suitability modeling using forest spatial data with sufficient thematic resolution needs to follow (Manton et al. 2005).

Past and current trajectories of intensive forest landscape use mean that management and restoration for biodiversity conservation as well as area protection are needed in the two western case study regions with a longer landscape history. However, implementation is unlikely to be successful where the forest use history is very long and intense (Angelstam et al. 2011*a*). At the same time, there is opportunity for intensified forest management for wood and biomass in the

easternmost case study regions with shorter landscape histories (Naumov et al. 2016, Angelstam et al. 2017*b*). However, caution should be applied because too much focus on intensification of forest management for wood production threatens forest biodiversity. This difference between the western and eastern regions supports the case for ensuring a balance between maximizing sustained yield and maintaining the potential for biodiversity conservation. A sole focus on maximizing sustained yield will lead to sustainable economic returns, but at the cost of causing harm to biodiversity (Triviño et al. 2015). This study is thus in line with Edwards et al.'s (2014) study on Borneo, which concluded that land-sparing where some forests are left unlogged and more species resulted in higher abundances of species and higher species richness than land-sharing with lower intensity forest management.

Maximum sustained yield and effective biodiversity conservation are not compatible. Thus, rather than focusing solely on the economic sustainability, a multi-perspective approach should be used to evaluate economic, ecological, and sociocultural sustainability. This means promotion of sustainable forest management policy, which is consistent with securing the provision of ecosystem services and integrated catchment management (e.g., Cook and Spray 2012), and requires the same set of steps. First, the extent to which networks of formally and voluntarily protected areas form functional green infrastructures that represent a regions' forest land covers types needs to be assessed (e.g., Angelstam et al. 2011*a*, Elbakidze et al. 2016). Second, regarding general guidelines about tree retention during thinning and final felling linked to voluntary forest certification (Elbakidze et al. 2016), what is the most efficient spatial distribution of retention trees across scales for biodiversity conservation? Third, which landscapes have few ecological (and social) values and are thus particularly suitable for intensive forest management (Andersson et al. 2013). Fourth, given that human habitat selection for recreation purposes is similar to the habitat characteristics for biodiversity conservation (Giergiczny et al. 2015), segregation of different forest landscape functions is also desirable to maintain forests' social values.

Three examples of forest landscape zoning toward functional green infrastructure are the Ekopark concept developed by Sveaskog Co. in Sweden (Angelstam and Bergman 2004, European Commission 2013*b*), the Latvian Ecoforest (E. Peterhofs, *personal communication*), and the Russian forest zoning system (Lazdinis and Angelstam 2005, Naumov et al. 2017). However, systems of forest land tenure, and the spatial configuration of them on the one hand, as well as the cultures and legacies of what forestry is and is expected to deliver (Brukas et al. 2011, Brukas 2015) on the other, effectively determine the extent to which segregated approaches are feasible. In Sweden and Latvia, small patches of ownership are a key challenge (e.g., Rendenieks et al. 2015*a*). Unless land owners collaborate across borders, zoning approaches are increasingly feasible with increasing size of forest management units within the same ownership. However, even if regions dominated by a single land owner, the sectorial silos of forestry and conservation imply that different stakeholders and actors have different and rival objectives. Thus, power relations and not evidence-based knowledge to balance production and biodiversity relationships may rule.

Acknowledgments

This study was made with funding from the Swedish Research Council Formas (grant number 2011-1737) to Per Angelstam and from the Swedish Institute (grant number 10976/2013) to Marine Elbakidze. We thank Kyle Eyvindson and Alexander Vinchevsky for valuable comments on the manuscript.

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