



Riparian buffer zones in production forests create unequal costs among forest owners

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Abstract

Riparian buffer zones (RBZs) are an important instrument for environmental policies for water and biodiversity protection in managed forests. We investigate the variation of the cost of implementing RBZs within different property size classes across the size range of non-industrial forest owner properties in Southern Sweden. Using the Heureka PlanWise decision support system, we quantified the cost of setting aside RBZs or applying alternative management in them, as the relative loss of harvest volume and of net present value per property. We did this for multiple simulated as well as real-world property distributions. The variation of cost distribution among small properties was 4.2–6.9 times higher than among large properties. The interproperty cost inequality decreased non-linearly with increasing property size and levelled off from around 200 ha. We conclude that RBZs, due to the irregular distribution of streams, cause highly unequal financial consequences for owners, with some small property owners bearing a disproportionately high cost. This adds to previous studies showing how environmental considerations differentially affect property owners. We recommend decision makers to stimulate the uptake of RBZs by alleviating these inequalities between forest owners by including appropriate cost sharing or compensation mechanisms in their design.

Keywords Forest management · Riparian buffer zone · Property size · Distributional fairness · Forest policy

Introduction

Riparian forests provide important structures for the maintenance of both terrestrial and freshwater biodiversity, prevent erosion, mitigate the leakage of chemicals to surface waters, and regulate the quality of surface waters (cf. Gundersen et al. 2010; Kuglerová et al. 2014). Beyond the landscape in which they are situated, functional riparian forests can contribute to solving the international public good suffering from excessive nutrient runoff, like is the case with terrestrial nutrient runoff to the Baltic Sea causing eutrophication

(Futter et al. 2010; Sagebiel et al. 2016). Therefore, it is commonly recommended to set-aside riparian buffer zones (RBZs) during forest harvesting in the form of unharvested strips of a certain width along the waterbody, although recommended characteristics of RBZs (e.g. width, allowed management, etc.) vary between different Nordic and Baltic countries (Ring et al. 2017). However, streams, rivers, and the hydrologically connective zone between forest soil and surface water are not evenly distributed due to topographical and soil heterogeneity in the forest landscape (Kuglerová et al. 2014).

Previous research has shown that at the landscape scale, the cost of implementing RBZs is proportional to the area that is set-aside at the landscape scale (Sonesson et al. 2021). Landscapes, however, are often not managed as a whole but rather by many property owners. Approximately 60% of European forests are privately owned in properties that vary widely in size (Živojinović et al. 2015; Weiss et al. 2019). There is a clear distinction in European regions with forests that are predominantly privately or predominantly publicly owned (Pulla et al. 2013). This means that the rate of co-occurrence of private forest properties is high within

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individual landscapes, which makes that those landscapes have a highly complex ownership structure. Often privately owned properties are very small; 40% of Austrian privately owned forest properties are < 3 ha, 62% of French privately owned properties are 1–4 ha, and 57% of the privately owned forest in Germany is in properties < 20 ha (Živojinović et al. 2015). In the southern part of Sweden (a region called Götaland), of which our case study area is a part, most forest (79%) is owned by non-industrial private forest owners (NIPF owners) and the average NIPF property encompasses 28 ha of forest (Swedish Forest Agency 2023). NIPF owned forest is used for commercial timber production in Sweden but is different from industrial and public forest because it is managed by private individuals rather than companies and governments. Industrial and public owners usually manage larger areas of forest of several hundreds of hectares or more (in average = 278 ha in Götaland 2021; Swedish Forest Agency 2023).

We can hypothesize that when implementing RBZs, some forest owners may be affected significantly more than others due to the higher proportion of forest land in the RBZs on their property. This distributional inequality is dependent on the spatial distribution of properties and streams and should be dependent on the scale at which there is an inequality in the distribution of properties and streams. This means that the coarser the distribution of RBZs is relative to the property size, the higher the distributional inequality of RBZs is among properties. This may lead then to the classic collective action problem of disproportionate burden sharing (Boadway and Hayashi 1999). The alleviation of inequalities due to RBZs through shared landscape planning has been proposed already more than 25 years ago but remained without widespread uptake (Fries et al. 1998). Forests ecosystems provide certain large benefits to the whole society while some forest owners need to bear the cost of the provision of these benefits if not alleviated by correcting policies (Lant et al. 2008; Fisher et al. 2008). By the logic of collective action theorem, such situations lead to the suboptimal provision of public benefits (Muradian 2013). At the same time, landscape-scale spatially optimized planning can be a relatively affordable way of increasing the efficiency of multi-objective forest management (Mazziotta et al. 2023), emphasizing the potential of planning beyond property boundaries. Therefore, to inform policymaking, we study the distribution of streams and forest properties of varying sizes and how it impacts the cost distribution of RBZs among forest owners.

We test the above hypothesis by quantifying the variation of the cost of implementing RBZs within different property size classes across the size range of non-industrial forest owner properties in Southern Sweden. We designed the RBZs according to the recommendations of the Swedish Forest Agency (see “Stream network and riparian buffer

zones” section). We use the forest decision support system Heureka PlanWise (Wikström et al. 2011) to simulate the forest development with and without hydrologically adapted RBZs and left unmanaged or under an adapted management regime dependent on the characteristics of the existent forest. We use an actual property map and a set of simulated fictional property maps to assess the distributional effects across the range of property sizes characteristic of non-industrial forest owners in Southern Sweden.

Methods

Study area

We studied 840 km² of production forests in a 1306 km² area in Hässleholm municipality, Southern Sweden. The study area contains a variety of monoculture and mixed forests that are common in the Baltic region. Most of the current forest has been regenerated by planting and is intensively managed for production. The most common tree species is Norway spruce (*Picea abies*, 50% of volume), followed by Beech (*Fagus sylvatica*, 14%), Birch (*Betula* spp., 10%), Scots pine (*Pinus sylvestris*, 10%), and Oak (*Quercus robur*, 7%). The soils in the study area are dominated by histosols (from peatlands) and arenosols (from unsorted glacial sediments), which are the dominating soil types in most of Southern Sweden (SLU 2020). Topographically, the study area is within the normal range of variation for Southern Sweden with an average elevation of 92 m (sd = 33 m), an average slope of 2.6° (sd = 2.5 degrees), and a ruggedness index of 1.0 (sd = 0.9; Riley et al. 1999; for further details see Appendix S1). The ownership structure is typical for Southern Sweden with predominantly small non-industrial private forest holdings (86% NIPF owned in Hässleholm, 78% NIPF owned in Southern Sweden). In particular, the ownership structure of the study area is relevant to the research question because there is a variety of property sizes from small to large, and this type of ownership structure is common in European forest landscapes (Pulla et al. 2013; Živojinović et al. 2015; Weiss et al. 2019).

Forest data

The basic forest characteristics were taken from a raster map of forests in Sweden (25 × 25 m) based on satellite and observational data from the national forest inventory (SLU 2010; Eggers et al. 2015). After segmenting the map into stands, some complementary attributes, needed to enable the simulations, were added by nearest neighbour matching stands based on species-wise volume, canopy height, basal area, and diameter at breast height to national forest inventory plots from the wider region. In total, there were

23,617 stands in a 725 km² area of forest. For each stand, we included the environmental conditions (location, elevation, slope, climate, site index, soil type, and soil moisture) and vegetation characteristics (tree species composition, mean age and height, and understorey vegetation type; see https://www.heurekaslu.se/wiki/Import_of_stand_register for a detailed description of all required variables).

Stream network and riparian buffer zones

Here, we define streams as all bodies of surface water flowing in a channel. This definition thus includes streams of all sizes and origins, from ditches to rivers. RBZs are currently required in Sweden to protect habitats and species of interest and to prevent chemical pollution of surface waters, but there are no specific legal or certification requirements concerning the width or management of the RBZ (Ring et al. 2017). The Swedish Forest Agency estimates that the average width of RBZs is 11 m, but that 31% of all riparian zones are not set-aside during harvesting (Swedish Forest Agency 2022). We followed best practices in stream and RBZ delineation as well as non-binding recommendations from the Swedish Forest Agency to create variable-width RBZs (10–15 m wide) that reflect hydrological characteristics of the riparian zone and groundwater discharge zone (i.e. where there is a disproportionately high influx of groundwater into the stream; Kuglerová et al. 2014; Swedish Forest Agency 2014; Laudon et al. 2016). Such variable-width RBZs are believed to increase the (cost-) effectiveness of water quality and biodiversity protection (Kuglerová et al. 2014; Laudon et al. 2016).

We used a 1 × 1 m resolution Digital Terrain Model (DTM, Lantmäteriet 2021) of all watersheds that overlap with Hässleholm municipality to model the stream network. We used the Python scripting interface of WhiteboxTools for this (Lindsay 2016c). We first pre-processed the DTM to make it suitable for stream delineation by removing elevated road segments, breaching embankments so that streams can pass roads, and then filling pits (i.e. correcting probable errors in the DTM; Lindsay 2016a, 2016b). Next, we calculated the upslope catchment size for each cell in the DTM (flow accumulation) using the D8 algorithm (i.e. water flows from each cell to its steepest downslope neighbour; O'Callaghan and Mark 1984) and, finally, extracted the streams using a 10-ha catchment area threshold (Ågren et al. 2015; Paul et al. 2023). Additionally, we identified groundwater discharge zones by extracting streams at a 6-ha catchment area threshold within a 50-m distance of the 10-ha threshold streams.

We verified the extracted 10-ha stream network by visiting 24 locations in the study area to look for 50 mapped streams between the 11 and 14 July 2022. Since the field-work was carried out in the driest part of the year, we looked

for stream channels and vegetation that indicated the presence of water. Stream channels did not have to be filled at the time of the visit. We found 47 out of 50 mapped stream channels (94%) during the field surveys.

To delineate variable-width RBZs, we calculated the depth-to-water index (Murphy et al. 2008) for the 10-ha and 6-ha stream networks. To set a depth-to-water threshold, we calculated the total area of fixed-width riparian buffer of 12.5 m (the average buffer width recommended by the Swedish Forest Agency, Swedish Forest Agency 2014) and chose a depth-to-water threshold that resulted in a similar total area of RBZs. This resulted in a depth-to-water threshold of 0.25 m and a total area of 3027 ha of RBZs (4% of the total forest area).

Finally, to be able to apply alternative management to the RBZs, we overlaid the RBZs with the forest map and used the buffer zone tool in Heureka PlanWise to split stands according to the RBZ borders (Lundström et al. 2018). This resulted in 35,599 stands in total.

Forest management

We simulated forest development using the PlanWise software from the Heureka forestry decision support system (version 2.20.0; Wikström et al. 2011). Heureka PlanWise simulates forest growth based on the input data for forests, a forest map of stands, a proposed set of management rules, and a set of sub-models that represent ecosystem processes that are calibrated using growth data from the Swedish National Forest Inventory. We used largely the default simulation settings that resemble common Swedish forest management; we describe deviations from the default and important choices in Table 1. The minimum rotation time was set to 1.2 of the legal threshold since shorter rotations are not common in practice (Lodin and Brukas 2021). We assumed that the climate changed according to the ECHAM5 A1B (business as usual emissions, climate change model: https://www.heurekaslu.se/wiki/Climate_Model). In Heureka, climate change includes the effects of increasing atmospheric CO₂ concentrations and temperature on tree growth based on the results of the process-based model BIOMASS (McMurtrie et al. 1990). We simulated two alternatives for forest management: one alternative without consideration of RBZs and one with.

In the first alternative, all stands, including RBZs, were managed for wood production under a clear-cut management system. This implies clear-cutting of mature stands, followed by artificial regeneration, a pre-commercial thinning, and, usually, one to three commercial thinnings. Clear-felled stands are replanted with the dominant species from the previous generation.

In the second alternative, we treated RBZs differently from their parent stands (Table 2). The RBZ was set-aside

Table 1 Overview of the variables and settings in the treatment schedule simulation

Variable	Description	Setting
Minimum rotation time	The minimum age at which a stand could be felled. We multiplied the legal minimum rotation time by a percentage to reflect that forest owners generally harvest later than the legal minimum. The legal minimum is dependent on tree species and growing conditions	1.2 times the legal minimum
Final felling delay	The number of 5-year periods by which final felling could be delayed compared to the minimum rotation time. We did this to reflect the fact that not all stands are harvested with the same timing	0–6 periods, one alternative is selected in the optimization
Number of thinnings	The number of thinnings applied during a single rotation	0–3 thinnings, one alternative is selected in the optimization
Thinning delay	The number of 5-year periods by which thinning could be delayed compared to the thinning guide	0–2 periods, one alternative is selected in the optimization
Climate	The climate change scenario that was applied to the forest growth simulations	ECHAM5 A1B (business as usual emissions, climate change)
Fertilization	If forest stands were fertilized to promote growth	No
Regeneration	How forest stands were regenerated after final felling	Plantation with the same species as in the previous generation No tree breeding effects

Table 2 Treatment of riparian buffer zones of different species and ages

RBZ category	RBZ treatment, generation 1	RBZ treatment, generation 2 and later
Spruce, > 50 years old	Clear-cut system	CCF
Spruce, < 50 years old	CCF	CCF
Other stands	Set-aside	Set-aside

in all stands except for those dominated by Norway Spruce. The RBZs of older spruce are known to be highly vulnerable to windfall after the clear-felling of the surrounding stand. For this reason, we decided not to differentiate the management of RBZs in the current spruce stands older than 50 years until the clear-cut. In the second-generation, following an ordinary regeneration, the management system of these RBZs was changed to continuous cover forestry (CCF) in the form of recurring selection fellings. In Heureka, CCF is implemented as selection felling where a percentage of trees is thinned from the stand. To younger spruce RBZs, we applied selection fellings from the start of the simulation period. The parent stands, regardless of species, were treated as in the first alternative.

The management was optimized by solving linear programming problems (Gurobi Optimizer 9.5.2 connected to Heureka PlanWise) at the municipality level based on 20 alternative treatment schedules per stand. The objective of the optimization was to reach the maximum net present value (NPV) while maintaining an even flow of harvest (harvest levels were not allowed to deviate more than 10% from

the mean harvest level over all harvest periods). The optimization model was the same for the alternative with RBZs as without the RBZs.

For further analysis, we extracted the NPV (€ ha⁻¹) and the total harvest volume over 100 years (m³ ha⁻¹, under bark) for each stand. The NPV is generally calculated as the sum of revenues (B) minus costs (C), each discounted at rate r for a near-infinite time horizon (T),

$$NPV = \sum_{t=0}^T \frac{B_t}{(1+r)^t} - \sum_{t=0}^T \frac{C_t}{(1+r)^t} \quad (1)$$

The revenues are generated from the sale of harvested timber at default prices in Heureka representing the recent actual prices in the region. The costs for thinning, harvesting, site preparation, and planting are also the default values in Heureka based on recent actual costs in the region. The revenues and costs were assumed to be fixed for the whole planning period. For a technical explanation with details for both even-aged and uneven-aged forest calculations see the software specifications at: https://www.heurekaslu.se/wiki/Net_present_value. The discount rate r was set to 2% which is a common value in economic analyses of Swedish forestry and is widely accepted by economists for long-term economic investments (Hansson et al. 2016; Drupp et al. 2018). We did a sensitivity analysis for discount rates between 1 and 3% to investigate how far our results were sensitive to the discount rate (Appendix S2). The near-infinite time horizon refers to that the NPV includes the NPV of the cash flow of the simulation period as well as the land expectation value. The land expectation value calculation assumes that

the third-generation of even-aged stands and the final harvest of uneven-aged stands are repeated in perpetuity. Heureka uses the Swedish Crown as currency, we used the 10-year average exchange rate of 9.89 Swedish Crown = 1 Euro from the European Central Bank to convert results to Euros.

Property maps

We overlaid the stand map with two sets of property maps. First, we used the real-world property map since it provides the most realistic distribution of properties in the area which is likely historically adapted to the topography of the landscape (in “Real-world property map” section). However, the topography might also have an influence on the property size (i.e. ease of constructing houses and thus population density is dependent on topography) and thus confound the relationship of property size with the cost distribution of implementing RBZs per sé. Therefore, to improve the generalizability of the results, we additionally simulated a range of fictional property maps, each of which with a distinct average property size and as little variation in property size (in “Simulated property maps” section).

Real-world property map

We overlaid the property map for Hässleholm with the stand map to attribute the forest stands to properties. Nine hundred and thirteen stands (2.5% of stands) did not intersect with a property with a unique identifier and were therefore excluded from this part of the analysis. The property map included a total of 4389 properties. Most properties in the real property map were small. The mean area of forest in a property was 16.2 ha (median = 5 ha, sd = 35 ha), and 95% of properties were smaller than 66.9 ha (Fig. 1). Of all the forest area in the study, 90% was in properties smaller than 200 ha (Fig. 1). We classified the property by forest area (Table 3).

Simulated property maps

We also simulated alternative fictional property maps spanning a wide range of average property sizes that can be found in privately owned forests. We used the Build Balanced Zones tool in ArcGIS Pro 2.9 (ESRI Inc. 2022) to create 49 different property maps with average property sizes ranging from ~25 to ~3800 ha, which covers most of the range of property sizes in Southern Sweden (Appendix S3).

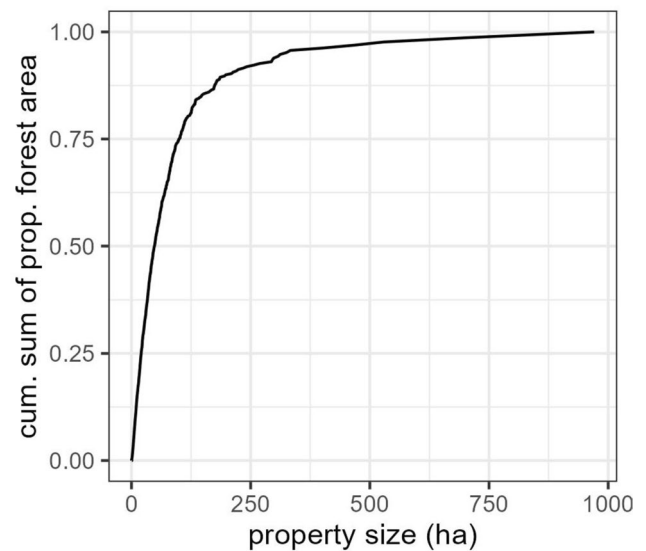


Fig. 1 Cumulative sum of the proportion of forest area in Hässleholm by property size showing that most forest is located in small properties

Table 3 Number of properties per property size class (by forest area) in Hässleholm according to the real property map

Property size class (ha of forest)	Number of properties
0–10	2860
10–25	752
25–50	432
50–75	160
75–100	85
100–200	80
200–500	17
> 500	3

Analysis

We used R version 4.2.2 (R Core Team 2022) to analyse the RBZ impact. For each simulated property or property size class in the real-world property map, we calculated the mean and the standard deviation of the property size (ha), the mean and the standard deviation of the properties’ RBZ proportion of forest land, the mean and the standard deviation of properties’ the difference in harvest volumes between the alternatives’ with and without RBZs (henceforth, harvest loss in $\text{m}^3 \text{ha}^{-1}$), and the mean and the standard deviation of the properties’ difference in NPV between the alternatives with and without RBZs (henceforth, NPV loss in € ha^{-1}). Then, we also calculated the landscape-level NPV loss per ha as the total cost of compensating all forest owners for the implementation of RBZs, as well as the landscape-level cost per hectare of compensating those

owners in the real-world property map who lose more than 5% and 10% of the NPV without RBZs.

Results

At the aggregated level, the NPV per ha with and without RBZs was 8654 and 8805 € ha⁻¹, respectively, and the harvest with and without RBZs was 697 and 714 m³ ha⁻¹. This corresponded to an NPV loss of 1.8% (151 € ha⁻¹) of NPV and 2.4% (17 m³ ha⁻¹) of harvest volume; both figures lower than the 4% of forest area confined to RBZ.

Cost variation and property size

As hypothesized, the variation of NPV and harvest loss was strongly dependent on the property size. The pattern was apparent for both the simulated and the real-world properties (Fig. 2). In the simulated property maps, the average SD of NPV and harvest loss per property over the five ownership simulations with the smallest mean property sizes was 6.6 and 6.1 times higher, respectively, than for the five ownership simulations (“size classes”) with the largest property sizes (Fig. 2; Table 4). In the real-world mixed property size map, the SD of NPV and harvest loss for properties with an area of 25–50 ha was 4.2 and 6.9 times higher than for the properties with an area larger than 500 ha (Fig. 2; Table 4). Note that the difference in average property size between

Table 4 SD of harvest and NPV loss of the smallest and largest properties

	Size class	Mean property size (ha)	SD of NPV loss (€ ha ⁻¹)	SD of harvest loss (m ³ ha ⁻¹)
Simulated	Smallest	25.3	191	20.7
	Largest	2292	29	3.4
Real-world	Smallest	35.3	160	22.2
	Largest	733.1	38	3.2

For the simulated property maps, the smallest property size in the table is calculated as the average of the five maps with the smallest mean property size, and the largest property size is the mean of the five maps with the largest mean property size. For the real-world property maps, the small properties are defined as all properties 25–50 ha and the largest as all > 500 ha

the small and large categories was smaller for the real-world property map than for the simulated property map.

In all cases, the majority of properties lost a small percentage of the NPV and harvest to the implementation of RBZs (Fig. 3). However, in both the simulated and real property maps, there was a much higher variation in the proportion of NPV and harvest that was lost per property in properties smaller than 25 ha and 25–75 ha (Fig. 3). There were only a few properties larger than 75 ha that lost more than 10% of the NPV or harvest, and the losses of properties over 200 ha were very close to the study area average. Surprisingly, there were a few properties on which the harvested volume and the NPV increased in the

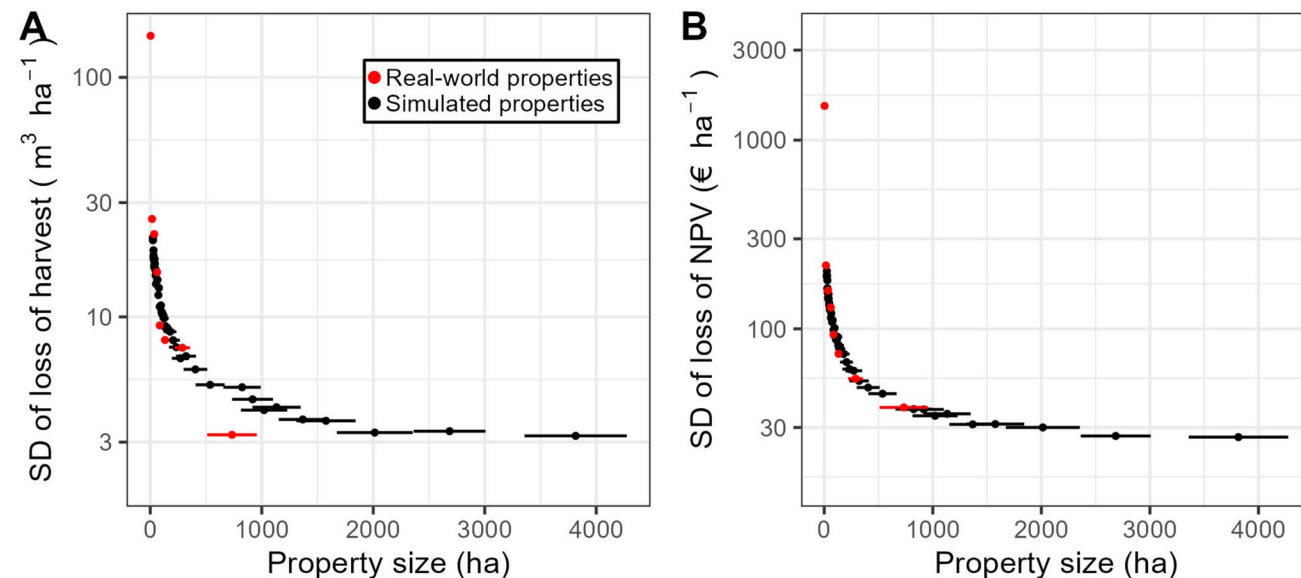


Fig. 2 Standard deviation of harvest loss (panel A) and NPV loss (panel B) over map-mean property size for simulated property maps (black dots), and for size classes of the real-world mixed-size property map (red dots). Horizontal lines indicate are 1 standard deviation

around the mean property size within each simulated property map or each real-world property map size class. The size classes of the real-world properties are as follows: 0–10 ha, 10–25 ha, 25–50 ha, 50–75 ha, 75–100 ha, 100–200 ha, 200–500 ha, and > 500 ha

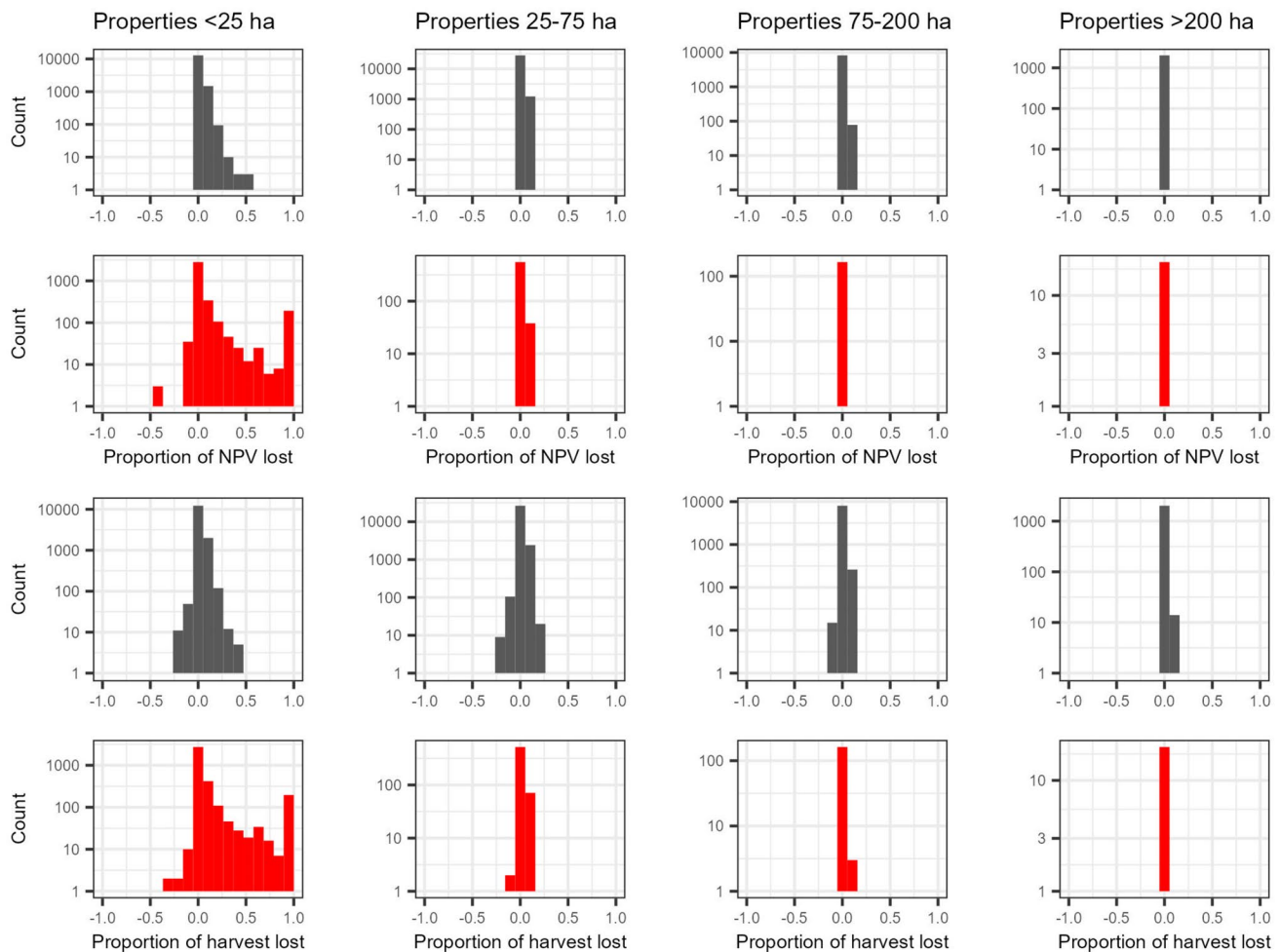


Fig. 3 Proportion of net present value and harvest lost due to the implementation of RBZs per property in the simulated property maps (black) and real property map (red). The Y-axis is log₁₀ transformed. Negative loss indicates an increase in NPV or harvest

RBZ alternative. All those properties were small and thus had limited harvest opportunities in even-aged management in the planning period, which was the management type for all stands in the scenario without RBZs. If the timing of harvesting is different between the two RBZ scenarios, or if CCF is applied to an RBZ in a small property (and thus is harvested more frequently), the amount of harvested timber can increase in some cases and earlier harvesting can increase the NPV due to reduced discounting. On the other hand, if the timing of harvesting changes so that more wood is harvested but at a later moment in the planning period, then the NPV can be lower while the timber harvest increases.

Compensating all forest owners would, in terms of NPV, at the landscape-level cost 151 € ha⁻¹ (1.8% of total NPV, i.e. the average NPV loss per hectare). The landscape-level NPV cost of compensating only those real-world property map owners who lost more than 5% of their NPV was 44 € ha⁻¹ (0.4% of the total NPV) and only compensating

those who lost more than 10% of their NPV cost 21 € ha⁻¹ (0.2% of the total NPV).

Discussion

For the majority of forest owners, the losses due to RBZ implementation would not differ much from the average of the whole municipality. Still, landscape heterogeneity causes the cost distribution to be unequal and some; particularly, small owners could lose 10–30% or even more of the harvest volume or NPV (as compared to 2.4% and 1.8%, respectively, at the municipality level). With increasing property size, the variation in losses due to RBZs strongly diminished: Among the smallest properties, it was 4.2–6.9 times higher than among the largest properties. At the landscape level, the total cost of implementing the RBZs was somewhat lower than the proportion of forest area with RBZs, which differs from previous results of Sonesson et al. (2021)

who found that RBZ implementation cost was generally proportional to the RBZ area. This is explained by the application of CCF on some of the RBZs and by the clear-cutting of existing trees before the system change to CCF, in some cases.

The variation in the cost of implementing RBZs among properties was not linearly related with property size. The cost variation decreased steeply at small scales. For properties larger than ~200 ha, the cost variation levelled off at the level of ~3–5 m³ ha⁻¹ and ~20–40 € ha⁻¹ (see Figs. 2 and 3). In the real property map, we used in this study, 90% of the forest was located in properties smaller than 200 ha. This means that a large number of forest owners in Hässleholm municipality would face relatively far-above-average costs of the implementation of RBZs while others would bear no costs at all. Since 79% of the forest in Southern Sweden is owned by NIPF owners with an average of 28-ha forest, such cost inequality is likely to perpetuate throughout the region. In any landscape with diverse forest ownership, riparian zones, and a policy requirement to protect those riparian zones, the intersection of the property boundaries and the streams will likely result in an unequal distribution of RBZs and related costs.

This potential cost inequality across properties is similar to what has been found for other ecosystem service trade-offs in Nordic forests. Pohjanmies et al. (2017) and Bakx et al. (2023) showed that achieving the same relative increase in carbon stock would, on average, imply higher relative cost for small property owners compared to any owner of several hundred-hectare large property. Pohjanmies et al. (2019) showed a similar pattern for the trade-off between conservation values and timber production. It is important to note, however, that those studies compared the average cost of environmental considerations of landscape-wide (unconstrained) planning alternatives with property-level (constrained) alternatives while we studied the distribution of those costs based on landscape-wide management planning without property-level constraints in management optimization. The importance of watershed-wide planning, i.e. across multiple properties, of forest stream protection has also been highlighted for the effective prevention of nutrient leaching to surface waters (Futter et al. 2010). We thus hypothesize that similar results can be expected from landscape-level planning of other ecosystem services if their provision potential is heterogeneously distributed across the landscape. If this holds, the financial inequalities between owners related to implementing multiple environmental consideration measures will accumulate.

Limitations

The results of this study are dependent on the spatial distribution of streams in the landscape and thus, considering

our methodology for defining streams and the topography of the landscape. The landscape that we used is however representative of the broader geographical area. In any situation where the distribution of streams is heterogeneous and the distribution of properties does not reflect this heterogeneity, similar results should be observable. Still, in different topographical contexts, the distribution of streams and therefore the cost inequality and scale at which it disappears will likely be different. Furthermore, the distribution of streams and wet soils is also dependent on the soil type, peat content of the soil, and seasonal hydrological dynamics (Ågren et al. 2015, 2021). Additionally, anthropogenic alterations to the landscape such as ditching have also led to a non-natural distribution of water channels in Swedish forests as the majority of channels have in fact been found to be ditches (Paul et al. 2023; Lidberg et al. 2023). This would also affect the distribution of streams worth protecting in real forest landscapes considering that landscape restoration might involve closing ditches.

The climate modelling in Heureka has several limitations that affect the absolute estimates of forest growth. Importantly, there is no inclusion of risks and natural disturbances and their occurrence frequency in relation to climate change in Heureka simulations. Several risks of disturbance, such as fire, drought mortality, pest outbreaks, or windfall, are predicted to increase with climate change in boreal forests (Price et al. 2013; Seidl et al. 2014). These disturbances are however challenging to model due to the stochastic nature of extreme events (Seidl et al. 2011). Overall, we can expect that we overestimated forest growth in our study. If this overestimation differs between RBZs and other forests, it could affect the magnitude of the property size-cost inequality relationship. Future studies and developments of forest management models should include water availability and natural disturbances to minimize such uncertainties.

In terms of management options, CCF in RBZs could reduce the cost of implementation while maintaining some ecosystem functioning compared to clear-felling, especially in wide buffers (Elliott and Vose 2016; Oldén et al. 2019a, 2019b; Sonesson et al. 2021). However, in this study, we did not apply CCF for any other than spruce-dominated RBZs, and those studies highlighted the need for larger RBZs if CCF was practised in the RBZ. The total area of hydrologically adapted RBZ in our study was linked to the area of recommended 12.5-m-fixed-width buffer zones. Studies have shown that to promote biodiversity and for proper ecosystem functioning in the riparian zone, much larger RBZs are needed to protect streams and riparian zones from the negative effects of forestry activities (Oldén et al. 2019a, 2019b; Jyväsjärvi et al. 2020). Wider RBZs, even when implemented with variable widths, will increase the costs. This means that we can expect the cost inequality to be

even more important because total variation will increase throughout the landscape.

Policy implications

International and national policy goals, such as the CBD Kunming-Montreal Global Biodiversity Framework, the EU Water Framework Directive, and the Swedish Environmental Objectives, stress the need for protecting water bodies in forest landscapes. Previous studies have shown that while RBZ implementation has increased there still is an implementation gap and thus water protection needs to improve to reach environmental goals in Swedish forest landscapes (Angelstam et al. 2011; Maher Hasselquist et al. 2020). Economic factors such as increased harvesting costs due to RBZ implementation or a loss of revenue from unharvested trees could represent one explanation for the lack of uptake by forest managers. For an improvement in effectiveness, it is important to stimulate adoption. From the literature, we know that accounting for unequal cost distributions in forest landscapes with a large share of small forest owners increases perceived fairness and thus effectivity (Loft et al. 2020). Perceived unfairness can reduce the willingness of affected people to implement environmental policies, especially if no compensation is provided (Clayton 2018; Maestre-Andrés et al. 2019). In this study, we show that the cost distribution of implementing RBZs in Southern Sweden would be highly unequal. The insights from the literature imply that addressing such inequality in the design of potential water protection policy would most likely improve acceptance, thus adoption rates and thereby effectivity.

Potential solutions to address fairness could be implemented in both private initiatives such as certification schemes (e.g. FSC and PEFC) and public policy instruments (e.g. regulation, landscape planning, or market corrections). In Sweden, certification is currently the most important driver of RBZ implementation (Ring et al. 2017) and strengthening RBZ requirements in certification could thus be the most straightforward way of increasing the uptake of RBZs by forest owners. However, this would not solve the problem of unequal cost distribution. Alternatively, the incorporation of redistribution mechanisms in landscape planning has been suggested to solve unequal cost distributions for ecosystem service production (Michanek et al. 2018). Specifically, a fee-fund system where losses are redistributed between forest owners in a landscape could mitigate inequality (Bostedt et al. 2021). This fee-fund system could in Sweden be implemented through forest owners associations which today already similarly redistribute costs for forest certification (Lidestav and Berg Lejon 2011). A second public policy instrument could be to introduce payments for ecosystem services to compensate owners directly for the costs of setting aside RBZs. This could be designed

similarly to the Finnish METSO programme, in which landowners are paid for voluntary nature conservation of forests with desirable structural characteristics, specifically targeting watershed protection (Matthies et al. 2015; Kangas and Ollikainen 2022). In this case, the compensation would be publicly arranged. The cost of such a compensation scheme would be highest if all owners were compensated but can be drastically decreased by only compensating those forest owners who bear most of the cost of RBZ implementation in the landscape. Because of the proportionality of costs to RBZ size, action-based payments would be a straightforward way to compensate owners for their losses. However, result or model-based payments have been suggested for agricultural policies and could provide an additional improvement over pure action-based payments (Bartkowski et al. 2021). Overall, stimulating the uptake of environmental protection measures through policy is complex, and more research is needed to evaluate the potential effects of minimizing unequal cost distribution on the uptake of RBZs.

Conclusions

To summarize, we compute the average costs of setting aside proposed RBZs at 1.8% of the total NPV and 2.4% of the harvest volume. These costs are relatively low compared to the set-aside area at the landscape level. At the property level, however, the variation in the cost for setting aside RBZs depends on property size. We estimate the variation of costs in small properties of ~25–35 ha to be 4.2–6.9 the cost variation in larger properties of ~700–2000 ha. This relationship between property size and cost distribution inequality was decreasing non-linearly and levelled off around ~200 ha. Given that the majority of private forest properties in Europe are smaller than that, the potential cost of implementing RBZs would be unevenly distributed among most forest owners. Considering the benefits of RBZs and the current implementation gap, addressing such unevenly distributed costs would in all likelihood increase acceptability and thus effectiveness of watershed protection in forests. This could happen through either public or private compensation or cost redistribution schemes. The cost of such schemes would depend on the politically accepted level of distributional inequality among forest owners.

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Availability of data and code The Heureka input data, simulation and optimization settings, the Heureka results, and the R code for the analysis and visualization will be made available in a public repository (<https://doi.org/10.5281/zenodo.10404253>). The python code for the topographical mapping of streams and the calculation of the depth-to-water index is available at: https://github.com/williamlidberg/Mapping-Mountainous-Arctic-Lakes-From-the-Air/blob/main/Hydrological_processing.py. We used forest data from the SLU forest map 2010 (available here: <https://www.slu.se/en/Collaborative-Centres-and-Projects/the-swedish-national-forest-inventory/foreststatistics/slu-forest-map/>) and the Swedish National Forest Inventory (only available upon request with the National Forest Inventory). Heureka is publicly available free software (<https://www.slu.se/SHA>). All documentation for Heureka can be found on the Heureka wiki (https://www.heureka.slu.se/wiki/Heureka_Wiki) and help pages (<https://www.heureka.slu.se/help/index.html?introduktion.html>). The terrain model that we used for the topographical mapping of streams is publicly available (<https://www.lantmateriet.se/sv/geodata/vara-produkter/produktlista/markhojdmmodell-nedladdning-grid-1/>).

Declarations

Competing interests The authors declare no competing interests.

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