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Comparing Conservation Value Maps and Mapping Methods in a Rural Landscape in Southern Finland

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Abstract

We tested to what extent conservation value maps are different if the valuation and mapping method is changed. We compared 66 different conservation value and 4 different ecosystem service maps. Using remote sensing and other georeferenced data, we produced 2 different habitat type maps, which were 50 % similar. We valued each mapped habitat type based on rarity corrected potential number of vascular plant species and naturalness using 6 different valuation alternatives. We mapped habitat type connectivity and complementarity using 2 main approaches. The habitat type valuation alternatives were quite similar, but if the habitat type naturalness was taken into account, differences were larger (correlations between maps 0.38–1.00). Different connectivity and complementarity calculations yielded different results, variation between different approaches being larger (correlations -0.15–0.44) than inside an approach (correlations 0.31–0.60). Conservation value maps were very different from ecosystem service maps (carbon storage, timber production potential, landscape value for recreation) (correlations -0.29–0.47). We show that valuation and mapping approach has a large effect on the conservation value map and the correlation between ecosystem service and conservation value maps depends on the used mapping approach. As different mapping approaches provide different maps, maps should be used with care.

Keywords:
ecosystem services, GIS, habitat type, naturalness, species richness

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1 Introduction

In many planning processes, such as land-use planning or environmental impact assessment, evaluation of the natural environment has been made in order to prioritize areas for development, forestry purposes or conservation (Margules & Usher, 1981; Smith & Theberge, 1986; Spellerberg, 1992). The value that is given to the natural environment in a conservation perspective can be defined as the conservation value. The overall conservation value of an entity is difficult to assess, since conservation value consists of many different parts. There are many different criteria for assessing the conservation value or different conservation values of an area. These criteria include but are not limited to biodiversity, naturalness (Angermeier, 2000) and ecosystem services (Daily, 1997). Selecting the criteria is crucial, since they define the actions to be performed in the conservation or in the land-use planning (e.g. Angermeier, 2000).

For different conservation values, there exist a plethora of different mapping alternatives. Biodiversity or species distributions can be mapped either directly or indirectly using remotely sensed data. Direct mapping means the identification of single species, assemblages or habitat types from the remotely sensed information; while in indirect mapping, environmental variables are used as surrogates (Nagendra, 2001; Turner et al., 2003). Naturalness has often been mapped using basic methods; for instance, by assessing the naturalness of vegetation by interpreting aerial imagery (Machado, 2004; Villarroya & Puig, 2012). Ecosystem services have been quantified by using direct observations, proxy data or process models (Egoh et al., 2012, Maes et al., 2012). The focus has often been in doing biophysically or monetarily accurate predictions (e.g. Troy & Wilson, 2006; Nelson et al., 2009) or in making spatial comparisons or prioritizations of the studied areas (e.g. Raudsepp-Hearne et al., 2010; Burkhard et al., 2012).

In many of the mapping approaches, surrogates or proxies are used. A widely used proxy data is a thematic map such as land cover or habitat type map. These are widely used e.g. in mapping ecosystem services (Egoh et al., 2012; Maes et al., 2012) or biodiversity patterns (e.g. Kerr & Ostrovsky, 2003; Turner et al., 2003). In this paper, habitat type is defined as a mappable land unit in which vegetation and environmental factors are fairly homogenous. Habitat types do not always equate to direct land cover classes, environmental factors need to be accounted as well. Thus, a specific approach is needed for habitat type classification (McDermid et al., 2005; Lucas et al., 2011). However, it has been argued that habitat type or other thematic maps are often used uncritically; for instance, their accuracy is not assessed (Newton et al., 2009).

When habitat types are mapped, they can be valued e.g. based on potential number of species and their rarity (Rossi & Kuitunen, 1996), the potential supply of ecosystem services (e.g. Troy & Wilson, 2006; Burkhard et al., 2012) or the ecological condition of the habitat type (Vanden Borre et al., 2011; Spanhove et al., 2012). In addition to the valuation, other factors can be included in the mapping tasks. These include e.g. connectivity and complementarity of sites, which are taken into account in spatial conservation prioritization approaches (Margules & Pressey, 2000; Moilanen et al., 2005; 2009b). Although spatial conservation prioritization is usually performed with species occurrence data, also habitat type maps have been used as input data (e.g. Lehtomäki et al., 2009; Arponen et al., 2012). Connectivity and other spatial composition and configuration calculations of thematic maps can also be computed with landscape metrics, which have been used in conservation value mapping (e.g. Schindler et al., 2008; Sundell-Turner & Rodewald, 2008).

Our objectives were 1) to test to what extent the conservation value maps are different, if different habitat type valuation alternatives, habitat type mapping alternatives, or connectivity and complementarity mapping alternatives are used; and 2) to compare these output conservation value maps to the maps of selected ecosystem services: timber production potential, carbon storage, and

recreational value. Our overall aim was to illustrate the extent of difference in conservation value maps when different decisions are made. These decisions include a priori decisions about conservation criteria and more detailed decisions about mapping methods. We discuss how, when and why maps are different and if the different mapping approaches are comparable. We acknowledged that the approaches we compared can yield very different maps but our goal was to demonstrate that several decisions need to be made in the mapping and all these decisions have effect on the final maps. These decisions include what criteria are used for mapping conservation values, by what methods these criteria are mapped and what input data is used. In our case, criteria included species richness and rarity together with naturalness, which were considered as being parts of conservation value, as well as ecosystem services. In mapping conservation values, several methods and input data options were compared.

2 Materials and methods

2.1 Study area

We studied a rural area with a surface of 390 km^2 from southern Finland (Kuitunen, 2013) located in the southern boreal vegetation zone (Ahti et al., 1968). The geographic coordinates (WGS84) of the site are 61° 16´–61° 30´ N and 24° 26´–24° 55´ E (Figure 1). The main land cover and land use types in the area are both coniferous forest (Scots pine, *Pinus sylvestris* and Norway spruce, *Picea abies*) and broadleaved trees (mainly Birch, *Betula spp.* and Aspen, *Populus tremula*) forest habitats followed by lakes, agricultural areas, and peatlands which are partly forest land or poorly productive forest land. Most of the forest area is used for timber production with rotation based forestry including regeneration cutting with either artificial or natural regeneration. Most of the peatlands are drained for forestry purposes.

2.2 Overview of the methods

In Figure 2, a flowchart of the used methods is presented. Overall, we used different methods and different datasets for mapping conservation values and ecosystem services. Our methodology is divided into three steps. In each step, we used different methodological alternatives to test what extent maps are different when different choices are made. First, two different habitat type classifications were used, namely classifications based on 1) object-based image analysis methodology and 2) thematic GIS-datasets. Second, each habitat type was valued based on the rarity corrected potential number of vascular plant species and habitat type naturalness. Overall, six different valuation alternatives were generated. Third, after valuation, habitat type connectivity and complementarity were computed using two different main methods: 1) landscape metrics and 2) spatial conservation prioritization. In total, 66 different conservation value maps were constructed. These maps were then compared with maps of selected ecosystem services: timber potential, carbon storage, and landscape value for recreation. In the remaining methods section, we summarize the used methods. More detailed explanation of used datasets and methods is given in the Appendix.

2.3 Classification of habitat types

We classified different habitat types using two different classification workflow alternatives modifying the approach given in Räsänen et al. (2014). We did not want to test *per se* which one of the methods is better but to test what kind of effect habitat type maps, which are based on different datasets, have on final conservation value maps.

In the alternative 1 (OBIA), we mapped different forest habitat types using a combination of eCoginition Developer 8.8 (Trimble, Sunnyvale, CA, USA) Multiresolution Segmentation and supervised random forest classification (Breiman, 2001). Segmentation input data consisted of color bands of aerial imagery and layers derived from airborne

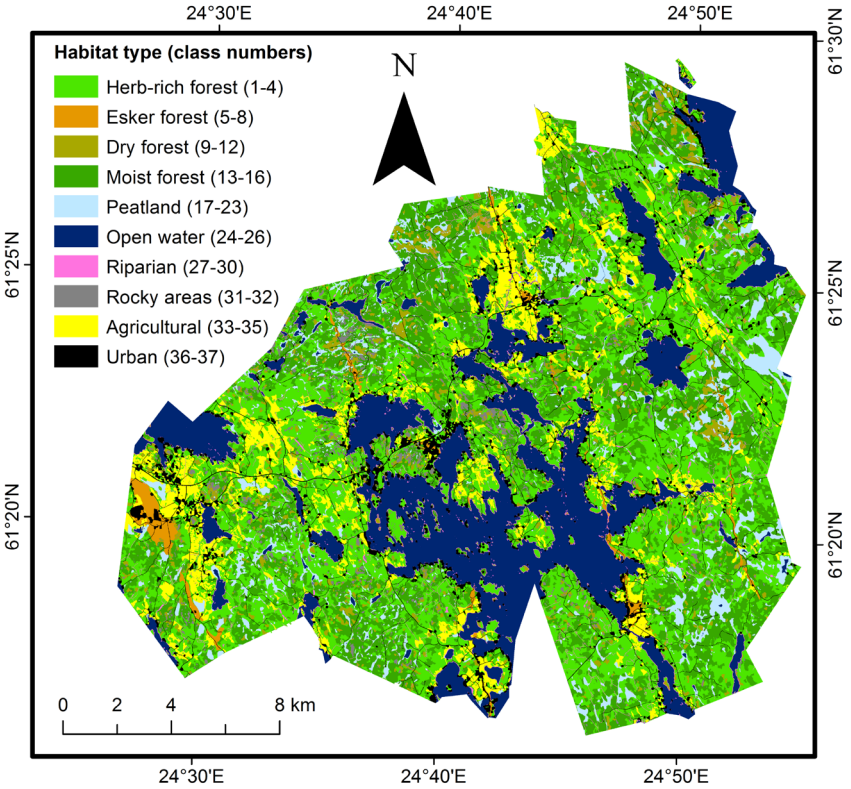

Figure 1: A simplified habitat type classification of the study area located in southern Finland. Habitat types are classified using classification alternative 1. Numbers in parentheses refer to habitat type class numbers that are explained in Table A1.

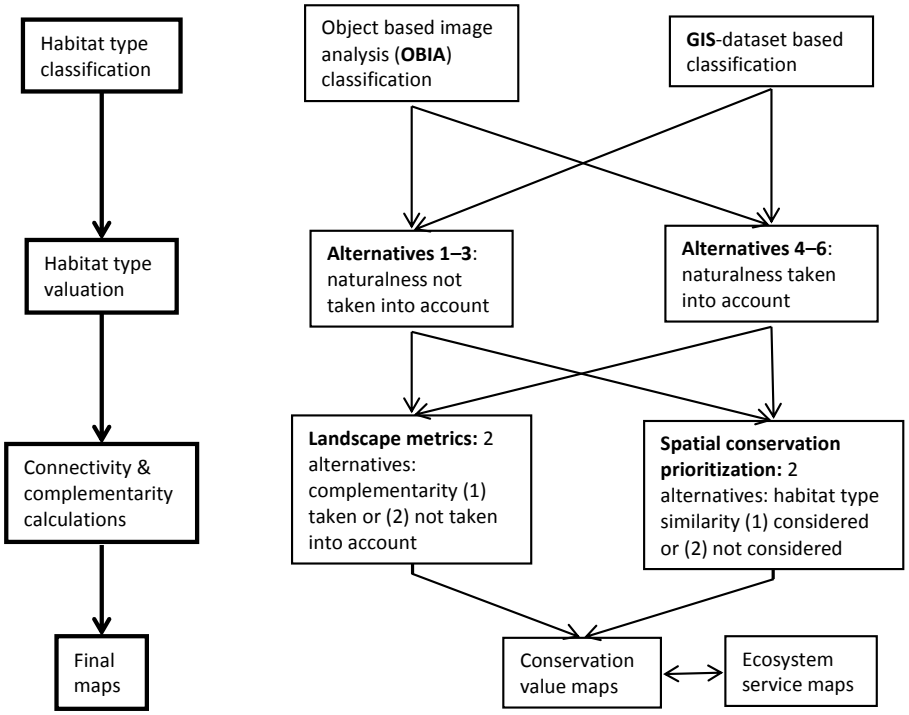

Figure 2: A flow chart of the used methods. Different stages are represented on the left. On the right, different alternatives in each stage are elaborated. First, we classified habitat types using two different alternatives. Second, we valued habitat types using six different alternatives divided into two groups. Third, we calculated connectivity and complementarity using two different approaches which both had two alternatives. Finally, we compared conservation value maps to maps of ecosystem services

laser scanner data. In the classification part, we calculated 122 different features from the layers, and used forestry planning polygons as training data. Other habitat types than forests were derived from various thematic GIS-datasets.

In the alternative 2 (GIS), we derived forest habitat types and part of the other habitat types from Finnish Multisource National Forest Inventory data (MS-NFI, Tomppo et al., 2013). The rest of the habitat types were derived from different thematic GIS-datasets as in the OBIA-classification. Both classifications were converted into 10 m resolution raster dataset for interoperability reasons.

We assessed the classification accuracy of OBIA and GIS-classifications with simple pixel-based cross-tabulation matrices using forestry planning polygons as the reference data. All area that was mapped as forests in the classification as well as in the reference was used in the classification accuracy calculation. For the OBIA-classification, an out of bag (OOB) error rate of the random forest classifier was also calculated on a segment level.

2.4 Valuation of habitat types

We valued different habitat types based on potential number of species and their rarity using six different methods. We compared alternative methods, since there are many ways how habitat types and their species composition can be valued. We wanted to test, if the valuation of habitat types has an effect on the resulting maps.

In all six methods, we used a database by Rossi and Kuitunen (1996), in which all established vascular plant species in Finland are given primary, secondary, tertiary, and quaternary habitat type preferences based on the best available literature (Hämet-Ahti et al., 1986). With established species we mean that the recent exotic species, cultivated species, or escapes were not listed or used. The objective of the mapping based on this database is to show the potential species composition of different areas. Produced maps can be used to show the locations of potentially high-value areas, which can then be checked with field work.

In the first method, the value of habitat types was the number of species that can potentially exist in the habitat type. In the second method, we gave more weight to species' primary habitats and weighted different species using range size rarity calculations (Williams et al., 1996). In the third method, we weighted different species based on their red list status. We further used a fourth, fifth, and sixth habitat type valuation method in which we corrected the habitat type values of valuations 1–3 based on their naturalness and ecological condition, where the habitat type values were reduced if they were not considered being natural or if it was considered that they were not in a good condition.

2.5 Habitat type connectivity and complementarity calculations

After attaining the values for the specific habitat types explained above, we mapped the value for the different areas using two different methods which utilized widely used software packages. First, we calculated landscape metrics per patch and valuated patches based on these metric values. Second, we used spatial conservation prioritization software for connectivity and complementarity calculations

In the landscape metrics (FRAGSTATS 4.1, McGarigal & Ene, 2012) approach, we calculated two different metrics for each patch: patch area and similarity index, which is a measure of the neighborhood similarity of the patch. Largest values were given to large patches with similar neighborhood. Similarities between habitat types were assessed by their species composition. We used two different alternatives: we either 1) took or 2) did not take habitat type complementarity into account. By complementarity we mean that we gave the overall highest values to patches that had the highest values of each respective habitat type. If complementarity was not taken into account, the overall highest values were given freely so that one habitat type could possibly have many of the most valuable patches. Furthermore, we performed additional complementarity analyses in which we modified habitat type valuations 4–6 so

that human habitat type patches were given the lowest values.

In the spatial conservation prioritization approach (Zonation 3.1, Moilanen et al., 2005, 2009a, 2012), raster cells instead of patches were valued. Cells were valued based on their habitat type values, and a connectivity measure similar to the similarity index in which similarity is measured in terms of species composition. In all spatial conservation prioritization calculations, we took connectivity into account; but we either 1) took or 2) did not take similarities of habitat types into account. If similarities between habitat types were taken into account, high value habitat types, (i.e. in our case habitat types with many (rare) species) were prioritized if they had similar species composition as lower valued habitat types. Finally, in additional calculations we modified valuation alternatives 4–6 so that human habitats were given the value of 0.

The major difference between landscape metrics and spatial conservation prioritization approaches is that the former one is dependent on patches whereas the latter one is not. While patches are simple and intuitive to human cognition, they take continuity poorly into account, although many ecological attributes have a continuous nature (McGarigal et al., 2009). Used spatial conservation prioritization approach favors contiguous areas which have similar neighborhoods; thus, it concentrates high value and low value areas to specific locations. We further wanted to evaluate the differences between these approaches and illustrate, how large discrepancies between maps can exist.

2.6 Mapping ecosystem services

We mapped three different ecosystem services from different ecosystem service main types, i.e., timber from provisioning services, carbon storage from regulating services, and landscape value of recreation from cultural services. We selected these three services, since these services represent different types of ecosystem services that landscapes can provide and since the services could be calculated easily from the datasets we had. We calculated timber and carbon storage using MS-NFI data. In recreational value calculation, we gave the highest values to those sites that have natural vegetation and are visible to the most important recreation routes. In this calculation, we used Viewshed analysis in ArcGIS 10.1 (Esri, Redlands, CA, USA) and 10 m digital elevation model derived from airborne laser scanner data.

2.7 Map quantization and comparison

Overall, 70 different maps were produced (Table 1). Comparisons were made between maps 1) that were based on different habitat type maps, 2) in which different habitat type valuation alternatives were used, 3) in which different complementarity and connectivity calculation alternatives were used, and 4) of conservation values and ecosystem services. For comparison purposes, maps were quantized to 20 quantiles. Exceptions were the maps of recreation value which were quantized to 16 quantiles because their distribution could not be divided into 20 quantiles. Additionally, in timber and carbon maps, areas that were not forests or peatland were given a value of 0. They were given the value of 0, since in the MS-NFI data, there were no information about the carbon stock or timber volume of these areas. After quantization, Pearson correlation coefficients between map pairs were calculated. Trade-off analyses were also performed, since timber volume is different type of an ecosystem service than the other considered ecosystem services and conservation values. In other words, in other ecosystem services and conservation values, the goal is to preserve; whereas in timber, the goal is to cut trees. Hence, in trade-off analyses, the map of timber volume was combined with conservation value or ecosystem service maps to find places with a high value for timber and a low value for conservation.

Table 1: Produced conservation value and ecosystem service maps that were used in the analysis.

Connectivity and complementarity measure / ecosystem service	Classification	Habitat type valuation	No. of maps
FRAGSTATS without complementarity	2 alternatives	6 different alternatives	12
FRAGSTATS with complementarity	2 alternatives	6 different alternatives	12
FRAGSTATS with complementarity, lowest value to human habitats	2 alternatives	alternatives 4 to 6	6
Zonation with connectivity	2 alternatives	6 different alternatives	12
Zonation with connectivity and zero value to human habitats	2 alternatives	alternatives 4 to 6	6
Zonation with connectivity and similarity	2 alternatives	6 different alternatives	12
Zonation with connectivity, similarity, and zero value to human habitats	2 alternatives	alternatives 4 to 6	6
Timber volume	-	-	1
Carbon storage	-	-	1

3 Results

3.1 Habitat type classification

The pixel-based classification accuracy of the habitat type classification alternative 1 (OBIA) was 47 % in the areas classified as forest habitat types (classes 1 to 12). A rather similar result was given by random forest OOB error rate which was 49 % and was calculated per object. The classification accuracy of the alternative 2 (GIS) was 23 % in areas classified as forest habitat types. Overall, the alternatives 1 and 2 were 50 % similar in their classifications. In regards to areas which were mapped as forests in both of the classifications, similarity was 22 %. When misclassifications between different forest type successional stages were not considered as errors, classification accuracies were higher: 60 % and 47 % with alternatives 1 and 2 respectively. Finally, the similarity between the alternatives 1 and 2 was 68 % and 56 % considering all data and forests respectively, when the misclassifications between successional stages were not considered as errors.

3.2 Habitat type valuation

The habitat type valuations could be divided into two distinct groups when they were compared. The first group included the valuations 1–3, i.e. without naturalness corrections, and the second group the valuations 4–6, i.e. with naturalness corrections. The differences inside these groups were rather small but the differences between the groups larger (Table 2). In the valuations 1–3, human habitats got generally quite large values but their values were among the lowest in the valuations 4–6 due to their naturalness correction (Table A2). Altogether, when averaged over all valuations, class 28 (riparian habitats) got the highest value, followed by classes 31 (non-calcareous rocky areas), 4 (mature herb-rich forest), and 33 (dry meadows). When only valuations 1–3 were considered, classes 33, 37 (industrial and urban areas), and 35 (cultivated areas) got the highest values. For the valuations 4–6, classes 28, 31, and 4 got the highest values.

Table 2: Pearson correlation coefficients between different different habitat type valuation alternatives

	1	2	3	4	5	6
1	1.00	0.82	0.70	0.50	0.43	0.25
2	0.82	1.00	0.65	0.22	0.43	0.12
3	0.70	0.65	1.00	0.48	0.55	0.58
4	0.50	0.22	0.48	1.00	0.82	0.81
5	0.43	0.43	0.55	0.82	1.00	0.80
6	0.25	0.12	0.58	0.81	0.80	1.00

3.3 Conservation value mapping

When conservation value was mapped using FRAGSTATS metrics without complementarity considerations, the highest values were given to large mature herb-rich forest patches. The location of the most valuable patches, however, depended on the used classification alternative. Moreover, when naturalness correction was not used, human habitat patches got high values. When complementarity calculations or Zonation were used, also other habitat types got large values. The location of the most valuable spots differed due to the classification alternatives and whether FRAGSTATS or Zonation was used. One of the major differences between FRAGSTATS and Zonation mapping was that in FRAGSTATS mapping values were given to habitat type patches (Figures 3a and 3b). In Zonation, a patch did not get equal value because even rather small homogeneous forest patches are divided into several rather small cells (pixels), and individual cells are valuated. Therefore, the high-value areas were usually divided between parts of the patches or inside a patch.

Generally, the differences between conservation value maps were low inside a habitat type valuation group if the same classification alternative and the same connectivity and complementarity calculations were used. Differences were a bit larger between the habitat type valuation groups (Figures 3d and 3f), or if the classification alternatives were compared (Figures 3d and 3e). Differences were notably larger, if different connectivity and complementarity calculations were used (Table 3, Figures 3a–3d). Despite of these general trends, there were some variations inside the different mapping alternatives.

In more detail, in the FRAGSTATS based maps without complementarity, differences inside valuation alternative groups 1–3 and 4–6 were small if the same classification alternative was used. Differences were larger between the valuation alternative groups and between the classification alternatives. In the complementarity based FRAGSTATS calculations, the results differed very little between the different valuation alternatives overall. In other words, there were no large differences between the habitat type valuation groups. There were, however, larger differences between the different classification alternatives and if human habitats were valued or not. Finally, the correlations between FRAGSTATS calculations with and without complementarity were intermediate when the same classification and valuation alternatives were used (Figures 3a and 3b).

In the Zonation calculations when similarity matrix was used for connectivity only, differences between the calculations based on the valuations 1–3 and 4–6 were minor when the same habitat type classification alternative was used. Differences between the classification alternatives (Figures 3d and 3e), and between the habitat type valuations 1–3 and 4–6 were larger (Figures 3d and 3f). In the Zonation calculations with connectivity and similarity, similar trends could be observed. There were, though, a bit more variation in regard to the valuation alternative. The correlations between the connectivity-only and the connectivity-and-similarity calculations were intermediate, when the same classification and valuation were used (Figures 3c and 3d).

The correlations between Zonation and FRAGSTATS outputs were low. The correlations between FRAGSTATS with complementarity and Zonation with connectivity and similarity were close to zero and often even negative (Figures 3a and 3c). In other comparison pairs, the correlations were positive but low. When human habitats were not valued in Zonation or FRAGSTATS, the correlations were also low being a bit higher when only connectivity was included in the Zonation calculations (Figures 3a–3d).

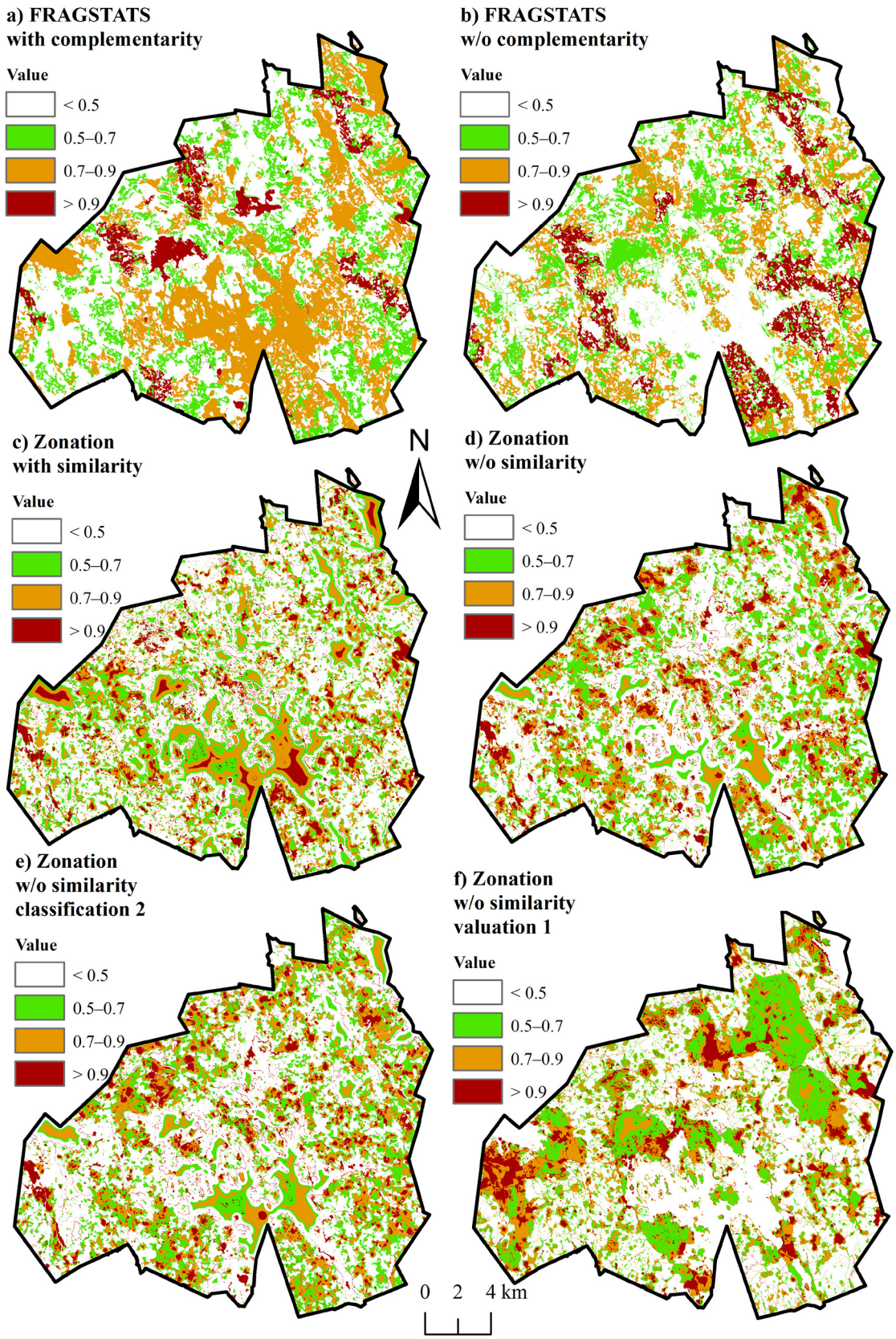

Figure 3: Different conservation value maps of the study area. Proportional value of the landscape is drawn. Areas in most valuable decile are drawn in black. Habitat type valuation alternative 6 (see Table A2) is used in a–e and alternative 1 in f. In a and c–e human habitats are given the lowest value. Classification alternative 1 (see Table A1) is used in a–d and f and alternative 2 in e.

3.4 Ecosystem service mapping

Of the ecosystem services, recreational value did not strongly correlate with the other services (Table 3, Figures 4a–4c). The different recreational value maps; i.e., based on the different classification alternatives, were almost similar. Timber production potential and carbon storage maps were rather similar with some exceptions. Carbon storage had the greatest values in (open) mires and in older forests. Timber production, instead, had largest values in older forests (Figures 4a and 4b).

The correlations of ecosystem services with all conservation value alternatives were usually low and partly negative possibly pointing out to the fact that hot spots of three ecosystem services were different from conservation value hot spots (Table 3, Figures 3 and 4). In general, the conservation value maps, where human habitats were not given value, were most similar to the ecosystem service maps. Vice versa, the conservation value maps, where human habitats were valued as other habitat types, were most different from the ecosystem service maps. The FRAGSTATS metrics based maps without complementarity (Figure 3b) were an exception in this general trend and were more similar to the ecosystem service maps than many other conservation value maps.

In more detail, all ecosystem services had the highest correlations with the conservation value maps calculated with Zonation with connectivity only and no value given to human habitats (Figure 3d and 3e). Also the FRAGSTATS maps without complementarity (Figure 3b) had almost similarly high correlations especially with recreation and timber. Timber and carbon had the strongest negative correlations with the conservation value maps calculated with FRAGSTATS with complementarity followed by the Zonation maps with connectivity and similarity. In addition to these, recreation differed as much also from the Zonation maps with connectivity only (Figure 3f). In these conservation value maps, human habitats were treated as other habitat types.

In the trade-off analysis, to find areas with high timber value and low value for biodiversity and ecosystem services conservation, results differed according to the selected conservation value map. There were some areas that had high value for timber and low value for conservation (Figure 4d). Hence, it could be argued that logging and conservation could be located in different sub-areas of the whole study area. Nevertheless, the location of the sub-areas may vary; and in the decisions about logging and conservation, several location-bound and also other factors should be taken into account.

Table 3: Correlations between different conservation value and ecosystem service maps. Abbreviation F refers to FRAGSTATS based maps and Z to Zonation based maps. Valuation refers to habitat type valuation alternatives. 'Inside group' means that correlations between conservation value maps in which valuation alternatives were compared inside groups; groups are valuation alternatives 1–3 and 4–6. In the 'groups compared' comparisons valuations of group 1–3 were compared to valuations of group 4–6. 'Human or not' refers to correlations between valuations where human habitats were either given or not given value using same valuation alternative (alternative 1–6). Classification refers to two habitat type classification alternatives (OBIA and GIS).

Maps compared				Correlation	
Comparison 1	Comparison 2	Valuation	Classification	Lower	Upper
F w/o complementarity	F w/o complementarity	inside group	same in both	0.93	0.97
F w/o complementarity	F w/o complementarity	groups compared	same in both	0.64	0.91
F w/o complementarity	F w/o complementarity	same in both	different	0.57	0.64

F with complementarity	F with complementarity	inside group	same in both	1.00	1.00
F with complementarity	F with complementarity	groups compared	same in both	0.96	0.98
F with complementarity	F with complementarity	same in both	different	0.54	0.57
F with complementarity	F with complementarity	human or not	same in both	0.53	0.70
F w/o complementarity	F with complementarity	same in both	same in both	0.31	0.48
Z w/o similarity	Z w/o similarity	inside group	same in both	0.79	0.96
Z w/o similarity	Z w/o similarity	groups compared	same in both	0.45	0.75
Z w/o similarity	Z w/o similarity	same in both	different	0.58	0.77
Z w/o similarity	Z w/o similarity	human or not	same in both	0.49	0.78
Z with similarity	Z with similarity	inside group	same in both	0.85	0.92
Z with similarity	Z with similarity	groups compared	same in both	0.38	0.64
Z with similarity	Z with similarity	same in both	different	0.58	0.72
Z with similarity	Z with similarity	human or not	same in both	0.99	1.00
Z w/o similarity	Z with similarity	same in both	same in both	0.44	0.60
F w/o complementarity	Z w/o similarity	same in both	same in both	0.10	0.28
F w/o complementarity	Z with similarity	same in both	same in both	0.17	0.37
F with complementarity	Z w/o similarity	same in both	same in both	0.12	0.34
F with complementarity	Z with similarity	same in both	same in both	-0.15	0.03
F, no value to human habitat	Z, no value to human habitat	same in both	same in both	0.16	0.44
Recreational value	Other services	-	-	0.25	0.36
Recreational value	Recreational value	-	different	0.96	0.96
Timber	Carbon	-	-	0.84	0.84
Recreational value	All conservation value maps	-	-	-0.08	0.33
Timber	All conservation value maps	-	-	-0.24	0.47
Carbon	All conservation value maps	-	-	-0.29	0.46

Correlation color codes	< 0.1	0.1 to 0.29	0.3 to 0.49	0.5 to 0.69	0.7 to 0.89	> 0.89

4 Discussion

Overall, our results show that conservation value maps can be very different when different decisions are made in the mapping. The extent of difference, however, varies as illustrated in Table 3. In this section, we discuss our results, as well as the strengths and weaknesses of our valuation and mapping methods.

4.1 Differences in habitat type classification and valuation

Our classification accuracies were low. The major reason for this is noisy training data. Forestry planning datasets are not principally intended to be used in this kind of tasks, and information about habitat types and boundaries might not always be accurate in them. Another reason is that we assessed the accuracy only in mapping forest habitat types, while other habitat types were derived straight from existing land use datasets. Forest habitat types can be difficult to interpret also by skilled professionals in the forest and their differences in remotely sensed data are often small. For instance, in the MS-NFI classification accuracy of the site fertility class (forest types) is 50 % and of the site main class (mineral soil and main peatland types) 84 %. In the former, most differences are not more than one class (Tomppo et al., 2009; Multi-Source National Forest Inventory of Finland, 2013).

Overall, the habitat type maps and the conservation value maps were different when different habitat type classifications were used. The correlations between conservation value maps where different habitat type classifications were used varied between 0.54 and 0.72, although habitat type maps had a similarity of 0.50. The reason, why correlations between conservation value maps were larger than correlations between habitat type maps, can be the fact that almost all of the differences between habitat type maps were in forest areas. Forest areas are quite homogenous e.g. in terms of their species composition compared to non-forested areas. The differences between conservation value maps would have been larger if also other habitat types than forest habitats would have been mapped using different methods.

While our habitat types were derived from literature describing species habitat type preferences (Rossi & Kuitunen, 1996), we did not map species communities as such. The used habitat type preference data does not directly tell about species richness or rarity in a specific patch but about the potential richness or rarity. Potential species richness does not always correlate with real species richness and different patches inside a habitat type might have different species richness (Hilli & Kuitunen, 2005).

Although our valuation alternatives 1–3 (and 4–6; Table 2, Figure 3) were rather similar, the valuation alternatives 1–3 and resulting maps were different from valuation alternatives 4–6 in which naturalness was taken into account. Hence, conservation criteria selection has a major effect on the conservation value maps. The major difference was that in the valuation alternatives 1–3, habitat types with strong human influence got high values whereas their values were defined to be low in the other valuation alternatives. Hence, the usage of valuation alternatives 1–3 in conservation decisions can be seen as counter-intuitive, since this could result in allocating more land under more prominent human influence. In different studies, it has been found that vascular plant species richness is higher in areas with greater human influence (Honnay et al., 2003) and higher in managed forests than in unmanaged forests (Paillet et al., 2010). In other words, it is advisable to include considerations of naturalness or other taxa to complement vascular plant species richness when decisions about conservation are made. The differences between the valuation alternatives were most evident when Zonation was used with similarity included. In this case, higher valued habitat types were replacing lower valued habitat types due to the similarities in the potential species composition.

Overall, it is feasible to compare different habitat type classifications and valuations, since there are so many different approaches, how habitat types are

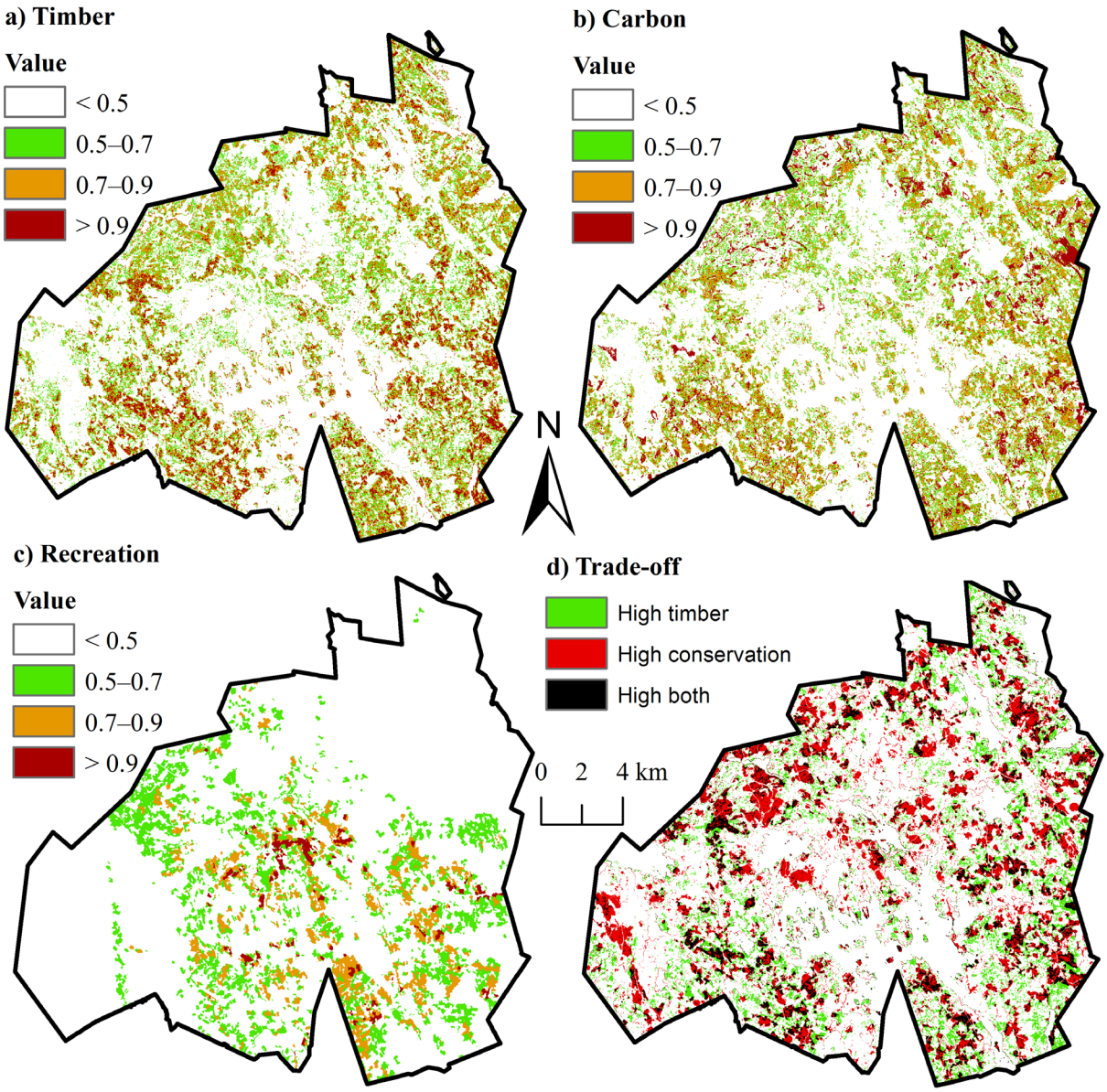

Figure 4: Different ecosystem service maps of the study area a–c and a trade-off image d. In a–c proportional landscape value is drawn with most valuable decile in red. In a and b only forest and peatland areas are considered. In c, classification alternative 1 is used. In d, the areas that belong to most valuable quartile of timber and/or conservation value are drawn using different colors. Only areas in forest and peatland are drawn; hence, there are also areas in highest quartile of conservation value that are not drawn. Conservation value is mapped using Zonation with connectivity only, classification alternative 2, and valuation alternative 6 with no value given to human habitats (Figure 3e). This conservation value map had the highest correlation with the timber map (0.47).

classified and valued. As we showed, classification and valuation do matter but their effect is nevertheless smaller than the effect of different connectivity and complementarity calculation methods.

4.2 Differences in connectivity and complementarity mapping methods

Zonation and FRAGSTATS produced very different outputs, which was quite expected. It is more important to discuss when, how, and why the

maps were different than to analyze if they were different. Differences were evident, for instance, inside separate patches. While FRAGSTATS gave uniform value to each patch, in Zonation based maps, there was large variation inside patches. Major reason behind the difference is that Zonation is less dependent on patches than FRAGSTATS. This can be seen as an asset: patches can be very different, when different classifications or scales are used. Differences in patches, then, have effects on calculated landscape metrics (Mas et al., 2010). Hence, an approach in which patches are not used can be more robust. This was confirmed also in our analysis, since we judged maps produced by Zonation as more convincing.

The fundamental difference between Zonation and FRAGSTATS raises the question, if it is feasible to compare maps produced with Zonation to maps produced with FRAGSTATS. Comparisons, however, illustrate the locations where maps have differences and show that connectivity and complementarity calculation stage in conservation value mapping is extremely important, because it causes the largest discrepancies between maps. One reason behind these large differences is that connectivity and complementarity mapping affects the configuration of maps. Even if habitat type valuation method is changed, maps are configured in the same way (i.e. the locations of patches do not change and the same habitat types get uniform values). Zonation, however, changed patched maps to continuous maps and our FRAGSTATS calculations gave a unique value to each patch in a specific habitat type.

Based on our analysis, Zonation produced credible maps; but the result maps have to be used with care in practical conservation decisions. For instance, Zonation prefers interior areas of large patches due to better connectivity. In some cases, however, edges between habitat types are not bad for connectivity, and edges may also be inhabited by different vascular plant species. Therefore in Zonation, an edge effect fix file can be added (Moilanen et al., 2012). However, only one edge effect fix file can be added and it is not, thus, habitat type specific. When habitat types are used in Zonation, using only one fix file is problematic, because for species that inhabit different habitat types different edges may be harmful.

In addition to the differences between different software, parameter or other settings inside software have a significant impact on the result. We used Zonation with similarity matrix implemented only for connectivity and for connectivity and habitat type similarity. It is not straightforward to judge whether habitat type connectivity or similarity or both should be included in the analysis. On the one hand, the high-value areas were patchier when similarity was included, and an aggregated reserve network has been argued to be biologically more valuable (Moilanen & Wintle, 2006). On the other hand, when similarity was included, high-valued habitat types were preferred.

Even if one knows that, e.g., large contiguous well-connected areas of old forests should be conserved (e.g. Hanski, 2000; Lehtomäki et al., 2009); there might be controversies where these areas are located. In our study, the location of large and well-connected (herb-rich) forest areas differed depending on the selected classification alternative. Furthermore, also different methods for connectivity calculations yielded different results. Therefore, there is urgent need for reliable mapping of valuable areas using remote sensing and landscape ecological knowledge. Additionally, conservation decisions should be, in the end, based on field observations and stakeholder involvement. Remote sensing and GIS methods may help in choosing the areas where to do field work (Rossi & Kuitunen, 1996; Hilli & Kuitunen, 2005). These areas could be, e.g., locations that have high conservation values in the maps.

4.3 Mapping ecosystem services and combining different maps

At least in European scale, it has been found that maps of single ecosystem services can differ drastically (Schulp et al., 2014). Furthermore, ecosystem services have often been mapped using land cover or other thematic maps as proxies, and, it has been shown that using land cover data only in

ecosystem service assessment might give inaccurate results both on coarse scale and fine scale analyses, since the provision of ecosystem services is not uniform inside one land cover type (Eigenbrod et al., 2010; Lavorel et al., 2011). Partly this shortcoming has been met, for example, by using biotope maps with high spatial and thematic detail (Vihervaara et al., 2012) or using GIS datasets of valuable natural areas and areas that potentially have impacts for ecosystem service provision (Kopperoinen et al., 2014). Nevertheless, as we showed in our analysis, also habitat type maps can be very different, which also has an effect on resulting ecosystem services maps. Therefore, the usefulness of thematic maps in ecosystem service assessments should be tested more thoroughly and spatially explicit data should be used if it is available.

In our map of carbon storage, we used habitat types as proxies for soil carbon. The amount of carbon in soils in forest areas is more or less constant but in peatland the thickness of peat varies. Thus, there is a double uncertainty in terms of soil carbon in peatland: both habitat type as well as the thickness of peat may differ from the mapped. Of our ecosystem service maps, the most accurate estimates were obtained for the recreational value, since different forest types were given the same scenic beauty value. In this map, however, the main doubt about accuracy is, if this kind of map is a sound estimate of recreational value. Only main recreational routes were considered, and different people regard different things as attractive in landscape (e.g. Kaltenborn & Bjerke, 2002).

In selecting overall, what areas should be conserved, multiple conservation values and ecosystem services need to be considered. In our analysis, it was found that also combined maps differ heavily from each other. For instance, the selection of possible areas for forestry regeneration cutting can be made by mapping areas with low conservation value and high timber value. It should be, however, known where these areas exist. If there are already large uncertainties in mapping the optimal areas for species conservation, even larger uncertainties can occur when the species conservation maps are combined with uncertain maps of ecosystem services (for uncertainties in ecosystem service maps, see Schulp et al., 2014). Therefore, when decisions about conservation are made, different uncertainties should be taken into account.

Overall in our study, the ecosystem services maps had low correlations with the conservation value maps. Yet, our three modeled ecosystem services were primarily services that are produced in forests and peatland; whereas in the conservation value maps, also other habitat types got high values. Ecosystem service maps were, then, the most similar with the conservation value maps with no value given to human habitats. In addition, the difference between conservation value and ecosystem service maps was partly caused by different mapping methods. In mapping conservation values, issues such as habitat type connectivity and complementarity were taken into account but these were not considered in ecosystem service maps, because they are not as important in ecosystem service provision. Therefore, conservation value maps and ecosystem service maps are not directly comparable. However, the differences between these maps show that very different conservation decisions can be made if different conservation criteria are chosen. All in all,, studied ecosystem services and conservation values should be selected so that they are relevant in the specific case using sound and accurate data.

5 Conclusion

In this study, we analyzed to what extent conservation value maps change, when mapping methods and input datasets are changed. As a basis, we used habitat type maps, to which we used different habitat type valuation and connectivity and complementarity mapping methods. Differences between maps were the largest when different methods for incorporating connectivity and complementarity, i.e. spatial conservation prioritization (Zonation) or landscape metrics (FRAGSTATS), were used. Also inside spatial conservation prioritization or landscape metric

approaches, there were quite large differences in maps, if evaluation of habitat type complementarity or similarities was changed. Moreover, differences between maps were considerable, if different conservation criteria, i.e. species richness and rarity or naturalness, were used, or different habitat type maps were used. Differences were rather small, when species richness and rarity weights were changed. Overall, we found that when mapping approaches are changed, there are large differences in the conservation value maps. Conservation value maps were also different from maps of selected ecosystem services (timber volume, carbon storage, recreational value). The extent of this difference depends, though, on how conservation values and ecosystem services are mapped.

Because the usage of different mapping methods greatly affects final maps, maps should be produced and interpreted with great care and different uncertainties should be considered. It has been acknowledged that maps of conservation values and ecosystem services are important tools in planning and decision making (e.g. Vanden Borre et al. 2011, Maes et al. 2012). When conservation decisions are made, maps should be produced in a dialogue with different stakeholders, and local context should be taken into account. In the mapping process, it should be asked, at least, why nature should be conserved in this context in the first place and what and how conservation values should be mapped. In other words, it should be decided how much different conservation criteria, such as species richness, naturalness and ecosystem services, should be weighted, what aspects of these criteria should be mapped, what datasets should be used and what kind of mapping methods are suitable in the case. The aim of the mapping and conservation should be decided first, since different conservation criteria represent different issues and the maps of these criteria are essentially different. Also after the decision about conservation criteria, very different maps can be produced, because the selection of the mapping method matters. When these questions are addressed through a thorough discussion, appropriate maps of conservation values can be produced.

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