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Shifting spatial priorities for ecosystem services in Europe following land use change

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ABSTRACT

Policy objectives to maintain ecosystem services are increasingly set. Methods to identify priority areas for ecosystem services can assist in the implementation of such policy objectives. While land use change is an important driver of changes in ecosystem services over time, most prioritization studies do not account for land use change or only assess negative effects. We assessed the effect of land use change on ecosystem services in Europe for a 40-year period and the subsequent consequences for identifying priority areas.

We quantified five services under current and future land use. For both time frames all sites were ranked based on their service provision using Zonation. To assess the sensitivity of the prioritization to land use change we compared the location of priority areas and the level of ecosystem services within priority areas in the two time frames.

Land use change shifts the location of priority areas. Overlap in priority areas over time ranges from 34.8% overlap for the top 1% priority areas to 75.4% overlap for the top 25% priority areas. Moreover, land use change affects the availability of ecosystem services in top priority areas: Compared to current top priority areas, future top ranked priority areas have lower pollination and carbon sequestration capacity. Capacity of erosion control and flood control are stable over time and nature-based tourism increases.

Shifts in priority areas are driven not only by local land use change, but also by land use change in the wider landscape, through connectivity effects and shifts in the relative importance of sites. The real management challenge lies in maintaining ecosystem services within landscapes where production and conservation objectives need to be reconciled and priority areas are affected by both local and landscape wide changes in land use. Moreover, we show that land use change has both local positive and negative effects on ecosystem service priorities, indicating that prioritization studies should not solely incorporate negative effects of land use change.

1. Introduction

Multiple competing demands for land exist, ranging from the production of agricultural and forestry products to the need for recreational spaces and a healthy living environment. Ecosystem Services (ESs) are affected by different facets of land use change including land cover conversion, (de)intensification of land management and changing the spatial arrangement of land cover types (Seppelt et al., 2016). In order to maintain ESs, policies aim to protect land with high values for ESs alongside biodiversity (Convention on Biological Diversity, 2010). To assist implementation of these policies researchers have developed approaches to prioritize areas for services (Cimon-Morin et al., 2013; Remme and Schröter, 2016; Verhagen et al., 2017).

Prioritization analysis ranks landscape units with respect to the occurrence of multiple ESs and is often part of a wider approach for systematic conservation planning, aimed at identifying the most cost-

effective protected area network (Moilanen et al., 2009). Priority areas are the highest ranked areas, often identified based on a predefined threshold, together providing the highest amount of ESs. Most prioritization analyses for ESs are based solely on the current state of land use and ESs (Luck et al., 2012). Such analyses, therefore, do not address the maintenance of ESs over time and do not support the development of management strategies to alleviate the impacts of land use change.

Within a conservation planning framework land use change is mostly considered as a threat to ESs (Cimon-Morin et al., 2016; Luck et al., 2012). Land use change can however have both positive and negative effects on ESs. Studies have indicated that for the European Union over time ES capacity increases for some services while decreasing for others, and is driven by changes in climate but especially by changes in land use (Polce et al., 2016; Schröter et al., 2005). At the local and regional level the effect of land use change varied from strongly positive to strongly negative, irrespective of the general trend

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Fig. 1. Schematic overview of the analysis. We have a total of three land use maps which are used as an input to calculate the capacity of five ecosystem services. These ecosystem service maps are the input for the prioritization analysis which results in a full priority ranking of the landscape per experiment. For the prioritization analysis in Zonation the ecosystem service input maps are split by flow zone, meaning that erosion, flood control and pollination each have multiple input maps. The maps presented here are only used for display purposes and do not represent the input or output maps of the analysis.

in ES capacity at the EU level (Metzger et al., 2006; Polce et al., 2016; Schröter et al., 2005; Stürck et al., 2015b). The local effects of land use change on ES capacity depend on the type of land use change, the service considered and the local biophysical context. Prioritization studies should thus account for both positive and negative effects of land use change on ESs.

A number of previous studies have linked land use change to ES models and prioritization techniques. Some studies directly link land use change to the prioritization analysis, without quantifying ESs under future conditions. One example of this are studies that exclude areas from the prioritization network based on projected land use change or development (Cimon-Morin et al., 2016; Troy and Wilson, 2006). Another example thereof are studies that assign positive and negative weights in the prioritization analysis to areas with projected land use change (Luck et al., 2009; Nagendra et al., 2013; Phua and Minowa, 2005; Pouzols et al., 2014; Wendland et al., 2010). The main limitation

of these approaches is that the effect of land use change on ESs is often assumed uniform irrespective of the local context. Other studies use the assessment of ESs as an intermediate step in the prioritization analysis. Quantifying ESs under both current and future land use simulations can account for non-uniform responses of ESs to land use change based on local conditions. This type of approach has been mostly used to study the effect of a single land use change process, such as deforestation and urbanization, on future ESs and prioritization (Reyers et al., 2009; Venter et al., 2009). Recently, Fan et al., (2016) prioritized areas in a watershed in Japan for multiple ESs and multiple land use change processes using this approach.

The provision of ESs can depend on local close by or global far-off ecosystems depending on the characteristics of ES flow (Fisher et al., 2009). ES flows are the connections between areas with ES capacity and those with ES demand. Not accounting for ES demand results in the identification of priority areas with high ES capacity but possibly low

ES provision to beneficiaries (Cimon-Morin et al., 2014; Verhagen et al., 2017) but data on ES demand is not commonly available (Maes et al., 2016; Wolff et al., 2015). This can be corrected for by incorporating flow zones as a proxy for ES demand (Verhagen et al., 2017). Flow zones constrain the prioritization by only considering ES capacity in areas that have a potential demand for that service. Air quality regulation is only a service to humans in populated areas and flood regulation only provides a service to humans in watersheds with human-use. Furthermore, an area can consist of multiple flow zones where each zone is limited by the extent of the ES flow, i.e. multiple watersheds. For local and regional services ES capacity can only contribute to a demand if the ES capacity is maintained within the flow zone. In other words, protecting flood regulation capacity in a watershed in Finland does not provide flood regulation in a watershed in Germany. Depending on the service considered, local losses of the service following land use change cannot be offset by distant gains in that service (Verhagen et al., 2017). Given that many ESs are supplied at sub-global scale it is of particular importance to analyze the effects of land use change on ES prioritization at both global and sub-global scale.

The aim of our study is to assess the effect of land use change on the priority ranking of areas for ESs. Here we define priority areas as areas above a certain rank (e.g. top 10%), which together provide the highest level of ESs. Using the European Union (EU) as our study area, we specifically ask: 1) to what extent does projected land use change shift the location of priority areas and does it affect the level of ESs maintained within these areas; and 2) how are such shifts spatially distributed across regions. Land use change projections in Europe are characterized by a combination of locally intensifying land management and large-scale agricultural land abandonment. These contrasts make Europe a suitable case study to test the interplay between potentially positive and negative impacts of land use change on prioritization for ESs. Our study provides spatially explicit information for managing priority areas by identifying on the one hand current priority areas that might be vulnerable to the negative effects of land use change, and on the other hand areas where favorable land management can provide opportunities for future priority areas for ESs.

2. Methodology

Our analysis consists of three major parts (Fig. 1): 1) modeling current and future land use (Section 2.1); 2) The quantification and mapping of ESs capacity (Section 2.2); and 3) the prioritization analysis (Section 2.3). In Section 2.4 we explain how the results of the different experiments are compared.

2.1. Current and future land use in Europe

We used an existing set of EU land use change scenarios that simultaneously project changes in land cover and land management between the years 2000 and 2040 (Stürck et al., 2015a). We briefly describe the main features of the land use simulations, and refer to Stürck et al. (2015a) for details.

We used a consistent modeling chain linking data on land cover, socio-economic drivers and land use change at an appropriate level of detail required to predict changes in related ESs (Lotze-Campen et al., 2017; Stürck et al., 2015a). Data on land cover and land management intensity in 25 EU member states (excluding Croatia, Cyprus and Malta) was compiled for the years 2000 (“current”, given a lack of consistent data for all variables of more recent date) and for two land use scenarios in 2040 (“future”). For the year 2000, CORINE land cover data were thematically aggregated to 16 land cover class at a 1 km resolution (Table 1). Land management intensity was assessed for croplands, grasslands and forests. Management intensity on croplands was determined using nitrogen application (Overmars et al., 2014), on

Table 1

Overview of land cover classes for the years 2000 and 2040. The land cover classes are based on the land cover classes in the DynaCLUE model (Stürck et al., 2015a; Verburg and Overmars, 2009). Information on land management intensity is not incorporated.

Land cover classes
Built-up area
Arable land
Pasture
Semi-(natural) vegetation
Inland wetlands
Glaciers and snow
Irrigated arable land
Recently abandoned arable land (only in future predictions)
Permanent crops
Forest
Sparsely vegetated areas
Beaches, dunes and sands
Salines
Water and coastal flats
Heather and moorlands
Recently abandoned pasture (only in future predictions)

grasslands using stocking density (Temme and Verburg, 2011) and in forests using wood removal (Verkerk et al., 2014). Land management intensity was provided as an output at regional (NUTS2) level and disaggregated to 1 km² cells (Stürck et al., 2015a).

For 2040, land use projections were derived from Dyna-CLUE model simulations (Verburg and Overmars, 2009). The Dyna-CLUE model combines top-down and bottom-up drivers to allocate land use change using outputs of global scale macro-economic models as an input. We used Corine Land Cover for the year 2000 to allow for a start-up period to capture the temporal path-dependence of land abandonment and regrowth more accurately (Verburg & Overmars, 2009). A detailed description of the land allocation algorithm in Dyna-CLUE can be found in Verburg and Overmars (2009). Scenarios for future land use change followed the IPCC SRES storylines (A1, A2, B1 and B2) but were adjusted to fit the European context (Nakicenovic and Swart, 2000; Paterson et al., 2012). In this study we used the two most diverging scenarios in terms of dynamics of agricultural area, namely A1 (high agricultural abandonment) and B2 (relatively low agricultural abandonment). In Europe land abandonment is one of the dominant land use change processes impacting a wide set of environmental indicators (van der Zanden et al., 2017). The A1 Scenario (“Libertarian Europe”) is characterized by a globalizing world with strong economic growth, moderate population growth and growing demand for food and feed. Environmental policies are not enforced. The B2 scenario (“European Localism”) is characterized by a fragmented world with moderate economic growth, moderate population growth and also moderate growth of the demand for food and feed. Environmental policies are in place but implemented at a regional level. The current land use map, the A1 future land use map and the B2 future land use map, all at 1 km resolution, were used as input to the ES models.

2.2. ES models

We quantified and mapped four regulating (carbon sequestration, erosion control, flood regulation and pollination) and one cultural ES (nature-based tourism) under current and future land use. Regulating and cultural ESs are generally thought of as conservation compatible, i.e. benefits obtained from these services do not negatively impact biodiversity conservation (Chan et al., 2006; Schröter et al., 2014). Each ES was quantified for the year 2000 and for the year 2040 (scenarios A1 & B2) resulting in a total of five ES maps per experiment. We next explain the quantification of ESs and how land use change affects

ESs. A full description of the quantification of ESs over time is provided in Stürck and Verburg (2016).

2.2.1. Carbon sequestration

Carbon sequestration ($\text{Mg C km}^2 \text{yr}^{-1}$) is an important global ES provided by soils and vegetation. Land use can result in both net carbon emissions and net carbon uptake. Total carbon sequestration is calculated as the sum of 1) changes in soil carbon sequestration under all land use measured using a dynamic bookkeeping approach (Schulp et al., 2008), and 2) changes in forest biomass sequestration under changing forest management and aging forests (Verkerk et al., 2014). Yearly tree biomass increment depends on the age class of the forest and on the total wood volume (Schelhaas et al., 2007). Management negatively affects carbon sequestration due to wood removal and in subsequent years results in higher carbon sequestration rates due to younger forest stands resulting in higher yearly wood increment (Verkerk et al., 2014). Changes in sequestration by soils thus only depend on land cover whereas changes in sequestration by biomass depend on both changes in land cover and land management. Initial carbon stocks and carbon sequestration rates are quantified using emission factors depending on country and land cover type. In order to solely focus on ESs and not disservices, all negative carbon sequestration values, i.e. emissions, were set to zero. Our carbon sequestration model only accounted for annual carbon sequestration rates and did not account for changes in stored carbon stocks in soil and vegetation.

2.2.2. Erosion control

Erosion control (t/ha/year) is defined here as the local protection of soils against erosion. We used the erosion control model described in Tucker et al. (2013) which quantified soil erosion based on the USLE equation (Wischmeier and Smith, 1978). The USLE equation combines data on climate, soil type, slope length and land cover type to calculate erosion risk. To distill the contribution of the land cover type to alleviating soil erosion risk we subtracted the actual soil erosion (i.e. the soil erosion with land cover) from the potential soil erosion (i.e. the soil erosion without land cover) (Luck et al., 2012; Verhagen et al., 2016). Erosion control is also influenced by land management on arable land such as contour farming or the presence of stone walls (Panagos et al., 2015) but not by the intensity of fertilizer use. Therefore, we did not account for management intensity and erosion control is only affected by changes in land cover type.

2.2.3. Flood regulation

Flood regulation is here defined as the interception and infiltration of runoff towards streams due to land cover. Flood regulation was quantified based on a look up table approach incorporating precipitation regime, catchment type (e.g. mountainous), location of land cover in the catchment and water holding capacity (Stürck et al., 2014). The water holding capacity is a factor of both the land cover type and management intensity of all land cover types. Flood regulation is quantified as a dimensionless index ranging from 0 to 1. Changes in land cover and management intensity directly affect flood regulation. All other parameters were assumed constant over time (Stürck et al., 2015b).

2.2.4. Nature-based tourism

Nature-based tourism denotes the capacity and attractiveness of ecosystems to support nature tourism. Nature-based tourism was quantified based on the method of Van Berkel and Verburg (2011) including as indicators elevation difference and the presence of lakes, rivers and protected areas (Van Berkel and Verburg, 2011). Nature-based tourism additionally depends on the surrounding landscape which was classified into dominant urban, agricultural or forest landscape and a mixed or mosaic landscape (Van Berkel and Verburg,

2011). We used an updated version of this method by van der Zanden et al. (2017). This updated version excluded areas with EU's high nature value farmland policies as an indicator because the future of this subsidy scheme is already included in the A1 and B2 land use scenarios. Over time nature-based tourism changes due to changes in land cover type, locally, or through changes in surrounding land cover. The quantification solely focused on the ecosystems' capacity to support nature-based tourism and does not account for use or demand through proxies like accessibility.

2.2.5. Pollination

Pollination, here limited to pollination by wild bees, is an important local flow ES for the production of certain food and biofuel crops. We mapped pollination flows between natural vegetation and croplands based on a combination of two existing studies (Serna-Chavez et al., 2014; Zulian et al., 2013). Pollination flow is sometimes assigned to croplands but from a conservation perspective pollination capacity needs to be assigned to natural vegetation, i.e. the main wild bee pollinators' habitat. Therefore we made some adjustments to the approach of Serna-Chavez et al. (2014). All land cover types were assigned a nesting suitability score based on a look-up table from Zulian et al. (2013), generally assigning high values to semi-natural vegetation and forest edges. However, pollination requires a service flow from habitat areas to croplands. Field studies across Europe showed that wild bee diversity on croplands was separately affected by the proximity of habitat areas and the management intensity of the cropland (Hendrickx et al., 2007). We categorized management intensity of cropland into three classes based on fertilizer application (low: $< 50 \text{ kg/ha}$, medium: $50\text{--}150 \text{ kg/ha}$ and high: $> 150 \text{ kg/ha}$) (Stürck et al., 2015a,b). To account for the proximity of cropland to natural vegetation we calculated the area of cropland (0–100%) in the eight cells directly surrounding each habitat site. In this neighborhood each cropland contributed proportionally, depending on the management intensity to account for the negative impacts of intensive management on pollinators. High intensity croplands have been found to host approximately only half the wild bee diversity of lowest intensity croplands (Hendrickx et al., 2007). Wild bees are important pollinators for many crops and studies on apple orchards highlight that increased species richness of wild bees resulted in increased fruit set (Garibaldi et al., 2013; Mallinger and Gratton, 2015). We therefore assumed that low intensity cropland had a maximum 100% area contribution, medium intensity cropland had a reduced 75% area contribution and high intensity cropland had a reduced 50% area contribution. Highest pollination values were thus assigned to cells with forests or semi-natural vegetation surrounded by low intensity croplands. Over time pollination capacity was affected by changes in the local land cover type of the habitat cells and also by changes in land cover and management intensity of croplands in the proximity of natural vegetation.

2.3. Prioritization approach

We used the software package Zonation v4, a tool developed to identify the most cost-effective network for conservation of biodiversity or other features (Moilanen et al., 2014, 2009). Zonation has been previously used to identify ES priority areas in e.g. England, Japan and the European Union (Casalegno et al., 2014; Fan et al., 2016; Verhagen et al., 2017). Zonation assigns a continuous ranking to all cells in a landscape based on the occurrence of features of interest (here: ESs), iteratively removing the cells with the lowest score for the features (Lehtomäki and Moilanen, 2013). After removing the lowest ranked cells from the prioritization the values of the remaining cells are updated based on the amount of ES in the lost cells relative to the amount of ES within the remaining cells. In other words, for abundant ESs the penalty of losing part of the ES value in the beginning is low but

increases when the ES becomes scarcer. Zonation differs from target based land planning approaches that have been used for ES prioritization including Marxan (Chan et al., 2006), Marxan with Zones (Schröter et al., 2014) or C-Plan (Cimon-Morin et al., 2016). Zonation is most useful when individual targets per ES cannot be set, an issue for most services (Remme and Schröter, 2016).

We used flow zones as a proxy for ES demand by only identifying priority areas in locations with ES demand (Verhagen et al., 2017). As a pre-processing step for the prioritization analysis the European map per ES is split into individual maps per ES specific flow zone. Zonation treats each flow zone input map as a separate protection features resulting in a better distribution of priority areas across flow zones in Europe with minimum impact on the overall protection of each ES (Verhagen et al., 2017). The identification of flow zones differs per ES and partly depends on the type of ES flow. Nonetheless there is yet no uniform way of identifying flow zones per ES and therefore we chose an approach that is both biophysically relevant but also computationally feasible. Carbon sequestration is a global flow ES and is not spatially restricted, due to the global climate system. Therefore, we use the entire EU as flow zone for this service. Nature-based tourism was also considered as a global flow ES, due to the global nature of today's tourism. We define nature-based tourism as nature recreation in larger areas lasting multiple consecutive days. For this type of nature-based tourism there is no absolute limit on the travel time or distance. In contrast, more localized day to day recreation services including urban leisure in green spaces can be restricted to individual city areas (Verhagen et al., 2017). As we restricted ourselves to tourism rather than recreation, we use the entire EU as flow zone for this service. Flood regulation and erosion control are dependent on the movement of water and soil particles between and within catchments. We used 5th order catchments with a minimum area of 2 km² to delineate flow zones for these two ESs (EEA 2008). For the prioritization analysis this means that at each cell-removal round (i.e. a step in which cells that contribute least are removed from the set of remaining priority areas) only the values of those cells belonging to the same catchment as where cells were removed from, are updated. Pollination is a very localized ES flow restricted by the flight distance of wild bee species (Ricketts et al., 2008). Following Verhagen et al. (2017), we split the pollination services map into 10x10km zones but only included zones containing agricultural land. This resulted in approximately 38,000 zones (and equally many input files) for pollination. Flow zones for carbon sequestration, erosion control, flood regulation and nature-based tourism are considered stable over time. For pollination, agricultural expansion and abandonment can result in shifting flow zones over time. To account for this flow zones were identified as locations with agricultural land under

current or future land use. Therefore potential shifts in the location of flow zones for pollination did not affect the results.

We ran a total of three Zonation experiments, one per land use projection (2000, 2040A1, 2040B2), using their respective set of five ES maps as input (Fig. 1). Every prioritization analysis requires the setting of parameters and here we mention the most important ones. See Appendix 2 for all settings. All experiments had equal settings, only the actual input (ES maps) differed per experiment. In Zonation, one needs to define how the individual ES maps are aggregated to calculate a combined ES value per cell. We used Zonation's aggregated benefit function, which sums the ES value of all features per cell thus prioritizing areas that can cost-effectively cover multiple ESs (Lehtomäki and Moilanen, 2013). Each input file needs to be assigned a weight. For each ES the number of input files is equivalent to the number of flow zones. The aggregated weight assigned to each ES is 100. For each service the aggregated weight is equally distributed over the number of flow zones of that service. Finally, Zonation requires the setting of costs for the analysis. Incorporating non-uniform costs results in more cost-efficient protected area networks for biodiversity (Naidoo et al., 2006). Here we however used land as a uniform cost measure because we were only interested in the effect of land use change on ES capacity and it is unknown how conservation costs will change over time. All Zonation analyses were run using the SurfSARA HPC cloud facilities for high performance computing.

2.4. Analysis of prioritization results

For each experiment the output comprised a full ranking of the landscape based on the occurrence of ESs at a 1 km resolution (Fig. 1). The following assessments were made: 1) we assessed the degree of congruence in priority rankings between experiments; 2) we determined the type of land use change current and future priority areas were subject to; and 3) we assessed how the level of ESs in priority areas changed.

We assessed the congruence between top ranked areas of each experiment calculated as the percentage of priority areas in experiment A that is also a priority area in experiment B. To assess the sensitivity of the result to the size of the priority area network chosen we analyzed the results for the top 1%, 2%, 5%, 10%, 17% and 25% ranked cells. We included the 17% threshold as it relates to the Aichi target 11 to protect 17% of the land globally for biodiversity and ESs (Convention on Biological Diversity, 2010). For each set of top X% priority areas we calculated the degree of spatial congruence (i.e. level of agreement in terms of sites included in the top X%) between all combinations of the three experiments. We did not consider changes in the specific ranking

Table 2

Overview and description of land use changes. Most land use change categories follow the classification from Stürck et al. (2015a). For visualization purposes some land use changes are grouped according to the classification in column 3.

Trajectory class	Trajectory Name	Description
Land cover Conversion	Stable	No change in either land cover or land management intensity
	Land abandonment cropland	Change from arable land to natural vegetation or recently abandoned land
	Land abandonment pasture	Change from pasture land to natural vegetation or recently abandoned land
	Crop2pasture	Change from arable land to pasture
	Crop conversion	Change from one category of arable land to a different category
	Recultivation pasture	Change from pasture to arable land
	Natural succession	Change from recent abandoned land or semi natural vegetation to forest land cover
	Recultivation forest	Change from forest to arable land
	Urbanization	Change from any land cover category to urban or peri-urban area
Change in land management	Cropland extensification	Stable arable land category, reduction in fertilizer
	Cropland intensification	Stable arable land category, increase in fertilizer use
	Pasture extensification	Stable pasture land cover, reduction in stocking density
	Pasture intensification	Stable pasture land cover, increase in stocking density
	Forest extensification	Stable forest land cover, reduction in wood extraction
	Forest intensification	Stable forest land cover, increase in wood extraction

of sites *within* each top X%.

We compared priority areas under current and future land use, separately for the A1 and B2 scenario, and classified them into three groups as only priority in 2000 (“current only”), only priority in 2040 (“future only”) or priority in both 2000 and 2040 (stable). We combined the three groups of priority areas with the land use change process in each location to analyze the relation between land use changes and changes in top priority ranking. Following Stürck et al. (2015a), we identified land change trajectories based on the change in land use between 2000 and 2040, but made some minor adjustments for the purposes of this paper (Table 2). Land change trajectories are hierarchically based on changes in land cover followed by changes in management intensity (Stürck et al., 2015a). Land management intensity for croplands (fertilization), grasslands (stocking density) and forests (wood removal) was quantified using a continuous scale. The continuous management intensity values were reclassified into three management intensity classes for cropland (low, medium, high) and two management intensity classes for grasslands (low, high) based on previously determined threshold values (Sturck and Verburg, 2016). For cropland and grassland sites, intensification or extensification (i.e. a reduction of management intensity) was recorded when the intensity class changed over time. For forests we used the original continuous management intensity classification and changes in land management intensity (intensification vs. extensification) were recorded based on a 5% change in wood removal between 2000 and 2040. As described in Section 2.2, the changes of ESs over time depend on changes in land cover and land management intensity.

For the assessment of ESs change in priority areas, we analyzed two different metrics of change in ESs. First, we calculated the % of each ES maintained for increasing percentages of land assigned as priority area. A comparison across the full priority ranking provides information on how ESs in priority areas are affected by land use change and to what extent changes in ESs are reflected across the full ranking. Second, we calculated the mean change in ES capacity for priority areas at the regional level, using NUTS3-regions (Eurostat/GISCO, 2011). NUTS3-regions are statistical units for the whole EU largely following administrative boundaries in which population ranges from 150,000 to 800,000.

3. Results

3.1. Shifting priorities for ecosystem services

Both land use change scenarios result in shifts in top priority areas

Fig. 2. Degree of overlap in priority areas for ecosystem services between the prioritization experiments. Results are presented for varying percentages of land assigned as priority (x-axis). The percentage overlap between priority areas is for the experiments 2000 and 2040 (A1) (green), 2000 and 2040 (B2) (orange) and 2040 (A1) and 2040 (B2) (blue).

over time (Fig. 2). Here we primarily focus on the top priority areas but see the Appendix 1 for a full priority ranking per scenario. The congruence between current priority areas and future A1 priority areas ranges from 34.8% for the top 1% priority areas to 75.4% for the top 25% priority areas. Thus for a priority network of around 40,100 km² (top 1%) only 13,900 km² retains its priority rank under both current and future land use, while 26,200 km² is no longer a top 1% priority area due to land use change. For the B2 scenario the contrast is not as strong (43.3% overlap with current in top1%), but this still means the majority of sites is no longer a top 1% priority site due to land use change (Fig. 2 orange bars). Priority area sets identified under projected land use for 2040 (A1 vs. B2, Fig. 2 blue bars) have much more overlap indicating that differences in land use change among scenarios have a smaller effect on shifting priorities.

Priority areas shared between all three prioritization experiments are spatially clustered and are predominantly located in the Mediterranean and in mountainous regions such as the Alps, the Carpathians and the Pyrenees. Priority areas unique to a specific experiment are much more dispersed and scattered across Europe without clear spatial patterns (Fig. 3). We identified three sets of shifting priorities: i) priority areas at risk, i.e. those priority areas unique to the year 2000 land use prioritization; ii) new priority areas, i.e. those priority areas unique to the year 2040 land use prioritization (A1 or B2); and iii) stable priorities, i.e. priority areas shared between two or three prioritizations of the years 2000 and 2040.

In general we observe two generic patterns of priority area distribution. First, we see areas where both stable and non-stable priority areas are clustered. Here the non-stable priority areas tend to be located at the edges of larger patches of stable priority areas (panel C in Fig. 3). This pattern is common around all locations with larger stable priority areas. Second, we see areas where both stable and non-stable priority areas are completely scattered without any spatial patterns (panel A in Fig. 3). This pattern is common in the absence of larger patches of stable priority areas and occurs predominantly in northern European countries.

Changes in priority areas at the cell level are only partly reflected in regional changes. Out of the 1296 NUTS3-regions 1071 primarily have stable priority areas, even in regions where the stable priority areas are scattered (Fig. 4). However, 60 NUTS-regions have a relative high share of priority areas that are identified as such only in the year 2000. These regions are located in Northern Scotland, Central France, Southern Fenno-Scandia and the Baltic states and especially in Poland and Czech republic. Here land use change has predominantly negative effects on ESs, resulting in a concentration of areas that initially are prioritized but are likely to forego priority under land use change in the absence of conservation measures. Only 13 NUTS-regions had a high share of new priority areas following land use change. The lower number of regions with new priority areas compared to the number of regions with foregone priorities indicates that newly established priority areas are not concentrated in specific regions.

The top priority areas are characterized by only a limited set of land use change processes (Fig. 5). We cannot identify one dominant land use change process across all three sets. Forest management intensification strongly overlaps with the location of priority areas at risk (33.5% or 69,620 km² of priority areas unique to the year 2000 land use). Forest management intensification negatively affects both carbon sequestration and flood control. Other ESs are not affected by forest management intensification, which possibly explains why a fraction of top priority areas remains stable, despite forest management intensification (17.4% or 86,312 km² of stable priority areas). New priority areas are largely characterized by land abandonment (17.0% or 35,329 km² of priority areas unique to the year 2040 land use) and forest succession (32.0% or 66,503 km² of priority areas unique to the year 2040 land use). Dominant land use change processes affecting

Fig. 3. Congruence in top priority areas (17%) between the three prioritization experiments. The left depicts the congruence in top 17% priority areas between the three experiments for Europe. The two right panels show the congruence of top 17% priority areas for two regional cutouts. Stable priorities are priority areas that are at least shared between two of the three experiments.

priority areas are thus forest intensification, land abandonment and forest succession whereas other land use processes do not show clear patterns with the shifts in priority areas.

3.2. Changing ecosystem services over time and space

Land use change shifts priority areas over time but it only results in declines of ES capacity for a limited set of ESs (Fig. 6). For the entire EU land use change results in declines of carbon sequestration and

pollination whereas erosion control, flood regulation and nature-based tourism are stable or show minor increases. This general pattern is also reflected in the top priority areas: future top priority areas have lower pollination and carbon sequestration capacity than current top priority areas. Pollination declines stronger at the EU scale (−18.0%) than at the scale of the top 17% priority areas (−14.3%). In contrast, carbon sequestration declines stronger at the scale of the top 17% priority areas (−17.4%) than at the scale of the EU (−16.5%). This means that the shift in priority areas could partly compensate for the losses in

Fig. 4. Share of priority areas per administrative unit (NUTS3-region) assigned to a specific class of priority areas namely: (A) priority areas only under current land use, (B) stable priorities under both current and future land use (A1 experiment) and (C) priorities only under future land use (A1 experiment).

Fig. 5. Overlap between land use change and three sets of priority areas, namely (A) priority areas only under current land use, (B) stable priorities under both current and future land use (A1 experiment) and (C) priorities only under future land use (A1 experiment). The area of the respective pie charts depicts the difference in land area. Results are presented for the top 17% priority areas.

pollination capacity, but was unable to compensate for the losses of carbon sequestration capacity in Europe. Land use change thus results in shifts in priority areas but the future capacity is only decreasing for a limited set of ESs.

Although the total capacity of three out of five ESs remained stable across priority areas in Europe (Fig. 6), at the regional level, land use change affected ES capacity (Fig. 7). In almost none of the NUTS3-regions ES capacity was stable over time within their priority areas. Most commonly, land use change results in trade-offs between ESs at the regional level and these trade-offs occurred all across Europe. In other regions land use change results only in increases or decreases in ES capacity. These regions tend to be spatially clustered: decreases in ES at the regional level are found in Eastern Europe and in France. Eastern Europe, especially Poland and Czech Republic, are characterized by large areas with forest and cropland intensification. ES capacity tends to increase in many Spanish regions, southern England and in Denmark characterized by forest expansion and decreasing management intensity of cropland and pastures. Land use change is thus driving changes in ES capacity at the regional level all across Europe and regional changes in ES capacity do not necessarily translate into changes in ES capacity at the EU level.

4. Discussion

4.1. Reflection on the results

In this study we prioritized areas in Europe for ESs. We did so for current and future land use based on results of dynamic models for land cover change, land management change and ESs over time. An important finding is that land use change resulted in shifts in the location of priority areas over time. These shifts in the location of priority areas can have different causes. For a number of services, i.e., pollination and nature-based tourism, not just local land use change, but also land use change in the neighborhood can affect local ES capacity. Especially for pollination cropland abandonment and cropland expansion can

strongly affect where ESs are supplied locally because of the required proximity between natural habitat and cropland areas for pollination. In a German agricultural landscape changes in ES capacity could not be fully explained without incorporating changes in landscape configuration besides changes in local land cover (Lautenbach et al., 2011). Effects of landscape configuration could be partly accounted for by connectivity measures in Zonation (Kukkala and Moilanen, 2016). However, for many ESs there is still limited knowledge on how configuration affects ES capacity (Verhagen et al., 2016). For all services local land use change changes the local ES capacity but because all results are nested it also changes the relative priority of other areas. For the prioritization analysis non-local changes in ES capacity can affect both the relative ranking of sites and the complementarity value of sites regarding ESs even without any change in ES capacity over time at the site itself. Accounting for flow zones partly accounts for this issue because only changes in ES capacity over time only affect the relative value and complementarity of sites within a flow zone for that particular service. To conclude, non-local changes in land use may affect the prioritization of an area 1) by changing its ES capacity through configuration effects, or 2) without changing its actual ES capacity, but because the ES capacity of an area is always relative to the ES capacity of other areas.

Besides land use change also forest aging affected the priority area allocation. For carbon sequestration forest aging affected the sequestration capacity over time. This effect is relatively limited because forest aging could only explain a 4% decrease in carbon sequestration over time without land use change (Schulp et al., 2008).

These interacting processes make it difficult to determine the causes of changes in priority at specific locations. In some areas changes in priority occurred while land use was stable while in other areas land use change did not affect the priority assigned. Nevertheless, from the overlay analysis it is clear that forest intensification is an important reason why areas of high priority under current land use are no longer top-ranked in 2040. In contrast, expansion of high intensity cropland (+12% in area in B2 scenario) does not directly affect current priority

Fig. 6. Percentage of ecosystem service (ES) maintained with increasing % of land assigned as priority area. The level of ES maintained is relative to the total amount of ES under current land use (2000), meaning that the lines for future land use (scenarios 2040 (A1) and 2040 (B2)) can exceed the 100% ES maintained (as is the case for nature-based tourism). The more concave the graph the smaller the fraction of land required to maintain a high % of that particular ES. Vertical dashed lines indicate the top priority levels ranging from 1% to 25%. Although the prioritization is performed for all ES combined, results are presented for each ES separately to highlight the differences between the ESs.

areas for ESs because these were already mostly located outside croplands. At the same time, agricultural abandonment and forest succession are important factors explaining new priority areas.

Previous studies have mostly looked at the impact of land cover change on ESs (Metzger et al., 2006; Polce et al., 2016; Stürck et al., 2015b). In the study of Polce et al. (2016), a set of three regulating services showed only minor changes in ES capacity between 2000 and 2050 for most areas in Europe. In two different studies, carbon sequestration capacity over time is expected to slightly increase as result of land cover change, especially agricultural abandonment (Schulp et al., 2008; Stürck et al., 2015b). The limited or negative impact of land use change on the aggregated ES capacity in our analysis can be

explained by the fact that we used a different set of ES models and because we fully accounted for changes in land management in addition to changes in land cover. Previous research already highlighted that different methods to map ESs across Europe also result in different ES maps (Schulp et al., 2014). This applies to current assessments of ESs but is also likely to hold for future predictions of ESs. More importantly, irrespective of the changes in aggregated ES provisioning at EU level, all studies find strong local and regional differences in the effect of land use change on ES capacity (Alcamo et al., 2005; Polce et al., 2016; Schulp et al., 2008; Stürck et al., 2015a). In other words, although the exact changes in ES capacity differ somewhat between studies, the main finding of this study - that land use change will result in shifting

Fig. 7. Projected change in all ecosystem services within priority areas per administrative (NUTS3)-region. Both panels show the change in ecosystem services under current and future land use for experiment A1, high agricultural abandonment (A), and experiment B2, low agricultural abandonment (B). An ES is considered to increase (decrease) over time if the difference between current and future ES supply is at least 5%. Results are presented for the ES capacity within the top17% priority areas.

priorities driven by local changes in ESs - is likely independent of the trends in ES capacity over time and the choice for particular ES models.

Within priority areas land use change resulted in both positive and negative changes in ES capacity over time at the local (cell), regional and EU level. Earlier studies seldom accounted for land use change directly, but rather assessed it as a threat to identified priority areas using negative weights (Cimon-Morin et al., 2016; Luck et al., 2012). In a study for a remote region in Canada expansion of urban areas resulted in decreasing ES capacity within priority areas over time (Cimon-Morin et al., 2016). In such circumstances, postponing the identification of a protected area network would lower the ESs over time (Cimon-Morin et al., 2016). When multiple land use change processes are included the effects on ESs are more diverse, with local increases and decreases. For example, in a study on a watershed in Japan, Fan et al. (2016) found no to slightly positive effects from land use change on ESs, with hardly any impact on the associated locations of priority areas. From this and our study we can conclude that accounting for the full array of possible land use change processes locally results in both positive and negative effects of land use on ESs without necessarily threatening ES capacity for the entire study region. Hence, in contrast to the common approach in prioritization studies it is not possible to assign uniform negative effects of land use change on priority areas for ESs because such an approach does not account for differing impacts of land use change on ES depending on the land use change, the ES studied and local positive and negative effects of land use change based on spatial configuration.

4.2. Reflection on the methodology

We used a straightforward approach to account for the effects of land use change on ES prioritization. Because of our primary interest in the effects of land use change we did not account for a number of other factors important for prioritization for planning purposes, such as non-uniform land costs. Consideration of land costs can also increase the efficiency of conservation networks (Kark et al., 2009). Here we used area as a uniform cost measure which is a common proxy in ES prioritization research (Remme and Schröter, 2016; Verhagen et al., 2017). Land costs can be approximated using different measures including land

prices, management costs and opportunity costs of foregone production (Naidoo et al., 2006). Threats including land use change are sometimes used as a surrogate for costs but should preferably be treated separately (Naidoo et al., 2006). Land costs are likely to change over time, partly as a result of changes in land management, land cover and societal demand. However, how these costs will change in the future is unknown. Therefore, we decided to use a uniform cost measure over space and time and only assess the direct effect of land use change on ES capacity.

We prioritized land in Europe based on the occurrence of five ESs. Although being a limited set, the ESs selected do cover the full array of possible responses to land use change with some being only sensitive to changes in local land cover, some being sensitive to changes in local and surrounding land cover and some being sensitive to land management. Conservation management for biodiversity and ESs especially in production landscapes should incorporate all these facets of land use change (Seppelt et al., 2016; Verhagen et al., 2016).

In our analysis we only focused on the top ranked priority areas as is most common in prioritization studies. In recent years prioritization studies have also looked at the full ranking of a landscape or used prioritization to identify areas most suited for development based on the lowest ranked areas (Kareksela et al., 2013; Nin et al., 2016). The focus on top priority areas is especially important from a conservation perspective in relation to threats from land use change. We found that current priority areas were mostly affected by forest management intensification and new priority areas had undergone land abandonment or forest succession. For the whole of Europe mean priority rankings were more strongly driven by changes in land cover compared to land management (Appendix 1). For example, the conversion of pastures and forests to cropland had a strong negative effect on mean priority rankings but hardly impacted top priority areas. In contrast, forest management intensification across the full priority ranking had almost no effect on the mean priority rank. This finding shows that it is important to distinguish between land use change processes affecting the full priority ranking versus land use change processes affecting priority areas.

In our analysis we did not account for impacts of climate change on

ESs over time. Recent studies looked at the combined effect of land use and climate change on ES capacity (Polce et al., 2016) and on ES prioritization (Fan et al., 2016). In Europe land use change had a stronger effect on ES capacity over time compared to climate change (Polce et al., 2016). Moreover, the timeframe of our study is relatively short. On longer time scales the relative impacts of climate change as compared to land use change may be larger.

Land use and land use change models at the European extent are never validated due to inconsistencies between datasets and large uncertainties in the most important land change processes in the Corine Land Cover multi-year data. However, the model application used in this study was part of an inter-model comparison of land change models at European extent (Alexander et al., 2017). This effort showed large variations between European land use models, with the DynaCLUE model falling within the range of that variation (Alexander et al., 2017). For smaller scale applications the DynaCLUE model has often been validated and generally has a high validation score (e.g., Pontius et al., 2008). In spite of this, the work presented here should be interpreted with care and be interpreted in terms of the variation in the effect of land use change on priority areas for ESs under two contrasting scenarios of land abandonment.

Our study has, indirectly, accounted for the demand of ESs through the use of flow zones. Previous studies have highlighted that changes in ES demand are likely larger than changes in ES capacity over time and might result in increased mismatches between areas of high ES capacity and areas of high ES demand (Alcamo et al., 2005; Stürck et al., 2015a). Whether changes in priority areas over time are more strongly driven by changes in ES capacity or by changes in ES demand remains an important question for future research with possibly important consequences for the management of ESs.

4.3. Implications for management

There is a need to operationalize the concept of ES in landscape planning and decision making (Balmford et al., 2011; de Groot et al., 2010). In general, the identification of priority areas cannot be easily translated into management practices. Identification of priority areas is often part of a wider framework of systematic conservation planning (Moilanen et al., 2009). The aim is to identify the most cost-effective protected area network where management for ES would focus on protection alongside biodiversity conservation. Whether the maintenance of ESs requires the designation of protected areas, or other measures, depends on the nature of the threats to the services.

However, for many ESs a certain use of the area is required to obtain benefits from ESs. According to some authors, the ES selected for prioritization should be compatible with biodiversity conservation objectives (Chan et al., 2006; Schröter et al., 2014). These areas should then be managed for a sustainable use of the area where benefits obtained from ES are not threatening biodiversity conservation. However, knowledge on what levels of use are compatible with biodiversity is missing and limiting use of ecosystems might be in direct conflict with meeting certain demands for these services requiring an expansion of sites providing ESs. Land management should aim to navigate these potential trade-offs regarding the conservation of biodiversity and ESs (Cimon-Morin et al., 2013).

A better understanding of the scale and impacts of land use change can provide guidance for the management of priority areas. Stable priority areas tend to be clustered in remote mountainous regions of Europe. Here the identification of priority areas may not need to result in the designation of protected areas. The real management challenges lie in minimizing land use impacts on ES capacity in more intensively-

used landscapes i.e. where production and other usage of the land co-occur and are often competing (Schulp et al., 2016). Here ESs are often supplied by relatively small patches of semi-natural vegetation. Similar for biodiversity, in these areas pressures of land use change, including intensification, are highest (Seppelt et al., 2016). Here the protection of patches providing local services might be required. In Europe, the Natura 2000 network is the most important land protection scheme for biodiversity with a relatively good performance for biodiversity protection (Kukkala et al., 2016; van Teeffelen et al., 2015). A simple overlay of the current Natura 2000 network (EEA, 2016) with priority areas that are at risk of land use change, indicates that around 75% of the priority areas at risk are found outside this network (not shown). This suggests that the current Natura 2000 network may need extension if it is to protect most of the current priority areas for ESs that are at risk of land use change. To test the effectiveness of protected area measures for ESs future research could develop scenarios where priority areas for ESs are coupled to land use simulation models, i.e. allowing no land use change in the top current priority areas for ESs.

Land use change processes, especially agricultural land abandonment and forest succession, resulted in the establishment of new priority areas. Land use planning and management could prioritize these areas to develop into a way that maximizes the gains in ES capacity. This coincides with the notion that conservation should not only aim at the protection of current priority areas but also stimulate favorable development/restoration of new areas with high potential. In Europe this relates especially to agricultural abandonment areas that can only achieve their potential upon appropriate management of re-wilding (Ceausu et al., 2015; Schulp et al., 2016).

Our results indicate that avoiding land use change at a single location is not sufficient to maintain the same level of ESs due to neighborhood interactions, requiring a landscape-scale approach (Mitchell et al., 2015; Verhagen et al., 2016). Moreover, many ES need to be supplied at a local or regional scale requires that priority areas are distributed across flow zones (Verhagen et al., 2017). While techniques are being developed to account for connectivity and minimum area requirements for ESs (Kukkala and Moilanen, 2016), these two factors suggest the importance of an integral land management approach.

Our study also found regional changes in ES capacity within priority areas over time (Fig. 7). Assessments and management of ES over time should therefore not only look at the EU wide impacts of land use change on ES capacity but also account for changes at the regional level. Many ESs have a regional or local flow with benefits being obtained locally or regionally, which makes this a logical management scale for this type of ESs. For most ESs, local losses in ESs capacity cannot be offset by distant gains. For biodiversity, non-coordinated management at smaller spatial scales to prioritize areas is known to result in sub-optimal outcomes (Kark et al., 2009; Pouzols et al., 2014). Further developing techniques to balance between the most economic outcome and the need to maintain most ESs at smaller spatial scales is a crucial step to effectively identify priority areas in a context of land use change.

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Appendix 1. Additional figures

This appendix is part of the publication: Verhagen et al. (2018). Shifting priorities for ecosystem services under land use change. Ecological Indicators. See Figs. A1 and A2.

Fig. A1. Full priority ranking for ecosystem services in Europe. The full priority ranking is depicted for A) 2000, B) 2040 (A1 scenario) and C) 2040 (B2 scenario).

Fig. A2. Change in mean priority rank following land use change. The results are separated by system, i.e. croplands, pasture and forest and semi-natural vegetation. The difference in the mean priority rank is between current land use and future land use (experiment A1).

Appendix 2. Zonation settings

This appendix is part of the publication Verhagen et al. (2018). Shifting priorities for ecosystem services following land use change. Ecological Indicators.

Below we provide an example of the settings file used for the prioritization analysis in Zonation. The settings were the same for all three prioritization experiments.

```
[Settings]
removal rule = 2
warp factor = 2000
edge removal = 0
initial removal percent = 0.0
annotate name = 0
mask missing areas = 1
area mask file = ../../../../dataWV/processed/
Zonation/MaskClueEUZonation.tif
post-processing list file = ppa_config.txt
```

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