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Including stakeholders' perspectives on ecosystem services in multifunctionality assessments

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ABSTRACT

Multifunctional landscapes are used and shaped by a range of different stakeholders. The high number of diverging values, interests or demands in such landscapes can lead to conflicts that impact sustainability goals. In this study, our aim was to include stakeholders' valuations of ecosystem services in multifunctionality assessments and thereby to identify different and possibly contradictory perspectives on landscape multifunctionality. Two European cultural landscapes, the *Vereinigte Mulde* (Germany) and the *Kromme Rijn* (The Netherlands), were used as case studies. Spatially explicit indicators of eleven ecosystem services were assessed and weighted according to their survey-based perceived importance for different stakeholder groups. While some significant differences between the groups were apparent, the results also revealed that all stakeholder groups acknowledge the importance of multiple ecosystem services. Stakeholder-specific multifunctionality hotspots occurred mainly in forests or grasslands and largely overlapped between the groups. Our study therefore clearly shows that the diversity of ecosystem services must be preserved in order to preserve the values that cultural landscapes offer to a wide range of people. While local solutions must be sought to resolve local land use conflicts over the use of ecosystem services, we conclude that multifunctionality can be declared a common goal.

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1. Introduction

Studies assessing landscape multifunctionality have recently been regarded as an important knowledge base to inform land management (Stürck and Verburg 2017; Manning et al. 2018). These studies assess the supply of multiple ecosystem services (i.e. benefits that people obtain from ecosystems) – including the provision of food and other raw materials, the regulation of environmental processes and the cultural experience of nature (MA 2005) – and aggregate them into one multifunctionality indicator (Höltинг et al. 2019a). Areas of high multifunctionality are thus regarded as areas where multiple benefits are provided to a range of stakeholder groups (Wiggering et al. 2003; O'Farrell and Anderson 2010).

Because multifunctional landscapes potentially meet diverse needs of society, increased multifunctionality is often associated with increased levels of human well-being (e.g. MA 2005; Rodríguez-Loinaz et al. 2015; Kremen and Merenlender 2018). However, in areas where many different land uses occur and multiple interests exist, stakeholders could use different ecosystem services and benefit to varying degrees from each of these services (Turkelboom et al. 2018; Plieninger

et al. 2019). Zoderer et al. (2019) for example found that benefits of local farmers are mostly associated with provisioning services, while residents preferred regulating services and visitors preferred cultural services. Since different interests in ecosystem services supply could result in conflicts, ecosystem service multifunctionality should not simply be directly linked to the *overall* levels of human well-being. Instead, further research is needed to differentiate between various groups of stakeholders and the benefits each group receives from multifunctional landscapes.

Until now, multifunctionality is predominantly assessed based on the potential supply of ecosystem services (i.e. the capacity of a particular ecosystem to provide ecosystem services within a given time period; Spake et al. 2017), including analyses of positive and negative correlations between ecosystem service supplies (i.e. synergies and trade-offs, respectively). The demand for ecosystem services, their actual use or the benefits that people receive from ecosystem services have been less of a focus, as reviewed in Höltинг et al. (2019a) (but see: Martín-López et al. 2012; Iniesta-Arandia et al. 2014; Darvill and Lindo 2015; Quintas-Soriano et al. 2019). However, it has



widely been argued that multifunctionality assessments could enhance their uptake in planning and management, when taking a more differentiated view on the beneficiaries in multifunctional landscapes, their values, interests and demands (Mastrangelo et al. 2014; Turkelboom et al. 2018; Plieninger et al. 2019).

Based on previous research (e.g. Raudsepp-Hearne et al. 2010; Byrnes et al. 2014; Darvill and Lindo 2015), Manning et al. (2018) developed a new approach to assess multifunctionality for different stakeholder groups. In this approach, indicators of ecosystem service supply are weighted according to the benefits that various stakeholder groups associate with individual services. The aggregation of the weighted ecosystem services results in multifunctionality indicators that differ between the stakeholder groups. Here, we are using the approach suggested by Manning et al. (2018) to present stakeholder-weighted multifunctionality assessments in two case study areas. We aim to demonstrate different perceptions of multifunctionality that are reflected in spatial patterns within our study landscapes. We hypothesize that ecosystem services are not equally valued by all stakeholder groups in an area and that different spatial patterns of multifunctionality can be observed for the different groups (Darvill and Lindo 2015).

In fact, management decisions are commonly made by the more influential stakeholder groups and often target marketable ecosystem services, while less influential stakeholder groups and non-marketable ecosystem services are overlooked (Turkelboom et al. 2018). Finding solutions that reduce such power inequalities and increase benefits for a wide range of stakeholders remains a great challenge in multifunctional landscapes. In this study, we have thus included both influential and non-influential stakeholder groups, as well as marketable and non-marketable ecosystem services. It has further been questioned whether stakeholders really appreciate multifunctionality or whether conflicts between them can actually arise due to the multifunctional character of a landscape (von der Dunk et al. 2011; Komossa et al. 2019; Zoderer et al. 2019). Recreationists, for example, who visit an area primarily to enjoy its tranquility, beauty and fresh air, might come into conflict with land users who disrupt its recreation potential (e.g. noise, dust or odour pollution from agricultural practices) (Martín-López et al. 2012). Nature conservationists, as another example, might come into conflict with recreationists who use the area for leisure activities (e.g. mountain biking, camping) without paying attention to the habitats of rare species (Young et al. 2005).

While we do not assess land use conflicts specifically, we identify the overlap between stakeholder-specific multifunctionality hotspots to discuss where conflicts could arise. We analyse interactions between the

ecosystem services to find out whether the stakeholder groups actually perceive existing trade-offs or synergies. Moreover, we tested spatial co-occurrences between stakeholder-specific multifunctionality hotspots and land use types to find out which land use classes are commonly considered multifunctional. Our study focuses on two landscapes that have previously been characterized as multifunctional landscapes and are representative for numerous cultural landscapes in Central and Western Europe (Verhagen et al. 2018; Karner et al. 2019; Komossa et al. 2019). Our specific research questions are: a) How do different groups of stakeholders in multifunctional landscapes value the importance of multiple ecosystem services?; and b) How do quantitative indicators of multifunctionality differ (regarding absolute values and spatial patterns) between different stakeholder perspectives?

2. Materials and methods

2.1. Description of the case study areas

The two case study areas (Figure 1) are densely populated peri-urban landscapes (i.e. formerly rural areas that are now shaped by urbanization; Shaw et al. 2020) and thus represent both residential and economic areas. They are comparable in size, known for their unique landscape scenery and biodiversity, and primarily used for recreation and agricultural production. The Kromme Rijn area (219 km^2 , 86.090 inhabitants) is located outside of Utrecht (The Netherlands). It is used for recreational purposes by the people of Utrecht, local residents and other visitors, as well as for dairy and fruit production (Verhagen et al. 2018; Komossa et al. 2019). Important landscape elements are open pastures, woody linear elements, orchards, the river beds of the meandering river 'Kromme Rijn' and cultural heritage sites (Tieskens et al. 2018). The Vereinigte Mulde area (241 km^2 , ca. 40.000 inhabitants) is located in Northwest Saxony (Germany). Large-scale farming systems exist, but 36.6% of the area is forested. The area is known for its unique biodiversity and its designated conservation zones (LfULG 2008). Moreover, the area is used for local recreation, also by urban dwellers from the nearby city of Leipzig.

2.2. Data gathering

2.2.1. Selection and indicators of ecosystem services

We selected eleven ecosystem services of social, economic and environmental value within the two regions (Table 1). The services cover provisioning, regulating, cultural and supporting services. The selection was based on previous stakeholder workshops and expert knowledge from the TALE project (www.ufz.de/tale/, Holzkämper et al. 2018; Karner et al. 2019). The

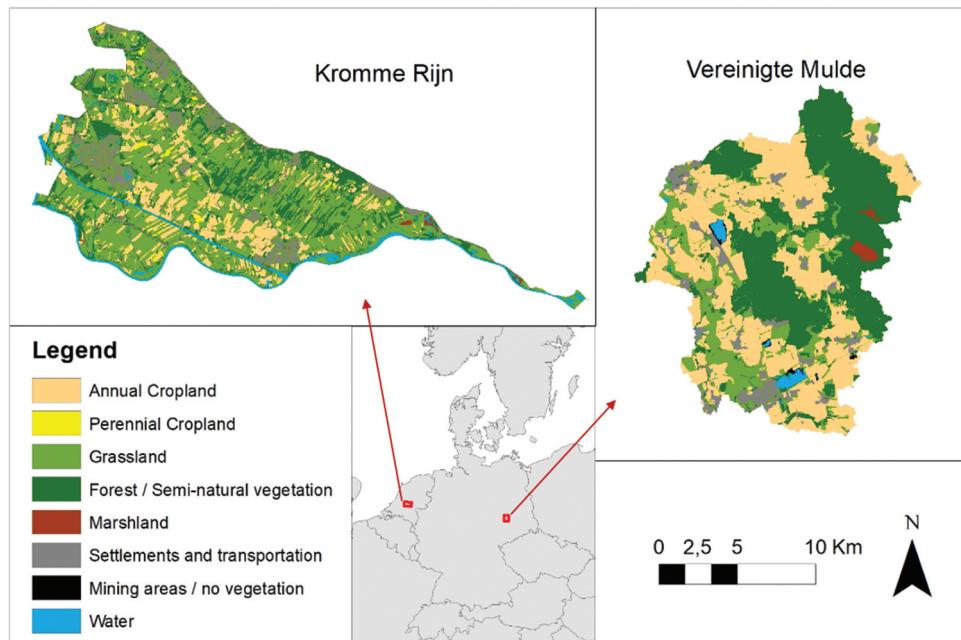


Figure 1. The two case study areas and their location in Europe. Spatial data on land use was provided by the German Surveying Authorities and the Land Use Database of the Netherlands.

Table 1. Indicators used to assess the supply of ecosystem services. Ecosystem service categories (MA 2005) and units are given.

Vereinigte Mulde			
Ecosystem services	Indicators used	Ecosystem services	Indicators used
Provisioning ecosystem services			
1. Crop production	Yields (dt/ha) of 15 crop types from the year 2017 (STLA 2017a, STLA 2017b); Calorie content (kcal/dt) of those crop types (FAO 2001)	1. Crop production	Yields (dt/ha) of 41 crop types (vegetables, cereals and strawberries), averaged for the years 2007–2016 (CBS 2019a, CBS 2019b, CBS 2017); Calorie content (kcal/dt) of those crop types (FAO 2001)
2. Timber production	Timber harvest (m ³ /ha*a), averaged for the years 2002–2012 for three forest types (coniferous, deciduous, mixed) (BWI 2016)	2. Timber production	Timber harvest (m ³ /ha*a) for the years 2006–2012 (NBI 2012)
3. Dairy production	Soil fertility (1–5) of grassland areas (LfULG 2015)	3. Dairy production	Total production of milk (€) based on the area of grassland, the amount of milk produced and the profits possible (Agrimatie, 2014; Verhagen et al. 2018)
4. Water supply	Water stored in the environment (mm) (Arnold et al. 2012; Strauch et al. 2017)	4. Water supply	Groundwater level (meter above NAP) (NGR 2019)
		5. Fruit production*	Fruit profits per farm (kg), based on the pollination potential and potential crop production deficits (CBS 2018; Heijerman-Peppelman and Roelofs 2010)
Regulating ecosystem services			
5. Climate regulation	Carbon stored below- and aboveground (MgC/ha) (BWI 2016; LfULG 2015)	6. Climate regulation	Carbon sequestration, calculated as the change in carbon soil and biomass stocks (MgC/ha) (Schulp et al. 2008)
6. Water retention*	Water retention index (0–1) (Copernicus 2015; MODIS 2016; Strauch et al. 2017)		
7. Pollination	Crop visitation by insect pollinators (0–1) based on habitat suitability and floral resources (Schulp et al. 2014; Zulian et al. 2013)	7. Pollination	Crop visitation by insect pollinators (0–1) based on habitat suitability and floral resources (Schulp et al. 2014; Zulian et al. 2013)
Cultural ecosystem services			
8. Landscape aesthetics	Aesthetic value (0–1) based on the hemeroby index, the heterogeneity of the landscape and the distance to water bodies (Paracchini et al. 2014)	8. Landscape aesthetics	The amount of estimated picture uploads in the landscape (n) (Tieskens et al. 2018)
9. Recreation	Number of recreation points (e.g. castle, viewpoints) and foot or cycle ways within 500 m distance (n) (OSM 2019)	9. Recreation	Number of recreation points (e.g. castle, viewpoints), recreation areas (e.g. parks) and foot or cycle ways within 500 m distance (n) (OSM 2019)
10. Tranquility	Tranquility increase (0–1) with increasing distance to railways, major and secondary roads (OSM 2019)	10. Tranquility	Inverse noise load as mapped for the Province of Utrecht in 2000 (Overheid 2006)
Supporting ecosystem services			
11. Habitat quality	Nesting quality based on predicted species distribution maps of 9 bird species (0–1) (Jungandreas et al., in review)	11. Habitat quality	Nesting quality for Newts (0–1) based on land cover/management maps and presence of hedges (Verhagen et al. 2018)

* Instead of water retention, another provisioning service, namely fruit production, was assessed in the Kromme Rijn case study (See S1).

supply of all ecosystem services within the two case study areas was spatially modelled using selected indicators. We based the selection of these indicators on the criteria of van Oudenhoven et al. (2018) and only chose the ones that are perceived as scientifically adequate, understandable and relevant for different stakeholder in the two landscapes, and that can be quantified for other times or in other areas (van Oudenhoven et al. 2018). Required data to estimate these indicators was gathered from public databases and derived from different biophysical models. A full description of the relevance, definition and models of each ecosystem service can be found in the Supplementary Material (S1). For further analyses, all ecosystem service indicators were converted to raster datasets of 25 m resolution and standardized between 0 and 1.

2.2.2. Identification of stakeholders and elicitation of weights for ecosystem services

The selection of stakeholder groups was based on the previous identification of governance structures and stakeholders in the case study regions (Nitsch et al. 2017). Four groups of stakeholders were identified as relevant: 1. Farmers and foresters, 2. Environmental organizations, 3. Decision makers (local governments and sector-focused governmental organizations) and 4. Recreationists (including tourists and local recreationists). These cover all generic types of stakeholder groups identified in Turkelboom et al. (2018): influential users, context setters and non-influential users. We expected different interests in terms of land use objectives and a divergence of ecosystem service valuations between these groups. The sample of local farmers and foresters in the Vereinigte Mulde area also represents their main activities (dairy farming, organic farming, horse keeping, state forestry, private forestry) in the area.

The representatives of the stakeholder groups ‘farmers and foresters’, ‘environmental organizations’ and ‘decision makers’ are considered experts in their fields. We expected low variances of valuations and thus aimed to survey five representatives of these groups within each case study (Table 2). In the Kromme Rijn area, however, only two farmers, two representatives of environmental organizations and four decision makers could be surveyed. This was a result of their unwillingness to engage about environmental issues given strong debate about their impacts on biodiversity in policy in the summer 2019. In order to guarantee comparability and to allow generalizations, we therefore merged the three groups into one group of ‘Land managers and context setters’ ($n = 8$). Varying levels of local knowledge and a higher within group variance regarding the ecosystem service valuation were expected for the recreationists. We therefore surveyed 50 recreationists in the Vereinigte Mulde area and 52 recreationists in the Kromme Rijn area. For the latter, however, only 47

Table 2. Stakeholder groups and number of participants per group surveyed in the two case study areas.

Case study area	Stakeholder group	Number of participants
Vereinigte Mulde (n = 65)	Farmers and foresters	5
	Environmental organizations	5
	Decision makers	5
	Recreationists	50
Kromme Rijn (n = 55)	Land managers and context setters (2 Farmers, 2 Environmental organizations, 4 Decision makers)	8
	Recreationists	47

responses could be used due to the incomplete answers of 5 participants.

To ask the different stakeholder groups about the importance of selected ecosystem services, we employed STREAMLINE, a new survey methodology tool for community engagement and participatory research that can be tailored to the needs of individual research projects (Metzger et al. 2018; www.streamline-research.com). It comprises a series of colourful laminated A3 canvases, which are designed to understand synergies and differences in future visions among different groups. Using publicly available materials on the STREAMLINE website (images, question set-ups) we developed three canvases to engage participants and to gather data in a relatively short period of time (S2). On the first canvas, respondents were presented a map of the area. They were asked to indicate their relationship to the area (e.g. visitor, resident, occupation, etc.) and to name in their own words a few ecosystem services important to them in the specific landscape. On the second canvas, the selection of eleven ecosystem services was presented to them and the participants were asked to indicate the level of importance of each ecosystem service from the perspective of their own well-being in that particular landscape. This was done using a five point Likert-scale ranging from very little to high importance (as in Schmidt et al. 2017; Zoderer et al. 2019). On the last canvas, the participants recorded their demographic information (such as age, gender, occupation) and postal code. For participants that were unfamiliar with the ecosystem service concept, the term ‘landscape function’ was used. Furthermore, if a respondent was not sure what a specific service was, a short explanation was provided by the interviewer.

In both case study areas, recreationists were surveyed using a convenience sampling approach attempting to balance for representation of gender, age groups, recreational activities (Corbin and Strauss 1990, S3: Data on recreationists). One or two interviewers were sampling at different locations and times within the area to cover the peri-urban gradients as well as diverse recreational settings. In the German site, surveys with recreationists were conducted at six locations on seven days spread between week and weekend days between April and July 2019. In the Dutch site, surveys with recreationists were conducted at four locations on five days, spread between week and weekend days between June and September 2019.

2.3. Data analysis

2.3.1. Assessing and mapping unweighted multifunctionality

A correlation analysis between all ecosystem service indicators was performed for both case study areas using the Spearman's correlation coefficients r , to assess the number of trade-offs or synergies (significance level $p = 0.01$). Spearman's correlation coefficients are widely used in ecosystem service assessments to measure the linear relationship between two ecosystem services (Lee and Lautenbach 2016). Trade-offs were classified as weak ($r > -0.3$), moderate ($-0.5 < r < -0.3$) or strong ($r < -0.5$), following Fagerholm et al. (2012). Likewise, synergies were classified as weak ($r < 0.3$), moderate ($0.3 < r < 0.5$) or strong ($r > 0.5$) (S4: Correlation analyses).

For comparison, multifunctionality was first calculated for each raster cell (25 m resolution) without taking into account the stakeholder weighting schemes. The Simpson Diversity Index (SDI, Simpson 1949) was applied to the unweighted ecosystem service indicators. Specifically, we used the 'diversity' function of the vegan package (Oksanen et al. 2013) in R (R Core Team 2013). In comparison to other multifunctionality indicators, the SDI takes into account the proportional supplies as well as the number of ecosystem services. It thereby combines richness and abundance components (Hölting et al. 2019b).

$$SDI = 1 - \sum_{i=1}^N p_i^2$$

N = total number of ecosystem services considered; p_i = the supply of each ecosystem service (i) proportionally to the supply of all ecosystem services in that area.

In addition, multifunctionality was calculated at the landscape scale using the average of all multifunctionality values within each case study area. This resulted in one multifunctionality indicator per case study area, as in Manning et al. (2018).

In order to analyse the spatial co-occurrence of multifunctionality and land use classes, multifunctionality was reclassified into five classes by quantiles and the proportion of each multifunctionality class per land use class was calculated for both case study areas. High resolution land use maps were available for both case studies, which were partly reclassified for this analysis (AdV 2010; Hazeu et al. 2010). As some of the ecosystem service indicators are partly based on land use (S5), a sensitivity analysis was conducted to check whether the results based on the complete set of indicators are robust and non-endogenous. To this end, we recalculated multifunctionality by using only the ecosystem service indicators that are not based on land use. The absolute differences between the original and the

sensitivity analyses of both case studies were calculated (S5: Co-occurrence of land use and multifunctionality classes: Fig. S5.1 for the results of the sensitivity analyses).

2.3.2. Evaluation of stakeholder surveys

The average values of ecosystem service weightings per stakeholder group were used to obtain a weighting scheme for each stakeholder group. Paired samples t-tests were conducted to test for significant differences between the resulting weighting schemes. One-way analyses of variances (ANOVA) were used to identify statistically significant differences in ecosystem service valuations between the stakeholder groups. Since the groups were of different sizes, the ANOVAs were performed once for all expert groups (number of participants = 5 to 8) and once for the two recreationists groups (number of participants = 47 to 50). In addition, we calculated standard deviations of all ecosystem service weightings per stakeholder group. The sum of these standard deviations represents the total variance *within* each stakeholder group. Variances *between* the groups were calculated as the sum of the standard deviations of two weighting schemes (S6: Weighting schemes and significance tests).

2.3.3. Assessing and mapping stakeholder weighted multifunctionality

In the next step, multifunctionality was assessed by taking into account the weighting schemes. The ecosystem service indicators were multiplied by the stakeholder-derived weights, as visualized in Figure 2 for four ecosystem services and two stakeholder groups only. Our method differs slightly from the one suggested by Manning et al. (2018) as our weighting factors do not reflect actual benefits, but the perceived importance of ecosystem services. However, the perceived importance can conceptually be linked to the demand for ecosystem services or to the potential benefits that stakeholders receive (Costanza et al. 2011; Wolff et al. 2015; Zoderer et al. 2019).

The multifunctionality of weighted ecosystem services was then calculated for each stakeholder group and case study area ($n = 6$) in a spatially explicit way using the SDI. Additionally, landscape scale multifunctionality indicators were calculated for both case study areas and each stakeholder group ($n = 6$), as described above. In order to compare the resulting maps, we used a method that evaluates their similarity on a scale between 0 (high similarity) to 1 (low similarity) (Map Comparison Statistics (MCS), Schulp et al. 2014). Hotspots of multifunctionality were defined by the upper 20% quantile of multifunctionality (similar to Stürck and Verburg 2017) and the percentages of hot-spot-overlap between weighted and unweighted multifunctionality maps were calculated. The spatial co-occurrence of land use classes and stakeholder weighted

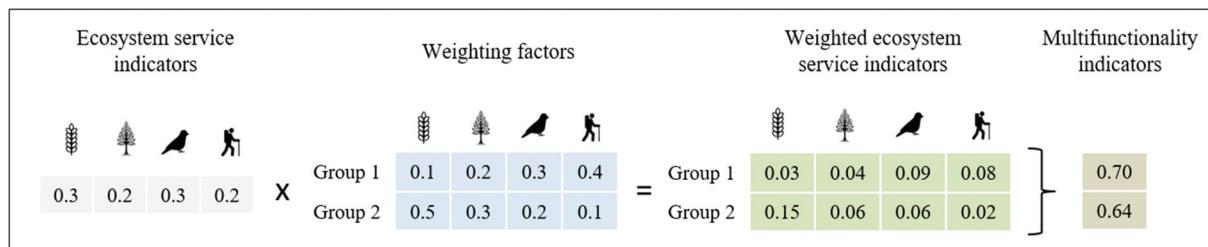


Figure 2. Assessing stakeholder weighted multifunctionality (modified after Manning et al. 2018). The ecosystem service indicators (here food production, timber supply, habitat quality and recreation potential) are multiplied with the weighting factors of each stakeholder group, which reflect the importance of the ecosystem services to the stakeholder group. In this example, Group 1 highly values the recreational potential of the landscape, whereas Group 2 values food production the highest. The application of the weighting factors results in weighted ecosystem service indicators, which are then aggregated to multifunctionality indicators by the use of the SDI.

multipfunctionality was assessed for both case study areas and each stakeholder group ($n = 6$), as described above. These results can be found in S5: Co-occurrence of land use and multifunctionality classes.

All spatial and statistical analyses were performed using R version 3.4.2. ArcMap version 10.7 was partially used for mapping purposes.

3. Results

3.1. Patterns of unweighted ecosystem multifunctionality

The maps of unweighted multifunctionality (Figure 3) clearly show where the diversity of ecosystem services provided within the landscapes is particularly high. In both landscapes, unweighted multifunctionality hotspots occurred primarily in forested areas. The Vereinigte Mulde is largely forested and a distinction is made between mixed, coniferous and deciduous forests, with mixed forests providing the greatest diversity of ecosystem services. The Kromme Rijn, on the other hand, has less forest, but a much higher proportion of grassland, where 20.5% of the multifunctionality hotspots are located. While grassland offers more hotspots than coldspots in both areas, the opposite is true for cropland. Both cropland and orchards in the Kromme Rijn provide more coldspots than hotspots of multifunctionality. Urban areas and watercourses in both areas also do not provide a significant diversity of ecosystem services. The sensitivity analyses for both case studies revealed some differences in the ranking of land use classes and the proportions of multifunctionality classes, which suggests some uncertainty in the results shown above. For the Vereinigte Mulde, the largest uncertainties were found for the land use classes 'Water', 'Other' and 'Mixed forests', where 10% of the multifunctionality classes were assigned differently compared to the results shown above (Fig. S5.1). The greatest uncertainties in the results of the Kromme Rijn case study are still in the land use classes 'Orchards', 'Water' and 'Urban' with absolute

differences of more than 15% of the assigned multifunctionality classes (Fig. S5.1).

3.2. Trade-offs and synergies

The majority of spatial correlations between ecosystem services in both case study areas were found to be moderate (S4). Strong positive correlations existed only between a few ecosystem services (Vereinigte Mulde: timber production and climate regulation, landscape aesthetics and dairy production, landscape aesthetics and pollination; Kromme Rijn: timber production and habitat quality). Strong negative correlations existed only in the Vereinigte Mulde between timber and crop production, as well as timber production and water supply. All in all, the number of trade-offs in both case study areas almost corresponded to the number of synergies (Vereinigte Mulde: 28 trade-offs/27 synergies, Kromme Rijn: 29 trade-offs/26 synergies). Most trade-offs were found between provisioning services and other service categories, while there were fewer trade-offs between regulating and cultural services. Habitat quality, the only supporting ecosystem service, was negatively correlated to most ecosystem services in the Vereinigte Mulde, but mostly positively correlated to other ecosystem services in the Kromme Rijn.

3.3. Stakeholder weightings of ecosystem services' importance

The average weighting schemes varied between the different groups of stakeholders in both areas (Figure 4). Comparing the groups clearly revealed that all groups, with the exception of the farmers and foresters of the Vereinigte Mulde, had preferences for regulating, cultural and supporting ecosystem services and valued provisioning ecosystem services less. Especially crop, timber and dairy production were valued lower than other services, with weightings not exceeding 3.5, except for the group of farmers and foresters of the Vereinigte Mulde. Of all

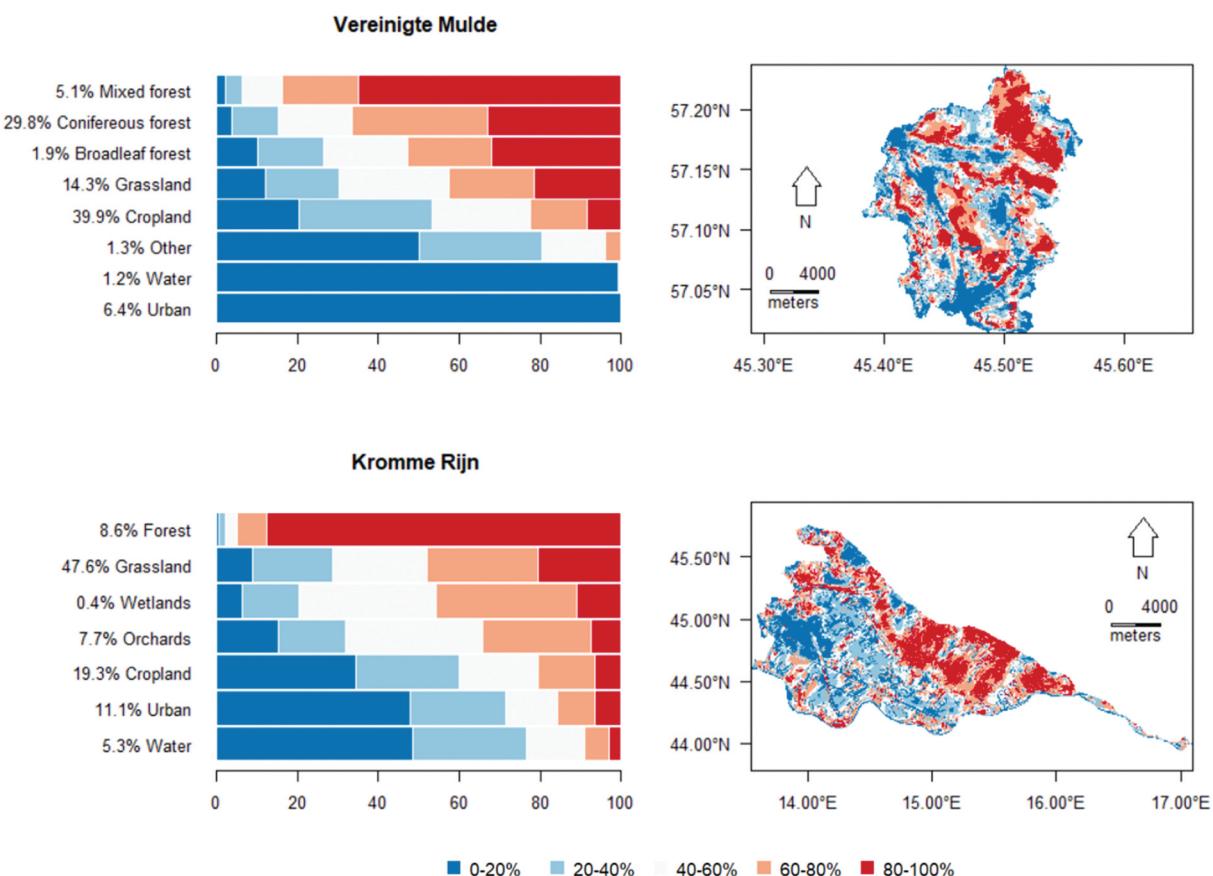


Figure 3. Patterns of unweighted multifunctionality and the co-occurrence with land use classes for both case study areas. Land use classes are depicted with the area percentages. 'Other' in the Vereinigte Mulde includes 'no vegetation', 'specialized crops', 'transport', 'wood', 'mining areas', and 'wetlands'. The colours depict multifunctionality classes, which are defined by five quantiles (Vereinigte Mulde: 0.000–0.823, 0.824–0.838, 0.839–0.849, 0.850–0.859, 0.890–0.889; Kromme Rijn: 0.308–0.666, 0.667–0.711, 0.743–0.744, 0.777–0.776, 0.870–0.867). Coordinates of the Universal Transverse Mercator (UTM) coordinate system are displayed.

the groups, the weighting scheme of the farmers and foresters of the Vereinigte Mulde was the most 'balanced'. They considered most of the services to be equally important. Their average weightings only varied between 4.0 and 4.8 with a standard deviation of 0.96 (S6).

In the case of the Vereinigte Mulde, the paired t-tests showed that there are significant differences between the final weighting schemes of farmers/foresters and decision makers, farmers/foresters and environmental organizations, and recreationists and decision makers ($p < 0.05$, S6.1). The total variance *within* the groups was strongest for environmental organizations (10.67), less strong for recreationists (9.42) and decision makers (9.04) and low for farmers and foresters (7.70). In case of the Kromme Rijn, the final weighting schemes of the land manager and context setter and the recreationists were not significantly different from each other ($p = 0.748$). The within-group variance was higher among recreationists (11.34) than among land manager and context setter (9.55).

The ANOVAs revealed whether the individual ecosystem service valuations differed significantly between the groups. In case of the smaller expert groups of both

case studies (farmers/foresters, environmental organizations, decision makers, land managers/context setters), the ecosystem service valuations were quite similar (S6.2). The strongest differences between these groups were found for the valuations of timber production and landscape aesthetics. However, the significance value was low ($p < 0.1$). In case of the two recreationist groups, small differences existed for the valuations of climate regulation and landscape aesthetics ($p < 0.1$), but only one significant difference was found for the valuation of recreation potential ($p < 0.001$, S6.3).

All group weightings, within- and between-group variances, the results of the t-tests and the ANOVAs can be found in the Supplementary Material (S6).

3.4. Patterns of stakeholder weighted ecosystem multifunctionality

Different absolute values (i.e. landscape multifunctionality indicators) and divergent spatial patterns of multifunctionality hotspots were identified after applying the weightings schemes (Table 3). In the Vereinigte Mulde, the landscape scale multifunctionality indicator was highest for farmers and

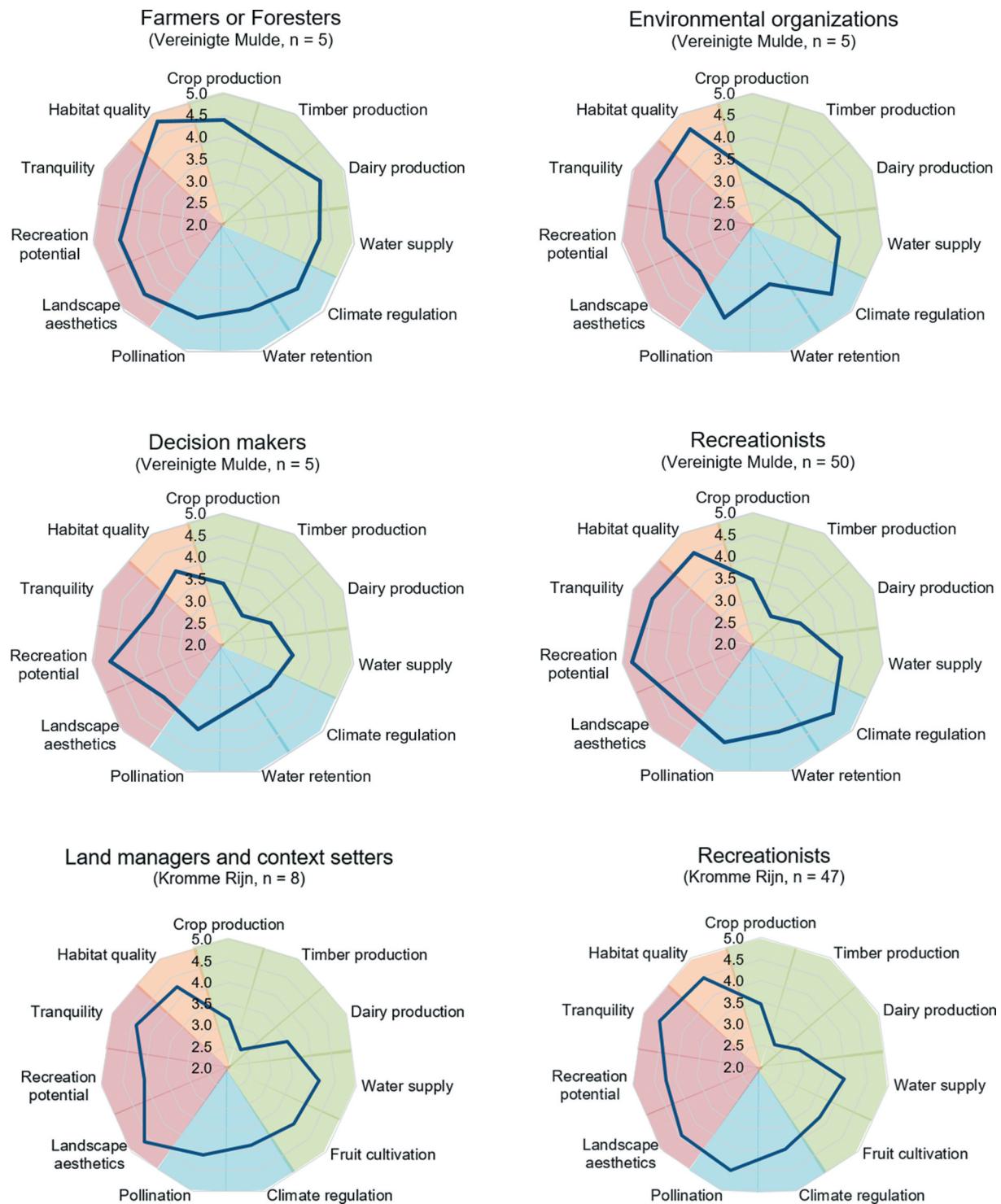


Figure 4. Average weighting schemes per stakeholder group for the Vereinigte Mulde (upper four plots) and the Kromme Rijn (lower two plots). The case study and the number of stakeholders per group are given in brackets below the stakeholder groups' names. Different colours were used to indicate the different ecosystem service categories (green = Provisioning services, blue = Regulating services, red = Cultural services, orange = Supporting services) as classified in the Millennium Ecosystem Assessment (MA 2005). Water retention was assessed only in the Vereinigte Mulde. Fruit production was assessed only in the Kromme Rijn.

foresters (0.82), followed by recreationists (0.77) and environmental organizations (0.67). The lowest value of landscape multifunctionality was calculated for decision makers (0.44). The spatial patterns of stakeholder weighted multifunctionality, as shown in Figure 5, also varied between the different groups.

For farmers/foresters and recreationists the hotspots of multifunctionality were more concentrated on certain parts of the landscape. For environmental organizations and decision makers hotspots were more wide-spread within the landscape. All in all, the stakeholder-specific hotspots largely overlapped

Table 3. Comparison of multifunctionality maps of the Vereinigte Mulde and the Kromme Rijn, including a multifunctionality indicator at the landscape scale (calculated as the mean value of multifunctionality per landscape); the percentage of hotspot overlap between weighted and unweighted multifunctionality maps; and a map comparison statistic, that evaluates the similarity of maps at a scale of 0 (high similarity) to 1 (low similarity).

Vereinigte Mulde	Landscape scale MF indicator	Overlap of hotspots with unweighted MF map	Map comparison statistics [0–1]			
			Unweighted MF map	Farmers and foresters	Environmental organizations	Decision makers
Farmers and foresters	0.821	71.9%	0.013	-	-	-
Environmental organizations	0.667	40.4%	0.162	0.156	-	-
Decision makers	0.435	26.8%	0.395	0.389	0.254	-
Recreationists	0.774	58.4%	0.054	0.051	0.112	0.343
Kromme Rijn	Landscape scale MF indicator	Overlap of hotspots with unweighted MF map	Unweighted MF map	Land managers and context setters		
Land managers and context setters	0.712	74.4%	0.034	-		
Recreationists	0.697	67.9%	0.051	0.053		

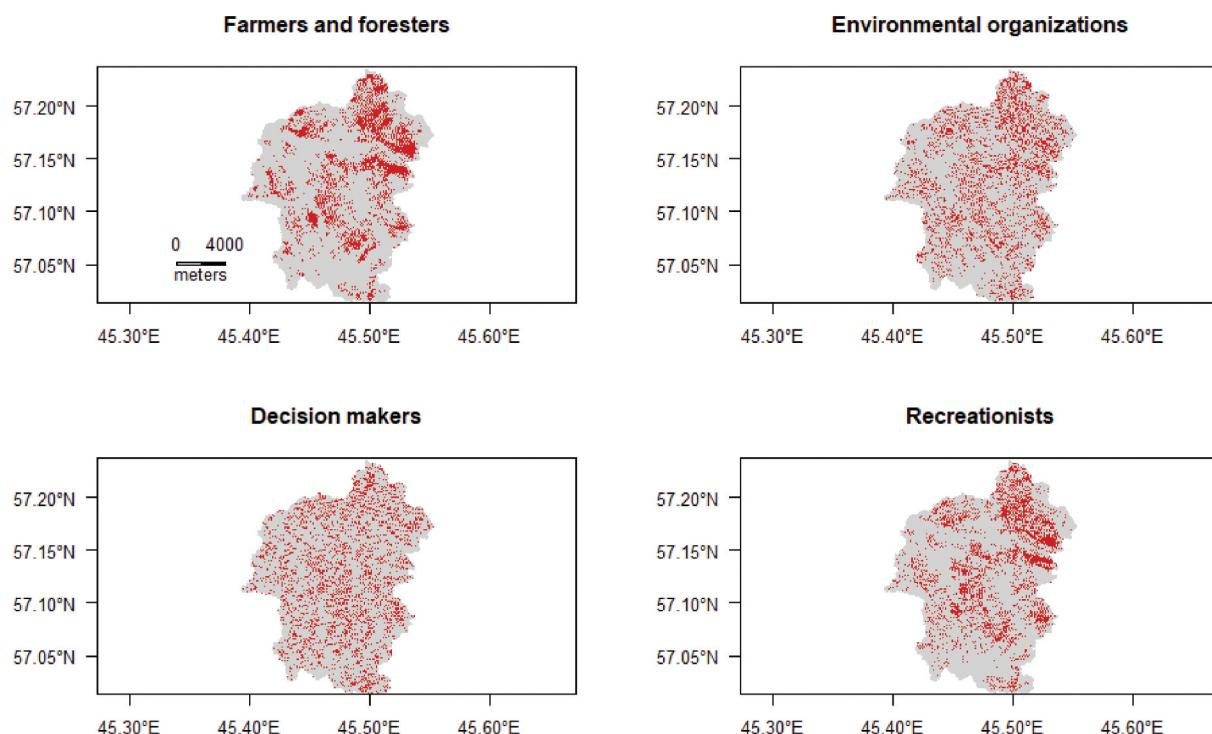


Figure 5. Stakeholder weighted multifunctionality hotspots, defined as the 80–100% quantile (red) in the Vereinigte Mulde area. UTM coordinates are displayed.

with the hotspots of the unweighted multifunctionality map (Figure 3), especially for farmers and foresters (71.9%). The least similarities were found between the maps weighted by decision-makers and farmers and foresters ($MCS = 0.39$). The most similar maps were those resulting from the weightings of recreationists as well as farmers and foresters ($MCS = 0.05$).

In the Kromme Rijn, the assessments of stakeholder-weighted ecosystem multifunctionality hardly differed between groups, both in terms of multifunctionality indicators at the landscape scale and in terms of spatial patterns (Figure 6). Landscape-scale multifunctionality was slightly lower for recreationists (0.69) than for the land managers and context setters (0.71). The two resulting maps were highly similar ($MCS: 0.053$). The

hotspots of multifunctionality identified for both groups tended to concentrate on the eastern parts of the study area. The percentage of overlap between weighted and unweighted multifunctionality hotspots was 64.3% for recreationists and 74.4% for land managers and context setters.

Minor differences between the groups were found regarding the co-occurrence of land use classes and multifunctionality hot- and coldspots (Vereinigte Mulde: Fig. S5.2, Kromme Rijn: Fig. S5.3). In the case of the Vereinigte Mulde, decision makers stood out, because the highest proportion of hotspots was not assigned to mixed forests but to coniferous forests. Moreover, hotspots for decision makers also occurred in urban areas and water bodies, while there were no hotspots in these land use classes for all other groups in that area (Fig.

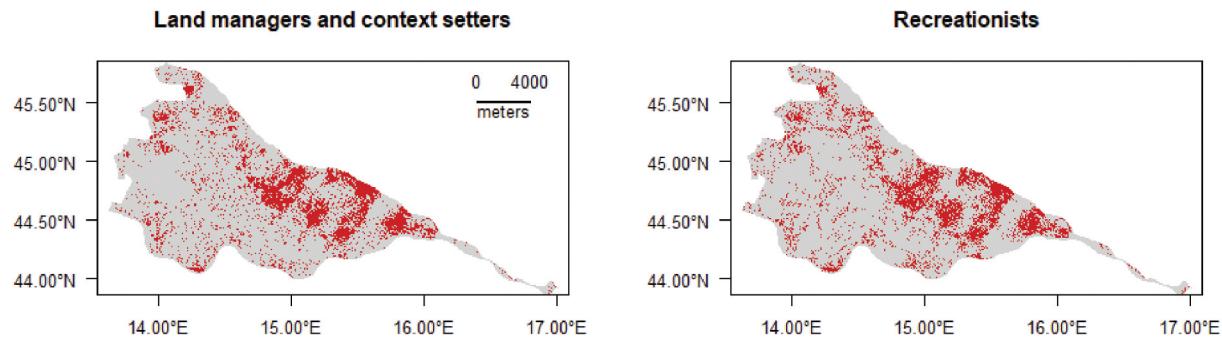


Figure 6. Stakeholder weighted multifunctionality hotspots, defined as the 80–100% quantile (red) in the Kromme Rijn area. UTM coordinates are displayed.

S5.2). In the Kromme Rijn, hotspots for land managers or context setters and recreationists largely fell within the same land use classes. Minor differences between the groups were found for urban areas, water bodies and wetlands, where higher proportions of hotspots were found for recreationists (Fig. S5.3).

4. Discussion

4.1. Differences and similarities of stakeholders perspectives on multifunctionality

The survey results of this study show that there are some differences between the stakeholder groups in both case study areas. Divergent valuations of ecosystem services can generally be explained by the different needs and interests of stakeholder groups (Chan et al. 2012; Turkelboom et al. 2018). Farmers, for example, rely on the economic returns of agricultural production and therefore attribute higher values to services such as crop, dairy or fruit production than other stakeholder groups, as shown in this study (Figure 4). Environmental organizations, on the other hand, aim to preserve habitats, ecosystems and the services they provide. The environmental organizations of the Vereinigte Mulde thus valued habitat quality, pollination and climate regulation the highest. Decision makers attached great importance to the recreation potential, as they are likely interested in increasing the attractiveness of the area for tourists to strengthen the region's economy. Finally, the recreationists of both case study areas attached great importance to the recreation potential and other cultural ecosystem services, not for economic reasons, but to enjoy their leisure time.

Deviations regarding the valuation of ecosystem services' importance also existed *within* stakeholder groups, because personal experiences and self-interest shape individual values and interests in ecosystem services (Massenberg 2019). In our case, the farmers and foresters were the group with the most common interests, while other groups had more diverging interests. The level of intra-group variance may also

depend on the number of participants per group. In case of the Kromme Rijn, for example, we assume that the within group variance of the land managers and context setters would have been higher if more people had participated in the survey. Those who were willing to participate might be more likely to have a similar perception on nature than other potential respondents.

While we found some differences between the groups, the statistical tests also showed that there was a large degree of agreement regarding valuations of individual services. Our results can therefore only partly confirm previous research, which suggest that stakeholder groups are associated with distinct ecosystem service demand bundles (Martín-López et al. 2012; Darvill and Lindo 2015; Zoderer et al. 2019). Moreover, the averaged weightings per stakeholder group were generally high, suggesting that all stakeholder groups recognize the value of maintaining multiple ecosystem services in a landscape and suggesting an awareness of the different land uses and the resulting goods and services (García-Llorente et al. 2012). One reason for this shared appreciation of multifunctionality could be that the different stakeholders have been socialized within the same environment and culture, which has influenced and shaped common values and interests (Massenberg 2019). For example, the two landscapes studied here are very well known as recreational areas and for providing natural habitats, but not necessarily for timber production. This perception is strongly communicated by society and reflected in our study results: both recreation potential and habitat quality were highly ranked by all stakeholder groups. The low level of social interest in timber production, on the other hand, was also reflected in our results. In contrast, people who grow up in different landscapes might be exposed to different values and might develop diverging interests in ecosystem services. This can be seen in our study by the evaluation of cultural ecosystem services: In the Kromme Rijn, landscape aesthetics was the more important cultural ecosystem service, while in the Vereinigte Mulde recreation itself was valued higher.

Regarding the perception of trade-offs and synergies, Plieninger et al. (2019) found that residents of rural agricultural landscapes perceive ecosystem services as mostly synergistic and that important trade-offs are often not recognized in socio-cultural valuations. In our study, we also found that stakeholders generally attached high values to many ecosystem services. Especially regulating and cultural ecosystem services, which were mainly synergetic in both landscapes, were considered as equally important. However, the relatively lower valuation of provisioning services suggests that there is an awareness of the trade-offs that existed primarily between provisioning and other ecosystem services (Lee and Lautenbach 2016).

4.2. Assessing multifunctionality: with and without stakeholder weights

The results of the unweighted multifunctionality assessments largely coincided with those of other multifunctionality studies: a high diversity of ecosystem services was found in land use types that are used less intensively in our study regions, such as mixed forests and grasslands (e.g. Felipe-Lucia et al. 2014; Allan et al. 2015; Rodríguez-Loinaz et al. 2015; Hölting et al. 2019b); synergies were identified primarily between regulating and cultural ecosystem services, while trade-offs existed mainly between provisioning services and other service categories (e.g. Queiroz et al. 2015; Rodríguez-Loinaz et al. 2015). The application of the weighting schemes enabled a differentiation between stakeholders in terms of how much and where they benefit from ecosystem multifunctionality. For example, the greatest differences were found between decision-makers and farmers or foresters in the Vereinigte Mulde, with farmers and foresters receiving greater overall benefits from the landscape than the group of decision-makers. We conclude that the supply of ecosystem services better matched their demand for ecosystem services, especially in the North-Eastern part of the study region where multifunctionality hotspots were identified for this group of stakeholders. Stakeholder-weighted multifunctionality maps therefore contribute to a better understanding of which areas are particularly important for certain actors in a landscape. A differentiation of multifunctionality indicators, as suggested by Manning et al. (2018), is therefore particularly useful, when spatial data on ecosystem services is available and spatial mismatches can be identified. The application of multifunctionality indicators at landscape level, on the other hand, is particularly interesting when different landscapes are compared for the same stakeholder groups or when different scenarios (e.g. management regimes) of a landscape are

compared, which has not been the focus of this study (but see Manning et al. 2018).

4.3. Implications of this study

Identifying stakeholder-weighted multifunctionality hotspots has two important implications. First, it is relevant for landscape planning. Areas that are important for a wide range of stakeholders can be selected as priority areas for the conservation of ecosystem services (O'Farrell and Anderson 2010; Raudsepp-Hearne et al. 2010). We found that land use systems that are generally less intensively used, such as mixed forests and grasslands, were benefit hotspots for all stakeholder groups. Conserving ecosystem services in these areas should therefore be a priority management aim. Landscape changes that may have a negative impact on ecosystem services (e.g. road construction) should be avoided here in order to maintain benefits for a wide range of stakeholders.

Second, comparing the spatial patterns of stakeholder-weighted multifunctionality maps is important for managing potential land use conflicts (de Groot 2006; Darvill and Lindo 2015). When hotspot areas overlap but stakeholders prioritize different ecosystem services, land use conflicts may occur. In this study, we found a particularly strong overlap between the multifunctionality hotspots identified for recreationists and farmers or foresters of the Vereinigte Mulde who valued individual ecosystem services differently. Their diverging and potentially contradictory ideas about land use in these areas (e.g. recreationists using the forests to enjoy tranquility and foresters using them to produce timber) could end up in an exemplary conflict over the use of ecosystem services for which solutions must be found locally.

Due to the fact that there was a high degree of agreement between the stakeholder weighting schemes, the final multifunctionality maps showed great similarities. It was therefore quite difficult to pinpoint areas where people benefit more than others, or to identify areas where conflicts between stakeholders could arise. Participatory mapping approaches and complementary qualitative methods could enable a deeper understanding of potential areas of benefit or conflict (Chan et al. 2012; Fagerholm et al. 2012; Karimi and Brown 2017; Riechers et al. 2017). While in our study the differences between stakeholder groups were small, we expect more explicit differences between stakeholders' valuations of ecosystem services in larger areas, which are shaped by various cultures and values, and in less multifunctional landscapes, where people are less aware of the multiple benefits that nature provides (Vos and Meekes 1999). It would thus be interesting to apply this method in either larger or non-multifunctional landscapes.

5. Conclusions

Differentiating between multiple beneficiaries in landscapes is a useful strategy to take account of diverging interests that could potentially lead to land use conflicts that compromise sustainability goals. The identification of stakeholder-specific multifunctionality hotspots is thus relevant for planning and decision-making. Here we have also shown that the added value of multifunctionality is widely recognized by multiple stakeholder groups and that multifunctionality can be declared a common goal. Our findings apply to a variety of cultural landscapes in Central and Western Europe, especially those which are accessible and agriculturally productive (Schulp et al. 2019). These landscapes have been shaped by humans throughout history and are valued by society for the diversity of ecosystem service they provide (Vos and Meekes 1999). Many cultural and multifunctional landscapes are now at risk of losing important ecosystem services (van Berkel and Verburg 2014; Schulp et al. 2019). We conclude that the most important way to preserve the value of landscapes for a wide range of stakeholders is to raise general awareness of environmental goods and services. Non-influential stakeholder groups, non-marketable and intangible ecosystem services should no longer be overlooked. Instead, landscape management plans are needed that balance interests and focus on the protection of common goods (i.e. cultural and natural resources accessible to all members of society).

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Data availability

The data that support the findings of this study are currently available on request from the corresponding author, LH. Additionally, the spatial data will be made openly available within the GeoNetwork Catalogue of the UFZ within 3 months from the date of publication.

Disclosure statement

The authors declare no conflict of interest.

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References

- AdV. 2010. Digitales Basis-Landschaftsmodell (Basis-DLM) des Amtlichen Topographisch-Kartographischen Informationssystems [WWW Document]. Arbeitsgemeinschaft Vermess. Länder Bundesrepub. Dtschl <http://www.adv-online.de/Adv-Produkte/> (accessed 2019 Apr 1).
- Agrimatie. 2014. Agrimatie - informatie over de agrosector. [WWW Document]. [accessed 2020 Jan 23]. <https://www.agrimatie.nl/Binternet.aspx?ID=4&Lang=0>.
- Allan E, Manning P, Alt F, Binkenstein J, Blaser S, Blüthgen N, Böhm S, Grassein F, Hözel N, Klaus VH, et al. 2015. Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. *Ecol Lett.* 18(8):834–843. doi:[10.1111/ele.12469](https://doi.org/10.1111/ele.12469).
- Arnold JG, Moriasi DN, Gassman PW, Abbaspour KC, White M, Srinivasan R, Santhi C, Harmel D, van Griensven A, Liew MV, et al. 2012. SWAT: model use, calibration, and validation. *Trans ASABE.* 55(4): 1491–1508. doi:[10.13031/2013.42256](https://doi.org/10.13031/2013.42256).
- BWI. 2016. Thünen-Institut, Dritte Bundeswaldinventur – Ergebnisdatenbank. [WWW Document]. [accessed 2019 Jul 2]. <https://bwi.info>.
- Byrnes JEK, Gamfeldt L, Isbell F, Lefcheck JS, Griffin JN, Hector A, Cardinale BJ, Hooper DU, Dee LE, Emmett Duffy J. 2014. Investigating the relationship between biodiversity and ecosystem multifunctionality: challenges and solutions. *Methods Ecol Evol.* 5(2):111–124. doi:[10.1111/2041-210X.12143](https://doi.org/10.1111/2041-210X.12143).
- CBS. 2017. Basisregistratie Gewaspercelen (BRP) | data overheid. [WWW Document]. [accessed 2019 Dec 17]. <https://data.overheid.nl/dataset/52172-basisregistratie-gewaspercelen-brp->.
- CBS. 2018. StatLine - Fruitteelt; oogst en teeltoppervlakte appels en peren per regio, 1997-2017. [WWW Document]. [accessed 2020 Jan 23]. <https://opendata.cbs.nl/statline/#/CBS/nl/dataset/71509NED/table?fromstatweb>.
- CBS. 2019a. StatLine - Groenteteelt; oogst en teeltoppervlakte per groentesoort. [WWW Document]. www.CBS.nl. [accessed 2019 Dec 17]. <https://opendata.cbs.nl/statline/#/CBS/nl/dataset/37738/table?ts=1576493298218>.
- CBS. 2019b. StatLine - Akkerbouwgewassen; productie naar regio. [WWW Document]. [accessed 2019 Dec 17]. <https://opendata.cbs.nl/statline/#/CBS/nl/dataset/7100oogs/table?fromstatweb>.
- Chan KMA, Guerry AD, Balvanera P, Klain S, Satterfield T, Basurto X, Bostrom A, Chuenpagdee R, Gould R, Halpern BS, et al. 2012. Where are cultural and social in ecosystem services? A Framework for constructive engagement. *BioSci.* 62(8):744–756. doi:[10.1525/bio.2012.62.8.7](https://doi.org/10.1525/bio.2012.62.8.7).

- Copernicus. 2015. Imperviousness Density 2015 — copernicus land monitoring service. [WWW Document]. [accessed 2020 Jan 23]. <https://land.copernicus.eu/pan-eur/opean/high-resolution-layers/imperviousness/status-maps/2015>.
- Corbin JM, Strauss A. 1990. Grounded theory research: procedures, canons, and evaluative criteria. *Qual Sociol.* 13(1):3–21. doi:[10.1007/BF00988593](https://doi.org/10.1007/BF00988593).
- Core Team R. 2013. R: A language and environment for statistical computing. Vienna (Austria): R Foundation for Statistical Computing.
- Costanza R, Kubiszewski I, Ervin D, Bluffstone R, Boyd J, Brown D, Chang H, Dujon V, Granek E, Polasky S, et al. 2011. Valuing ecological systems and services. *F1000 Biol Rep.* 3. doi:[10.3410/B3-14](https://doi.org/10.3410/B3-14).
- Darvill R, Lindo Z. 2015. Quantifying and mapping ecosystem service use across stakeholder groups: implications for conservation with priorities for cultural values. *Ecosyst Serv, Best Pract Mapp Ecosyst Ser.* 13:153–161. doi:[10.1016/j.ecoser.2014.10.004](https://doi.org/10.1016/j.ecoser.2014.10.004).
- de Groot R. 2006. Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landsc Urban Plan, Landscapes sustainability.* 75(3–4):175–186. doi:[10.1016/j.landurbplan.2005.02.016](https://doi.org/10.1016/j.landurbplan.2005.02.016).
- Fagerholm N, Käyhkö N, Ndumbaro F, Khamis M. 2012. Community stakeholders' knowledge in landscape assessments – mapping indicators for landscape services. *Ecol Indic.* 18:421–433. doi:[10.1016/j.ecolind.2011.12.004](https://doi.org/10.1016/j.ecolind.2011.12.004).
- FAO. 2001. Food balance sheets - A handbook. Annex 1 Food Composition Tables. Rome: Food and Agriculture Organization of the United Nations.
- Felipe-Lucia M, Comín F, Bennett E. 2014. Interactions among ecosystem services across land uses in a floodplain agroecosystem. *Ecol Soc.* 19(1). doi:[10.5751/ES-06249-190120](https://doi.org/10.5751/ES-06249-190120).
- García-Llorente M, Martín-López B, Iniesta-Arandia I, López-Santiago CA, Aguilera PA, Montes C. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: an ecosystem service approach. *Environ Sci Policy.* 19–20:136–146. doi:[10.1016/j.envsci.2012.01.006](https://doi.org/10.1016/j.envsci.2012.01.006).
- Hazeu GW, Schuiling C, Dorland GJ, Van Oldengarm J, Gijsbertse HA. 2010. Landelijk Grondgebruiksbestand Nederland versie 6 (LGN6): vervaardiging, nauwkeurigheid en gebruik.
- Heijerman-Peppelman G, Roelofs PFMM. 2010. Kwantitatieve Informatie Fruitteelt 2009/2010 (Bloembollen, Boomwekerij & Fruit No. 2009–41), Praktijkonderzoek Plant & Omgeving. Wageningen: UR Wageningen.
- Höltung L, Beckmann M, Volk M, Cord AF. 2019a. Multifunctionality assessments – more than assessing multiple ecosystem functions and services? A quantitative literature review. *Ecol Indic.* 103:226–235. doi:[10.1016/j.ecolind.2019.04.009](https://doi.org/10.1016/j.ecolind.2019.04.009).
- Höltung L, Jacobs S, Felipe-Lucia MR, Maes J, Norström A, Plieninger T, Cord AF. 2019b. Measuring ecosystem multifunctionality across scales. *Environ Res Lett.* 14 (12):124083. doi:[10.1088/1748-9326/ab5ccb](https://doi.org/10.1088/1748-9326/ab5ccb).
- Holzkämper A, Bussani L, Cord AF, de Miguel Á, Jungandreas A, Karner K, Mitter H, Tabares DR, Schmid E, Schönhart M, et al., 2018. Indicators of ecosystem services and biodiversity and methods for evaluation (TALE No. 2.1), Towards multifunctional agricultural landscapes in Europe: Assessing and governing synergies between food production, biodiversity, and ecosystem services – TALE. Leipzig (Germany): UFZ.
- Iniesta-Arandia I, García-Llorente M, Aguilera PA, Montes C, Martín-López B. 2014. Socio-cultural valuation of ecosystem services: uncovering the links between values, drivers of change, and human well-being. *Ecol Econ.* 108:36–48. doi:[10.1016/j.ecolecon.2014.09.028](https://doi.org/10.1016/j.ecolecon.2014.09.028).
- Jungandreas A, Strauch M, Václavík T, Volk M, Cord AF, in review. Land sharing supports more endangered farmlands birds according to stakeholder-driven land-use scenarios.
- Karimi A, Brown G. 2017. Assessing multiple approaches for modelling land-use conflict potential from participatory mapping data. *Land Use Policy.* 67:253–267. doi:[10.1016/j.landusepol.2017.06.004](https://doi.org/10.1016/j.landusepol.2017.06.004).
- Karner K, Cord AF, Hagemann N, Hernandez-Mora N, Holzkämper A, Jeangros B, Lienhoop N, Nitsch H, Rivas D, Schmid E, et al. 2019. Developing stakeholder-driven scenarios on land sharing and land sparing – insights from five European case studies. *J Environ Manage.* 241:488–500. doi:[10.1016/j.jenvman.2019.03.050](https://doi.org/10.1016/j.jenvman.2019.03.050).
- Komossa F, van der Zanden EH, Verburg PH. 2019. Characterizing outdoor recreation user groups: A typology of peri-urban recreationists in the Kromme Rijn area, the Netherlands. *Land Use Policy.* 80:246–258. doi:[10.1016/j.landusepol.2018.10.017](https://doi.org/10.1016/j.landusepol.2018.10.017).
- Kremen C, Merenlender AM. 2018. Landscapes that work for biodiversity and people. *Science.* 362(6412):eaau6020. doi:[10.1126/science.aau6020](https://doi.org/10.1126/science.aau6020).
- Lee H, Lautenbach S. 2016. A quantitative review of relationships between ecosystem services. *Ecol Indic.* 66:340–351. doi:[10.1016/j.ecolind.2016.02.004](https://doi.org/10.1016/j.ecolind.2016.02.004).
- LfULG. 2008. 65E Vereinigte Mulde und Muldeauen (Gebietsbeschreibung FFH Gebiete in Sachsen No. EU-Meldenr.: 4340-302). Fauna-Flora-Habitat-Gebiete in Sachsen. Dresden: Sächsisches Landesamt für Umwelt, Landwirtschaft und Geologie.
- LfULG. 2015. Auswertekarten Bodenschutz 1:50.000 [WWW Document]. Karten GIS-Daten Zum Fachthema Boden. [accessed 2019 Mar 1]. <https://www.boden.sachsen.de/karten-wms-wfs-dienste-und-gis-daten-zum-fachthema-boden-19148.html>.
- MA. 2005. Millennium ecosystem assessment. Washington (DC): Island Press.
- Manning P, Van der Plas F, Soliveres S, Allan E, Maestre FT, Mace G, Whittingham MJ, Fischer M. 2018. Redefining ecosystem multifunctionality. *Nat Ecol Evol.* 2(3):427–436. doi:[10.1038/s41559-017-0461-7](https://doi.org/10.1038/s41559-017-0461-7).
- Martín-López B, Iniesta-Arandia I, García-Llorente M, Palomo I, Casado-Arzuaga I, Amo DGD, Gómez-Baggethun E, Oteros-Rozas E, Palacios-Agundez I, Willaarts B, et al. 2012. Uncovering ecosystem service bundles through social preferences. *Plos One.* 7(6): e38970. doi:[10.1371/journal.pone.0038970](https://doi.org/10.1371/journal.pone.0038970).
- Massenberg JR. 2019. Social values and sustainability: a retrospective view on the contribution of economics. *Sustain Sci.* 14(5):1233–1246. doi:[10.1007/s11625-019-00693-w](https://doi.org/10.1007/s11625-019-00693-w).
- Mastrandolo ME, Weyland F, Villarino SH, Barral MP, Nahuelhual L, Laterra P. 2014. Concepts and methods for landscape multifunctionality and a unifying framework based on ecosystem services. *Landsc Ecol.* 29:345–358. doi:[10.1007/s10980-013-9959-9](https://doi.org/10.1007/s10980-013-9959-9).
- Metzger MJ, Murray-Rust D, Houtkamp J, Jensen A, La Riviere I, Paterson JS, Pérez-Soba M, Valluri-Nitsch C. 2018. How do Europeans want to live in 2040? Citizen visions and their consequences for European land use. *Reg*

- Environ Change. 18:789–802. doi:10.1007/s10113-016-1091-3.
- MODIS. 2016. LAI data sets for land surface and climate modelling (HDF-EOS files, 2007-2016). [WWW Document]. [accessed 2019 Dec 17]. <http://globalchange.bnu.edu.cn/research/lai#download>.
- NBI. 2012. Database Zesde Nedersalndse Bosinventarisatie (NBI6) 2006-2012. [WWW Document]. [accessed 2019 Jul 2]. <Httpsprobosnlpublicatiesoverige1094-Mfv-2006-Nbi-2012>.
- NGR. 2019. Grondwater. Nationaal georegister – PDOK. [WWW Document]. Grondwater. [accessed 2019 Dec 17]. <http://nationaalgeoregister.nl/geonetwork/srv/dut/catalog.search#/metadata/4d4f5838-1eb2-4fe9-81cd-2ba37537a233; https://webkaart.provincie-utrecht.nl/viewer/app/Webkaart?debug=false&version>.
- Nitsch H, Rutz C, Mehic Z, Hagemann N, Holzkämper A, Romero MN, Schönhart M, van der Zanden EH, Wilaarts B. 2017. Policy analysis – report of work package 1 (TALE No. 1.1). Towards multifunctional agricultural landscapes in Europe: Assessing and governing synergies between food production, biodiversity, and ecosystem services – TALE. Leipzig (Germany): UFZ.
- O'Farrell PJ, Anderson PM. 2010. Sustainable multifunctional landscapes: a review to implementation. Curr Opin Environ Sustain. 2(1-2):59–65. doi:10.1016/j.cosust.2010.02.005.
- Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, et al. 2013. vegan: Community Ecology Package.
- OSM. 2019. Open street map. Planet dump. <https://planet.osm.org>. [WWW Document]. [accessed 2019 Mar 1]. <https://www.openstreetmap.org>.
- Overheid. 2006. Geluidbelasting - Contouren provinciale wegen 2000 | data overheid [WWW Document]. Open Data Van Overheid. [accessed 2019 Dec 17]. <https://data.overheid.nl/dataset/57528-geluidbelasting—contouren-provinciale-wegen-2000#panel-1>.
- Paracchini ML, Zulian G, Kopperoinen L, Maes J, Schägner JP, Termansen M, Zandersen M, Perez-Soba M, Scholefield PA, Bidoglio G. 2014. Mapping cultural ecosystem services: A framework to assess the potential for outdoor recreation across the EU. Ecol Indic. 45:371–385. doi:10.1016/j.ecolind.2014.04.018.
- Plieninger T, Torralba M, Hartel T, Fagerholm N. 2019. Perceived ecosystem services synergies, trade-offs, and bundles in European high nature value farming landscapes. Landsc Ecol. 34(7):1565–1581. doi:10.1007/s10980-019-00775-1.
- Queiroz C, Meacham M, Richter K, Norström AV, Andersson E, Norberg J, Peterson G. 2015. Mapping bundles of ecosystem services reveals distinct types of multifunctionality within a Swedish landscape. AMBIO. 44(S1):89–101. doi:10.1007/s13280-014-0601-0.
- Quintas-Soriano C, García-Llorente M, Norström A, Meacham M, Peterson G, Castro AJ. 2019. Integrating supply and demand in ecosystem service bundles characterization across Mediterranean transformed landscapes. Landsc Ecol. 34(7):1619–1633. doi:10.1007/s10980-019-00826-7.
- Raudsepp-Hearne C, Peterson GD, Bennett EM. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci U S A. 107(11):5242–5247. doi:10.1073/pnas.0907284107.
- Riechers M, Noack EM, Tscharntke T. 2017. Experts' versus laypersons' perception of urban cultural ecosystem services. Urban Ecosyst. 20(3):715–727. doi:10.1007/s11252-016-0616-3.
- Rodríguez-Loinaz G, Alday JG, Onaindia M. 2015. Multiple ecosystem services landscape index: A tool for multi-functional landscapes conservation. J Environ Manage. 147:152–163. doi:10.1016/j.jenvman.2014.09.001.
- Schmidt K, Walz A, Martín-López B, Sachse R. 2017. Testing socio-cultural valuation methods of ecosystem services to explain land use preferences. Ecosyst Serv. 26:270–288. doi:10.1016/j.ecoser.2017.07.001.
- Schulp CJ, Burkhard B, Maes J, Vliet JV, Verburg PH, Yue GH. 2014. Uncertainties in ecosystem service maps: a comparison on the European scale. Plos One. 9(10): e109643. doi:10.1371/journal.pone.0109643.
- Schulp CJ, Levers C, Kuemmerle T, Tieskens KF, Verburg PH. 2019. Mapping and modelling past and future land use change in Europe's cultural landscapes. Land Use Policy. 80:332–344. doi:10.1016/j.landusepol.2018.04.030.
- Schulp CJ, Nabuurs G-J, Verburg PH. 2008. Future carbon sequestration in Europe—Effects of land use change. Agric Ecosyst Environ. 127:251–264. doi:10.1016/j.agee.2008.04.010.
- Shaw, B. J., van Vliet, J., & Verburg, P. H. 2020. The peri-urbanization of Europe: A systematic review of a multi-faceted process. Landscape Urban Plan. 196:103733. doi:10.1016/j.landurbplan.2019.103733
- Simpson EH. 1949. Measurement of Diversity. Nature. 163:688. doi:10.1038/163688a0.
- Spake R, Lasseur R, Crouzat E, Bullock JM, Lavorel S, Parks KE, Schaafsma M, Bennett EM, Maes J, Mulligan M, et al. 2017. Unpacking ecosystem service bundles: towards predictive mapping of synergies and trade-offs between ecosystem services. Glob Environ Change. 47:37–50. doi:10.1016/j.gloenvcha.2017.08.004.
- STLA. 2017a. Statistischer Bericht C II 2 – j/17, Bodennutzung und Ernte im Freistaat Sachsen. (No. C II 2 – j/17). Kamenz: Statistisches Landesamt des Freistaates Sachsen.
- STLA. 2017b. Statistischer Bericht C I 3 - j/17, Anbau und Ernte von Gemüse und Erdbeeren im Freistaat Sachsen. Kamenz: Statistisches Landesamt des Freistaates Sachsen.
- Strauch M, Jungandreas A, Witing F, Schürz C, Volk M. 2017. Towards multifunctional agricultural landscapes in Europe: SWAT as a key to asses synergies and trade-offs between ecosystem services and biodiversity. Presented at the International SWAT Conference. Warsaw, Poland.
- Stürck J, Verburg PH. 2017. Multifunctionality at what scale? A landscape multifunctionality assessment for the European Union under conditions of land use change. Landsc Ecol. 32:481–500. doi:10.1007/s10980-016-0459-6.
- Tieskens KF, Van Zanten BT, Schulp CJ, Verburg PH. 2018. Aesthetic appreciation of the cultural landscape through social media: an analysis of revealed preference in the Dutch river landscape. Landsc Urban Plan. 177:128–137. doi:10.1016/j.landurbplan.2018.05.002.
- Turkelboom F, Leone M, Jacobs S, Kelemen E, García-Llorente M, Baró F, Termansen M, Barton DN, Berry P, Stange E, et al. 2018. When we cannot have it all: ecosystem services trade-offs in the context of spatial planning. Ecosyst Serv. SI: Synthesizing OpenNESS. 29:566–578. doi:10.1016/j.ecoser.2017.10.011.
- Van Berkel DB, Verburg PH. 2014. Spatial quantification and valuation of cultural ecosystem services in an agricultural landscape. Ecol Indic. 37:163–174. doi:10.1016/j.ecolind.2012.06.025.

- van Oudenhoven APE, Schröter M, Drakou EG, Geijzendorffer IR, Jacobs S, van Bodegom PM, Chazee L, Czucz B, Grunewald K, Lillebø AI, et al. 2018. Key criteria for developing ecosystem service indicators to inform decision making. *Ecol Indic.* 95:417–426. doi:[10.1016/j.ecolind.2018.06.020](https://doi.org/10.1016/j.ecolind.2018.06.020).
- Verhagen W, van der Zanden EH, Strauch M, van Teeffelen AJA, Verburg PH. 2018. Optimizing the allocation of agri-environment measures to navigate the trade-offs between ecosystem services, biodiversity and agricultural production. *Environ Sci Policy.* 84:186–196. doi:[10.1016/j.envsci.2018.03.013](https://doi.org/10.1016/j.envsci.2018.03.013).
- von der Dunk A, Grêt-Regamey A, Dalang T, Hersperger AM. 2011. Defining a typology of peri-urban land-use conflicts – A case study from Switzerland. *Landsc Urban Plan.* 101:149–156. doi:[10.1016/j.landurbplan.2011.02.007](https://doi.org/10.1016/j.landurbplan.2011.02.007).
- Vos W, Meekes H. 1999. Trends in European cultural landscape development: perspectives for a sustainable future. *Landsc Urban Plan.* 46:3–14. doi:[10.1016/S0169-2046\(99\)00043-2](https://doi.org/10.1016/S0169-2046(99)00043-2).
- Wiggering H, Müller K, Werner A, Helming K. 2003. The concept of multifunctionality in sustainable land development, in: sustainable development of multifunctional landscapes. Berlin (Heidelberg): Springer; p. 3–18. doi:[10.1007/978-3-662-05240-2_1](https://doi.org/10.1007/978-3-662-05240-2_1).
- Wolff S, Schulp CJ, Verburg PH. 2015. Mapping ecosystem services demand: A review of current research and future perspectives. *Ecol Indic.* 55:159–171. doi:[10.1016/j.ecolind.2015.03.016](https://doi.org/10.1016/j.ecolind.2015.03.016).
- Young J, Watt A, Nowicki P, Alard D, Clitherow J, Henle K, Johnson R, Laczkó E, McCracken D, Matouch S, et al. 2005. Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodivers Conserv.* 14:1641–1661. doi:[10.1007/s10531-004-0536-z](https://doi.org/10.1007/s10531-004-0536-z).
- Zoderer BM, Tasser E, Carver S, Tappeiner U. 2019. Stakeholder perspectives on ecosystem service supply and ecosystem service demand bundles. *Ecosyst Serv.* 37:100938. doi:[10.1016/j.ecoser.2019.100938](https://doi.org/10.1016/j.ecoser.2019.100938).
- Zulian G, Maes J, Paracchini ML. 2013. Linking land cover data and crop yields for mapping and assessment of pollination services in Europe. *Land.* 2:472–492. doi:[10.3390/land2030472](https://doi.org/10.3390/land2030472).