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Analysis of historic changes in regional ecosystem service provisioning using land use data

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ABSTRACT

Land use change has a major impact on goods and services that our environment supplies for society. While detailed ecological or biophysical field studies are needed to quantify the exact amount of ecosystem service supply at local scales, such a monitoring might be unfeasible at the regional scale. Since field scale monitoring schemes for ecosystem services or ecosystem functioning are missing, proxy based indicators can help to assess the historic development of ecosystem services or ecosystem functioning at the regional scale. We show at the example of the historic development (1964–2004) in the district of Leipzig/Germany how land use/land cover data can be used to derive regional scale indicators for ecosystem functions. We focus thereby on two hypotheses: (1) the ecosystem functioning has degraded over time and (2) changes in land use configuration play an important role in this degradation. The study focuses on indicators for ecosystem functions related to (i) water purification by riparian buffer strips, (ii) pollination, (iii) food production and (iv) outdoor recreation. Each indicator builds on the analysis of land use configuration and land use composition information and is tested on sensitivity/robustness with respect to parameters which had to be estimated based on expert knowledge. We show that land use composition is an important aspect in our ecosystem service assessment. Although our study region is faced with a maximum land use change of 11% in the major land use classes between 1964 and 2004, we see a decrease of ecosystem function indicators up to 23%. The regional assessment shows an overall trend for degradation of ecosystem functioning from 1964 to 1984. This trend is reversed between 1984 and 1994 but the process slowed down until 2004 without reaching the level of 1964.

1. Introduction

The environment supports human existence and human well-being, e.g. us, with a number of goods and services. These are for instance products like food, fiber and fuel but also services like water retention, pollination, pest control or cultural values. These goods and services are summarized as ecosystem services (cf. MA, 2005 for a widely used list of these goods and services). As these environmental processes get their value by the benefit people obtain from it (Díaz et al., 2006; MA, 2005) it is a clearly anthropocentric concept: Without a benefit there is no service. The provisioning of ecosystem services depends on ecosystem functions which can be defined as ‘the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly’ (De Groot, 1992). This definition treats ecosystem functions as a term referring to ecosystem properties alone, exclusive of ecosystem goods and services – other authors (e.g. Christensen et al., 1996) have defined the term in a broader sense that encompasses a variety of phenomena, including ecosystem properties, ecosystem goods, and ecosystem services (compare the discussion in Hooper et al., 2005, p. 7). In the following, we will use the term in the definition by De Groot, 1992. Since the values of ecosystem services are typically external to the valuation framework of decision-makers, suboptimal decisions and allocations of sparse resources may result. Assessments of ecosystem services or related ecosystem functions can help to incorporate the value of ecosystem services into decisions and help thereby to achieve decisions that can account of a range of ecosystem services and functions including their interrelations. At present, however, there is a lack of a sound methodology to measure various ecosystem services in a similar way as market goods and services, which can be taken from large number of publications on concepts and methods (e.g. Boyd and Banzhaf, 2007; Fisher et al., 2009; Nicholson et al., 2009; Wainger et al., 2010; Wallace, 2007). Therefore, it is important to develop methods and indicators that help quantifying these ecosystem services or related ecosystem functions. While process studies at the field and landscape level are important tools, methods for regional ecosystem service or ecosystem functioning assessments are an important complement. Due to the multi-functionality of the landscape ecosystem

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services, assessments should aim at studying several ecosystem services in parallel (Chan et al., 2006; De Groot, 2006; Nelson et al., 2009, Raudsepp-Hearne et al., 2010). To be cost efficient, regional assessments should build on indicators derived from available data. At the regional to global scale, the mapping of ecosystem services has been a major topic in the research community (Troy and Wilson, 2006; Eigenbrod et al., 2010). Quite often these ecosystem service mapping has been based on proxy-based maps (e.g. Costanza et al., 1997; Sutton and Costanza, 2002; Chan et al., 2006; Troy and Wilson, 2006; Egoh et al., 2008). Eigenbrod et al. (2010) distinguish thereby between land cover based proxies and proxies based on logical combinations of likely causal variables. Land cover based ecosystem service mapping approaches frequently use the benefit transfer approach. This approach estimates values for ecosystem services at one site by transferring estimates from other sites. The method can be understood as a lookup table approach: values are once assigned to objects with specific characteristics and later used to assign values for objects with similar properties. Values are assigned to land use or land cover classes as a result of one or more studies and are later used for the valuation of land use or land cover classes in other regions (cf. Plummer, 2009 for a detailed discussion). A famous example for the valuation of land use classes is the work of Costanza et al. (1997) who performed an expert based valuation of ecosystems at the global scale, which was discussed intensely (Bockstael et al., 2000; Costanza et al., 1998; Toman, 1998). A couple of studies use these values to assess ecosystem services at regional scales (e.g. Hu et al., 2008; Kreuter et al., 2001; Li et al., 2007) which seems questionable – the transfer of regional valuation studies (e.g. Troy and Wilson, 2006) seems more reliable. Other approaches to map ecosystem services based on land use or land cover data use the results of meta-analysis (cf. Nelson and Kennedy, 2009; Lindhjem and Navrud, 2008; Bergstrom and Taylor, 2006). Proxies based on logical combinations of likely causal variables are another way of estimating and mapping ecosystem services (e.g. Chan et al., 2006; Egoh et al., 2008; Lonsdorf et al., 2009; Önal and Yanprechaset, 2007; Tallis et al., 2008; Troy and Wilson, 2006). Willemen et al. (2008) and Willemen and Hein (2010) estimated landscape functions which can be related to a number of ecosystem functions and ecosystem services based on proxy variables. Regression models were developed for a part of these landscape functions (tourism, arable production and plant habitat) which allow the estimation of suitability maps.

The delivery of some of the ecosystem services is dependent on their spatial context (cf. Kremen et al., 2007; Schröder and Seppelt, 2006) – pollinators, e.g. need different habitat characteristics for nesting and foraging. Additionally, demand and supply of some of the services show a spatial dependency as well: having suitable areas for organisms relevant for pest control or pollination somewhere in the landscape is not necessarily similar to having suitable areas for organisms relevant for pest control or pollination nearby arable fields. Therefore, both spatial configuration and spatial composition of land use are important for regional ecosystem service assessments.

At present, there are only a few studies at the regional scale which study the development of multiple ecosystem services over more than two time steps and apply techniques different from very simple benefit transfer approaches – one example is Pederson et al. (2006). We aim at closing this gap by analyzing historic land use, taken the district of Leipzig, East Germany as example. Since important data for an assessment of the demand for ecosystem services are missing – especially for GDR times – we concentrate on ecosystem service related ecosystem functions instead. Nevertheless, we aim at including demand related factors there ever possible. We focus thereby on two hypotheses: (1) the supply of ecosystem services has degraded over time and (2) changes in land use configuration play an important role in this degradation.

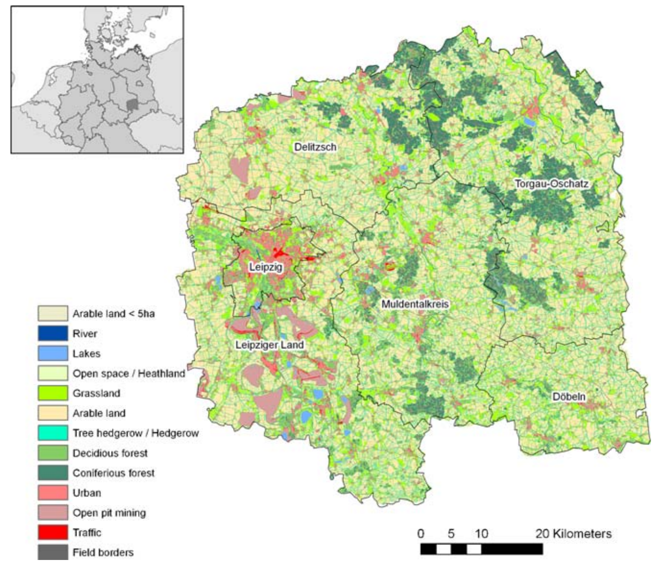

Fig. 1. The district of Leipzig is located in the northern part of Saxony (inset). The main map shows the land cover of 1994 a well as the 6 administrative units.

2. Data and methods

2.1. Case study

The study region, the District of Leipzig (cf. Fig. 1) comprises an area of approximately 4386 km^2. It is located in the Free State of Saxony, Eastern Germany and therefore in a temperate and relatively dry climate – mean annual precipitation 556.6 mm, mean annual temperature 10.0 °C (Institute for Meteorology of the University of Leipzig, 2008). While the relief is rather plain in the northern part of the district, the southern section, which encompasses most of the study area, is characterized by the foothills of the Erzgebirge mountain range. It is a predominantly rural and agricultural landscape. Agriculturally used land currently takes up 66% of the area, while forest cover is below German average. Brown coal mining has developed in the area since the 1920s. The open pit mining has lead to significant changes in the relief. Nowadays, the majority of the mining pits was flooded or is in the process of being flooded. Flooded mining pits are used as attractive recreation sites.

Central Europe has been an agriculturally dominated region for centuries. However, major changes in the structure of agriculture took place in the region within the last 60 years, caused in particular by the agricultural policy of the German Democratic Republic (GDR). The mid-20th-century ideal of modern agriculture – not only of the GDR – was to be realized by industrialized production methods, including mechanization, specialization, automatization, fortified application of chemicals, collectivization and large-scale structuring of farms (Schöne, 2005). Production units were redivided into smaller units in the mid-1980s, a trend that was further followed up after the two German states had been reunified in 1990.

District and city of Leipzig both show a decreasing population trend. There was no information available on the number of inhabitants in the area of the District of Leipzig before 1984. However, statistical data concerning the Free State of Saxony (Statistical Office of the Freestate of Saxony, 2008) show a constant decrease in population since 1955. It is assumed that the trend of population numbers in the District of Leipzig is consistent with the trend at the level of the Free State of Saxony, and has been decreasing between 1964 and 1984 as well. Particularly in the first years after the fall of the Berlin wall in 1989, the region suffered from a very high emigration rate – a process which left its traces until today.

Table 1
The 20 land use classes which underlie our analysis.

Lakes/rivers	Grass/rangelands	Urban
Watercourse Standing water body	Pasture Open space/heathland Woodland/single tree Tree hedgerow/hedgerow Arable field boundaries Grassland boundaries	Built-up land Special use area/open cast mine Interstate Country road/federal highway Other road Private road Other traffic area
Wetlands	**Temperate/boreal forests**	**Cropland**
Peatland	Deciduous tree dominated forest Coniferous tree dominated forest	Arable field Arable field <0.5 ha

Table 2
Overview about the indicators. See Table 1 in the appendix for pseudo-code of the calculations.

Indicator	Description and pseudo-code	Unit
Food production		
I_{Food}	sum of arable land cells within the two highest soil fertility classes	[m^2]
Water quality		
$I_{WQ,1}$	sum of the area of the buffer strip cells	[m^2]
$I_{WQ,2}$	sum of arable land in the region uphill of the buffer strips; uphill is defined as cells routing through the buffer strip cells	[m^2]
$I_{WQ,3}$	sum of arable land on slopes > 3 degree in uphill buffer strip region	[m^2]
$I_{WQ,4}$	potential erosion in uphill buffer stripe region, calculated using the RUSLE equation (Renard et al., 1994)	[t/a]
Outdoor recreation		
$I_{OR,1}$	sum of areas with suitable habitat in not noise polluted areas; only areas above a minimum area threshold are considered	[m^2]
$I_{OR,2}$	sum of areas with suitable habitat in not noise polluted areas which are accessible given a specific time constraint; only areas above a minimum area threshold are considered	[m^2]
Pollination		
$I_{Poll,1}$	Area of the potential nesting sites	[m^2]
$I_{Poll,2}$	Euclidian distance between potential nesting habitats and the nearest arable land cell	[m]
$I_{Poll,3}$	number of visitations of native pollinator on arable files, based on a k-nearest neighbor approach	[–]

2.2. Data

For the assessment of dynamic processes, multi-temporal data is required. Land cover data has been derived for the time steps of 1964, 1984, 1994 and 2004 by manual classification with the aid of auxiliary data (Horstkötter, 2003; Küster, 2003). It was based upon the Color Infrared Biotope and Land Use Mapping of Saxony 1992/1993 (LfULG, 1993) and was adjusted to the four time steps with the aid of satellite images (Corona – KH-4A, resolution 1.8 m × 1.8 m, panchromatic, May, 1965, Ruffner, 1995; USGS, 2008 and Landsat TM 5, 30 m × 30 m) and plane survey sheets Model (Cadastral Office Saxony, 1997 with a scale of 1:25,000). Furthermore, a digital elevation model (Cadastral Office Saxony, 1997) with a resolution of 20 m × 20 m was used. Hence, each land use data set is a blend of various sources.

After blending the different land cover data sources, the four data sets could be refined to a cell resolution of 10 m × 10 m. Land cover types were split up into 20 classes (cf. Table 1). For a part of the land cover types a reclassification as land use classes was possible, e.g. sealed areas were differentiated into built-up land, interstate highways, country road/federal highways, other roads, private roads and other traffic area–form hereon, the will therefore us the term land use/land cover to refer to our data. The high spatial resolution of the data allows the examination of parameters of landscape configuration, whose spatial dimension is located at the micro scale (Schulp et al., 2008), e.g. forested riparian strips and bee habitats adjacent to arable fields.

In addition, statistical data on population changes and yields was provided by the Statistical Office of the Free State of Saxony. GDR data was not always available, or was not available for the administrative units used in this study. We therefore consider it advantageous to use land use-data as a proxy, as it is not affected by political changes and shifts of administrative units.

2.3. Methods

Indicators for regional ecosystem services assessments used here are derived from expert knowledge of processes, using parameter values (or ranges) from recent publications supported by a sensitivity analysis. Table 2 provides a summary of the indicators, Table 1 in the online appendix includes pseudo code descriptions of how the indicators are calculated.

2.3.1. Water quality regulation

Water quality regulation consists of a set of processes: filtration, in-stream purification as well as erosion control (Brauman et al., 2007). We focus on the potential of the landscape to avoid diffuse emissions into the river network by riparian buffer strips. Buffer strips act as a sink for nutrients and other agro-chemicals like pesticides transported in surface runoff (Dorioz et al., 2006; Patty et al., 1997). Reviews about the effect of buffer strips show a remarkable variation on the effect size. The magnitude of the sediment retention is known to depend on the width of the buffer strip. Dosskey (2001) and Sabater et al. (2003) assume a reduction of nitrate loadings of 5–30% per meter width of a buffer stripe. A literature review by Castelle et al. (1994) indicated buffer strip widths from 3 m to 200 m to be effective, depending on site-specific conditions. A buffer width of at least 15 m was found to be necessary to protect wetlands and streams under most conditions. Buffer strips have been identified as non-agricultural and non-urban land use classes directly connected to the river system. This includes the following land use classes: pasture, open space/heathland, woodland/single tree, tree hedgerow/hedgerow, arable field boundaries, grassland boundaries, deciduous tree dominated forest, coniferous tree dominated forest and peatland.

A first step is the identification of buffer strip cells along the river network. Indicator I_{WQ1} sums the area of all buffer strip cells and identifies thereby the supply of the related ecosystem function. But this weights all cells equally while ignoring the variance in sediment and pollutant flow through the different cells. To estimate demand for water quality regulation we assumed that the demand is proportional to the upstream area of agricultural land and to the erosion potential estimated by the RUSLE (Revised Universal Soil Loss Equation; Renard et al., 1994). We quantified this effect by adding the contributing areas with agricultural land use, the contributing areas with agricultural land use on slopes higher three degrees and the estimated erosion potential. For the estimation of the actual sediment and pollutant retention one needs to estimate the uphill area of the buffer strip as well as the load origination in this uphill area. Analogue to water basins we calculated the uphill areas for the buffer strips using the D8 flow direction approach (Jenson and Domingue, 1988) with the 20 m × 20 m digital elevation model as the data source. By combining the flow-direction raster and the buffer strip cells we identified the contributing areas for the buffer strips. Indicator $I_{WQ,2}$ equals the sum of all arable land cells up stream of any buffer strip cell, derived from flow direction calculation. $I_{WQ,3}$ equals the area of arable land up stream with a slope larger than 3% and $I_{WQ,4}$ equals the sum of the potential sediment [t/a] of the upstream arable land cells derived from the universal soil loss equation (Renard et al., 1994).

2.3.2. Food production

Food production is one of the most evident ecosystem services. As land is the basic requirement for the cultivation of any crop, a region's potential for food production in agricultural areas depends directly on the fertility of available arable land. Conventional agricultural management includes the application of fertilizers, pesticides, fungicides and herbicides. However, the input of chemicals is of anthropogenic origin and therefore not considered a part of the ecosystem service food production. Rather, as food production depends heavily on the farmer's management actions like the application of fertilizer or pesticides as well as on ecosystem functions, the challenge for this service consists in separating one factor from the other. Soil fertility maps (LfULG, 2007) capture the integrated effect of local soil, relief and weather conditions and are therefore used as a proxy for the ecosystem service food production and its related ecosystem functions. To assess changes, we selected arable land cells for each of the four time steps and overlaid the soil fertility map with them. The indicator I_{Food} is defined as the area of arable land in the two highest soil fertility classes. This allowed us to keep track how the use of the environment to produced food has changed over time.

2.3.3. Outdoor recreation

Outdoor recreation is considered a cultural ecosystem service by the Millennium Ecosystem Assessment (MA, 2005). It values the function of ecosystems to provide opportunities for nature-based recreational activities, such as walking, bird-watching, camping, fishing, swimming, and nature study. The availability of outdoor recreation possibilities in an area is critically important to the well-being of its residents (Chan et al., 2006).

To evaluate the recreation services delivered by a region one has to consider the potential of the area to produce the ecosystem function which might be consumed as the outdoor recreation service (supply side) as well as the possibility for people to access the service (demand side). Correspondingly, our analysis is twofold: first, we assess the suitable outdoor recreation area and second we assess the accessibility of these suitable areas.

We defined suitable areas by two indicators. $I_{OR,1}$ is defined as the sum of natural or semi-natural land use type excluding noise polluted areas around streets, i.e. the areas that provide the ecosystem function. Buffer sizes around streets were defined based on estimated noise levels. The noise level at the streets was based on traffic census data (Kathmann et al., 2007). We assumed a distance-conditional level decrease of 4 dB(A) per distance doubling. We considered all values above the orientation value for residential areas of 55 dB(A) (German Industrial Standard DIN 18005) as not suitable for recreational. Traffic census data are only available for the time after the German reunification – traffic volume for the GDR time is expected to be lower than today but cars and pavement are expected to emit higher noise levels than nowadays. Given these uncertainties, we decided to treat noise buffers conservatively and applied the same buffer size for all time periods. Especially for 1964 this might lead to an underestimation of the areas suitable for recreation.

Since the recreational value increases with area size we analyzed the frequency distribution of suitable areas. Since small interruptions of suitable habitat are not of major concern for the recreational quality we connected nearby patches. This patch connection was only applied to areas outside the noise buffers.

$I_{OR,2}$ considers the accessibility of the potential outdoor recreation. It sums only the area of suitable habitat outside of noise polluted areas which are accessible given a specific time constraint. We used half an hour, on hour and one and half hours as thresholds. The accessibility was estimated based on cost distance surfaces. For the cost distance surface we assumed an average velocity based on the fastest available kind of movement – e.g. for streets we assumed movement by car, for dirt roads we assumed movement by bicycle and on other terrain we assumed walking. The average velocity on streets depends on the type of street as well as on the location of the street inside of a city or outside of a city.

Since the vast majority of the population of the study area resides in the city of Leipzig, we based our accessibility calculations on the cost weighted distance between Leipzig and the potential recreation areas.

2.3.4. Pollination

Pollination by insects helps to sustain and potentially increase the production of the majority of the global leading crops. While cereals do not profit from pollination, important fruit, vegetable, nut, spice, oil and stimulant crops profit from pollination (Klein et al., 2007).

The demand for the service pollination is generated by the decision of the farmer to plant crops which depend on or profit from pollination. In the district of Leipzig, important crops which profit from pollination are rapeseed and fruits like apples and strawberries. The supply with pollination depends on the amount of honey bees, wild bees and other pollinators which visit the arable fields or orchards. The majority of crops are most effectively pollinated by bees (Klein et al., 2007). The importance of wild bee conservation is of increasing concern, as extensive losses of honey bee populations have occurred in the last 20 years (De la Rua et al., 2009). Maintaining diverse wild bee communities is expected to stabilize and improve the delivery of pollination service (Greenleaf and Kremen, 2006; Hoehn et al., 2008; Klein et al., 2003; Winfree and Kremen, 2009). Since the distribution of honey bees depends on decisions by beekeepers, it is questionable how much of it can be considered as an ecosystem service; therefore, we will focus on pollination by wild insects.

The supply with pollination service by wild pollinators depends on the distribution of nests which can be estimated by the distribution of potential nesting habitats. To meet supply and demand for pollination we must consider the spatial configuration of nesting habitats and pollination dependent crops. Because no information is available on the mapping of crop types to arable land, we were not able to differentiate between different arable land use types.

Indicators for the ecosystem service pollination therefore have to consider the ecological demands of wild bees. Wild bees live solitarily in nests, which they burrow into the ground, dead wood or similar material (Westrich, 1996). Annually tilled arable fields are therefore not suitable as a wild bee nesting habitat. Besides nesting habitats, foraging habitats are also of concern. Wild bees cannot rely on only one species to forage on, as the blossom period of each plant species is limited in time (Svensson et al., 2000; Morandin et al., 2007). Besides grasslands and woody habitats, edge habitats, e.g. road sides, arable field boundaries or riparian strips, play a particular role, as they often host wild plant species and have a rich biodiversity. Other small-scale-habitats like single trees or hedgerows also serve as wild bee habitats, as they are usually not cultivated and therefore host wild herb species. It is known that landscape configuration, as well as landscape composition, has a substantial impact on the availability and the spatial distribution of insects in a region (Steffan-Dewenter and Tscharntke, 1999; Steffan-Dewenter, 2002, 2003) as well as on the pollination service (Kremen et al., 2007; Ricketts et al., 2008).

A first indicator $I_{Poll,1}$ measures the supply side of the service (the areas which provide the ecosystem function) by adding the

Table 3
Overview about the parameter values that have been changed during the sensitivity analysis. All combinations of the different parameter values have been studied.

ESS	Parameter	Description	Reference value	Values for sensitivity analysis
Recreation	Base buffer size [m]	The buffer size around interstate roads with the highest traffic load. These buffers are used to clip noise-polluted areas. The buffers around other roads are scaled relative to this value	1000	500, 750, 900, 1000, 1100, 1250, 1500, 2000
	Minimum area size [ha]	The threshold for the minimum area of suitable habitat. Smaller areas are not considered for outdoor recreation.	50	25, 50, 100
Pollination	beta	The decay parameter which describes how fast visitation probability decreases with distance between nesting and foraging habitat.	−0.00103	−0.00103, −0.00053
	knn	The number of nearest neighbors around each nesting habitat cell which are assumed to be pollinated.	30	20, 40
	maxSearchDist [m]	The maximum search distance considered for pollination visits.	500	200, 400, 500, 600, 750, 1000

size of potential nesting habitats. In addition, visitation probability can be used as an indicator for pollination service. We estimate the pollination service by calculating the visitation probability in two ways: (1) based on the distance between potential nesting habitats and all arable fields ($I_{Poll,2}$) and (2) the visitation probability based on the distance between potential nesting habitats and the k-nearest cells with foraging habitats ($I_{Poll,3}$).

Edge habitats were classified as potential nesting habitats. Specific edge habitats of arable fields and grassland were identified during the land use classification. Forest edge habitats have been identified by the r.le.tex command in the GRASS module re.le (Baker and Cai, 1992). The tool identifies edges by comparing the attributes of adjacent cells – if two adjacent cells contain different attribute values these are identified as edges. In our case, land use classes there used as attributes during the edge selection process. Arable fields and grasslands have been classified as potential foraging habitats. The distance d between potential nesting habitat and arable fields was calculated as the Euclidian distance between each arable land cell and the nearest nesting habitat cell. The indictor $I_{Poll,2}$ was defined as the sum all distances d between potential nesting and potential pollination habitat.

This simple distance based approach does not consider that bees do not fly unnecessary way but forage as close as possible to their nesting habitat. Therefore, we modified the approach by considering only a limited number of foraging cells for each nesting habitat cell. We used a self implemented k-nearest neighbor approach to select the foraging cells closest to each habitat cell. Again, the distance between nesting cell and foraging cell was calculated as the Euclidian distance d. To transform the distance into a visitation probability, we used an exponential decline in distance between nest and arable field as a functional relationship (Ricketts et al., 2008). The mean decline parameter beta for temperate studies from the Bayesian analysis of Ricketts et al. (2008) was used to parameterize the exponential decay model – we used the average decay rate for temperate climates as the decay parameter beta. For calculating the indicator $I_{Poll,3}$, the visitation probability from all nesting habitats to arable fields are summed up.

To speed up execution time we limited the search space to a reasonable distance maxSearchDist (cf. Table 1 in the online appendix for a pseudo-code implementation). The number of neighbors considered *knn* as well as the maximum search distance *maxSearchDist* has been chosen under incorporation of preliminary results from field observations (Bernd Gruber and Jeroen Everaars, personal communication). A further speed up was obtained by employing the obvious parallelization potential of the algorithm by distributing the calculations based on groups of habitat cells. The parallelization was performed using the mpi4py library for Python 2.6 (http://mpi4py.scipy.org/).

2.3.5. Sensitivity analysis

To test the robustness of the approaches we applied a sensitivity analysis to the calculation of pollination and recreation (cf. Table 3 for an overview of the parameters studied). The parameter ranges there in most cases chosen as 50% and 200% of the mean value which was selected based on expert knowledge. Exceptions are the number of nearest neighbors, for which it did not seem appropriate to test such large ranges, as well as the decay parameter beta for which two values given (Ricketts et al., 2008) were used. The larger beta value is the decay parameter estimated for all wild bees while the smaller beta represents the decay parameter estimated based on studies on wild bees in temperate climates. We changed all parameters for each of the two indicators in combination, i.e. we did not change one parameter at once but looked at the effects of combined parameter changes. Since we are interested in the robustness of the results we focused on the effect of parameter changes on the differences of the indicator between periods. This sensitivity has then been compared with the differences between the periods. In other words, we looked at how much the trend over time was influenced by changes in parameter values.

3. Results

3.1. Land use composition

Land use composition has changed moderately since 1964 (cf. Fig. 2 and Table 4). In general, changes from 1994 to 2004 are rather small. While arable land has declined over time, grassland and urban areas have increased from 1964 to 1994 while forest areas have been more or less stable. The area covered by open mining pits has increased by approximately 66 percent between 1964 and 1984 but has been reduced afterwards. Abandoned open

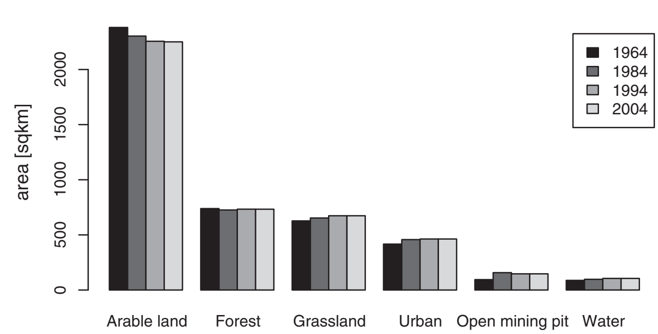

Fig. 2. Change of land use composition in the district of Leipzig.

Table 4
Relative change in land use classes between the time periods.

	64–84 [%]	84–94 [%]	94–04 [%]	64–04 [%]
Arable land	−3.3	−2.1	−0.2	−5.5
Forest	−1.7	1.0	0.0	−0.8
Grassland	4.2	3.1	0.0	7.4
Urban	9.8	1.2	0.0	11.1
Open mining pit	66.9	−6.8	0.0	55.5
Water	11.1	8.2	0.0	20.2

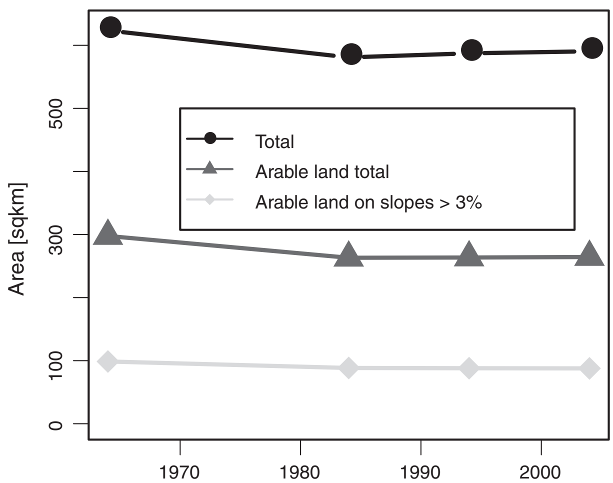

Fig. 3. Development of uphill buffer strip areas in the district of Leipzig ($I_{WQ,1}$, $I_{WQ,2}$, $I_{WQ,3}$).

mining pits have been flooded, thus increasing the lake area in the region.

3.2. *Indicators tailored for ecosystem service/ecosystem function assessment*

All results show an overall trend for degradation ecosystem functioning from 1964 to 1984 which is reversed between 1984 and 1994 and slowed down till 2004 without reaching the level of 1964.

3.2.1. *Water quality regulation*

The upstream area of buffer strips decreased by 7% between 1964 and 1984 and increased by 1.5% afterwards (cf. Fig. 3). The arable land in these upstream areas has decreased by 12% in the first period and has remained at this level afterwards. The same general trend can be detected for the development of arable land on slopes with an inclination over 3% for which a higher erosion potential can be assumed. A similar trend is gained by the analysis of the potential erosion classes generated by the RUSLE (cf. Fig. 4): The decrease of arable land in the upstream areas of the buffer stripes is highest for areas with a low erosion potential.

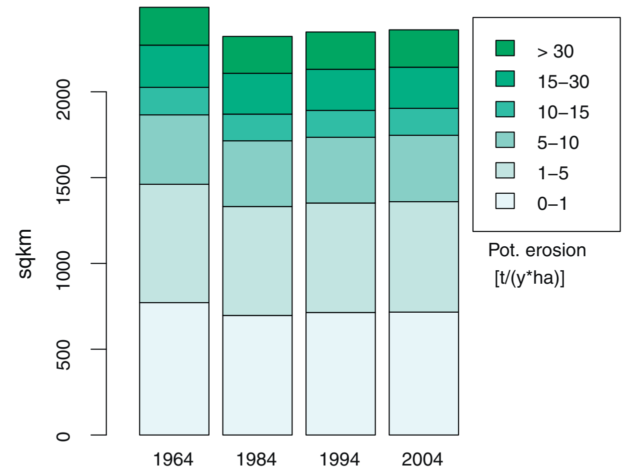

Fig. 4. Spatial distribution of the uphill buffer strip areas in zones of different potential erosion ($I_{WQ,4}$). The erosion potential $[t/(ha \times a)]$ has been calculated by the RUSLE.

3.2.2. *Food production*

The area used as arable land has decreased by around 4% from 1964 to 2004. This is partly due to abandonment of farming on low quality soils but urban sprawl has lead to a loss of high quality soils as well (cf. Table 5). Small fields have been abandoned or pooled between 1964 and 1984 – afterwards a slight increase of small fields can be detected (statistical data of the state of Saxony). Higher efficiency on larger fields along with technological progress has lead to an increase in productivity (statistical data of the state of Saxony) which in term resulted in a higher food production. But this should not be confused with an increase in the ecosystem service food production since the ecosystem service has to exclude effects of fertilizer or pesticide application, human workforce and other non-environmental production factors. Thus, while the produced quantity of food has increased, the corresponding ecosystem service has slightly declined, given the decline in arable land.

3.2.3. *Outdoor recreation*

Suitable outdoor recreation area has decreased from 1964 to 1984 and increased afterwards without reaching the 1964 level (cf. Fig. 5). Between 1964 and 1984 mainly small patch sizes have been lost while bigger patches seem to have stayed mainly untouched. New roads are an important factor determining the loss of potential outdoor recreation area – on the other side they allow better access to the remaining areas. Accessibility of potential outdoor recreation patches has increased between 1964 and 1984 (cf. Fig. 6).

3.2.4. *Pollination*

The distance between arable land and potential nesting habitats of wild bees has increased from 1964 to 1984 (cf. Fig. 7). The increasing field sizes lead to a loss of potential nesting habitats in the surroundings of arable fields. After 1984 the trend has reversed but the situation still has not returned to the situation of 1964. Still,

Table 5
Change of soil fertility class distribution on arable land in the district of Leipzig. The last row describes the total amount of arable land.

Soil fertility	1964		1984		1994		2004	
	km^2	% of 64	km^2	% of 64	km^2	% of 64	km^2	% of 64
Very low	4.05	100	3.64	89.8	3.65	90	3.64	89.8
Low	494.85	100	494.48	99.9	493.42	99.7	489.51	98.9
Average	366.21	100	340.43	93	337.80	92.2	337.39	92.1
High	1049.92	100	1027.31	97.8	1011.85	96.4	1010.35	96.2
Very high	392.18	100	387.55	98.8	383.83	97.9	382.09	97.4
Aggregated	2307.22	100	2253.4	97.7	2230.54	96.7	2222.99	96.3

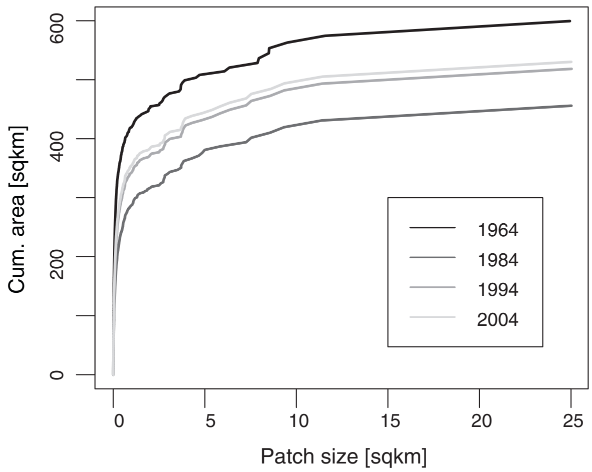

Fig. 5. Development of suitable outdoor recreation areas in the district of Leipzig ($I_{OR,1}$). The figure shows the cumulated area up to a specific patch size.

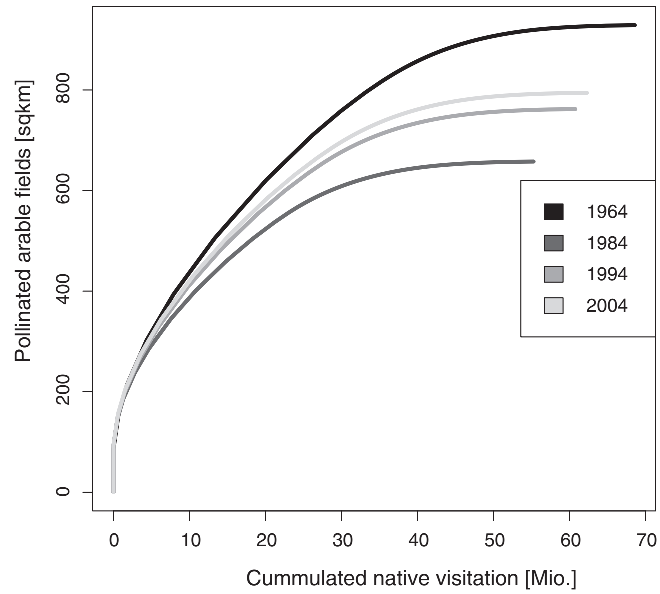

Fig. 8. Development of visitations by native pollinators on arable fields and the development of the pollinated area ($I_{Poll,3}$). See Section 2.3 for details.

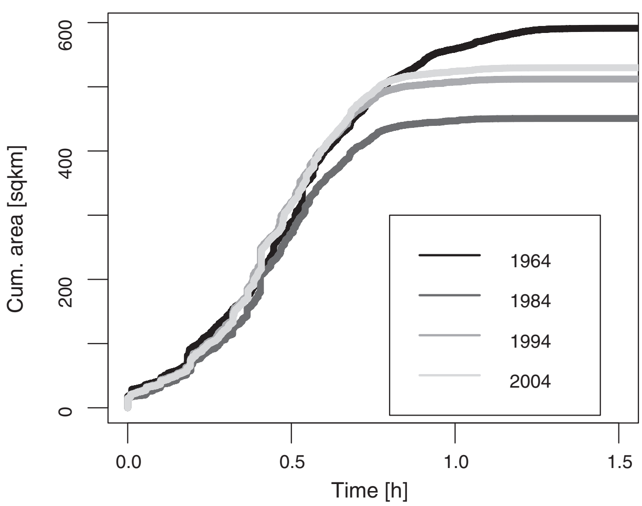

Fig. 6. Development of the accessibility to suitable outdoor recreational areas ($I_{OR,2}$). The lines show the cumulated area which can be reached from the city of Leipzig in a specified time.

the distances those wild bees would have to cover to visit the next arable field are within the known foraging distance. However, wild bees are expected to forage, and thereby pollinate, at the edges of fields – following the optimal foraging theory they should avoid unnecessary ways to optimize their foraging productivity (Pyke, 1978). The number of visitations by native pollinators as well as the area pollinated by them shows a strong decrease between 1964 and 1984 (cf. Fig. 8). Again, a recovery starts between 1984 and 1994 while this process slows down between 1994 and 2004 without reaching the initial situation.

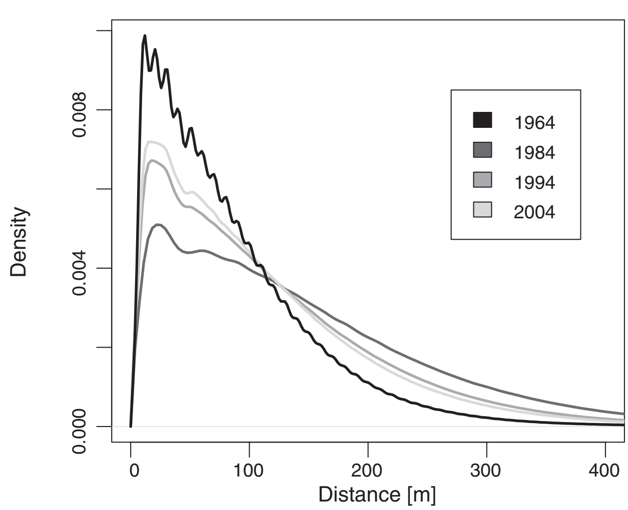

Fig. 7. Distribution of the distance between arable land and potential wild bee nesting habitats in the district of Leipzig ($I_{Poll,2}$). See Section 2.3 for details.

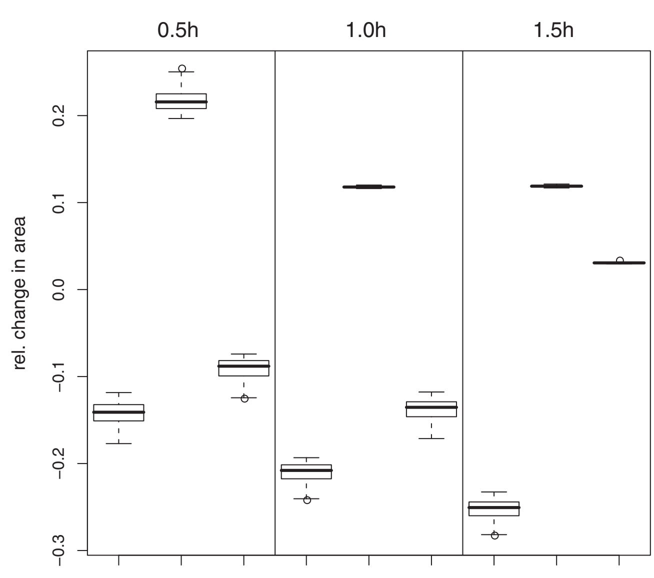

Fig. 9. Results of the sensitivity analysis for outdoor recreation ($I_{OR,2}$). The y-axis refers to the relative change of suitable outdoor recreation area which can be reached in 0.5 h, 1.0 h, 1.5 h. Each of the boxplots refers to results of all parameter combinations given in Table 2 – for each parameter combination; the relative change between two time steps has been calculated.

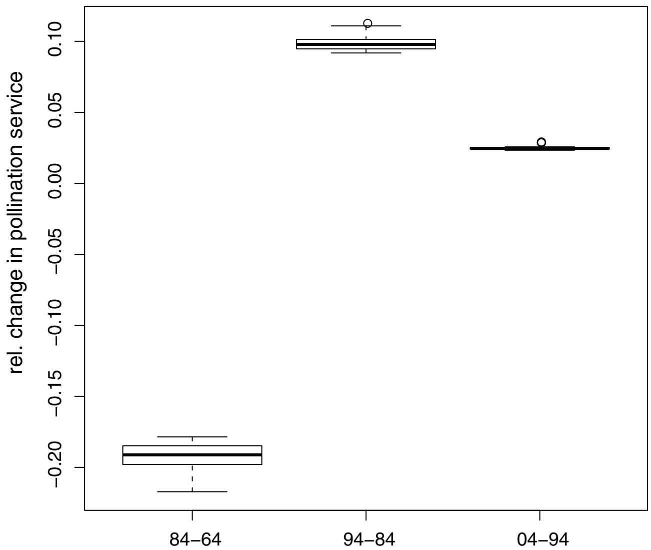

Fig. 10. Results of the sensitivity analysis for pollination ($I_{Poll,3}$). The y-axis refers to the relative change of visitations by natural pollinators on arable fields. Each of the boxplots refers to results of all parameter combinations given in Table 2 – for each parameter combination; the relative change between two time steps has been calculated.

3.3. Sensitivity analysis

The results of the sensitivity analysis for outdoor recreation (cf. Fig. 9) as well as for pollination (cf. Fig. 10) show that the trend in the ecosystem functioning related to both ecosystem services over time is much stronger than the effect of the parameter changes. While the parameter changes lead to changes in the absolute values of the ecosystem functions, the changes of the relative changes between the periods are negligible compared to the changes between the periods. We therefore consider the results as being robust.

4. Discussion

4.1. Development of ecosystem services/ecosystem functions over time

Given the results presented above, we still face a lower supply of the ecosystem services studied here as a result of changes in the landscape introduced in the 1960s (cf. Table 6). While we see a tendency after 1984 towards a general improvement of the considered ecosystem functions, the situation still has not recovered from the land-use changes from the agricultural industrialization in the 1960s. The rate of improvement observed between 1984 and 1994 decreased by 2004 to such an extent that it seems unlikely that the initial situation will be reached in the near future.

This trend is most obvious for pollination, where we observe a clear change for the worse regarding the spatial match of demand and supply. The trend for water quality regulation is less pronounced but still clear, while outdoor recreation shows a negative (loss of suitable areas) as well as a positive (increasing accessibility) tendency. The results for food production show a tendency towards the loss of fertile arable land but interpretation is rather difficult due to the effect of other production factors. If we compare the results of indicators which consider only the supply side of ecosystem service/ecosystem functioning ($I_{OR,1}$, $I_{WQ,1}$, $I_{Poll,1}$) with those that include the demand side as well, we detect significant differences in the magnitude of the effect, but the direction of the effect stays the same.

Agricultural intensification in its various aspects – e.g. land consolidation – is considered to have a strong negative impact on the performance of the ecosystem services/ecosystem functions assessed in this study. Agricultural intensification raised the demand for water quality regulation, even though the total area of arable land declined by 5.5% (1964–2004). Since detailed information on fertilizer or pesticide use was not available we had to focus our analysis on changes in land use composition like increasing field sizes. The observed decline of small scale habitats like habitat islands and edge habitats can be explained as a result of agricultural intensification and the resulting enlargement of field units. Compared to 1964, the area of small arable fields (<0.5 ha) in 2004 has decreased by 34.3%. We expect a related decline of field margins, which leads to a further reduction of the expected pollination potential of the landscape. Nevertheless, rapeseed – a field crop which profits from pollination services (Klein et al., 2007) – is increasingly cultivated in the district of Leipzig. On the other hand, agricultural intensification has increased food production intensity, resulting in higher total food production on a slightly smaller area.

Other important factors of land use change have been open pit mining, urban sprawl and road construction. Open pit mining led to an increase of standing water bodies and thereby outdoor recreation potential, but also favored the loss of fertile arable land and natural areas. Likewise, urban sprawl leads to a decrease of fertile arable land. Road construction had a mixed effect, leading to a decrease of potential outdoor recreation area but increasing the accessibility to the remaining areas.

Table 6
Summarized ecosystem service related indicators in the district of Leipzig. The values are always expressed as the difference between the first and the second period divided by value of the first period. The interpretation of the direction of the change depends on the type of indicator.

	64–84 [%]	84–94 [%]	94–04 [%]	64–04 [%]
Food production				
Arable land on fertile soils[a] I_{Food}	−1.9	−1.4	−0.2	−3.4
Outdoor recreation				
Pot. recreation area: $I_{OR,1}$	−24	+13.7	+2.3	−11.5
Pot. recr. area accessible in 0.5 h: $I_{OR,2}(0.5\,\text{h})$	−5.8	+18	−0.8	+10.3
Pot. recr. area accessible in 1 h: $I_{OR,2}(1h)$	−20	+13.5	+3.3	−6.3
Pot. recr. area accessible in 1.5 h: $I_{OR,2}(1.5\,\text{h})$	−23.8	+13.6	+3.5	-10.4
Water quality regulation				
Buffer strip area: $I_{WQ,1}$	−6.5	+1.3	+0.02	−5.2
Arable land in buffer strip upstream area: $I_{WQ,2}$	−11.6	+0.1	−0.3	−11.2
Arable land on slopes > 3% buffer strip upstream area: $I_{WQ,3}$	−10.4	−0.5	+0.3	−11.1
potential erosion : $I_{WQ,4}$	+0.8	+0.36	+2.8	+3.9
Pollination				
Nesting cell area: $I_{Poll,1}$	−29.5	+16.1	+4.3	−14.7
Distance between arable fields and nesting habitats: $I_{Poll,2}$	+66.1	−21.4	−7.1	+23.3
Visitation by native pollinators on arable fields: $I_{Poll,3}$	−19.4	+9.9	+2.5	−10.1

[a] Soil fertility classes high and very high.

Analyzing these trends raises the question of whether it is possible to compare ecosystem services over time. Since ecosystem services relate to the value society assigns to the goods and services produced by nature, the same delivery of service might be valued quite differently over time. If we think for example of honey bees, their most important service has changed from honey production to pollination. In our case the situation gets even more complicated since we need to assign services in two very different political and social systems: the GDR and the FRG. To avoid these problems, we focused on comparing the potential of the land-use pattern to provide ecosystem functions and refrained from assigning monetary values.

4.2. Methodological aspects and data issues

The four indicators have been applied to newly derived land use data. The number of land use/land cover classes used (20) per se is no limitation to the results of the study. Additional land use classes would have been helpful to distinguish between different field crops as well as between habitats of different quality for nesting and foraging of pollinators. But German data privacy protection laws make it nearly impossible to get information on actual crop rotation systems at a finer resolution than the NUTS 5 level. A potentially incorrect mapping of land use types which appear only over a very short term (e.g. conversion and re-conversion of a plot of land), could be largely prevented by using additional data sources from several years. Even so, visual interpretation errors of satellite images and aerial photographs can never be eliminated with absolute certainty.

The temporal scale of 40 years, split up into time steps of 10–20 years, is reasonable as consequences of changing management practices are likely to be visible in the landscape structure from one time step to another. In addition, this is a timeframe over which management decisions can make an impact on ecosystem functions.

Our perspective on water quality regulation is limited to the effect of buffer strips – in stream purification or purification during groundwater passage has not been considered. However, buffer strips have been shown to be an effective water quality management instrument (Ullrich and Volk, 2009). We can, therefore, expect to cover an important fraction of diffuse emissions. Our indicator works in the absence of data on fertilizer or pesticide application. This low data requirement allowed the application for historic conditions in which information on fertilizer application and pesticide use are absent. Generally, we view the potential of our indicator as a fast pre-screening tool for large areas which can be used to identify areas for further investigation. Our indicator could be used to parameterize process models like SWAT (Arnold and Fohrer, 2005) or SWIM (Krysanova et al., 2005).

The problems involved in the estimations for the food production result from the fact that food production involves ecosystem services in interaction with other agricultural production factors. The increasing productivity shows that farmers have been able to compensate for a moderate loss of fertile arable soils. A more advanced analysis of the development of the ecosystem service food production would involve the application of an agricultural production function under incorporation of fertilizer and pesticide application. Unfortunately, these data are again hard to get at a reasonable spatial resolution due to legislative restrictions in Germany.

The calculations involved in the indicator for outdoor recreation face two problems. First, we have no valid data for the noise levels during the GDR area and can therefore not adapt the buffer sizes accordingly. We used the conservative approach of transferring the buffer sizes from the most recent period. We do expect an even clearer decay of outdoor recreation otherwise. Second, we have to face the question of whether better accessibility implies more service or if it implies less service per area due to potential overcrowding. We can expect the relationship to be non-linear. Before a threshold is reached, more service will be delivered but after reaching the threshold decay of the service has to be expected. Our conclusions are drawn under the assumption that the threshold has not yet been reached. Given the decreasing population trend in the district of Leipzig, it seems reasonable that it will also not be reached in the coming years. Another approach would have been to use information on tourism. But the application of this approach to our case study region is problematic: While tourists do visit the district of Leipzig, the majority of the visits are motivated by cultural aspects. Tourists aiming at outdoor recreation are expected to visit the outstanding hiking and climbing possibilities of the Saxony Switzerland or the Erzgebirge mountain range 100–200 km away. This might change in the future since the local tourism industry is attempting to increase tourism by upgrading the water sport facilities on the flooded former open pit mining areas. But for the time being it can be considered safe to neglect the few tourists visiting for outdoor recreation reasons. Thus, we focus on the accessibility of the suitable areas for local outdoor recreation by the inhabitants. Still, it must be noted that accessibility is not the only property that should be taken into account to represent the demand side. Accessibility goes hand in hand with a higher intensity of use of the service which might in term reduce the benefit obtained per person. At least some inhabitants might favor a less crowed space over better accessibility.

For pollination we detected a clear decay of the potential of the landscape to supply the service. Whether the weakened relationship between arable fields and potential nesting habitats of wild bees really indicates a decay of the service depends on a number of additional facts: First, the spatial distribution of crops is unknown. Therefore, we cannot calculate the true number of potential nesting habitats surrounding fields which would profit from pollination. We could circumvent this problem by analyzing average, worst and best case scenarios of crop patterns. The best case scenario would distribute orchards and rapeseed as close to potential nesting habitats as possible, while the average would assume a random distribution, and the worst case scenario would place rapeseed as far away as possible from the potential nesting habitats. Second, the land cover data are not sufficient to decide whether a cell is really suitable for wild bee nesting – we do not know if dead wood is available, if the soil is suitable, etc. This information would also be valuable to estimate the wild bee density – to our knowledge bee density estimation is an unsolved question even in field experiments. This bee density information together with information about the correlation between visitation frequency and yield increase would allow a proper quantification of the nesting habitat to arable field relationship. Currently, we can only diagnose a decay of the relationship without assigning a proper value to it. However, even detecting this shift gives valuable hints for potential future conflicts and points to further research needs.

Our approach on estimating pollination services is an extension of the work by Lonsdorf et al. (2009) in that it considers that bees tend to forage in the neighborhood of their nesting sites if possible. In this sense, their work overestimates the pollinated area and the visitation by pollinators. But in contrast to our approach, Lonsdorf et al. (2009) distinguish between different pollinator species and use additional information about the distribution of foraging habitats to estimate the spatial pattern of pollinator abundance. Clearly, these are important issues which could be included in future versions of our pollination indicator if the necessary data become available.

A general problem of proxy-based approaches is the validation of their results. A validation would necessitate additional data. Water quality data would be available but the huge changes in East Germany after the wall came down make it nearly impossible to single out the effect of buffer strips – waste water treatment plant technology has changed, new treatment plants have been build, the percentage of connected inhabitants has increased, farming practices and ownerships of arable land have changed, and a large emigration together with a breakdown of industry have changed water quality significantly. Outdoor recreation could be validated by conducting a recreation-focused survey but this is beyond the scope of the presented work. For a validation of the pollination results, field work on nesting site distributions, bee densities or visitation rates at arable fields is necessary. Since these data are not available for our case study region, we have to rely on relationships based on expert knowledge or a review of the literature.

A first step towards the validation of the results is the analysis of their robustness. For the present work, we focused on $I_{Poll,3}$ (pollination) and $I_{OR,2}$ (outdoor recreation). We selected these two indicators because they depend on parameters that could not be estimated from observed data. Food production is a simple GIS operation with no obvious degrees of freedom which could be studied in a sensitivity analysis. Uncertainty in the underlying GIS data is clearly a source of uncertainty but their analysis is also beyond the scope of this paper. For water quality regulation most sensitivity can be expected to originate from the underlying digital elevation model as watershed calculations are highly sensitive to the terrain data. Because a Monte Carlo analysis would be extremely computationally demanding, we decided not to pursue the analysis for this indicator. While results for the two indicators studied for each time step differed during the sensitivity analysis, the trend remained stable – effects of parameter changes on the trend were negligible. We therefore consider the results as robust and to support our first hypothesis that ecosystem functions decreased over time in our case study region.

During our analysis we were able to detect developments that would not have been noticed by an analysis of the changes in land use composition alone. Although our study region is faced with a maximum land use change of 11% in the major land use classes between 1964 and 2004, we see a decrease in ESS indicators of up to 23%. Only our indicator for the development of food production is independent of configuration aspects. This supports our second hypothesis that an analysis of the decline of ecosystem functioning in our case study region has to consider land use composition effects in addition to land use composition effects.

The potential of the landscape to provide pollination services can only be assessed if the spatial pattern of nesting and foraging habitats is accessed. Likewise, the value of buffer strips for water quality regulation can only be estimated if the spatial pattern of diffuse emissions from agriculture is taken into account. Similarly, an assessment of outdoor recreation potential needs to consider patch size, effects of noise pollution and accessibility.

However, as not all ecosystem services depend on land use configuration it is not necessary to include configuration aspects in all studies. We expect services like soil formation, water supply regulation, timber production or carbon sequestration to depend primarily on land use composition. Assessments of these services obviously will not profit from a consideration of land use configuration aspects.

5. Conclusions and outlook

We derived land use/land cover data from satellite images and auxiliary data for four time steps and used a set of indicators to estimate how the changes between the different time steps relate to selected ecosystem functions. The aim of the study was the detection of trends over time not the exact quantification of ecosystem service provisioning or ecosystem functioning. The results show a clear tendency over time with land consolidation between 1964 and 1984 as a major – but not the only – driving force. The results depend on the analysis of land use configuration aspects – an analysis of land use composition would have drawn a different picture for three of the four ecosystem services that we studied. The sensitivity analysis indicated that the results are robust against parameter changes.

Nevertheless, the question remains how close the indicators get to reality. Eigenbrod et al. (2010) compared proxy-based mapping approaches with ecosystem services mapped from primary data and inferred significant biases caused by using proxy-based approaches. The land use/land cover based benefit transfer they use shows an especially high bias. Methods that used likely causal factors as proxies (like Chan et al., 2006) also suffered from departures from results based on primary data. Nevertheless, proxy-based assessments are considered beneficial since the high costs of primary data collection is likely to outweigh its benefits (Eigenbrod et al., 2010). In a similar manner, our work identified trends in ecosystem service provisioning based on spatial configuration. Due to missing data, a real validation is currently not possible. Instead, our work illustrates the data that would be needed from field studies to properly validate the results.

Since the previously described set of indicators is both computationally fast and not very data intense we see potential to use it in the context of land use optimization frameworks like LUPO (Holzkaemper and Seppelt, 2007) or to transfer it to available land use data sets like CORINE. Additionally, the indicators might be used in the context of decision support systems like FLUMAGIS (Volk et al., 2007) or the Elbe-DSS (Lautenbach et al., 2009).

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2010.09.007.

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