Land Use Policy 29 (2012) 521–535

Contents lists available at SciVerse ScienceDirect

Land Use Policy

journal homepage: www.elsevier.com/locate/landusepol

Rural–urban gradient analysis of ecosystem services supply and demand dynamics

Franziska Kroll[a,*], Felix Müller[a], Dagmar Haase[b,c], Nicola Fohrer[a]

[a] Christian Albrechts University of Kiel, Institute for the Conservation of Natural Resources, Ecology Centre, Olshausenstr. 40, 24098 Kiel, Germany
[b] Humboldt University of Berlin, Department of Geography, Rudower Chausee 16, 10099 Berlin, Germany
[c] Helmholtz Centre for Environmental Research – UFZ, Department of Computational Landscape Ecology, Permoserstr. 15, 04318 Leipzig, Germany

ARTICLE INFO

Article history:
Received 13 April 2010
Received in revised form 9 July 2011
Accepted 10 July 2011

Keywords:
Ecosystem services
Rural urban gradient
Land use
Corine land cover
Urban sprawl

ABSTRACT

Urban regions are important places of ecosystem service demands and, at the same time, are the primary source of global environmental impacts. Although there is broad agreement on the importance of incorporating the concept of ecosystem services into policy strategies and decision-making, the lack of a standardized approach to quantifying ecosystem services at the landscape scale has hindered progress in this direction. Moreover, tradeoffs between ecosystem services and the supply/demand ratio of ecosystem services in urban landscapes have rarely been investigated. In our paper, we present a method to quantify and map the supply and demand of three essential provisioning services – energy, food, and water – along the rural–urban gradient of the eastern German region Leipzig–Halle. This urban region has experienced significant socio-economic dynamics and land use changes since the German reunification in 1990. The results show that both the demand and the supply of ecosystem services changed considerably during the time span under consideration (1990–2007). We identified an increasing supply/demand ratio of food and water but a decreasing supply/demand ratio of energy. In addition, the pattern of ecosystem demands shows a levelling of rural–urban gradients, reflecting profound modifications of traditional rural–urban relationships. The changes of ecosystem service supply gradients are determined more by land use intensity, such as the intensification of agricultural production, than by land cover changes such as urban sprawl. The comparison of supply/demand ratios and rural–urban patterns of ecosystem services can help decision-makers in landscape management in striving for a sustainable balance between resource supply and demand.

Introduction

The integrative concept of ecosystem services

The development of strategies for sustainable managed landscapes requires a sophisticated understanding of social, economic and ecological processes, as well as of their mutual dependencies (Kates et al., 2001; Swart et al., 2002; McMichael et al., 2003). The concept of ecosystem services, defined as the benefits humans obtain from ecosystems (MA, 2005), fulfils this need through its integration of environmental and socio-economic concepts (Müller and Burkhard, 2007). Although broad agreement exists on a general understanding of the ecosystem service concept (Fisher et al., 2009), there is ongoing discussion about a universal definition as well as an overall classification system of such services (e.g., Boyd and Banzhaf, 2007; Wallace, 2007; Costanza, 2008; Wallace, 2008; Fisher and Turner, 2008; Fisher et al., 2009; TEEB, 2009). These studies discuss, among others, the distinction between services, functions and benefits (Boyd and Banzhaf, 2007; TEEB, 2009), structures and processes (Wallace, 2007), intermediate and final services (Boyd and Banzhaf, 2007; Costanza, 2008), and the need for one common classification system, rather than a multitude of competing classifications (Wallace, 2007; Costanza, 2008). One recent study proposes a common international classification of ecosystem goods and services (CICES) (Haines-Young and Potschin, 2010), taking into account previous classification systems developed by the Millennium Ecosystem Assessment (2005), United Nations etc. (2003) and TEEB (2009). The CICES classification also includes abiotic elements, such as mineral resources and wind, solar, and hydroelectric energy. In this paper, we follow the ecosystem service classification of CICES while assessing services at the landscape scale.

For the status assessment of ecosystem services, the needs and desired level of service provision of the society must also be

* Corresponding author. Tel.: +49 431 8801221; fax: +49 431 9904083.
E-mail addresses: fkroll@ecology.uni-kiel.de (F. Kroll), fmueller@ecology.uni-kiel.de (F. Müller), dagmar.haase@ufz.de (D. Haase), nfohrer@hydrology.uni-kiel.de (N. Fohrer).

0264-8377/$ – see front matter © 2011 Elsevier Ltd. All rights reserved.
doi:10.1016/j.landusepol.2011.07.008

considered; these are identified as the human demands for the services (Paetzold et al., 2010). Following the definition of Burkhard et al. (2011), the demand for ecosystem services is understood as the sum of all ecosystem goods and services that are currently consumed or used in a particular area over a given time period. The impacts on this demand are manifold, and include governmental policies, population dynamics, economic factors, marketing and advertising, cultural norms and governance features (Curran and de Sherbinin, 2004). In contrast to the demand for ecosystem services is their supply, defined as "the capacity of a particular area to provide a specific bundle of ecosystem goods and services within a given time period. Here, capacity refers to the generation of the actually used set of natural resources and services" (Burkhard et al., 2011, p. 2). The supply is directly determined by environmental resources and services, but is also influenced by human actions and decisions such as governmental policies and technical progress. In this regard, we consider the effects of human inputs (e.g., fertilizer, seeding, and the construction of power plants) as an inseparable part of the ecosystem service supply.

Ecosystem services in urban regions

As the majority of the human population is located in cities (UNPD, 2005), urban regions are the focal points of human ecosystem service demands and, simultaneously, the primary source of global environmental impacts (Jenerette et al., 2006b; Bai, 2007). Not only is the number of urban dwellers constantly increasing, but so too is the resource consumption per capita (Kennedy et al., 2007). This leads to an increasing dependency of urban regions on imported materials and goods such as water, food, and fuel (Decker et al., 2000). Although specific ecosystem services – namely the reduction of noise and air pollution, microclimate regulation, aesthetic and recreational services – contribute directly to quality of life in cities (Bolund and Hunhammar, 1999), the increasing imports of essential goods and services through globalization ensure that the growth of today's cities is independent from the ecological constraints of the cities' direct rural hinterlands (Rees, 1992, 1996; Luck et al., 2001). For instance, the monetary value of imported goods into Germany has tripled since 1990, with energy and food products being among those with the highest import increases (German Federal Statistical Office, 2010). However, cities still depend on their direct hinterland for the provision of water, the most important and largest material flux into urban ecosystems (Decker et al., 2000). For this reason and also because of the export of ecological degradation to remote regions, the comparison of ecosystem service demand in urban regions and the corresponding supply from their rural hinterlands provides important arguments for establishing sustainable policy strategies (Wackernagel et al., 2006).

Quantification methods of ecosystem services demand and supply

One of the first concepts to relate supply and demand with respect to ecosystem services was the ecological carrying capacity (Rees, 1992, 1996), which has been defined by Chambers et al. (2000) as the number of individuals of a given species that a defined habitat can support infinitely. Whereas the carrying capacity focuses on the resource provision of a given habitat, the concept of the ecological footprint represents a refinement of this idea, as it calculates the resource consumption of a population, expressed in units of global average hectares of biologically productive land and sea area (Rees, 1992; Wackernagel and Rees, 1996). This metric has evoked criticism because it does not take into consideration the spatial heterogeneity of resource provision of land and sea areas (Van den Bergh and Verbruggen, 1999; Opschoor, 2000; Templet, 2000; Anderson and Lindroth, 2001). Indeed, as the ecological footprint is calculated in global average hectares, the resource demand and supply can only be compared at the global level. Another criticism of the "ecological footprint" approach targets its aggregation of all ecosystem services, which renders it impossible to consider tradeoffs between the services (Van den Bergh and Verbruggen, 1999).

The monetary valuation of ecosystem services is a further attempt to indirectly relate the supply of ecosystem services to their demand (e.g., Costanza et al., 1997; Kreuter et al., 2001; Turner et al., 2003; Troy and Wilson, 2006; Petrosillo et al., 2009). However, there are several problems related to this method, foremost among which is the low or missing market value placed on most ecosystem services, particularly regulating and cultural services. The monetary value of most ecosystem services is determined by the amount and spatial distribution of the demand, which requires an examination of the spatial demand and supply patterns (McDonald, 2009). As a result, the spatial and temporal variability in the relationship between demand and supply is another challenge for the monetary valuation of ecosystem services (Limburg et al., 2002; Turner et al., 2003; Jenerette et al., 2006). For these reasons, several authors have stressed the urgent need for a non-monetary quantification of ecosystem services in physical service units, which should be used to underpin any monetary valuation (Boyd and Banzhaf, 2007; Burkhard et al., 2009; Vandewalle et al., 2009; TEEB, 2009; German Federal Statistical Office, 2009).

However, standardized approaches to a non-monetary quantification of ecosystem services – particularly their supply and demand – at the landscape scale are still missing. One problem that hinders the development of a standardized approach is the difficulty of reflecting spatial accuracy while still guaranteeing the comparability of different case studies.

Objective of this paper

Set against the preceding background, in this paper we present a method of ecosystem services quantification and mapping which aims at providing a generalization that is applicable to further case study regions. Although there might exist more elaborate quantification methods focusing on single ecosystem services, the target here is to find a balance between precision, broad applicability to a variety of landscapes, and adaptability to varying data availability. Only this general approach can provide comparable and standardized results and guarantee their use in the political process.

In this paper, the demand and supply of three ecosystem services in an urban region are directly compared by linking each to the regional land cover structure. The quantification of ecosystem services in physical units is made spatially explicit by incorporating land cover, land use intensity, soil, climate, and population data. In contrast to the ecological footprint approach, land serves here only as a proxy variable for the calculation of ecosystem services, and is not the measured variable itself. As a direct and easily understood linkage exists between the supply and demand of provisioning services, these are chosen for this study.

Three essential provisioning ecosystem services – water, food and energy – are calculated for three points in time: 1990, 2000 and 2007. This approach is taken first to show the temporal variations in the services, and second, to enable the visualization of tradeoffs between them. The eastern German urban region of Leipzig–Halle has been chosen as the case study region. This area has experienced severe socio-economic modifications and land use changes since the fall of the Berlin Wall in 1990 (Kroll and Haase, 2010). These changes, which will be described in more detail in the 'The case study' section, led to urban sprawl and an increasing urbanization of the former rural hinterland. In order to consider the impacts of urban sprawl, land use change, and socio-economic dynamics in the rural–urban continuum, the rural–urban gradient approach has been applied. Generally, rural–urban gradients

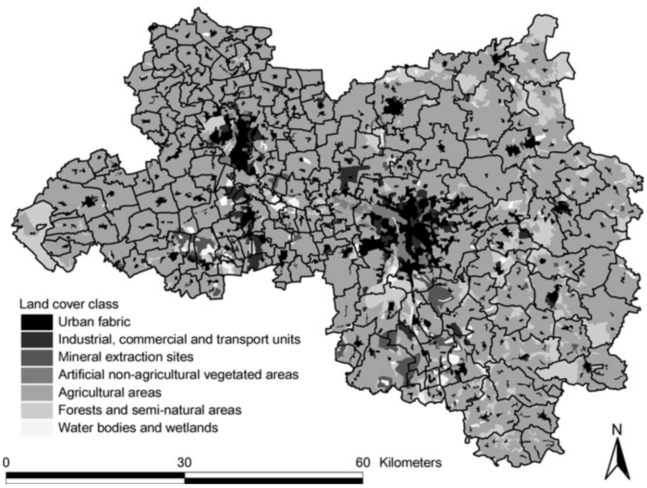

Fig. 1. CORINE land cover map (German Federal Environmental Agency, 2004) and municipality borders of the case study region Leipzig–Halle. The map is showing the pattern of the main CORINE land cover classes in 2000 and the administrative municipality borders in 2001.

are widely used in landscape science and ecology to describe spatial land use patterns (e.g., Foresman et al., 1997; Luck and Wu, 2002; Hahs and McDonnell, 2006) and ecosystem structure and functions in rural–urban regions (e.g., McDonnell and Pickett, 1990; Blair, 1996; McDonnell et al., 1997). We seek to combine both by developing rural–urban gradients of ecosystem services. Hence, the aim of this paper is to quantify and analyze supply and demand of three essential ecosystem services along a rural–urban gradient, and thereby to answer the following questions:

1. Is it possible to identify the impacts of socio-economic dynamics, land cover and land use changes on the rural–urban pattern of ecosystem service demand and supply?
2. Does ongoing urban sprawl reduce service provision in the rural hinterland?
3. Is the presented method for quantifying and mapping ecosystem services in physical units transferable to other regions and can tradeoffs be made visible?

The paper is structured as follows: in the next section, the case study region Leipzig–Halle is introduced. Next, the quantification and mapping methods for the energy, water, and food supply and demand are demonstrated. The quantification results are subsequently presented in the form of maps and rural–urban gradients. In the last section, the results are discussed in detail before conclusions are drawn.

Methods

The case study

Demographic and economic development

The case study region, measuring 4360 km^2 of size, comprises the two eastern German cities of Leipzig and Halle as well as their surrounding suburban and rural municipalities (Fig. 1). Being situated in the former German Democratic Republic, this region experienced a drastic deindustrialization (Nuissl and Rink, 2005) and lost a considerable part of its population during the post-socialist transition period after German reunification in 1990 (Banzhaf et al., 2009). The two cities, Leipzig and Halle, which together had comprise 870,621 inhabitants in 1990, lost approximately 15% of their population by 2007, whereas their suburban municipalities gained in population during the same period. The rural municipalities that are not directly adjacent to one of the two cities suffered from a severe population decline (Table 1). The emigration from core cities and rural areas was identified as a typical eastern German migration pattern after 1990 and has been linked to the decline of employment opportunities (Kroll and Haase, 2010). Nuissl and Rink (2005) and Couch et al. (2005) identify a number of reasons for the strong suburbanization of the early 1990s, which accompanied an overall decline in the population: the difficult restitution process of inner city buildings, the lack of efficient spatial planning instruments, fiscal subsidies, and finally, increasing household numbers due to a tendency towards smaller household sizes.

Even though the region lost the majority of its industrial jobs during the economic restructuring that took place after 1990, urban sprawl began with the construction of shopping malls and industrial estates on the green fields at the urban fringe (Nuissl and Rink, 2005), and the housing sector followed. In recent years, in contrast to Halle, the city centre of Leipzig has regained in population, particularly in the central neighbourhoods, driven by small and young households (Kabisch et al., 2010).

Land cover changes

The case study region is situated in a lowland area on fertile loess soils (mainly Cambisols, Luvisols and Chernozems), resulting in a high percentage of arable land (73% in 2000, see Table 2) and a rather small amount of forests and semi-natural area (10% in 2000, see Table 2) around the two cities. The region has a medium-continental climate (Haase, 2009) with an average annual precipitation ranging from 480 mm in the north-west to 750 mm in the south-east. The average annual temperature ranges from 7.5 °C to 10.0 °C (see Table 3 for data sources).

Fig. 1 shows the region's land cover pattern in 2000, when the peak of urban sprawl had already ended. The evidence of urban sprawl is indicated by the overall declining population that is not accompanied by a decrease in urban land consumption (cf. Kroll and Haase, 2010). All land cover changes are listed in Table 2, subdivided into urban, suburban and rural municipalities. It can be seen that industrial, commercial and transport land units increased the most in the suburban municipalities, followed by urban fabric, which exhibited a similar pattern of growth. Simultaneously, agricultural areas and mineral extraction sites declined. The latter have been replaced mainly by forests, semi-natural areas and large bodies of water.

The region of Leipzig–Halle is well-suited to be a case study region for the analysis of ecosystem service demand and supply dynamics, as it underwent massive societal, economic and demographic changes during the post-socialist transition period after 1990. These changes are presumed to have had a strong impact on the demand of ecosystem services in the region. In addition, the supply of ecosystem services is expected to have changed considerably due to the land cover changes described above.

Quantifying and mapping ecosystem services

The quantification and mapping of three ecosystem services, namely, the provision of food, water, and energy, has been carried out for three points in time: 1990, 2000 and 2007. 1990 reflects the situation shortly after the collapse of the German Democratic Republic and at the starting point of societal and economic transitions. The next time step, 2000, illustrates the effects of the intervening 10-year period of massive demographic and economic changes, and the resulting land use and land cover changes that accompanied them. The last time step, 2007, was chosen as representative of the most recent, "post-transition" developments, such as the ongoing but decelerating urban sprawl and reurbanization trends.

Table 1
Population development in the cities, suburban and rural surroundings of Leipzig and Halle taking the earliest time step from 1990 as 100%.

Year	City		Suburban municipalities adjacent to the city		Rural municipalities not adjacent to the city	
	Leipzig	Halle	Leipzig	Halle	Leipzig	Halle
1990	100	100	100	100	100	100
2000	88	80	118	135	96	96
2007	91	76	121	130	88	90

Data source: Statistical Office of the Free State of Saxony and Statistical Office of Saxony-Anhalt.

As the land cover pattern is one of the most important factors that affect a landscape's capacity to provide ecosystem services (Burkhard et al., 2009), we used land cover maps as a basis for quantifying and mapping ecosystem services supply and demand. For this purpose, we utilized the CORINE Land Cover maps of 1990, 2000 and 2006 (German Federal Environmental Agency, 2004, 2010). The European CORINE Land Cover mapping system includes 44 land cover classes (EEA, 1994) of which 24 occur in the Leipzig–Halle case study region. While the rather coarse scale of 1:100,000 represents a constraint on the data set when used at a regional level, its easy access, pan-European comparability, and availability for different points in time are substantial advantages.

As not every hectare of a given land cover type provides the same amount of ecosystem services, additional biotic and abiotic information were included in the analysis, including land use intensities, soil quality and climate data. For the calculation of ecosystem service demands, statistical data on population numbers and resource consumption of households, industry and commerce, mining, agriculture and forestry were applied. An overview of the data used is given in Table 3, and this topic is further explored in the subsequent methodology sections.

Quantifying and mapping energy supply and demand

The energy supply was calculated in GJ of final energy per hectare land cover type. First, the renewable and nonrenewable energy resources used for energy conversion purposes were identified and assigned to the corresponding land cover types. Although the provision of lignite as a subsurface asset is not considered to be an ecosystem service in the CICES study (Haines-Young and Potschin, 2010), it is included in the present study as it constitutes an important part of the natural capital of the Leipzig–Halle region and competes with other energy supplies.

Solar energy was assigned to urban, industrial and commercial areas, as we assume that solar power plants in the region have been mainly located on buildings during the time span under consideration. Information on the location, installed power and installation date of solar power plants was extracted from the website of the corporate energy provider Vattenfall Europe AG (Vattenfall Europe AG, 2009); the company publishes this information in accordance with the German Act on Granting Priority to Renewable Energy Sources. The same dataset also contains information about wind and water power plants. Due to the lack of adequate data, solar thermal power plants could not be considered in this study. Energy provision per hectare was finally calculated at the municipality level for each point in time as follows:

$$\frac{P_{\text{solar}} \times H \times 3.6}{A_{\text{urban+industry}}} = \frac{E}{A_{\text{urban+industry}}} \tag{1}$$

The symbols, corresponding parameters and units used in the equations in the 'Quantifying and mapping ecosystem services' section are listed in Table 4.

Wind energy was assigned to arable land, since wind power plants are assumed to be mainly located on this land cover type. Again, wind power was calculated at the municipality level for each point in time:

$$\frac{P_{\text{wind}} \times H \times 3.6}{A_{\text{agriculture}}} = \frac{E}{A_{\text{agriculture}}} \tag{2}$$

As the CORINE Land Cover map does not consider rivers as such, an additional river map including information about the river width was used (see Table 3), so that river area could be incorporated into the formula for the calculation of water energy supply:

$$\frac{P_{\text{water}} \times H \times 3.6}{A_{\text{river}}} = \frac{E}{A_{\text{river}}} \tag{3}$$

Bio-energy provision was assigned to two land cover classes: arable land (bio-fuel crops) and forests (wood). As data on energy crop cultivation and wood cultivation do not exist at the local level, average values per Federal State were estimated using the data sources listed in Table 3. Consequently, it was necessary to assume that bio-energy cultivation on arable and forest land was equally distributed in each of the two Federal States

Table 2
Land cover pattern in % of the main CORINE land cover classes in the Leipzig–Halle region in 1990, and their changes expressed as Δ 1990–2000 and Δ 1990–2006 in %, subdivided into urban, suburban and rural municipalities.

	City (Leipzig and Halle)			Suburban municipalities adjacent to Leipzig and Halle			Rural municipalities not directly adjacent to Leipzig or Halle		
	~430 km^2			~700 km^2			~3230 km^2		
	% 1990	Δ % 1990–2000	Δ % 1990–2006	% 1990	Δ % 1990–2000	Δ % 1990–2006	% 1990	Δ % 1990–2000	Δ % 1990–2006
Urban fabric	33.7	+0.6	+0.7	6.8	+1.0	+2.1	5.6	+0.3	+0.3
Industrial, commercial and transport units	7	+1.2	+2.7	2.1	+2.2	+2.8	1.3	+0.3	+0.5
Mineral extraction sites	1.1	−0.9	−0.9	4.8	−1.4	−2.9	3.6	−1.2	−2.0
Artificial non-agricultural vegetated areas	9.4	±0.0	+1.3	0.8	±0.0	+0.3	0.1	±0.0	±0.1
Agricultural areas	44.3	−2.1	−5.1	78.9	−4.7	−6.0	78.3	−1.2	−1.9
Forests and semi-natural areas	3.8	+0.7	+0.6	6.1	+2.2	+1.6	10.5	+1.2	+1.5
Water bodies and wetlands	0.7	+0.5	+0.7	0.5	+0.7	+2.1	0.6	+0.6	+1.5

Table 3
Data used to quantify energy, water and food supply and demand.

Data	Time steps, time series stand	Scale	Publisher/source
Population			
Population number	1990, 2000, 2007	Municipality	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Administrative borders of municipalities	2001	Municipality	German Federal Statistical Office (GENESIS)
Land use			
CORINE Land Cover	1990, 2000, 2006	1:100,000	German Federal Environmental Agency (2004, 2010)
Soil			
Soil map of Saxony including usable field capacity and actual root depth	2007	1:200,000	Saxon State Department of the Environment and Geology
Soil survey map Germany No. CC 4734 Leipzig	2009	1:200,000	German Agency for Geology and Natural Resources Hanover
Soil fertility map of Saxony	2007	1:200,000	Saxon State Department of the Environment and Geology
Soil fertility map of Saxony-Anhalt	2008	1:200,000	Environmental Agency of Saxony-Anhalt
Climate			
Precipitation, Evapotranspiration (ETP) as grass reference ETP	1961–1990	1 km × 1 km	Environmental Agency of Saxony-Anhalt; German Meteorological Service (1999), Döring et al. (1995), Müller-Westermeier (1995, 1998), Müller-Westermeier and Kreis (2002)
Water			
Groundwater level	2001	1:200,000	Federal Department for Flood Water Protection and Water Management Saxony-Anhalt
River map	1978	1:1,000,000	German Federal Environmental Agency
Water consumption of households, services, industry, mining, agriculture	1990 (1991 in Saxony), 2000, 2007	Federal State	Environmental-Economic Accounting of the Länder, Statistical Office of the Free State of Saxony, Statistical Office of Saxony-Anhalt, Saxon State Department of the Environment and Geology (2004)
Energy			
Total area of bio energy crops (differentiated after crop types)	1990–2007	Federal State	Fachagentur für Nachwachsende Rohstoffe e.V.; Department of agriculture, forestry and horticulture Saxony-Anhalt; Ministry of Agriculture and Environment Saxony-Anhalt (2002, 2007), Saxon State Ministry of the environment and agriculture (2007)
Energy yield per ha crop type	2007	No scale	Institute for Energy and Environment (2007)
Wind, water and solar energy plants with installed power	1990–2007	Exact locations	Vattenfall Europe AG (2009)
Lignite extraction	1990–2007	Lignite extraction site	Saxon State Department of the Environment and Geology (2004), Saxon Upper Mining Authority (2002, 2007)
Energy consumption of households, services, industry, mining, traffic	1990 (1991 in Saxony), 2000, 2007	Federal State	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt; Environmental-Economic Accounting of the Länder
Energy consumption of agriculture and forestry	1991, 1999, 2003	Germany	German Federal Statistical Office: Environmental-Economic Accounting, Schmidt and Osterburg (2009)
Food			
Crop yield	1991, 2000, 2007	District	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Crop composition	1991, 1999, 2007	District	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Fruit yield, livestock, game, fish	1991, 2000, 2007	Federal State	Saxon State Ministry of the Environment and Agriculture; KTBL (2005)
Food consumption per person	1990, 2000, 2007	Germany	German Federal Statistical Office

(Saxony and Saxony-Anhalt) as an approximation. The calculation is expressed by the following formula:

$$\frac{A_{\text{energy crops}} \times E_{\text{crop}}}{A_{\text{agriculture}}} = \frac{E}{A_{\text{agriculture}}} \tag{4}$$

Lignite, which is extracted from several large, open cast mining sites, is intensively used and represents the only non-renewable energy source in the region. Although the extracted amount of lignite per hectare, as well as its energetic value, differ from site to site, an average value per hectare mineral extraction site was calculated, using the data sources listed in Table 3. An efficiency factor of 36% was assumed for the conversion of raw brown coal to final energy (Bundesverband für Braunkohle, 2010), leading to the following formula:

$$\frac{C \times E_{\text{coal}} \times 0.36}{A_{\text{mineral}}} = \frac{E}{A_{\text{mineral}}} \tag{5}$$

Energy demand was calculated in GJ of final energy per hectare of each land cover type, in order to be directly comparable with the energy supply. To accomplish this calculation, statistical data on the energy demand of households, services, industry, traffic, mining and agriculture were applied (see Table 3). The energy demand of households and services was assigned to the urban area and calculated on the municipality level as follows:

$$\frac{ED_{\text{pop}} \times POP}{A_{\text{urban}}} = \frac{E}{A_{\text{urban}}} \tag{6}$$

For the energy demand arising from industry (assigned to industrial and commercial areas) and transportation (assigned to road and rail networks, ports and airports), average values per Federal State were calculated due to the lack of data at the local level:

$$\frac{ED_{\text{industry}}}{A_{\text{industry}}} = \frac{E}{A_{\text{industry}}} \tag{7}$$

and

$$\frac{ED_{\text{traffic}}}{A_{\text{traffic}}} = \frac{E}{A_{\text{traffic}}} \tag{8}$$

For the energy consumption of arable land, pastures, orchards, berry patches, and mineral extraction sites, it was necessary to fall back on German average values (cf. Schmidt and Osterburg, 2009). The calculated energy demand and supply values per hectare of each land cover type and, where available, per municipality or Federal State, were then linked to the CORINE Land Cover maps of the three different points in time.

Table 4
Parameters, their units and symbols used in the equations for the ecosystem service supply and demand calculations in the 'Quantifying and mapping ecosystem services' section.

Symbol	Name of parameter	Unit
P_{solar}	Installed power of solar energy plants	MW
P_{wind}	Installed power of wind energy plants	MW
P_{water}	Installed power of water energy plants	MW
H	Full load hours	hours
$A_{urban+industry}$	Urban, industrial and commercial area	hectare
A_{urban}	Urban area	hectare
$A_{industry}$	Industrial area	hectare
$A_{traffic}$	Traffic area	hectare
$A_{agriculture}$	Agricultural area	hectare
A_{river}	River area	hectare
$A_{energy\ crops}$	Area of bio energy crops	hectare
$A_{mineral}$	Area of mineral extraction sites	hectare
E	Final energy	GJ
W	Water	m^3
F	Food	GJ
E_{crop}	Final energy produced per crop type and hectare	GJ/hectare
E_{coal}	Energetic value of brown coal	GJ/tonnes
C	Brown coal extracted	tonnes
ED_{pop}	Final energy demand per person	GJ
$ED_{industry}$	Final energy demand of industry	GJ
$ED_{traffic}$	Final energy demand of traffic	GJ
WD_{pop}	Water demand per person	m^3
$WD_{industry}$	Water demand of industry	m^3
WD_{mining}	Water demand of mining	m^3
$WD_{agriculture}$	Water demand of agriculture	m^3
FD_{pop}	Food demand per person	GJ
POP	Population number	–

Quantifying and mapping water supply and demand

The groundwater recharge was chosen as an appropriate indicator to represent the ecosystem service water supply in the case study region. In order to calculate it, a method was required that is sensitive to the impacts of land cover changes on the water supply. The TUB-BGR approach (Wessolek et al., 2004) was chosen since it fulfils this requirement and is also applicable to easily accessed data. The output of this method is the mean annual percolation rate, meaning that it includes groundwater recharge as well as interflow. The following input data were used:

average corrected annual precipitation (mm),
average corrected precipitation in summer (mm),
average potential evapotranspiration (ETP) as grass reference ETP (mm),
groundwater level (dm),
usable field capacity (mm),
actual root depth (dm).

Based on these input variables, the TUB-BGR approach calculates the mean annual percolation rate ($mm/m^2\ a^{-1}$) using regression functions for arable land, grassland, broad-leaved forest, coniferous forest, and mixed forest. Hence, the CORINE Land Cover classes had to be reclassified into these five categories. For water bodies, urban fabric, industrial, commercial and transport units, no percolation was assumed as an approximation, although the sealing rate of artificial surfaces in the region varies between 40 and 60% for large housing estates, 60–80% for 1990s housing estates and older villages, and 80–100% for centres, commercial space and roads (Haase and Nuissl, 2007). Therefore, although infiltration and groundwater recharge is drastically lower in urban areas compared to other areas (Decker et al., 2000), the groundwater recharge for these areas is slightly underestimated in this study. For a detailed analysis of the relation between groundwater recharge and percentage of sealed surface in Leipzig, see Haase (2009). The calculated water supply was then mapped in m^3 per hectare with a 1 km × 1 km resolution. Negative values were set to zero in the supply map, but were incorporated into the demand map.

Water demand was also calculated in m^3 per hectare of land cover type. The groundwater consumption of the vegetation by capillary rise was taken from the produced percolation map (negative percolation rates). For the water consumption of households, services, industry, mining and agriculture, statistical data were applied (Table 3). Water demand of households and services was assigned to urban areas and calculated at the municipality level, applying the following formula:

$$\frac{WD_{pop} \times POP}{A_{urban}} = \frac{W}{A_{urban}} \tag{9}$$

For the water demand of industry, mining and agriculture, average values per hectare for the whole region were calculated as follows:

$$\frac{WD_{industry}}{A_{industry}} = \frac{W}{A_{industry}} \tag{10}$$

$$\frac{WD_{mining}}{A_{mining}} = \frac{W}{A_{mining}} \tag{11}$$

$$\frac{WD_{agriculture}}{A_{agriculture}} = \frac{W}{A_{agriculture}} \tag{12}$$

Quantifying and mapping supply and demand of food for human consumption

Food supply was quantified in GJ per hectare of land cover type. This includes an additional qualitative aspect of the produced food in comparison to yields in tonnes per hectare. Statistical data on crop type and yields (wheat, rye, barley, oat, triticale, silage maize, fodder grass, rape seed, turnips, potatoes, apples, pears, prunes, cherries, and berries), as well as livestock, game and fish were employed to calculate average values of food production per unit of arable land, orchards, berry patches, pastures, forests, and water bodies (for data sources and scales see Table 3). The amount of grass produced per hectare of pasture was converted into the quantity of meat and milk that could hypothetically be produced with the vegetation from this area, taking into consideration the actual livestock composition of the region. For pastures, forests, orchards and berry patches, average food supply values at the Federal State level were calculated per hectare of land cover type. Regarding food production on arable land, average yields per hectare were calculated at the district level while considering the percentages of different crop types. Fodder crops (silage maize, grass, oats and triticale) were assigned a tenth of their caloric value. The value of 10% was assumed, representing an average level of feed conversion efficiency of livestock, which depends on the animal species, age, and fodder type (Webster, 1980; Wenk et al., 1980). The percentage of non-food crop production (e.g., bio-fuel crops) on arable land was estimated using the data sources in Table 3, which allowed us to restrict the calculation of crops cultivated for food production. In order to differentiate food production on arable land areas of variable soil fertility, soil fertility maps that incorporate an index of the soil's yield potential (in units from 1 to 5) were applied (see Table 3). A bi-variate regression and correlation analysis was conducted to investigate the relationship between the aggregated soil fertility index at the district level and the crop yield at the district level with $n = 24$ districts of Saxony and Saxony-Anhalt. In cases where a significant correlation between crop yield and soil fertility index was found, a correction of the results was conducted using the soil fertility maps and the determined regression curves. After food production in tonnes per hectare was calculated, the result was multiplied with the caloric value of each food, based on standard nutritional factors supplied by the United Nations FAO statistics database (FAO, 2009).

Table 5
Energy supply and demand in GJ of final energy per hectare including all relevant land cover types for the three time-points 1990, 2000, and 2007. The table includes average values for the whole case study area (e.g., for arable land) as well as ranges for values calculated at a larger scale.

	1990 Supply (GJ/ha)	Demand (GJ/ha)	2000 Supply (GJ/ha)	Demand (GJ/ha)	2007 Supply (GJ/ha)	Demand (GJ/ha)
Urban fabric	–	300–3300	0 to <0.1	190–2300	0–62	190–2600
Industrial and commercial units	–	6000	0 to <0.1	3800	0–62	3770
Transport units	–	700–1100	–	1000–1400	–	760–1300
Mineral extraction sites	19,150	1300	5750	1300	9040	1300
Arable land	–	8.7 (year 1991)	4.4–5.6 (bio energy) 3–75 (wind energy)	6.9 (year 1999)	9–12.5 (bio energy) 4–75 (wind energy)	7.5 (year 2003)
Pastures	–	3.2 (year 1991)	–	3.3 (year 1999)	–	3.5 (year 2003)
Fruit trees and berry plantations	–	20.4 (year 1991)	–	18.4 (year 1999)	–	21.6 (year 2003)
Forests	–	0.04	1–1.8	0.04	4.4	0.04
Rivers	–	–	12–69	–	12–293	–

Food demand was calculated in GJ/ha of urban area, using the average German food consumption per capita (see Table 3 for data source). Food demand at the municipality level was calculated as follows:

$$\frac{FD_{\text{pop}} \times POP}{A_{\text{urban}}} = \frac{F}{A_{\text{urban}}} \tag{13}$$

Rural urban gradients

Using the resulting composite maps of ecosystem services, we computed rural urban gradients of the supply and demand of each ecosystem service for 1990, 2000, and 2007. Twenty concentric rings with an outer radius of 1–20 km and a diameter of 1 km each were drawn around the city centre of Leipzig using the software package ArcView 3.2 by ESRI. For each ring, an average ecosystem service value per hectare was calculated. The application of concentric rings around the city centre, as used by Schneider and Woodcock (2008) and Solon (2009) among others, implies a generalization, given that the spatial structure of a metropolitan area does not exhibit the same pattern in all directions from the city centre (Zheng, 1991). But for the purposes of this paper, that is, to analyze major trends and variability between urban, suburban and rural ecosystem services supply and demand, concentric rings are considered to be a suitable approach.

Results

Energy supply and demand

The energy supply in the case study region changed considerably from 1990 to 2007. In 1990, the only energy source in use was lignite, which was extracted from the large mines located primarily south of Leipzig. Considering the amount of lignite extracted, its energetic value, its efficiency in conversion to electric power and the area that was in use for lignite extraction, an average value of 19,150 GJ final energy supply per hectare of mineral extraction site was calculated for 1990 (Table 5). By 2000, this value had decreased by more than two-thirds due to a reduced amount of extracted coal, but several renewable energy sources had now appeared in the region, namely energy from wind, biomass, and water. These energy sources combined to represent a small amount of renewable energy that was produced on arable land, forest, and river areas, with wind energy being the most important of these renewable energy sources (Table 5). By 2007, lignite extraction had increased slightly. This rising importance is predicted to continue due to the economic competitiveness of lignite as a primary energy source in Germany and the forecasted decline of electricity production by nuclear power plants and stone coal (Stoll et al., 2009). Also, more energy crops were cultivated in 2007 resulting in the doubling of average bio-energy production per hectare of arable land and forest area, compared to 2000 (Table 5). In addition, more wind-, water-, and solar power plants were constructed (Fig. 2). However, the contribution of renewable energy sources to the total final energy supply in the region was still below 7% in 2007 (Table 6), meaning that changes in lignite extraction dominated the total changes of energy supply over the whole time span of the study.

The energy demand maps reveal another picture (Fig. 3). Here, the land cover types with the highest values can be found mainly in the urban centres: the industrial and commercial units have the largest energy demand per hectare, followed by urban fabric and transportation areas. The energy demand per hectare of urban fabric (related to households and services) depends on the population

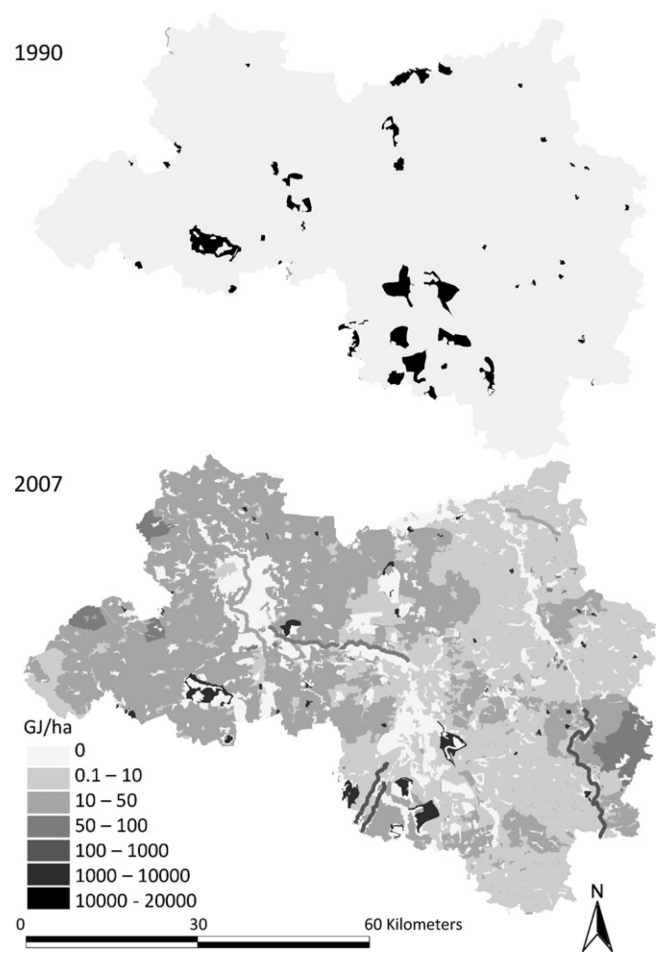

Fig. 2. Energy supply in GJ of final energy per hectare in the case study region Leipzig–Halle, years 1990 (top) and 2007 (bottom).

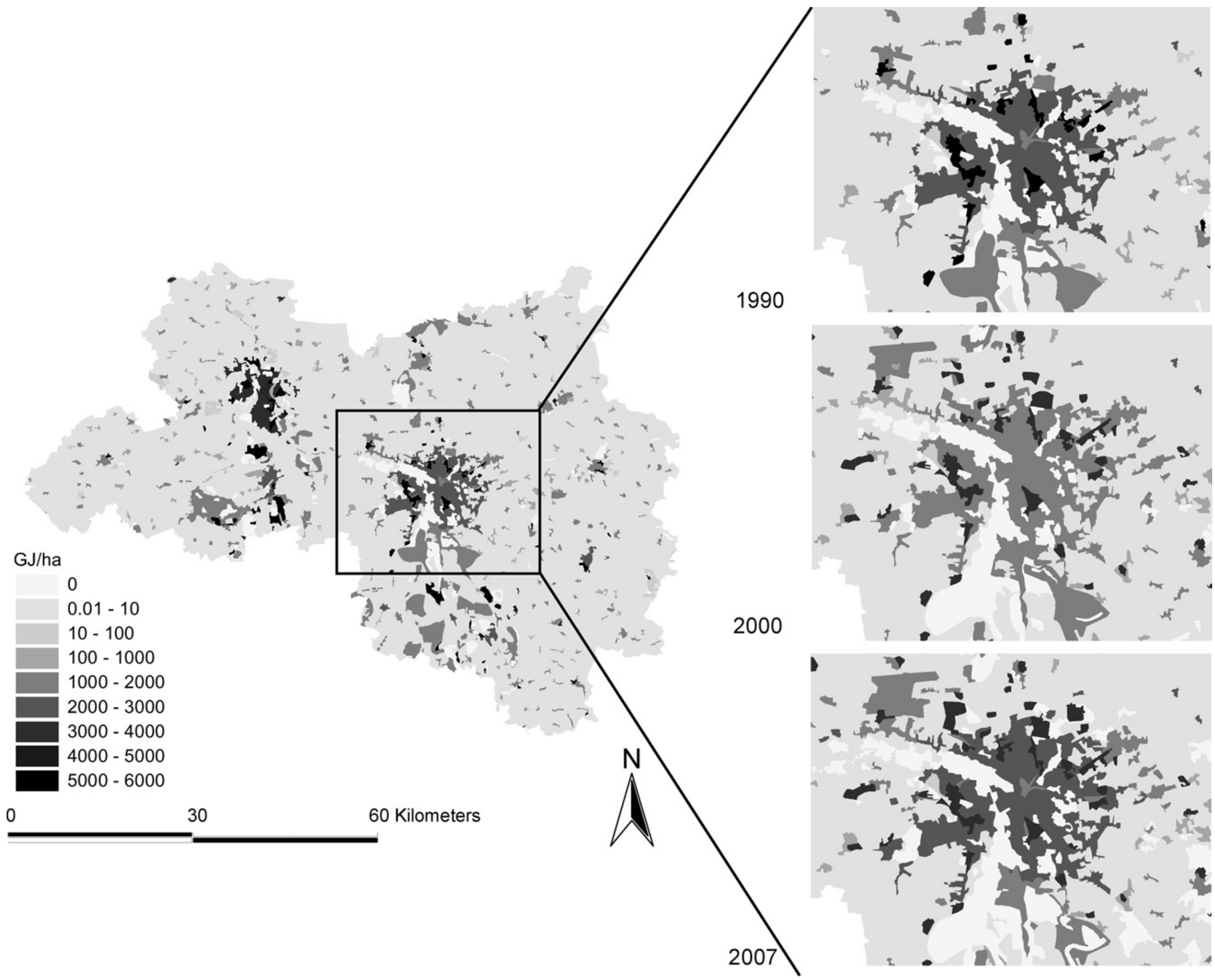

Fig. 3. Energy demand in GJ of final energy per hectare in the case study region Leipzig–Halle, year 1990 (left) and detailed energy demand maps of the urban and suburban region of Leipzig, years 1990, 2000, and 2007 (right).

density. Therefore, it was the highest in the city centres of Leipzig and Halle. The average energy demand per hectare of arable land was rather low in each year, but it was exceeded by the average supply per hectare arable land only in 2007 (Table 5). The energy demand per hectare of industrial and commercial area decreased by about one third between 1990 and 2000. The energy demand of households and services in urban areas also dropped sharply from 1990 to 2000, before increasing slightly between 2000 and 2007. This recent increased energy demand in the urban area of Leipzig can be traced to a rising per-capita energy demand as well as reurbanization processes. By contrast, the energy demand per hectare of transport units increased between 1990 and 2000 (Table 5).

Table 6 summarizes the energy demand and supply for the three points in time over the entire case study area. In addition, it captures the proportion of renewable energies out of the total supply. We see that the energy supply was twice as high as the demand in 1990. That ratio decreased significantly to 0.54 in 2000, as the total energy supply was reduced substantially more than the total energy demand. However, during the next time period (2000–2007), both the total energy demand and the supply increased slightly again, as did the supply/demand ratio.

The rural–urban pattern of energy demand and supply for the Leipzig region is shown in Fig. 4. Energy demand is the highest in the city centre and then decreases along with the population density. In 1990, energy demand increases again at a distance of 4 km from the city centre, where several industrial and commercial units could be found, but then drops sharply. In 2000 and even more distinctly in 2007, the sharp decline in the energy demand gradient is not observed before a distance of 7 km from the city centre, due to the new industrial and commercial units that have been constructed on the urban fringe. In the more rural areas, energy demand is also higher at the two later points in time. These facts, together with the lower energy demand per hectare in the city centre, result in more gradual energy demand gradients in 2000 and 2007 as compared to 1990. Meanwhile, the energy supply gradient of 1990 differs strongly from the other two gradients, mainly because of the abandonment of mining in several extraction sites in the suburban and rural regions after 1990.

Table 6
Total final energy supply and demand in the case study and renewable energies in % of the total final energy supply in 1990, 2000, and 2007.

	1990	2000	2007
	Final energy (10^6 GJ)		
Energy supply	294	62	74
Energy demand	142	115	120
Ratio supply/demand	2.07	0.54	0.62
	Renewable energies in % of total supply		
	0	4.5	6.2

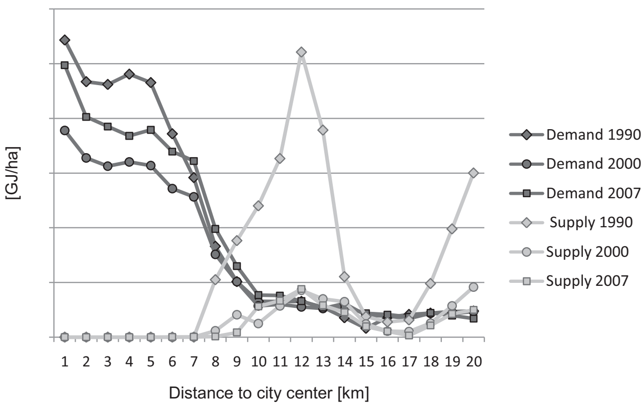

Fig. 4. Rural–urban gradients of energy demand and supply for Leipzig, showing the three points in time 1990, 2000, 2007. Each point represents the average value in the concentric ring at the respective distance from the city centre.

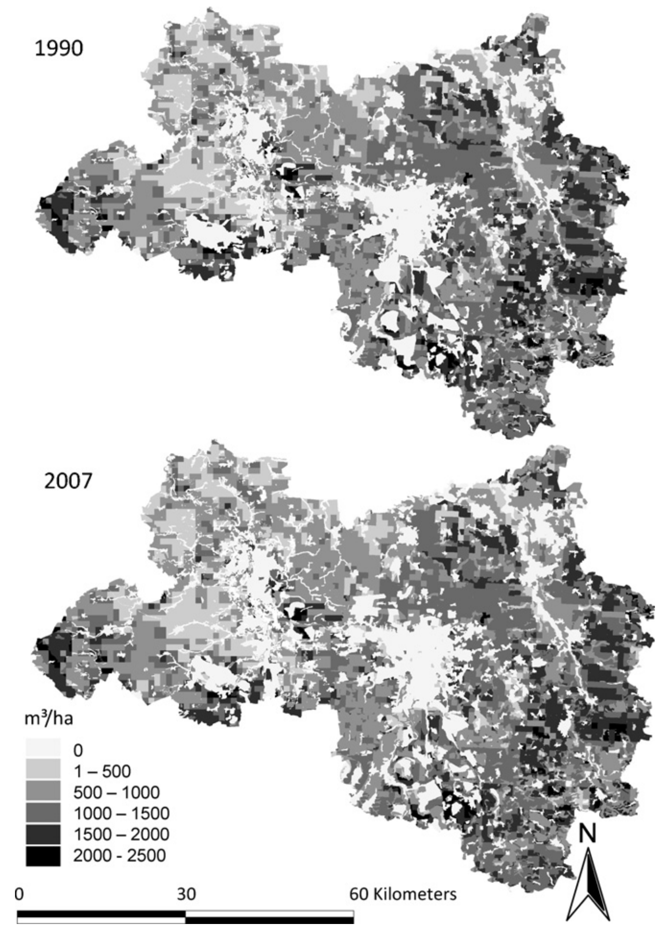

Fig. 5. Water supply in m^3 per hectare in the case study region Leipzig–Halle, year 1990 (top) and year 2007 (bottom).

Water supply and demand

Fig. 5 shows maps of the water supply (groundwater recharge and interflow) per hectare for 1990 and 2007. The ranges of water supply per hectare of land cover type are indicated in Table 7. The water supply varies not only with the land cover type, but also with precipitation, ETP and soil characteristics, producing a wide range of supply values within each land cover type. Since the ETP of arable land was the lowest among all the water-supplying land cover types, the groundwater recharge under arable land was the highest, followed by broad-leaf forests, mixed forests, green areas and coniferous forests. For the water supply quantification, average climate data for the time span 1961–1990 were used, which results in constant water supply values per land cover type over the whole time span. All changes in water supply at the landscape level over time therefore arose from land cover changes. The main land cover changes influencing the water supply were the increase of industrial and transportation area and urban fabric at the urban fringe of both cities. In some areas featuring appropriate climatic and soil conditions and a high groundwater level, vegetation consumed groundwater by capillary rise. This is especially the case along the flood plains in the urban region of Leipzig and in the north-eastern part of the case study region.

Water demand on artificial surfaces was the highest on industrial and commercial units, but the water demand on mineral extraction sites was also high due to the lowering of the groundwater, which is necessary for coal mining. The water demand per hectare of urban fabric varied with the population density, and was therefore highest in the city centres of Leipzig and Halle. Both the total population size as well as the per-capita usage of freshwater decreased considerably between 1990 and 2000, leading to a reduction of water demand per hectare of urban area during the same time span. In spite of the population increase in the city centre of Leipzig after 2000, water demand further decreased because of the ongoing decline in per-capita usage. Likewise, water demand per hectare of industrial and commercial areas continuously decreased from 1990 to 2007 (see Fig. 6).

Table 7 shows the total water supply and demand in the case study region for the three points in time and the supply/demand ratio. The total water supply slowly, but continuously, decreased over the whole time period. Thus, the steady increase of urban, industrial, commercial and transportation areas seems to have had a greater impact than the reduction of mineral extraction sites. Nevertheless, the supply/demand ratio increased by one third from 1990 to 2007 due to the considerable reduction in water demand in the industrial, commercial and urban areas (Table 8).

Table 7
Water supply and demand in m^3 per hectare including all relevant land cover types for the three points in time 1990, 2000, and 2007. The table includes average values for the whole case study area (e.g., for mineral extraction sites) as well as ranges for values calculated at a larger scale.

	1990		2000		2007	
	Supply (m^3/ha)	Demand (m^3/ha)	Supply (m^3/ha)	Demand (m^3/ha)	Supply (m^3/ha)	Demand (m^3/ha)
Urban fabric	–	440–3300	–	220–1550	–	200–1500
Industrial and commercial units	–	24,800	–	21,800	–	17,800
Mineral extraction sites	–	20,000	–	20,000	–	20,000
Arable land and heterogeneous agricultural areas	0–2500	0–1500	0–2500	0–1500	0–2500	0–1500
Green areas[a]	0–1800	0–1800	0–1800	0–1800	0–1800	0–1800
Broad-leaved forest	0–2150	0–1500	0–2150	0–1500	0–2150	0–1500
Coniferous forest	0–1500	0–1900	0–1500	0–1900	0–1500	0–1900
Mixed forest	0–1825	0–1750	0–1825	0–1750	0–1825	0–1750

[a] Including urban green, pastures, areas of fruit trees and berry plantations, natural grassland and shrub, sparsely vegetated areas.

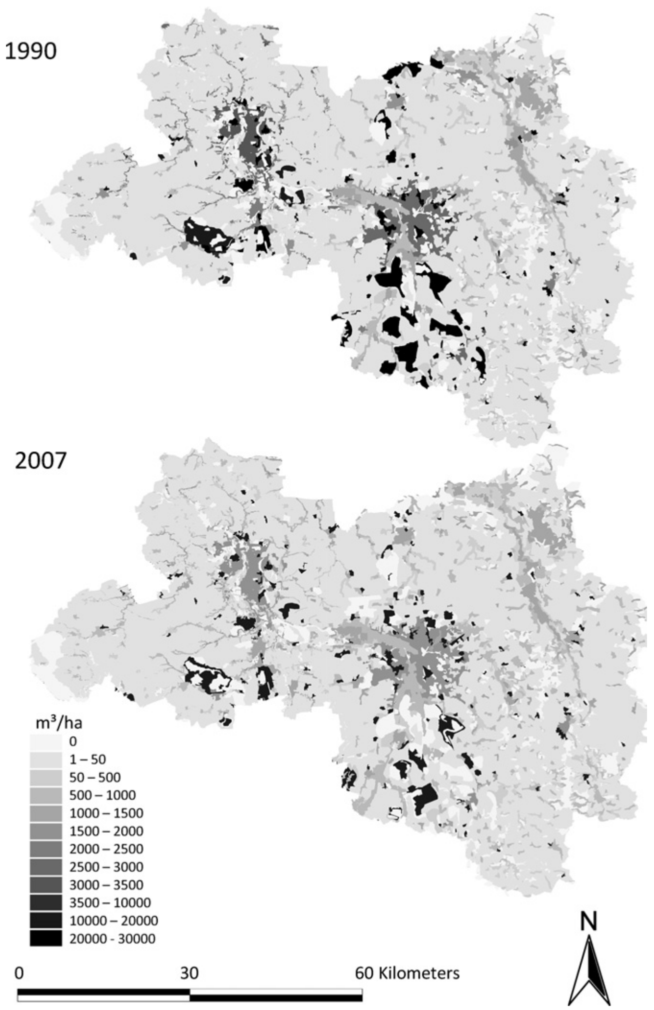

Fig. 6. Water demand in m^3/ha in the case study region Leipzig–Halle, year 1990 (top) and 2007 (bottom).

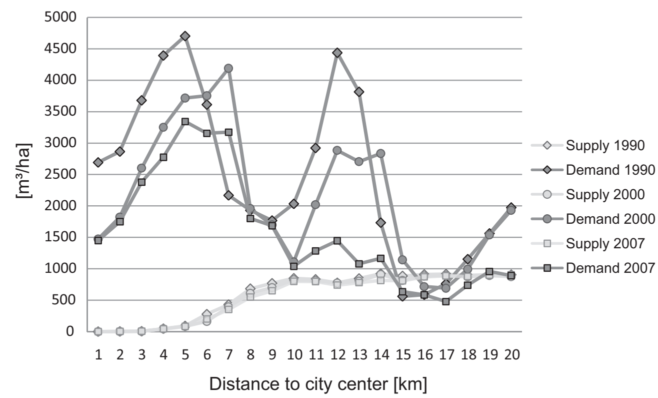

Fig. 7. Rural–urban gradients of water demand and supply for Leipzig, showing the three points in time 1990, 2000, 2007. Each point represents the average value in the concentric ring at the respective distance from the city centre.

The rural–urban gradients of water supply and demand are shown in Fig. 7. The demand gradients exhibit two major peaks at a distance of 5–7 and 12–14 km due to the concentration of industrial and commercial units as well as mineral extraction sites at these distances from the city centre. Both land cover types dominate the trends of the water demand gradients due to their high water demand per hectare. Thus, while the overall level of water demand decreased continuously from 1990 to 2007, it is also evident that the area of high water demand per hectare is shifting away from the centre.

In comparison to the water demand gradients, the water supply gradients show a low variability from one point in time to another. The water supply per hectare increases at a distance of 5–10 km from the city centre before dropping back to the same level. This small increase is caused by the high groundwater level in and around the city centre of Leipzig and the resulting water consumption of the vegetation. Only a small decrease of water supply is observable in the suburban regions at a distance of 6–9 km from 1990 to 2007.

Table 8
Total water supply and demand in the case study region in years 1990, 2000 and 2007.

	1990	2000	2007
		(10^6 m^3)	
Water supply	369	365	362
Water demand	592	490	398
Ratio supply/demand	0.62	0.74	0.91

Food supply and demand

For the quantification of the food supply, a correlation analysis between soil fertility and the yield of different crops at the district level was conducted, in order to plot statistical crop yield data on a spatially explicit soil map. In Table 9, the Pearson correlation coefficients for crop yields and the soil fertility index of 24 districts in Saxony and Saxony-Anhalt are presented. They show a notable variation among the different crop types as well as among the three points in time. The temporal differences presumably stem from variations of other impact factors on the yield such as human inputs and climatic conditions. As the correlations between soil fertility and crop yields are stronger for grains than for root vegetables and rape seed, the latter crops seem to be more sensitive to climatic constraints than to soil fertility influences.

Based on regression curves that were plotted for those relationships with a significant correlation (as marked in Table 9) and on additional statistical data for other produced food types, food supply maps were produced (Fig. 8). As the food supply was calculated in GJ per hectare, the results depend on the yield in tonnes and on the food's caloric value. For instance, fruit trees have the highest yield in tonnes per hectare, but the caloric value of fruit is rather low compared with grains, meat or fish. Nevertheless, fruit trees and berry patches provided the highest food supply per hectare in 2000 and 2007, followed by arable land and pastures. Water bodies and forests played only a minor role in the food supply of the region.

The food supply and demand values for each land cover type are listed in Table 10. Regarding the arable land cover type, the composition of cultivated crops was essential. The more bread grains that were cultivated and conversely the fewer root crops, fodder crops, and biofuel plants, the higher the food supply in GJ per hectare. This is the first reason for the food supply differences observed between the administrative districts in Fig. 8, as crop composition data on the district level were used. The second reason for this variability is soil fertility, which is highest in the western and southern parts of the case study region. The temporal variability of the food supply is just as substantial as the spatial variability. The food supply per hectare increased from 1990 to 2000 and again from 2000 to 2007 due to a higher productivity per hectare and a decrease in fodder production. It is evident that these two processes had a stronger influence on food production than the parallel ongoing increase of biofuel crop cultivation on arable land.

Table 9
Pearson's correlation coefficients *r* between crop yield and soil fertility for different crops in 1991, 2000, 2007 with $n = 24$ districts of Saxony and Saxony-Anhalt (see Table 3 for data sources).

	Yield									
	Year	Winter wheat	Winter rye	Winter barley	Oat	Triticale	Potatoes	Turnips	Rape seed	Silage maize
Soil fertility index	1991	0.52**	0.22	0.52**	0.47**	0.31	0.12	0.3	0.29	0.03
	2000	0.58**	0.64**	0.76**	0.29	0.48**	0.25	0.16	0.09	0.23
	2007	0.57**	0.33*	0.51**	0.03	0.44*	0.14	0.29	0.03	0.61**

* $p < 0.05$.
** $p < 0.01$.

Table 10
Food supply and demand in GJ per hectare including all relevant land cover types for the three points in time 1990, 2000, and 2007. The table includes average values for the whole case study area (e.g., food supply on pastures) as well as ranges for values calculated at a larger scale (food supply on arable land, food demand on urban fabric).

	1990 (1991)[a]		2000 (1999)[a]		2007	
	Supply (GJ/ha)	Demand (GJ/ha)	Supply (GJ/ha)	Demand (GJ/ha)	Supply (GJ/ha)	Demand (GJ/ha)
Urban fabric	–	4.5–35.6	–	2.6–28.7	–	2.4–28.5
Arable land	23–37.6	–	27.3–43.5	–	30.6–50.5	–
Fruit trees, berry plantations	25.5	–	65.6	–	55.8	–
Pastures	9.4	–	13.7	–	17.3	–
Forests	0.01	–	0.02	–	0.1	–
Water bodies	1.3	–	0.5	–	0.4	–

[a] Statistical data on crop yield for the earliest time-point are from 1991 because of data unavailability for 1990. Statistical data on crop composition are from years 1991, 1999 and 2007.

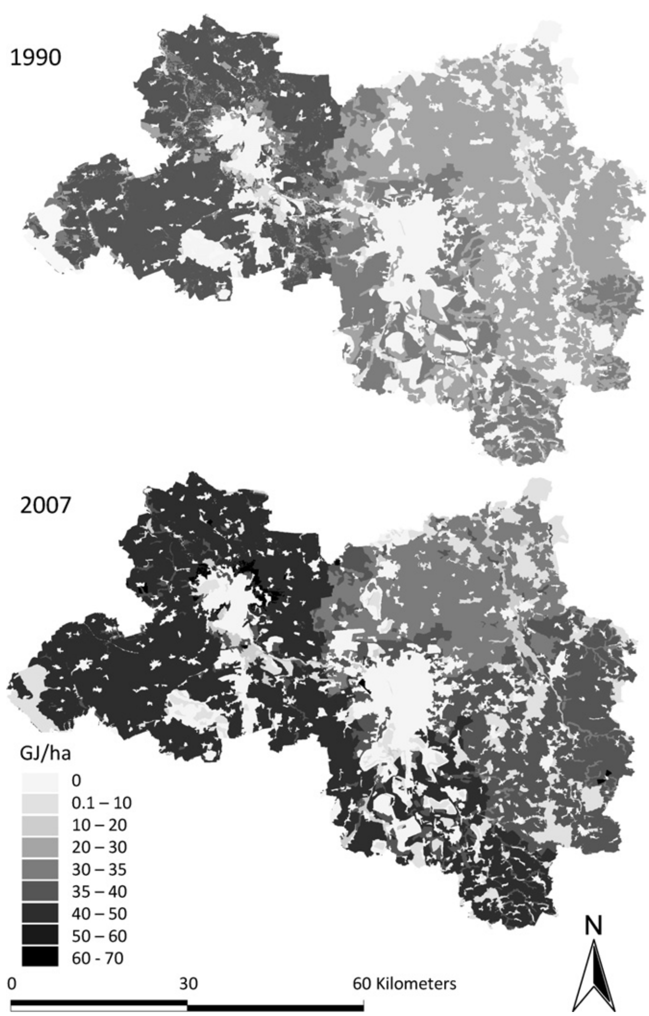

Fig. 8. Food supply in GJ per hectare in the case study region Leipzig–Halle, year 1990 (top) and 2007 (bottom).

Food demand depends on both per-capita food consumption and the total population. While the total population number decreased during the time period under investigation, food consumption per person increased from 0.54 GJ in 1990 to 0.55 GJ in 2000 and then to 0.56 GJ in 2007 (see Table 3 for data source). However, the total population decrease had a stronger impact, resulting in an overall decrease in food demand per hectare of urban area (Table 10). The city centre of Leipzig is an exception to this observation, due to the recent reurbanization trend (Fig. 9).

Table 11 presents the total food supply and demand in the whole case study region and the supply/demand ratios. The increase in the food supply in combination with the decreasing food demand results in a growing supply/demand ratio. Already in 1990, the food supply in the region was almost 12 times higher than the food demand. Of course, this ratio is only valid for the caloric supply and demand of the food, which was very high due to the large amount of cultivated grains and their reasonably high yields per hectare. Thus, the choice of the unit of measure had an important influence on the results in this case.

The rural–urban gradients of food demand and supply for Leipzig are illustrated in Fig. 10. The food demand per hectare is the highest in the city centre and decreases continuously as we move away from the centre. Again, the gradients show a decrease in demand from 1990 to 2000 due to the population decline. Because of the reurbanization of Leipzig, food demand in the city centre slightly increased again from 2000 to 2007. Only at a distance of 7–12 km from the city centre, where new suburban settlements

Table 11
Total food supply and demand in the case study region and the food supply/demand ratios for the three points in time 1990, 2000, and 2007.

	1990 (1991)[a]	2000 (1999)[a]	2007
		(10^6 GJ)	
Food supply	9.46	11.27	12.10
Food demand	0.80	0.76	0.75
Ratio supply/demand	11.8	14.8	16.1

[a] Statistical data on crop yield for the earliest time-point are from 1991 because of data unavailability for 1990. Statistical data on crop composition are from years 1991, 1999 and 2007.

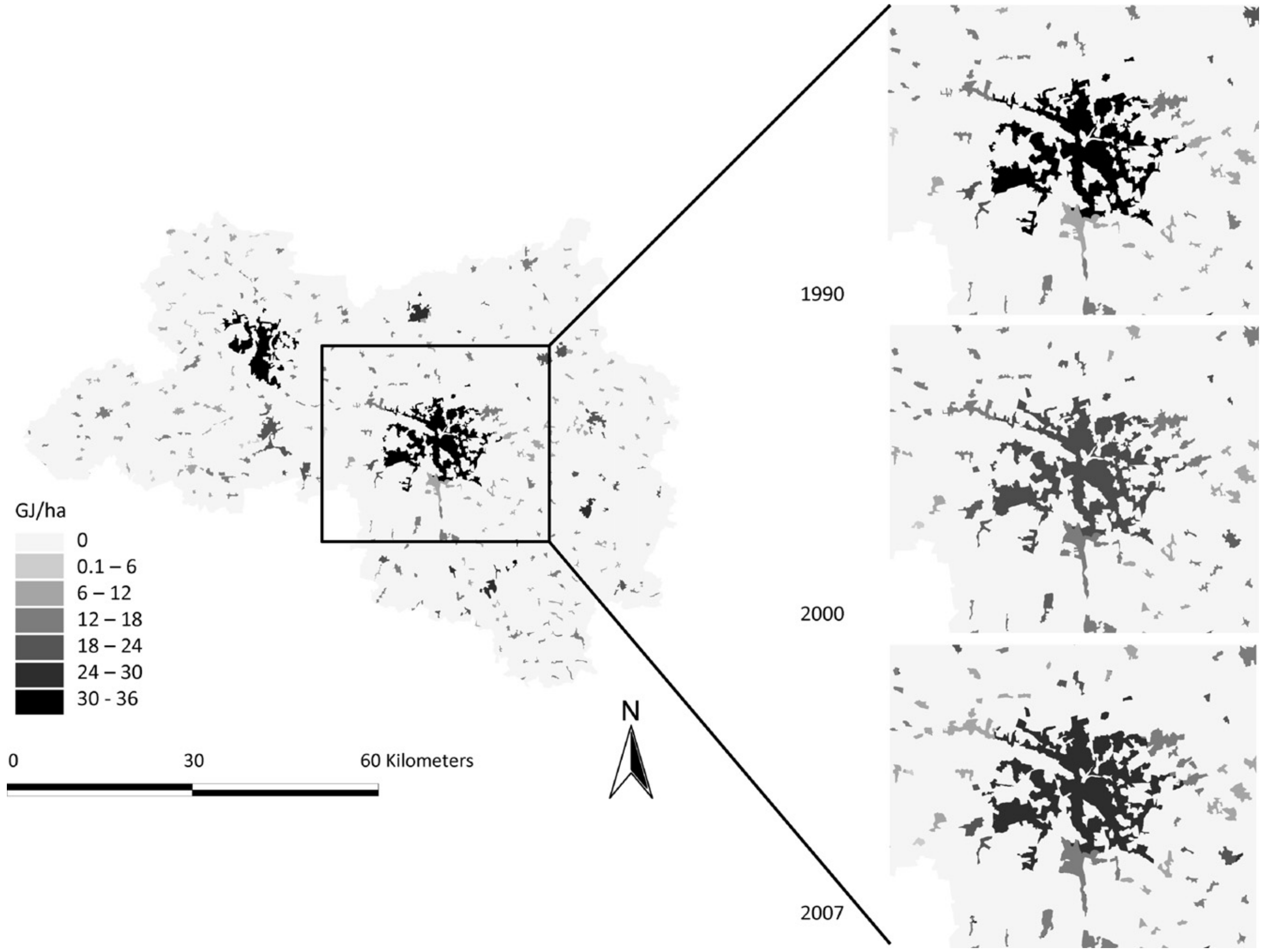

Fig. 9. Food demand in GJ per hectare in the case study region Leipzig–Halle, year 1990 (left) and detailed food demand in the urban and suburban region of Leipzig in 1990, 2000, and 2007 (right).

were built, did the food demand in 2000 and 2007 exceed the demand in 1990. The food supply increased considerably from 1990 to 2000 and again from 2000 to 2007 in rural areas. Influences of urban sprawl on the food supply can be identified in 2007, when it decreased at a distance of 6–12 km compared to 2000, in spite of higher productivity per hectare.

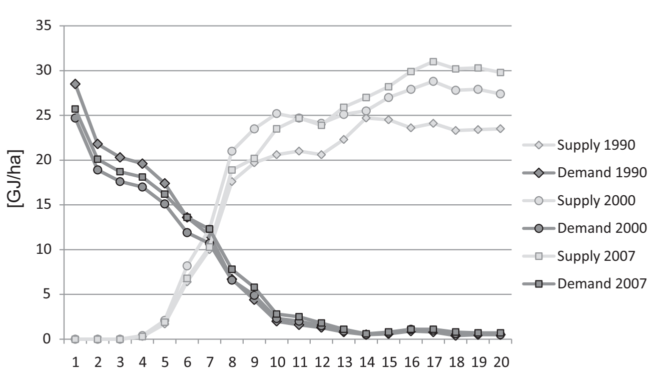

Fig. 10. Rural–urban gradients of food demand and supply for Leipzig, showing the three points in time 1990, 2000, 2007. Each point represents the average value in the concentric ring at the respective distance from the city centre.

Discussion

In the case study region Leipzig–Halle, the demand for ecosystem services underwent significant changes from 1990 to 2007. In the city centre, the population decline led to a considerable decrease in the demand for all three services investigated in this study. In parallel to that development, the suburbanization of industry, commerce and population caused an increase in the demand for ecosystem services in the suburban and rural regions. Both processes resulted in a levelling of rural–urban demand gradients, indicating decreasing differences between urban, suburban and rural resource consumptions.

The supply side of ecosystem services also exhibited clearly visible changes, which show different trends for each of the investigated ecosystem services. Four major processes took place that influenced the food supply in the case study region. Two of them, the expansion of urban area on fertile soils and the increase of biofuel crop cultivation on arable land, had a negative impact on food production. However, the other two had a much stronger and more positive influence on total food production, namely the increase in productivity per hectare and the decrease in fodder production. Consequently, the loss of agricultural land in the region due to urban sprawl did not have a visible net effect on the overall food production, as it was more than outweighed by the increased productivity. We conclude that land use intensity changes had much stronger impacts on ecosystem service supply than changes in the land cover pattern in this case. However, the expansion of urban

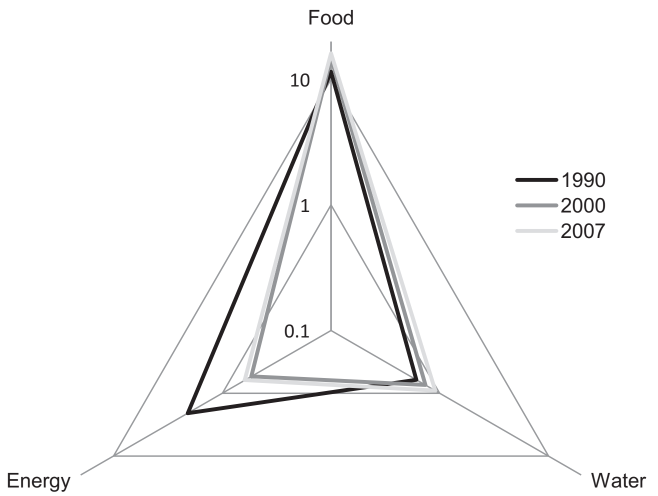

Fig. 11. Supply/demand ratios of food, energy and water for the whole case study area in a logarithmic scale.

area occurred on the most fertile soils located around the city centres, especially around Halle.

During the last two decades, more and more land cover types have been discovered to be usable for energy production purposes. Nowadays, the energy providing capacities of almost all land cover types are used, be it for installing solar-, water-, or wind power plants or transforming crops and wood into biofuel. Nevertheless, the amount of energy that was produced by renewable energy sources remains low in comparison to the non-renewable energy source lignite. In 2007, only a very small proportion of the energy demand was met by a renewable energy supply, and therefore we cannot speak of a sustainable situation here, regardless of the supply/demand ratios.

The water supply in the region decreased, mainly in the suburban areas, despite the decommissioning of several mineral extraction sites, whose areas then contributed to the water supply instead of the demand. Still, these land cover changes did not fully compensate for the reduced water supply due to soil sealing.

Fig. 11 summarizes the temporal dynamics of the supply/demand ratios for all three investigated ecosystem services. As described above, the supply of the different services influenced each other in manifold instances, causing several possible tradeoffs between them. The identification of such tradeoffs through the ecosystem service approach has the potential to provide helpful information for regional decision-making regarding landscape management. First, the increase in biofuel crop cultivation negatively impacted food production; secondly, the decreased lignite extraction negatively influenced the energy supply, but positively influenced the water supply. Accordingly, Fig. 11 shows a decreasing energy supply/demand ratio, but an increasing water supply/demand ratio. However, the rising water supply/demand ratio is caused more by changes in demand rather than changes in supply. The food supply/demand ratio increased as well, despite the growth of biofuel crop cultivation on arable land. Hence, no tradeoffs between the services were visible during the time span under consideration. This observation could also be helpful information for decision makers in the region: If there are no tradeoffs identifiable between bio-energy and food production, it may be reasonable to increase energy crop cultivation.

For decisions of that kind, tradeoffs between provisioning, regulating and cultural ecosystem services and between ecosystem integrity variables should be considered in the discussion. The maximization of resource provision for direct human use should not be the only objective of decision makers involved in landscape management, as this may negatively influence ecosystem functioning and ecosystem integrity which are the basic conditions for the provision of each ecosystem service. Thus, for example, the higher use of pesticides in biofuel crop cultivation in comparison to food crop cultivation can cause a negative impact on biodiversity. In this sense, Luck et al. (2001) argue that "the capacities of terrestrial ecosystems to assimilate gaseous waste may be more limiting to humans than any other constraint on their resource production" (p. 791), emphasizing the importance of carbon sequestration in comparison to resource provision.

It is evident that adequate methods for the quantification of regulating services and ecosystem integrity at the landscape scale have yet to be developed. Additionally, it will be necessary to refine the modelling of ecosystem service demand by incorporating lifestyle and household variables into the analysis. Variables such as household size, housing area, age structure, and income clearly vary along the rural–urban gradient (Kroll, 2009; Kroll and Kabisch, 2011), and are assumed to have an important impact on the rural–urban pattern of resource consumption (Lutzenheiser, 1997; Liu et al., 2003; Lenzen et al., 2004; Williams, 2005). Another aspect that has not yet been incorporated into the analysis is the trade of resources and the role of indirect water and energy consumption by households. In the present study, the water and energy used in distant regions of the world to produce the goods that are consumed in the case study area are not considered. Incorporating this information would presumably decrease the energy and water supply/demand ratios considerably, revealing more clearly the overexploitation of resources and the prevailing unsustainable way of life in the case study area. Even concerning food, for which the supply in our case study area would be more than adequate to sustain the population, the region still depends on imports, as the variety of locally grown food does not meet the requirements of the population. The same observation applied to energy, due to the high demand for natural gas and petroleum in the region.

Hence, the supply/demand ratios presented in this paper demonstrate only the potential and not the actual self-sufficiency of the region concerning the three investigated services, as imports and exports are not considered. Whether self-sufficiency is the overall aim or, instead, the maximization of specific services according to the land's potential and stakeholder preferences, is a matter for discussion. In the latter case, it would be important to guarantee that the cluster of regions that trade ecosystem services among one another do not run an ecological deficit. Ecosystem service quantification and mapping at the landscape scale, which are applicable for all European regions, can aid in that discussion. Our approach may also help to identify thresholds or boundaries of the sustainability "choice space" defined by the interaction of biophysical limits and socio-economic values at the landscape scale (Potschin and Haines-Young, 2006). There are various ecological scales at which ecosystem services are provided as well as many different institutional scales at which these services are managed, and scales often do not coincide (Hein et al., 2006). Consequently, not all aspects of ecosystem service supply at the landscape scale discussed in this paper can be influenced by planners and politicians working in institutions that are operating at the regional scale. For instance, regional stakeholders do not have the capability to influence global economic market conditions or subsidies for food and energy, both of which have an important influence on the dynamics of ecosystem service supply at the regional level. However, regional stakeholders can influence general conditions for renewable energy, food, and water provision by regional spatial planning. Also the demand side can be influenced through the use of information campaigns, the improvement of water- and energy distribution networks, and the promotion of resource efficient industries.

Conclusions

In this paper, we presented a method to identify the impacts of socio-economic and land use changes on the demand and supply of provisioning ecosystem services in a rural–urban landscape. Both the demand and the supply of three ecosystem services under investigation changed considerably between 1990 and 2007. This observation is valid not only for the total ecosystem service demand and supply in the case study region Leipzig–Halle, but equally for its rural–urban distribution. Due to the suburbanization of the population and industry, in concert with an overall shrinking of the population, the rural–urban demand gradients became more gradual, making the distinction between urban and rural consumption characteristics more subtle. Urban sprawl did not clearly impact the provision of ecosystem services, which underscores the fact that land use intensity changes had a more important impact on ecosystem service supply than did changes in land cover such as the decrease in agricultural area. The observed change of land use intensity is the reason that tradeoffs between the services are not yet identifiable.

Such information, together with a comparison of the supply/demand ratios of different ecosystem services, can help decision makers in modifying the regional landscape in order to achieve a sustainable balance of resource supply and demand.

As all data used in this study are publicly available, it should be possible to extend the quantification and mapping approach to other European regions in order to produce a broader and more comprehensive picture of ecosystem service demand and supply across all European urban regions. This could be a first step towards incorporating ecosystem services into national accountings and policy implementation.

Acknowledgements

We wish to thank Regina Hoffmann-Müller and Christopher Stolz for their very helpful review of an earlier version of this manuscript. Many thanks to Jennifer Anderson for proofreading the manuscript. This work was financially supported by the EU Integrated Project PLUREL (contract no. 036921).

References

Anderson, J.O., Lindroth, M., 2001. Ecologically unsustainable trade. Ecological Economics 37, 113–122.
Bai, X., 2007. Integrating global environmental concerns into urban management. Journal of Industrial Ecology 11 (2), 15–29.
Banzhaf, E., Grescho, V., Kindler, A., 2009. Monitoring the urban to peri-urban development with integrated remote sensing and GIS information: a Leipzig, Germany case study. International Journal of Remote Sensing 30 (7), 1675–1696.
Blair, R.B., 1996. Land use and avian species diversity along an urban gradient. Ecological Applications 6 (2), 506–519.
Bolund, P., Hunhammar, S., 1999. Ecosystem services in urban areas. Ecological Economics 29, 293–301.
Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63 (2–3), 616–626.
2010. Bundesverband für Braunkohle (Date: 10.02.2010) http://www.debriv.de/pages/glossar.php?page=505&archiv=W.
Burkhard, B., Kroll, F., Müller, F., Windhorst, W., 2009. Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. Landscape Online, 15.
Burkhard, B., Kroll, F., Müller, Netkov, S., Müller, F., 2011. Mapping ecosystem service supply, demand and budgets. Ecological Indicators, doi:10.1016/j.ecolind.2011.06.019.
Chambers, N., Simmons, C., Wackernagel, M., 2000. Sharing Nature's Interest. Ecological Footprints as an Indicator of Sustainability. Earthscan Publ. Ltd., London.
Couch, C., Karecha, J., Nuissl, H., Rink, D., 2005. Decline and sprawl: an evolving type of urban development – observed in Liverpool and Leipzig. European Planning Studies 13 (1), 117–136.
Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. Biological Conservation 141, 350–352.
Costanza, R., D'Arge, R., de Groot, R.S., Faber, S., Grasso, M., Hannon, M., Limburg, K., Naeem, S., ONeill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of world's ecosystem services and natural capital. Nature 387, 253–260.
Curran, S.R., de Sherbinin, A., 2004. Completing the picture: the challenges of bringing "consumption" into the population–environment equation. Population and Environment 26 (2), 107–131.
Decker, E.H., Elliott, S., Smith, F.A., Blake, D.R., Rowland, F.S., 2000. Energy and material flow through the urban ecosystem. Annual Reviews Energy Environment 25, 685–740.
Döring, J., Müller, J., Jörn, M., 1995. Klimatische Kennzeichnung des mitteldeutschen Schwarzerdegebietes. In: Körschens, M., Mahn, E.G. (Eds.), Strategien zur Regeneration belasteter Agrarökosysteme des mitteldeutschen Schwarzerdegebietes. B. G. Teubner Verlagsgesellschaft, Stuttgart, Leipzig, pp. 533–568.
European Environmental Agency, 1994. Corine Land Cover report (Date: 12.01.2009) http://www.eea.europa.eu/publications/COR0-landcover.
Food and Agriculture Organization, 2009. Standard Nutritive Factors Used In Food Balance Sheets (Date: 08.01.2010) http://www.fao.org/economic/ess/publications-studies/publications/nutritive-factors/it/.
Fisher, B., Turner, R.K., 2008. Ecosystem services: classification for valuation. Biological Conservation 141, 1167–1169.
Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecological Economics 68, 643–653.
Foresman, T.W., Pickett, S.T.A., Zipperer, W.C., 1997. Methods for spatial and temporal land use and land cover assessment for urban ecosystems and application in the greater Baltimore-Chesapeake region. Urban Ecosystems 1, 201–216.
German Federal Environmental Agency, DLR, DFD. 2004. CORINE Land Cover 2000. Daten zur Bodenbedeckung Deutschland, DVD.
German Federal Environmental Agency, 2010. German Aerospace Center 2010. CORINE Land Cover 2006. http://www.corine.dfd.dlr.de/intro_de.html (Date: 02.15.2010).
German Federal Statistical Office, 2009. Umweltnutzung und Wirtschaft. Bericht zur Umweltökonomischen Gesamtrechnung, Wiesbaden.
German Federal Statistical Office, 2010. Export, Import, Globalisierung. Deutscher Außenhandel und Welthandel, 1990 bis 2008, Wiesbaden.
German Meteorological Service (Ed.), 1999. Klimaatlas Bunderepublik Deutschland. Offenbach am Main.
Haase, D., Nuissl, H., 2007. Does urban sprawl drive changes in the water balance and policy? The case of Leipzig (Germany) 1870–2003. Landscape and Urban Planning 80 (1–2), 1–13.
Haase, D., 2009. Effects of urbanization on the water balance – a long-term trajectory. Environmental Impact Assessment Review 29, 211–219.
Hahs, A.K., McDonnell, M.J., 2006. Selecting independent measures to quantify Melbourne's urban–rural gradient. Landscape and Urban Planning 78, 435–448.
Haines-Young, R., Potschin, M., 2010. Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting. Report to the European Environmental Agency, Nottingham.
Hein, L., van Koppen, K., de Groot, R., van Ierland, E., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecological Economics 57, 209–228.
Institute for Energy and Environment, 2007. Schlüsseldaten Klimagasemissionen (Date: 08.01.2009) www.ufop.de/downloads/Bericht_Klimagas.pdf.
Jenerette, G.D., Marussich, W.A., Newell, J.P., 2006. Linking ecological footprints with ecosystem valuation in the provisioning of urban freshwater. Ecological Economics 59, 38–47.
Jenerette, G.D., Wu, W., Goldsmith, S., Marussich, W.A., Roach, W.J., 2006b. Contrasting water footprints of cities in China and the United States. Ecological Economics 57, 345–358.
Kabisch, N., Haase, D., Haase, A., 2010. Evolving Reurbanisation? Spatio-temporal Dynamics as Exemplified by the East German City of Leipzig. Urban Studies 47 (5), 967–990.
Kates, R.W., Clark, W.C., Corell, R., Hall, J.M., Jaeger, C.C., Lowe, I., McCarthy, J.J., Schellnhuber, H.J., Bolin, B., Dickson, N.M., Faucheux, S., Gallopin, G.C., Grubler, A., Huntley, B., Jager, J., Jodha, N.S., Kasperson, R.E., Mabogunje, A., Matson, P., Mooney, H., Moore, B., O'Riordan, T., Svedin, U., 2001. Environment and development – sustainability science. Science 292, 641–642.
Kennedy, C., Duddihy, J., Engel-Yan, J., 2007. The changing metabolism of cities. Journal of Industrial Ecology 11 (2), 43–59.
Kroll, F., 2009. Peeling the urban onion – rural–urban gradients of socio-economic and land use features in European city regions. In: Breuste, M., Kozová, M., Finka, M. (Eds.), European Landscapes in Transformation – Challenges for Landscape Ecology and Management. European IALE Conference 2009, Salzburg, Bratislava.
Kroll, F., Haase, D., 2010. Does demographic change affect land use patterns? A case study from Germany. Land Use Policy 27 (3), 726–737.
Kroll, F., Kabisch, N., 2011. The relation of diverging urban growth processes and demographic change along an urban–rural gradient. Population, Space and Place, doi:10.1002/psp.653.
Kreuter, U.P., Harris, H.G., Matlock, M.D., Lacey, R.E., 2001. Change in ecosystem service values in the San Antonio area, Texas. Ecological Economics 39, 333–346.
KTBL, 2005. Faustzahlen für die Landwirtschaft. KTBL-Schriftenvertrieb im Landwirtschaftsverlag, Münster-Hiltrup.
Lenzen, M., Dey, C., Foran, B., 2004. Energy requirements of Sydney households. Ecological Economics 49, 375–399.
Limburg, K.E., O'Neill, R.V., Costanza, R., Farber, S., 2002. Complex systems and valuation. Ecological Economics 41, 409–420.
Liu, J., Daily, G.C., Ehrlich, P.R., Luck, G.W., 2003. Effects of household dynamics on resource consumption and biodiversity. Nature 42, 530–533.

Luck, M.A., Jenerette, G.D., Wu, J., Grimm, N.B., 2001. The urban funnel model and the spatially heterogeneous ecological footprint. Ecosystems 4 (8), 782–796.
Luck, M., Wu, J., 2002. A gradient analysis of urban landscape pattern: a case study from the Phoenix metropolitan region, Arizona, USA. Landscape Ecology 17, 327–339.
Lutzenheiser, L., 1997. Social structure, culture, and technology: modeling the driving forces of household energy consumption. In: Stern, P.C., Dietz, T., Ruttan, V.W., Socolow, R.H., Sweeney, J.L. (Eds.), Environmentally Significant Consumption: Research Directions. National Academy Press, Washington, DC.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being. Millennium Ecosystem Assessment (MA). Island Press, Washington, DC.
McDonald, R., 2009. Ecosystem service demand and supply along the urban-to-rural gradient. Journal of Conservation Planning 5, 1–14.
McDonnell, M.J., Pickett, S.T.A., 1990. Ecosystem structure and function along urban–rural gradients: an unexploited opportunity for ecology. Ecology 71 (4), 1232–1237.
McDonnell, M.J., Pickett, S.T.A., Groffman, P., Bohlen, P., Pouyat, R., Zipperer, W., Parmelee, R., Carreiro, M., Medley, K., 1997. Ecosystem processes along an urban-to-rural gradient. Urban Ecosystems 1, 21–36.
McMichael, A.J., Butler, C.D., Folke, C., 2003. New visions for addressing sustainability. Science 302, 1919–1920.
Ministry of Agriculture and Environment Saxony-Anhalt, 2002. Energetische Biomassenutzung in Sachsen-Anhalt, Magdeburg.
Ministry of Agriculture and Environment Saxony-Anhalt, 2007. Biomassepotenzialstudie 2007 für das Land Sachsen-Anhalt, Magdeburg.
Müller, F., Burkhard, B., 2007. An ecosystem based framework to link landscape structures, functions and services. In: Mander, Ü., Wiggering, H., Helming, K. (Eds.), Multifunctional Land Use – Meeting Future Demands for Landscape Goods and Services. Springer, Berlin, Heidelberg, New York, pp. 37–64.
Müller-Westermeier, G., Kreis, A., 2002. Die bodennahe Lufttemperatur und der Niederschlag im 20. Jahrhundert dargestellt anhand von Karten. In: Deutscher Wetterdienst (Ed.), Klimastatusbericht 2001. Offenbach am Main, pp. 12–19.
Müller-Westermeier, G., 1998. Die mittleren klimatologischen Bedingungen in Deutschland (Teil II). In: Deutscher Wetterdienst (Ed.), Klimastatusbericht 1998. Offenbach am Main, pp. 30–32.
Müller-Westermeier, G., 1995. Numerisches Verfahren zur Erstellung klimatologischer Karten. Berichte des Deutschen Wetterdienstes, vol. 193. Offenbach am Main.
Nuissl, H., Rink, D., 2005. The 'production' of urban sprawl in eastern Germany as a phenomenon of post-socialist transformation. Cities 22 (2), 123–134.
Opschoor, H., 2000. The ecological footprint: measuring rod or metaphor? Ecological Economics 32, 363–365.
Paetzold, A., Warren, P.H., Maltby, L.L., 2010. A framework for assessing ecological quality based on ecosystem services. Ecological Complexity 7, 273–281.
Petrosillo, I., Zaccarelli, N., Semeraro, T., Zurlini, G., 2009. The effectiveness of different conservation policies on the security of natural capital. Landscape and Urban Planning 89 (1–2), 49–56.
Potschin, M., Haines-Young, R., 2006. "Rio+10″, sustainability science and landscape ecology. Landscape and Urban Planning 75, 162–174.
Rees, W.E., 1992. Ecological footprints and appropriated carrying capacity: what urban economics leaves out. Environment and Urbanization 4 (2), 121–130.
Rees, W.E, 1996. Revisiting carrying capacity: area-based indicators of sustainability. Population and Environment 17 (3), 195–215.
Saxon Upper Mining Authority, 2002. Jahresbericht 2002, Freiberg.
Saxon Upper Mining Authority, 2007. Jahresbericht 2007, Freiberg.
Saxon State Department of the Environment and Geology, 2004. Der Braunkohlebergbau im Südraum Leipzig, Bergbau in Sachsen Band 11, Dresden.
Saxon State Ministry of the Environment and Agriculture, 2007. Energie für die Zukunft, Sachsens Potenzial an nachwachsenden Rohstoffen/Biomasse, Dresden.
Schmidt, T., Osterburg, B., 2009. Aufbau des Berichtsmoduls Landwirtschaft und Umwelt in den Umweltökonomischen Gesamtrechnungen II, Table Part of the Final Report. R&D project financed by the German Federal Statistical Office, Braunschweig.
Schneider, A., Woodcock, C.E., 2008. Compact, dispersed, fragmented, extensive? a Comparison of urban growth in twenty-five global cities using remotely sensed data. Pattern Metrics and Census Information. Urban Studies 45 (3), 659–692.
Solon, J., 2009. Spatial context of urbanization: Landscape pattern and changes between 1950 and 1990 in the Warsaw metropolitan area, Poland. Landscape and Urban Planning 93 (3–4), 350–361.
Stoll, R.D., Niemann-Delius, C., Drebenstedt, C., Müllensiefen, K. (Eds.), 2009. Der Braunkohlentagebau. Springer, Heidelberg, Berlin.
Swart, R., Raskin, P., Robinson, J., 2002. Critical challenges for sustainability science. Science 297, 1994.
TEEB, 2009. The Economics of Ecosystems and Biodiversity: The Ecological and Economic Foundations (Date: 11.02.2010) www.teeb-web.org.
Templet, P.H., 2000. Externalities, subsidies and the ecological footprint: an empirical analysis. Ecological Economics 32, 381–383.
Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46, 493–510.
Troy, A., Wilson, M.A., 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. Ecological Economics 60, 435–449.
United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development and World Bank, 2003. Handbook of National Accounting System of Integrated Environmental and Economic Accounting (SEEA). United Nations.
UNPD, 2005. World Urbanization Prospects: The 2005 Revision. United Nations Population Division, New York.
Van den Bergh, J., Verbruggen, H., 1999. Spatial sustainability, trade and indicators: an evaluation of the 'ecological Footprint'. Ecological Economics 29, 61–72.
Vandewalle, M., Sykes, M.T., Harrison, P.A., Luck, G.W., Berry, P., Bugter, R., et al., 2009. Review Paper on Concepts of Dynamic Ecosystems and Their Services (Date: 11.05.2009) http://www.rubicode.net/rubicode/RUBICODE_Review_on_Ecosystem_Services.pdf.
Vattenfall Europe, 2009. Vattenfall Europe AG. EEG-Anlagenstammblatt (Date: 12.10.2009) www.vattenfall.de/cps/rde/xchg/trm_de/hs.xsl/166.htm.
Wackernagel, M., Kitzes, J., Moran, D., Goldfinger, S., Thomas, M., 2006. The ecological footprint of cities and regions: comparing resource availability with resource demand. Environment and Urbanization 18, 103–112.
Wackernagel, M., Rees, W.E., 1996. Our Ecological Footprint: Reducing Human Impact on the Earth. New Society Publishers, Philadelphia.
Wallace, K., 2007. Classification of ecosystem services: problems and solutions. Biological Conservation 139, 235–246.
Wallace, K., 2008. Ecosystem services: multiple classifications or confusion? Biological Conservation 141, 353–354.
Webster, A.J.F., 1980. The energetic efficiency of growth. Livestock Production Science 7 (5), 243–252.
Wenk, C., Pfirter, H.P., Bickel, H., 1980. Energetic aspects of feed conversion in growing pigs. Livestock Production Science 7 (5), 483–495.
Wessolek, G., Trinks, S., Duijnisveld, W.H.M., 2004. Entwicklung von Boden- und nutzungsspezifischen Regressionsgleichungen zur Ermittlung der Sickerwasserrate aus dem Boden: Das TUB-BGR-Verfahren, Bundesanstalt für Geowissenschaften und Rohstoffe, Geozentrum Hannover, TU Berlin, Berlin, Hannover.
Williams, J., 2005. One-person households – a resource time bomb? In: Tiezzi, E., Brebbia, C.A., Jorgensen, S.E., Gomar, D.A. (Eds.), Ecosystems and Sustainable Development, vol. 81, pp. 409–417.
Zheng, X.P., 1991. Metropolitan spatial structure and its determinants: a case-study of Tokyo. Urban Studies 28 (1), 87–104.