Ascertainment and Assessment of ES

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4.1 Indicators and Quantification Approaches[1]

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4.1.1 Introduction

The need for applications and tools of the–frequently mainly conceptually used–ecosystem service (ES) ideas has become more and more obvious during the last years (Daily et al. 2009). Practical applications are necessary to further develop and improve the conceptual base of ES on the one hand. On the other, tools for environmental and resource management are needed in order to further establish ES in decision-making processes (Kienast et al. 2009). The recognition and the appropriate quantification of ES are fundamentals for their valuation, independently whether the valuation is conducted with biophysical, social or economic methods. Their application and integration is one of the biggest challenges of contemporary ES science (Wallace 2007).

The supply of ES is based on geo-biophysical structures and processes, which are changing in intensity as well as in spatial and temporal distribution. Anthropogenic impacts, especially land-use and land-cover changes or climatic variations are among the major factors determining the qualities and quantities of ES supply. Land-use patterns and changes in land cover can be surveyed, spatially analysed and regionally assessed. They deliver direct measures for human activities (Riitters et al. 2000) and clearly demonstrate the relations between ES supply and demand (Burkhard et al. 2012). Spatially explicit identification and mapping of ES distributions and the analysis of their spatio-temporal dynamics therefore enable the aggregation of highly complex information. The respective ES visualisations can support decision-makers in the environmental sector by providing powerful tools to support sustainable landscape planning and ES trade-off assessments (Swetnam et al. 2010). Spatially explicit ES quantification and mapping have therefore been named as one of the key requirements for the implementation of the ES concept in environmental institutions and decision-making processes (Daily and Matson 2008).

One key problem of each ES quantification is, besides the difficult and comprehensive data acquisition, the selection of an ES categorisation system which is appropriate for the specific study region and the particular research question. Most of the currently available ES classification systems (e.g. de Groot et al. 2010a; Wallace 2007) distinguish the three classes with regulating ES, provisioning ES and cultural ES. Some authors additionally include *habitat services* (de Groot et al. 2010a; TEEB 2010). Habitat services are, however, often assigned to the category of ecosystem functions, which in the Millennium Ecosystem Assessment (MEA 2005a) were called *supporting ecosystem services*. Many ecosystem functions or habitat properties do not deliver direct or final ES. Therefore, the distinction between ecosystem functions and ES has become more common and accepted. This distinction also proved to be advantageous for the avoidance of double counting of closely correlating functions and services, for example in monetary valuations.

Numerous methods and tools for the characterisation of ecosystem functions and services in landscapes have been developed especially within the last 10 years. Additionally, existing methods and data collection programmes are ready to be integrated in the ES concept due to their thematic diversity (e.g. monitoring within the long-term ecological research (LTER) network; Müller et al. 2010). They include measurements, monitoring programmes, mapping activities, expert interviews, statistical analyses, model applications or transfer-functions (de Groot et al. 2010b). Natural structures and processes (e.g. flows of energy, matter and water) are central in biophysical assessments. These approaches are different from monetary valuations, where the actual assessment of values is carried out by monetisation. Monetary ES approaches such as cost-benefit analyses (CBA) or willingness-to-pay (WTP) surveys are applicable and well-established concepts (Farber et al. 2002). However, results are often disappointing especially for nonmarket goods and services such as many regulating ES, ecosystem functions or biodiversity characteristics (Ludwig 2000; Spangenberg and Settele 2010).

1 Section 4.1 is in main parts based on the paper of Burkhard et al. (2012).

Suitable ES indicators are needed for all quantification approaches. These indicators have to be quantifiable, sensitive for land-use changes, temporally and spatially explicit and scalable (van Oudenhoven et al. 2012). Indicators are tools for communication, enabling the reduction of information about highly complex human-environmental systems. After Wiggering and Müller (2004), indicators in general are variables delivering aggregated information about certain phenomena. They are selected to support specific management purposes by providing integrating synoptic values, depicting not directly accessible qualities, quantities, states or interactions (Dale and Beyeler 2001; Turnhout et al. 2007; Niemeijer and de Groot 2008).

4.1.2 Ecosystem Service Supply and Demand Assessment at the Landscape Scale–the 'Matrix'

Different landscapes can be characterised by different ecosystem structures, functions and consequently by varying capacities to supply ES (Burkhard et al. 2009), depending on the natural settings as well as human activities (e.g. land use) within the research area. Different land-use patterns, heterogeneous population distributions as well as multiple ecological and socio-economic conditions cause varying demands for ES (► Fig. 3.2).

In this chapter, a method for the assessment of different land-cover types' capacities to support ecosystem functions (assessed based on the ecological integrity concept and respective indicators for ecosystem structures and processes; for detailed information see Müller 2005; Burkhard et al. 2009, 2012), to supply multiple ES and to identify demands for ES will be shortly introduced. The method has been applied in different case studies, for example for the assessment of ES in boreal forest landscapes in northern Finland (Vihervaara et al. 2010), in urban–rural regions in central eastern Germany (Kroll et al. 2012) or for the calculation of flood regulation capacities in a Bulgarian mountainous region (Nedkov and Burkhard 2012).

The approach is based on an *assessment matrix*, which links relative and mainly non-monetary ES supply capacities or ES demand intensities to different geospatial units (e.g. different land-cover types). Based on this interrelation analysis, resulting ecosystem function and ES scores can be visualised in maps. Differentiations between ES supply and demand but also between ES potential and *de facto* flows (ES actually used by humans) are needed (see below). Supply and demand of/for different ecosystem goods and services are often spatially and temporally decoupled and managed by transport, trade and storage opportunities in today's globalised world. Nevertheless, calculations of these two variables deliver data that are highly relevant for ES budget assessments for specific spatial or temporal units. Self-sufficiency rates and ES flows within and between regions can be calculated on this basis. Ecosystem functions and several regulating ES such as nutrient regulation, erosion control and natural hazard protection are exceptions. They are normally not transportable and therefore, a physical connection between the service providing unit (SPU) and service benefiting/demand area (SBA) must exist (Nedkov and Burkhard 2012; Syrbe and Walz 2012; ► Sect. 3.3).

Such information, especially in a regionalised form, and the related ecological and socio-economic data are highly relevant for environmental management and for ES-based landscape planning. Thus, requests for appropriate tools are numerous (Kienast et al. 2009). When assessing the *potential* of a landscape, a land-use type or an ecosystem, usually the (hypothetical) maximum of ES supply under the given conditions is being assessed. Often it is not considered whether there is a human use of these ES or not. *Flows* of ES on the contrary describe the capacity of a defined spatial unit to supply a specific ES set (ES bundle) actually used by humans within a given time period (after Burkhard et al. 2012; see Box). This distinction becomes relevant for certain ES, for example when assessing protected ecosystems. These systems undoubtedly supply numerous goods and services. However, e.g. in the case of core zones in national parks, where any human activity may be prohibited, many of these ES (e.g. timber, game) cannot be used. Of course, ecosystem functions, such as nutrient cycling or biodiversity, take place anyway. They provide positive effects on ecological integrity within the protected area itself, but often also on adjacent ecosystems.

Conceptual Background for ES Supply and Demand (after Burkhard et al. 2012)

- *ES supply* refers to the capacity of a particular area to provide a specific bundle of ecosystem goods and services within a given time period. For detailed analyses, a differentiation between ES potentials and actual ES flows is needed.
- *ES demand* is the sum of all ecosystem goods and services currently consumed or used in a particular area over a given time period. Up to now, demands are assessed not considering where ecosystem services actually are provided. These detailed provision patterns are part of the
- *ES footprint* which (closely related to the ecological footprint concept; Rees 1992) calculates the area needed to generate particular ecosystem goods and services demanded by humans in a certain area and a certain time. Different aspects of ecosystem service generation are considered (production capacities, waste absorption, etc.) for assessing the ES footprint.

For many regulating ES, it can be assumed that ES potentials and flows are comparable (► Sect. 2.1).

Ecosystem functions, ES supply, ES demand and ES budgets in different land-use types can be assessed by the help of *ES matrices*. The first matrix in ◘ Fig. 4.1 contains ecosystem functions (ecological integrity) and ES on the *x*-axis. The geospatial units (here CORINE land-cover types; EEA 1994) are placed on the *y*-axis (after Burkhard et al. 2009, 2012). All relevant ES capacity scores are entered, using a relative scale between 0 (equivalent to no relevant capacity to support the respective ecosystem function or to supply the respective ES), 1 (low relevant capacity), 2 (relevant capacity), 3 (medium relevant capacity), 4 (high relevant capacity) and 5 (maximum capacity in the study area) at the intersections. Based on the 44 different CORINE land-cover classes and 39 ecosystem functions and services, altogether 1716 capacity scores have to be given (◘ Fig. 4.1). Due to this high number of scores needed and the related high assessment efforts, existing databases or expert evaluations need to be harnessed. These data can successively be checked and replaced by more exact information resulting from modelling, measurement, monitoring or in-depth interviews (Burkhard et al. 2009).

The matrix in ◘ Fig. 4.1 shows clear patterns of ES capacity distributions across the different land-cover types. Especially, the forest land-cover types (including broad-leaved, coniferous and mixed forests) show high scores for a multitude of ES. Such multifunctionality is typical for forest ecosystems. Also the other generally more natural land-cover types such as natural grasslands, wetlands and water bodies are characterised by high ES capacities. Strongly anthropogenically influenced ecosystems, such as urban fabrics, industrial or commercial units and transport units (in the upper part of the matrix), show comparably low ES capacities. Of course, these areas also supply ES, but in comparison with the other land-cover types, their ES supply is rather low (► Sect. 6.4).

The whole ES concept is a highly anthropocentric approach. Fisher et al. (2009) defined that only those services with a clear benefit to human societies can be denoted as ES. Services without direct human benefits should be termed as ecosystem functions or intermediate services. Thus, a societal demand should be identifiable for all individual ES. Data about actual anthropogenic uses of each ES are needed for their assessment (see definitions in Box 1). Major parts of this information can be derived from statistics, modelling, ecological and socio-economic monitoring or from interviews. ◘ Figure 4.2 shows a respective matrix, which, comparable to the ES supply matrix (◘ Fig. 4.1), provides exemplary information about the ES demands within the different CORINE land-cover classes. The *y*-axis contains regulating, provisioning and cultural ES. The ecological integrity variables are not relevant here because they (per definition) do not provide direct benefits to humans. The scores were given in a similar manner as in the ES supply matrix; 0 (light pink) denotes no relevant human demand within the particular land-cover type and 5 (dark red) illustrates maximum demand.

◘ Figure 4.2 clearly shows that the overall highest demands for manifold ES are located within the

		Ecological integrity								Regulating services							
		1 Exergy capture	2 Entropy production	3 Storage capacity	4 Cycling & Nutrient loss reduction	5 Biotic water flows	6 Metabolic efficiency	7 Heterogeneity	8 Biodiversity	9 Global climate regulation	10 Local climate regulation	11 Air quality Regulation	12 Water flow regulation	13 Water purification	14 Nutrient regulation	15 Erosion regulation	16 Natural hazard regulation
1	Continuous urban fabric	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2	Discontinuous urban fabric	1	1	1	1	1	1	1	1	0	0	0	0	0	0	0	0
3	Industrial or commercial units	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0
4	Road and rail networks	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0
5	Port areas	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	3
6	Airports	1	1	0	2	1	1	1	1	0	0	0	0	0	0	0	0
7	Mineral extraction sites	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0
8	Dump sites	0	0	5	0	0	0	2	1	0	0	0	0	0	0	0	0
9	Construction sites	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0
10	Green urban areas	4	3	2	3	2	1	3	3	1	2	1	2	1	1	2	0
11	Sport and leisure facilities	4	3	2	3	2	1	2	2	1	1	1	2	1	1	1	0
12	Nonirrigated arable land	5	4	4	1	3	4	3	2	1	2	0	1	0	0	0	1
13	Permanently irrigated land	5	4	3	1	5	2	3	2	1	3	0	0	0	0	0	1
14	Ricefields	5	4	3	1	5	1	3	2	0	2	0	2	0	0	0	0
15	Vineyards	3	2	2	0	3	1	3	2	1	1	0	1	0	0	0	0
16	Fruit trees and berries	3	2	3	2	4	2	4	3	2	2	2	2	1	1	2	2
17	Olive groves	3	2	3	1	3	2	3	2	1	1	1	1	1	1	1	0
18	Pastures	5	5	4	2	4	5	2	2	1	1	0	1	0	0	4	1
19	Annual and permanent crops	4	3	3	2	3	2	2	2	1	2	1	1	0	0	1	1
20	Complex cultivation patterns	4	3	3	1	3	2	4	3	1	2	0	1	0	0	0	1
21	Agriculture & Natural vegetation	3	2	3	2	3	2	3	3	2	3	1	2	1	0	3	1
22	Agro-forestry areas	4	3	4	4	4	3	4	4	1	2	1	1	1	1	2	1
23	Broad-leaved forest	5	4	5	5	5	4	3	4	4	5	5	2	5	5	5	3
24	Coniferous forest	5	4	5	5	4	4	3	4	4	5	5	2	5	5	5	3
25	Mixed forest	5	4	5	5	5	4	3	5	4	5	5	2	5	5	5	3
26	Natural grassland	4	3	5	5	4	4	3	5	3	2	0	1	5	5	5	1
27	Moors and heathland	4	3	5	5	4	5	3	4	3	4	0	2	4	3	0	2
28	Sclerophyllous vegetation	3	2	2	4	2	3	3	4	1	2	0	1	0	0	0	1
29	Transitional woodland shrub	3	2	2	4	2	3	3	4	0	1	0	0	0	0	0	0
30	Beaches, dunes and sand plains	1	0	1	0	1	1	3	3	0	0	0	1	0	0	0	5
31	Bare rock	0	0	0	0	0	0	3	3	0	0	0	1	1	0	0	1
32	Sparsely vegetated areas	1	1	1	1	1	0	2	3	0	1	0	1	0	0	0	1
33	Burnt areas	0	0	3	0	0	0	2	1	0	1	0	0	0	0	0	0
34	Glaciers and perpetual snow	0	0	0	0	0	0	2	1	3	3	0	4	0	0	0	0
35	Inland marshes	4	3	5	3	4	4	3	2	2	2	0	2	0	4	0	4
36	Peatbogs	4	3	5	5	4	4	3	4	5	4	0	3	4	3	0	3
37	Salt marshes	3	2	5	3	4	3	2	3	0	1	0	0	0	2	0	5
38	Salines	0	0	0	0	0	0	1	1	0	2	0	0	0	0	0	0
39	Intertidal flats	1	1	1	4	0	2	2	3	0	1	0	0	0	1	0	5
40	Water courses	3	1	1	3	0	3	4	4	0	1	0	1	3	3	0	2
41	Water bodies	4	2	4	3	0	4	4	4	1	2	0	2	0	1	0	1
42	Coastal lagoons	5	4	4	3	0	5	4	4	0	1	0	0	0	0	0	4
43	Estuaries	5	4	2	3	0	5	3	3	0	0	0	0	3	3	0	3
44	Sea and ocean	3	2	1	4	0	3	2	2	5	3	0	0	0	5	0	0

					Provisioning services														Cultural services					
		17 Pollination *	18 Pest and disease control *	19 Regulation of waste *	20 Crops	21 Biomass for energy	22 Fodder **	23 Livestock (domestic)	24 Fibre	25 Timber	26 Wood Fuel	27 Fish, seafood & edible algae	28 Aquaculture	29 Wild foods & resources	30 Biochemicals & Medicine	31 Freshwater	32 Mineral resources ***	33 Abiotic energy sources ***	34 Recreation & Tourism	35 Landscape aesthetics & Inspiration	36 Knowledge systems	37 Religious & Spiritual experience	38 Cultural heritage & Cultural diversity	39 Natural heritage & Natural diversity
1	Continuous urban fabric	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	3	3	1	5	1	0
2	Discontinuous urban fabric	0	0	0	1	1	1	0	0	0	0	0	0	1	0	0	0	1	3	2	1	5	1	0
3	Industrial or commercial units	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	1	0
4	Road and rail networks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
5	Port areas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	0	1	0
6	Airports	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
7	Mineral extraction sites	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	5	0	0	1	0	0	0
8	Dump sites	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
9	Construction sites	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
10	Green urban areas	1	1	1	0	0	0	0	0	0	1	0	0	1	0	0	0	0	3	3	1	0	2	0
11	Sport and leisure facilities	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	1	0	0	1	0
12	Nonirrigated arable land	0	2	2	5	2	3	0	5	0	0	0	0	0	1	0	0	0	1	1	2	0	3	0
13	Permanently irrigated land	0	1	2	5	1	2	0	2	0	0	0	0	0	1	0	0	0	1	1	2	0	3	0
14	Ricefields	0	1	1	5	0	2	0	0	0	0	0	0	0	0	0	0	0	1	1	2	0	4	0
15	Vineyards	0	1	1	4	1	0	0	0	0	1	0	0	0	0	0	0	0	5	2	2	0	5	0
16	Fruit trees and berries	5	3	2	5	1	0	0	0	4	4	0	0	0	0	0	0	0	5	2	2	0	4	0
17	Olive groves	0	3	2	4	1	0	0	0	4	4	0	0	0	0	0	0	0	5	2	2	0	4	0
18	Pastures	0	2	4	0	1	5	5	0	0	0	0	0	0	0	0	0	0	3	2	2	0	3	0
19	Annual and permanent crops	0	2	2	5	1	5	5	5	0	0	0	0	0	1	0	0	0	1	1	2	0	2	0
20	Complex cultivation patterns	0	3	2	4	1	3	0	4	0	0	0	0	0	2	0	0	0	2	2	2	0	3	0
21	Agriculture & Natural vegetation	0	3	2	3	2	2	3	4	3	3	0	0	3	1	0	0	1	2	2	3	0	2	3
22	Agro-forestry areas	3	3	3	3	2	2	3	2	3	3	0	0	0	0	0	0	0	3	2	2	0	3	0
23	Broad-leaved forest	5	4	4	0	1	1	0	0	5	5	0	0	5	5	0	0	0	5	5	5	3	3	5
24	Coniferous forest	5	4	4	0	1	1	0	0	5	5	0	0	5	5	0	0	0	5	5	5	3	4	5
25	Mixed forest	5	5	5	0	1	1	0	0	5	5	0	0	5	5	0	0	0	5	5	5	3	4	5
26	Natural grassland	0	1	2	0	0	1	3	0	0	0	0	0	5	0	0	0	0	3	4	5	1	4	3
27	Moors and heathland	2	2	3	0	2	0	2	0	0	2	0	0	1	0	0	0	0	5	4	5	1	2	5
28	Sclerophyllous vegetation	2	2	3	0	0	0	2	0	0	2	0	0	1	3	0	0	0	2	3	4	1	2	4
29	Transitional woodland shrub	2	2	3	0	1	0	2	1	0	2	0	0	1	0	0	0	0	2	3	4	1	2	2
30	Beaches, dunes and sand plains	0	1	1	0	1	0	0	0	0	0	0	0	0	0	0	1	0	5	4	4	0	2	2
31	Bare rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	4	3	4	0	2	0
32	Sparsely vegetated areas	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	0	2	0
33	Burnt areas	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
34	Glaciers and perpetual snow	0	1	1	0	0	0	0	0	0	0	0	0	0	0	5	0	0	5	5	3	2	0	0
35	Inland marshes	0	2	3	0	0	5	2	0	0	0	0	0	0	0	0	0	0	0	2	4	0	2	0
36	Peatbogs	2	3	4	0	2	0	0	0	0	0	0	0	0	0	0	0	0	4	2	3	0	2	4
37	Salt marshes	0	2	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	3	2	3	0	2	0
38	Salines	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	3	0	2	0
39	Intertidal flats	0	2	3	0	1	0	0	0	0	0	0	0	0	0	0	0	0	4	2	3	0	2	0
40	Water courses	0	3	5	0	3	0	0	0	0	0	3	0	4	0	5	0	5	4	4	4	0	3	5
41	Water bodies	0	3	5	0	0	0	0	0	0	0	3	5	4	0	5	0	5	5	4	4	0	3	4
42	Coastal lagoons	0	3	5	0	1	0	0	0	0	0	4	5	4	0	0	0	0	4	4	4	0	2	4
43	Estuaries	0	4	5	0	2	0	0	0	0	0	5	5	4	0	0	0	0	4	5	4	0	2	3
44	Sea and ocean	0	3	5	0	3	1	0	0	0	0	5	5	0	0	0	1	3	4	5	4	0	4	2

* These ecosystem services are named because they can be of high importance in some ecosystems although the potential of double-counting must be noted.
** Potential double-counting when fodder is used for feeding on the same farm.
*** These services are often not counted as ecosystem services; but they can be of high importance for policy decisions, land-use management strategies and scenarios.

Fig. 4.1 Land-cover types (*y-axis*) and ecosystem functions and services (*x-axis*) illustrating the capacities of different land-cover types to support ecosystem functions and to supply ES on a scale from 0 (no relevant capacity; *pink*) to 5 (maximum relevant capacity; *dark green*); exemplarily assessed for a central European 'normal landscape' (after Burkhard et al. 2009, 2012)

highly human-modified land-cover types in the upper part of the matrix. Urban areas as well as industrial and commercial areas are the land-cover types with the highest demand scores. It also becomes obvious that in the more natural land-cover types (lower part of the matrix), generally lower demands for ES can be found. This can of course be justified by the lower population numbers and related lower consumption rates in these areas. Agrarian land-cover types show high demands for regulating ES (e.g. nutrient regulation, water purification, erosion control). Similarly to the ES supply matrix, ES demand maps can also be compiled based on the ES demand matrix.

Taking the information from the ES supply and demand matrices as starting points, sources and sinks for individual ES can be identified. As both components–supply and demand–were normalised to the same relative units (0–5), ES budgets can be calculated by subtracting the ES demand scores from the ES supply scores. And also the resulting ES budget scores can be illustrated in a matrix and

	Regulating services											Provisioning services														Cultural services					
	9 Global climate regulation	10 Local climate regulation	11 Air quality regulation	12 Water flow regulation	13 Water purification	14 Nutrient regulation	15 Erosion regulation	16 Natural hazard regulation	17 Pollination	18 Pest and disease control	19 Regulation of waste	20 Crops	21 Biomass for energy	22 Fodder	23 Livestock	24 Fibre	25 Timber	26 Wood fuel	27 Fish, seafood & edible algae	28 Aquaculture	29 Wild foods & resources	30 Biochemicals / medicine	31 Freshwater	32 Mineral resources	33 Abiotic energy sources	34 Recreation& tourism	35 Landscape aesthetics	36 Knowledge systems	37 Religious experience	38 Cultural heritage & diversity	39 Natural heritage & diversity
1 Continuous urban fabric	3	5	5	4	1	1	1	5	3	5	3	5	5	1	5	3	3	2	5	5	5	5	5	4	2	4	4	3	4	4	2
2 Discontinuous urban fabric	3	5	5	5	2	2	1	4	4	4	2	4	4	2	4	3	3	3	4	4	4	5	5	3	3	4	4	3	3	2	3
3 Industrial or commercial units	5	1	5	4	3	3	1	5	4	3	4	5	5	5	5	5	5	5	4	4	4	5	5	5	5	1	1	4	1	3	1
4 Road and rail networks	4	2	4	4	0	0	3	4	1	2	0	0	4	0	0	0	2	0	0	0	0	0	1	2	0	2	2	1	1	1	0
5 Port areas	3	2	2	5	3	0	4	5	1	4	3	2	5	2	2	2	5	2	2	2	1	1	3	3	1	2	2	2	1	2	1
6 Airports	5	2	4	1	2	1	1	5	0	5	1	2	5	0	2	1	1	0	1	1	1	1	3	2	0	1	1	1	1	1	0
7 Mineral extraction sites	0	0	0	2	0	0	4	3	0	0	3	0	3	0	0	1	2	0	0	0	0	0	2	1	0	0	0	0	0	0	0
8 Dump sites	2	2	3	0	2	0	0	5	0	3	5	0	1	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
9 Construction sites	0	2	1	2	2	2	2	3	0	1	2	0	4	0	0	4	4	0	0	0	0	0	2	4	0	0	0	0	0	0	0
10 Green urban areas	0	2	1	0	0	0	0	2	2	3	0	1	1	0	1	0	0	0	0	0	0	0	2	0	0	4	4	2	0	2	1
11 Sport and leisure facilities	0	2	3	0	1	0	0	3	0	3	0	2	3	1	2	1	1	1	2	2	2	3	3	1	1	3	3	1	0	2	0
12 Nonirrigated arable land	2	2	1	2	0	3	2	3	3	3	2	1	1	0	0	0	0	0	0	0	0	1	0	0	1	0	0	1	0	1	0
13 Permanently irrigated land	2	2	1	2	5	3	2	3	3	3	2	1	2	0	0	0	0	0	0	0	0	1	5	0	1	0	0	1	0	1	0
14 Ricefields	4	3	1	5	5	3	5	3	1	3	2	1	2	0	0	0	0	0	0	0	0	1	5	0	0	0	0	2	0	3	0
15 Vineyards	2	5	1	0	4	3	5	3	2	3	1	1	2	0	0	1	1	0	0	0	0	2	4	0	0	0	0	2	0	3	0
16 Fruit trees and berries	1	2	1	0	2	3	1	3	5	3	1	1	2	0	0	1	1	0	0	0	0	2	3	0	0	0	0	2	0	2	0
17 Olive groves	1	2	1	0	2	2	0	3	2	3	1	1	1	0	0	1	0	0	0	0	0	2	1	0	0	0	0	2	0	2	0
18 Pastures	3	1	0	1	2	1	0	2	0	1	3	0	1	3	1	0	1	0	0	0	0	1	2	0	2	0	0	1	0	1	0
19 Annual and permanent crops	1	1	1	1	2	5	1	2	2	3	1	1	2	0	0	0	0	0	0	0	0	1	1	0	1	0	0	1	0	1	0
20 Complex cultivation patterns	1	1	1	1	2	5	1	2	3	3	2	1	2	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	0	1	0
21 Agriculture & natural vegetation	2	1	1	0	2	3	1	1	2	3	1	1	2	0	0	0	0	0	0	0	0	1	2	0	2	0	0	1	0	0	0
22 Agro-forestry areas	1	1	1	0	2	3	0	1	2	1	0	1	1	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0
23 Broad-leaved forest	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0
24 Coniferous forest	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0
25 Mixed forest	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0
26 Natural grassland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
27 Moors and heathland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0
28 Sclerophyllous vegetation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
29 Transitional woodland shrub	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
30 Beaches, dunes and sand plains	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	1
31 Bare rock	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
32 Sparsely vegetated areas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
33 Burnt areas	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
34 Glaciers and perpetual snow	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
35 Inland marshes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
36 Peatbogs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
37 Salt marshes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
38 Salines	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0
39 Intertidal flats	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
40 Water courses	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	3	0	0	0	0	0	0
41 Water bodies	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
42 Coastal lagoons	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
43 Estuaries	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
44 Sea and ocean	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	4	0	0	0	0	0	0

◘ **Fig. 4.2** Demand for ecosystem services (*x-axis*) within different land-cover types (*y-axis*) on a scale from 0 (no relevant demand; *light pink*) to 5 (maximum demand; *dark red*); exemplarily assessed for a central European 'normal landscape' (after Burkhard et al. 2012)

in maps. ◘ Figure 4.3 shows the ES budget matrix for the different CORINE land-cover types. Each field in the ES budget matrix was calculated based on the scores in the ES supply matrix (◘ Fig. 4.1) and the ES demand matrix (◘ Fig. 4.2). Therefore, the assessment scale ranges from −5 = demand clearly exceeds supply (undersupply), via 0 = demand = supply (neutral budget), to +5 = supply clearly exceeds demand (oversupply). Empty fields indicate land-cover types with neither a relevant ES supply nor a relevant demand for ES.

◘ Figure 4.3 shows a clear pattern of ES undersupply in the regions with high anthropogenic

		Regulating services											Provisioning services														Cultural services					
		9 Global climate regulation	10 ocal climate regulation	11 Air quality regulation	12 Water flow regulation	13 Water purification	14 Nutrient regulation	15 Erosion regulation	16 Natural hazard regulation	17 Pollination	18 Pest and disease control	19 Regulation of waste	20 Crops	21 Biomass for energy	22 Fodder	23 Livestock	24 Fibre	25 Timber	26 Wood fuel	27 Capture fisheries	28 Aquaculture	29 Wild foods & resources	30 Biochemicals / medicine	31 Freshwater	32 Mineral resources	33 Abiotic energy sources	34 Recreation& tourism	35 Landscape aesthetics	36 Knowledge systems	37 Religious experience	38 Cultural heritage & diversity	39 Natural heritage & diversity
1	Continuous urban fabric	-3	-5	-5	-4	-1	-1	-1	-5	-3	-5	-3	-5	-4	-1	-5	-3	-3	-2	-5	-5	-5	-5	-5	-4	-1	-1	-1	-2	1	-3	-2
2	Discontinuous urban fabric	-3	-5	-5	-5	-2	-2	-1	-4	-4	-4	-2	-3	-3	-1	-4	-3	-3	-3	-4	-4	-3	-5	-5	-3	-2	-1	-2	-2	2	-1	-3
3	Industrial or commercial units	-5	-1	-5	-4	-3	-3	-1	-5	-4	-3	-4	-5	-4	-5	-5	-5	-5	-5	-4	-4	-4	-5	-5	-5	-4	-1	0	-4	-1	-2	-1
4	Road and rail networks	-4	-2	-4	-4	0	0	-3	-4	-1	-2			-4				-2						-1	-2		-2	-2	-1	-1	0	
5	Port areas	-3	-2	-2	-5	-3		-4	-2	-1	-4	-3	-2	-5	-2	-2	-2	-5	-2	-2	-2	-1	-1	-3	-3	-1	-1	0	-2	-1	-1	-1
6	Airports	-5	-2	-4	-1	-2	-1	-1	-5		-5	-1	-2	-5	1	-2	-1	-1	0	-1	-1	-1	-1	-3	-2		-1	-1	-1	-1	0	
7	Mineral extraction sites				-2			-4	-3			-3		-3			-1	-2						-2	4	5			1			
8	Dump sites	-2	-2	-3		-2			-5		-3	-5		0										-2								
9	Construction sites		-2	-1	-2	-2	-2	-2	-3		-1	-2		-4			-4	-4						-2	-3							
10	Green urban areas	1	0	0	2	1	1	2	-2	-1	-2	1	-1	-1		-1			1			1		-2			-1	-1	-1		0	-1
11	Sport and leisure facilities	1	-1	-2	2	0	1	1	-3	1	-2	1	-2	-3	-1	-2	-1	-1	-1	-2	-2	-2	-3	-3	-1	-1	2	-2	-1		-1	
12	Nonirrigated arable land	-1	0	-1	-1		-3	-2	-2	-3	-1	0	4	1	3		5						0			-1	1	1	1		2	
13	Permanently irrigated land	-1	1	-1	-2	-5	-3	-2	-2	-3	-2	0	4	-1	2		2						0	-5		-1	1	1	1		2	
14	Ricefields	-4	-1	-1	-3	-5	-3	-5	-3	-1	-2	-1	4	-2	2								-1	-5			1	1	0		1	
15	Vineyards	-1	-4	-1	1	-4	-3	-5	-3	-2	-2	0	3	-1			-1	-1	1				-2	-4			5	2	0		2	
16	Fruit trees and berries	1	0	1	2	-1	-2	1	-1	0	0	1	4	-1			-1	3	4				-2	-3			5	2	0		2	
17	Olive groves	0	-1	0	1	-1	-1	1	-3	-2	0	1	3	0			-1	4	4				-2	-1			5	2	0		2	
18	Pastures	-2	0		0	-2	-1	4	-1		1	1		0	2	4		-1					-1	-2		-2	3	2	1		2	
19	Annual and permanent crops	0	1	0	0	-2	-5	0	-1	-2	-1	1	4	-1	5	5	5						0	-1		-1	1	1	1		1	
20	Complex cultivation patterns	0	1	-1	0	-2	-5	-1	-1	-3	0	0	3	-1	3		4						1	-1			2	2	1		2	
21	Agriculture & natural vegetation	0	2	0	2	-1	-3	2	0	-2	0	1	2	0	2	3	4	3	3			3	0	-2		-1	2	2	2		2	3
22	Agro-forestry areas	0	1	0	1	-1	-2	2	0	1	2	3	2	1	2	3	2	3	3				-1	-2			3	2	2		3	
23	Broad-leaved forest	4	5	5	2	5	5	5	3	5	4	4		1	1			4	4			4	5				5	5	5	3	3	5
24	Coniferous forest	4	5	5	2	5	5	5	3	5	4	4		1	1			4	4			4	5				5	5	5	3	4	5
25	Mixed forest	4	5	5	2	5	5	5	3	5	5	5		1	1			4	4			4	5				5	5	5	3	4	5
26	Natural grassland	3	2		1	5	5	5	1		1	2			1	3						5				-2	3	4	5	1	4	3
27	Moors and heathland	3	4		2	4	3		2	2	2	3		2		2			2			1				-2	5	4	5	1	2	5
28	Sclerophyllous vegetation	1	2		1				1	2	2	3				2			2			1	3			-1	2	3	4	1	2	4
29	Transitional woodland shrub		1							2	2	3		1		2	1		2			1					2	3	4	1	2	2
30	Beaches, dunes and sand plains				1				5		1	1		1											1		4	3	4		2	1
31	Bare rock				1	1			1																1		4	3	4		2	
32	Sparsely vegetated areas		1		1				1		1	1														-1		1	4		2	
33	Burnt areas		1									1																	5			
34	Glaciers and perpetual snow	3	3		4						1	1												5			5	5	3	2		
35	Inland marshes	2	2		2		4		4		2	3			5	2												2	4		2	
36	Peatbogs	5	4		3	4	3		3	2	3	4		2													4	2	3		2	4
37	Salt marshes		1				2		5		2	2				2											3	2	3		2	
38	Salines		2								1	1															2	2	2		1	
39	Intertidal flats		1				1		5		2	3		1													4	2	3		2	
40	Water courses		1		1	3	3		2		3	5		3						2		4		5		2	4	4	4		3	5
41	Water bodies	1	2		2		1		1		3	5								2	5	4		5		5	5	4	4		3	4
42	Coastal lagoons		1						4		3	5		1						3	5	4					4	4	4		2	4
43	Estuaries					3	3		3		4	5		2						4	5	4					4	5	4		2	3
44	Sea and ocean	5	3				5				3	5		3	1					4	5				1	-1	4	5	4		4	2

Fig. 4.3 Ecosystem service supply-demand matrix showing budgets in the different land-cover types; based on matrices in Figs. 4.1 and 4.2. Scale from –5 (*dark red*) = demand clearly exceeds supply = undersupply; via 0 (*pink*) = demand = supply = neutral budget; to 5 (*dark green*) = supply clearly exceeds demand = oversupply. Empty fields indicate land cover types with neither a relevant ES supply nor a relevant demand for ES (after Burkhard et al. 2012)

influences, especially in the urbanised areas and the industrial and commercial units. The more natural land-cover types, particularly the forests, show characteristic patterns where the ES supply often exceeds the demand. More detailed information about the locations of actual ES supply (SPUs) and related flows to areas of ES demand (SBAs) could be integrated in ecosystem service footprint calculations (see Box 1). No experience with this approach is available up to now. Highly complex import and export balances would be needed, for which data on required scales are not easily available.

The following case study application from the central eastern German region Leipzig-Halle shows how empirical ES quantifications can be transferred to the relative 0–5 scale, and how the results can be illustrated in spatially explicit ES maps. The study took place as a part of the EU project PLUREL (*Peri-urban Land Use Relationships*, ► www.plurel.net/). More detailed information about the different ES quantification methods and the map compilation can be found in Kroll et al. (2012) and in Burkhard et al. (2009, 2012). The following maps from the Leipzig-Halle case study region include CORINE land-cover maps for the years 1990 and 2006 and spatial distributions of the provisioning ES 'energy' supply, demand and supply–demand budgets (◘ Figs. 4.4 and 4.5). The quantifications for the ES 'energy' refer to final energy units in gigajoule per hectare per year. Lignite as the major energy source in this region was included within the provisioning ES category. We are aware that current ecosystem functions are not involved in the generation of lignite and that the integration of natural resources is seen critical by many authors. We are following the CICES system (► http://cices.eu/) here, which includes abiotic outputs from natural systems in their accompanying ES classification. Moreover, open-pit lignite mining has enormous impacts on ecosystem structures and processes in the study area's landscapes. Thus, this ES is of high relevance for landscape planning and therefore cannot be neglected.

The energy supply map from the year 1990 (◘ Fig. 4.4, top right) shows that the large lignite open-pit mines were the only regional energy source at this time with a final energy contribution of 20,000 GJ ha^{-1} $year^{-1}$. In the year 2007 (◘ Fig. 4.5, top right), a clear reduction of the open-pit mine areas and their energetic outputs are visible. New energy sources such as wind power, biomass, solar energy or waterpower were developed, resulting in a more heterogeneous distribution of energy supply in the region.

The demands for the energy provisioning ES (◘ Figs. 4.4 and 4.5, bottom left) show a clear sink function of the industrial and commercial units and the urban areas. The pit mines themselves also have a high demand for energy. The demand for energy was generally decreasing by 20 % between 1990 and 2007, mainly due to the decline of energy-intensive industrial activities and energy saving measures. The ES supply–demand budget maps (◘ Figs. 4.4 and 4.5, bottom right) illustrate the abovementioned source-sink functions of the rural and urban areas. Based on such information and data, decisions for regional ES provision and landscape planning can be supported.

4.1.3 Conclusions and Outlook

The high applicability of the ES matrix approach presented here arises from its potential for visualisation and from the comparison of the effects of different land-use activities on ecosystem functions and services. Thereby, assessments of trade-offs between different land-use types are possible. Various ecosystem functions and services can be displayed and huge amounts of data resulting for example from expert interviews, statistics, measurements and modelling can be integrated. The normalisation to the standardised relative 0–5 scale integrates different biophysical dimensions (e.g. Joule, tons, diversity indices) or economic units (e.g. Euro, Yuan) and makes them (to a certain degree) comparable.

The application of freely available spatial data such as CORINE enables the coverage of large landscape units with a unified land-cover classification system in almost all European countries. Issues with the land-cover classification system, the spatial data resolution and generalisation problems lead to uncertainties of the assessments. Further data with higher spatio-temporal or thematic resolution can, like in the ES assessments, easily be integrated.

The matrix approach is also linked with technical and thematic uncertainties, especially if the majority of the ES scores are based on expert opinions. The uncertainties are based upon the selection of a suitable and representative case study area, the selection of relevant land-cover classes (matrix *y*-axis), spatial and geo-biophysical data acquisition, the selection of relevant ecosystem functions and services (matrix *x*-axis) and related indicators, the indicator quantification in the matrices based on

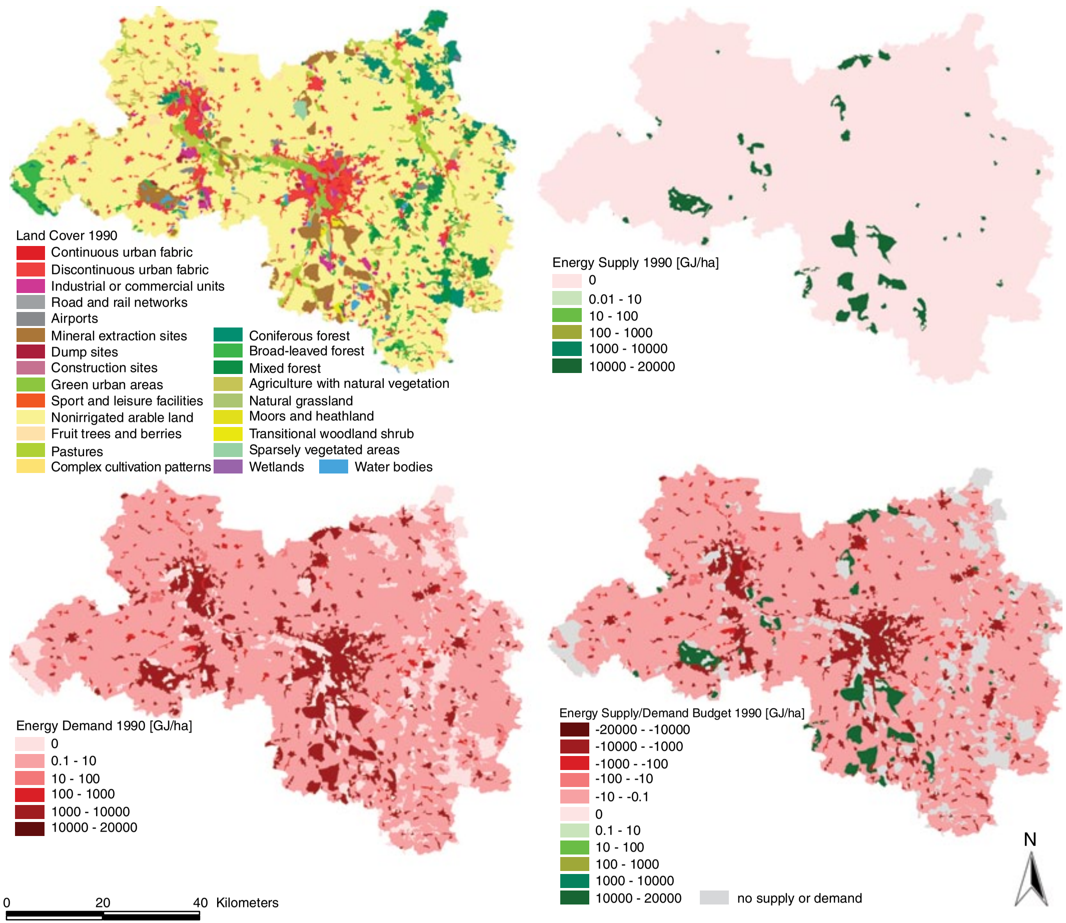
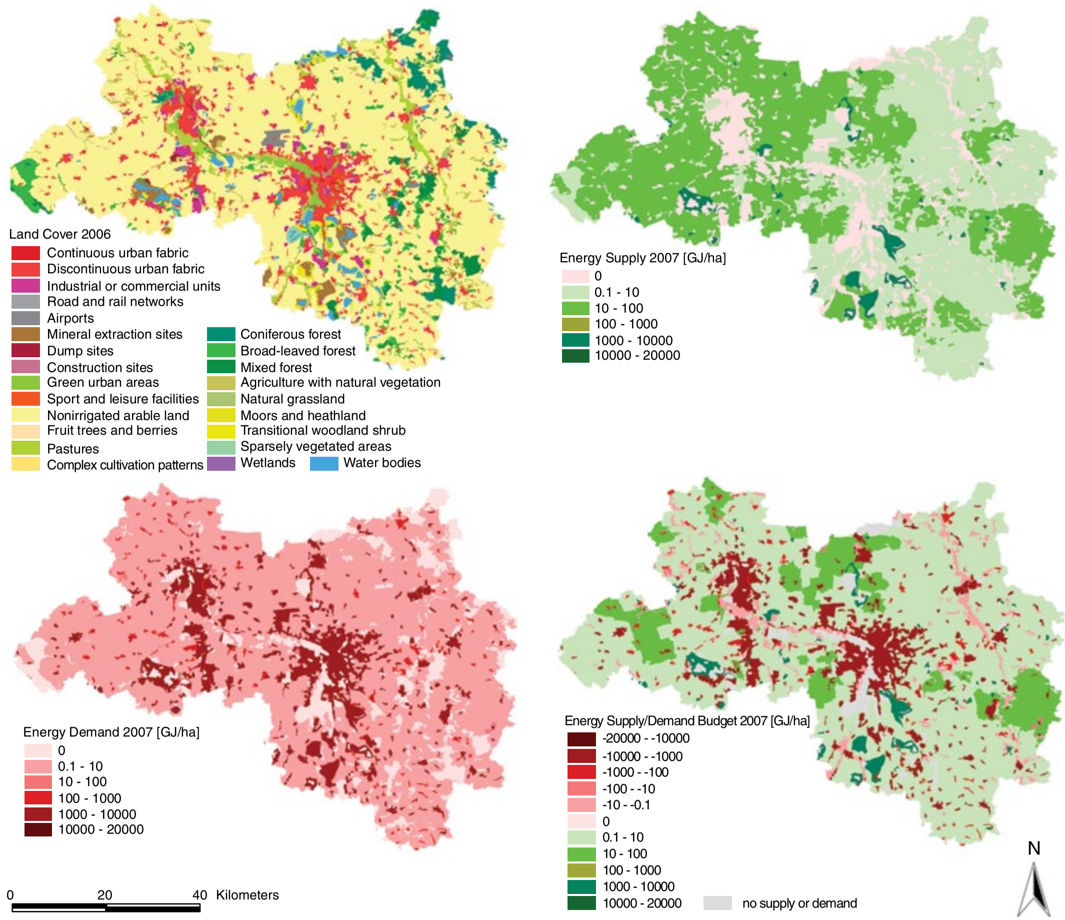


Fig. 4.4 CORINE land-cover map 1990 (*top left*); energy supply map (*top right*), energy demand map (*bottom left*) and energy budget map (*bottom right*) for the Leipzig-Halle region in the year 1990 (energy data in GJ ha^{-1} a^{-1}; after Burkhard et al. 2009, 2012)



■ **Fig. 4.5** CORINE land-cover map 2006 (*top left*); energy supply map (*top right*), energy demand map (*bottom left*) and energy budget map (*bottom right*) for the Leipzig-Halle region in the year 2007 (energy data in GJ ha^{-1} a^{-1}; after Burkhard et al. 2012) the 0–5 scale, the linkage of the assessment values with the spatial units (map compilation) and the interpretation of the results by the end user. A detailed discussion of the different sources of uncertainties can be found in Hou et al. (2013).

Further developmental steps are needed to tackle these problems. One key issue is the inclusion of additional ES in the quantitative classifications, as shown in the energy budget case study example. Direct measurements, official statistics, simulation models or specific surveys, for example in the class of cultural ES, are needed to fill these data gaps. Moroever, regional geological, geomorphological, pedological, climatic and geobotanical site conditions as well as additional human system inputs (e.g. fertiliser, energy, materials) strongly influence ES potentials and flows. These effects should be integrated in future assessments (besides land-cover and land-use intensity) in order to minimise the assessments' uncertainties. Thereby, more exact ES scores (0–5) can be provided for example to actors in participatory processes.

Nevertheless, there are limits of intersubjectivity in such an optimisation. Related to the high amount of data needed to derive the different ES matrices, it will probably not be possible to completely abdicate from expert opinions. This statement can of course be interpreted as a critical argument. But it can also be seen positively because expert-based approaches have the advantage of relatively rapidly delivering target-oriented results which immediately can be applicable in decision-making processes.

One major demand from environmental planning is to make predictions about potential future developments' effects. Therefore, one key step in the future improvement of the matrix approach is the coupling with computer models (► Sect. 4.4.3). This would enable assessments of scenarios and their spatial specifications regarding the supply and demand of ES. This would seriously increase the applicability of the ES concept in practice. Due to the enormous complexity of such efforts, only common, transdisciplinary and cross-regional efforts will lead to positive outcomes.

4.2 Approaches to the Economic Valuation of Natural Assets

B. Schweppe-Kraft, K. Grunewald

4.2.1 Principles of Economic Valuation

"It is not with money that things are really purchased. (John Stuart Mill 1848)"

Economic science is, briefly put, the art of the rational and economical use of scarce resources for the fulfillment of human values and needs. Since ecosystem services are limited and their use is often at least partially mutually exclusive (trade-offs), rules are needed to make rational choices between alternatives that affect ES more or less strongly. Here, economic science seeks to maximise the general welfare, taking into account intergenerational welfare, distribution and consensual ethical rules.

Ecosystem services become economic goods, or obtain economic value, by providing benefits, and by being scarce. Not only such goods as food, water and recreational opportunities provide benefits; so, too, do the nonmaterial assets that are part of human preference and thus relevant as benefits. The right of species to exist and the value we ascribe to that right are–besides other more direct benefits–of economic importance, as soon as they become a part of individual preference. Thus, the habitat function of an ecosystem for wild species may constitute a sociocultural ES in this sense.

Scarcity means that the provision or maintenance of an ES is associated with costs (Baumgärtner 2002). An example are the costs of measures to maintain 'healthy' landscapes that provide sufficient opportunities for recreation, fertile soils, fresh water, etc. (► details in Sect. 6.5). Almost 50 % of the biological diversity in Germany relies on traditional or nonintensive forms of land use that are usually not economically competitive on the world market. The resources for conserving such anthropogenic biotopes and habitats are scarce. Costs can arise even if no money is paid, for example from the limitation of agricultural and forestry use in protected areas. These so-called opportunity costs are, generally speaking, benefits which the society or the individual must do without, in favour of other goals or benefits.

Ecosystems continually provide people with services. They are similar in this respect to the human-made productive assets that are used to provide us with goods and commodities. Such assets are the basis of our welfare, unless they are consumed or destroyed. The same holds true for natural assets as well: 'We must live from the interest, and should not consume [natural capital]' (Hampicke and Wätzold 2009). Destroyed or degraded ecosystems are restorable, if at all, only after a long period of time. The costs of restoration generally exceed the cost of maintenance many times over. The genetic information lost by species extinction is irreversible. Nonetheless, the economic value of the depreciation of natural capital is not easy to determine.

Unlike buildings, industrial plants or machinery, natural capital usually provides us with a number of different benefits simultaneously, each of which has to be evaluated separately. These generally include so-called public goods, such as air quality regulation, recreation in the open countryside, etc. One of the characteristics of public goods is that they cannot be privately appropriated. Therefore, there are no functioning markets which could lead to an optimum level of supply based on individual supply and demand. Market prices can be interpreted as values in the sense of willingness-to-pay and as costs, expressing scarcity. All this is lacking in the absence of markets.

In addition, each single ecosystem is embedded in a tight network of ecological dependencies with other natural assets. In such a situation, the assessment of physical changes can already be a problem, long before we arrive at the point of valuation. Moreover, there are also creeping impacts which occur later, and when they occur, then sometimes in an erratic and irreversible way. Which means, that methods, like the discounting method, are required to compare current and future costs and the difficult problem of valuing nonmarginal changes has to be solved.

If economists valuate goods or services, they as a rule assign them instrumental value, based on their usefulness for achieving a defined objective. This means that both economic valuation and the ES concept approach the issue from an anthropocentric perspective (Hampicke 1991). In addition, economic valuation is based on 'methodological subjectivism' (Baumgärtner 2002). All valuations must (or at least should, see below) build on the preferences of each individual citizen.

Economic assessments are always focused on choices between alternatives. Ecosystem services, like any other goods and services assessed in an economic cost-benefit-analysis, are not evaluated in isolation, but always in terms of their relative advantage in comparison with other goods, which, due resource scarcity, must be dispensed with. The relative advantage of one asset compared with others is its economic valuation, which, for practical reasons, is not expressed in terms of specific goods (e.g. 'How many glasses of beer is something worth to me?'), but rather in terms of the maximum amount of income which one will forego, or the maximum willingness-to-pay/ minimum willingness-to-accept, of individuals. All methods of economic evaluation, including the market-based and cost-based methods, try in principle to value (real) income changes and willingness-to-pay more or less accurately, or at least to find plausible proxies for such valuations.

Economic valuation, must, in accordance with its own principles and methodological standards, always focus on specific alternatives, e.g. restoration or no restoration of an alluvial floodplain; maintaining a grassland or converting it into farmland; urban living conditions with or without an adjacent park, etc. Economic valuations of ES are often part of a so-called cost-benefit analysis, which attempts, as far as possible, to evaluate all the economic impacts of the implementation and of the nonimplementation of a project or programme, or of various project or programme alternatives. To this end, all relevant effects of the various alternatives must first be predicted. As regards public goods, such as recreation, urban living conditions or urban climate, this encompasses an assessment of the number of persons who will benefit or suffer disadvantages due to a change with respect to these goods. Moreover, all costs, savings, income increases and income declines must be determined, including all costs and benefits measured in income equivalents (willingness to pay or to accept) which will result from the changes in public goods.

Discounting Future Costs and Benefits

The future development of costs and benefits can vary significantly between different project alternatives. Dike-shifting involves high investment costs; the future benefits include flood damage avoidance, reduced nutrient concentration in the water and restored habitats. No dike-shifting means more financial scope for consumption today, but higher damage cost, higher spending on prevention of nutrient loads and less benefit from additional biodiversity in subsequent years.

In order to make differences in temporal cost-benefit distributions comparable, all future values are discounted to their present value and then summed up (the discounted cash-flow method, illustrated by the example of nature conservation; see Herrmann et al. 2012).

The discounting of future values is justified by the consideration that (a) investments help increase production; and (b) people are willing to forego consumption today to save and invest in order to ensure a higher level of supply in the future. The model of discounting is thus fundamentally based on the assumption of future growth. If the availability of goods and services is to increase in the future, it makes sense to rate the same quantity of goods higher in the present than in the future, when the quantity and quality of available goods and services will have risen, due to investment and growth. A no-growth perspective, however, does not per se mean that any calculation based on discounting would be obsolete. In such a case, additional sustainability criteria for each of the different periods could act as limits showing where discounting is still feasible and where it is not. Nevertheless, a generally accepted method for such a case does not exist yet.

The choice of the interest rate depends, among other factors, on the type of investment that constitutes the basis for comparison. Private investments in innovative goods can achieve a very high return on capital. The rate of return of saving deposits marks the lower limit of interest rates for private investments. A prerequisite for the operation of private markets are complementary products provided by the public sector, such as infrastructure, education, jurisdiction, social security, etc. If all these costs were attributed to private market activities, the real value of the return of investments could be reduced further.

The German Federal Environment Agency suggests using interest rates of between 3 and 1.5 % in cost-benefit analyses, the latter figure for cross-generational considerations of over 20 years (UBA 2007).

Some authors (Baumgärtner et al. 2013) propose working with different interest rates, arguing that environmental goods and ecosystem services should be discounted at lower interest rates than other goods. The underlying assumption is that the supply of environmental goods and ES will deteriorate, making them more valuable per unit, or that consumer demand for environmental goods will increase with growing incomes.

However, it should be noted that the tendency to support low interest rates for environmental and growth-critical reasons, can also have negative results for environmental and natural assets in the context of concrete decisions. In the abovementioned example of dike shifting, a low discount rate leads to high values for all future benefits, such as avoided flood damage, extended habitat areas, reduced maintenance costs, or additional opportunities for recreation. But a low discount rate also means that the time of taking action, e.g. making an investment in natural capital, becomes ever more irrelevant to the value of its outcomes. At a discount rate of 3 %, the net present value (NPV) of an infinite constant stream of benefits to begin immediately is 80 % higher than one that is to start in 20 years. At an interest rate of 1 %, the value of the stream of benefits beginning today would only be 20 % higher than one which were to start in 20 years. Hence, a low interest rate can also be taken as a reason for reluctance to initiate environmental projects.

Conclusion: It is the state of the art to use different discount rates and different costing/calculation periods, and to compare the different outcomes with a critical view of the underlying assumptions.

The final step in a cost-benefit analysis, as in any economic evaluation, is the aggregation of individual values to a total value. This is done by adding all positive and negative income effects (costs and benefits) including the observed income equivalents (willingness to pay). This means that, for example, the social value of the preservation of the recreational function of a landscape and of the habitat function of its ecosystems for flora and fauna is nothing but the sum of individual willingnesses to forego income in favour of the maintenance of these functions. The social value of a land development project, e.g. an industrial plant, would result from the net income growth caused by the new plant, minus the willingness to pay for the lost recreation and conservation functions, minus the agricultural land rent (which is usually included in the price paid for the land by the new owner), minus all other external costs not included in the price, such as increased flood damage or flood regulation costs caused by the additional water run-off due to imperviousness of the land surface.

The process of evaluation and aggregation is somewhat similar to an election (Osborne and Turner 2007), but with some differences:

- The individual can only vote in accordance with the scope of his own interests (How often does he really use a recreational area? What is the share of the income generated that accrues to him?).
- The strength of a vote can differ (a greater or lesser increase in individual incomes or of income equivalents measured by willingness-to-pay).
- The individual is not directly asked to vote; rather, his 'vote' is ascertained from the extent (positive or negative) of the net income effect accruing to him.
- The net income effect does not have to be investigated for each person individually, it is sufficient if the sum is known.
- Representative sampling methods are applied to determine the benefits of public goods (▶ Sect. 4.2.3).

Economic valuation methods differ from the 'one man, one vote' rule, inasmuch as every individual valuation of public goods is in fact tied to the amount of individual earnings, i.e. valuation results can depend on income distribution. Normally, it is not the purpose of a cost-benefit analysis to examine the fairness of distribution. In industrialised countries, this is no problem, for income distribution is as a rule irrelevant to the results of a cost-benefit analysis. Different weightings for individual willingness to pay in order to compensate for income disparities usually affect the overall results only slightly. This may be different if the effects of an international scope are assessed. Ignoring income inequalities on an international scale can easily result in ethically unacceptable valuation approaches.

The abovementioned principles of economic valuation:

- Are based on individual preferences
- Assess values as relative advantages, expressed in terms of changes in income or income equivalents (willingness-to-pay)
- Involve the formation of a social value by simple aggregation of individual values

They do not mean that economic valuation completely denies the notion of values that are not simply individual, but which rather have supra-individual worth, such as divine commandments, animal rights, or the notion of binding rules for a harmonious human-nature relationship. Cost-benefit analysis accepts such values, but treats them as individual ones, assuming that they are solely valid for the person that proclaims them. A person who assumes, for example, that animal rights should be ranked higher than the pursuit of any additional welfare gains, cannot demand that all economic advantages measured in a cost-benefit analysis be set to zero. He can, however, demand that his own individual foreseeable future income growth be assessed as his willingness-to-pay against e.g. any further species extinction.

Accordingly, individuals and their choices based on individual preferences tied to their economic limits (income) on the one hand constitute elementary declarative units. That means that the economic value is determined by the subjective evaluation of individuals ascertained by means of a survey of representative samples. In the strict sense, expert judgments can only be integrated into cost-benefit analyses if they can be interpreted as approximations to the preferences of individuals which cannot be measured directly. In this view, the economic value assigned to an ES is not a quality that is inherent to that object (e.g. an ecosystem), but rather a value which depends on the overall context, not only the economic context.

The valuation of the ES 'fresh drinking water', can, for example, depend on the following aspects (Baumgärtner 2002): How much clean water is there in total? How is the supply of clean drinking water distributed in space and time? How is the access to this resource regulated? What competing demands for water exist, besides its use in households? What kind of institutional restrictions exist? What kind of alternatives are there to water use in various use areas, and what would they cost? How much would it cost to import clean water from other regions? How much does technical water purification cost?

The failure of the market, private production and private consumption to generate socially-acceptable or optimal results–i.e. a market failure–is, according to economic doctrine, the occasion for an economic evaluation. This may be the case if:

- Production and consumption cause losses of benefits or price increases for others (so-called negative external effects). Examples: intensifying agriculture by removing hedgerows impairs the recreational capacity of a landscape; diking along a river can prevent flooding of areas behind the dike, but increases the flood risk upstream and downstream.
- Public goods are involved, i.e. those which benefit a large number of people without or with only limited possibilities of excluding anyone from those benefits. Example: recreational use of the open landscape, of public bathing waters, the existence value of species/biodiversity, or possible future pharmaceutical use of a certain kinds of species. In such cases, due to the lack of user payment, there are no incentives for market activities to maintain the provision, to prevent overexploitation, or to protect the asset from detrimental external effects.
- The costs of current activities accrue over the long term, e.g. to future generations, and therefore are not taken into account by present market participants. For example soil erosion, CO_2 emissions by intensive agricultural use of peat soils.

In the case of market failure, economic valuation has the function of informing about all costs and benefits accruing to people now and in the future, and enables decision-makers to reduce external costs and maintain provisioning with public goods to an optimal extent, thus maximising welfare under consideration of all relevant costs and benefits.

Like public surveys and public participation, cost-benefit analysis can help ascertain public opinion more precisely and make individual preferences more obvious than can be done by general elections only. In addition, it can reveal a malfunction of the democratic system, for example, the lopsided influence of powerful interest groups which are able to effect political decisions against the public interest (e.g. environmentally counter-productive subsidies; Brown et al. 1993).

Economic valuations need not necessarily be carried out with monetary units (Abeel 2010). Money can even be a hindrance. It can, for instance, promote the idea that only the world of market goods (production and consumption) really counts, whereas the actual goal is to correct the results of the market, by making it clear that the production of goods entails hidden costs that can obscure their true prices. Often, we are persuaded to produce things that we would rather do without for other, nontraded goods, e.g. for biodiversity and healthy ecosystems, if we knew enough about the issues, or if it became obvious that national income consists to a considerable degree of the costs of repair of damage to the environment and nature (Leipert 1989).

Money as a valuation unit may moreover suggest that the valuated goods will in fact be priced and thereafter traded. Nonetheless, the decision as to how to deal with market failure is up to policy makers, and is completely independent of the valuation process. Whether market failure is to be corrected by public supply, by do's and don'ts, by incentives, by taxes, duties or user fees or by the creation of markets, is a matter for public decision making. Economic valuation does not imply converting public goods into commodities to be traded on the market, either directly or indirectly.

Another misconception may be that the value of an ES that is calculated and determined for a specific social, economic or ecological environment could be transferred to other situations with no adaption, like the price of a good trade on the world market, for instance a smart phone. Such an understanding, however, would overlook the fact that many ecosystem services are tied to their point of origin, so that no distribution can take place. However, distribution in response to demand is a prerequisite for the emergence of a common price level on the market.

On the other hand, valuation in monetary terms can be highly practical. A monetary value allows a trade-off involving costs, income and various other goods, including public goods, based on the views of a representative sample of citizens. Other valuation methods, such as benefits analysis (Zangemeister 1971; Hanke et al. 1981) and similar types of so-called multi-criteria analysis (Zimmermann and Gutsche 1991), also use decision-making models based on trade-offs (▶ Sect 4.1.). However, such models often depend on the opinions of a limited selection of experts and/or 'citizen experts' (Dienel 2002), which are not representatives. Although in certain cases, expert-based models may have a high problem-solving competence, the social values upon which they often implicitly build have not been validated.

Various decision-support instruments, such as cost-benefit-analyses, expert-based multi-criteria analyses or discursive processes of active citizenship, should be used in accordance with their respective strengths and weaknesses. A representative group of citizens mixed with some experts could for instance provide useful advice for the best use of a fixed local budget for various urban green-space management measures; however, when it comes to the preparation of a concept for reducing soil erosion in a district (Grunewald and Naumann 2012), an expert-based cost-effectiveness analysis would likely be better grounds for sound decision-making. The cost-benefit analysis, after all, shows its strengths when actions are to be taken that might affect a great number of people physically and financially in very different ways. This is the case, for instance, when decision support is needed on the question as to how much money a city should spend overall on green-space management. Another example would be the design of a well-balanced programme of measures for reducing soil erosion that should also take into account other effects, e.g. upon species preservation, the landscape, or water pollution, in such a way that the costs of the measures will best be outweighed by their benefits.

Example

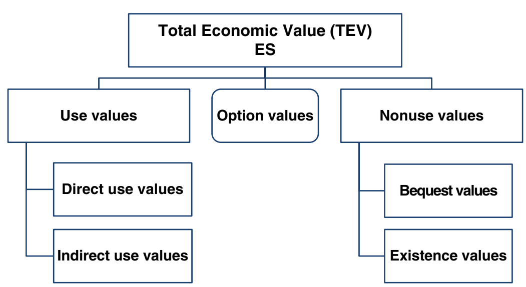
Grossmann et al. (2010) applied a cost-benefit analysis on proposals for a bundle of nature-based flood prevention measures by increasing the retention area through dyke-shiftings (▶ Sect. 6.6.3). They calculated the avoidance of flood damage, valuated the water purification effect of an enlarged alluvial floodplain by comparing it with the cost of alternative measures for reducing water pollution, and asked people about their willingness-to-pay for the benefit of the enhancement of conservation and recreation. The value of the ES thus assessed was three times as high as the cost of the measures.

4.2.2 The Total Economic Value

The most widely accepted approach for the economic valuation of ES is the concept of Total Economic Value (TEV, Pearce and Turner 1990) (◘ Fig. 4.6). The various benefits of ecosystems are classified as either *use values* or *nonuse values*. Use values are further subdivided into *direct* and *indirect use values* and *option values*. Nonuse values are broken down into *existence values* and *bequest values*.

■ Direct Use Values

Direct use values accrue from the direct use of ES for consumption and production, e.g. food, firewood, medicine, timber, drinking water, cooling water, etc. The use of a landscape for recreation, leisure activities, tourism or scientific or educational purposes is also considered a direct use of ES (Baumgärtner 2002). Direct use can be consumptive–example: firewood–or nonconsumptive, as with recreation. Direct use values are linked to provisioning services and goods, as well as with some sociocultural ES, such as for recreation, cultural identity, landscape aesthetics and knowledge services.



◘ **Fig. 4.6** The concept of total economic value (TEV). (Adapted from Pearce and Turner 1990; Bräuer 2002)

■ **Indirect Use Values**

Indirect use values arise when ecosystem services interact directly or indirectly with human activities. Examples are flood control by means of water-retention measures in alluvial floodplains, the self-purification effect of water bodies, or the water-filtration capacity of soils. The so-called regulatory services generally fall into this category. The economic value of these services is measured as the change in the costs and benefits of the use that is affected by them, e.g. reduction of flood damage, benefits from additional use as a swimming location, or the decreased costs of the drinking water supply; see, by analogy, the concept of *final ecosystem services* by Boyd and Banzhaf (2007) (► Sect. 3.2).

■ **Option Values**

Option values express the fact that there is a willingness to preserve the possibility of later use of ES, regardless of whether this will really take place or not. Option values and values to be realised in the future correspond largely to the so-called *Potentialansatz* (capacity approach) in German landscape planning (► Chap. 2 and ► Sect. 3.1). The option value can also be interpreted as an insurance premium that people are willing to pay to maintain the possibility of future use (Weitzman 2000). Option values are especially significant in the context of landscapes and ecosystems of high cultural significance and singularity, such as the Brocken peak in the Harz Mountains in Germany, or with respect to the uncertainty of a future economic use of species and their genomes (e.g. Norton 1988).

■ **Bequest Values**

The bequest value expresses the willingness of people to forego parts of their present income in order to preserve things for future generations. This heritage can refer to sociocultural ES, but also to provisioning services.

■ **Existence Values**

Existence value reflects the willingness-to-pay for the preservation of things regardless of whether there is any likelihood of their future use or not, just in order to preserve their existence. Such values are often ascribed to assets thought to have an intrinsic value, such as living species, e.g. in the concept of animal rights.

These different kinds of values, named above, are conclusive. Their sum is the overall economic value of an ecosystem. However, in field studies, it is often impossible to clearly separate the different values from one another.

Investigations at Natura 2000 sites have revealed that more than 50 % of their TEV were constituted by indirect use values and nonuse values (Jacobs 2004). That means that from a conservationist point of view, these values, especially the option, bequest and existence values, are the most critical ones. On the other hand, the problems of reliable evaluation increase as one moves from direct use values to nonuse values.

4.2.3 Valuation Methods and Techniques[2]

Use Values

■ **Market Prices**

If assets provided directly by nature can also be found on markets in the same or a similar quality–e.g. mushrooms, fish, game–the market price can be used as a proxy for their value (the *market-price method*). One important precondition for the applicability of this method is that product qualities and the demand for marketed and non-marketed products are similar. This is not always the case, however. For example, experience shows that blueberries which are picked in the woods on a hike taste particularly good, this special kind of appropriation seems to give them an extraordinary quality, so that they could be rated considerably higher than purchased blueberries. On the other hand, the picking is an activity that is incidentally performed, without significant additional effort. One might also pick the berries when demand is low, and therefore have to valuate them at a price well below their market price. The same is true of self-caught fish. As an actively appropriated product, it might have a higher value than comparable market products, but it could also serve as an incidental by-product of the fishing activity itself, which is the

2 For a systematic presentation of economic valuation methods that is also addressed to noneconomists see ► www.ecosystemvaluation.org.

■ **Fig. 4.7** The economic value of the provisioning service of a field (here cornfield near Sulingen in Lower Saxony) can be measured on the basis of the income loss resulting from abandoned agricultural use. © Burkhard Schweppe-Kraft

actual ES provided–recreational activity. If the fish is used by the family of the angler, their possibly differing preference for fish may also be important for the valuation.

The market-price method could, for example, be suitable for the valuation of the effect of an alteration in forest management on all the wild fruits to be found there, or it could be appropriate for the valuation of the improved water quality in a lake on the composition of its fish population (less biomass, but a higher proportion of game fish). In both cases, the changes on the supply side are only one side of the coin, for the extent to which the additional supply will really be used must also be assessed. Finally, the question should be answered, e.g. on the basis of surveys, to what extent the value of the products is thought to lie above or below the market price level.

- **Change of Value Added, Profits, Return on Sales Minus Cost of Production**

The majority of market goods created with the help of ecosystem services, such as drinking water, wood products, food, etc., is produced in combination with labour and capital. If the ES change, e.g. additional land used for agricultural production, causes increased sales of goods, the additional value of sales is not the only determining factor for their valuation; rather, it is the difference between the additional sales and the costs of the use of capital, precursor products, production facilities and labour power, including a normal remuneration of the labour input of the entrepreneur. The difference remaining after this calculation corresponds in the case of e.g. cropland more or less to the cost for the lease of the land being assessed, or a comparable plot. Therefore, the ground rent (lease) is often used as a proxy for the net value of the productive input of ecosystem services that are combined with certain plots of land (Hampicke et al. 1991).

Example

What loss in the value of agricultural production would result from the abandonment of this field (■ Fig. 4.7)? From the total loss of market revenue, one must first subtract the variable costs. In addition, adjustments with respect to labour and capital inputs will occur in the mid- or long-term and have to be considered in the evaluation. After these adjustments, the loss of ground rent (lease) remains as a permanent loss. This is determined on the basis of various favourable and unfavourable factors, such as soil fertility, water supply, climate, slope, etc. When evaluating large-scale soil loss in developing countries, one would have to assume significantly higher income losses, due to a lack of alternative employment opportunities. Nothing in the world would suffice to persuade us to do without the entirety of the agricultural land on earth–its loss would have a value of 'minus infinity' (Costanza et al. 1998).

If a corn (maize) field is converted into a species-rich damp meadow, for example due to conservation measures, a comparison of these two different provisioning services–corn and hay, respectively–would require a calculation of the difference between the proceeds from the sales of these two products, and of the above-described production costs. For the corn, this difference would be positive; for the hay, probably neutral or even negative.

For a comparison of the total economic value (TEV) of intensive–e.g. corn–and extensive farming systems–e.g. a meadow–a correct valuation of the services corn and hay could be critical. The difference between the profits is often significantly less than the difference between the sales proceeds, one reason being that intensive farming systems often require higher inputs. The different valuation of provisioning services, in one case on the basis of sales proceeds, in the other on the basis of sales proceeds minus costs, explains why in the study by Ryffel and Grêt-Regamey (2010), the calculated total value of species-rich grassland is less than that of intensively used grassland, while in the study by Matzdorf et al. (2010), the species-rich grassland comparatively outperforms the farmland (► Sect. 6.2.4).

An assessment of provisioning services on the basis of sales proceeds would mean that not only ES would be evaluated, but the value added by labour and capital, too, would be included. A correct application of the cost-benefit analysis must always subtract the costs necessary for production from the value created, to calculate the net yield. In the case of provisioning services, this means the respective earnings minus the wages for the work of the contractor plus the rent paid for the land (see environmental services ► Sect. 2.1).

Implicitly, the above calculation of provisioning services involving profits or rents is based on the assumption that the labour thus 'freed' and–at least in the medium to long term, even the capital thus 'freed'–will find uses elsewhere, and will there generate added value that corresponds to the costs. Cost-benefit analyses carried out in industrialised countries are, due to the flexibility of the markets for labour and capital, generally based on this simplifying assumption. Deviations should be clearly identified and explained. In many regions in developing countries, however, the necessary alternative opportunities are not available, particularly for the factor labour. If the destruction of the services of an ecosystem, e.g. the loss of soil fertility, or overfishing, drives the people who had depended on these services into long-term unemployed, the cost-benefit analysis would have to include as the value of the supply service concerned not only the lost profits, but the entire value, including labour and possibly capital costs. In industrialised countries like Germany, adjustment problems and deadlines are more likely to be the factors to be taken into account with respect to the factor capital.

Therefore, when determining the cost of a change in agricultural production or the abandonment of agricultural use the calculations for the short or medium term are often based on contribution margins. A contribution margin is the market revenue minus the variable costs. As the term implies, the contribution margin per hectare states the contribution that the production on one hectare of land makes to cover the fixed costs of a business, for example, to the interest payments due on the loan for stables (see case study in ► Sect. 6.2.3). A contribution-margin calculation assumes that unused capital is inflexible, i.e. it cannot be used elsewhere just as profitably. In the short term, such a method of calculation is justified; in the medium term however, adaptation possibilities have to be assumed. After the technical depreciation period of the capital involved–at the latest–it is advisable to shift to such values as lease or long-term profit outlook for

■ **Fig. 4.8** Fruit growing areas are particularly dependent on pollination services. © Burkhard Schweppe-Kraft

the calculation of production losses. The correct handling of the costs of capital can be crucial for the actual calculated results. For example, in a case of the rewetting and use abandonment of previously farmed peat soils, Röder and Grützmacher (2012) calculated costs of € 40/t of saved CO_2 emissions, on the basis of contribution margins. If only the lease costs of, say, € 250/ha were used in the calculation, a much more favourable value of around € 9/t of CO_2 would result. Assuming a 20-year adjustment period with adaptation rates at a consistent level and a calculated interest rate of 3 %, costs of over € 17/t would result. Calculation examples from studies based on all three types of calculations can be found in the literature. This shows that major methodological differences occur not only in the evaluation of ES generally, but also that great tension is possible simply with the very conventional cost calculations, which are based on different, and often highly questionable, assumptions.

The example of using land-lease as an approximation for the long-term value of the agricultural production function of an ecosystem (provisioning services) again shows dramatically that economic valuations generally apply only to relatively small changes: The higher total value of grassland compared to farmland, which can be calculated on the basis of the study by Matzdorf et al. (2010) (► Sect. 6.2.4), applies only to the case of the current distribution between grassland and farmland. If, due to the currently high total economic value (TEV) of grassland, ever more farmland were to be transformed into meadowland, the supply of the various public and private goods produced using these land areas would gradually increase so greatly that the prices and the willingness-to-pay for any additional margins of these goods would fall. The total economic value per unit of converted farmland could pull even with the TEV per additional unit of grassland, and then even exceed it. This could in fact be accomplished relatively quickly, for example in the case of the species-protection function/service. For the preservation of biodiversity often optimally requires a mix of grassland and farmland, and not a grassland monoculture.

This also shows why the value of the sum of all ES cannot be calculated from the value to be set for a relatively small change to be assessed. Multiplying the total stock of farmland in the industrialised countries by the respective lease values per hectare, the result is by no means the value that society would be willing to pay for the preservation of the agricultural production output of these areas; the true figures would be significantly higher. With the increasing loss of production areas, prices would rise to an extreme degree, and the social upheaval thus provoked would have uncontrollable consequences.

Example

Within the EU, the service pollination is estimated at a value of some € 14 billion (Gallai et al. 2009). This is the value of agricultural products which are highly dependent on insect pollination. This knowledge does not help much for concrete valuations. In assessing the changes in pollinator populations in specific growing regions, the decisive factor is whether the populations there already constitute a limiting factor for production, or whether they are extant in abundance. So far for example, we know relatively little about how flower strips within fruit-growing areas impact on the net yields (■ Fig. 4.8).

■ Change in Production Costs

The cost of production method also ascertains the change in the difference between the sales proceeds and costs of production, but it does so for the special case that product quantities and revenues remain constant, and that only the costs of production change. The typical example of this case is the reduced effort required to provide clean drinking water if a farm field, which generates pollution is replaced by grassland. Another example would be an increased use of fertilisers to compensate for reduced soil fertility, which has resulted, for example, from intensive use, or soil erosion caused by the removal of hedgerows and other small structures.

In these cases, the production cost method was used directly to valuate the supply capacity of ecosystems (water supply, agricultural production), and also indirectly to assess the impact of regulatory services (reduction of soil pollution, and of soil erosion by small structures) upon the respective provisioning service.

■ Damage Costs, Mitigation Costs, Adjustment, Repair, Replacement Costs

Many regulating services influence the effects of natural hazards (flooding, avalanches and mudflows, storm damage, etc.) and anthropogenically induced risks (climate change, air pollution, urban climate stress). For the evaluation, the damage and damage prevention costs and the adaptation, repair, replacement or avoidance cost can often be used. Here, the extent to which damages (including medical expenses), or the cost of prevention and repair (rehabilitation) can be changed by ecosystems and ES is examined. Examples include the prevention of flood damage through restoration of floodplains, or avoidance costs for the treatment of respiratory diseases caused by the dust-filtration effect of urban green spaces.

It is a general economic principle that a goal should be achieved at minimum cost. If a damaged item is of lower value than the cost of its repair, it is more beneficial to all concerned to monetarily compensate the aggrieved person than have the damage repaired. This principle applies not only to the compensation for damage to passenger cars, but also to evaluation in the determination of total economic value (TEV). The same applies if the damage-avoidance costs are higher than the damage. Here, too, it is cheaper to pay the lower insurance compensation for a damaged asset than the higher cost of completely avoiding the potential cause of damage. Such situations are referred to as the *least-cost principle*.

Often, only a portion of the value of an ecosystem services can be quantified by damage or repair costs, just as medical costs often reflect only the cost of treatment, but not the physical or mental suffering of the patient. If, due to increased use intensification in an area, there are no more skylarks or partridges there, the cost of resettlement or avoidance of that loss may be significantly less than its ethical and aesthetic significance. Other methods, such as willingness-to-pay analyses, should be used if damage or avoidance costs can measure only part of the total economic value of a service.

Example

During the mid-1990s, Pimentel et al. (1995) assessed the on-site and off-site costs of erosion in the USA, and arrived at a figure of about \$100/ha/yr. If this order of magnitude of replacement and damage costs is compared with the cost of erosion-mitigation measures, a very positive cost-benefit ratio of 1:5 results; the soil erosion hazards due to water and wind are thus reduced from 17 $t/ha^{-1}\,a^{-1}$ to 1 $t/ha^{-1}\,a^{-1}$. Using an analogous approach for a loess-covered, predominately agricultural area in Saxony, Grunewald and Naumann (2012) ascertained a cost-benefit ratio of approximately 1:2 (► Sect. 6.6.2).

■ Alternative Costs

Closely connected with the above methods is the so-called alternative-cost approach. This method often valuates not the costs in fact incurred, but rather those of theoretically possible options which might be used in order to achieve a goal in an alternative manner. An example might be the evaluation of the additional self-cleaning capacity of a renaturated water body, using the two potential alternatives of, on the one hand, the measures necessary to reduce pollutant input from agriculture, and on the other, the building of additional wastewater treatment capacity to achieve the same water-quality effect. The erosion protection provided by hedgerows and small structures could, for example, be valuated not only via the production-cost method, as above, but also on the basis of the cost of soil conservation measures on the field which are equally effective.

Whether or not a corresponding alternative-cost approach is permissible depends on whether the social goals are formulated in a sufficiently binding manner or not. Strictly speaking, the alternative-cost approach only leads to correct results if the objectives are formulated in such a binding manner that the necessary measures for their alternate achievement will actually be implemented in the not-too-distant future. An example of such a binding social goal is the EU Water Framework Directive (WFD), which mandates the attainment of a certain level of water quality (▶ Sects. 3.3.2 and 6.6.2). If farmland is converted to grassland, the nutrient input into the groundwater and the surface waters is reduced, and the specified goals of the WFD become more attainable. A corresponding contribution to the reduced water pollution can be achieved by various measures in farming, or by improvements in the treatment stages. Under the least-cost principle, an alternative measure, which allows both similar relief at the lowest cost and at the same time has a realistic chance of implementation should be selected as the value of reduction of nutrient immissions due to conversion into grassland. Matzdorf et al. (2010) used a value of between € 40/ha and € 120/ha for the valuation of the reduced nutrient inputs through the preservation of grassland, based on the evaluation of data of cost-effective measures to reduce nitrogen emissions by Osterburg et al. (2007) (▶ Sect. 6.2.4).

Measures for rewetting and restoring formerly farmed peat soils halt the mineralisation of organic soil components, and thus lead to a significant reduction of greenhouse-gas emissions. The evaluation of this regulatory service 'rewetted peat soils' is possible both on the basis of damage costs and on the basis of alternative cost. In accordance with the Stern Report, the methodological convention of the German Federal Environment Agency (UBA 2007) suggests a preliminary cost estimate of approximately € 70/t of CO_2, based on a combined damage-/mitigation-cost analysis. In case of the use of wind power, 1 t of avoided CO_2 emissions costs approximately € 40; on the European carbon market, a ton of CO_2 cost € 6–7 in early April 2012. Which of the above values is to be used for the valuation of the CO_2 emissions saved by rewetting will depend on how future developments are to be assessed (▶ Sect. 6.6.4).

It can be assumed that the required reduction of CO_2 emissions cannot be implemented solely using the current favourable measures that enable the current low prices on the carbon market. Achieving the goal at these costs is thus unrealistic. Measures in the cost category of CO_2 avoidance through wind power would seem, for example, to be more realistic. If we assume, moreover, that the goal of limiting the temperature increase to 2°C will fail to be attained by a wide margin, which seems increasingly likely, even the € 70 damage costs would have to be considered too low. The example shows that even with realistic assumptions, there can be very widely divergent evaluation approaches. Evaluations should therefore always disclose the assumptions upon which they are based, and whenever possible, alternative calculations under different assumptions should be undertaken.

Example

At the beginning of the 1990s, the city of New York was forced to take action, since it no longer met the established drinking-water quality standards. A water filtration and treatment plant was to be built for $ 6–8 million, and operating costs of about $ 300 million per year would have been added. As an alternative, the issue of improving the ecological functions of ecosystems in the Catskill Mountains, the drinking-water catchment area for the city, was examined. This cost was estimated at a one-time investment of € 1–1.5 billion. Faced with a balancing of interests between the cost of improving the ecosystems on the one hand and the development of purification technology as a substitute for the reduced ES of degraded ecosystems on the other, the decision was made in favour of the ES option (Chichilnisky and Heal 1998).

■ Real Estate Prices–Hedonic Pricing

The evaluation approaches presented above have, under the MEA (2005a) system and the ES classification (▶ Sect. 3.2), respectively, been oriented primarily towards provisioning and regulating services. The *hedonic pricing method* is oriented towards the sociocultural services recreation and aesthetics, or beyond that and in more general terms, towards the subjectively evaluated welfare functions of green elements and green spaces in the residential environment.

Under the hedonic pricing method, the goal is to ascertain the effect of near-residential green spaces on real-estate prices by statistical analysis. Hoffmann and Gruehn (2010) come to the conclusion that in densely populated inner-city districts, the green features of the residential environment accounts for 36 % of the property value. In less densely populated, smaller towns, the effect is less (► Sect. 6.4).

The hedonic pricing method covers only that portion of the use of urban green spaces that accrues indirectly to the property owners. Any benefits above this portion would have to be ascertained by other methods, by carrying out an additional willingness-to-pay analysis, or on the basis of the statistical data estimates of a demand function, similarly to a travel-cost analysis.

The Travel-Cost Approach

The term *travel-cost analysis* covers a whole package of different methodological options, which are primarily used for the evaluation of recreation areas. Here, the relationships between the number of trips to a region or a certain type of area and the amount of the cost per trip are analysed statistically. In the newer versions of the method–also the quality of the area for recreation (e.g. landscape, landscape diversity, facilities with recreational infrastrucure) are taken into account. On this basis, a demand function for recreation in the area or area type in question is assessed. Based on a comparison of the behaviour of visitors with high- and low-access costs, respectively, it is possible to deduce that the willingness-to-pay for the first visit undertaken within a given monitoring period to a particular area or type of area is higher than for later visits. Visitors with low access costs do not need to exercise this higher willingness-to-pay for the first visit in real terms, and thus realise a so-called consumer surplus. The sum of all consumer surpluses yields the total net benefits of recreation in the assessed areas. The consumer surplus constitutes the willingness-to-pay that an individual has for a recreational activity, minus its actual cost.

In some proposed methods and evaluation studies (Ewers and Schulz 1982; UBA 2007; to some extent too, Getzner et al. 2011), the actual costs of a recreational activity are regarded as its benefits. Certainly, assuming rational behaviour, the benefits must generally be at least as high as the cost paid for them; however, as discussed above in connection with the costs for the production of agricultural products, the purpose of a cost-benefit analysis is to ascertain the difference, or the ratio of costs to benefits, for each alternative. With such a difference ascertainment, the result of a recreational activity the benefits of which are just as high as the costs, would always be neutral; the net benefit, i.e. the difference between benefits and costs, would always be zero. This result would emerge in all studied alternatives, regardless of whether the recreation areas were of average quality, are actually upgraded, or would be devalued by impacts. For it we dispense with the counterbalancing of the costs, and show the cost only in their indicator function for the minimum benefit, we will arrive at completely nonsensical evaluation results when comparing options. For example, if the construction of a bypass road were to lead to an increase in the expense of money or travel-time to be paid by the inhabitants for access to their recreation areas, this would not be recorded as an obstacle to their recreation, but rather as an increase in their recreational benefits. Hence, the simple calculation of cost is unsuitable for the evaluation of recreational benefits. The goal must be to calculate the consumer surplus, the difference between the benefits (or willingness-to-pay) and the costs.

Under the travel-cost method, which uses this approach, willingness-to-pay is derived from the observed actual behaviour of a large number of different recreation-seekers, using statistical methods. This, like the land-price method, is one of the so-called revealed-preference methods, based on an investigation of factually evident preferences, in contrast to the *stated-preference methods*, in which the preferences are directly queried.

Example

In the Eibenstock-Carlsfeld region in the western Ore Mountains of Saxony, a survey was carried out via interviews among visitors and tourist-service providers on their appreciation of the landscape scenery (Grunewald et al. 2012). The questions concerned the qualitative landscape characteristics and preferences, travel expenses and willingness-to-pay for the maintenance and appearance of the landscape. For this purpose, the monetisation approaches of the travel-cost and willingness-to-pay methods were used. The study comprised face-to-face interviews with 95 summer and 105 winter tourists; travel costs were recorded for a total of 584 individuals. The goal was the analysis and monetary valuation of sociocultural ecosystem services related to landscape aesthetics, in order to provide a foundation for the improved landscape planning and management.

The tourists' aesthetic perception of the landscape elements in the region is influenced primarily by visible, near-natural landscape elements, such as the forest and water bodies, and by their harmonic composition. An undisturbed landscape was the principal reason for travelling to the region and spending vacations there. Altogether, tourists paid about € 5.5 million per year in travel costs (extrapolated to the total number of tourists visiting the region), they are willing to pay € 170,000 per year in addition for the protection and management of ecosystems. The results show that the visitors valued public goods and services highly, a factor which will have to be considered more strongly in future planning (Grunewald et al. 2012).

- **Hunting Leases, Fishing Licences, etc.**

For some recreational activities, such as hunting or fishing, there are prices to be paid in the form of fishing licences and hunting leases. These, unlike such expenses as those for fishing equipment or the fuel used to reach a fishing spot, are an expense associated with no real costs, or only minimal ones. A payment that is not remuneration for any labour or capital cost is referred to as a 'surplus.' Even the rent for agricultural land is such a 'surplus.' By paying for a hunting lease or fishing licence, the sportsman shows that his benefit from the fishing or hunting activity is at least equal in value to that payment. In this case, as with the land-price method, this share of the benefits accrues not to him, the user, but rather to the owners of the land leased. The benefits that can be calculated from fishing or hunting leases is the lower limit of the actual benefits from that activity.

If we also wish to ascertain the net benefits to the anglers and hunters over and above this minimum, it would be necessary to apply other methods, such as the travel-cost approach or contingent valuation. It is important in cases of changes in the conditions for recreational use, to always also ascertain the possibilities of substitution. Generally, there are also other places where recreational activities may be carried out. In such cases, the increase in travel costs to remaining alternative fishing or hunting areas would be a first rough measure for the welfare loss caused by the degradation or the loss of another area. With a more precise travel-cost analysis, it would be possible to capture also the 'consumer surplus' over and above simple cost effects.

- **Admission Prices**

A method for calculating leisure and recreational use which was in the past particularly common is the *admission-price method*. Here, the recreational opportunity to be valuated–from city parks to national parks–is compared with similar recreational activities for which a price of admission is charged. One problem with this method is that people who spend time in fee-based recreational facilities, such as former horticultural exhibitions or amusement parks, may have different preferences from those of people who use free leisure facilities, such as urban forests or natural parks, so that it is difficult to find truly comparable situations. For example, admission-charging swimming pools and guarded beaches often have a distinctly different character than free swimming spots. Moreover, the price of admission reflects the lowest level of willingness-to-pay among those who avail themselves of the service; some visitors would be willing to pay a higher ticket price. Because of these problems, a valuation based on admission prices should also be supplemented by some other alternative valuation method, such as travel-cost or willingness-to-pay analysis.

- **The Willingness-to-Pay Analysis (Contingent Valuation), Choice Analysis**

In addition to, or as an alternative to the above methods, any direct or indirect use value can theoretically be assessed on the basis of direct interviews using contingent valuation or the choice analysis. These valuation techniques are used for the ascertainment of both use and nonuse values (see below). Applied to the same evaluation object, travel-cost and willingness-to-pay analyses often provide relatively similar results (Löwenstein

1994; Luttmann and Schroeder 1995; Whitehead et al. 1995). In cases where specialised knowledge is required for an evaluation, e.g. for the evaluation of changes in soil fertility, erosion, effects on water quality, flood damage, etc., complementary expert-based methods should also be used, in addition to the willingness-to-pay analysis, in which, since it is a representative approach, largely nonexperts are interviewed.

Methods for the Detection of Nonuse Values

▪ Contingent Valuation, Choice Analysis

Preferences for nonuse values, such as the desire to preserve species and habitats as a 'value in and of itself' (existence value), or so that they can be used and experienced by future generations (bequest value), can, like option values, currently only be ascertained by direct, representative surveys. The main methods for this are the willingness-to-pay analysis and the choice analysis.

The willingness-to-pay analysis asks how much money or income an individual would be willing to do without, as a maximum, in the form of a generally mandatory landscape-maintenance tax, so that nature might be preserved, or a specific conservation programme might be implemented. In a choice analysis, the respondents are presented with different options about the future, which they can, by means of various procedures, either accept or reject. Each option here describes various conditions related to the natural environment, and an income-relevant quantum, such as a surcharge or deduction for income tax purposes. By means of statistical analysis, willingness-to-pay with respect to the various parameters can be derived from the various 'decisions' thus made.

There is an extensive body of scientific literature on the validity of stated preference methods and the possibilities for improving and securing their validity (e.g. Hoevenagel 1994; Marggraf et al. 2005).

A number of results regarding willingness-to-pay for conservation measures in Germany are now available (◘ Fig. 4.9; ► Sect. 6.6.1). They involve extensive activities, such as national programs for the conservation of biodiversity (an average of € 231 per household per year) down to such local activities as measures for the conservation of the dusky large blue butterfly on 64 ha in Landau, the Palatinate (€ 22 per household per year). The fact is that today, every household pays an average of around € 16–20 per year for conservation via public expenditures for nature conservation that are based on their tax payments.

Some authors argue that concrete locally visible measures should be queried as much as possible, this provides a more realistic assessment of willingness-to-pay (Fischer and Menzel 2005). On the other hand, results regarding smaller, more specific measures always leave the question unanswered as to how the group of those questions regarding willingness-to-pay is to be defined: only at the municipality level, or that of the district, of the entire state, or nationwide? When questioned at the local level, one has to deal with the effect that measures in sparsely populated areas tend to always obtain a lower value than measures in densely populated areas, because of the smaller population, and hence the smaller potential willingness-to-pay group. For the valuation of nature as an 'intrinsic value,' this would be a substantively unacceptable result. Moreover, it has been demonstrated that the evaluation of specific measures always includes the implicit distributional assumptions of the respondents ('If I pay for Measure A, I assume that others will pay for Measure B'; Degenhardt and Gronemann 1998). As an evaluation of ◘ Fig. 4.9 shows, a lower willingness-to-pay does tend to be expressed for special measures than for comprehensive measures; however, at the local and regional levels, the willingness-to-pay per measures unit is considerably higher. In the case of the preservation of the dusky large blue butterfly (*Glaucopsyche nausithous*) in Landau, the conversion of the willingness-to-pay results of the population to a per-ha of measure-implementation value yields € 6656/ha/yr. However, in a nationwide programme examined by Meyerhoff et al. (2012), values of only € 1000/ha for the specific grassland-part of the programme, exclusively, were obtained and 300 €/ha if the whole programme was valued.

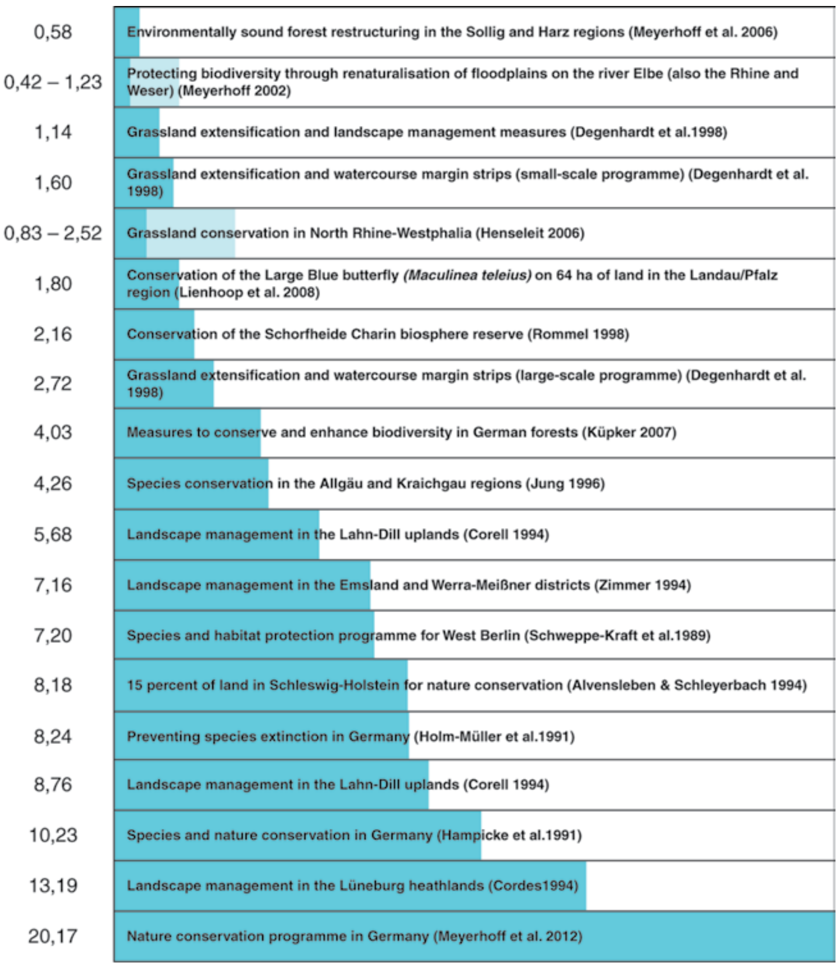


Fig. 4.9 Willingness-to-pay for conservation programmes encompassing various spatial and substantive factors (in €/mo.). When comparing the data, one matter to consider is that no adjustment was made for inflation. (Adapted and supplemented from BfN 2012 (references other than Meyerhoff et al. 2012 see there))

Actual per ha costs of conservation measures are usually below these figures.

For concrete decisions on conservation projects or interventions at the state or federal levels, the effect due to different population densities, regional preferences or implicit distributional assumptions are not particularly helpful. Such decisions should therefore be based on willingness-to-pay analyses, with which comprehensive programmes have been evaluated. Special willingness-to-pay for individual measures within these programmes could then be roughly evaluated on a pro rata basis, for instance per area segment, or, more accurately, through more detailed expert-based scoring methods (Schweppe-Kraft 1998).

Restoration-Cost Method

A nonpreference-based method for the assessment of existence values is the restoration-cost method. It is especially applied for the evaluation of the functions or services of habitats for the preservation of biodiversity. Under this method, the costs which would accrue if one were to first destroy a habitat and then restore it, are ascertained.

If restoration is required by law, this method is only used to ascertain what a measure, such as the construction of a road, would additionally cost in the form of mandatory compensatory measures. If restoration is not required, it ascertains the costs which would be incurred if society were to recognise in the future that restoration were necessary or desirable. Under economic theory, this approach is acceptable, since international conventions and policy statements such as the European Biodiversity Strategy have made a commitment to a 'no-net-loss' strategy with respect to the conservation of biological diversity. This means that we can–hopefully–assume with a relatively high degree of probability that such a restoration will in fact occur in the future.

A particular challenge in restoration-cost methodology is the monetisation of interim losses of function. Unlike technical infrastructure, the restoration of the biodiversity of ecosystems is not completed with the conclusion of the restoration of physical initial conditions (e.g. termination of intensive use, rewetting), but rather well, beyond that, require a number of years or even centuries. A number of different methods exist for evaluating the interim loss of function (Schweppe-Kraft 1998; Dietrich et al. 2014). In the USA, a discounting procedure within the framework of the so-called habitat equivalency analysis has been widely used since about 1995 for the quantification of damages. Previously, this method had already also been proposed for use in Germany for the assessment of tree damage and damage to habitats (Buchwald 1988; Schweppe-Kraft 1996; ► Sect. 6.6.1). The restoration-cost approach is also used in the German impact-regulation system (Köppel et al. 2004).

If this method is used to assess the approximately 10 % of Germany, which are of particular significance for the conservation of biological diversity, we obtain values of between 50 cents/m^2 for farmland with endangered segetal plants and almost € 200/m^2 for intact raised bogs. The total value of this 10 % of the land area in Germany comes to approximately € 740 billion, which, at the time of calculation, equaled some 80 % of the value of German productive capital (◘ Table 4.1).

Such economic valuation methods as cost-benefit analyses have the goal of evaluating the macroeconomic benefits of measures. For local decision-making, however, other quanta are often determinant, such as the effect on regional income and employment, as assessed by Job et al. (2005, 2009) for selected protected areas (◘ Table 4.2).

Benefit Transfer

Here, results from other primary studies in which ES-values have already been collected are transferred to the study area and to the services to be tested. There are four stages of *benefit transfer* (Wronka 2004; TEEB 2010): direct transfer, corrected transfer, transfer of evaluation functions and meta-analysis. However, this distinction is of a more or less technical nature. Whether a direct transfer leads to acceptable results, or whether a transfer with an evaluation function is required, depends on the particular problem.

Standard values and simplified evaluation method for the transmission of the value of ES are relatively easy to determine, if the value of ecosystem services is independent of the respective location. One example of this is the value of CO_2 emissions and carbon sequestration. Both have global effects that are independent of the source. The problem in this case more likely involves the correct estimation of the physical effects, which, for example, in the case of the conversion of grassland to farmland, depends on the scope and on the share of organic matter in the soil. Standard restoration costs for the species and habitat-protection functions or services must be defined relatively independently of the location, since the place of compensation is almost always different from the place of impact. For example, nutrient inputs such as nitrates and phosphorus pollute not only the local waters, but ultimately end up in the North or Baltic Seas. Hence, for the nutrient decomposition and fixing services too, uniform values make sense. The same is true for soil erosion (► Sects. 5.3 and 6.6.2). The long-term preservation of the safety of the food supply is a global issue. Long-term shortages or surpluses can therefore also be evaluated on a global scale. The locally differentiating feature would then be the respective agricultural suitability, including soil fertility as an essential input factor.

Benefit transfer becomes more problematical if the value of the service is highly site-dependent.

4

Table 4.1 Compensation values for habitats in Germany, calculated analogously to the *Habitat Equivalancy Analysis* method, taking into account average recovery costs and times (Schweppe-Kraft 2009)

Habitat type	€ per sqm	Area ratio in %	Total value in € million
Heath	41.83	0.22	34,790
Dry and nutrient-poor grassland	8.06	0.27	8037
Molinia meadows	18.51	0.04	2591
Dump floodplain meadows and tall herb communities	6.14	0.10	2315
Extensively used hay meadows	6.14	0.48	10,991
Fens and swamps	9.80	0.03	1088
Extensively used grassland	2.66	1.19	11,897
Extensively used arable land	0.49	1.26	2318
Extensively used vineyards	13.31	0.02	982
Orchard meadows	9.75	0.93	34,125
Extensively used fish ponds	48.93	0.01	1541
Hedges, shrubberies and copses	16.28	2.00	122,100
Natural and near-natural forests	18.44	1.96	135,430
Wood-pastures	20.64	0.09	6594
Low and medium forests	4.47	0.49	8172
Natural and semi-natural forest edges	22.79	0.01	786
Natural and semi-natural forest borders	2.82	0.00	22
Raised bog, natural and near-natural	195.46	0.18	131,914
Transitional bogs and degraded raised bogs	127.42	0.21	100,023
Near-natural standing waters and streams	48.93	0.66	120,698
Total	–	**9.48**	**736,416**

Table 4.2 Economic effects of protected area tourism. (Job et al. 2005, 2009)

	Berchtesgaden National Park (2002)	Altmühltal Nature Park (2005)
Number of visitors	114,100	910,000
Average daily expenditure per capita	€ 44.27	€ 22.80
Gross sales	€ 51 million	€ 20.7 million
Income 1st and 2nd sales stages	€ 4.4 million	€ 10.3 million
Employment equivalent	206 people	483 people

Examples are the recreational performance of landscapes and the prevention of flood damage. A comparably attractive landscape will provide very different recreational services, depending on whether it is located near a metropolitan area, within a familiar tourist area, or in a sparsely populated rural area. The value of the water-retention capacity of forests or floodplains is critically dependent on how extensively and densely populated the flood-prone areas in the drainage portion of the respective watershed are (► Sect. 3.3).

In assessing the capacity of ecosystems to conserve biodiversity using contingent valuation, the question of transferability depends, among other things, on whether the the biodiversity target or programme assessed was local or regional/national in scope (see above).

4.2.4 Conclusion

Economic valuation should be viewed as one decision-supporting method among others. Its main focus of application should be in cases in which the issue is to balance environmental assets and aspects of long-term sustainability, e.g. recreation, biodiversity protection, quality of the residential environment, the self-cleaning capacity of the waterways or soil fertility, against short-term income prospects. It can be used both in decision-making with regard to projects and programmes with negative effects on ES, and for such issues as the amount of money one should invest for the restoration and maintenance of ES.

Some methods of economic valuation are not particularly controversial; for example, there is little doubt that it is useful to have a monetary estimate of the damage costs available when implementing measures that affect the risk of flooding. Nor should there be any fundamental objection against the comparison of costs for reducing the nutrient inputs in agricultural operations into the water, with equivalent measures to increase the self-purification capacity of water bodies.

However, other methods–particularly the stated-preference methods–are indeed controversial. Can we really assume that the statements made by respondents with regard to their willingness-to-pay for maintaining public assets actually reflect their real preferences? How should questions be formulated, and which assets should one ask about, so that the results will be useful in real standard decision-making situations? There is certainly still a great deal of research that needs to be done. According to the existing results, the willingness-to-pay for environmental public goods is usually much higher than what citizens would have to pay in the form of lost income for the maintenance or the provision of these goods.

To date, we are still a long way from having easily applicable valuation approaches for all ES. The criticism that economic valuations address only some aspects of problems therefore often has less to do with the concept of economic valuation. The underlying concept of 'total economic value' (TEV) is based on the preferences of the individual–which is certainly not the worst premise in a democracy–but within that limitation, it sees a very broad range of needs, desires, and motives with respect to the protection and utilisation of nature, which may well also have an altruistic or ecocentric base. If only some of the relevant aspects are to be assessed, as is often the case, this is more likely due to the lack of opportunity, or the necessary resources, to fully ascertain all the effects of the alternatives to be evaluated and assessed. Scientific/ecological impact assessment is often more problematic than economic valuation, as the case of flood protection shows.

In the development of transferable standard assessments or assessment procedures, we are still at the beginning of the development. On the one hand, more primary studies are needed in many areas on which reliable benefit transfer methods could be developed–the travel-cost analysis, which ascertains the quality of areas, has hardly been used at all in Germany; on the other, the development of standards with which those primary studies can be checked for validity is necessary.

Economic valuation is an 'art' that requires a high level of knowledge in the environmental and economic area. Not every economic valuation meets scientific standards. For the uninitiated, this is rarely visible, which can lead to an impression of arbitrariness. De Groot et al. (2002) pointed out that depending on the methods and spatial characteristics in each case, the monetary results of the evaluation of individual ES will vary widely (cf. also above, for the evaluation of agricultural supply capacity). Scientific minimum standards for evaluations could prevent apparent arbitrariness and thus facilitate the acceptance of economic evaluations–especially among those who are not supporters of the interests of the environment and nature.

One of these standards would be the requirement for a generally comprehensible, nontechnical summary, in which not only the total economic value and/or the overall cost-benefit ratio, but also the respective partial values including the explanation of the methods used and their key assumptions would be documented.

Overall, the ES studies which are now extant in large numbers, and which compare the costs and the benefits of measures for the protection of nature and biodiversity, have shown that the usefulness of such measures often significantly exceeds the associated costs. Hence, more conservation and safeguarding of ES lead to an overall gain in welfare.

A critical practice of economic valuation which discloses its assumptions and methods could help business and society find a more sustainable way to manage nature, ecosystem services and biodiversity.

4.3 Scenario-Development and Participative Methods

R.-U. Syrbe, M. Rosenberg, J. Vowinckel

4.3.1 Basics and Fields of Application

Our ecosystems underlie accelerating transitions (Bernhardt and Jäger 1985; Antrop 2005). Some of the reasons are the increased utilisation of renewable energies, globalisation, demographic change and the irresistible urban sprawl. Using scenarios, we can analyse the consequences of these changes for ecosystem services and determine how people are able to intervene in terms of control (Carpenter et al. 2006).

The development of scenarios is only one approach to investigate future trends. Other examples of methods of foresight research are Delphi studies (Dörr 2005), prognoses (Jessel 2000), forward projections (Bork and Müller 2002), the analysis of planning documents, and landscape experiments (Oppermann 2008). However, the discussion of scenarios is deemed to be the key method for argument about sustainability (Walz et al. 2007). It allows a comprehensive examination of the temporal, spatial, and dimensional aspects of ecosystem services (► Sect. 3.3.) since particularly the evaluation of intergenerational justice requires a reasonable view into the future and studies about long periods. Last but not least, the scenario method is a bridging framework for interdisciplinary collaboration on the field of social-environmental research (Santelmann et al. 2004).

Scenarios may be used 'to explore plausible futures for ecosystems and human well-being based on different assumptions about driving forces of change and their possible interactions' (MEA 2005b). A simple definition is 'scenarios are hypothetical sequences of events, constructed for the purpose of focusing attention on causal processes and decision-points' (Rotmans et al. 2000). The aim of a scenario is, therefore, to identify and to compare possible options of action. Instead of only following a single future trend, a tree of possibilities can be explored (Oppermann 2008) enabling to assess the desired and manageable ones among them.

Due to their decision-preparing function, scenarios are part of an action framework and, therefore, suitable tools:

- To draft capabilities in order to prepare for coming occurrences
- To estimate the risk potential of strategies in order to demand for action
- To draft options for action and to compare them in order to choose the most feasible
- To describe the effects of individual measures to other fields of action in order to evaluate the suitability of that measures in a broader area of consideration

Depending on the application purpose, the elaboration of scenarios can be done by experts alone (analytically) or by participation together with actors from policy, economy, NGOs, and the public. The following description of the methodical framework is restricted to the analytical version. Selected participative procedures are presented in a case study below. Both versions can be applied in two forms of expression: Either scenarios are narrated in so-called storylines (Rotmans et al. 2000) using mainly qualitative statements, or quantitative scenarios are calculated depending above all on model simulations. Analytical scenarios are often quantitative, whereby participative approaches have got predominantly a qualitative character. There is also a difference between projective and normative

Well-Known Scenarios About Environmental Issues

Environmental issues were often central for scenarios with both quantitative and qualitative approaches according to the overviews given by Alcamo (2008) and Albert (2009). The first quantitative scenarios used hydrological models (Aurada 1979). A more recent prominent example is the so-called World Water Vision (Gallopin and Rijsberman 2000). The study of Wolf and Appel-Kummer (2005), funded by the German Federal Agency for Nature Protection, addressed consequences of demographic change to nature protection. Several analyses dealt with the impacts of land-use change within rural areas (Dunlop et al. 2002; Nassauer et al. 2002; Haberl et al. 2004; Bastian et al. 2006; Bolliger et al. 2007; Lütz et al. 2007; Tappeiner 2007; Tötzer et al. 2007). But also shoreline and sea issues were central for scenarios, such as the North Sea (Burkhard and Diembeck 2006) or the Great Barrier Reef near Australia (Bohnet et al. 2008).

An increasing number of recent publications evaluate environmental scenarios using landscape functions or ecosystem services such as Dunlop et al. (2002), Nassauer et al. (2002), Fidalgo and Pinto (2005), and Seppelt and Holzkämper (2007). The Fourth Assessment Report (Pachauri and Reisinger 2008) of the Intergovernmental Panel on Climate Change (IPCC) addresses the effects of climate and socio-economic changes to a large number of ecosystem services at the global level. Examples of integrated man-environmental research through scenarios are the Millennium Ecosystem Assessment (MEA 2005b), which includes foresight and backsight analyses of 50 years, and the Global Environment Outlooks of UNEP, of which the fourth generation is available (UNEP 2007) and the fifth one is under revision (UNEP 2011).

scenarios. The former searches for the implications of assumed trends and the latter starts with (desired) future goals and explores how to act in order to meet them (Nassauer and Curry 2004).

4.3.2 Framework of Scenario Development

The methodical framework presented below is particularly designed for scenarios of landscape development that should be evaluated by ecosystem services. The framework was tested on the county of Görlitz within the Landscape Saxony 2050 research project (funded by the Saxon Department of Science and Arts). The scenario methodology consists on a combination of approved single procedures and fits them to the problems of landscape development. The methodical basis includes the works of Reibnitz (1991), Gausemeier et al. (2009) from business science and Alcamo (2008) from environmental science.

The framework uses an explorative forecast approach. This approach is open-ended, i.e. there is no direction and range of developments set from the beginning. Quantitative and qualitative approaches can dominate or be combined. The framework consists maximum of seven steps. Depending on the main question and application task, not all steps have to be run-through completely. ◘ Figure. 4.10 gives an overview of this method.

Step 1 comprises, first, the organisational preparation of scenario process, second, the formulation of a principal question and, third if necessary, a specification by core topics. The principal question defines the overall objectives. A time horizon and the delineation of the study area belong to that. If the principal question is rather complex, the object of investigation should be confined by core topics. Regarding the case study, the time horizon (2050) and the study area (Görlitz County) were fixed, but the principal question was defined rather broadly as 'How will the ecosystem services be altered due to future landscape change?' Therefore, the principal question had to be specified using the two core topics 'biodiversity' and 'renewable energy' that were treated separately. Both topics were very important in political and social debates.

Step 2 consists of the selection of driving forces and ecosystem services that should be considered. That is, the scenario expert team has to select which drivers are interesting to the principal question in respect to the core topic and the impacts they have on the ecosystem services (ES). Therefore, the selection of drivers and ES has to be done simultaneously since both depend on each other. A good selection and precise definition of driving forces is crucial for the whole scenario development because if the

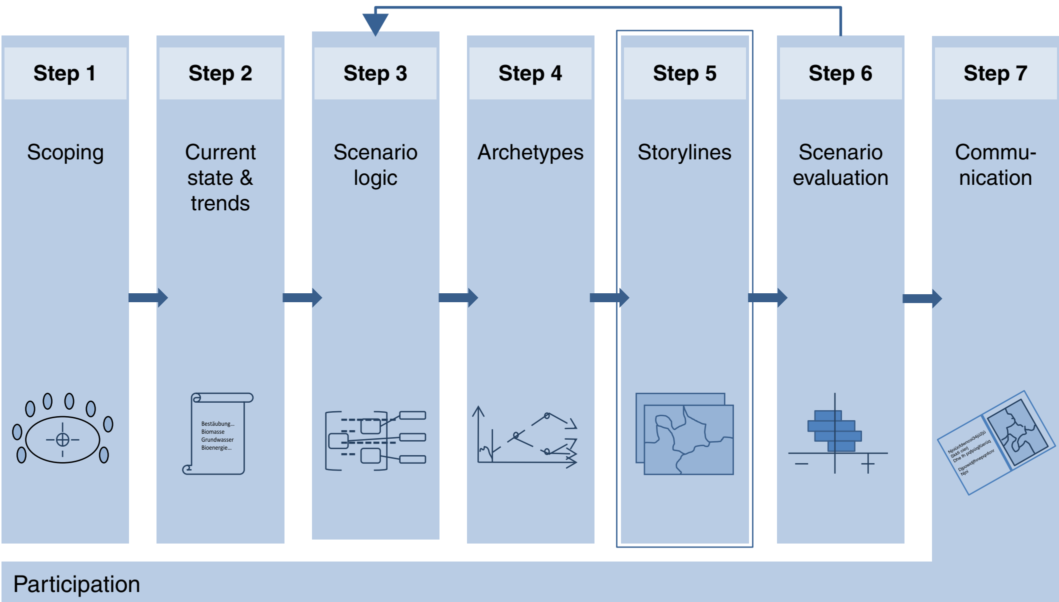


Fig. 4.10 Working steps of the described scenario framework. © IÖR/Syrbe

selection is to broad it hampers the communicability of scenarios. If the drivers are too imprecise and cannot be described by clear indicators, they will complicate the discussion as well as the quantitative processing. One bad example would be choosing 'energy and mining' as a driving force since several directions of development could be implied. On closer consideration, hundreds of driving forces can be identified. But only a small number (<10) must be considered and each of them should be describable by a single measure and a known actual value. For this, thorough investigations are necessary, which will also be useful later on.

Step 3 defines the logical scenario structure. The main purpose of scenario development is to draft different future visions. To do so, the drivers that are to be variable within the scenario process need to be chosen. A differentiation can be achieved connecting the variable drivers with diverse trends. Of course, this differentiation is only possible for a small number of drivers. Empirically it does not make sense to use and vary more than three key drivers concerning the amount of work and the straightforwardness of the whole process. The other drivers are defined to be unvaried between several trajectories. The unvaried drivers are called framework conditions and must be described as accurately as possible using also external prognoses or expertises. On the contrary, the variable drivers open up the possibility space of scenarios and, thus, are called key drivers.

Step 4 implements the abovementioned logical scenario structure. Therefore, an overview of the current situation is needed. An initial ES assessment should be made using the middle pillar of the EPPS framework (► Sect. 3.1.2) unless it already exists. The key drivers have to be connected through a small number (commonly two by four) of trends concerning their future development as it is interesting for the principal question and also relevant for altering the ES under consideration. The trends may not only be linear but can also be defined accelerating, retarding or erratic. The description does not need to be exact, but rather generic. An established way of description is using pictograms for the several trend types (◘ Fig. 4.11). Not all trend combinations can be combined because contradictions are possible. An appropriate number of plausible combinations (so-called bundles) must be selected. These bundles guide the initial ways to develop and describe scenarios in detail, which will be done in step 5.

Step 5 contains the wording and specification of scenarios. The selected bundles enable to derivate several future trajectories. They receive short

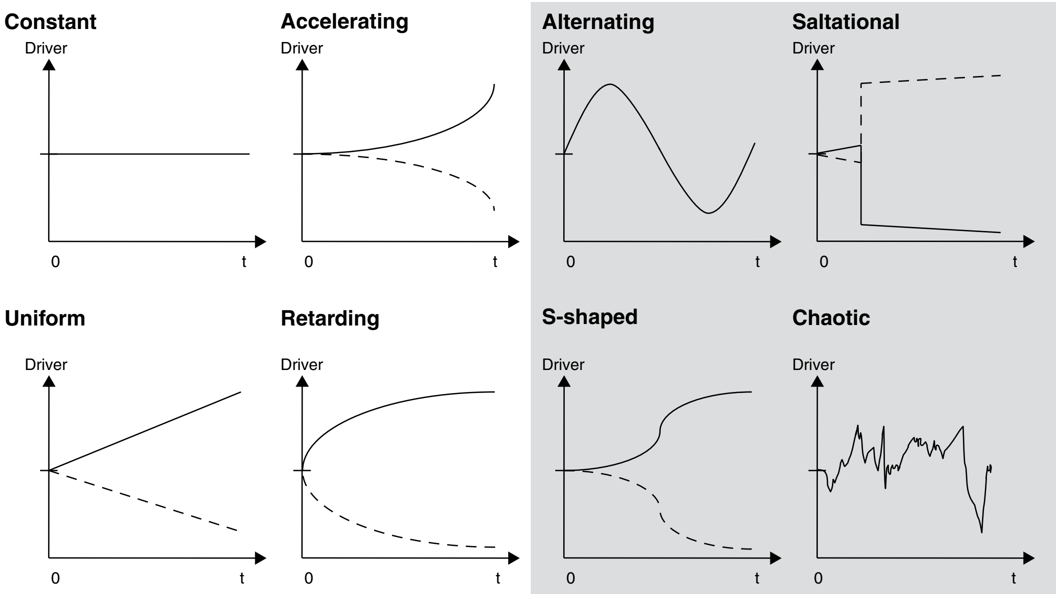


Fig. 4.11 Pictograms for the trend types of key drivers; *strait line*: positive alteration; *dashed line*: negative alteration; *white background*: basing types; *grey background*: combined types. © IÖR/Syrbe

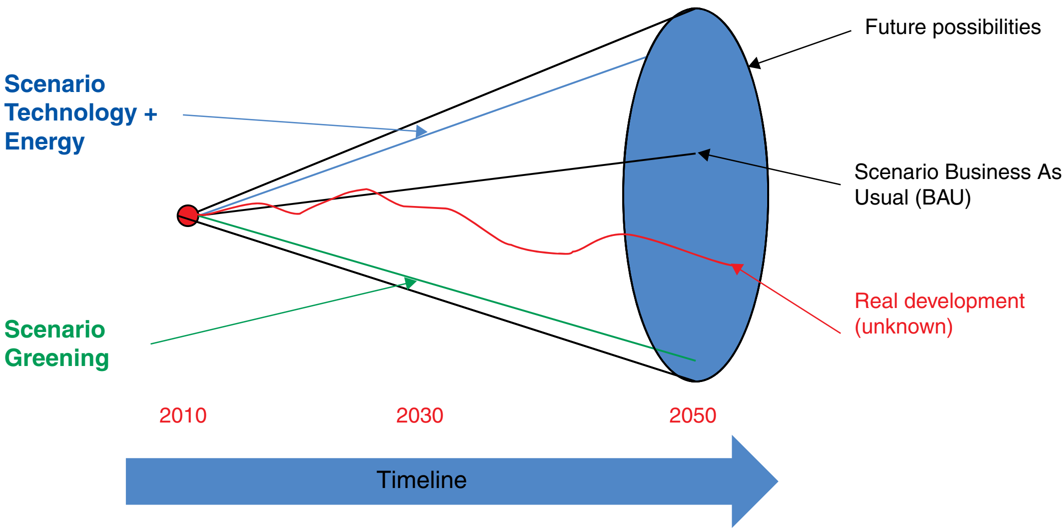


Fig. 4.12 Scenario funnel with schematically hinted trajectories. © IÖR/Syrbe

names characterising the assumable end points in future, the so-called archetypes. For instance, the archetypes of Landscape Saxony 2050 scenarios read 'Business as usual (BAU)', 'Greening' and 'Techno + Energy' Fig. 4.12. The core result of this step is a storyline that describes the future situation (sometimes also the steps towards it) and that give reasons for the most important conclusions. To achieve this goal, the interdependencies between all drivers (variable and framework conditions) have to be analyzed. The so-called cross-impact analysis can be treated with the help of a matrix to ensure

Nonlinear Phenomena of Scenario Development

There are some known nonlinear but nevertheless typical phenomena in connection to scenario method: First, particular situations may lead to a strong determination of a previously open development. The so-called lock-in-phenomena arise e.g. from exhausting resources or decision of a competition. Sometimes one option among competing technologies can win and outlive all the others. Second, a seldom but powerful incident could change all options of development. These so-called 'wild cards' should be discussed separately from the main round because many participants are not frank enough to accept them, even though their treatment may be important for taking precautions for the future.

that all possible two-dimensional effects are considered. Simulation models, balances, and other quantitative methods resulting in tables and numerical values are frequently used in expert scenarios to figure out multidimensional interdependencies. The participative scenario framework prefers stakeholder discussions to work out qualitative results. Admittedly, these results are not quantitatively representable but often more complex. A proven tool to facilitate the discussion is scenario mapping. To draw items into a map gives an overview of spatial dependencies and helps to figure out possible environmental conflicts as well as the points of interest for the actors. These maps are an essential basis for a subsequent evaluation (step 6) and instructive abstracts of scenario outcome.

Step 6 is the evaluation part of scenario outcome. Storylines, tables and maps underlie a comparing evaluation to give answers to the principal question and to ensure the scenario process quality. The evaluation can be spatial or nonspatial depending on how the scenarios are mapped. The evaluation of scenario outcome regarding ecosystem services does not need to be restricted to singular values. Rather, the future cross-impacts of the services, their so-called trade-offs (▶ Sect. 3.1.2) as well as synergies should also be unfolded. Risks and suitability areas should be delineated and compared. The main purpose of this step is to draw conclusions from scenario results for management options and possible future strategies. The aim is not only to figure out the best storyline but also the best measures that will accomplish this. It is possible that a repetition from step 4 onward is necessary to specify them anew and to rethink the scenarios therewith.

Step 7 comprises all measures of scenarios' communication and participation with the concerned actors (or customers). The participation tools are specified in the next section (▶ Sect. 4.3.3). Although it is placed as the last step, participation shall start with the beginning of a scenario development and pervade throughout the whole process. This way, the methodology can have some loops between mainly expert-oriented steps and steps with more participation. At the interface between both modes of work, data must be translated into easily comprehensible presentations, and meanings have to be quantified the other way around. Lastly, the scenario results have to be published at the end of the process to enhance public awareness and (hopefully!) application.

4.3.3 Participation and the Case Study Görlitz

'Participation' is the cooperation of actors, stakeholders or interested individuals within a scenario development or during an assessment; the concerning method is called participative. The main reason for the inclusion of decision makers by participative methods within an assessment or a scenario development is the social appreciation of the results. Another good reason for participation is that assessments are most helpful if the users take part in it (Carpenter et al. 2006). Additionally, participative scenario workshops reveal educational effects for the participants (Alcamo 2008). Therefore, it is recommendable to involve young people, particularly if long-term scenarios are being developed.

The cooperation with participants that are lay people within methodically sophisticated methods is challenging regarding the quality of communication. Experts must be able to interest people and engage them to get involved in the cause. The crucial problem is to ensure a comprehensible flow of information from scientific knowledge to messages in normal language and vice versa. Therefore, a pool of hints shall be proposed, which may be extended in several ways.

Types of Participation for Development of Scenarios or Ecosystem Services Assessment

- Workshops (with group work, presentations and perhaps stage discussion)
- Small group participation events such as world café or focus group interview
- Personal interviews (survey with prepared questions or thematic guideline)
- Public surveys (oral, by letter or on the web)
- Stalls at exhibitions, fairs or congresses
- Excursions (empirically with high motivational effect)
- Culturale events (cinema show, theatre and suchlike) with following discussion
- Teaching units in schools, other educational institutions, or outdoor
- Internet forum, blog, etc.

The participative work on scenarios, mainly using a workshop, is called a scenario exercise. It is the methodical core of the whole scenario development. The most important steps of ► Sect. 4.3.2. have to be handled therein. The scenario exercise should be combined with the working steps that are executed only among experts as well as with alternative forms of participation (► Box 'Types of Participation'), in order to minimise time exposure for the participants, to activate them without boring them, and to ensure a high degree of involvement also for those who are not keen on debates.

Elements of a Scenario Exercise

- Invitation of genuinely interested participants
- Introduction: explanation of aims and methodical steps
- Mind opener to stimulate creativity (e.g. quiz)
- Brain-storming to catch maverick ideas
- Suggestion talk(s) by experts
- Ballot about alternative proposals (e.g. by stick points)
- Plenar discussion for central decisions
- Working groups developing particular scenarios
- Breaks with social events (e.g. dinner)
- Plenar presentation of working group's results with final discussion
- Protocol shipment of the final results to all participants

The actual scenario exercise can consist of several elements (► Box 'Elements of a Scenario Exercise'). All essential information including the time frame must be communicated with the invitation beforehand to avoid the worst case: unsatisfied participants frequently discussing off-topic issues or query the meaning of the exercise in general. The first important topic on the schedule should be an introducing explanation of sense, aim, and background of the exercise, eventually completed by a short lesson on scenarios. Second, a so-called mind opener can help to get the participants in the right mood to bear creative ideas and to break away from their everyday problems, as well as to prevent them from judging prematurely. Therefore, unexpected questions, a quiz, or a flashback into the past can be recommended. These elements can also be used later to make the event less formal. The actual scenario discussion shall be done preferably in working groups. Intermediate results have to be retained periodically to ensure the progress of discussion. Spontaneous ideas should be recorded neutrally at this point and systematised only later. Because one-day workshops can be very exhausting and will only be successful for good teams, Ringland (1997) recommend two half-day rounds instead, which can be separated by an informal evening event. Graphical, textual, cinematic and interactive media help to facilitate the discussion if they are specially geared to the participants.

Some of frequently made mistakes should be mentioned. A possible participants' irritation due to incomplete information has already been noted. Additionally, frustration can arise from overloaded presentations, a boring schedule, or too slow progress in scenario elaboration. To avoid such undesirable situations, breaks should be inserted that can be used by the scenario experts to develop intermediate results further and enrich them by additional information (i.e. from simulation models) to get a faster progress and make the meeting more interesting for the participants. The hope to get quantitative data by a negotiation among actors would be mostly disappointed: data requested from participants remain often incomplete and vague; therefore, they must be completed and sophisticated by experts work. Often, a successful participation process needs more preparation time than execution time (Walz et al. 2007).

Tips for Planning a Successful Scenario Exercise

Timely invitation of participant
- Information about the venue, aims, duration, and fee as well as possible cost reimburse
- Invitation shall be motivating, provoking, exciting, or funny
- Homework (i.e. a questionnaire) can save a working step and prepare for the topic

Introduction by the scenario team
- Aims and schedule of the whole project and of the particular event
- Introduction should be short, but include organisational information (breaks, meals, etc.)
- Introduction highlights the possibilities of participation

Mind opener to activate creativity (possibilities)
- Enquiring wishes or nightmares for future
- Asking to draw an own desire scenario
- Provoking (i.e. through theses or artistic illustrations)
- 'Fairy question': 'What do you want to ask a time traveler from the future?'

Brain storm to obtain creative ideas before people hear lectures
- Ballot about drivers or evaluation criteria
- Nomination of surprising incidents to be regarded ('wild cards')
- Risks and problems for future

Key note lectures from scenario team and external experts
- Participants get comparable information as basis for discussion
- Sharing the most recent state of the art about trends and drivers
- Current state of the study area

Group work to draw particular scenarios
- Avoid strong/weak division of working groups to not confine the creativity of the weak group
- Group division should consider the interests of members
- Each group needs a moderator from the scenario team
- Job description must be prepared for groups and moderators
- Each group elects a presenter at the beginning

During two projects ('Landscape Saxony 2050' and 'LÖBESTEIN') in the East Saxon county Görlitz, Germany, additional experiences from scenario workshops were collected and will be shared below (► Box 'Experiences from Görlitz as Regional Example'). The authors developed participative scenarios about the increasing use of renewable energies and the protection of biodiversity there.

4.4 Complex Analyses, Evaluation and Modelling of ES

4.4.1 Background

K. Grunewald, G. Lupp

"To make simple things complicated, is daily routine, to make the complicate things simple, this simply is creativity. (Charles Mingus)"

Nature, our environment, and society are complex systems. Complexity means that, the reaction of a system is not predictable as a whole even if we know single reactions and interactions of its components precisely. The characteristics of complex-

Experiences from Görlitz as Regional Example

In the beginning, a world café event, where participants visited several thematic tables to discuss input variables (drivers, trends, wishes, aims, standards, values) in brief sequence, was organised.

The workshop preparation was done by Internet surveys. Online tools such as ► http://kwiksurveys.com/ are available that are easy to design and able to provide statistically edited results. Unfortunately, a personal email address of all participants must be known. Preconditions to use this tool are the participants' accordance and engagement. The tool worked well among the internal and external experts but not with the other participants. Therefore, survey forms (as PDF, per email of fax) were sent out in order to involve all interested actors. However, long word/excel query catalogues could not be used succesfully since they were not returned on time and fully completed except by the respective expert team.

In the workshop, statements from several experts were discussed and enriched by additional thoughts. However, the self-introduction round of participants occasionally escalated to time-consuming talks. Good experiences were made with three questions asking for one-sentence answers from all participants in the beginning (who are you, how do you feel about the topic, what is your intention). The selection of trends, drivers, and trajectories is not suitable for a full auditorium and should be implemented in other ways (see suggestions above). A good scope was to deliver several proposals that the actors could choose by participation in specific working groups or table discussions. After a certain period of difficult discussions, a change through playful insertion was appreciated. Group works with about 5 participants each were most efficient. Many participants were skilled in handling maps and used them to discuss allocation questions intensively. Therefore, well-prepared maps and drawing utensils were valuable. The moderators must keep in mind the time frame as well as those participants who don't impose themselves in discussion and activate them directly.

ity are numerous elements that interact with each other and the reaction as a whole is unpredictable (Riedel 2000). Examples for complex systems and limitations for their predictions are, for instance, weather forecasts, the prediction of market trends at the stock exchange, but also the reactions of ES. Disturbance of complex ecosystems might lead to severe and irreversible new states (SRU 2007). Land management can be considered a complex system. Land use and forestry affect nature in many ways, e.g. water cycling, soil fertility, biodiversity or regional value adding (► Chap. 6).

Complex or Complicated?

An airplane is a complicated thing. It consists of many different parts. However, it does not contain a real secret. This means, difficult tasks can be solved by knowledge.

A five-course meal is complex. You have to know the different ingredients. But when you prepare the different dishes, it does not necessarily means that you are getting a delicious meal. Systems with many different interactions not working on a simple 'if-then' principle are dynamic and multilayered and, thus, are complex.

The aim of the ES concept is to cope with the challenge of interactions and complexity of ecosystems and to describe impacts and consequences for human well-being. A comprehensive assessment of ES demands enormous efforts and is only partially adequate to serve as a basis for politicians and stakeholders to support decisions by involving all different demands.

By breaking down, abstracting and weighting complex issues are simplified. Therefore, just like in a caricature, they are easier to understand through simple and concise means. With simple means and a few lines, significant and striking attributes of a person or a situation can be drawn. Complex systems can only be determined by observations of patterns. They can be observed in the abiotic and biotic environment or in society (e.g. different soil substrates, routines, behaviour). ES patterns can be analysed with a matrix of supply and demand for certain land-use types (► Sect. 4.1) and within Contingent Economic Assessments (Examples in ► Sect. 4.2 and ► Chap. 6).

Visions and intentions like the concept of 'sustainability' and the 'ES-concept' could be seen as a tool to influence patterns and types of land uses. If new patterns occur in complex systems, a tipping point has been crossed. One of the goals of research on ES is to figure out tipping points and how they are influenced by human activities. It is one of the core challenges to determine the development of those systems (scenarios, alternatives, ▶ Sect. 4.3; modelling, ▶ Sect. 4.4.3) and forms a basis for regulation and steering (policy, incentives, planning, governance ▶ Chap. 5).

The ES concept is intended to support solving and balancing complex problems with tools and methods. It strives for integrated approaches by analysing, assessing, and weighting ES based on scientific methods by using all available information while including human needs. The ES concept requires weighting between quick and cheap assessment procedures (e.g. rough estimations based on 'rapid evaluation tools') and more detailed, elaborated, time demanding, as well as more expensive examinations (intensive assessment of all different ES aspects).

In the following section, a broad application of the ES concept will be presented using a case study on 'impacts of an increased biomass production for energy purposes'. It shows how ES can be selected and assessed, how different approaches for evaluation ES can be used, and how regional stakeholder can participate in these processes. Finally, the ES model 'InVest' is presented demonstrating its use and describing strengths and weaknesses of this model.

4.4.2 Energy Crop Production–A Complex Problem for Assessing ES

G. Lupp, O. Bastian, K. Grunewald

The increased production of biomass for energy purposes is a prime example for the increased use of ecosystems driven by strong political interest. The European Commission has set mandatory targets for all member states for the use of renewable energies. The share of renewables has to double between 2010 and 2020 according to this policy. Half of the renewables share is to be derived from biomass (Commission of the European Communities 2007). With respect to conflicts and minimising impacts, the EU commission has developed a biomass action plan and requested all member states to develop national biomass action plans. The German biomass action plan (BMELV/BMU 2009) emphasises climate protection, regional value adding, the strengthening of rural and peripheral regions, and the protection of biodiversity, soil fertility, waters as well as air quality as the core goals for biomass production using annual energy crops or woody biomass.

To achieve these goals and to minimising conflicts, stakeholders have to be included, and the acceptance for biomass has to be increased by informing and involving the lay public through adequate communication and consultation (BMELV/BMU 2009). Although ES are not explicitly mentioned in the document, ES have to be secured and enhanced in a sustainable way when energy crops or woody biomass are cultivated. This document already indicates possible methodological steps and approaches for assessing impacts of biomass production on ES.

In order to improve ES and biodiversity protection in sustainlable land-use management practices, we suggest the following steps (see also ◘ Fig. 4.13 and Lupp et al. 2011):

First, relevant economic and ecological elements, especially ES, have to be selected. In the case study food and feed production, provision of energy derived from wood and energy crops, variable cross margins for farms, biodiversity, carbon fixation, pollination, provisioning of drinking water, water discharge regulation, erosion control and outdoor recreation opportunities were chosen.

In our work, we follow the '**DPSIR**-steps' (*Driving Forces, Pressures, State, Impact, Response*) according to the OECD (2003). This approach involves a system-analysis view and describes a methodological procedure for characterising the impacts of socio-economic activities on the environment and ways to reduce or halt these impacts (BAFU/BFS 2007).

In the first step, the *Driving Forces* of an increased energy crop cultivation and timber extraction are assessed by analysing energy policies, regulations set by legal instruments and incentives

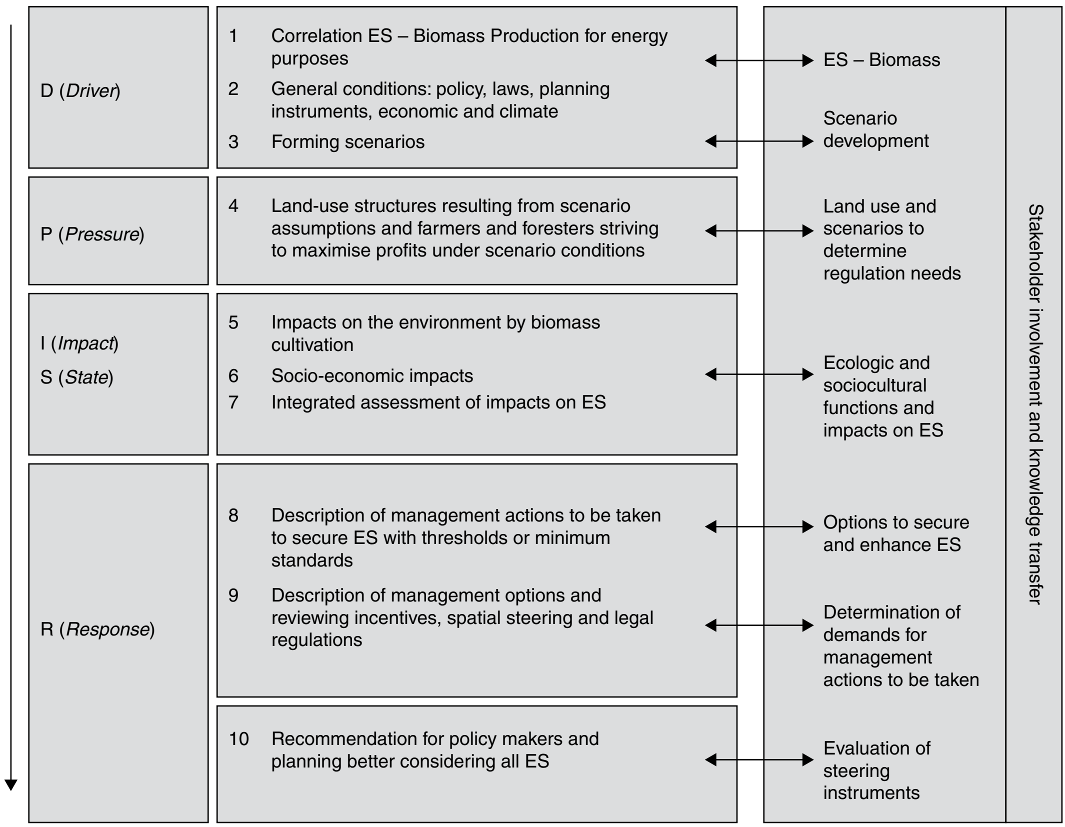


Fig. 4.13 Schematic approach to assess and evaluate ES in different scenarios

as well as economic situations and climate conditions. Based on these findings, land-use scenarios are developed. By using scenarios, future land-use patterns (*State*), their impacts (*Impacts*), and *Pressures* on ES can be determined. Using this data, necessary actions to maintain or improving the provisioning of ES can be identified and possible options for improved regulations (*Responses*) can be developed (Fig. 4.13).

To cope with the challenges and adaptation of land management concepts, regional approaches at the landscape level seem to be among the most promising since influencing factors and the demand for specific solutions may differ (Rode and Kanning 2006). Case study regions to be selected should provide heterogeneity. Although certain factors might have global impact, different landscape units might react completely different.

To address dimension and different landscape scales, different types of units should be assessed reaching impacts on regional level down to individual land parcels. The latter is important for putting objectives into practice by farmers and foresters to carry out precise management actions to support certain species, e.g. to maintain deadwood in forests for birds and insects or provide patches for skylark (*Alauda arvensis*) in intensively managed fields as nesting habitats.

Different energy crops and the way they are cultivated lead to specific impacts on ES, some examples can be found in Table 4.3. But also different natural conditions or landscape character might influence impacts on ES.

In an integrated assessment, different ES can be compared with each other. For example, so-called spider-web diagrams can be a suitable instrument to describe them (Fig. 4.14).

Table 4.3 Impacts of energy crop cultivation on ES on the example of corn, rape and cereals compared to cultivation of woody biomass in short rotation coppices (derived from Lupp et al. 2011, modified)

ES	Factor	Impacts of annual energy crops (corn, rape or cereals)	Impact of woody biomass cultivation in short rotation coppices (SRC)
Provisioning Services			
Providing biomass for energy purposes	Land use, crop rotation	Increasing food prices, increasing prices for land tenure and land buy; crop diversity is reduced, monocultures increase, in some cases conversion of high nature value grassland into fields	In some cases cultivation of SRC on high nature value grassland
	Income of farmers, income by land tenure for land owners	Increased opportunities to gain revenues for farmers, however mainly driven by incentives and payments under the German Renewable Energy Act; option to react to different market demands every vegetation period	New options to gain revenues, however depending on incentives and payments under the German Renewable Energy act; No possibilities to change crops for 20–30 years, no flexibility to react on changing demands
Water provisioning	Groundwater level, moisture content in soils	Groundwater levels depend on cultivated crops, water quality can be affected by fertilisers and pesticides	Groundwater level and regeneration rates may be lower since SRC demand more water and have higher interception rates; Better water quality due to lower fertilising and herbicide use
Regulatory Services			
CO_2-storage	Energy input for planting, fertilizing, pesticide treatment and harvest	Substitution of fossil fuels, however quite a lot of energy input for cultivation of annual crops	Substitution of fossil fuels; energy input increases, when SRC wood chips receive additional treatments (e.g. technical drying)
	CO_2-sink, emission of greenhouse gases	Greenhouse gases are emitted when high nature value grasslands are converted, energy crops are cultivated on marshlands and drained bogs, additional greenhouse gas emissions when nitrogenous fertilisers are used	Can be used as carbon sink, humus and subsoil biomass is accumulated, however, greenhouse gases are emitted, when wetlands are drained for SRC or high-growth forests are converted to SRC
Nutrient and humus regulation	Nutrient input	Often high nutrient input and excessive use of biocides	SRC have lower nutrient input compared to annual crops; negative impacts, when high nature value grassland is converted to SRC
	Nutrient leaching	High nutrient leakage for some crops and inappropriate farming practices	Lower leaching rates compared to annual crops
Reduction of soil erosion	Vegetation cover	High risks for erosion by wind and rain, when there is no cover by crops	Almost permanent cover by woody plants reduces erosion caused by wind and rain, quick and Intensive root penetration lowers soil erosion, neighbouring fields benefit from SRC

Table 4.3 Continued

ES	Factor	Impacts of annual energy crops (corn, rape or cereals)	Impact of woody biomass cultivation in short rotation coppices (SRC)
Water retention	Water runoff	Intensive water runoff when there is no vegetation cover (relevant especially for corn)	Lower water runoff, water retention effects increase with increased height and size of SRC
Water purification	Pesticides in groundwater bodies, cost for water purification	High nutrient input in groundwater leads to increased costs for water purification, water quality might not fulfill minimum standards set by law, water has to be provided from less affected regions	Lower nutrient input in groundwater and water purification compared to annual crops
Groundwater recharge	Groundwater level	Higher groundwater recharge rates compared to perellial crops, SRC or forests	Lower groundwater recharge rates due to dense vegetation cover, higher interception rates compared to annual crops, groundwater levels might be lowered
Pest control	Resilience Amount of biocides used	Higher pest risks, greater hazards due to increased spreading of European corn borer (*Ostrinia nubilalis*) or western corn rootworm (*Diabrotica virgifera virgifera*), increased use of biocides or genetically modified plants	Higher resilience for pests, pest infestation is often less harmful for perennials, however problems with rust (*Melampsora*), beetles (*Melasoma populi*) and rodents (e.g. mice)
Pollination	Amount of pollinating insects	Corn and cereals are wind pollinated and provide no nourishment for pollinating insects, rape has only a short flowering period in spring, Intensive energy crop cultivation reduces accompanying vegetation, therefore no larger amount of permanent sources for pollinating insects	Trees are harvested before they are mature, some willow species can be used as nectar source
Biodiversity	Cultivated species, types and proveniences, cultivation patterns and forms of land use	Intensive cultivation, few different species, genetically modified species, few accompanying vegetation, harvest before maturation	Few high yield (hybrid) species, genetically modified species, few structures within a SRC, however longer rotation periods with 2–5 (20) years
	Bird species	Intensive cultivation provides little habitat for species dependent on less intensive land management practices, e.g. ortolan bunting (*Emberiza hortulana*), skylark (*Alauda arvensis*), early harvest in summer before matureness reduce feed for species depending on grains (e.g. migrating birds like geese)	Habitat and nesting for bird species depending on hedges and woodland, however mainly ubiquist species; conversion of fields reduce habitat for ground nesting birds

Table 4.3 Continued

ES	Factor	Impacts of annual energy crops (corn, rape or cereals)	Impact of woody biomass cultivation in short rotation coppices (SRC)
	Accompanying vegetation and fauna	Few accompanying vegetation, few habitats for animals (e.g. insects and small mammals), Intensive soil treatment and biocides	Accompanying vegetation and fauna increases but mainly ubiquist species; when managed with longer rotation periods, SRC can contribute to some extent for structuring landscapes and contribute to a green infrastructure; however loss of habitats, when SRC replace low impact agriculture, fallow or marginal land or high nature value grasslands
Sociocultural Services			
Ethical values	Share of food production compared to energy crop cultivation	The use of cereals and other plants suiting for human nutrition is perceived negative from an ethical perspective (increasing prices for food and starvation in developing countries)	No direct ethical conflicts, since woody biomass derived from SRC is non edible; however indirect effects: Farmland is converted for energy production and lost for food production
Identity	Cultural heritage, traditional farming practices	Traditional land-use forms and specialised crops disappear, cultivation of less productive species diminish	New landscape elements, can sometimes pick up traditional land-use forms or act as modern interpretations, willow cultivation (*Salix spec.*) used to be a common landscape element used for basket-weaving in the past, today in the context of modern, sustainable energy production
Education value	Energy crops in research projects	Only few species, research mainly on increasing yields	New land-use form, object of numerous and multidisciplinary research
Landscape aesthetics	Diversity of landscape patterns	Mainly monocultures and large unstructured fields, shortened crop rotation, however seasonality and different appearance of fields during the year; attractive yellow rape blooming in spring	Can structure large agricultural landscapes
	Visibility, vistas, viewing unique, landscape features	Visibility is reduced and vistas blocked in summer (e.g. high corn stands), distant views in winter	No visibility, vistas are blocked except a few months after initial planting and for a few months after harvesting

Based on data collected by Kort et al. 1998; McLaughlin and Walsh 1998; Börjesson 1999a, b; Heidmann et al. 2000; Liesebach and Mulsow 2003; Londo et al. 2004; Windhorst et al. 2004¸Bringezu and Steger 2005; Burger 2005; NABU 2005; Rode et al. 2005; SRU 2007; Bardt 2008; Lee et al. 2008; Ericsson et al. 2009; Hillier et al. 2009; Rowe et al. 2009; Cherubini and Stromman 2010; Greiff et al. 2010

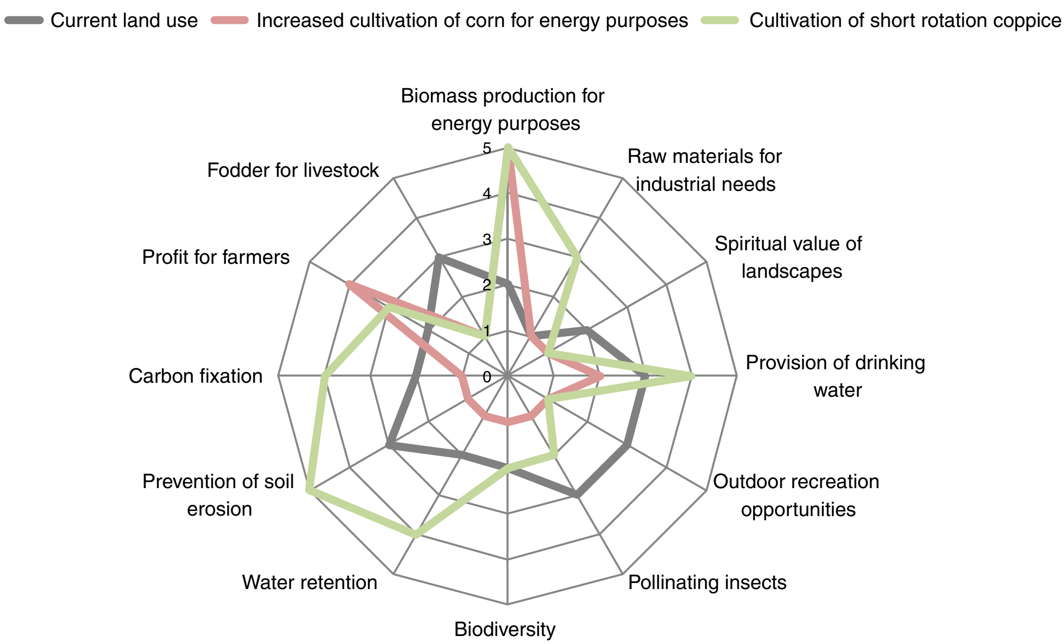


Fig. 4.14 Exemplary diagram of ES modification by energy crop cultivation

Scenarios

Scenario analyses aim to determine impacts of biomass production. Undesirable effects (Trade-offs, *disservices*) should be eliminated or at least be minimised. As demonstrated in ► Sect. 4.3, scenarios are suitable to evaluate the time and space aspect and to compare and weight different resulting developments or different options for action. Scenarios also provide many possibilities to involve stakeholders.

In the Moritzburg small-hilly landscape 10 km north of Dresden, expert-based scenarios were created describing impacts of different policies resulting in distinct laws and incentives like EU common agricultural payments for farmers. These assumptions lead to scenarios allowing for impact assessments for different possible developments. The three scenarios are:

- First scenario: Abandonment of livestock
- Second scenario: Biomass production for energy purposes
- Third scenario: Optimising ES from a nature conservation point of view

All three scenarios lead to different land-use patterns. In the 'Biomass' scenario, the share of corn increases. High-nature-value grassland along rivulets will be replaced by short-rotation coppice. Land use will be intensified to compensate the loss of agricultural land needed for biomass production. The third scenario 'optimising ES' will result in diversified land-use patterns.

Biophysical Approaches

To assess the impacts of an increased cultivation of energy crops on biodiversity and ES, expert-based approaches of landscape planning can be used. They are described in many methodological handbooks e.g. in Bastian and Schreiber (1999). Usually, semiquantitative assessments of the landscape functions, a subject of protection, a potential or risks, or–speaking in terms of provisioning–ES are carried out. Usually a five-step Lickert scale is used stretching from 'very good condition' to 'very bad condition'. Items evaluated are e.g. erosion sensitivity, scenic quality or biodiversity that might be affected by large scale monocultures like rape or corn (◘ Table 4.3). Often impacts on eye-catching species like skylark or lapwing are analyzed. They serve as umbrella species for certain types of habitats or groups. Choosing them helps raising awareness among different stakeholder groups and lay persons for more conceptual approaches like biodiversity or ES.

▪ Monetary Approaches

Many ES can have economic values, e.g. a demand for ES on markets exists or the provisioning or maintenance has costs (Baumgärtner 2002), e.g. forest growth simulators like SILVA 2.2 also integrate economic evaluations (Pretzsch 2001). For agriculture, econometric decision models exist and also provide information on economic effects of different management objectives (Kächele and Zander 1999). With these models, decisions of foresters and farmers can be described and effects of legal or regulatory frameworks can be implemented (e.g. mix of different tree species or crop rotation). The models describe developments when managers would solely act in rational profit maximising terms.

Another option is to use opportunity costs (► Sect. 4.1). They quantify losses, which derive from maintaining low impact practices on fields in favour of biodiversity. For example, it can be calculated how much money would be lost if a farmer does not cultivate small patches in large-scale fields to provide habitat for skylark (Brüggemann 2009).

▪ Demand-Based Approaches

One option to assess the demand for ES are surveys among the population. For example, the author-conducted interviews at different locations within the study area led to interesting results. The provisioning of drinking water and habitat for plants and animals is considered to be very important for the interviewees, while providing renewable energy from biomass is almost irrelevant.

▪ Transdisciplinarity and Participation

Transdisciplinary approaches are characterised by close cooperation between researchers and practitioners. The idea is to implement the work to solve real-life problems (Müller et al. 2000). Participation means active involvement of stakeholders and other interest groups in decision-making (UBA 2000; Förster et al. 2001). Therefore, it is useful to integrate key stakeholders in each research step, to let them participate, and to involve them actively in the project process. For example, it is possible to involve them in the scenario work, e.g. by letting them decide about key drivers (► Sect. 4.3). To mobilise stakeholders from different institutions, activating interviews can be conducted to see which way the wind is blowing and to produce interest in an active participation in workshops (L.I.S.T. 2011).

Our own results in the Upper Lusatian Landscape and the Ore Mountains showed that stakeholders and land users often do not decide on the basis of maximising profits, but also consider non-monetary values like traditions, attitudes, and even ethical values. They are often convinced that providing different ES for society is very important, even if they are unfamiliar with the concept of ES.

▪ Regulation of Energy Crop Cultivation

The cultivation of energy crops and woody biomass is mainly regulated by market prizes, incentives paid to farmers under the European Union Common Agricultural Policy and direct or indirect payments under the German renewable energy act (EEG 2008). It is therefore necessary to assess different steering instruments to see whether they can regulate energy crop production effectively and what impacts occur on ES.

It can be stated that only single ES are considered in laws and incentives in Germany, and they not as a whole. Often, no Safe Minimum Standards are defined. Laws often demand that 'deterioration has to be prevented' or 'good farming practices' have to be used. However, 'good farming practices' are more a mere code of conduct rather than safe minimum standards (Hafner 2010).

4.4.3 Application of Models of InVEST to Assess Ecosystem Services

M. Holfeld, M. Rosenberg

Models are representations of reality. They might be images, intellectual and linguistic constructs or mathematical formulas. The modelling of ecosystem services initially provides an abstract representation of ecosystems, of processes taking place and of potential changes. This is already covered by ecosystem models to a quite good extent. The challenge, however, is to incorporate the demand and benefit into the models.

In this respect, the following model approaches are currently of special relevance: *Integrated Valuation of Ecosystem Services and Tradeoffs* (InVEST, ► www.naturalcapitalproject.org), *Artificial Intelligence for Ecosystem Services* (ARIES, ► www.ariesonline.org), the BGS *ecosystem services model* (► www.bgs.ac.uk) and *Multi-scale Integrated Models of Ecosystem Services* (MIMES, ► www.uvm.edu). All these approaches aim to simplify reality so that the integrated relationships of ecosystem services can be considered.

In this section, the open source modelling approach InVEST will be introduced and experiences of its application for a case study will be discussed. According to the developers, InVEST is suitable to be used for an integrated assessment of ecosystem services at a local, regional or global scale. It has been used around the world in numerous local and national projects and studies, as well as in day-to-day decision-making processes (Daily et al. 2009; Nelson et al. 2009; Tallis and Polasky 2009; Bhagabati et al. 2012). Examples of its application include the Willamette Basin in Oregon, Oahu on Hawaii, British Columbia, California, Puget Sound in Washington State, the Eastern Arc Mountains of Tanzania, the upper Yangtze River Basin in China, Sumatra, the Amazon Basin and the Northern Andes in South America as well as Ecuador and Colombia. In the course of the realisation of the case studies, the focus is set on the identification and protection of important areas for biodiversity and ecosystem services, as well as on the demonstration of their relations.

Characterisation of the Model Approach of InVEST

InVEST was developed as a scenario tool to support decision-making in environmental planning processes. The basis of the evaluation of ecosystem services is ecological characteristics and methods of assessing economic values (Nelson et al. 2009; Tallis and Polasky 2009). InVEST is usable in combination with ArcGIS (ESRI), which provides the cartographic representation of the ecosystem services evaluation. Meanwhile, a cooperation with Idrisi is also under development (► www.clarklabs.org/about/Clark-Labs-Receives-Grant-from-Moore-Foundation.cfm).

The development and administration of the meta-model is realised by the *Natural Capital Project* with participation of several well-known American research institutions, as well as by *Nature Conservancy* and by the WWF (*World Wildlife Fund*) (Natural Capital Project 2012). Depending on the needs and expertise of the user different models with varying levels of complexity will be provided–from the simple analysis of existing relationships using a small amount of data up to a complex model, which includes different scenarios and feedback on the comprehensive analysis of ecosystem services (Nelson et al. 2008; Daily et al. 2009). However, currently only simplified procedures are offered, so that the models only require a small amount of input data.

Nevertheless, the open source model InVEST is already taking into account significant aspects of a two-dimensional modelling approach of ecosystem services. These include the spatial mapping and localisation of services and welfare effects in GIS, an integrated view of supply services, regulatory services as well as sociocultural services (TEEB 2009; Tallis et al. 2011). Furthermore, basic abiotic and biotic environmental parameters are incorporated into the assessment process. Thus, the quantification of ecosystem services within the individual models is not only based on the land use of the past, present and future, but incorporates additional parameters when necessary.

Based on the 14 models currently included, InVEST enables a biophysical and partly economic evaluation of a selection of ecosystem services of terrestrial as well as maritime systems. In ◘ Table 4.4 the seven terrestrial models for the description of services and products of land and freshwater are presented and assigned to corresponding classes of ecosystem services.

In addition to the final results of the individual models, partial results and intermediates are also taken into account. However, those partial results cannot be clearly assigned in every case to an eco-

Table 4.4 Terrestrial InVEST-models for assessment of ecosystem services (Tallis et al. 2011; date: May 2012)

InVEST-Modules	Ecosystem Services	Indicators, partial results and intermediates	► Sect. 3.2
Biodiversity	Habitat quality	– Habitat quality – Relative level of habitat degradation – Relative habitat rarity	R.11
Carbon storage and sequestration	Economic value of carbon sequestered	– Amount of carbon stored – Difference of carbon stored in future and current landscape – Volume and biomass of forest management	V.6; V.8; R.2; R.3
Reservoir hydropower production	Economic benefit of hydropower production	– Total water yield per sub-watershed – Mean water supply yield volume per sub-watershed – Total energy produced per watershed (in kWh)	V.12; R.5
Water purification: nutrient retention	Economic benefit of nutrient retention by filtration	– Total water yield per sub-watershed – Total amount of nutrient retained by each sub-watershed (in kg) – Mean amount of nutrient retained	R.5; R.6
Sediment retention	Avoiding costs of sedimentation (for dredging and water treatment)	– Total potential soil loss per sub-watershed – Mean sediment retained on each sub-watershed	R.7
Managed timber production	Net present economic value of timber production	– Volume and biomass of forest management	V.6; V.8
Crop pollination	Potential value of the pollinator supply for each agricultural land use to crop production	– Potential likely abundance of a pollinator species nesting in the landscape, given the availability of nesting sites there and of food – Relative farm value of crop production on each agricultural cell due to wild pollinators	R.10

system service as presented in ► Sect. 3.2. An assignment of models according to productive, regulatory or sociocultural ecosystem services or welfare effects will not occur. Nevertheless, each individual model and its background is explained briefly. A categorisation according to the welfare effect is not possible as some of the models do not describe a direct performance, product, or process of ecosystem services, but rather demonstrate risks–and, therefore, describe impacts on the functionality of an area at a certain land use (e.g. sediment trapping).

The programme language of all models listed is Python, which is also usable within ArcGIS. However, for calculations based on InVEST basically no knowledge of Python programming is needed, instead the usage of InVEST-models requires basic to intermediate skills in handling ArcGIS (Tallis et al. 2011). Furthermore, the computer system used has to meet certain requirements. For example, the regional and language settings need to be changed to 'English (USA)' in the system panel. This ensures that decimals are determined by a point, not a comma (as with German settings). Otherwise, incorrect results or even system crashes can be caused as the model scripts are unable to collect and process commas of the input parameters. Furthermore, a recent ArcGIS licence is required, while some models even require the ArcInfo licence level. In addition, installation and activation of the ArcGIS *Spatial Analyst* extension is required. Moreover, the model for the assessment of pollination as well as all models for assessing the maritime system require additional Python library extensions, such as *Numeric Python*, *Scientific Library for Python*, *Python for Windows* and *Matplotlib* as well as for ArcGIS 9.3, the *Geospatial Data Abstraction Library*.

Model InVEST

The scenario tool InVEST can be downloaded from the website of the *Natural Capital Project* (► www.naturalcapitalproject.org). The installation of the programme is very user-friendly as an entire folder structure with all scripts and training data will be unpacked–given the appropriate installation file is selected for downloading. New users of InVEST benefit from a structured data provision, as they can open the programme and test the models without a lot of prior skills or background knowledge. To apply InVEST to your own research the input data for each individual model needs to be adjusted to the specific study area according to the requirements for each model. Partly, some of these data can be found in open source databases of different state agencies or individual studies. For such data, the format needs to be adjusted in analogy to the demodata. This includes compliance with the original names of column headers and with the conventions for objects according to the instructions of the user manual, also taking into account general limitations of data management in geoinformatics. Furthermore, it needs to be considered that the computation time of the models depends on the resolution of the raster data at the beginning and at the end of the modelling process. Thus, in order to accelerate the calculation a lower spatial resolution (grid cell size) is recommended.

Continuous development of the individual models of InVEST aims to lead to a steady improvement in modelling. In this context, users need to consider the increasing demands on hardware and software. Currently, an ArcGIS 9.3 or 10 licence is required, because specific calculation algorithms of it are used in the models of InVEST.

Example of Use

While working on the project 'Landscape Saxony 2050' (► www.ioer.de/index.php?id=812) at the Leibniz Institute of Ecological Urban and Regional Development almost all terrestrial and one maritime model of InVEST were selected and applied to the study area–the district of Görlitz in Eastern Saxony, Germany. Those models include reservoir hydropower production, sediment retention, aesthetic quality, biodiversity–habitat quality and rarity, carbon storage and sequestration, managed timber production and crop pollination. When the simplest level of complexity in InVEST is used, most of these models are based on a matrix in which average performance parameters are assigned to the individual land use. The variables can represent both absolute values like stored carbon in tons per hectare, as well as relative values, with the highest value usually being defined as 1, while all other values are represented in their proportion to that. Depending on the programming of the individual models calculations are taking place at different levels of complexity. These calculations begin by adopting variables for land use as defined in the matrices (i.e. as in the fixation of carbon), and end with aggregated, buffered, overlaid calculations (as in biodiversity) or with neighbourhood analysis (as in aesthetics), where a decreasing influence is computed based on land use. Results are either relative values between 0 and 1, absolute values with indicators and/or economised assessments of the provided ecosystem services in the form of raster maps and tables.

In the following example, the biodiversity model and its calculation has been selected out of the mentioned InVEST models for assessments of ecosystem services, and will be processed for the district of Görlitz. This particular model has been chosen as it is characterised by high complexity, but also because of its variety of possibilities to integrate additional parameters in the calculation process, and, furthermore, because of the key significance of biodiversity as well as the possibility to represent a comprehensive topic in a highly simplified form.

Using the model biodiversity, two assessments can be carried out: habitat quality and the degree of exposure of habitat rarity. The latter describes the current decrease of the area of a habitat (in this case of land use) within a certain space compared to an earlier time. However, the actual risk or the consequences of habitat rarity are not determined or identified.

The selected area of investigation with an extent of approximately 2106 km^2 is located in the border area of Germany, the Republic of Poland and the

Attributes of LULC_value_habitat_and_sensitivity

OID	LULC	NAME	HABITAT	L_hway	L_froad	L_sroad	L_droad	L_lroad	L_urb	L_agra	L_rail
0	0	unknown	0	0	0	0	0	0	0	0	0
1	11	continuous urban fabric (high density)	0,1	0	0	0	0	0	0	0	0
2	12	continuous urban fabric (very high density)	0	0	0	0	0	0	0	0	0
3	13	discountinuous urban fabric	0,3	0,2	0,2	0,1	0	0	0	0	0,2
4	14	Industry and commercial units	0,1	0,1	0,1	0	0	0	0	0	0,1
5	15	Infrastructure	0	0	0	0	0	0	0	0	0
6	20	fill-up/pit	0,3	0,2	0,2	0	0	0	0	0	0,2
7	30	recreation	0,4	0,2	0,2	0,1	0	0	0,2	0	0,2
8	40	arable land and crop	0,4	0,2	0,2	0,1	0	0	0	0	0,2
9	41	agricultural crop land	0,2	0,2	0,2	0,1	0	0	0	0	0,2
10	42	arable crop	0,6	0,1	0,1	0	0	0	0	0	0,1
11	43	uncultivated land	0,4	0,1	0,1	0	0	0	0	0	0,1
12	50	pastures	0,8	0,3	0,3	0,2	0,1	0,1	0,5	0,2	0,3
13	60	forest area and woodland	0,9	0,1	0,1	0	0	0	0,2	0,2	0,1
14	61	broad-leaved forest	0,9	0,1	0,1	0	0	0	0,2	0,1	0,1
15	62	coniferous forest	0,8	0,1	0,1	0	0	0	0,3	0,1	0,1
16	63	mixed forest	0,9	0,1	0,1	0	0	0	0,2	0,1	0,1
17	64	clear cutting and reafforestation zone	0,7	0,1	0,1	0	0	0	0,3	0,2	0,1
18	65	transitional woodland shrub	0,8	0,1	0,1	0	0	0	0,2	0,1	0,1
19	70	water bodies	0,9	0,6	0,5	0,4	0,3	0,2	0,5	0,9	0,5
20	81	open construction sites	0	0	0	0	0	0	0	0	0
21	82	sparsely vegetated areas	0,7	0,2	0,2	0,1	0	0	0,2	0,1	0,2
22	91	moors and wetlands	0,9	0,5	0,4	0,3	0,2	0,1	0,2	0,5	0,4
23	92	arid environment	0,9	0,5	0,4	0,3	0,2	0,1	0,2	0,1	0,4
24	99	do not exist in 1992 or 2005	0	0	0	0	0	0	0	0	0

Record: 1 Show: All Selected Records (0 out of 25 Selected) Options

LULC	code of land use and land cover
HABITAT	basic habitat quality
L_hway	relative sensitivity to the threat of highway
L_froad	relative sensitivity to the threat of federal roads
L_sroad	relative sensitivity to the threat of state roads
L_droad	relative sensitivity to the threat of district roads
L_lroad	relative sensitivity to the threat of local roads
L_urb	relative sensitivity to the threat of urban areas
L_agra	relative sensitivity to the threat of agricultural areas
L_rail	relative sensitivity to the threat of railway lines

Fig. 4.15 Land-use classes, habitat value and sensitivity of habitat types to each threat (screenshot of the example of use of InVEST in the district of Görlitz)

Czech Republic. The district has a wide variety of habitats, which reach from lowlands to highlands. Open brown coal mining and recultivation have brought large-scale changes. Noteworthy are cultural and historical particularities, such as folk architecture (Umgebindehäuser) and the culture of the Sorbs, a Slavic ethnic group. Rare species such as otters, cranes, eagles and more recently even wolves, find suitable habitats here. In addition, the region is both demographically and economically affected by a strong change (► Sect. 4.3).

By selecting the chosen model from the toolbox of InVEST, a dialogue box opens. There, the input data and the folders for the results need to be defined. Thus, the existing paths of the sample data provided by InVEST need to be replaced with actual data of the chosen study area. The input data for the delineation of habitats are based on maps of land use and land cover, for which the habitat types and land use mapping (BTLNK) of the Free State of Saxony from 1992 and 2005 are used. These maps reflect a variety of land use classes. In order to simplify the modelling, the classes are all combined into one aggregated BTLNK mapping with 25 classes (BTLNK_25). Eventually, their contents are provided in a grid with a resolution of 20 m.

In addition, a relative habitat value (Habitat) for each land-use class needs to be defined within a spreadsheet in relation to the other classes (Fig. 4.15). Those values range from 0 (unsuitable) to 1 (perfectly suitable as Habitat). In order to define the habitat values for this case study, non-species-specific information according to Bastian and Schreiber (1999) are used. These are parameters that do not document habitat qualities of specific species or groups of species (species of open land, forest or of aquatic and wetland sites), but assign general assessments to individual habitat types with regard to their importance for species and area protection.

In addition to determining the general habitat quality of each land-use, threats that may affect this habitat quality are also determined such as highways, federal roads, state roads, district roads and local roads as well as railway lines, which were extracted from the Digital Landscape Model (ATKIS Base-DLM) on a scale of 1:25,000 for both years of the investigation and converted into a raster format. The areal threats of additionally considered urban and agricultural areas are based on the coverage of BTLNK_25.

Thus, the dimension of degradation, which is solely caused by the respective sensitivity of habitat types to each threat, has been defined between 0 and 1 (◘ Fig. 4.15) based on an evaluation of the influence of the mentioned threats on the habitat quality of the identified land-use classes. The value 1 presents the highest impact, the value 0 no or imperceptible degradation. Thus, a land use that is not displayed as Habitat (Habitat = 0) has no coefficient of degradation by threat.

Finally, the threats have been characterised based on their relative importance or weight and impact across space–range in kilometers and whether the impact of the threat decreases linearly or exponentially across space. A value of 1 is a linear decline in impact and 0 an exponential decline. The maximum range is based on the findings of Baier (2000); the remaining parameters were defined by the authors.

After completing and confirming the input data, the calculation is started. In this process, the individual steps are recorded in a separate process window. Based on the information provided by the habitat values of individual land-use classes from ◘ Fig. 4.15, a reclassification of land-use maps is taking place (H_j as general habitat quality). Simultaneously, the area sizes of individual land-use classes in the study area for the base year 1992 are compared to 2005 (the degree of hazard habitat rarity). For this application, the Eq. 4.1 is used.

$$R_j = 1 - \frac{N_j}{N_{j\,base\ year}} \tag{4.1}$$

R_j represents the degree of change of the individual land uses in the study area compared to the base year, N_j defines the area size of individual land uses in the base year and the current year. Is $N_j \geq N_{j\ base\ year}$, so $R_j \leq 0$ and the result is $R_j = 0$, otherwise there is a change in land use and R_j is greater than 0. The output of the calculation results in a grid, which values of R_j are each projected onto all present land-use areas (i.e. for the second point of time). A partial result of this calculation step is a map representing the area development of land-use classes (a so-called exposure of change in use) between a base year (here 1992) and a later point in time (in this case study 2005).

Taking into account the sensitivities of each present land use (◘ Fig. 4.15), the impact of each grid cell on its surrounding grid cells is determined within a second step by using the maximum range and impact across space for each threat and each grid cell. The individual effects of each threat on the grid cells are then summed up, which may show the impact of a threat on habitat quality. Considering the weights of the individual threats (◘ Fig. 4.15) the impacts of the threats on habitat quality are aggregated. The result of the summed degradation (D_{xj} as degradation of habitats) can be represented and compared in a raster map for the respective reference year.

In the last step, according to Eq. 4.2, the specific habitat quality (Q_{zj}) is calculated as an index by merging the aggregated degradation D_{xj} (including the half-saturation value k) with the reclassified land-use classes represented as habitat quality values (H_j). The half-saturation value is determined as half of the highest grid cell degradation value in the study area. The exponent z corresponds to the value 2.5.

$$Q_{zj} = H_j \left(1 - \left(\frac{D_{zj}^z}{D_{zj}^z + k^z} \right) \right) \tag{4.2}$$

As a first result of the modelling of the biodiversity by InVEST, the risk level of the habitats (in this case of the land-use classes) is presented in terms of their area sizes. In the context of the case study in the district of Görlitz, it was found that between 1992 and 2005 in particular the following land-use categories were affected by a strong reduction of their extent in proportion to their respective total area in the base year: reforested areas, fallow ground, mining areas as well as areas for transport and infrastructure. In addition, a decrease of the extent of meadows and pastures was discovered.

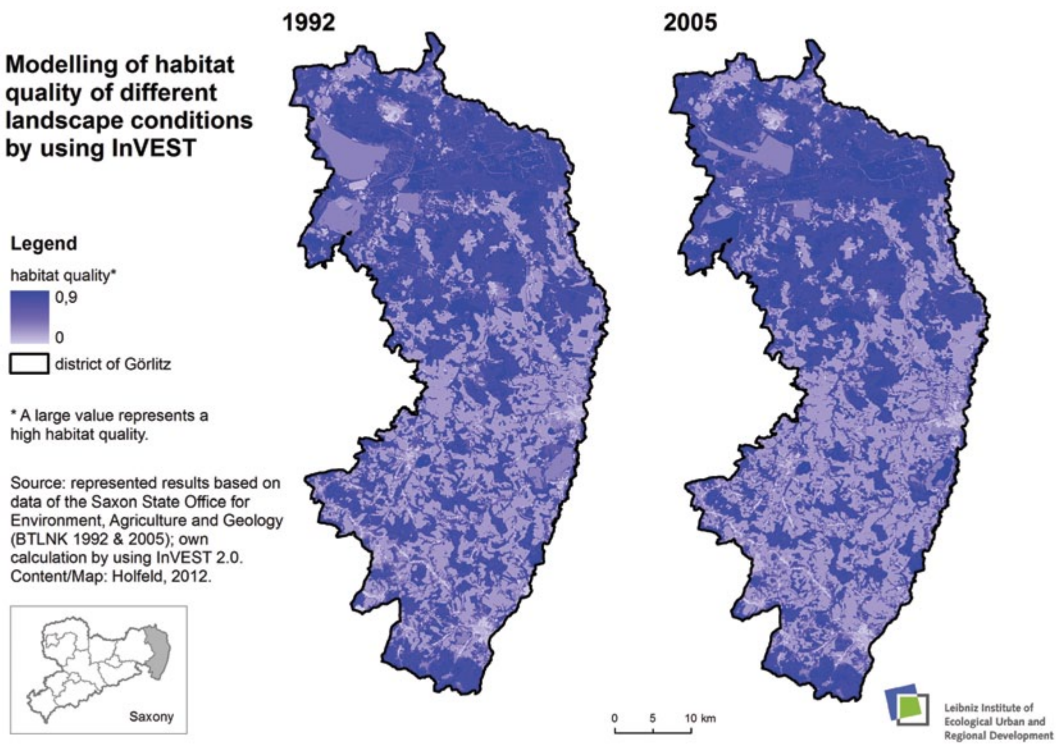


Fig. 4.16 Results of the assessment of InVEST model habitat quality in the diestrict of Görlitz for 1992 (*left*) and 2005 (*right*)

Next, the aggregated degradation as impact of threats is presented for the study area. Thus, the highest negative influences are detected at the border of urban areas and along main traffic infrastructure (highways and federal roads). The intermediate areas show no or hardly any perceptible threats. The same is found for the urban areas of the study area, which cannot be affected by any threat as they have not been assigned to the habitat function in the model.

Based on the result of the degradation, and taking into account the given habitat value of each land use, the *specific habitat quality* of each grid cell is mapped (Fig. 4.16). Thus, the highest habitat values are found mainly in the wooded north compared to the strongly agricultural influenced south of the district.

The lowest habitat values are found in the large urban areas as well as in a linear manner in the settlements along the main roads. A comparison between the base year 1992 and the year 2005 based on bluegray-scale values in the map (Fig. 4.16, left versus right) is hardly possible and also not possible as they rely on different databases. In order to compare the habitat quality of both points in time, the sum over all grid cells of a year must be calculated. The model completes this calculation automatically and writes its result into a log file, in which all input parameters are logged as well. Thus, the summed quality for 5,304,420 grid cells in the base year 1992 is 2,857,030. The total value for 2005 is 2,884,710. The habitat quality as an overall value for the district of Görlitz has improved slightly between the two assessment years, although spatially differentiated large-scale degradation in habitat quality is determined, for example, due to changes in land use. However, their scope has been fully compensated by other sub-areas within the study area. Many steps are similar to the approaches of conventional landscape planning.

Discussion

The results of the analysis of biodiversity with InVEST offer a simplified representation of the real habitat quality in the study area. Using the input data of Bastian and Schreiber (1999) average habitat values depending on the habitat type have been used for the district of Görlitz.

As an intermediate the degradation of habitats (degree of threats of habitats) was calculated, which show the downgrading of the habitat quality due to selected infrastructural threats. Within the modelling, it is basically assumed that the impacts of individual threats are adding up. In reality, however, their effect might be significantly higher (Tallis et al. 2011). Furthermore, it should be mentioned that the result is only one example out of many concerning habitat qualities, depending on the selection and consideration of individual threats as well as the considered habitats or species (Nelson et al. 2008, 2009).

Due to the manner of spatial location, the examination of habitat rarity seems hardly useful. However, the consideration of the change in land use within the biodiversity model is to be regarded as absolutely reasonable. But for this, a simple transition matrix between the different land uses would be sufficient. The current form of presentation is to be considered as very critical. Land-use types, which experience no absolute reduction or absolute increase in areal extent for the entire study area, are not assigned any degree of hazard. This includes land uses that are subject to areal change in land use in one part of the study area being fully compensated in another part.

As shown by the example of the biodiversity model, due to the low complexity of its individual models InVEST is easy to operate–as long as the user has at least basic working knowledge of geographic information systems. Through the results, some simplified relationships between land use and biodiversity or ecosystem services can be discovered (Polasky et al. 2008; Daily et al. 2009; Nelson et al. 2009; Tallis and Polasky 2009). Here the focus is more directed at the ecosystem services considering supply and demand than on biophysical processes. According to the current state of development of the models, an economic value can be assigned to an individual basis for a produced unit or for a specific process, which is used as a valuation basis for the study area. Thus, it is possible to evaluate the ecosystem services appropriately despite spatially separated locations for the demand and the provision of a service. However, the demand oriented approach is currently not available for all models contained in InVEST. Likewise, it needs to be considered if, for example, there is no water reservoir (modul: hydropower production), no service of energy can be provided.

The modelling with graded levels of complexity based existing approaches for specific modelling of landscape functions–such as SWAT or USLE (Tallis and Polasky 2009), allows to define the choice of the model complexity on the availability of data or on the user group. While simple models contribute to a better understanding of the relationships of the ecosystem services, the more complex models are intended to estimate the precisely measured services. Along with the desired development of the models, including further parameters, the demand for providing better data as well as the operability of InVEST increases (Tallis and Polasky 2009). Therefore, the provision of data and data sources in a central database would be desirable for different study areas in order to minimise the research work.

Due to the relevance to ArcGIS, results can be represented spatially in different scales (Daily et al. 2009). In order to do so it is crucial to have sufficient specific and differentiated information as input data for a certain study area. Furthermore, it has to be noted that the size of the study area depends on the considered ecosystem services (Tallis and Polasky 2009). For example, water-based services or pollination are of greater importance at a local scale (► Sect. 3.3) while climate-regulating processes require a global scale.

In addition to cartographic outcomes, results can be exported in a tabular form. The present results, however, are not suitable for professional use, such as in the development of detailed water and landscape plans or environmental audits as many functions and interactions are still negligible (Tallis et al. 2011). Similarly, the balance of costs and benefits of different models of InVEST is controversial even among developers, and certain ecosystem services, such as biodiversity (habitat quality), cannot be represented economically. The monetisation is furthermore criticised, because its assessment depends on spatial, temporal and sociocultural aspects that within InVEST cannot yet be considered as differentiated as their findings (Tallis and Polasky 2009). In general, average parameters are used for each evaluation of ecosystem services, which limit the validity of the result depending on the aspect to be researched and the scale of the study area.

With InVEST, the *Natural Capital Project* provides an evaluation process with great potential, even though it currently still has certain modelling weaknesses. One positive aspect is that InVEST is offered as an open source model, although its algorithms are sometimes highly complex. The open approach also allows less-experienced programmers to understand the calculation steps. The open development of the individual models ensures that both experts and laymen may submit suggestions to improve the modelling. At the same time, providing InVEST free of charge is supporting the rapid spread and development. The disadvantage of the continuous development of the models is that developers are always focussing on the latest versions of ESRI ArcGIS in order to incorporate the latest features from ArcGIS. Thus, increased system requirements of hardware are needed, but also the latest ArcGIS licences.

In conclusion, despite the identified criticism and existing weaknesses, it can be summarised that InVEST is a remarkable method to evaluate small as well as large-scale ecosystem services and to compare different regions, especially as the effort to define the input data is still small and the use of the individual models is relatively easy. However, the modelling procedures and results always need to be examined critically in order to avoid false conclusions.

Conclusion

Models provide exceptional opportunities to analyse and evaluate ecosystem services. With them, the landscape change that has already taken place as well as scenarios of future developments can be subject of an assessment. Therefore, decision-makers as well as the affected population can identify relationships and interactions of their action. Thus, the knowledge and communication of ecosystem services is strengthened. The various existing approaches to evaluate ecosystem services focus on different questions of content, spatial and/or temporal nature and still show significant deficits (Nelson et al. 2009).

With InVEST, an instrument is currently being developed, which is close to achieving the existing requirements for an evaluation of ecosystem services. In contrast to Burkhard et al. (2009) and Koschke et al. (2012), who already allow a holistic view of the ecosystem services within demarcated areas, the InVEST approach is also observing other biotic and abiotic parameters in addition to land use. However, the integration of those parameters is still at the beginning and needs further development in order to allow differentiated analyses of the ecosystem services (Nelson et al. 2009). Besides the development of computational algorithms within the models, well structured access to quantifiable data needs to be build up as the data availability is still quite limited. Simultaneously, methods are required, which allow the often individually evaluated ecosystem services to be compared and weighed up against other and to communicate their results (Holfeld et al. 2012).

4.5 Communicating ES

K. Anders

4.5.1 The Importance of Communication

In recent years, an entire new field of research, that of sustainability communication, has emerged which investigates the possibilities of communication regarding environmental issues. It encompasses a broad gradient of the issues which have been handled in various ways in various disciplines, in terms of their theoretical foundations, methodological approaches, and practical areas of application (Michelsen and Godemann 2005). In the present chapter, we will be able to examine only a few systemic decisions. The basic fact is that without appropriate communication, ecological issues will have no chance of validation in society. Only by way of communication can the relevant information in the social systems even be selected, informed and understood. Communication is therefore the key process of societal autopoiesis for social systems, i.e. it is through communication that they produce and reproduce themselves (based on Luhmann, this range of issues has e.g. been precisely circumscribed by Schack 2004).

However, how this process actually proceeds can be influenced only to a limited degree (Ziemann 2005). The feasibility of communication is widely overestimated; the definition of communication is often mechanically reduced to a more or less complicated relationship between the broadcasters in the receiver. The German phrase commonly used today, 'I'm communicating this or that,' erroneously even suggests the possibility of engaging in communication with no counterpart. However, the difficulties involved in being in control of the communications process do not imply that it is fundamentally unshapeable. Rather, one's own role as a participant in that process can certainly provide opportunities to put forward arguments, positions and assessments. In order to identify free spaces for the societal validation of ES for a number of very different fields of application–from advertising to discourse–i.e. if we are to assume that communications, in spite of its internal dynamics, is a shapeable process (Schack 2004), we will first have to take a more detailed look at the intentions connected with the term 'ecosystem services.'

4.5.2 'Ecosystem Services' as an Umbrella Term for Communicative Intent

The concept of ecosystem services is based on a very large number of different properties of ecosystems and landscapes. The initially very summary systematics of supply, regulation and sociocultural ES (► Sect. 3.2) does not follow any scientific–analytical or systematic–necessity; rather, it is designed to ensure that asymmetric processes and perspectives attained public recognition within the context of a certain topicality. A similar strategy was used several years earlier around the concept of biological diversity, in which genetic diversity, species diversity and landscape diversity were brought together without the relationship between these various levels having been clearly defined. Wilson and Piper (2010) characterised the ES use of language 'as a route to better understanding their importance and also of improving their protection.'

As a result, the term 'services' has been variously used, and the term broadly stretched. The authors of the Millennium Ecosystem Assessment admit as much: 'The condition of each category is evaluated in somewhat different ways, although in general a full assessment of any service requires consideration of stocks, flows and resilience of the service' (MEA 2005a, p. 29). While the term in such areas as supply functions has generally remained relatively closely oriented towards the usual use of the language about a service (for people; the implicit anthropomorphism is justified pragmatically), cultural services must be located more in the network of interrelationships between humankind, nature and the landscape (MEA 2005a; Freese and Anders 2010). Regulatory services, on the other hand, involve first of all the self-organisational capacity of an ecosystem; the advantages for people are thus indirect.

This leads to a difficulty of operationalisation: various processes incorporated under ES are to be found in particular landscapes and very different qualities, which resulted a problem of evaluation criteria. There are ES which can basically be provided in unlimited quantity (e.g. soil formation), while others undoubtedly violate the principles of sustainability, if their activation is not kept within limits. Often, these services are rendered at the cost of others (Trade-offs; Stallmann 2011; ► Sect. 3.1.2), resulting in requirements for a balancing of interests which have to date remained methodologically unresolved as long as the concept of planning contexts is to be used. This series of imprecisions recalls Luhmann's assessment of ecological communications in the sciences (◘ Fig. 4.17):

> "The carelessness in the choice of words and the lack of awareness of theory-related decisions of great consequence are among the most notable characteristics of this literature–as if care for the environment could justify carelessness in the speech concerning it. (Luhmann 2008, p. 8)"

Whether ecosystem services have indeed become part of a discursive framework or pattern of inter-

Fig. 4.17 At the meeting of the German section of the International Association for Landscape Ecology (IALE-D) in 2010 in Nürtingen, the artists Christiane Wartenberg and Robert Lenz presented a shelf with two kinds of honey. One set of jars contained real bees' honey, labeled with the exact information regarding the place of production and also regarding the landscape development issues connected with it. Next to it was 'artificial honey'–jars with drypoint etchings of the most common terms in environmental research, from 'acceptance' to 'invasive art.' What was kept apart in this art exhibition–natural space, use, and scientific research–should also be separated more carefully in the debate over 'ecosystem services.' © Kenneth Anders

pretation, as described, e.g. by Brand and Jochum (2000), in other words, whether for example the expectation that aspects of the protection of nature and resources might better be validated has indeed been fulfilled, is a question that deserves closer examination.

The attractiveness of the concept within the environmental sciences, the business and financial world and also among policy-makers, would any case appear to be still on the increase, which should, however, not be confused with greater validation for the processes thus described. It is certainly possible, that the term 'ecosystem services' will become established without this fact having any consequences for society's relationship with the environment.

4.5.3 Government and the Market Instead of Communications?

The term communications itself is not a factor in the Millennium Ecosystem Assessment. Rather, the scientific community sees itself as a communicating actor in this context; its target system is the policy-making establishment. While the executive summary of the study for 'decision-makers' does raise the issue of participation and transparency as 'ecosystem-services'-related demands directed towards policy-makers, this is framed merely in terms of the requirements of administration, not as the constituent element of societal communications (MEA 2005b). Even a theoretically rooted concept of 'the public' is something which is not to be found in the debate around ES. Once in a while, there are merely indications about the use of publicly available information (Ruhl et al. 2007), which do correspond to basic demands for transparency in such areas as planning processes. The reason for this systematic blindness may be found in economic calculation: Unlike the 'tragedy of the commons,' the tragedy of ecosystem services is seen not as a problem of overconsumption, but rather of underproduction (Ruhl et al. 2007).

As a result, it would appear that the societal appreciation for ES will become tangible only when the market conditions for the same have been created. Communication is thus not excluded; rather, it is assumed that for ecological problems, the tool is available: the successfully established, symbolically generalised communications medium known as 'money'. That is not the place to pass judgment on the prospects for the success of this idea. However, the identification recognition of ecological processes and services, and the emergence of corresponding markets, can only be achieved through communication, in other words, the medium money cannot be transferred to ecological plans and actions merely on the basis of an assertion to that effect. The authors of the Millennium Ecosystem Assessment evidently assume that it is only necessary to convince policy-makers of the plausibility of their arguments, in order to create the necessary laws and regulations. Büscher and Japp (2010) pointed out in this context "that in the current public debates over problem solutions with respect to the 'ecological crisis', sociological arguments play no role. The salvation of the world is, as it were, to be carried out with no concept whatever of 'society'".

4.5.4 Communications Efforts as an Approach to the Shaping of Environmental Sciences

In order to be able to arrive at a statement in spite of these yet uncertain questions, let us use the term 'communications efforts' in order to do justice to the reasonable desire for the shaping of communications. 'Communications efforts' means the intent to effectualise scientific knowledge with respect to the significance of ecosystems for people outside the scientific system. Here, a changed self-consciousness is palpable in environmental research, where communications efforts have been massively enhanced in recent years. Today, we often expect that, given a general feeling of insecurity, environmental scientists should not so much bring particular ascertainments into the discourse, but should rather enter into an exchange with policy-makers regarding the weighing of ecosystemic contexts, and should assume a vanguard position in that respect. In this context, the term 'pro-active' has become fashionable.

An author such as Luhmann would doubt that this new awareness is based on any realistic analysis of the possibilities of the scientific community, for '… other functional systems must assume the task of sorting out what is useful and what is useless' (Luhmann 2008, p. 108). Precisely this step towards action is usually taken only rarely (Bechmann and Stehr 2004), which is in turn no coincidence, for research after all, due to the construct of 'consensual knowledge' (Bechmann and Stehr 2004, p. 30), is always in danger of weakening its own position as a systemic element by giving up its own medium, according to which information is selected according to the criterion of true/false. In other words, the core business of the scientific community is the question: 'Is this statement true, or is it not true?' Once one abandons the realm of this core business, one is treading on slippery ground. In order to survive in such a situation, scientists ultimately have to assume two roles: one as communicators in the sustainability discourse, and another as communicators within the scientific system. One good example is the Stern Report, *The Economics of Climate Change* (Stern 2007), in which, especially with regard to the effects of disturbed climate-regulatory functions, a political agenda ranging from the trade in emissions rights through a reduction in deforestation to targeted climate adaptation has been developed from out of the scientific community, in spite of a high degree of uncertainty.

Kuckartz and Schack (2002) pointed out that the goal of environmental communication encompasses a broad range of gradients which is not sufficiently reflected: the attempt to achieve acceptance for laws or to promote ecological products, involves very different consequences than the desired changes in behaviour, or even the claim to enable people to orient themselves amongst the complex issues of ecological action. In one case, public relations and advertising predominate; in the second, by contrast, education. This diversity also applies to communications regarding ES. In the following, we will therefore discuss several more or less established forms of scientific or planning related communications efforts with regard to their suitability to generate societal responses for certain ES.

▪ The Classical Transfer of Knowledge

The transfer of knowledge should build an elementary bridge between the scientific community and other systems by 'publishing'–literally: 'making public'–the results of research. In other words, a communication is to be made available beyond the bounds of professional circles. In this area too, the efforts of environmental research and planning have been greatly enhanced in recent years. The goal of eliminating knowledge gaps (e.g. Schmidt et al. 2010) is appropriate, since the preparation and accessibility of sufficient information ultimately permits communication–even if such activity is in and of itself not communication. It is for precisely this reason that totalitarian systems denied the release of information, since they will be unable to control the results in the public communicative sphere. Beyond the concept of public participation in planning (Schmidt et al. 2010), it is according to the participatory intent of the authors necessary to ascertain that public opinion comes into being in the first place only through communication, and that this is precisely what the task of planning processes consist of. Communication efforts are realised through the fact of the accessibility of information; hence, it is demonstrated that certain functions of ecosystems are indispensable for human beings, or that the loss of the same would affect the general interest. Here, environmental scientists can certainly assume an active role without departing from their home turf. This includes the description of ecosystemic contexts such as soil formation, water retention or important nutrient chains (i.e. regulatory services), and also knowledge on land and water use (supply services), or descriptions of the wealth of interaction between people in the landscape (sociocultural services).

In all these cases of knowledge transfer, what is needed is not so much professional marketing strategies and campaigns, as clear statements and a generally comprehensible language based in precisely this kind of clarity. There are enough historical models for this, in which environmental scientists convey information directly, and, for good reason, do without any aggregated preparation of the same by means of 'communications profiles.' Knowledge transfer has traditionally been carried out with a high level of quality under the *Leitmotiv* of 'welfare effects' (e.g. Albert 1932; Hornsmann 1958; Altrogge 1986). The discontent around this classical role of science is often described as disillusionment regarding its societal effect. There are two variants of this; while for example, Barkmann and Schröder (2011) target the lack of the reception of scientific knowledge in society, many other authors assume that environmental knowledge is basically sufficient, but that there is a lack of corresponding behaviour resulting from it (e.g. Wehrspaun and Schoembs 2002). Indeed, the attitude of classical knowledge transfer does not ensure that the knowledge provided will also be societally used. On the other hand, the question is justified, in terms of the concept explained at the outset: To what extent is it even possible to force such an assurance?

▪ The Transfer of Knowledge and Transdisciplinary Contexts

Beyond the 'classical' domain of knowledge transfer will–in the context of transdisciplinarity, i.e. with regard to the methods used and even with regard to the concrete research questions of a partially open process–conceptual deficits once again dominate the picture (a systematisation approach of Jenssen and Anders 2010). While knowledge transfer is correctly criticised with regard to obsolete models of the relationship between the broadcaster and the receiver (Karmanski et al. 2002), there is a lack of dialogic work methods in most research processes in which actors determinant for the landscape can weigh the relevance of the research knowledge produced and bring their own forms of knowledge–hence also their relationship to various ES–into play. Under the conditions of transdisciplinarity, knowledge transfer thus becomes an active communication task, i.e. scientists have to accept the existing heterogeneity

Totholz
im Wald ist
Mist,
die reine
Parasitenzucht.

Der Kampf
um die Rohstoffe
hat begonnen.

Naturnahe artenreiche
Wälder – wenn es die
nicht mehr gibt,
vergessen wir, wie
der Wald aussieht
und nehmen
Kiefernmonokulturen
auch als Wald hin.

Die
Kiefer
ist
der
märkische Brotbaum.

Fig. 4.18 By means of just four positions on forest development, we can already gain a hint of the contradictions one encounters with respect to ES, if one wishes to communicate about them. In the Schorfheide-Chorin Landscape Workshop, held in the state of Brandenburg between 2006 and 2009 as part of the BMBF collaborative research project Sustainable Development of Forest Landscapes in the North German Plain (NEWAL-NET), there were over 100 such positions. Much could be gained by bringing some order into this diversity in order to create space to help enunciate aspects hitherto ignored © Kenneth Anders

of knowledge, and subject their own work to the resulting validity conflicts (Fig. 4.18). For reasons of quality, too, debates will become necessary, for where representatives of various disciplines and areas of practice collide, it is difficult to manage the professional standards introduced, so that valid knowledge can only be selected by means of intensive and critical discussion. With regard to ES, this means that those contradictions are invisible which emerge from the fact that landscapes are used, enjoyed and protected simultaneously. Environmental sciences can therefore not themselves per se assume the role of the advocate of various ES. The appellative stance of the Millennium Ecosystem Assessment proves ineffective in the face of the reality of such processes. Rather, environmental scientists need to clearly defined their role in communications processes, i.e. either withdraw to the relatively passive position of the 'classical scientist' (and add to that the internal dynamics of communications), or else subject themselves to the contradictions that in fact emerged from the social, economic and ecological dimensions of sustainability–in the landscape and elsewhere. The latter occurs only rarely, and is the result of an understanding that posits the knowledge is only monopolised within the scientific communities, and that nonscientific perspectives cannot claim any knowledge-related status, but are only described in terms of identity, habit, individual experience, interest or sensitivity. What then remains of communication is understood as a means for generating acceptance of consensus (critically assessed by Adomßent 2004), which again moves closer to the mechanistic understanding described at the outset.

Social Marketing and Considering Lifestyles with Respect to Consumer Behaviour

One approach common in Germany for raising societal awareness of sustainability issues is social marketing (e.g. Buba and Globisch 2009). The methods developed here can also be used for various ES. For example, their recognition for the area of agriculture could occur by seeing not farmers, but rather the consumers themselves, as the perpetrators of reduced biological and landscape diversity (Adomßent 2004)–at least as long as the farmers lack any possibility of financing practices for the preservation of forms of diversity on the market. Diversity is thus seen as a product to be created, and no longer as an issue existing and endangered; in that way, it can become an object of marketing.

Compared with social-scientific analyses of environmentally relevant consumer behaviours and the societal complexity upon which they are based (e.g. Brand et al. 2001), social marketing constitutes a narrowing of the perspective, with the goal of linking social-scientific research with business concepts of customer acquisition in order to ultimately effect behaviour change. This too is accompanied by a changed self-awareness on the part of the scientific community–away from critical analysis and towards 'change management' (Buba et al. 2009). First of all, social groups with certain value patterns, consumer habits and some culturally determined characteristics are identified, using a process similar to that of 'sinus-Milieus' (everyday-life worlds; cf. e.g. Theßenvitz 2009). Subsequently, the identifications obtained are used to construct target groups which are then to be won to the intended goals by means of adapted media codes; in common parlance, one might say, 'if you want to reach people, you have to go to where they're at.' This apparently simple truth becomes a distortion if one realises that communication is a process in which all participants are moving, and no one is waiting 'at' anywhere.

From the point of departure of lifestyle research, Lange (2005) described social marketing as a modest, and hence realistic, horizon of expectations, by means of which consumer behaviour could be influenced; a thorough examination of the range of possibilities available to consumerism is provided e.g. by Bilharz (2009). However, even Lange has doubts about the expectation that such consumption patterns could be permanently rooted by means of the targeted influencing of lifestyles. For lifestyles can neither be politically controlled, nor is it possible to constructively use distinction effects, e.g. for the role of eco-pioneers. Social distinction is part of social dynamics, and therefore contributes just as much to the erosion of cultural patterns as it does to their formulation. The weak correlation of lifestyle and action moreover points to the limited possibilities in our society to even practice sustainable consumer habits at all, so that the ball is now in the court of the structural-policy decision-makers. Kuckartz and Schack (2002) have confirmed empirically that the actors in environmental communication no longer even see changes in attitude and consciousness as a task to be addressed. In view of the various ES, this situation is becoming ever more acute, since not all processes compiled under its heading can be affected directly by individual consumer behaviour. Moreover, since a major share of our actions result not from lifestyle-related patterns, but rather from overall societal ones, the decision regarding the use of certain ES–especially regulatory services–can under no circumstances be left to the free market, but rather must be regulated by law (Bilharz 2009). For example, soil protection can vary obviously be better provided by legislation than by a market for intact soils.

In this respect, social marketing, too, deserves to be handled with greater care with respect to its expected effects and to the suitable fields for its application than is currently the case. The representatives of this school of thought emphasised that in addition to a designing of social groups as the object of marketing, they are expressly working towards the self-determined assumption of responsibility by these groups (Buba and Globisch 2009). However, it is doubtful that the tautology of conventional marketing can be broken by the awakening and satisfaction of needs, for the selected information and its preparations already anticipate the principles of power and validity established in the respective lifestyle circles–precisely what we have to thank for the lack of sustainability in the practice of our lives. It is conceivable that representatives of 'Lifestyle of Health and Sustainability' (LoHaS), or a 'consumer materialist' might be motivated by social marketing to make a certain decision with respect to items of purchase; however, the expectation that representatives of these target groups will as a result change their attitudes simply because we have tried to speak to them in their language, is misplaced, since just that avoids calling into question the guiding ideas and mythologies of the hitherto dominant institutional practices (Brand 2005, p. 153). Moreover, the fact is that the actors participating in communication ultimately are always open in terms of their decision-making (Ziemann 2005), and also, communication necessarily causes changes in one's own perception, as a result of which the scientists involved themselves emerge from the process with modified perspectives. In other words: if one wants to promote communication while at the same time excluding its internal dynamics, we will fail to communicate.

- **Campaigns**

In this context, efforts to generate public validation of ES by means of campaigns are conceivable. Here, the conceptual lack of clarity of the term is initially an obstacle. As Lisowski (2006) has demonstrated, at least in the European context, the linear sequence of planning, strategy and campaigns as a way of achieving democratic influence is rarely encountered; rather, campaigns develop 'evolutionarily,' along existing financial and professional spaces. Hence, certain aspects may be successful, while others fade into the background. The precondition is the existence of organisations, which represent a certain interest for the public. Their practice is also known in the area of environmental communications. Campaigns for the establishment of wilderness areas, for the preservation of endangered species and habitats, for the protection of certain landscape types, for food produced under conditions respecting the ecosystems, etc. are an everyday occurrence. They may affect decisions and help promote societal developments, as in the Stand-By Campaign (Schack 2004). Finally, Frankel (1998) demonstrated a 'greening of communications' for industrial advertising. However, it is precisely the term ES that shows us clearly that while advertising refers effectively to the respective organisations or companies that control certain landscape processes, it hardly refers at all to the ecosystems themselves (cf. the WWF Tiger campaign, described in Conta Gromberg 2006). In this respect, this form of communication suffers from an authenticity problem, since suspicion regarding motives always arises (Japp 2010). Moreover, organisations with conflicting purposes are free to promote their own respective campaigns, in which ultimately different environmental goals are pursued and addressed. Since not all functions and processes in the utilised landscape are per se mutually noncontradictory, campaigns may certainly be a possible tool for highlighting ES, but they are an unlikely tool for use in planning processes–contradictions are not considered campaign-capable.

- **Education for Sustainable Development and Education for Landscape Policy**

The goal of education for sustainable development, a transgenerational, self-organising debate, and personal skills in addressing the issue of sustainability, would appear to be close to the intent of the concept of ES, and even to offer an adequate solution to the above-described asymmetry of subsumed functions: Placing concepts in relationship with one another, permitting diversity of perspective, and acting responsibly constitute the key points within which adapted and adequately contextualised accesses to this issue could be created. What is meant here is not education for sustainable development as an 'advertisement for sustainability' (Siemer 2007), as a sub-function of social marketing, or as self-praise for environmental policy, but rather as communication. However, that would require that the autopoietic process in education itself be promoted, in other words, that its results not be prejudiced. Yet it is precisely this precept that is violated by many works purporting to promote 'education for sustainable development', instead, they rely on old concept of environmental education, albeit in new garb. For example, role-playing in which children basically provide a 'constructive solution' to a conflict have nothing to do with the purpose of the concept as described here–to promote open learning processes. The frequent restriction of the approach to questions of consumerism, too, ultimately does not result in a satisfactory proximity to the ecological aspects of the service involved. Communication of ES through education for sustainable development thus does not automatically lead to success, but rather depends on the concrete formulation of the programme. It may even cause confusion and frustration, if the individual scopes for action ultimately remain schematic which has, in personal experience, often proven to be the case.

Such approaches suffer most from their own abstraction and lack of spatial rootedness, for action always takes place in spaces of action upon which the contents are to refer in their full complexity. Scenarios which do not incorporate the logic of the locality remain ineffective. World cafés, in which the moderators stifle critical positions which stem from spatial contexts, rather than seizing upon them and using them, thereby miss their chances for success. It is not sufficient to sow a species-rich meadow or to wet a low-lying area, even if these are, beyond any doubt, good deeds. Rather, the relationship to the landscape space and the relationships existing within them is indispensable, even if the resulting balance sheet may be depressing. The logic of the school garden is useful; however, it does not yet yield any understanding of the relationship of tension between various ES.

De Haan and Kuckartz (1998) describe a 'distance gap' with respect to the perception of critical environmental situations which they interpret from various perspectives–the role of the media, interest in faraway places, or a globalised environmental consciousness. According to this thesis, environmental impacts increase with distance, while one's own surroundings remain intact. This is in fact often unwittingly reinforce by certain manners of work in education for sustainable development, due to a predominant focus on global contexts which affect humankind as a whole (cf. the development of the problem horizon in Rieß 2010, or the main syndromes of global change in de Haan and Harenberg 1999), and the corresponding environmental behaviour generally begins and ends in the perception of consumer options. In order to make use of the methods of education for sustainable development for the communication of ES and make them fertile in the participatory planning process, precisely this principle needs to be reversed. Sustainability conflicts are primarily to be found before one's own door. Such a paradigm shift would however require a critical debate, a fearless scientific description of this conflict and open questions. It seems that such precepts tend to be an exception in the present environmental communications process.

One promising path in this concept is provided by the European Landscape Convention (ELC 2000), which Germany has never signed or ratified, and which as a matter of course sees a spatial connection in education on landscape policy (as justified in a case example tested by Kulozik 2009). This approach, oriented towards the peculiarity of concrete landscapes and the changes taking place within them also promotes development of the topic of ES (► Sect. 3.4), since it:

1. Takes the particular landscape conditions of various processes subsumed under the heading of ES, i.e. a specific ecosystemic balance or dis-balance, as its point of departure
2. Seeks a connection with the perception of the landscape held by its own inhabitants; i.e. based on the communication process, it qualifies, processes and develops further precisely those potentials which have a prospect for gaining a response from the communicative counterpart

An orientation towards the simple and internally logically structured agenda of the landscape convention for communications regarding various ES is to be recommended, even if the demands raised herein have not yet been politically established. Such and orientation can be easily prepared by means of education about the landscape; it allows for the integration of partners such as artists, land users, conservationists, local politicians, etc., and it is evidently–like all development of the landscape–open-ended with regard to outcome. In the context of concrete landscapes, there is no need for protection against cheap arguments, since the contradictions and interdependencies of one's own space are considerably more easily recognisable than are globally conveyed contexts: behind every practical action in the landscape is an actor with societal conditions demanding a certain action. Michelsen (2002) states in this regard 'that the context of knowledge acquisition is also a decision-making factor about the relevance of knowledge for action.'

Precisely this situation makes landscape an ideal context for education. The fact that such approaches are nonetheless the exception in Germany is on the one hand due to the lack of any corresponding discursive framework–the term 'landscape' is hardly present at all in the German discourse over sustainability–and on the other, to the mistaken idea that dealing with particular landscapes will ultimately lead to a dissipation of forces, so that the overarching whole–global change–risks getting lost in the process. To this, one might respond that skill in dealing with ES can only emerge in the caring dealing with particular cases and, once it has taken shape, will always grow beyond its original dimensions.

- **Landscape Workshops–A Point of Attachment for Local Discussions, Regional Debates and Societal Discourses**

As social beings, we have various social connections. We live in a family, share in the life of a village community or an urban neighbourhood, belong to a professional grouping, and are citizens of a country. In the communications regarding ecological matters, the various levels, languages, logics and issues emerging from this situation have not been sufficiently considered to date. The oft-cited slogan 'Think globally–act locally,' which was also used for the Agenda 21 campaign, easily blurs the various communications processes which, while occurring parallel to one another, often occur without mutual reference, and with each constructing its own environment.

For the inhabitants of a major city, rural space is their nearby environment, while the inhabitants of those rural areas tend to see it as their own space which they themselves shape. Depending on the circumstances, different sustainability issues may use different symbolic places. Issues which have become established in society as a whole by way of the mass media may have been completely ignored by village communities; on the other hand, societal discourses often screen off regionally specific conditions. The limits to scientific communications efforts resulting from this situation cannot here be systematically developed, but it is certainly recommended that the level at which an ES is to be validated be precisely identified.

A local conflict, e.g. regarding a rewetting project, will have to use the scope of communications existing in a certain place; the rhetoric of climate change will seldom be of use here. On the other hand, if an international agreement on climate protection is at issue, the situation is reversed. Considerable problems may arise even at the point of transition from the space of action at the level of a cultural landscape to that of the purely local level. It is possible, by means of landscape workshops (Anders and Fischer 2010), to attempt over a lengthy period of time to continually link local, regional and societal discourses, and to thus influence them with regard to their perception of ES.

Since actors who can convincingly convey such matters as topics from the mass media into a concrete local space are few in number–generally, this is only done successfully on a temporary basis by the appearance of prominent political figures–overall societal contributions to the debate usually bypass the regions. In such cases, still there is a possibility of combining local aspects into perspectives for action at the concrete level, and to thus inject them into the debate. This approach is close to an understanding of communications science as communicating science (Ivanišin 2006), which is ultimately oriented towards the qualification of space-related discourse.

Outlook

Let us here summarise the essential statements as theses:

- Communication is a precondition for the validation of ES; however, it can only be shaped to a limited extent, i.e. the initiator of a communications process does not have sole control over its outcome.
- The term 'ecosystem services' brings together, with communicative intent, various processes of ecosystems and landscapes which have not hitherto been satisfactorily linked, a fact which has ultimately resulted in confusion in communication.
- The political sphere and the market cannot replace communication; rather, they are themselves societal subsystems, differentiated by communications. There are approaches in the environmental sciences to use the media of these systems, which requires considerable change in the self-understanding of science, but for which there has to date been no sufficient justification.

The legitimate demand to nonetheless shape communications has resulted in the formation of various schools and approaches in the context of sustainability communications:

- *Classical knowledge transfer* is today often dismissed as 'popular science.' However, the means available here permit a precise provision of scientific results for extra-scientific communication, and should therefore continue to be used.
- *Transdisciplinary knowledge transfer* is a worthwhile undertaking, but it does require that the

environmental sciences abandon, for the sake of communication, their claim to a monopoly over the concept of knowledge. Without debates, transdisciplinary processes will moreover suffer from a loss of quality due to the erosion of professional standards.

- *Social-marketing and target-group-specific communications strategies* should be critically examined with respect to the extent of their reach. Their core business is that of consumer patterns and behaviour forms which are very close to consumerism–e.g. the acceptance of laws and societal practices.
- *Campaigns* can be used effectively, but ultimately they constitute more of a service institution than an ecosystem service.
- In the context of *education for sustainable development*, global perspectives often dominate; they are important, but they should be conveyed in their own space. The communication regarding particular ES in their mutual interrelationships can be very successful in the context of landscape-policy education.
- *Local regional and societal discourses* are very difficult to link, since they constitute different environments and establish different issues. In place of the question, 'Which target groups do I want to address?' It is more promising for communication to ask, 'Which public do I want to address, i.e. within which issue contexts will I want to place a contribution which is to be communicated?'

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