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Assessment of ecosystem integrity and service gradients across Europe using the LTER Europe network

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ABSTRACT

Better integration of knowledge from ecological, social and economic science is necessary to advance the understanding and modelling of socio-ecological systems. To model ecosystem integrity (EI) and ecosystem services (ES) at the landscape scale, assessment matrices are commonly used. These matrices assign capacities to provide different services to different land cover types. We revised such an existing matrix and examined the regional heterogeneity in EI and ES provision in Europe and searched for spatial gradients in their provision to elucidate their suitability for large-scale EI and ES mapping in Europe. Overall, 28 sites belonging to the Long-Term Ecological Research network in Europe participated in this study, covering a longitudinal gradient from Spain to Bulgaria and a latitudinal gradient from Italy to Sweden. As a primary outcome, an improved and consolidated EI and ES matrix was achieved with 17.5% of all matrix fields updated. For the first time, this new matrix also contains measures of uncertainty for each entry. EI and ES provision assessments were more variable for natural and semi-natural than for more anthropogenically dominated land cover classes. Among the main types of EI and ES, cultural service provision was rated most heterogeneously in Europe, while abiotic provisioning services were more constant. Longitudinal and latitudinal EI and ES gradients were mostly detected in natural and semi-natural land cover types where temperature and precipitation are major drivers. In anthropogenically determined systems in which cultural services play a dominant role, temperature and precipitation gradients were less important. Our results suggest that this matrix approach to assess EI and ES provision principally works on broad spatial scales; however, local assessments for natural systems seem to be less generalizable than assessments from anthropogenically determined systems. Provisioning and regulating

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services are more generalizable than cultural services. Particularly in natural and semi-natural systems, spatial gradients need to be considered. We discuss uncertainties associated with this matrix-based EI and ES assessment approach and suggest that future large-scale studies should include additional land cover information and ecosystem disservices and may determine ES fluxes by differentiating between ES provision and consumption.

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1. Introduction

Modelling of complex socio-ecological systems is one of the major challenges in contemporary transdisciplinary research (Filatova et al., 2013; Walker et al., 2006). This task requires a comprehensive, interdisciplinary integration of ecological, social and economic aspects with specific conceptual as well as simulation models (An, 2012; Burkhard et al., 2010) and hence, implicates an outstanding need for data (Wallace, 2007). Ecosystem integrity (EI) and ecosystem services (ES) are suitable conceptual models to deal with complex socio-ecological systems in a systematic and integrative manner (Johnston et al., 2011; Kandziora et al., 2013; Paetzold et al., 2010) and both concepts have attracted increasing interest of scientists and decision makers, especially during the last decade (de Groot et al., 2010; Portman, 2013). This reflects the increasing awareness in society and political frameworks that safeguarding ecosystem functioning, and sustaining a balance between supplies and demands of ES are prerequisites for long-term human well-being (Haines-Young and Potschin, 2010). This view is also mirrored in a number of environmental legislations, such as the Habitats Directive, the Water Framework Directive and the European Marine Strategy Framework Directive in the European Union alone, as well as similar legislations worldwide.

Concerning the ecological component of coupled socio-ecological systems, both ecosystem structures and processes are relevant for ecosystem functions. These ecosystem functions, the so called 'supporting services' in the Millennium Ecosystem Assessment (2005) and Müller and Burkhard (2010), can be assessed with the concept of EI (Müller, 2005). This approach has proven to be successful in describing the interrelationships between ecosystem functioning, biodiversity and the delivery of ES (Haines-Young and Potschin, 2010; Kandziora et al., 2013; Schneiders et al., 2012). Ecosystem services, in turn, provide a logical linkage between ecosystems and social systems, describing and quantifying the societal appropriation of ecosystem functions. Therefore, ES provide a good model of complex socio-ecological systems. Based on different degrees of ecosystem integrity, capacities to supply particular services can vary strongly across landscapes (Burkhard et al., 2009). The individual ES supply potentials are therefore linked to natural conditions, e.g. land cover (vegetation foremost), soil conditions, fauna, topography and climate as well as human impacts (e.g. land use intensity, pollution).

The main challenge in the application of the ES concept is the generation of appropriate data to quantify ES (Feld et al., 2009; Wallace, 2007). As the ES concept is very holistic and comprehensive, a wide variety of information has to be taken into account. This information should be sufficiently detailed, in a relevant resolution and at appropriate spatial and temporal scales (Hou et al., 2013). Especially for comparative studies at large spatial scales, collecting quantitative data on all aspects of EI and ES is illusory. Therefore, a number of investigations have successfully used structured expert-based local ecosystem assessments to gain the required data (e.g. Burkhard et al., 2009; Palomo et al., 2013; Vihervaara et al., 2010). These assessments combine measured data and expert knowledge on local ecosystems and thereby provide comparable semi-quantitative data that can be used to assess EI and ES in diverse landscapes and at large spatial scales. Such an approach will likely also be followed by the 'Mapping and Assessment of Ecosystems and their Services in Europe' (MAES) programme, which is one of the key actions of the EU Biodiversity Strategy to 2020.[1]

To classify and compare ES across larger spatial units, a consistent framework of land cover types is needed in which ES can be assessed. CORINE land cover (CLC; EEA, 2006) provides such a unifying classification system for 27 countries in Europe. This system has been used successfully in ecosystem assessment studies before (Burkhard et al., 2009, 2012). In these studies, a two-dimensional matrix linking different land cover types with capacities for EI and ES was developed. This matrix was based on expert knowledge and allows spatially explicit analyses of EI and ES.

However, a potential drawback of using such a fixed matrix approach to compare EI and ES at the continental scale is that it disregards regional heterogeneities in the provision of ES by individual land use classes. Only ES with limited regional heterogeneity, or with a heterogeneity that can be explained by additional co-variables, are suitable for the matrix assessments at larger spatial scales.

To address these issues related to matrix-based assessments of EI and ES at large spatial scales, we address the following questions in this study: (1) How much do land cover type-specific assessments of EI and ES vary among regions in Europe and (2) is the provision of certain ES linked to the same land cover classes everywhere in Europe? To answer these questions, we evaluated EI and ES based on an assessment matrix and local experts' knowledge about different local ecosystems. In the next step we analyzed the heterogeneity in EI and ES components in different land cover types in Europe, which allowed distinguishing and categorizing individual EI and ES components and CLC classes according to their suitability for large-scale assessments. In this context, we ask (3) if EI and ES provision is heterogeneous in Europe, and whether this heterogeneity can be explained by spatial gradients. If so, spatial information could be used as a co-variable to facilitate comparative analyses of EI and ES provision at large spatial scales.

To address such problems, a network of ecosystem experts from all major European ecosystems is indispensable. The European Long-Term Ecological Research network (LTER-Europe; Parr et al., 2002) is well suited for this task. The local experts that monitor a wide range of environmental variables have an excellent overview of EI and ES at their sites. Therefore, educated expert assessments based on long-term monitoring should provide more reliable and robust results than singular measuring campaigns that produce "snap-shot information" only. These local expert assessments are particularly suitable to reveal spatial patterns and gradients of EI and ES in different land cover classes across Europe. This knowledge is integral for the development of large-scale EI and ES assessment schemes.

2. Materials and methods

Our study is based on two data sources: the most recent CLC maps for Europe (see Section 2.1) and a matrix relating each of the

[1] http://biodiversity.europa.eu/ecosystem-assessments/european-level.

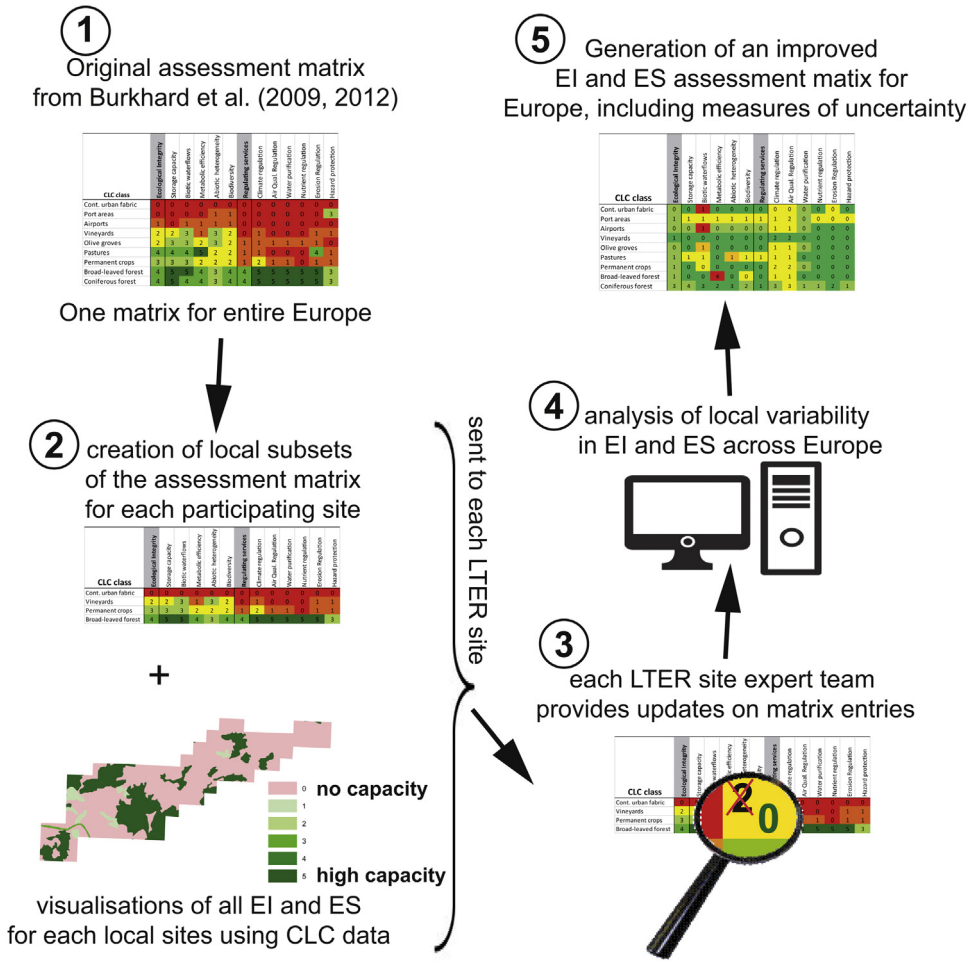

Fig. 1. Work flow of this study, from the original ecosystem integrity (EI) and ecosystem service (ES) assessment matrix to the improved and consolidated assessment matrix. CLC = CORINE land cover, LTER = long-term ecological research.

44 CLC classes in Europe with individual EI (see Section 2.3) and ES (see Section 2.4) components (39 individual EI and ES components assessed; see supplementary data S1). This matrix includes the ranked contribution of each CLC class to individual EI and ES components and is mainly based upon expert assessments (range from 0: 'no capacity to provide demanded particular service' to 5: 'very high capacity to provide demanded particular service'; for a detailed description how this matrix was conceived, see Burkhard et al., 2009, 2012, 2014).

For each participating LTER-Europe site in this study, CLC maps were extracted from the European CLC database and a subset of the complete matrix including only CLC classes present at the individual site was created (Fig. 1). The maps and assessment matrices were sent to the local expert teams, who were asked to adjust the original matrix entries based on their knowledge on EI and ES provision in the CLC categories at their site. To standardize this adjustment process, the local experts were provided with a specific protocol that they had to follow (supplementary data S2). This general approach to use ES matrices was already applied in several other case studies (e.g. Burkhard et al., 2009, 2012; Kaiser et al., 2013; Vihervaara et al., 2010) and also inspired other ES studies (e.g. Baral et al., 2013; Maes et al., 2011). The adjustments suggested by the local site expert teams were used to improve the European EI and ES assessment matrix and to analyze spatial gradients in EI and ES provision across Europe.

2.1. CORINE land cover (CLC) data

In 1985 the 'CoORdination of INformation on the Environment' programme was initiated in the European Union. Later the programme was taken over by the European Environmental Agency. Originally, the main goal was to map 44 European land cover classes belonging to four main aggregated CLC classes ((1) wetlands and water bodies, (2) forests and semi-natural areas, (3) agricultural areas, (4) artificial surfaces) at a scale of 1:100,000 for 26 European countries. This proved to be a huge task and nowadays it has become a milestone in the search for a pan-European consistency of environmental information across the member states. The original CORINE has been updated and enlarged twice: in the years 2000 (32 European countries) and 2006 (38 European countries). Despite its relatively coarse scale, it has been widely used, such as to evidence land cover changes across Europe (Feranec et al., 2007), for forest mapping (Pekkarinen et al., 2009), to estimate carbon storage (Cruickshank et al., 2000) and for mapping land function dynamics (Verburg et al., 2009). Several methods have been proposed to assess ES using CLC maps (Burkhard et al., 2009, 2012; Drius et al., 2013; Koschke et al., 2012; Kroll et al., 2012). Although several drawbacks have been pointed out about the CORINE thematic accuracy (EEA, 2006), this system is still the best European-wide data source available for land cover. Nevertheless, the present work also addresses suitability issues of CORINE maps to assess the spatial dimensions of ES provision. The most recent update of the CLC map

Table 1
LTER sites participating in this analysis and number of CORINE land cover (CLC) classes present within each site. The numbering of sites corresponds with the numbers given in Fig. 2.

No.	Country	LTER site	Longitude	Latitude	*N* CLC classes
1	AT	Reichraming	14.45481	47.84881	7
2	BG	Srebarna	27.07806	44.11279	9
3	DE	Bayrischer Wald	13.39707	48.98947	7
4	DE	Bornhoved	10.24382	54.09545	12
5	DE	Darß-Zingst	12.67636	54.37955	1
6	DE	Friedeburg	11.71509	51.61908	6
7	DE	Greifenhagen	11.43060	51.63267	7
8	DE	Harsleben	11.07514	51.84234	7
9	DE	Rhine-Main-Observatory	8.99938	50.15883	18
10	DE	Siptenfelde	11.04760	51.65116	7
11	DE	Uckermark	13.65327	53.35717	13
12	DE	Wanzleben	11.44482	52.08000	5
13	ES	Doñana	-6.41177	36.99470	9
14	ES	Sierra Nevada	-3.13842	37.06871	21
15	FI	Lake Päijänne	25.47392	61.65385	13
16	IT	Coastal Dunes	14.96689	41.98173	7
17	IT	Majella	14.11011	42.07657	5
18	IT	Venice Lagoon	12.32919	45.40108	13
19	LT	Curonian Spit	21.07451	55.47174	11
20	NL	Wadden Sea	5.56010	53.28800	28
21	PO	Brenna-Zlewinia	18.93082	49.65997	2
22	PO	Slowinski	17.46824	54.69565	2
23	RO	Braila	28.02388	44.99032	25
24	RO	Neajlov	25.55024	44.33154	14
25	SE	Aneboda	14.90854	59.75431	1
26	SE	Gammtratten	12.02408	58.05441	1
27	SE	Gardsjon	18.11430	63.85922	1
28	SE	Kindla	14.55433	57.11396	1
	Sum: 11	Sum: 28			Mean ± SD: 9.0 ± 7.2

for 2006 was selected for this study as the basic source for spatial information on land cover.[2]

2.2. LTER Europe network

Participating sites of this study belong to the European Long-Term Ecological Research network (LTER-Europe[3]). LTER-Europe is the umbrella organization for more than 300 LTER sites and Long-Term Socio-Ecological Research (LTSER) platforms. Focusing on ecosystem research and monitoring, the local site experts have an excellent overview of ecosystem structures and functions at their sites, as well as the societal demand for ES. This makes the LTER network an ideal testing ground for this type of study. A good overview of the thematic orientation of LTER sites and data that is available is provided by the metadata base Drupal Ecological Information Management System (DEIMS).[4] A total of 28 LTER sites from 11 countries participated in this study (Table 1), spanning an East–West gradient in Europe from Bulgaria to Spain and a North–South gradient from Sweden to Italy (Fig. 2). On average, 9 CLC classes were present at each site (range: 1–28). Altogether the sites include 43 different CLC types, and thus all CLC classes that occur in Europe except CLC class 'glaciers and perpetual snow'. The most common CLC classes were different types of forests, occurring at more than 70% of all sites, while other CLC types occurred only at a limited number of sites (Table 2).

2.3. Concept of ecosystem integrity (EI)

Ecosystem integrity refers to the self-organizing capacity of ecological systems as well as their resistance against non-specific ecological risks (Müller, 2005), which varies depending on the system's developmental stage and due to occurring disturbances, caused for example by human land use activities or land cover change (Drius et al., 2013; Fränzle et al., 2008). The key components to represent EI are ecosystem structures (such as biodiversity, abiotic heterogeneity) and ecosystem processes related to energy balance (exergy capture, entropy production, metabolic efficiency), water balance (water flows) and matter balance (storage capacity, nutrient loss) (Müller, 2005). It is assumed that with these key components, the capacity for self-organization and for increasing the complexity of ecosystem structures and processes can

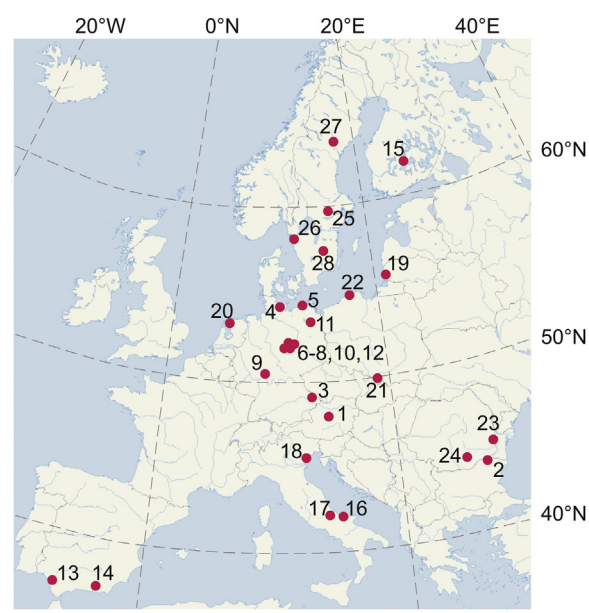

Fig. 2. Location of the 28 sites that participated in this study. Numbers given in the map correspond to the site numbering in Table 1.

[2] http://www.eea.europa.eu/data-and-maps/data/clc-2006-vector-data-version.
[3] www.lter-europe.net.
[4] http://data.lter-europe.net/deims.

Table 2
Representation of CORINE land cover (CLC) classes at the participating LTER sites (total: 28).

CLC code	CLC class	*N* sites	% sites
311	Broad-leaved forest	20	71.4
312	Coniferous forest	20	71.4
112	Discontinuous urban fabric	17	60.7
243	Agriculture and natural vegetation	17	60.7
211	Non-irrigated arable land	16	57.1
313	Mixed forest	15	53.6
231	Pastures	12	42.9
324	Transitional woodland shrub	12	42.9
121	Industrial or commercial units	9	32.1
242	Complex cultivation patterns	9	32.1
321	Natural grassland	9	32.1
512	Water bodies	9	32.1
411	Inland marshes	8	28.6
222	Fruit trees and berry plantations	7	25.0
511	Water courses	7	25.0
131	Mineral extraction sites	6	21.4
331	Beaches, dunes, sand plains	6	21.4
141	Green urban areas	5	17.9
142	Sport and leisure facilities	5	17.9
123	Port areas	4	14.3
132	Dump sites	4	14.3
221	Vineyards	4	14.3
111	Continuous urban fabric	3	10.7
122	Road and rail networks and associated land	3	10.7
124	Airports	3	10.7
323	Sclerophyllous vegetation	3	10.7
332	Bare rocks	3	10.7
333	Sparsely vegetated areas	3	10.7
421	Salt marshes	3	10.7
521	Coastal lagoons	3	10.7
523	Sea and ocean	3	10.7
322	Moors and heathland	2	7.1
334	Burnt areas	2	7.1
133	Construction sites	1	3.6
212	Permanently irrigated land	1	3.6
213	Rice fields	1	3.6
223	Olive groves	1	3.6
241	Annual crops associated with permanent crops	1	3.6
244	Agro-forestry areas	1	3.6
412	Peatbogs	1	3.6
422	Salines	1	3.6
423	Intertidal flats	1	3.6
522	Estuaries	1	3.6

be described based on a feasible number of indicators. For more detailed descriptions and applications of related integrity indicators see Müller (2005), Müller and Burkhard (2010) and Fränzle et al. (2008). Cycling of energy, matter and water, specific diversity of functional key species and suitable abiotic conditions are key components for the description of ecosystem functioning (de Groot et al., 2010), which again are a prerequisite for ES supply. In some studies, ecosystem structures and functions are also referred to as supporting ES (MA, 2005), but this nomination is discussed controversially as a societal demand is commonly regarded as a crucial aspect in the definition of an ES (Boyd and Banzhaf, 2007; Fisher et al., 2009). This demand is not required in the EI concept.

2.4. Concept of ecosystem services (ES)

Natural ecosystems constitute an indispensable capital for human well-being, since they assure basic needs such as water and food supply (Costanza et al., 1997; Costanza and Daily, 1992). These natural benefits were originally described by Daily (1997) as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" and since then much effort was used to better define the concept of "ecosystem services" (e.g. Boyd and Banzhaf, 2007; Fisher et al., 2009). Even though the definition of this concept is not univocal, the urgent need to incorporate the ES perspective when defining environmental planning frames is largely recognized (MA, 2005; Egoh et al., 2007; Feld et al., 2009). The ES concept links ecosystem structures and processes (described by EI) to the benefits humans derive from goods and services produced by ecosystems. Thus, they are suitable models of complex socio-ecological systems. Different classifications of services, including both material and spiritual values of natural environments, have been proposed to date (MA, 2005; Costanza et al., 1997; de Groot et al., 2010). For instance the ES 'cascade' model by Haines-Young and Potschin (2010) is used to describe interrelations in socio-ecological systems. The selection of ES for the assessment in this study was based on a combination of the most recent ES categorisation systems (de Groot et al., 2010; MA, 2005; CICES[5]). The three common ES categories of regulating, provisioning and cultural ES were used to group together 31 different ES (see Table S1). More detailed descriptions of the individual services and potential indicators for quantifications can be found in Kandziora et al. (2013) and Burkhard et al. (2009, 2012, 2014).

2.5. Statistical analyses

The adjusted EI and ES assessment matrices that were returned from the local expert teams were compiled into one database, from which a new synthetic, updated matrix was derived by calculating average values (rounded to integers) and standard deviations of the individual expert assessments. Only CLC classes that underwent at least three independent assessments were updated (31 CLC classes out of 43 in total). For less common CLC classes the original matrix entries were retained. Differences in the adjustment frequency, i.e. the proportion of matrix entries that were changed in each of the four main aggregated CLC classes and the four main EI and ES groups (EI, provisioning services, regulating services, cultural services), were analyzed with one-way Analysis of variance (ANOVA), respectively, followed by Tukey HSD post hoc tests. Homoscedasticity of the data was checked with Levene tests and met ($P > 0.05$).

As a next step, longitudinal and latitudinal gradients in the local assessments of the four main ES types were analyzed with linear mixed models using the function 'lmer' in the add-on package 'lme4' (Bates et al., 2013) within the framework of R 2.13.1 (R Development Core Team, 2011). *P*-values for the models were estimated using the function 'cftest' in the add-on package 'multcomp' (Hothorn et al., 2013). One model was fitted for each of the four main groups of EI and ES. In these models, the mean value of the local residuals of all components within an EI and ES main group was used as the dependent variable; independent variables were CLC class and the class-specific effects of longitude and latitude using the interaction terms of CLC class:longitude, as well as CLC class:latitude. To consider the nested data structure of different CLC classes within one site, the variable LTER site ID was added to the models as a random factor. The spatial gradients were then determined from the interaction terms in the model. This approach, in which all CLC classes were analyzed in one model, was chosen to avoid alpha error inflation, which would occur if all CLC classes were analyzed individually. Only CLC classes present at least at five LTER sites were considered in this analysis. With less independent assessments, no robust spatial trends can be estimated. To avoid model over-fitting in this dataset, with its limited number of replicate assessments per CLC class ($5 \leq n \leq 20$), we decided neither to add additional predictor variables like altitude and areal coverage of individual CLC classes, nor to consider non-linear effects. Such expanded, more detailed models would be desirable; however, they

[5] http://cices.eu/; including a (provisional) classification of abiotic outputs from natural systems, e.g. minerals and wind energy.

can only be realized with larger datasets. This is what we aim for in future studies.

3. Results

From the adjusted assessment matrices returned from the local LTER site expert teams, an updated synthesis version of the CLC versus EI and ES assessment matrix was generated (supplementary data S3). Only those CLC classes were updated, for which at least three independent assessments were available. Thus, the update concerned 1209 ranking cells of the matrix (31 CLC classes being present at least three times versus 39 individual EI and ES components that are distinguished). The values of 211 of these 1209 cells (17.5%) were changed in the new matrix compared to the original matrix (Table 3). This resulted in changes in 22.6% of the four group averages (ecosystem integrity, provisioning services, regulating services and cultural services). There were twice as many cases with increases than decreases in the assessment values. In most of the cases, the rankings were changed by ±1 step, but changes up to +4 and −3 steps also occurred.

3.1. Heterogeneity in EI and ES assessments

The need for local adjustments of the capacities to supply individual EI and ES components varied between the CLC classes (Fig. 2A). For example, in salt marshes, on average each expert team of a LTER site in which this CLC class occurred altered 15 of the 39 ES components (38.5%), while on average only less than one out of the 39 ES components (1.7%) were adapted for road and railroad networks. The four main groups of CLC classes showed significant differences in the frequency of local adaptation (Fig. 2B; ANOVA: $F_{3,27} = 6.33$, $P = 0.002$). The assessments for waters and wetland areas needed most local adjustments (Fig. 2B), except sea and ocean which ranges second to last in the list. CORINE land cover classes associated with forest and semi-natural areas needed similar amounts of local adjustments. Artificial surfaces, including all urban and industrial CLC classes, had the least need for local adjustments in the EI and ES assessment, whereas agricultural areas were intermediate.

Among the individual EI and ES components, pollination was adjusted most frequently by the local LTER site expert teams (Fig. 3A). On average, each participating site management team altered the relevance of pollination in 30% of the locally occurring CLC classes. However, in sites dominated by aquatic and wetland habitats, this proportion was lower than at entirely terrestrial sites. Overall, cultural services exhibited the greatest need for local adjustments (Fig. 3B; ANOVA: $F_{3,35} = 6.71$, $P = 0.001$), followed by regulating services. Provisioning services and EI indicators showed the least need for local adjustments.

Contrasting the frequency of adjustment for individual EI and ES components in each CLC class with the heterogeneity between the replicate assessments of the local expert teams showed a high degree of correlation between the two (Fig. 4). This analysis further allows separating the CLC classes and EI and ES components in four different sectors, indicating their suitability for analyses on large spatial scales. These sectors are characterized by below- and above-average values in adjustment frequency and adjustment heterogeneity, respectively. Sector 1: CLC classes and EI and ES components were rarely adjusted and the heterogeneity in the replicate assessments was low. These are highly homogenous CLC classes or EI and ES components across Europe, and thus particularly qualified for assessments on a large spatial scale. Sector 2: CLC classes and EI and ES components were infrequently adapted, but replicate assessments by the local expert teams were more heterogeneous. Thus they have an intermediate potential for large-scale analyses.

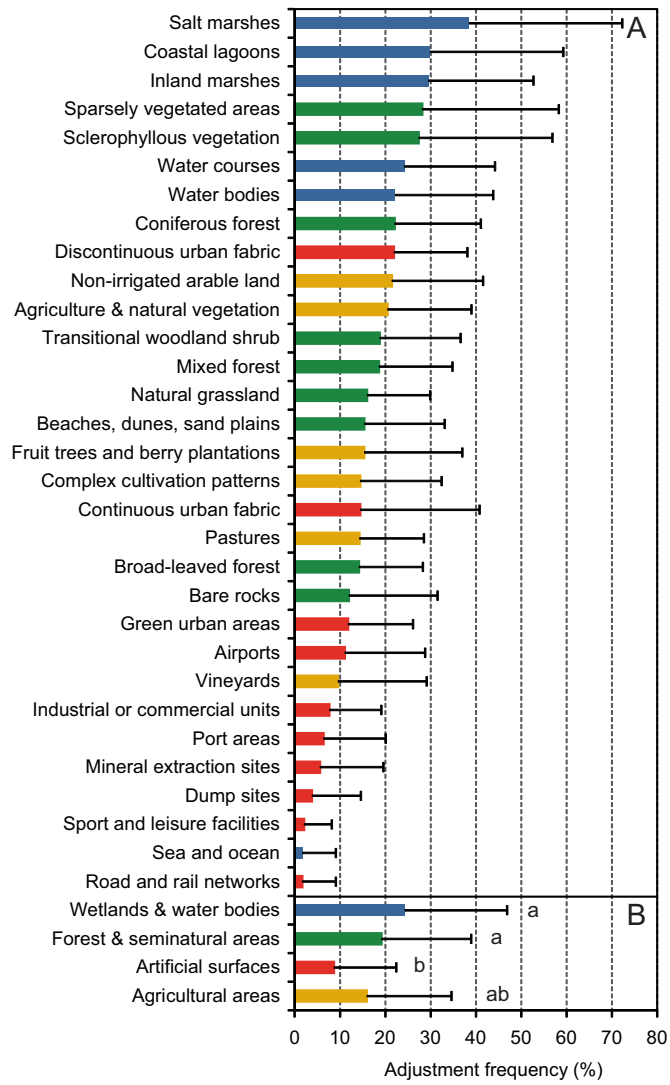

Fig. 3. Proportion of matrix entries (mean ± standard deviation) changed for each CORINE land cover (CLC) class separately (A) and for each aggregated groups of CLC classes (B). The colors of individual CLC classes in (A) match the colors of the four aggregated groups of CLC classes in (B). Lower-case letters indicate significant differences between CLC class groups in Tukey HSD post-hoc test.

Sector 3: CLC classes and EI and ES components may be suitable for large-scale analysis after their updating. These CLC classes and EI and ES components are characterized by a frequent and consistent need for adjustment in the parameterization between all LTER sites, probably because the original matrix entries deviated from the prevalent estimation of these parameters. Sector 4: CLC classes and EI and ES components are less suited for large-scale analyses, as they are prone to frequent and heterogeneous adjustments. These frequent, however inconsistent adjustments of the local experts may reflect either spatial gradients in the parameterization of individual EI and ES components or other types of underlying variability, e.g. different intensity of land use and land protection schemes.

All land cover classes in aquatic and wetland habitats except sea and ocean were located in sector 4 (Fig. 5A). Sea and ocean (CLC code 523), in contrast, fell into sector 1, as only few and very consistent updates were suggested by the local expert teams. Most types of artificial surface areas fell into sector 1 except CLC classes continuous urban fabric (111) and discontinuous urban fabric (112), falling into sectors 2 and 3, respectively. All agricultural and semi-natural habitats including forests showed average adjustment frequencies

Table 3
Summary of changes in the assessment matrix.

N matrix cells (all CLC ≥ 3)	All cells		Ranking cells only		Group averages	
	N 1333	% 100	*N* 1209	% 100	*N* 124	% 100
Cells corrected +1	141	10.6	121	10	20	16.1
Cells corrected +2	20	1.5	19	1.6	1	0.8
Cells corrected +3	5	0.4	5	0.4	0	0
Cells corrected +4	1	0.1	1	0.1	0	0
Cells corrected −1	64	4.8	57	4.7	7	5.6
Cells corrected −2	4	0.3	4	0.3	0	0
Cells corrected −3	4	0.3	4	0.3	0	0
Cells corrected −4	0	0	0	0	0	0
Total cell corrections	239	17.9	211	17.5	28	22.6

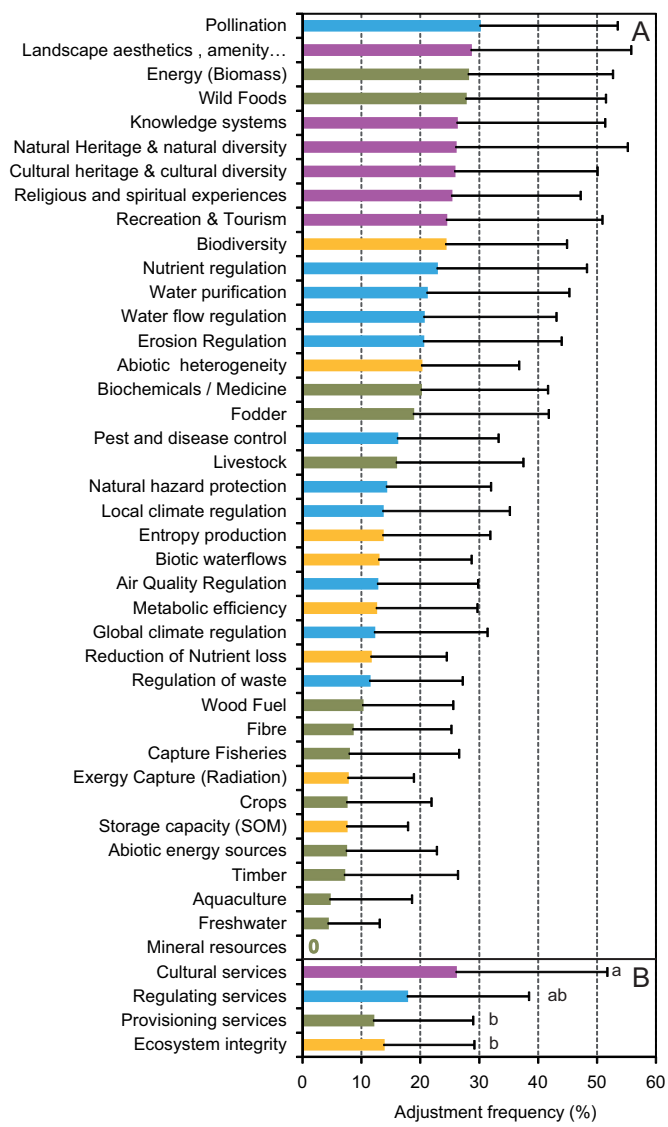

Fig. 4. Proportion of matrix entries (mean ± standard deviation) changed for each individual ecosystem service types and for aggregated groups of ecosystem service types (B). The colors of individual CLC classes in (A) match the colors of the four aggregated groups of ecosystem service types in (B). Lower-case letters indicate significant differences between CLC class groups in Tukey HSD post-hoc test.

and heterogeneities and are thus located close to the intersection of the sector border lines. Only sparsely vegetated areas (333) and sclerophyllous vegetation (323) were more variable and fall into sector 4.

Most EI indicators were located in sector 1, except abiotic heterogeneity (Fig. 5B, i1) and biodiversity (i2). Provisioning and regulating services differed widely. This was especially evident for some provisioning services; for instance, mineral resource (p1) and freshwater provision (p2) were ranked very consistently, while other provisioning and regulating services were highly heterogeneous, such as pollination (r1), energy from biomass (p3) and wild foods (p4).

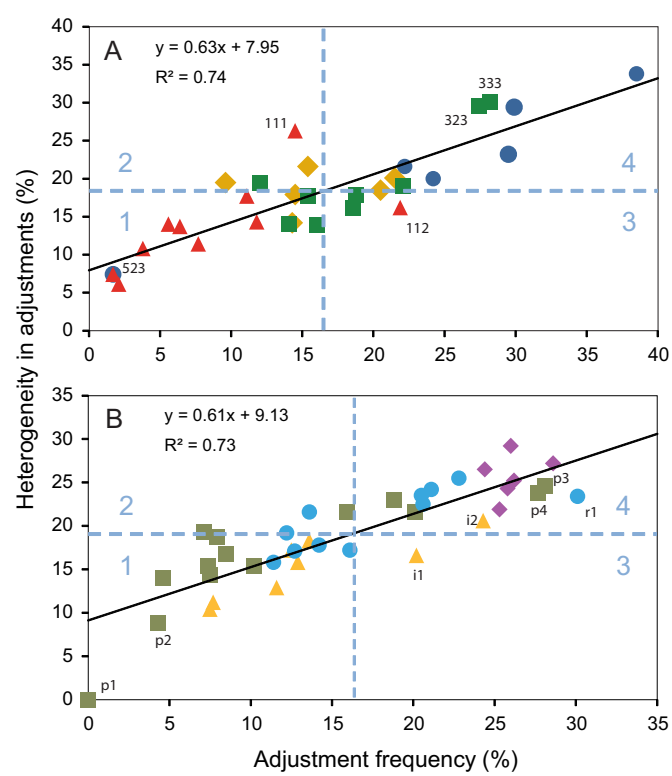

Fig. 5. Frequency of adjustments and heterogeneity in the replicate assessments of (A) CORINE land cover (CLC) classes and (B) ecosystem service types. Sectors 1–4 differentiate quadrants of below- and above-average means and SD. Colours of data points indicate CLC and Ecosystem service groups, the colour code is identical to Figs. 3 and 4. In (A), numbers refer to CLC class codes (Table 2), in (B) provision of mineral resources (p1), freshwater (p2), energy from biomass (p3), wild foods (p4); capacity for abiotic heterogeneity (i1) and biodiversity (i2); regulation of plant pollination (r1).

Table 4
Results of linear mixed models testing for latitudinal and longitudinal gradients in the four ecosystem service groups (A) ecosystem integrity, (B) regulating services, (C) provisioning services and (D) cultural services. Only CLC classes containing significant ($p < 0.05$, highlighted in bold) or nearly significant ($0.1 > p > 0.05$, highlighted in italics) spatial trends are shown.

CLC code	CLC class	Latitude			Longitude		
		Est.	SE	*P*	Est.	SE	*P*
(A) Ecosystem integrity							
112	Discontinuous urban fabric	0.01	0.01	0.214	−0.02	0.01	**0.023**
312	Coniferous forest	0.03	0.01	**0.004**	<0.01	0.01	0.842
321	Natural grassland	0.05	0.02	**0.003**	0.02	0.01	*0.051*
331	Beaches, dunes, sand plains	−0.03	0.02	*0.076*	<0.01	0.01	0.648
(B) Regulating services							
312	Coniferous forest	0.02	0.02	0.336	0.03	0.02	***0.095***
324	Transitional woodland shrub	0.04	0.02	**0.029**	−0.02	0.01	**0.047**
411	Inland marshes	−0.05	0.02	**0.018**	0.03	0.01	**0.012**
511	Water courses	0.13	0.09	0.155	0.03	0.01	**0.018**
512	Water bodies	−0.02	0.02	0.235	0.09	0.03	**0.009**
(C) Provisioning services							
131	Mineral extraction sites	−0.03	0.02	***0.093***	<0.01	0.01	0.477
242	Complex cultivation patterns	−0.03	0.02	**0.046**	<−0.01	0.01	0.921
243	Agriculture & natural vegetation	−0.04	0.01	**0.001**	0.01	0.01	0.201
312	Coniferous forest	−0.05	0.01	**<0.001**	0.04	0.01	**<0.001**
411	Inland marshes	−0.04	0.01	**0.004**	<0.01	0.01	0.744
511	Water courses	−0.04	0.01	**<0.001**	0.09	0.02	**<0.001**
512	Water bodies	0.16	0.06	**0.008**	0.02	0.01	**0.040**
(D) Cultural services							
112	Discontinuous urban fabric	−0.06	0.03	**0.016**	−0.01	0.02	0.508
311	Broad leaved forest	−0.01	0.02	0.615	0.03	0.02	**0.035**
312	Coniferous forest	0.02	0.02	0.450	0.06	0.02	**0.019**
313	Mixed forest	−0.07	0.03	**0.012**	0.04	0.02	***0.062***
411	Inland marshes	−0.16	0.03	**<0.001**	0.04	0.02	**0.015**

3.2. Gradients of EI and ES supply

Significant latitudinal and longitudinal gradients in the provision of the four main EI and ES groups were detected in 13 and 11 CLC classes, respectively (Table 4). Such gradients were found disproportionally often in aquatic and wetland CLC classes (11 significant gradients in 3 CLC classes that were included in the analysis, 11/3 = 3.7). To a lesser extent, spatial gradients were found in forests and semi-natural areas (9 significant gradients in 6 CLC classes that were included in the analysis, 9/6 = 1.5). Fewest significant gradients were detected in the CLC classes representing anthropogenically transformed agricultural land and artificial surfaces (each 2 significant gradients and 5 CLC classes in the analysis, 2/5 = 0.4).

The direction and slopes of the spatial gradients of the EI and ES main groups in the individual CLC classes varied (Table 4). Ecosystem integrity increased northwards in natural grassland and coniferous forest, while EI of discontinuous urban fabric decreased towards the East (Table 4A and Fig. 6A and B). Regulating services increased northwards in transitional zones between shrubs and woodland, but decreased in inland marshes. Additionally, regulating services decreased eastwards in transitional areas between shrubs and woodland, but increased eastwards in all aquatic and wetland areas in this analysis (i.e. inland marshes, water courses and water bodies; Table 4B and Fig. 6C and D). Provisioning services decreased northwards in all CLC classes (complex cultivation patterns, agriculture and natural vegetation, coniferous forest, inland marshes, water courses) except for water bodies. At the same time, provisioning services of coniferous forests, water courses and water bodies increased towards the East of Europe (Table 4C and Fig. 6E and F). Cultural services of discontinuous urban fabric, mixed forest and inland marshes decreased northwards, whereas for broad-leaved forests, coniferous forests and inland marshes, an increase towards the East of Europe was detected (Table 4D and Fig. 6G and H).

4. Discussion

Ecosystems in Europe are complex and diverse, and thus the assessment of ecosystem services at broad spatial scales is a challenge (Anton et al., 2010; Keene and Pullin, 2011). It has even been claimed that due to the great diversity of ecosystems, the creation of one classification system for all ecosystems is impossible (Zhang et al., 2010). Nevertheless, the effort to assess ecosystem services at different scales has to be made in order to inform policy. This in turn needs the best scientific information available to pass legislation ensuring a sustainable use of ecosystem services on the long-term (Maes et al., 2012). Despite the rapid growth in research efforts on ecosystem services and their relationship to human society (Dick et al., 2011; Seppelt et al., 2011; Vihervaara et al., 2010), few studies exist that deal with spatial scaling, standardization and testing of ES assessment methods. In the present study, we tested the applicability of the matrix assessment method developed by Burkhard et al. (2009) for EI and ES mapping at the continental scale in Europe. We support the finding by Dick et al. (2014) that land cover is a suitable base layer to map ecosystem services and demonstrated that in principle the EI and ES assessment using a conversion matrix translating land cover classes into EI and ES is a feasible approach to this task also at the European scale. In particular, abiotic provisioning services and artificial surfaces proved to be highly constant across Europe in our study. They exhibited the least need for local adjustments of the EI and ES assessment matrix and showed the least heterogeneity between the individual local assessments. They are thus very suitable for large-scale mapping and comparisons.

These artificial surfaces are commonly designed to deliver few, but very specific services. Their EI and regulating services are commonly very low (Kroll et al., 2012), but their cultural values and historic relevance are widely acknowledged (Bolund and Hunhammara, 1999). In most artificial surfaces, the heterogeneity of EI and ES provision across Europe is low. For instance,

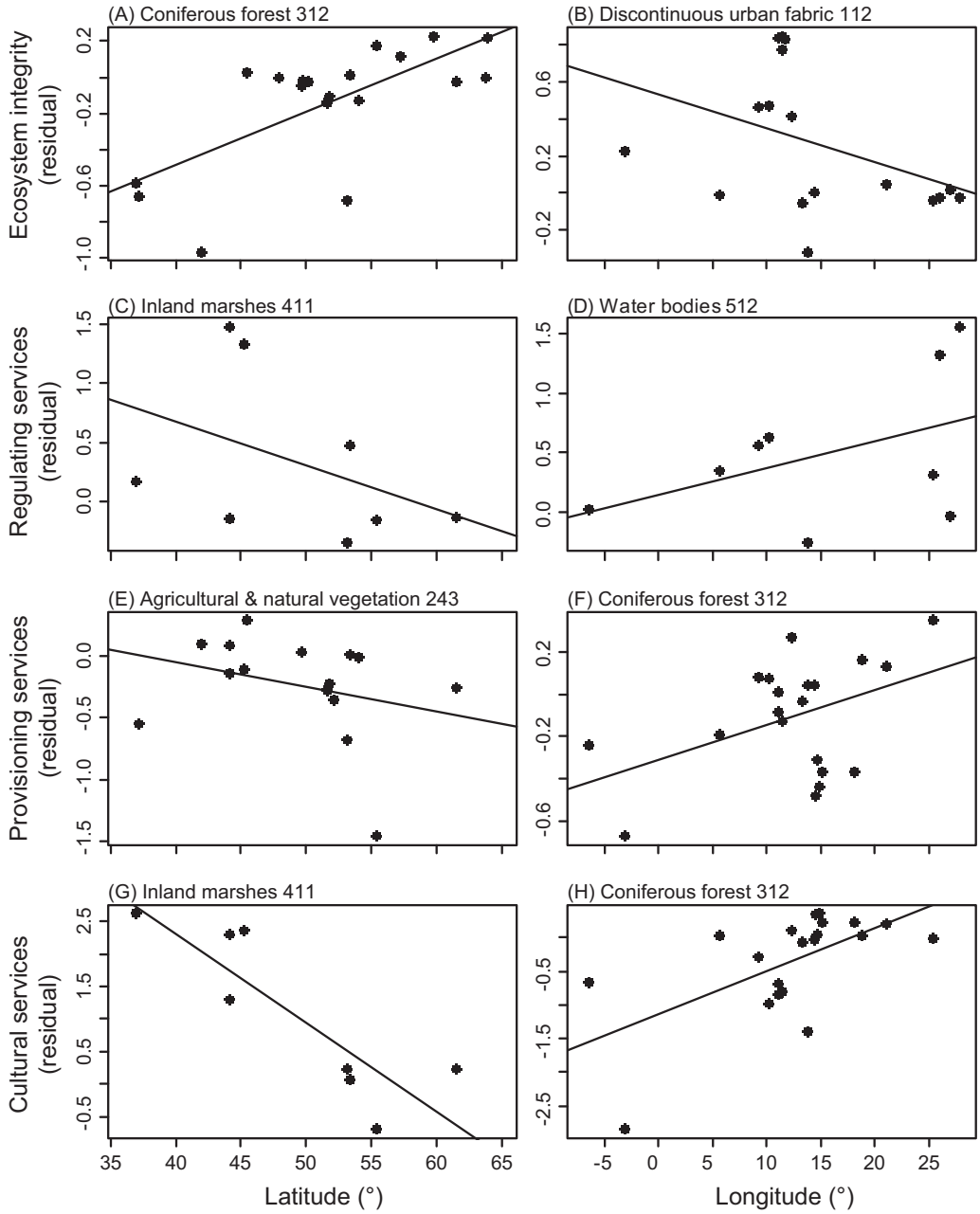

Fig. 6. Selected significant latitudinal and longitudinal patterns of the four main ecosystem integrity and ecosystem service groups. For accompanying statistics, see Table 4.

everywhere in Europe roads or football fields have very similar, specific properties and people have very similar expectations to what each should serve for.

With increasing naturalness of the land cover classes, the heterogeneity of EI and ES provision increased correspondingly. Most likely this is because in natural systems the local exploitation options as well as exploitation intensities are more diverse. In natural systems, all forms of use from complete legal protection (no use, and therefore no or limited provision of an array of ES) to intensely exploited systems occur while agricultural areas and artificial systems, in contrast, are fully exploited per definition. Secondly, less related research has been done so far in natural systems (Drius et al., 2013). The scarcity of prior studies describing EI and ES in natural areas has meant that EI and ES have so far been rather diffusely defined in such land cover classes. This consequently opens the door to a higher degree of subjectivity in the local expert assessments, and may have contributed to high heterogeneity in the local assessments. The EI and ES of agricultural CLC classes, in turn, are much better studied (van Zanten et al., 2013) and thus the entries in the original assessment matrix may have already been more substantiated.

Considering individual EI and ES components, provisioning services (especially abiotic provisioning services) and EI were predefined best and also showed the lowest degree of heterogeneity between individual local assessments. We suggest that this is because they are most easily and directly assessable. The amount of minerals that can be extracted and the amount of wind at a site is accessible by direct quantitative measurements. Also EI components are well studied, as they directly relate to ecosystem functions. Assessing such ecosystem functions has a long tradition among ecologists who constitute a large proportion of scientists working in the field of ES assessment. Hence their concepts are well established, minimizing the subjectivity in the individual local assessments.

The highest variability among the four main groups of EI and ES was found in the provision of cultural services across Europe. This high variability in the assessment of cultural services could be, again, partly due to the still limited number of research studies dealing with such services (Daniel et al., 2012; Feld et al., 2009) and thus, the conceptual framework of cultural ES being the least consolidated of the different ES types. Furthermore, as physical, emotional and mental benefits derived from cultural ecosystem services are often subtle (Kenter et al., 2011) and manifested indirectly (Anthony et al., 2009), their value is intrinsically difficult to assess and strongly depends on the cultural (and even individual) background of the assessing person (Daniel et al., 2012). In this study at the European scale, the cultural background of the local expert teams was highly diverse. Hence, the roles of "landscape aesthetics, amenity and inspiration" and "natural heritage/natural diversity" were assessed particularly heterogeneously by the local expert teams. Nevertheless, if EI and ES assessments do show a substantial amount of spatial variability, mapping remains feasible if gradients can be detected that explain this variability. This study only considered latitudinal and longitudinal gradients, as the moderate replicate number for repeated assessments of individual CLC classes did not allow more detailed analyses. Such geographical gradients were detected predominantly in natural and semi-natural CLC classes such as waters, wetlands and forests, while they rarely occurred in more anthropogenically-shaped landscapes.

We suggest that the increased occurrence of longitudinal and latitudinal EI and ES gradients in natural systems is caused by key processes in natural systems being linked to temperature and precipitation. Temperature has a strong North–South gradient, and seasonal temperature and precipitation distribution additionally have an East–West component in Europe, from continental to oceanic climate. In anthropogenically-shaped CLC classes, few geographical gradients were found. Such CLC classes predominantly deliver cultural services. Cultural backgrounds in Europe follow no strict East–West or North–South gradients, but are, in terms of geography, patchily distributed. To determine gradients here, other economic (e.g. per-capita GDP) or demographic (e.g. population density, degree of urbanization) variables seem more promising.

Even though no causal relationships between EI and ES provision and geographical gradients were determined in this study, some interesting patterns warrant speculations on their background.

Integrity of ecosystems was higher in Northern Europe for coniferous forests and grasslands. Accordingly, the intensification of land use of Europe started historically in the South from where it expanded northwards. Forests in Fennoscandia remained rather natural up to the recent few hundred years (Kouki, 1994). Despite the current intensive logging practice in these boreal forests, tree species compositions still are close to those of natural forests compared to Central and Southern Europe (Lindbladh et al., 2013).

We further found that regulating services of all freshwater habitats (inland marshes, water courses, water bodies) increase towards the East of Europe. In the more continental climate with warmer summers and colder winters, such areas may have a special role for local climate regulation.

In general, provisioning services decreased northwards. Not surprisingly, this relationship was predominantly found in forest and agricultural systems, where productivity is strongly coupled to temperature and length of vegetation period. The only exception to this trend applied to water bodies, where provisioning services increased northwards. This may be related with the more important role of professional and sport fishing there, as most of the valuable freshwater fish species in Europe are cold-water adapted, such as salmonids and coregonids (Kottelat and Freyhof, 2007).

An increase in provisioning services was detected towards the East in coniferous forest, water courses and water bodies. In the eastern parts of Europe, the degree of urbanization is lower and people still use a greater diversity of natural and semi-natural habitats, and maybe also with a greater intensity (e.g. for berry and mushroom picking, angling, etc.). In line with this, the cultural services of a range of these habitats including forests and marshlands were also ranked higher in East Europe. This greater attraction to and valuation of natural ecosystems may be one reason for our finding, that in Eastern Europe, urban areas seem to be associated with lower EI than urban areas in Western Europe.

4.1. Uncertainties of EI and ES assessments using this matrix approach

One general critical comment on the method is often derived from the fact that only semi-quantitative assessments – here expert knowledge – are used, instead of fully quantitative, measured data. To our conviction, this potential disadvantage concerning the quantitative accuracy is more than compensated for by the practicability of the approach. These analyses require such an enormous and diverse amount of data that a total, holistic quantification of all services used (e.g. by models or measurements) can hardly be reached in the forthcoming years. Hence, the immediate, high demand for applications of the ES concept in environmental management requires pragmatic solutions (Daily, 1997). Therefore, the utilization of expert knowledge is the only possibility to attain the demanded information fast and on an intersubjective level so far. The inclusion of the 28 expert teams in this study helped to improve the precision of the respective EI and ES valuations. As a further asset of this study, with the repeated assessment of EI and ES in individual CLC classes, a measure of uncertainty for each assessment can be provided (see supplementary data S3).

Also the suitability of CORINE land cover information as the only data source for land cover classification may be debated (Hou et al., 2013). Especially spatially limited landscape elements, below the relatively coarse resolution of this spatial information, such as small streams or hedge rows in an agricultural landscape, are not represented in the CLC maps. Still, such spatially restricted landscape elements can provide important services. The rather coarse resolution of CLC makes this kind of approach especially suitable for assessments on the meso- and macro-scale, while CLC data is not adequate for detailed studies at small spatial scales.

Furthermore, EI and ES are not exclusively determined by land cover. Instead, land use intensity and other information are crucial to fully describe EI and ES (Burkhard et al., 2009). For example in forests, species composition, age structure and stand density as well as the availability of young successional stages are relevant for the assessment of EI and ES (Swanson et al., 2011). Ecosystem integrity and ES do not only depend on the land cover element under consideration, but will also be affected by land use outside the boundary of a given spatial feature (Daniel et al., 2012). Moreover, the degree of landscape fragmentation, altitudinal gradients and human population density may be relevant co-variables explaining observed variability in the EI and ES assessments. Nevertheless, given all these shortcomings, the overall variability in the judgments of the experts was astonishingly small.

4.2. How to improve in the future

From the limitations and uncertainties currently associated with our study approach, next steps to improve our conceptual study design can be deducted, as follows:

(1) This study has shown which cells in the matrix have been objects of intensive changes by the regional experts (see supplementary data S3). These points will be selected as focal objects of future elaborations to clarify the reasons for their

variability and to modify the matrix model accordingly in order to reduce uncertainties. For example, the implementation of socio-economic data, topographical relief of landscapes and information on land use intensity should be able to explain additional proportions of variability in local assessments. Where current CLC classes are too broad, sub-division of such classes may be necessary with the help of additional spatial data sources (e.g. several mixed forest classes based on forest species inventories or several classes of non-irrigated arable land based on soil and husbandry maps).

(2) Numerous, variable adjustments were proposed especially for cultural services. Here we have to check whether that high variability is a consequence of the factual service provision at the given location or of the regional valuation of the service provision by the experts. There might be a further need to improve the valuation schemes and specify assessment parameters in these ecosystem types, both in these CLC classes as well as in other rather diffusely defined ecosystem types.

(3) Another line of research may deal with individual EI and ES components being provided at different spatial and temporal scales (Costanza, 2008; Rodríguez et al., 2006). The provision of some EI and ES occurs at the local scale of individual land units (e.g. crop production), while other services (e.g. pest control) are typically provided at a greater spatial grain, integrating individual land units at a landscape level (Geijzendorffer and Roche, 2013). Also the underlying ecosystem functions and processes from which services are derived typically operate at distinct scales, with specific spatial and temporal characteristics (Müller, 1992; Nielsen and Müller, 2000). Examples of this include "global" climate regulation, "landscape" aesthetics or the "field" related harvest of crops. Furthermore, groundwater regulations are operating on larger scales than individual CLC units. On the other hand, ES demand shows broad spatio-temporal variability: markets of agricultural goods, for instance, are often globalized, while "local" climate regulation demands are addressed at the landscape level. The consequences of these issues with scale and the de-localized effects of ecosystem provision and consumption have not been adequately investigated up to now. Therefore, multi-scale approaches, including additional land cover information at different spatial scales may be fruitful.

(4) Furthermore, integration of the so-called ecosystem disservices into EI and ES assessments would be helpful for trade-off analyses. Ecosystem disservices comprise unwanted or economically harmful characteristics of ecosystems, such as existence of herbivorous insects, competition for water or the emission of greenhouse gasses. Dunn (2010) suggested that the occurrence of ecosystem disservices is higher in disturbed ecosystems, and thus inversely related to EI. In this context, Zhang et al. (2007) pointed out that agricultural ecosystems deliver, and at the same time are also threatened by, ecosystem disservices. Such disservices (e.g. crop pests) can greatly affect the profitability and sustainability of agricultural production.

(5) Finally the differentiation between ES provision and consumption would help to get a more complete picture of ES fluxes. Such models help to differentiate between local, small-scale provision and use of ES versus long-distance import and export of services (Burkhard et al., 2012, 2014).

(6) For applied purposes, scenario applications are widely used to assist decision makers in valuing the outcomes of different management alternatives. For these prospections, EI and ES models have an enormous potential to improve the quality of the potential future layouts. In ES indication, different modelling systems are in use (e.g. ARIS, INVEST, see Bagstad et al., 2013; Grêt-Regamey et al., 2008; Nelson et al., 2009), which are continuously improved. A stronger linkage between service indicators and models will be very helpful to enhance the applicability of the overall approach.

5. Conclusions

We identified a high potential for the combination of long-term research with spatial ES assessments. The European LTER network, containing many of the most important ecosystem types in Europe, represented an excellent training ground to refine and test shared and replicable methods to evaluate EI and ES. We demonstrate that the use of CORINE land cover classes to assess EI and ES is principally feasible also at larger spatial scales. With this study, we provide an updated EI and ES assessment matrix for Europe. For the first time, measures of uncertainty for individual assessments are also available. Our analyses further allow distinguishing between more and less suited EI and ES components and CLC classes for large-scale assessments. Where high degrees of variability impedes direct use of certain CLC classes as well as certain EI and ES components for direct mapping at larger spatial scales, the search for spatial gradients in EI and ES in different land cover types across Europe is one way to explain and reduce this intrinsic variability. Furthermore, with this regionalization of the EI and ES assessment matrix, regional hotspots for individual EI and ES components can be detected. Finally, we point out caveats in current EI and ES assessment schemes and propose future lines of research to develop realistic, target-oriented large-scale EI and ES models to inform environmental management and administration.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolmodel.2014.06.019.

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