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REVIEW

Blind spots in ecosystem services research and challenges for implementation

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Abstract

Ecosystem service research is high on the policy agenda. Strategies to synthesize individual success stories and derive generalized results to provide guidance for policymakers and stakeholder is central to many science-policy initiatives, such as Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services and The Economics of Ecosystems and Biodiversity. However, generalization requires the documentation of basic information on methods and results of case studies, which might not be present throughout all case studies. We used a quantitative review based on a random sample of studies published in the ISI Web of Knowledge between 1996 and 2016 to identify blind spots in ecosystem service research that might hinder the generalization. We structured our analysis along critical questions about five facets that characterize the holistic ideal of ecosystem services research: (i) social-ecological validity of ecosystem data and models, (ii) consideration of trade-offs between ecosystem services, (iii) recognition of off-site effects, (iv) comprehensive and shrewd involv ement of stakeholders, and (v) relevance and usability of study results for the operationalization of the ecosystem service concept in practice. Results show that these facets were not addressed by the majority of case studies including more recent studies. Clusters of ecosystem services studied together were prone to different blind spots. To effectively operationalize the concept of ecosystem services, the blind spots need to be addressed by upcoming studies. A list of critical questions is provided to raise the awareness of the blind spots both for synthesis of existing knowledge and for future research agendas.

Keywords Operationalization · Stakeholder involvement · Good modeling practice · Quantitative review · Off-site effects · Trade-offs

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Introduction

The concept of ecosystem services (ES) centers around the multiple interactions between ecosystems and human well-being. Since its introduction by early studies (Westman 1977; Ehrlich and Ehrlich 1981), the concept has evolved intensely and integrates ecological, economic, and cultural aspects (Folke 2006; Balvanera et al. 2017). This integrated view distinguishes the ES concept from narrower environmental management perspectives (Baker et al. 2013). The topic has attracted a lot of research in the last 15 years. A look at the available bibliometric research databases shows that the body of literature on this topic is still rapidly growing (McDonough et al. 2017). Although the ES concept has been adopted in high-level policy frameworks, such as the Convention on Biological Diversity,[1] the Intergovernmental Platform on Biodiversity and Ecosystem Services (Díaz et al. 2015), and the EU biodiversity strategy,[2] there is currently a mismatch between the considerable conceptual understanding of the ES concept in science, and the limited practical application thereof (Díaz et al. 2015). This mismatch is reflected in the ongoing discussion on how the ES concept could be improved, mainstreamed, and operationalized (e.g., Cowling et al. 2008; Daily et al. 2009; Balmford et al. 2011; Bennett et al. 2015; Fischer et al. 2015). This discussion has at least three different strands: (1) the specification of the ES concept itself, (2) the available knowledge on ES for practical implementation of the concept, and (3) best practice for implementation of the concept. There is still vivid debate around the ES concept itself (Norgaard 2010; Sagoff 2011; Bennett et al. 2015; Fischer et al. 2015; Rieb et al. 2017); nevertheless, we want to focus in this paper on the knowledge base needed for practical implementation of the concept.

To successfully transfer knowledge from ES case studies to environmental policies, it is necessary to develop a sound knowledge base with respect to effects of global change on the provisioning of ES. Initiatives such as The Economics of Ecosystems and Biodiversity (TEEB[3]), the economics of land degradation (ELD[4]), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES[5]), and the World Bank's Global Partnership for Wealth Accounting and the Valuation of Ecosystem Services (WAVES[6]) aim at providing such a knowledge base, thereby relying on the quality of results from individual case studies. Seppelt et al. (2011), however, raised concern about the soundness and usefulness of results of many ES studies. They aligned their criticism along four dimensions, biophysical realism, trade-offs, off-side effects and stakeholder involvement, and the limited consideration studies paid to each of these facets. ES-based management requires more than a snapshot quantification. The biophysical quantification of ES has to be dynamic (i.e., able to respond to changing environmental or technical conditions), and to be supplemented by an analysis of synergies/trade-offs between ES, and consideration of export of negative or positive decision effects (so-called off-site effects). Similarly, the social context of ES, the preferences, traditions, realizable management options, and governance structures must be taken into account, typically through elicitation and consultation with local stakeholders and decision-makers. Here, we extended the analysis of Seppelt et al. (2011) by (1) analyzing changes between May 2010 and March 2016, (2) extending the analysis with respect to the relevance and usability of study results for the operationalization of the ES concept in practice, and (3) identifying patterns across clusters of studies with respect to the ES analyzed and study characteristics. The aim of our study is to assess current practice in the field of ES research and to highlight blind spots that affect ES studies. Thereby, we hope to stimulate establishing quality standards for upcoming research, improve the operationalization of the ES concept, and contribute to a robust knowledge base for decision support in environmental management.

Methods and data

We investigated how current ES assessments were conducted in practice, by analyzing papers reporting on such assessments. We focused on case studies, and excluded opinion papers, reviews, or methodological papers without an application of the approach since these describe how things should be done instead of how they were done. Seppelt et al. (2011) reviewed publications found through an ISI Web of Knowledge search of articles with the search phrase "ecosystem service" OR "ecosystem services" OR "ecosystem valuation" in the title published up to 24 May 2010. This search yielded 460 articles, of which 153 case studies were analyzed by Seppelt et al. (2011). We extended their database with a sample of articles selected by the same search phrases published between 25 May 2010 and 31 March 2016. Additionally, we included one marine study before 2010 that was excluded by Seppelt et al. (2011), who focused only on terrestrial systems. The query returned a total of 2101 articles between 25 May 2010 and 31 March 2016, from which we randomly selected 807 articles. The random sampling was stratified by year to ensure a similar relative coverage per year. All 807 articles were read by the authors and analyzed with respect to a set of predefined questions which were used to fill attributes of a database. Three hundred fifty-one of the articles in our sample could be used for the purpose of the analysis.

[1] https://www.cbd.int/
[2] http://ec.europa.eu/environment/nature/biodiversity/comm2006/2020.htm
[3] http://www.teebweb.org/
[4] http://www.eld-initiative.org/
[5] http://www.ipbes.net/
[6] https://www.wavespartnership.org/en

Three hundred twenty-one papers of the 807 had to be excluded because they were review papers or opinion or methodological papers without an application in a study. Eighty-eight papers were excluded because they were off-topic. Other studies were excluded as they were not available to us, were not written in English, or were repeated studies already included in the database. In total, our analysis was therefore based on 504 case studies: 153 for 01/1996–05/2010 and 351 for 06/2010–03/2016—the references of studies used in the sample are provided in the appendix. The original data set from Seppelt et al. (2011) was updated for new attributes referring to relevance and usability, type of stakeholder involvement, and trade-off analysis (see Table S4 for all attributes considered and the supplementary material for the full data base).

We used hierarchical Ward clustering (Legendre and Legendre 2003) to identify patterns across studies with respect to (i) the ES studied (*studies by ES cluster*) and (ii) the indicator values of Table S4 (*studies by indicator cluster*). In addition, we formed clusters of ES that were studied together (*ES cluster*)—here, we clustered the ES and not the studies as before. To avoid the double zero problem, the Jaccard similarity measure (Legendre and Legendre 2003) was used to measure the similarity of studies and ES. The double-zero problem occurs since the absence of an ES category in two studies does not indicate similarity between both studies while the presence of an ES category in both studies indicates similarity. The Gower distance (Gower 1971) was used to describe the dissimilarity between studies based on the values of the different indicators variables of Table S4 (*studies by indicator cluster*). The number of clusters was in all cases decided based on the Mantel correlation between the clustered data for the various cut levels of the dendogram and the raw distance matrix (Legendre and Legendre 2003). Mantel correlation is meant here in its simplest case, i.e., the equivalent of a Pearson r correlation between the values in the two distance matrices.

All analysis was done in R 3.4.1 (R Core Team 2017) using the packages cluster 2.0.6 (Maechler et al. 2017), gclus 1.3.1 (Hurley 2012), gplots 3.0.1 (Warnes et al. 2016), lattice 0.20–35 (Sarkar 2008), RColorBrewer 1.1–2 (Neuwirth 2014), reshape2 1.4.2 (Wickham 2007), sp. 1.2–5 (Pebesma and Bivand 2005), vcd 1.4–3 (Meyer et al. 2016), and vegan 2.4–3 (Oksanen et al. 2017).

Results and discussion

Distribution of case studies over space and across ecosystem service categories

A global operationalization of ES requires case studies of similar quality to be conducted across the world. However, the geographical distribution of ES studies was highly skewed: the top 13 countries in our sample made up for 50% of the total number of case studies, the top 31 for 75% of the total number of case studies (cf. Figure 1). Most ES research was undertaken in the USA, China, and the EU. In recent years, the share of studies conducted in the USA and China has been reduced while the share of studies conducted in countries of the EU has increased (cf. supporting online material)—this is mainly due to an increase in ES studies that cover the EU25 and EU27 countries. The distribution of case studies shows a pattern related to development status (UNDP 2014). From the 31 countries in which 75% of the studies in our sample were conducted, the majority (87%) took place in countries with a very high human development index (HDI). The only exceptions were the People's Republic of China, the Federative Republic of Brazil, and the United Mexican States (countries with a high HDI) as well as the United Republic of Tanzania (low HDI). A quasi-Poisson generalized linear model with a log link identified a significant positive relationship between the number of ES case studies per country and the human development index (HDI, slope at the link: scale 5.7, p value: 5e-05, explained deviance: 29%). This uneven distribution of case studies limits the ability of regional and local decision-makers outside the most studied countries to identify most urgent threats for ES provisioning since the knowledge base is relatively narrow in large parts of the world. Especially worrisome is the lack of studies in low-developed countries in which societies depend much more on ES than in higher developed countries.

Marine systems were clearly underrepresented in the sample: only 21 studies (~ 4%) dealt with marine ES. Eighty-five percent of these marine studies were published in 2014 and 2015, indicating an increasing awareness of the ES community on the importance of this blind spot.

Beside geographical distribution, a representative range of ES should be covered. A comprehensive ecosystem assessment is only possible if a representative set of ES is quantified, not particularly those that are easy to estimate. An uneven coverage of ES categories provides an incomplete picture of the system under study (Baveye 2017) and might lead to sincere shortcomings with respect to conclusions drawn from the results if the uneven coverage is not taken into account.

Coverage of ES categories in research was uneven across time (cf. Figure 4 and supporting online material). So far, research focused on "food provisioning," "climate regulation" (mainly carbon sequestration), "biodiversity and nursery" (mainly species richness indicators), and "opportunities for recreation and tourism." Other ES such as "ornamental species," "biochemical products and medicinal resources," or "spiritual & artistic inspiration" were rarely considered. Possible reasons for this unequal distribution across categories are perceived importance of the ES categories by researchers and/or stakeholders, different research background of study leaders, or financial, logistic, and scientific challenges to

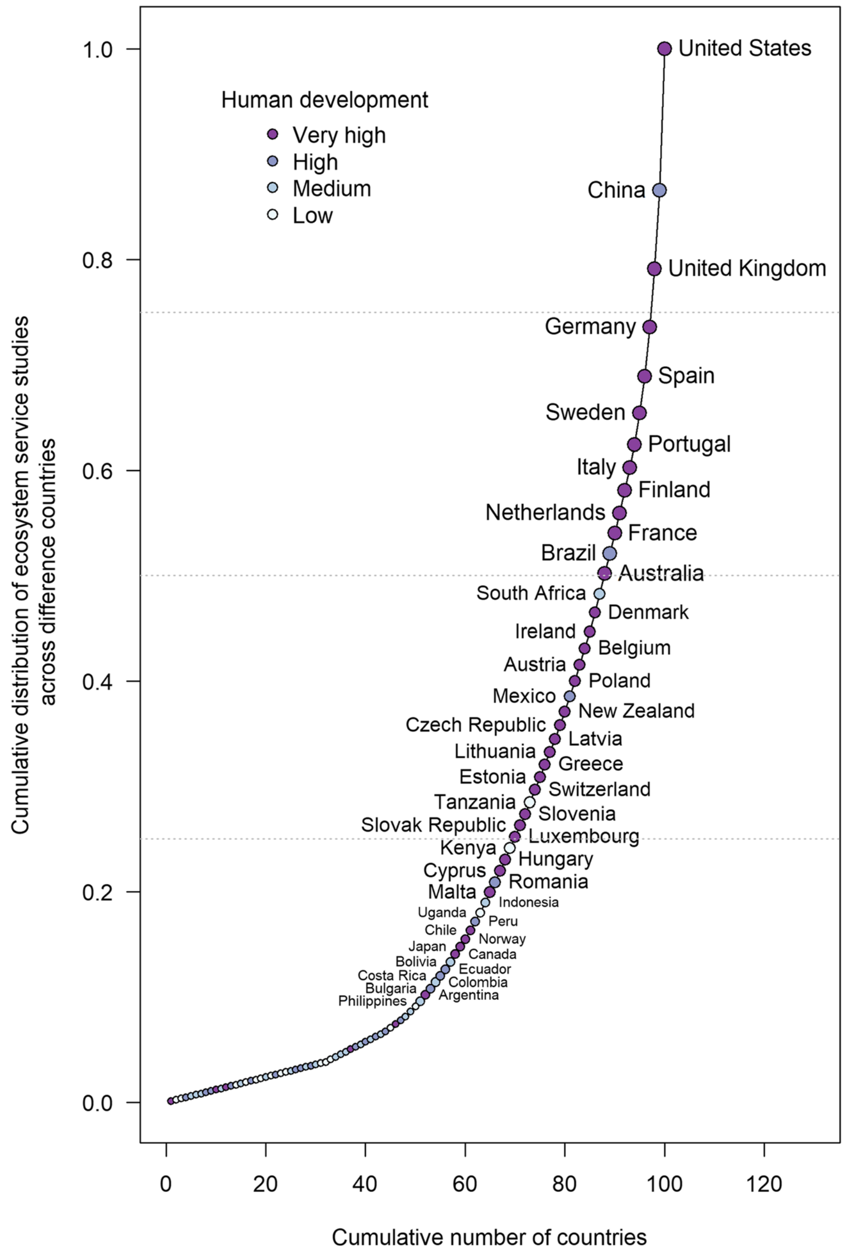

Fig. 1 Cumulative distribution of case studies across countries. The y-axis shows the cumulative sum of case studies by the different countries. The gray horizontal lines indicate the 25th, 50th, and 75th percentile. The top 50 countries with most ecosystem service case studies in the sample are labeled. Development status of the countries is color coded according to the human development index (HDI) for 2015 (http://hdr.undp.org/). Global studies and marine studies in the open ocean were excluded here. Short country names as used by the World Bank are provided instead of official country names to improve readability

quantify some ES in the field. Potentially, the uneven distribution might simply reflect the importance of the different ES categories for human well-being or at least their perception by scientists or stakeholders. There is good reason to believe that, e.g., ornamental species are in most cases of less concern than food security or avoidance of hazardous events. However, several studies (e.g., Vieira et al. 2014; Jacobs et al. 2015; Oleson et al. 2015; Tilliger et al. 2015) have stressed the importance of other cultural ecosystem services than recreation and tourism for the public, which is not in line with the reflection of these services across case studies.

The cluster analysis based on the ES that were studied in the case studies revealed 11 clusters (cf. Table 1 and Fig. 2). These clusters of studies focusing on similar combinations of ES categories could represent different research communities inside of the ES community. Indicator values from Table S4 differ clearly between the studies by ES clusters. The frequency of studies in each of those studies by ES cluster differed by development status (measured by the HDI) of the countries the study was located in (cf. Figure 3). Studies from cluster “Tourism and recreation,” “soil services,” “water quality and habitat provisioning,” and “water quality, food provisioning, air quality” were mainly conducted in countries with the highest HDI. This implies that information on these ES clusters is even more biased than the uneven distribution of studies across countries suggests. Studies in cluster “climate regulation, soil retention, water quality, habitat provisioning” have been studied in countries with an HDI of around 0.7 (which

Table 1 Studies by ecosystem service study cluster: cluster of case studies based on the ecosystem services analyzed. In quotation marks, labels are given that are used when referring to the clusters in the text

Cluster number	Ecosystem services mainly studied by case studies in the cluster	Number of studies in cluster 1
1	"Forest services"	40
2	"Food and water provisioning"	94
3	"Diversity of Ecosystem services"	163
4	"Climate regulation, water provisioning, air quality"	70
	Climate regulation, water provisioning, to a lower degree air quality	
5	"Habitat provisioning and other services"	209
	Habitat provisioning, food provisioning, timber and fiber provisioning, climate regulation, tourism and recreation potential, hazard mitigation	
6	"Climate regulation, soil retention, water quality, habitat provisioning"	58
7	"Biological regulation" (pollination and biological pest control)	47
8	"Tourism and recreation"	76
	Tourism and recreation together with other cultural services	
9	"Soil services"	22
	Soil services, to a lower degree food provisioning and biological regulation	
10	"Water quality and habitat provisioning"	26
	Water quality and habitat provisioning (mostly in aquatic systems)	
11	"Water quality, food provisioning, air quality"	31
	Water quality, to a lower degree food and air quality	

mainly represents ES studies in China). Studies in countries with a low HDI focused on the provisioning of food and water as well as forest services. This is not too surprising, given the stronger dependency of relatively large parts of the population of these countries on these ES. However, it indicates that trade-off analysis in these countries might overlook other important services.

Social-ecological validity

Social-ecological validity means that measurements, modeling and monitoring of ecosystem functions, and the social dimension related to ES supply and demand are close to the phenomenon measured and not abstract and unsubstantiated proxy indices (Seppelt et al. 2011; Fischer et al. 2015). We used six indicators for social-ecological validity: data source, model type, indicator used, system boundary, uncertainty, and validation (cf. Figure 5, Table S4).

A third of the case studies (34%) used look-up tables (cf. Figure 5a)—often together with simple land-cover classes as proxies, which further limits the reliability of a look-up table approach. The rest of the case studies used statistical models (29%), process models (13%), and GIS models (12%). Modeling approaches differ with respect to their ability to account for feedbacks, non-linear effects (Grêt-Regamey et al. 2014), and spatial and temporal variability (Dale and Polasky 2007; Eigenbrod et al. 2010a) and to predict system behavior under changing boundary conditions (Bennett et al. 2015; Rieb et al. 2017). There is no general best modeling approach—depending on the research question, purpose of study, and the available data, different approaches might be preferred (Bennett et al. 2012). Given the complexity of human-environment interactions and the importance of feedbacks in ecosystem processes as well as in social processes, it has to be assumed that models that ignore such feedbacks are limited in their ability to correctly predict changes in both ES demand and supply. Especially, the ability to predict non-linear system behavior is limited in models without feedbacks. Look-up table approaches and most GIS (geographic information system) approaches are not able to incorporate feedbacks and are therefore limited with respect to social-ecological validity. Substantial shortcomings associated with the use of look-up tables have been reported by Konarska (2002) and Eigenbrod et al. (2010b). The types of model categories used differed across the different ecosystem service categories (cf. Figure 4): process models and statistical models were seldom used in the assessment of cultural services and the less frequently assessed provisioning services such as ornamental species or genetic resources. The application of the different methods was unequally distributed across the ES study clusters from Table 1. The use of look-up tables was most widespread in cluster "diversity of ES" (52% of studies) in which studies analyzed many different ES together. The use of GIS models was highest in cluster "climate regulation, water provisioning, air quality" (21%) and "habitat provisioning and other services" (21%), statistical models were used most frequently in clusters "biological regulation" (63%) and "soil services" (62%), while process models were used most often in clusters "climate regulation, soil retention, water quality, habitat provisioning" (27%), "soil services" (25%), and

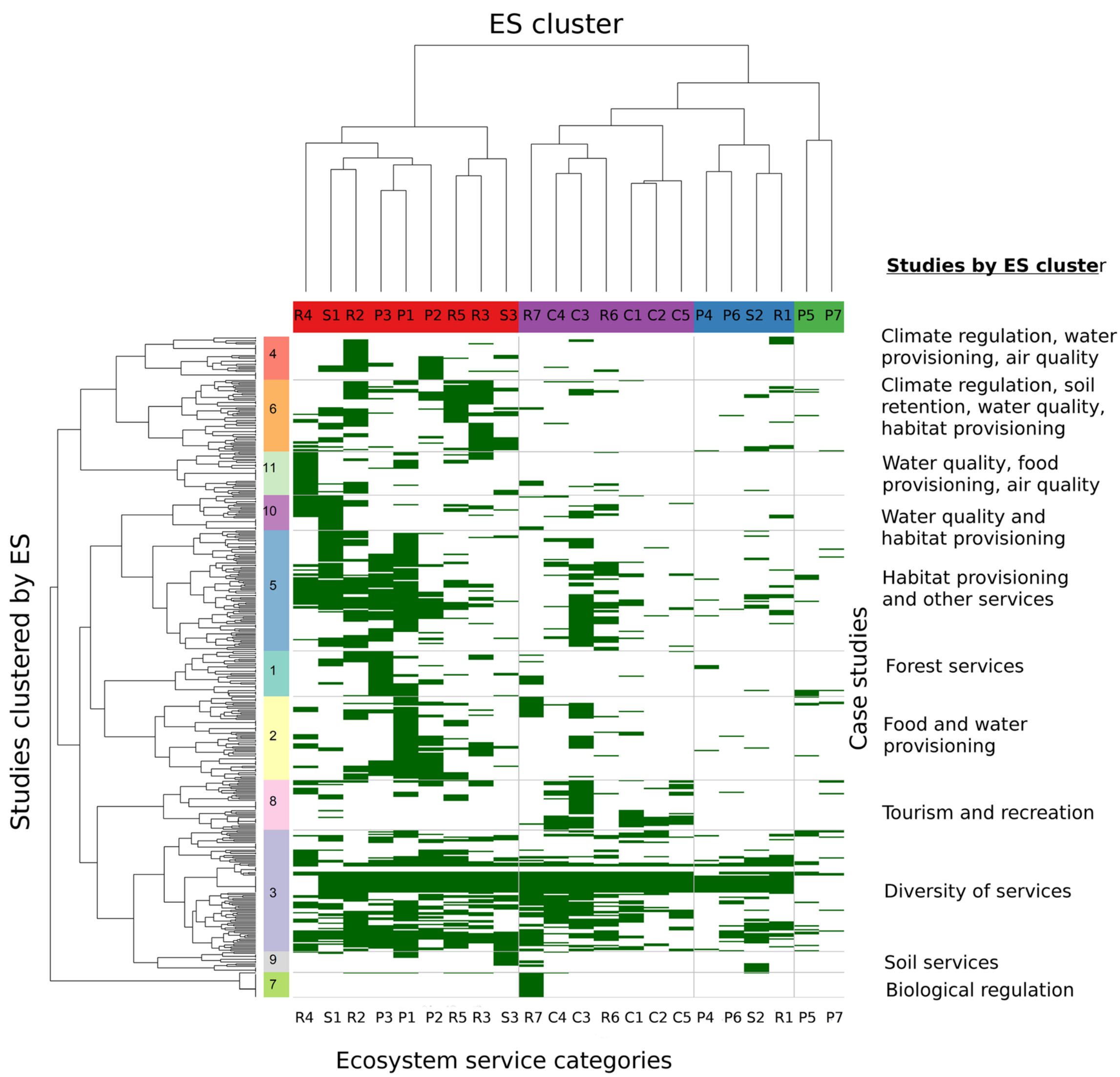

Fig. 2 Case studies clustered with respect to ecosystem services (ES) studied. The studies have been ordered to reflect similarities between ES (ES cluster) at the x-axis and similarities between studies (studies by ecosystem services cluster) at the y-axis. Membership of studies to the different types of clusters is indicated by the colors at the margin of the plot as well as by the two dendrograms. The numbers shown at the y-axis correspond to the cluster numbers presented in Table 1—cluster labels are provided at the far right. The central matrix of the figure indicates if an ecosystem service has been studied (green) in a case study or not (white). Horizontal and vertical lines have been added to simplify the identification of clusters in the central matrix. The ecosystem service categories shown at the x-axis are as follows (in order of appearance): R4: water quality regulation; S1: biodiversity and nursery; R2: climate regulation; P3: provisioning of fiber, fuel, and other organic raw materials; P1: food provisioning; P2: freshwater provisioning; R5: soil retention and erosion protection; R3: water quantity regulation; S3: nutrient cycling; R7: biological regulation; C4: esthetic: appreciation of natural scenery; C3: opportunities for tourism and recreational activities; R6: natural hazard mitigation/ disturbance regulation; C1: cultural heritage and identity; C2: spiritual and artistic inspiration; C5: science and educational services; P4: provisioning of inorganic resources; P6: provisioning of genetic material; S2: soil formation; R1: air quality regulation; P5: provisioning of biochemical products and medicinal resources; P7: provisioning of ornamental species

"water quality, food, air quality" (27%). There was no obvious pattern that the use of specific models was biased towards higher or lower human development status of the country where the case study was located. Use of process models and of statistical models was a bit more frequent in countries of highest human development status, and statistical models were observed more frequently in least developed countries. However, it seems that issues such as data availability did not hinder the use of process models in countries of lower human development status.

Frequently, neither results obtained from ES models were validated (86%) nor were their uncertainties quantified (49%) (cf. Figure 5b, c). Assuming that all models are wrong but some are useful (Box 1976), the usefulness of the models for the purpose of a study needs to be demonstrated. The only way to estimate the reliability of any type of model is a test

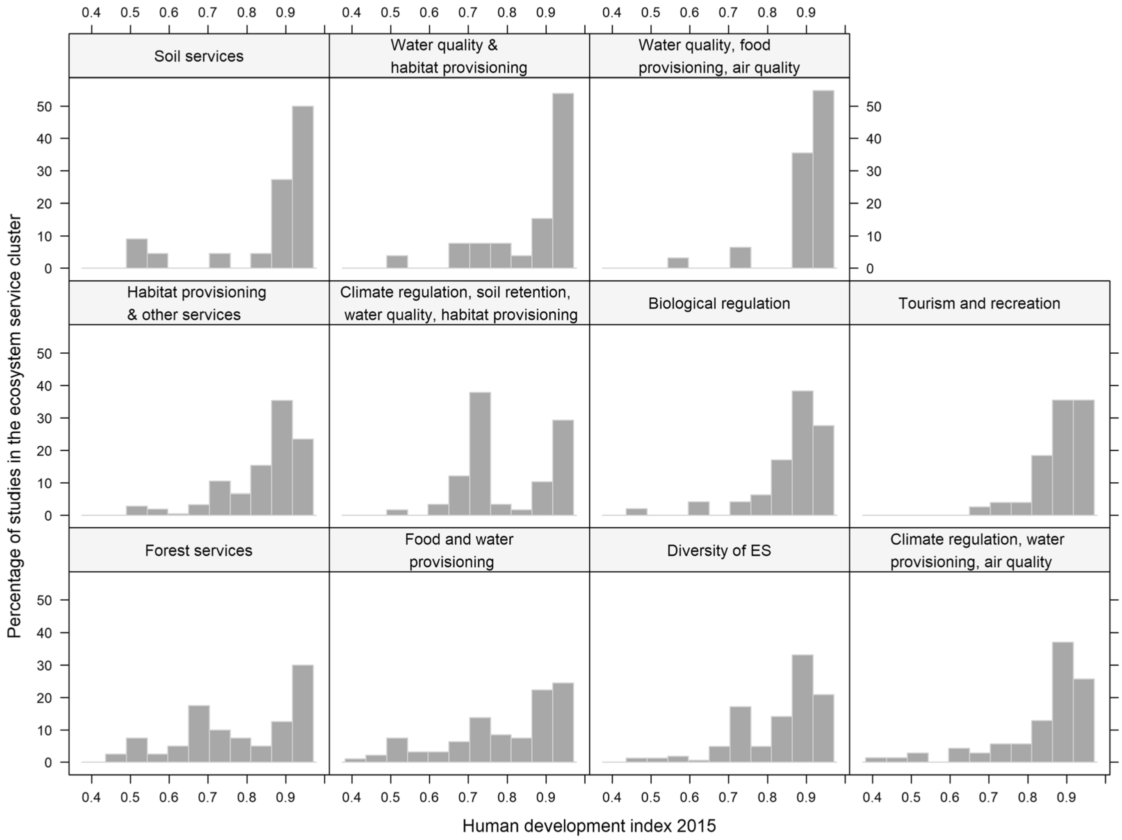

Fig. 3 Distribution of studies in the 11 ecosystem service study clusters across the human development index (HDI). Each panel shows the distribution of studies in the cluster across the HDI—percentages add up to 100 for each panel

against independent data and an analysis of the uncertainty of model predictions (Kirchner et al. 1996; Jakeman et al. 2006; Dormann et al. 2008; Bennett et al. 2012; Laniak et al. 2013; Hou et al. 2013; Hamel and Bryant 2017). Such a validation of results with independent data and an assessment of the attached uncertainty is a necessary prerequisite to judge conclusions drawn from model results. It should be mandatory to discuss results in the light of the quantified uncertainties and help thereby decision-makers to decide if results are reliable enough to support the decision to be made. Uncertainties were most often quantitatively assessed in statistical models (40%), followed by process-based models (31%), GIS models (25%), and look-up-table approaches (22%). Consideration of uncertainty differed substantially across the ES study clusters from Table 1: studies in cluster “tourism and recreation” showed the lowest percentage of studies that considered uncertainty quantitatively (16%), while clusters “soil services”, “biological regulation” and “forest services” contained the highest number of studies that considered uncertainty quantitatively (50%, 47%, and 40%, respectively). The highest share of studies using validation was found in clusters “forest services” (34%) and “water quality, food provisioning, air quality” (25%) and lowest share in “biological regulation” (5%).

Secondary data have been used exclusively by 56% of the studies (cf. Figure 5d). Secondary data might not be well suited for the purpose of the ES study since they have in most cases been sampled and potentially aggregated for a different purpose. Therefore, spatial and temporal extent and resolution as well as thematic resolution are often suboptimal. Relying on suboptimal data to assess the provisioning of ES or the demand for those ES may hamper the social-ecological validity of the results. Interestingly, a strong difference between demand-side and supply-side studies was observed: only 16% of the demand-side studies relied on secondary data compared to 60% of the supply-side studies. Use of primary data was highest in ES study clusters “biological regulation” (68%), “tourism and recreation” (80%), “soil services” (80%), and “water quality and habitat provisioning” (62%), and lowest in ES study clusters “habitat provisioning and other services” (25%) and “climate regulation, soil retention, water quality, habitat provisioning” (23%).

Studies split relatively even between administrative (39%) and biophysical (44%) system boundaries. The social-ecological validity of a study depends on an appropriate system boundary definition. Depending on whether the focus is on the supply or the demand side of ES, both a biophysical and an administrative system boundary definition might be appropriate. Twelve percent of the studies used a combination of demand- and supply-side-system-boundary definitions. These were studies in which either demand or supply was

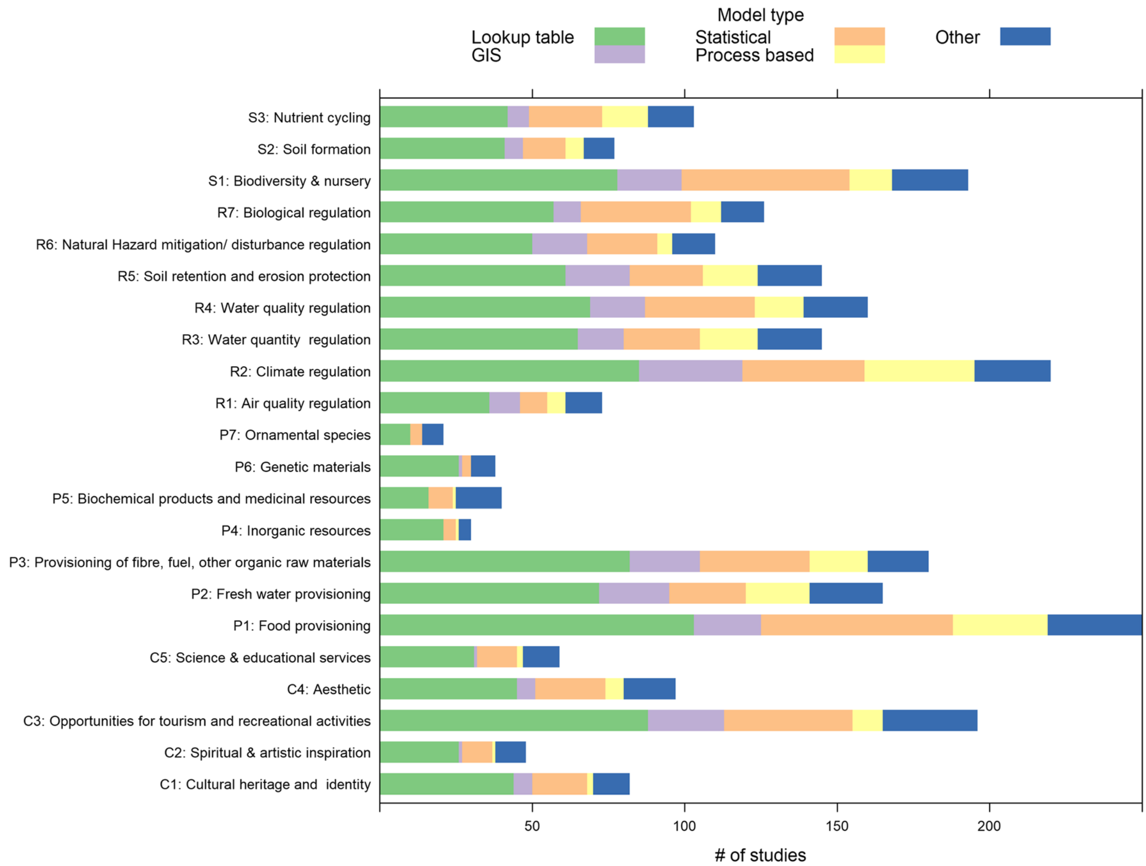

Fig. 4 Number of studies in which the different ecosystem service categories have been studied. In addition, the main model types used to quantify the service are indicated. The category "other" includes meta-models and conceptual models

assessed for different spatial units or where different system boundaries for different ES were used. Examples are administrative units for food provisioning based on agricultural statistics and water provisioning based on watershed boundaries. Over time, the share of studies in our sample that used purely biophysical system definitions has decreased while the share that defined the system boundary on both administrative statistics and biophysical boundaries has increased (cf. Figure 6). Demand-side studies had a higher share of administrative system boundary definitions (52%), while supply-side studies had a slightly higher share of biophysical system boundaries such as watersheds (47%). While it is not possible to classify all those cases as problematic, it raises the question if system boundaries for ES assessments are drawn for practical purposes such as data availability and to what extent this influences the validity of results. Studies that relied on secondary data used administrative system boundaries more frequently (44%) compared to those that used primary data (31%). Administrative boundaries were encountered with the highest proportion in ES study clusters "climate regulation, water provisioning, air quality" (48%) and "habitat provisioning and other services" (49%) and biophysical boundaries in clusters "water quality and habitat provisioning" (63%) and "water quality, food and air quality "(70%).

Our sample further indicates that monetization of ES was performed in a third of studies: most studies used biophysical indicators (56%), followed by monetary indicators (32%) and rankings (15%). Indicator use has changed clearly over time: studies applying monetary indicators have decreased, studies using biophysical indicators have been dominant since 2011, and further since 2015 studies applying ranking indicators have gained importance (cf. Figure 6). Biophysical indicators were most frequently used in ES study clusters "biological regulation" (89%) and "water quality, food provisioning, air quality" (75%), ranking indicators most frequently in ES study clusters "tourism and recreation" (36%), "diversity of ES" (23%), and "soil services" (25%), while monetary indicators were used most frequently in ES study clusters "diversity of ES" (39%) and "habitat provisioning and other services" (35%). Monetary indicators were used slightly more often in countries with an HDI around 0.7 (e.g., China, which contributed the majority of studies in this HDI range).

Trade-offs

Ecosystem services can be expected to have mutual relations (Bennett et al. 2009). These relationships between ES can take the form of trade-offs, synergies, or no-effects (Lee and

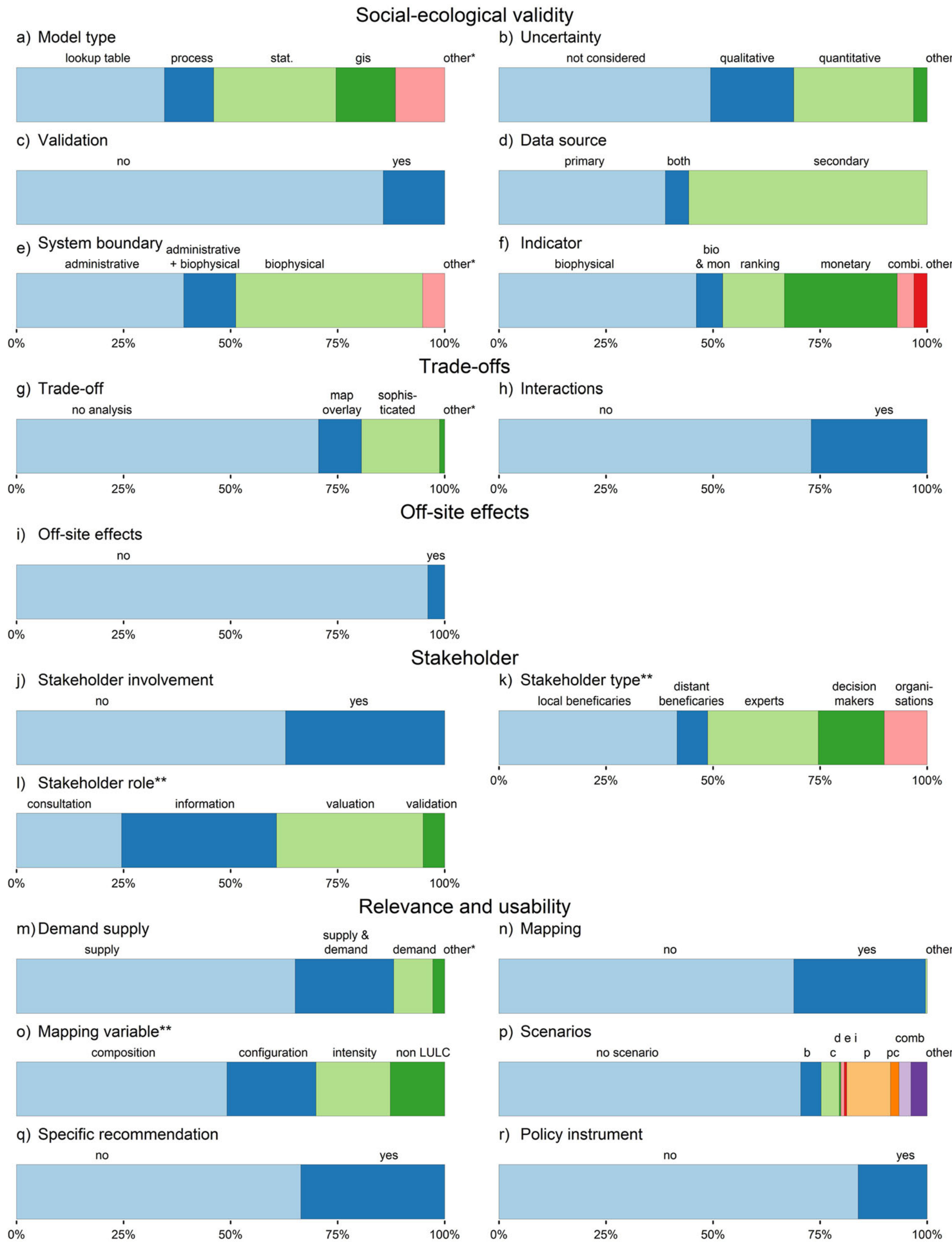

Fig. 5 Distribution of the studies for different indicator levels. Colors solely serve to separate categories. For the indicator "scenarios" the following types have been distinguished: b, behavioral changes; c, climate change; d, demographic changes; e, economic changes; i, invasive species; p, policy changes; pc, policy changes and climate change combined; comb, combination of several scenario types in the same study. *"Other" refers to cases in which insufficient information was given in the article to make a clear assignment. **For stakeholder role and stakeholder type, the percentage refers only to the number of studies that involved stakeholders. For mapping variable, the percentages refer only to the studies that mapped ecosystem services. Single studies could use several mapping variable categories, stakeholder types, or role categories

Lautenbach 2016; Cord et al. 2017a). Analyzing the relationships between ES or between management actions and ES is a crucial step for identifying promising management pathways (Bennett et al. 2009; Mouchet et al. 2014; Lee and Lautenbach 2016). The strength and even the direction of the relation might change in space and time, and can be triggered by changing management strategies. In addition, the relations between ES can be non-linear and comprise critical thresholds (Koch et al. 2009; Sabatier et al. 2013). Trade-offs and synergies can originate from common drivers or from interactions between ES (Bennett et al. 2009). Examples for interactions are the beneficial effects of forests on crop-pest control (Gagic et al. 2011) or pollination (Ricketts et al. 2008) and the positive effects of mosaics of land use on biodiversity (Tscharntke et al. 2012b). While not all ES are linked by interactions, these are often of high relevance in multifunctional landscapes (Raffaelli and White 2013)—ignoring them in environmental management likely leads to suboptimal decisions.

The majority of studies did not analyze trade-offs (70%), 10% of the studies used a simple map overlay to assess bundles of ES or trade-offs between them, while 19% of the studies followed a more complex approach to analyze trade-offs (cf. Figure 5g). Approaches followed in trade-off analysis included optimization approaches (Chan et al. 2006), the analysis of the trade-offs of different scenarios or management

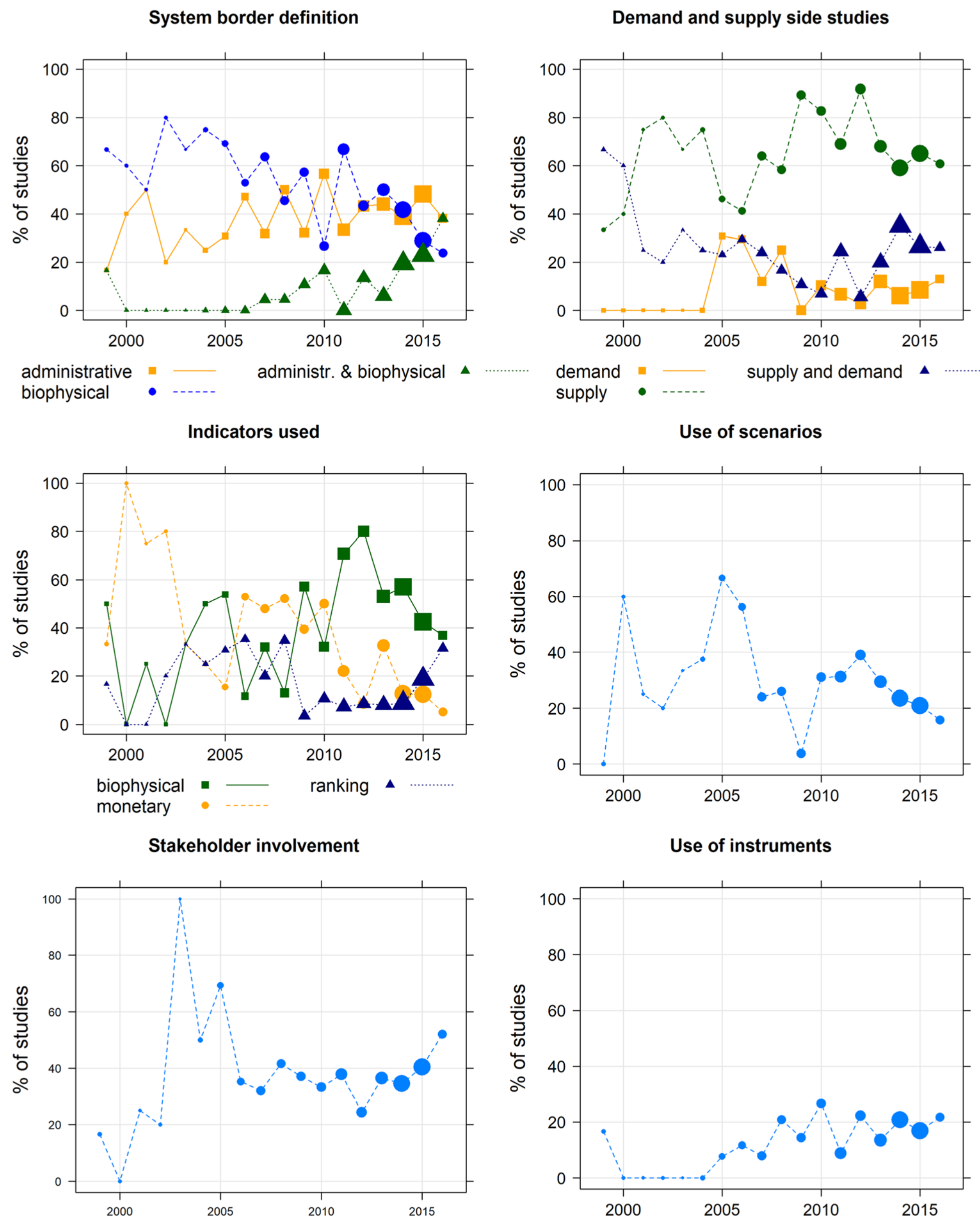

Fig. 6 Percentage of studies in a year that belonged to a specific attribute category. Values before 1999 have been dropped due to the low case numbers. Point size indicates the number of studies per year in the sample. The supporting online material contains additional time series plots. For indicators used, numbers do not add up to 100% since some studies used several indicators. Ranking indicators refer to the ranking of ecosystem services or of scenario outcomes by stakeholders or experts. For stakeholder involvement, values are relative to the number of studies per year that involved stakeholders—point sizes reflect here the number of studies per year that involved stakeholders

alternatives (Cordier et al. 2011; Grêt-Regamey et al. 2013; Kirchner et al. 2015), or the statistical analysis of survey data (Maskell et al. 2013) including trait-based analysis (Lavorel et al. 2011). Most studies that considered trade-offs focused on either trade-offs between a limited set of management actions (54% of trade-off studies) or trade-offs in space (49%)—15% of the trade-off studies considered both trade-offs between management options and trade-off in space. Eighteen percent of the trade-off studies assessed trade-offs between beneficiaries, and 11% assessed trade-offs in time. Bundles between ES were only addressed by 4% of the trade-off studies, 5% used prioritization approaches to identify where protection of ES would be of most concern, and 2% of the trade-off studies assessed trade-offs based on the potential of the system to provide ES based on a Pareto frontier approach (Seppelt et al. 2013). The share of studies that did not consider trade-offs was highest in the ES study clusters "biological regulation" (89%), "tourism and recreation" (84%), "soil services" (81%), and "diversity of ES" (78%). Trade-off analysis by map overlay had the highest share in ES study cluster "climate regulation, water quality, air quality" (22%), and more sophisticated trade-off analysis had the highest share in ES study clusters "water quality, food provisioning, air quality" (30%), "food and water" (30%), "water quality and habitat provisioning" (26%), "forest services" (24%), and "climate regulation, soil retention, water quality, habitat provisioning" (24%).

Interactions between ES were largely and constantly ignored across time: only 26% of the studies considered interactions between ES (cf. Figure 5h). Even of the studies that looked at many different ES categories, very few actually considered interactions between services. Considering interactions was expectedly rare among studies that used look-up tables (17%) and GIS models (14%), while it was more common in studies that used statistical approaches (40%) or process models (49%). In the case of process models, the interactions were most frequently built into the model. ES study cluster “biological regulation” did not consider interactions between ES at all which was related to the fact that the majority of studies in this cluster only analyzed a single ES (95%). Low frequency of studies with interactions between ES was also observed in ES study clusters “climate regulation, water quality, air quality” (12%), and “tourism and recreation” (13%).

The majority of studies that did not analyze trade-offs did also not incorporate interactions between ES (92%). This was particularly the case for trade-off analysis based on simple map overlays (75%), while 75% of studies that applied a more sophisticated trade-off analysis incorporated interactions between services. Examples of studies that used sophisticated trade-off approaches, without considering interactions between ES, were studies that analyzed trade-offs between different management options but based on assessment models that did not incorporate interactions between ES(e.g., van Wilgen et al. 1998; Chan et al. 2006; Jacobs et al. 2015; Hossain and Dearing 2016), studies that analyzed trade-offs in ES demand (e.g., Buckley et al. 2012; Martín-López et al. 2012), or studies that analyzed trade-offs based on field data (e.g., Jessop et al. 2015).

Off-site effects

Only 20 studies (4%) incorporated off-site-effects (also called tele-coupling, teleconnections, or off-stage ES burden: Seppelt et al. 2011; Liu et al. 2013, 2016; Yu et al. 2013; Pascual et al. 2017; cf. Figure 5i). From the perspective of global sustainability, it is important that place-based ES assessments do not overlook effects on distant social-ecological systems (Pascual et al. 2017). The decision to avoid local damage of ES might lead to a replacement of damaging activities to distant land systems with overall unsustainable impacts (Kissinger et al. 2011; Schröter et al. 2018). Taking these off-site effects into account, even in lump-sum, approaches such as water or carbon footprints might reveal underlying causes of developments and enable a better stakeholder selection as well as an improved system understanding (Hilborn 2013). Without consideration of such off-site effects, there is substantial risk for the spatial spillover rebound effect (Maestre Andrés et al. 2012), insofar as policies intending to protect one type of biodiversity or ES in a certain area have even stronger negative impacts on such biodiversity or ES in another region. We believe that international agreements on ES need to be wise about the re-distribution of impacts in globally connected markets and that ES studies accordingly need to provide off-site analyses for their study systems. So far, we simply do not know how relevant off-site effects are in ES assessments—we can only suppose based on research from the land science community and the few published ES studies that incorporating this topic that this is a highly relevant blind spot of ES research.

Stakeholder involvement

There is evidence that stakeholder participation can enhance the quality of environmental decisions through the input of more comprehensive information (Reed 2008). Further reasons for engaging stakeholders in research include the gaining of knowledge from those most deeply connected to a particular resource or issue or community, achieving buy-in by those most likely to be affected by the research results, building stronger connections between science, policy, and society, and ensuring that research addresses real-world needs (Durham et al. 2014). The need to engage stakeholders in quantification of ES is particularly important, as ES inherently involve people whose (current) preferences and experiences define the (current) benefits of nature’s services. Frequently, perception of ES differs between scientists and stakeholders as well as between different stakeholder groups (e.g., Hicks et al. 2013). Förster et al. (2015) suggest that ES assessments require a co-design approach to adapt to decision makers’ needs, and adjust the assessment process accordingly for providing the relevant information for decision-making.

Stakeholders were involved in 37% of the case studies in the sample (cf. Figure 5j). Beside two peaks in 2003 and 2005, this share has been relatively stable over time (cf. Figure 6). Stakeholders were more frequently involved in studies investigating the demand for ES (89%) than the supply side (26%). Stakeholder involvement was highest in ES study cluster “tourism and recreation” (63%) and lowest in ES cluster “biological regulation” (5%). It was also low in ES study cluster “climate regulation, soil retention, water quality, habitat provisioning” (20%) and “soil services” (19%).

The type of stakeholders involved differed among studies (cf. Figure 5k). Overall, local beneficiaries were included most often (42% of studies), while distant beneficiaries were almost never included (7%). Experts as stakeholders were favored (26%) over decision-makers (15%) and organizations (10%). Most studies included only one type of stakeholders (53%), 30% included two types of stakeholders, 12% included three, and 4% included four stakeholder categories.

Additionally, the role of stakeholders in the ES assessment varied between studies (cf. Figure 5l). Thirty-six percent of the studies that involved stakeholders used them in an informing role where stakeholders were asked to provide additional information about ES (that were already chosen), e.g., about the use of ES, location of ES, or biophysical underpinning of ES.

In 34% of the studies in our sample, stakeholders had a valuing role where they were asked to help to prioritize ES or bundles of ES, or scenarios that were part of the study. In 24% of studies, stakeholders had a consulting role and were asked beforehand to help identify relevant ES and/or beneficiaries that should be included in the study. Only 5% of studies involved them in a validation role where stakeholders were asked to validate ES assessment methods or outcomes. While an informing role was most important at the beginning of the study period, this changed in 2012 and 2014–2016—in these years, more studies involved stakeholder in a valuing role (cf. supporting online material).

Relevance and usability

While studies focusing on single aspects of ES assessments can be of importance for decision-making and management, the promise of ES research has been the integrated assessment of the provisioning of multiple ES and societal needs (MA 2005). Decision-making and environmental management that aims to overcome sectoral perspectives by integrating the manifold perspectives of feedbacks in coupled socio-environmental systems would seriously profit from the comprehensive integration of ES supply, the demand for ES, and policy options affecting demand and/or supply for ES. Goods and services provided by ecosystems have to reach their potential users—otherwise, the benefits cannot be realized. A complete ES study should therefore aim at assessing both demand and supply for ES (Orenstein and Groner 2014; Wei et al. 2017). While the supply side can be measured as biophysical indicators, social demand for ES can be valued using economic valuation techniques in real or hypothetical markets (Turner et al. 2010; Bateman et al. 2011), or based on non-monetary indicators such as the assessment of people's perceptions of the importance of different services (Walz et al. 2016).

The majority of ES case studies (70%) focused on the supply side of ES, while 12% studied only the demand side of ES, and 18% looked at aspects of both sides at the same time (cf. Figure 5m). The share of purely demand side studies in our sample has been declining from 2008 onwards while combined supply and demand side studies showed an increasing tendency since 2010 (cf. Figure 6). Following Wolff et al. (2015), it seems that we need to improve the understanding of the demand for ES. The focus on the supply side of ES was lowest in the ES study cluster "tourism and recreation" (50%) and ES study cluster "habitat provisioning and other services" (61%). It was highest in ES study clusters "climate regulation, soil retention, water quality, habitat provisioning" (76%), "soil services" (75%), "biological regulation" (74%), "water quality, food, air quality" (73%), and "climate regulation, water quality" (73%). The highest share of studies that considered both demand and supply was observed in ES study cluster "habitat provisioning and other services" (32%) and lowest in "soil services" (12%).

Mapping of ES is considered an important information tool for policy support (Maes et al. 2012a, b; Verhagen et al. 2014). Mapping of ES provides an easily understandable overview about the spatial heterogeneity of service provisioning or service demand, which is essential information for spatial planning. A rich diversity of approaches is available to map ES, ranging from proxy-based approaches over phenomenological, niche-based, and trait-based approaches to full process–based mapping approaches (Lavorel et al. 2017a). Their representation of uncertainty is underdeveloped, however. In total, 31% of the studies mapped ES (cf. Figure 5n). Land use composition was used by 49% of these studies, management intensity was used by 17%, and landscape configuration was used by 20% of these studies (cf. Figure 5o). Non-land use–related indicators—such as terrain or soil type—were used by 13% of the studies to map ES, in most cases in addition to land use–related indicators for example by mapping hydrological services or erosion control. Fifteen percent of the studies in the sample that mapped ES included all three components of land use, while 33% of these studies relied only on land use composition indicators, ignoring the serious shortcomings of simple land use composition–based proxies for mapping ES (Eigenbrod et al. 2010a, b). Mapping was observed most frequently in the ES study clusters "habitat provisioning and other services" (48%) and "climate regulation, soil retention, water quality, habitat provisioning" (47%) and lowest in "soil services" (6%) and "water quality, food, air quality" (9%). Mapping based purely on land use composition indicators was observed most frequently in the ES study clusters "diversity of ES" (30%), "climate regulation, water provisioning, air quality" (31%), "habitat provisioning and other services" (32%), and "water quality and habitat provisioning" (30%). The share of studies that used additional indicators beside land use composition to map ES was highest in "climate regulation, soil retention, water quality, habitat provisioning "(51%), "forest services" (57%), and "food and water provisioning" (52%).

Land systems are dynamic by nature and change over time. Similarly, demand and supply for ES are likely to change with time. A sustainable management of land systems has therefore to consider potential future changes of the system (McCauley 2006). Changes in the demand and/or the supply side of ES have to be expected due to various reasons, such as land use and climate change (e.g., Prather et al. 2013; Polce et al. 2016), demographic changes (shrinking/growing population, aging of population), behavioral changes (e.g., changes in consumption behavior: Tscharntke et al. 2012a), economic development (e.g., the financial crisis and its effects on resource availability for environmental conservation), or policy changes (e.g., bioenergy: Banse et al. 2008; Landis et al. 2008; Campbell and Doswald 2009; Lautenbach et al. 2013).

However, the majority of studies (70%) only focused on the current state and did not consider changes to ES over time (cf. Figure 5p). ES assessments were treated mainly as a static analysis without considering changes on both the demand and the supply side of ES. The use of scenarios in the case studies has been widely fluctuating between 67% in 2005 and 4% in 2009—the use of scenarios in studies after 2012 has decreased from 40 to 21% in 2015 (cf. Figure 6). Scenarios were used most frequently in the ES study clusters “food and water provisioning” (44%), “climate regulation, water provisioning, air quality” (37%), “habitat provisioning and other services” (37%), “water quality, food, air quality” (36%), and “forest services” (34%) – they were seldom used in “biological regulation” (5%).

Scientific results may translate into specific recommendations on best practice. Only a third of the studies (33%) did provide any kind of specific recommendation (cf. Figure 5q). This is in line with the results from Laurans et al. (2013), who looked specifically at ES valuation studies and found that specific recommendations were seldom derived from economic valuations. Studies that involved stakeholders provided specific recommendations more frequently than studies that did not involve stakeholders (41% vs. 29%). Specific recommendations were also more frequently provided in demand side than in supply side studies (44% vs. 28%). Studies that mapped ES provided also more frequently specific recommendation than did studies that did not map ES (40% vs. 30%). Specific recommendations were provided most frequently in ES study clusters “climate regulation, water provisioning, air quality” (58%) and “habitat provisioning and other services” (45%) and seldom in “soil services” (19%), “water quality and habitat provisioning” (19%), “diversity of ES” (22%), and “water quality, food and air quality” (22%).

The majority of the case studies (84%) did not consider any policy instrument (cf. Figure 5r). The relatively broad category of payments for ES (PES) was mentioned mostly (in 14 studies). Studies that assess the effects of policy instruments on ES provide highly relevant information for decision-makers and stakeholders. The lack of studies on specific policy instrument clearly hampers transferring the ES concept from science to policy and practice. However, the consideration of policy instruments in ES studies has increased from 2007 onwards (cf. Figure 6). The consideration of policy instruments was highest in ES study cluster “climate regulation, water provisioning, air quality” (21%) and lowest in “forest services” (11%), “biological regulation” (11%), and “water quality and habitat provisioning” (11%).

Studies clustered by indicators

In addition to the clusters of case studies based on similarity between the ES studied by the case studies, we clustered studies based on their similarity in the indicator values from Table S4. We identified four clusters of studies, in the following denoted as *studies by indicator clusters 1–4* (Table 2). Cluster 1 had a relatively high share of process model studies (44%) together with studies employing statistical models (26%) or look-up table approaches (18%), cluster 2 was dominated by look-up table studies (79%), cluster 3 had a high share of GIS models (52%), and cluster 4 a high share of statistical models (79%). A large share of the studies in the studies by indicator cluster 2 belonged to the cluster “diversity of ES” for blind spot indicator cluster 3 (Table 2). The distribution of studies in the clusters did not vary by development status of the countries the studies were located. The case studies in indicator cluster 1 considered interactions between ES in 63% of the time, used scenarios in 88% of the time, and focused on the supply side in only 51% of the studies. Studies in the indicator cluster 2 evenly used the different types of indicators, considered interactions between ES only in 12% of the case studies, and assessed trade-off less frequently. The case studies in the indicator cluster 3 frequently assessed uncertainty quantitatively (40%), frequently were supply-side studies (73%), involved stakeholders less frequently (23%), mapped ES frequently (88%), and dominantly used biophysical indicators (73%). Cluster 4 had the lowest use of secondary data (14%), a low use of scenarios (12%), and mapped ES only seldom (5%)—quantitative assessment of uncertainty was frequent (35%).

How to overcome the blind spots?

Many of the blind spots are related to decisions made during the design of the studies. To overcome the blind spots identified here, careful experimental design is of key importance. The term experimental design is used here in a broader sense including not only the design of field experiments but also the design of scenarios, stakeholder integration, and simulation experiments. For example, it is important to consider how to reach specific recommendations at the experimental design phase, c.f. Förster et al. (2015). A best practice example is the analysis by Kirchner et al. (2015), who demonstrate how stakeholder involvement, model integration, trade-off analysis, scenario analysis, and optimization approaches can be combined to provide recommendations for improved targeting of agri-environmental schemes to provide a more balanced and efficient supply of ES and foster rural development. If no policy instruments are tested, thematic, temporal, and spatial resolution, extent and grain not chosen in accordance with decision-makers’ requirements then specific policy recommendation cannot be derived or will fail. Borner et al. (2007) show how a farm-level bio-economic model can be used in combination with scenarios and a trade-off analysis to derive recommendation with respect to the implications of policy instruments. Similarly, it is essential to consider the dynamic nature of social-ecological systems—e.g., by

Table 2 Main properties of the studies by indicator clusters. ES = Ecosystem services.

Indicator	Cluster 1 ($n = 89$)	Cluster 2 ($n = 198$)	Cluster 3 ($n = 102$)	Cluster 4 ($n = 116$)
Main model type used	Process (44%)	Look-up table (79%)	GIS (52%)	Statistical (79%)
Specific recommendation given	39%	27%	40%	34%
Quantitative uncertainty assessment	24%	20%	40%	35%
Stakeholder involvement	41%	44%	23%	36%
Interaction between ES	63%	12%	22%	29%
Dominant study by ES cluster	"Habitat provisioning and other services" (26%), "food and water provisioning" (19%)	"Diversity of ES" (29%), "habitat provisioning and other services" (18%)	"Habitat provisioning and other services" (25%), "climate regulation, soil retention, water quality, habitat provisioning" (23%)	–
Purely supply side studies	51%	66%	73%	68%
Validation	17%	10%	21%	14%
Scenarios	88%	10%	39%	12%
Dominant indicator	Biophysical (33%) and monetary (36%)	Relatively even (Biophysical 32%, ranking 22%, monetary 34%)	Biophysical (73%)	Biophysical (57%)
Trade-off—sophisticated/map overlay	30%/8%	11%/10%	20%/20%	22%/4%
Offsite effects	9%	3%	5%	1%
Mapping ES	15%	26%	83%	5%
Policy instruments considered	27%	18%	6%	11%
Use of secondary data	73%	59%	83%	14%
System boundary definition purely biophysical	49%	37%	32%	62%

scenarios—to derive recommendations. Properties of available secondary data need to be taken into account—if these are not appropriate for the policy question at hand, the study will likely not be successful with respect to policy impact. ES categories and indicators need to be selected not only depending on data or model availability but also on their usefulness for recommendations. If not all desired parameters can be measured or modeled at preferred extent, resolution, and grain, it might be worth to check if useful results could still be derived. Excluded aspects of the studied socio-ecological system need to be reported and reflected in policy recommendations since this might heavily influence the validity of the conclusions. This refers also to the exclusion of off-site effects and possible changes in system conditions.

While not every ES study will be able to consider off-site effects, it is essential to reflect on this decision when it comes to drawing conclusions. However, it should in most cases be possible to check the plausibility of off-site effects and report them qualitatively in the discussion. Examples are the import of fish as a side effect of marine protection (Hilborn 2013), the import of forest products due to forest protection (Meyfroidt et al. 2010), carbon emissions of long distant eco-tourism (Weaver 2006; Eijgelaar et al. 2010), and bioenergy imports due to increasing bioenergy demand (Miyake et al. 2012; Mingorrı 2014). Examples of more advanced approaches are virtual water or CO_2 equivalent emissions, the ecological footprint (Wackernagel et al. 2002), the land footprint (Bruckner et al. 2015), or the water footprint (Hoekstra and Chapagain 2006). The framework provided by Schröter et al. (2018) provides helpful guidance how to conceptualize off-site effects into ES assessments. Research instruments such as ecological based life cycle assessments that incorporate ES or risk-adjusted wealth accounting systems are currently missing (Pascual et al. 2017).

Validation of results requires that independent data are used for testing the model or existing data are re-used in a cross-validation setup. Problems arise due to hierarchical dependencies, random effects, and spatial and temporal autocorrelation as well as by data scarcity. Therefore, it is essential to develop an applicable validation strategy and potentially adjust the experimental design. Similarly, a proper treatment of uncertainties should be planned at the design phase to ensure that information and resources are available. Substantial knowledge already exists in many disciplines related to ES research to identify, quantify, and communicate uncertainties which can be transferred and applied in ES studies (Hamel and Bryant 2017).

Experimental design of a case study is also essential to derive policy relevant results from trade-off analysis. The possibility to identify trade-offs and synergies is limited by the design of the case study. If biophysical interactions between ecosystem components or social interactions between beneficiaries are not included, it will be hard to identify tipping points or non-linear relationships between services. Sims et al. (2014) show the interaction between climate change, pine beetle dynamics, and disturbance-mitigating ES and highlight the importance of incorporating biophysical interactions and scenario analysis in ES assessments. If only a few possibilities are tested, recommendations might miss important compromise solutions (Seppelt et al. 2013). Kragt and Robertson (2014) provide a best practice example how Pareto frontier analysis could be used to assess trade-offs between different management options. Schipanski et al. (2014) show how trade-offs of management options can be assessed under consideration of temporal dynamics.

For a more realistic mapping of ES, it is essential to go beyond using land use composition as a proxy for ES. Lavorel et al. (2011) show how ES models based on plant traits and abiotic characteristics can be used to improve understanding of ecological constraints for the delivery of multiple services as well as to identify opportunities for enhancing the multifunctionality of landscapes. Mandle et al. (2015) and Schirpke et al. (2014) provide food for thought how trade-offs between beneficiaries can be estimated based on the derivation of beneficiaries specific areas where ES provisioning is demanded.

Clearly not every ES study needs to address all mentioned blind spots: studies aiming at methodological improvements do not necessarily have to involve stakeholders or provide specific recommendations for decision-making. Also, not all ES assessments need to incorporate seldom-investigated ES such as genetic resources or ornamental species. However, there is clearly the possibility to improve many ES case studies and the wider ES knowledge base with respect to some of the blind spots mentioned. The indicator cluster analysis (Table 2) shows that there are groups of studies that are better in addressing different blind spots. Partly this was related to the ES categories studied by the clusters, probably linked to the different research traditions of the diverse ES research community. Validation of results and a quantitative assessment of uncertainties are for example commonly applied in hydrological and ecological modeling—which might explain why these blind spots are addressed more frequently in ES assessments involving those services. Learning across the boundaries of these clusters might help to overcome the blind spots and additionally might help to reach the level of inter- and transdisciplinarity that is needed for multifaceted ES assessments. Not all concepts might be easily transferable across different ES research communities. However, we are optimistic that an interdisciplinary exchange will stimulate new exciting developments leading to better science and improved management support.

We provide a list of critical questions (Table 3) as a guideline for authors and reviewers to check on how blind spots of ES research might be incorporated in ES assessments. In addition, we recommend the use of evidence-based approaches (Mupepele et al. 2016). Real-world problems in ES assessments will necessitate compromises and shortcuts. Therefore, we do not assume that each study will comply with all facets of our guidelines. However, these compromises should be clearly stated and not suppressed by the urge to influence policy.

Potential blind spots not covered in our analysis

We did not address all potential blind spots in our analysis. One of these aspects is the co-production of ES by both ecological and social systems (Reyers et al. 2013; Bennett et al. 2015). Without explicitly including social interactions with or anthropogenic modifications of ecosystems as an integral feature of the ecosystem in an ES assessment, decision-making based on ES frameworks risks being ineffective or even counterproductive (Chan et al. 2012; Comberti et al. 2015). In the large majority of food production systems, for example, the maintenance and enhancement of food provisioning depends primarily on land management practices. Cultural heritage is another example of co-production of services: landscapes such as classical pastoral landscapes in Portugal are clearly a co-production of land management and ecological processes (Daniel et al. 2012). Future research should therefore not only look into the ecological validity of models but also investigate how such models incorporate what we know about social systems, leading to a better "social-ecological" validity of ES assessments.

Co-production of ES by social systems was not investigated frequently. Especially lookup table approaches missed this aspect, but even studies employing process models capturing effects of land management such as LPJ (Sitch et al. 2003; Bondeau et al. 2007; Zaehle et al. 2007) or SWAT (Soil and Water Assessment Tool; Gassman et al. 2007) in most situations did not restrict the ES estimate to the ecosystem contribution. Statistical models used to quantify the potential for recreation and tourism did frequently include accessibility or infrastructure as co-variates. However, in the majority of cases the values reported included both the contribution by ecosystems and manmade infrastructure—i.e., the importance of the co-production of the service was not explicitly addressed.

The importance of monitoring ES has been stressed repeatedly (Carpenter et al. 2006; Tallis et al. 2012). Monitoring ES should include both demand- and supply-side indicators to allow consistent information on changes in the coupled socio-environmental system in which ES are provided and

Table 3 Critical questions for reviewing ecosystem service assessment studies and guiding questions for assessing the use of studies in policy and synthesis work, e.g., IPBES, or TEEB-like studies. Not all questions are limited to one blind spot. Questions can be used to critically assess existing studies—asking them at the design phase of a study might help to derive more reliable and applicable results with higher impact. ES = Ecosystem services

Critical questions for ecosystem service assessments with respect to…
Social-ecological validity:
1. How did ecosystem structures, processes, and functions translate into ES?
2. How was the model tested or validated in this system for these indicators?
3. How robust were the results in the face of uncertainty?
4. Was the data suitable for the conclusions drawn? How were conclusions supported by the available data?
5. Were effects on species and ecosystems functioning underpinning the ES considered? Which effects were ignored?
6. Were non-linear or interactive effects considered or could their existence be safely excluded? How could non-linear or interactive effects affect validity of results and conclusions?
7. Were the results reliable given the system boundary? Was the system boundary suitable for the analysis and the recommendations drawn?
8. Was the co-production of ES taken into account? How strong could that have affected trade-offs and interactions between services?
9. How did social-technological modification translate into ES?
Trade-offs:
10. Which interactions between ES were shown to be causal (by literature and/or measurements/field work)?
11. How did trade-offs depend on assumptions regarding different scenarios, management options, or changing environments?
12. Which secondary effects of changes in the supply of ES on the economic system or markets were considered?
13. Were trade-offs between beneficiaries and aspects of distributional justice considered in the trade-off analysis? Which groups of beneficiaries were potentially not adequately represented in the trade-off analysis?
14. Was the number of compared options sufficiently large or was the search space too limited to effectively support decision-making?
Off-site effects:
15. Which environmental processes on larger temporal and spatial scales were considered?
16. How did trade-mediated effects on larger spatial scale determine the results?
17. Which differences in the valuation of ES were studied when beneficiaries were distributed over different locations?
18. Which indirect effects could be triggered by recommended policy actions?
19. Were cross-scale or cross-location impacts considered?
20. Were stakeholders aware of ES burdens elsewhere?
Stakeholder work:
21. How were stakeholder groups set up and how were their roles described (transparency)?
22. How can the results and statements derived from stakeholder work be tested, e.g., did they match or contradict observed behavior and why?
Relevance and usability:
23. Were specific management actions or the effects of a policy instrument analyzed? Which management actions and policy instruments could be used to operationalize the conclusions?
24. Were specific scenarios developed and quantified or was the dynamic nature of the land system ignored?
25. Were specific recommendations given? Were results presented in a format applicable for decision-making?
26. Were governance indicators (in particular political stability, control of corruption, and government effectiveness) considered?

used. Without it will be impossible to detect trends in the potential of ecosystems to provide ES or trends in changing demands for those ES. Furthermore, it will not be possible to assess the effects of management actions and policy instruments on ES. However, existing monitoring systems have not been designed for the majority of ES—this leads to the use of secondary data and potential biases when using those to infer ES from those proxies. Monitoring was seldom a topic in the case studies in our sample. Only a few case studies have investigated longer-term developments of ES. Of those, case studies that analyzed longer-term developments in ES mostly monitored proxies such as land use and investigated the effect on ES by means of models (Lautenbach et al. 2011; Carreño et al. 2012; Renard et al. 2015; Dittrich et al. 2017; Lavorel et al. 2017b; Locher-Krause et al. 2017). This is of course a serious blind spot that limits our ability to detect long-term trends and impacts of policy actions. Initiatives such as "Group on Earth Observations – Biodiversity Observation Network" (GEO BON; Tallis et al. 2012) and citizen science approaches (Schröter et al. 2017) may succeed in promoting the use of monitoring of ES. Remote sensing offers interesting possibilities for that (Cord et al. 2017b).

Potential shortcomings

Our analysis was based on a random sample of papers, resulting from a search specified on phrases in the title of journal articles published in the ISI Web of Knowledge. Potentially, this could have introduced a bias that limits our ability to infer about the population of all published ES studies. The random sample itself should however be large enough to allow unbiased estimates of parameters of the underlying population. By limiting the search to the ISI Web of Knowledge, we ignored a richness of published material in books and the gray literature. Experts and policymakers might prefer policy reports instead of scientific papers as a source of information. In addition, since we excluded studies not published in English we might have introduced a spatial bias: some non-English-speaking countries might have higher number of case studies published as gray literature which we ignored by our sample design. From our perspective, however, our selection is justified by the quality check that articles published in ISI-listed journals should have undergone. To put it differently: while clearly many good and excellent studies have been published only as gray literature, we assume that on average the scientific quality of gray literature would be lower and that failure to the blind spots mentioned here would even be more widespread. Testing this assumption would be the challenge for a follow-up study.

Key policy reports such as IPBES or TEEB build on existing work by summarizing key findings. While the panel experts can be expected to downweight lower-quality studies, the reports will nevertheless be affected by blind spots in the literature: if regions or ES have not been assessed in studies, the knowledge base is likely to affect the synthesis. Similarly, the low number of studies involving off-site effects or scenarios limits the possibility that such aspects are adequately reflected in the synthesis work. While the key policy reports reflect the validity of the knowledge base to provide adequate advice, synthesis work would be even more reliable if the mentioned blind spots were addressed more frequently by the scientific community.

Comparison with more specialized reviews

Our finding that we lack studies in low-developed countries in which societies depend much stronger on ES than in higher developed countries is in line with findings from Christie et al. (2012), Delgado and Marín (2015), McDonough et al. (2017), and Schmidt et al. (2016). Schmidt et al. (2016) quantified how blind spots in accounting for natural capital and ES differ between regions of the globe. McDonough et al. (2017) analyzed the country of origin of ES publications, while Delgado and Marín (2015) and Vihervaara et al. (2010) analyzed where the ES study took place. Christie et al. (2012) analyzed the use of monetary and non-monetary valuation techniques for biodiversity. Findings of these studies show geographical patterns similar to our results and highlight research need in society's poorest nations.

The number of papers identified concerning trade-off assessments and prioritization is largely in line with findings by Martinez-Harms et al. (2015) and Howe et al. (2014). Martinez-Harms et al. (2015) also showed that most trade-off studies used simple approaches such as correlation analysis or map overlays between different ES. Our findings about the limited use of trade-off analysis and models that incorporate interactions between ES are also in line with the review of Bommarco et al. (2013) on ecological intensification building on ES. The authors found that nearly all studies had examined a single service process in isolation and that it had never been tested whether suites of below- and above-ground services contribute synergistically or trade-off in their contribution to crop yield and quality.

Conclusions

The number of published ES studies has continued to grow over the last years. However, shortcomings with respect to social-ecological validity, trade-off analysis, off-site effects, stakeholder involvement, and relevance and usability still persist. Certain aspects, such as the use of instruments, or the inclusion of the demand side in assessments, were addressed more frequently in recent studies. The continued uneven coverage in geographical space and of ES categories hinders the efficient steering of resources for the conservation of ES. One has to be careful to assess the current state of knowledge on ES—it is not sufficient to look at the number of studies on specific ES categories or at the number of studies in a region. Instead, it is important to filter out the case studies that fulfill specific requirements. The presence of blind spots seems clustered: clusters of ES studied together were prone to different blind spots—learning across those clusters offers great potential for improvement. To effectively operationalize the concept of ES, the mentioned blind spots need to be addressed by upcoming studies. The list of critical questions provided (Table 3) can help to raise the awareness of the blind spots both for synthesis of existing knowledge and for future research agendas. Furthermore, it might help to stimulate critical thinking to further improve ES research. The scientific community has come a long way and delivered important results for policymakers and managers—still, we all should strive to identify ways to improve our research and how to provide specific policy recommendations from our results.

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