Chapter 8
Tools for Mapping and Quantifying Ecosystem Services Supply

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8.1 Introduction

The concept of ecosystem services highlights the contribution of ecosystems to human well-being while bridging ecological and social systems (Daily and Matson 2008; Haines-Young 2009; de Groot et al. 2010; Fisher et al. 2011; Chung and Kang 2013; Bryan et al. 2013). They are classified into four categories, which include provisioning services, regulating services, cultural services, and supporting

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B. R. Bakshi (ed.), *Engineering and Ecosystems*,
https://doi.org/10.1007/978-3-031-35692-6_8

services (Millennium Assessment (MA) 2005).[1] Ecosystem services represent the interface for the management of social-ecological systems at different scales (MA 2005; Müller et al. 2010) and quantifying their benefits within the context of socio-ecological systems has increasingly become a focus of research (Raymond et al. 2009; Haines-Young et al. 2012; Schmitt and Brugere 2013; Posner 2015; Reed et al. 2017).

The actual or realized benefits, which society receives from ecosystem services, will depend not only on the supply of ecosystems services but also on the demand from society (Burkhard et al. 2012). Humans have an important role in the delivery of ecosystem services and make critical contributions to the flow of ecosystem services between areas of supply and demand. Therefore, the potential benefits derived from ecosystem services will also depend on stakeholders' management strategies, capacity, access, and need within the context of a range of social, economic, and institutional contexts (Villamagna et al. 2013).

While the importance of differentiating between demand and supply is recognized, existing tools for quantifying ecosystem services primarily focus on ecosystem services supply or implicitly integrate demand without specifically looking at flows and/or treat the beneficiaries as homogenous. Ecosystem services supply focuses on the capacity of natural ecosystems to provide relevant ecosystem goods and services within a given time period (Burkhard et al. 2012; Crossman et al. 2013). However, increasingly more research and tools have been devoted to the quantification of ecosystem services demand and the flows of ecosystem services between supply and demand locations. Ecosystem service supply remains important because it is directly derived from the amount and quality of ecosystems, irrespective of the demand or value assigned to the potential service. It is therefore an essential part of ecosystem services assessments.

The objective of this chapter is to review the research and tools for quantifying ecosystem service supply focusing on commonly used tools. We begin by discussing and reviewing the major types of mapping methods for characterizing single ecosystem services. We then describe how multiple ecosystem services can be considered and ways in which important priority areas for ecosystem services provision can be identified. While the review focuses specifically on ecosystem services supply, the distinction between supply and demand in many modeling papers may not be specifically made. We conclude by discussing the research gaps and future challenges in quantifying ecosystem service supply.

[1] Besides the most widely used MA classification, there are also other classification systems which treat ecosystem services slightly differently, especially the MA's "supporting services" class. For instance, TEEB replaced "supporting service" with "habitat service" (TEEB 2010), while CICES (Common International Classification of Ecosystem Services) does not include "supporting service" leaving only three categories (Haines-Young and Potschin-Young 2018). FEGS-CS (Final Ecosystem Goods and Services Classification System) classify 21 final ecosystem service categories and 358 unique FEGS codes (Landers and Nahlik 2013). More details are provided in Chap. 2.

8.2 Quantifying Ecosystem Services

A wide range of methods have been developed for ecosystem services assessments as discussed in multiple comprehensive reviews (Feld et al. 2009; Seppelt et al. 2011; Hernández-Morcillo et al. 2013; Blattert et al. 2017; Cord et al. 2017). These approaches can be vastly different, even for the same services, and the values quantified can vary from biophysical values to monetary values (La Notte et al. 2012; Reed et al. 2017), biophysical values such as erosion control (Vihervaara et al. 2012), or social values (Raymond et al. 2009; Bryan et al. 2010, 2011; Brown 2013).

Many approaches focus specifically on the spatial characteristics of ecosystem services, such as where services are generated (supply) and where services are received and distributed (demand, Fig. 8.1). From the perspective of ecosystem services supply, many quantification methods exist that are derived from natural sciences. For example, the science of catchment management or ecosystem management provides direct quantifications of ecosystem services provision, even though it was developed long before the concept of ecosystem services was popularized. Thus, in this chapter, we have focused specifically on tools and techniques, which have only been developed from the perspective of quantifying—and in particular mapping of—ecosystem services.

Mapping is a practical and useful tool for integrating and revealing complex spatial information across different scales (Martínez-Harms and Balvanera 2012; Crossman et al. 2013). Given the advantages of mapping approaches, the number of studies on mapping ecosystem services has been growing in recent years. Ecosystem services maps can explicitly reveal the spatial distribution of ecosystem services (Egoh et al. 2008), such as service hotspot areas (Eigenbrod et al. 2010; Leh

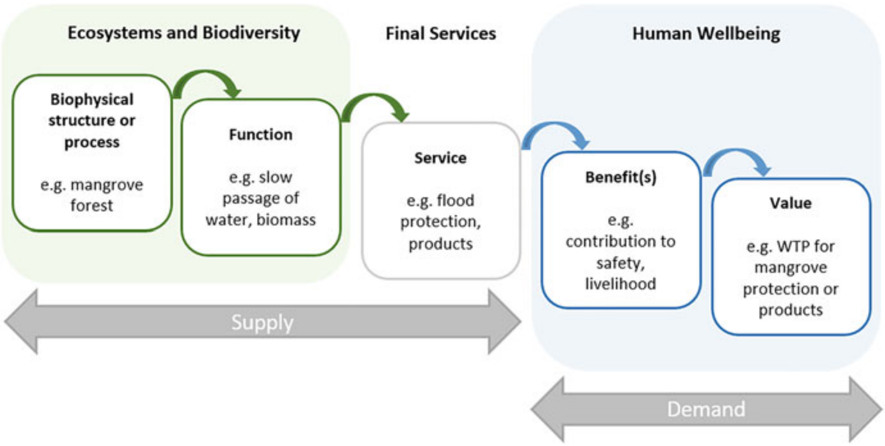

Fig. 8.1 Ecosystem services cascade and relationship between ecosystem services supply and demand. (Adapted from Braat and de Groot (2012))

et al. 2013) and trade-offs and correlations between multiple ecosystem services (Mouchet et al. 2017), which can support decision-making and help communication with stakeholders. With the continued development of ecosystem services mapping methods, many comprehensive off-the-shelf tools have been developed, among which, InVEST (Sharp et al. 2020), ARIES (Villa et al. 2009), and SoIVES (Sherrouse et al. 2011) are widely used. A longer list of tools can be found in a review by de Groot et al. (2018; Table 8) and in the Ecosystems Knowledge Network Tool Assessor (https://ecosystemsknowledge.net/tool). In this chapter we describe the five main types of ecosystem services mapping and quantification methods: (1) Primary data; (2) Causal relationships; (3) Expert knowledge; (4) Participatory mapping; and (5) Biophysical models.

8.2.1 Primary Data

Ecosystem services supply can be mapped directly using primary data. Primary data are derived from field survey or samples (Martínez-Harms and Balvanera 2012) and or remote sensing to represent ecosystem services values. Primary data are most often used in quantifying provisioning services, such as timber (Delphin et al. 2013) and food (Wang et al. 2018a) (Fig. 8.2a). Although primary data offer

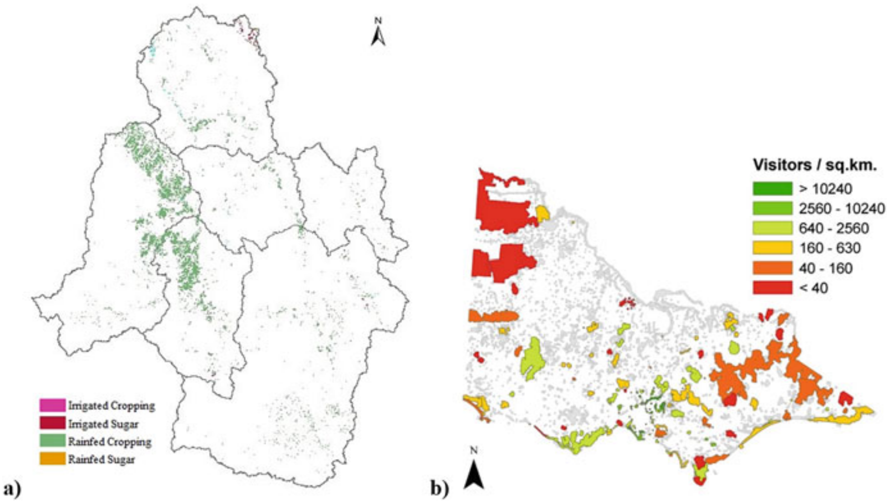

Fig. 8.2 (**a**) Many ecosystem services, especially provisioning services, are commonly produced for other purposes such as a map of agricultural areas. Maps of agricultural land cover classes such as above from Queensland Australia represent primary data, which can be used to map ecosystem services. (Adapted from Wang et al. 2018a). (**b**) Causal relationships use readily available information to characterize ecosystem services. This map of the spatial patterns of Victoria's protected areas popularity was produced using official visitation statistics. (Adapted from Levin et al. 2017)

the most accurate information, such data are not available for all types of ecosystem services. For example, the majority of regulating and supporting services are closely related to complex ecosystem processes and functions, while cultural services are nonmaterial and thus difficult to represent with primary data. Furthermore, availability of primary data is a critical limitation (Martínez-Harms and Balvanera 2012). Primary data cannot always be mapped, particularly if there are issues around data confidentiality or data protection.

8.2.2 *Causal Relationships*

The term causal relationship describes how readily available information can be used to characterize ecosystem processes and services and is one of the most frequently used methods to map different ecosystem services (Martínez-Harms and Balvanera 2012; Schägner et al. 2013). For instance, air quality regulation services of a city could be mapped based on urban greenspace distribution and the vegetation attributes described by remote-sensing-derived leaf area index (Ortolani and Vitale 2016). Recreational services are usually mapped by social and ecological data such as the national parks numbers, tourism statistics, and public access levels (Raudsepp-Hearne et al. 2010; Paracchini et al. 2014) (Fig. 8.2b).

Causal relationships represent a method which utilizes proxies to quantify ecosystem services provision and provide a useful way to estimate ecosystem services when direct ecosystem services indicators are absent. Many of these mapping and quantification techniques implicitly incorporate ecosystem services demand without explicitly modeling supply (i.e., visitor numbers at national parks). However, causal relationship methods need considerable knowledge for understanding the generating processes of ecosystem services (Eigenbrod et al. 2010; Schägner et al. 2013). Uncertainties are produced and the outcomes are not accurate if there are poor causal relationships between the data and the ecosystem service it is meant to represent.

8.2.3 *Expert-Based Model Knowledge*

Expert knowledge is one of the most widely used approaches; it is a simple and effective way to map ecosystem services (Egoh et al. 2008; Müller et al. 2010; Grêt-Regamey et al. 2017). This method incorporates advice from different experts and stakeholders to map ecosystem services. For example, Haines-Young et al. (2012) use "expert" and "literature-driven" methods to establish the multiple links between land cover and use and potential ecosystem service outputs in different geographical contexts across Europe. One of the more popular methods is the application of land cover proxies where each land cover is given a specific value for ecosystem service provision (Jacobs et al. 2015; Burkhard et al. 2012, 2014) to create a

Land cover	Ecosystem Services: Crop production	Water provision	Raw materials	Recreation and tourism	Landscape aesthetics	Spirituality and Cultural	Erosion prevention	Pollination	Carbon sequestration	Local climate and air	Stormwater retention	Habitat for biodiversity
Water bodies	3.4	4.25	3.3	4.3	3.5	2.75	2.1	1.75	3.25	3.5	3.05	3.8
Bare soil	0.6	0.35	1.85	1.25	0.35	0.25	0.7	0.35	0.5	0.4	1.2	1.1
Impervious	0.3	0.25	0.25	1.65	1.65	0.8	2.5	0.45	0.35	0.35	0.35	1.55
Roads	0.25	0.2	0.2	0.35	0.45	0.35	0.4	0.4	0.25	0.25	0.25	0.25
Oil palm	4.5	3.4	4.25	2.25	3.4	2.25	3.9	3.7	4	4.1	3.1	3.75
Rubber	4.5	3.2	4.4	3.2	3.25	3.8	3.9	3.8	4	4	3.4	3.5
Rubber mix	4.4	3.15	4.35	4.1	4.1	4.3	4	3.5	4	4.2	4	4.1
Fruit mix	4.4	3.1	4.3	4.15	4.4	3.6	4	4.35	3.8	4.1	4	4.3
Other ag.	4.3	3.1	4.05	3.65	3.9	4	4.05	4.15	4	4.2	3.7	3.9
Forests	0.8	0.9	4.1	4.4	4.35	4.4	4.3	4.05	4.5	4.2	4.4	4.45

Fig. 8.3 Matrix model/look-up-table for mapping ecosystem services supply with a land cover proxy for Kuala Lumpur, Malaysia. Dark red represents the highest potential to supply ES, while yellow shows the least. These values are then mapped based on their corresponding land cover classes. (Data supplied by Gangul Nelaka)

matrix/look-up-table of ecosystem services versus land cover (Fig. 8.3), which can then be converted into an ecosystem services map based on this relationship. Other examples of this approach include a study by Swetnam et al. (2011) who developed scenarios integrating local stakeholders and experts to define the extent of changes in land cover classes under different sets of drivers. Palomo et al. (2013) defined ecosystem services according to expert-advice and questionnaires and then mapped ecosystem services flows. However, the high levels of subjectivity and the lack of quantitative assessments for ecosystem services are the main disadvantages of the expert knowledge approaches (Hamel and Bryant 2017).

8.2.4 *Participatory Mapping*

Public Participation Geographic Information System (PPGIS) is used to map ecosystem services using quantitative or qualitative social surveys (Brown 2013; Shoyama and Yamagata 2016). A common approach is to ask participants to identify ecosystem service locations using a point or polygon on a web-based or paper map (Fig. 8.4). These points are then aggregated or interpolated to create a raster surface representing ecosystem service provision. Social Values for Ecosystem Services (SoIVES) is a very popular ecosystem services mapping tool that also uses quantitative social surveys of both point locations and preferences for difference locations and land covers, in conjunction with Maxent create raster surfaces of ecosystem services (Sherrouse et al. 2011). Both approaches map ecosystem services from the perspective of a specific stakeholder group.

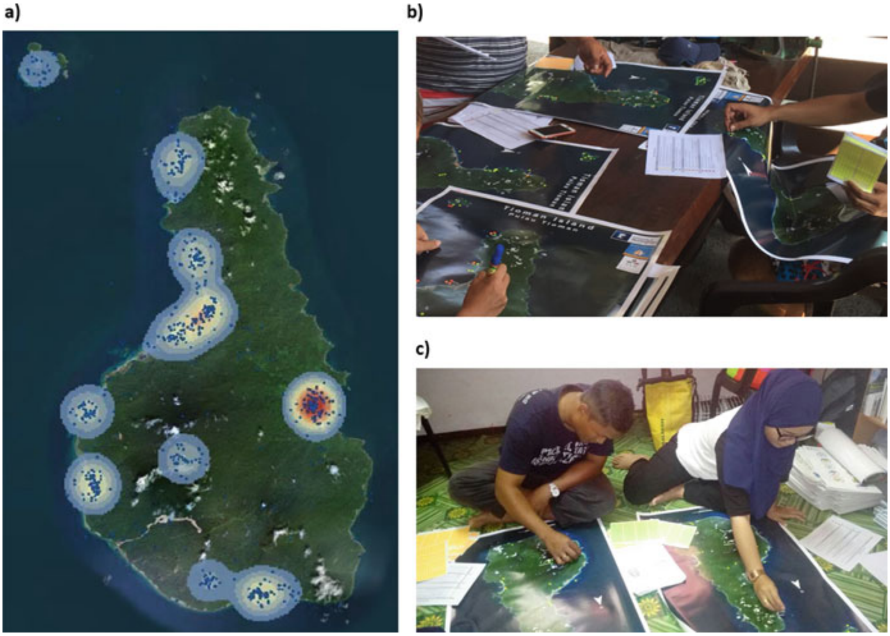

Fig. 8.4 PPGIS mapping with a quantitative social survey carried out on Tioman Island Malaysia. (Figures and data adapted from Lechner et al. (2020)). (**a**) Map of recreational landscape values. Multiple survey participants identified recreational landscape values with sticker dots (**b**, **c**), which were first digitized, then combined, and finally interpolated to produce a recreational landscape value surface

8.2.5 Biophysical Models

The most data intensive mapping approach uses biophysical models to describe the biophysical processes and functions of ecosystems (Kareiva et al. 2011; Petz 2014; Runting 2017). Various models from different disciplines and theories are utilized for ecosystem services assessments and are integrated with GIS. Commonly, these biophysical models are based on process or mechanistic models, which are composed of multiple equations, which approximate real world biophysical processes such as erosion or hydrological flows. However, machine learning, or other statistical approaches that mimic biophysical processes, can also be used. These models are commonly used for mapping regulating services and supporting services (Martínez-Harms and Balvanera 2012; Baral et al. 2013a; Pulighe et al. 2016). For instance, the Universal Soil Loss Equation (USLE) can be used to simulate the mechanisms associated with the interaction between soil, precipitation, and vegetation to assess the soil loss and retention for the soil conservation service (Sánchez-Canales et al. 2015; Grafius et al. 2016), while the Carnegie-Ames-Stanford approach (CASA) model can be used to simulate photosynthesis processes to estimate net primary productivity for carbon sequestration service (Dai et al.

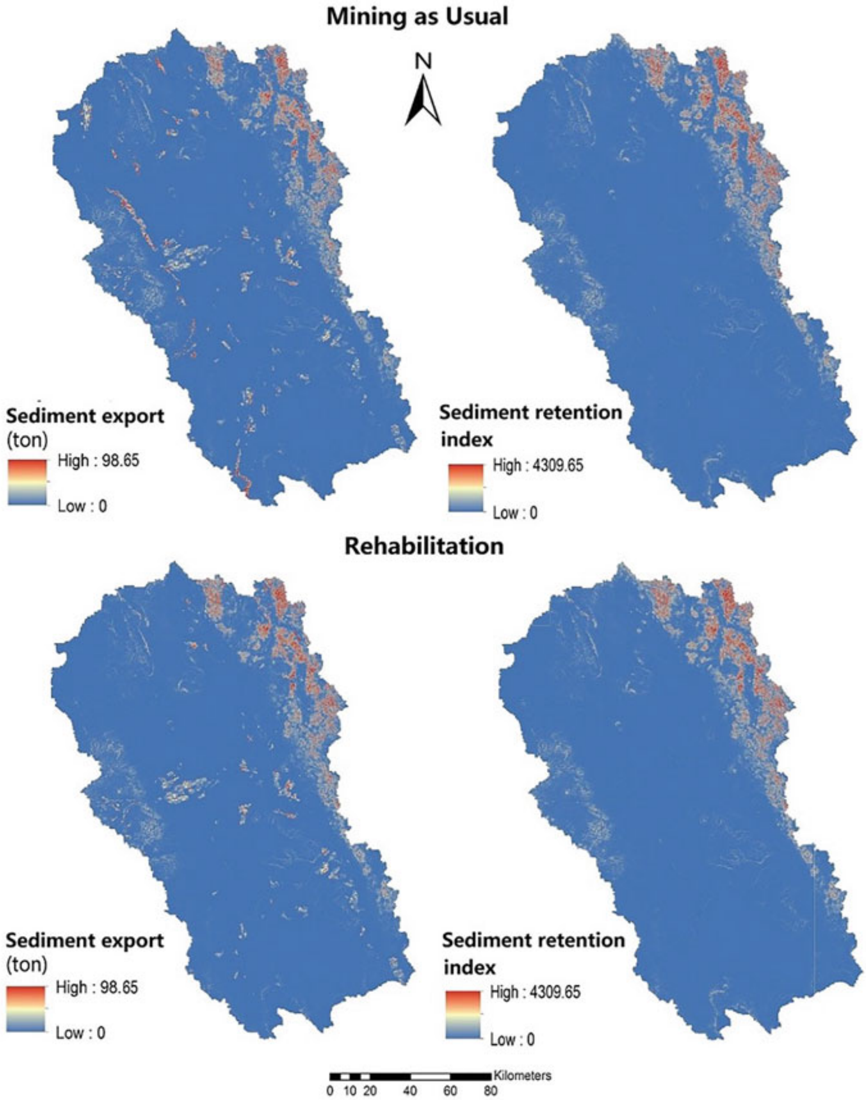

Fig. 8.5 Sediment retention service mapping under two different scenarios: rehabilitation and mining as usual scenarios in an Australian mining region (Wang et al. 2018b) using process-based Sediment Delivery Ratio model in InVEST

2017). However, there are still challenges associated with how well these models characterize a biophysical process (Surfleet and Tullos 2013; Sharp et al. 2020) and/or models selection (Lavorel et al. 2011; Petz and van Oudenhoven 2012) due to a lack of understanding ecosystem processes, subjectivity, and data availability. Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) is one of the most widely applied ecosystem services mapping tools (Sharp et al. 2020) and it includes several biophysical models to characterize a range of ecosystem services such as sediment retention, urban cooling, and flood risk (Fig. 8.5).

8.3 Comparing Ecosystem Service Supply

Ecosystem services assessments typically assess the spatial or temporal change in services to identify optimal land use or management strategies. Several approaches are used to compare multiple ecosystems and identify their spatial trends. In this section, we describe approaches for (1) comparing multiple ecosystem services and (2) identifying priority areas.

8.3.1 Comparison of Multiple Services

A single ecosystem can provide multiple ecosystem services and these services can interact resulting in services trade-offs and synergies (Fig. 8.6). Trade-offs occur when one service decreases as another service increases (Rodríguez et al. 2006; Grêt-Regamey et al. 2013). Synergies are defined as the situation when the changes are positive for both (or many) ecosystem services and trade-offs describe the opposing situation (Rodríguez et al. 2006; Haase et al. 2012; Crossman et al. 2013). Haase et al. (2012) proposed an evaluation matrix of ecosystem services correlations, which describe ecosystem services synergies, trade-offs, losses, and other single-aspect changes.

Ecosystem services trade-offs and synergies have been mapped and assessed in various studies. Among the four types of ecosystem services, provisioning and regulating services are most frequently assessed. For instance, the interactions between water provision and sediment retention services have been assessed in multiple fields of different countries (Williams and Hedlund 2014; Früh-Müller et al. 2016; Fernandez-Campo et al. 2017; Hao et al. 2017; Hamel et al. 2019). There are also many studies focusing on cultural services interactions (Turner et al. 2014; Ament et al. 2017). Different agricultural practices can lead to different correlations among crop yields, soil carbon, and nutrient retention (Qiu and Turner 2013; Kragt and Robertson 2014; Nelson et al. 2009). Correlations between ecosystem services are also commonly assessed in response to future change scenarios. For instance, climate changes can cause changes in hydrological ecosystem services and their interactions with other services (Agropolis et al. 2013; Jiang et al. 2017; Mandle et al. 2017). Analyses of ecosystem services current and future trade-offs and synergies are commonly integrated into land use planning and natural resources management (Castro et al. 2014; Witt et al. 2014; Keith et al. 2017) to help decision-makers balance the protection and promotion of different ecosystem services.

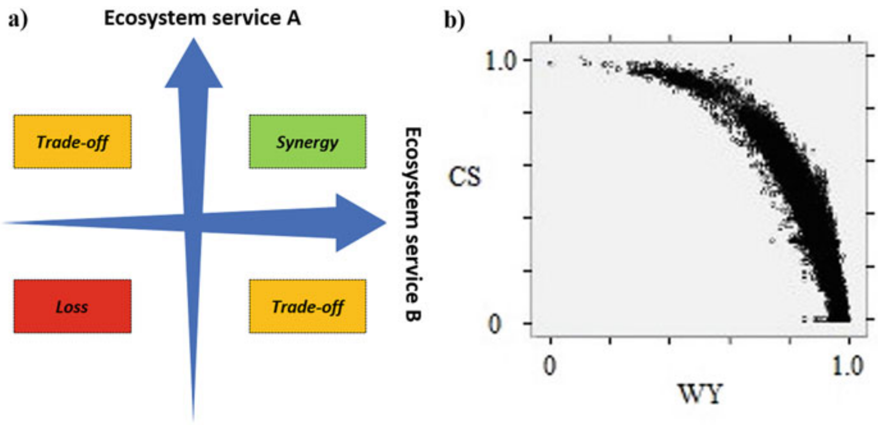

Fig. 8.6 (**a**) Evaluation matrix of ecosystem services trade-offs and synergies. (Adapted from Haase et al. 2012). (**b**) The pixel-scale correlation between carbon sequestration (CS) and water yield (WY) within the whole mining lease of Currugh mine in Queensland, Australia. The chart shows a trade-off between carbon sequestration (CS) and water yield (WY) with a Spearman's rank correlation coefficient of −0.88 ($P = 0.01$). (Adapted from Wang et al. 2020)

Quantitative approaches for measuring ecosystem services trade-offs and synergies include aspatial and spatial methods. Aspatial methods range from aggregated metrics such as correlation coefficients to graphical tools such as radar plots, parallel coordinate plots, or scatterplots (Weil 2017). Correlation coefficients such as the Pearson and Spearman's rank coefficients have been widely used to assess the ecosystem services interactions (Agropolis et al. 2013; Castro et al. 2014; Oñatibia et al. 2015; Staes et al. 2017). Regression models have also been applied to characterize pair-wise relationships between ecosystem services. For instance, Jia et al. (2014) utilized a logistical regression model to analyze ecosystem services trade-offs and synergies for a Grain-for-Green area in China. Maes et al. (2012b) applied multinomial logistic regression models to assess the correlations between ecosystem service supply, biodiversity, and habitat conservation status in Europe.

In addition to aggregated statistical methods for assessing trade-offs, a range of useful tools or indicators have been developed to map these trade-offs. Beyond traditional mapping methods such as hotspot or chloropleth maps (Weil 2017; Burkhard and Maes 2017), interactive tools have become popular to visualize trade-offs and synergies (Natural Capital Project 2020; Fredriksson et al. 2020). For instance, Pang et al. (2017) developed the Landscape simulation and Ecological Assessment (LEcA) tool to analyze synergies and trade-offs among five ecosystem services in Sweden. Trodahl et al. (2017) utilized Land Utilization and Capability Indicator (LUCI) to evaluate the trade-offs between water quality and agricultural productivity in New Zealand. Other useful examples can be found on the Natural Capital Project's visualization website (Natural Capital Project 2020). Where many ecosystem services are assessed, the concept of ecosystem services bundles has been used to describe services that always concurrently appear together; these are commonly identified by cluster analysis methods (Raudsepp-Hearne et al. 2010; Turner et al. 2014; Ament et al. 2017; Mouchet et al. 2017).

8.3.2 Identifying Priority Areas

Assessing the spatial patterns in ecosystem service distribution can be valuable in identifying priority – areas for management (Lourdes et al. 2022). Such assessments have been applied to individual and multiple ecosystem services, although variation in the spatial distribution between multiple ecosystem services can be high (Schröter and Remme 2016). In order to meaningfully assess the spatial patterns of multiple ecosystem services, individual services are converted to a common scale (i.e., rescaled) for standardization such as a minimum-maximum normalization (e.g., 0–1) (Dou et al. 2020; Lavorel et al. 2011; Maes et al. 2012a; Mokondoko et al. 2018), or z-score normalization or z-standardization (Jopke et al. 2015; Weil 2017). The type of rescaling method applied is tied strongly to the objective of the assessment, taking note that the absolute or initial values assigned to ecosystem service will change when standardized.

Hotspot mapping is a common cluster analysis method used to distinguish the abundance and distribution of ecosystem services across a landscape. The terms “hotspot” and “coldspot” respectively denote areas of high service provision and low service provision for a single ecosystem service (Egoh et al. 2008). Methods for delineating ecosystem service hotspots are diverse; Schröter and Remme (2016) provide a comprehensive review on the methods available. Popular methods include the top richest cells method and Getis-Ord Gi* statistic. Although both methods delineate ecosystem service hotspots, the two methods highlight unique approaches to identifying spatial patterns in ecosystem services. The top richest cells (quantile) method divides grid cells, ranked from high to low service value, into classes with an equal number of cells (Bai et al. 2011; Dou et al. 2020; Eigenbrod et al. 2010; Orsi et al. 2020). The class with the highest values is defined as a hotspot, with class sizes for hotspots ranging from 5% to 30% of the total cells. For example, Orsi et al. (2020) delineated hotspots as the highest 20% of cells supplying an ecosystem service. While the Getis-Ord Gi* statistic (Getis and Ord 1992) utilizes a spatial clustering method to delineate hotspots. This method identifies areas where high value cells are highly concentrated within a specified distance/neighbourhood (i.e. high values within a neighborhood of low values or vice versa) (Bagstad et al. 2016; Li et al. 2017; Sylla et al. 2020), distinguishing hotspots and coldspots with varying degrees of clustering (i.e. significance). The differences in these two approaches are further detailed by Bagstad et al. (2017).

Areas of overlap between multiple ecosystem services, or ‘multiple service hotspots’, can be identified by summing hotspots produced through a range of methods, for individual services. The applications of both hotspot mapping and aggregation and comparison of multiple services are diverse (Anderson et al. 2009; Dou et al. 2020; Maes et al. 2012a; Mokondoko et al. 2018). Pan et al. (2020)

assessed areas of overlap between several hydrological ecosystem services for the integrated management of a river basin, while Bogdan et al. (2019) mapped social values, delineating multiple service hotspots for cultural ecosystem services. Bagstad et al. (2016), on the other hand, combined biophysical ecosystem service values and social values, delineating multiple service hotspots for more inclusive management of the Southern Rocky Mountains.

8.4 Data Sources and Uncertainty

One of the big challenges for ecosystem services mapping is the requirement for data and the varieties of sources and principles by which they were created (Crossman 2017). The types of data sources can be distinguished into two categories, primary and secondary data (Egoh et al. 2012; Martínez-Harms and Balvanera 2012; Crossman et al. 2013). Primary data are those derived from sampling in the field, such as field measurements, surveys, and interviews, while secondary data are defined as those derived from readily available information not typically verified in the field including literature-based or modeled data (Martínez-Harms and Balvanera 2012). Along with the types of data sources, data can be classified as biophysical or socioeconomic. Biophysical data are related to the natural and biophysical systems, such as hydrological data, remote sensing, topographical, and land cover data. Socioeconomic data are the data related to social and human activities, such as crop production, population, road lines, and economic data (Martínez-Harms and Balvanera 2012).

Among different type of datasets, land cover data are the most widely used. Land cover change is one of the greatest drivers of changes in ecosystems and their services and, as noted in Sect. 8.3, is commonly used as proxy for mapping ecosystem services (Petter et al. 2013; Baral et al. 2013b; Nahuelhual et al. 2014; Abram et al. 2014; Tolessa et al. 2016). Land cover spatial data can be acquired through different ways. There are many well-established land use and land cover database in many countries, which are acquired through particular land use and cover mapping projects. For instance, the CORINE land cover database of Europe (Haines-Young 2009; Burkhard et al. 2012) and the national land cover dataset (NLCD) of USA (Lawler et al. 2014; Yoo et al. 2014) are widely used for ecosystem services assessments. Where existing data is unavailable, land cover changes can be mapped using remote sensing (Krishnaswamy et al. 2009; Tolessa et al. 2016; Zaehringer et al. 2017). Besides land cover spatial data, other data such as hydrological (Vigerstol and Aukema 2011; Terrado et al. 2014), topographic (Sherrouse et al. 2011; Fernandez-Campo et al. 2017), and climate data (Bangash et al. 2013; Jiang et al. 2017) are also are frequently utilized for ecosystem services mapping.

Scale is always a critical issue in landscape ecology and geographic research (Lechner et al. 2012a) and has a significant effect on ecosystem services mapping (Nemec and Raudsepp-Hearne 2012; Di Sabatino et al. 2013). Some ecological processes are scale dependent (i.e., species environment relationships), while other processes occur at multiple scales (Lechner et al. 2012b; Grêt-Regamey et al. 2014). Ecosystem services supply can also be mapped at different grain sizes and extents, which include pixel, local, regional, national, and global scales (Martínez-Harms and Balvanera 2012). Among them, local and regional scales are the most frequently assessed in which the extents of a study are commonly defined by the boundaries of a biogeographic or hydrologic system such as a mountain range (Grêt-regamey et al. 2012), watershed (Band et al. 2012), forest (Pohjanmies et al. 2017), or urban area (McPhearson et al. 2013).

Mapping scale can influence the results of ecosystem service assessment. For instance, Grafius et al. (2016) mapped ecosystem services at two different scales in three urban areas of the UK, and found sensitivity to scale was dependent on the type of service. Hou et al. (2017) assessed ecosystem service interactions at the pixel and town scales through a case study in the central Loess Plateau of China, which revealed that scale could apparently affect ecosystem services synergies and interactions. Grêt-Regamey et al. (2014) estimated the effects of scale on ecosystem services mapping through four case studies of different countries and suggested a four-step approach to address the scale issues. There are also many other studies focusing on scales of ecosystem services mapping (Larondelle and Lauf 2016; Raudsepp-Hearne and Peterson 2016; Calderón-Contreras and Quiroz-Rosas 2017; Xu et al. 2017), which all demonstrate the scale dependency issue and emphasize the importance of considering scale effects when mapping ecosystem services.

Beyond input data and spatial scale, several other sources of uncertainty affect the quantification and mapping of ecosystem services, including the context and framing of the assessment and, in the case of modeled data, the model structure, parameters, and technical implementation (Hamel and Bryant 2017). Managing these uncertainties involves understanding the potential use of ecosystem services information, potentially through codevelopment approaches, and applying proved analytical methods developed in the field of integrated environmental modeling (Pappenberger and Beven 2006; Petersen et al. 2013; Hamilton et al. 2019; Hamel and Bryant 2017),

8.5 Challenges for Quantifying Ecosystem Services

One of the primary goals for quantifying and mapping ecosystem service is for integration into planning and management (Egoh et al. 2008; Raymond et al. 2009; Potschin and Haines-Young 2012; Grêt-Regamey et al. 2017). There are many examples of mapping ecosystem services for urban planning focusing on the whole urban area (Lauf et al. 2014; Kaczorowska et al. 2015; Albert et al. 2016; Larondelle and Lauf 2016; Pickard et al. 2017; see Lourdes et al. 2021 for a regional review) or particular parts such as urban green spaces (Pulighe et al. 2016; Engström and Gren 2017). Also, ecosystem services have been mapped for conservation planning and natural resource management (Tallis and Polasky 2009; Bottalico et al. 2015;

Gunton et al. 2017) commonly together with biodiversity (Guerry et al. 2012; Sumarga and Hein 2014).

Although studies on ecosystem services mapping and quantification are growing, there are still a number of challenges. Some significant review papers have summarized the multiple challenges and bottlenecks associated with ecosystem services mapping (Crossman et al. 2013; Malinga et al. 2015; Brown and Fagerholm 2015) and characterized the types of ecosystem services mapped for multiple purposes (Willemen et al. 2015; Klein and Celio 2015; Drakou et al. 2015). Building on these existing reviews, we outline four key research challenges and gaps, which need to be considered and should be a focus for future research.

1. Gaps in Data Availability

Data availability can affect the quality of ecosystem services maps and the types of ecosystem services mapping tools available. Where primary data is not used, the application of proxy data is used, which can lead to uncertainties for mapping outputs (Eigenbrod et al. 2010; Schägner et al. 2013). Little is known about how the errors associated with proxy-based methods might affect the inferences drawn from analyses because quantifying the impacts of such errors is difficult without comparisons to primary data (Vrebos et al. 2015). A major challenge for ecosystem services mapping is to develop approaches, which adequately characterize ecosystems when using limited available data. This is especially important for data poor regions in the Global South where both primary data such as land cover maps may be of poor quality or unavailable and basic biophysical information such as the properties of soil may have never been measured or are highly uncertain, thus restricting the use of process-based biophysical models.

2. Inconsistency in Mapping Approaches

Although there are various approaches applied to mapping ecosystem services, there is still a need to understand the uncertainties and biases introduced by different mapping methods (Crossman et al. 2013; Crossman 2017). Different indicators and approaches have been used to map the same service, which can lead to apparent differences in the outputs (Schulp et al. 2014). Also, the same service may be mapped differently according to different research objectives such as mapping only ecosystem supply versus quantifying flows and potential values of ecosystem services. Additional guidance for ecosystem service mapping for decision-making and interpreting outputs will help understand the differences between methods and the implications for the development of decision-support tools (Bagstad et al. 2013; Hamel et al. 2020). Because different actors have different mandates and motivations, understanding information needs will ensure that ecosystem services information is used effectively (Bremer et al. 2020). It is important to note; there is not one optimal method, but the approach taken should be adapted to the decision-making context and uncertainty understood.

3. Assessing Uncertainties in Ecosystem Services Mapping

Uncertainty assessments are commonly conducted in many disciplines from hydrology (Benke et al. 2008), economics (Gilboa et al. 2008) to landscape ecology (Lechner et al. 2012a) as uncertainty can seriously affect the outcome of an analysis. Currently, few studies focus explicitly on analyzing uncertainty in ecosystem services mapping (Grêt-regamey et al. 2012, 2014; Schulp et al. 2014; Hamel and Guswa 2015; Wang et al. 2018a, b). A recent special issue in the journal *Ecosystem Services* in 2018 demonstrated best practices and challenges for "transparent, feasible and useful uncertainty assessment in ecosystem services modelling" (Bryant et al. 2018). Uncertainty was characterized for different models (Maldonado et al. 2018) and different scenarios (Ashley et al. 2018; Monge et al. 2018) and new methods were introduced for uncertainty assessment such as through the application of machine learning (Willcock et al. 2018). However, there are still a lot of challenges for successfully assessing uncertainties of ecosystem services mapping. As ecosystem services mapping draws on methods from a range of disciplines each with their own methods for assessing uncertainty (e.g., social sciences to hydrology), these methods should be incorporated into ecosystem services quantification approaches. It is important that any approach which addresses uncertainty is effective without being so time-consuming that it would be impractical to apply (Hamel and Bryant 2017).

As ecosystem service modeling methods become more complex, incorporating and assessing multiple ecosystem services in more complex ways, can cause errors and uncertainty to propogate and magnify. Approaches that incorporate multiple ecosystem services remain limited in several ways. Many approaches identify only high or low values of ecosystem services provision relative to a study area (i.e., top pixel values) or neighborhood (i.e., Getis-Ord Gi). However, the normalization process ignores the absolute values of these ecosystem services from the societal perspective. Not all ecosystem services are equivalent in their value to society and thus not all high valued ecosystem services should be considered equal when it comes to combining multiple ecosystem services.

4. Mapping Across Temporal Scale

While ecosystem services are commonly mapped across space, there are still relatively few studies, which have mapped historical changes in ecosystem services which have included high temporal resolution (i.e., numerous timesteps), even though abiotic (i.e., rainfall) and biotic (i.e., phenology) systems are dynamic. Although mapping future ecosystem service scenarios is relatively common, ecosystem services hotspots, trade-offs, and priority areas will change in both time and space. Mapping these temporally can add to the predictive capability of the outcomes. Such an approach is especially important in highly dynamic landscapes such as mining sites and agricultural and urban landscapes, which develop very quickly, especially in the Global South.

8.6 Conclusions

This chapter provided an introduction to the current tools for mapping and quantifying ecosystem services supply. While ecosystem service approaches have progressed rapidly in recent years, there are still many challenges. This is especially the case in the Global South where there is rapid land use change, resulting in the loss of biodiversity and ecosystem services. In urban landscapes, modelling aproaches still need further development. For example, InVEST, one of the most widely used ecosystem services modeling package, only recently released a suite of tools for modeling urban ecosystem services. In addition, moving beyond the realm of quantifying and mapping ecosystem services, consideration needs to be given for how the outcomes will be used and by whom. This also poses challenges in terms of how the outcomes should be represented including ways to clearly communicate uncertainties of both the input and output data and the ways in which the models have been validated. Knowledge gaps between practitioners and stakeholders could be reduced by building collaborative connections and including stakeholders early on in the mapping and decision-making process. This ensures that the needs of the end users are met and the underlying questions to be mapped are understood. Co-production of maps could result in output maps that are fit for purpose, easily understood by relevant stakeholders or end users, and could lead to better results around translating mapping outcomes into effective policy, planning, and management.

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