

The Institute for Natural Resource Conservation
of the Christian-Albrechts-Universität zu Kiel

Socioeconomic contexts for the spatial variations of ecosystem
services and the associated uncertainties

Dissertation
submitted for the Doctoral Degree
awarded by the Faculty of Agricultural and Nutritional Sciences
of the
Christian-Albrechts-Universität zu Kiel

Submitted by

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Kiel, 2014

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Date of Oral Examination: July 2, 2014

Printed with the approval of
the Faculty of Agricultural and Nutritional Sciences
of the Christian-Albrechts-Universität zu Kiel

Zusammenfassung

Ökosystemleistungen basieren auf ökologischen Prozessen und Funktionen und werden stark durch sozio-ökonomische Faktoren beeinflusst. Da die Ökosystemleistungen wesentliche Güter und Dienste für die menschliche Wohlfahrt bereitstellen, sind sie sehr gut geeignet, um die Interaktionen innerhalb von Mensch-Umweltsystemen zu reflektieren. Bei der Anwendung von Ökosystemleistungsabschätzungen treten vielfach starke Unsicherheiten auf, die vor einer Anwendung erkannt und nach Möglichkeit reduziert werden sollten. In diesem Kontext untersucht diese Arbeit die Wechselwirkungen zwischen Ökosystemleistungen und sozioökonomischen Variablen auf dem regionalen Skalenniveau, und es wird ein methodischer Rahmen zur Unsicherheitsanalyse entworfen, der in den Untersuchungsräumen angewendet wird.

Im ersten Kapitel wird das Konzept der Ökosystemleistungen vorgestellt und es werden die Forschungsfragen für diese Dissertation herausgearbeitet. Hierbei werden die Themenfelder der Ökosystemleistungs-Quantifizierung und –Regionalisierung vorgestellt, es wird die Rolle der Ökosystemleistungen in Mensch-Umwelt-Systemen herausgearbeitet und es werden die stärksten Unsicherheits-Quellen bei der Quantifizierung von Ökosystemleistungen diskutiert. Aus diesen Ausführungen werden Wissenslücken abgeleitet und die Hypothesen und Forschungsfragen dieser Arbeit werden aufgelistet.

Im zweiten Kapitel werden die sozioökonomischen Einflüsse auf Biodiversität, Ökosystemleistungen und Wohlfahrtskriterien auf der regionalen Skala in der chinesischen Provinz Jiangsu auf der Basis des DPSIR-Modells (Driver-Pressure-State-Impact-Response) untersucht. Daneben werden die Beziehungen zwischen den einzelnen Sektoren dieses Modells analysiert. Die Resultate zeigen, dass Urbanisierung und Industrialisierung im Vergleich mit intensivst genutzten Agrargebieten durchaus positive Einflüsse auf Biodiversität, Produktivität der Landwirtschaft, Tourismus und Lebensstandards der Bevölkerung haben können. Ähnliches gilt für die Förderung der Agrartechnik und die Unterstützung einer modernen Agrarforschung, die ebenso positiven Einfluss auf die ökologische Ausstattung der Landschaft ausüben können. Weiterhin wird dargestellt, dass die regionalen Kohlenstoffspeicher durch nicht-agrarische Nutzungsformen gefördert werden, während die Ausdehnung des Farmlandes und die Steigerung der Agrarproduktion negativ auf Biodiversität und Services wirken. Diese Zusammenhänge werden anhand von statistischen und geostatistischen Analysen dargestellt.

Das Kapitel 3 charakterisiert urban-rurale Gradienten in Bezug auf Ökosystemleistungen und sozioökonomische Variablen im Umkreis der Städte Leipzig und Kunming (China). Daneben werden die unterschiedlichen Gradiententypen zueinander in Beziehung gesetzt und verglichen. Schließlich werden die Unsicherheiten des Verfahrens herausgearbeitet. Die Ergebnisse zeigen eine Reihe von Regelmäßigkeiten der räumlichen Muster in beiden Untersuchungslandschaften. So zeigen die Habitatqualitäten und die f-Transpirationsraten in Leipzig und Kunming ähnliche Tendenzen einer Zunahme mit zunehmender Entfernung vom Stadtzentrum. Viele andere Ökosystemleistungs-Variablen zeigen gegenläufige Ausprägungen, während die Verkehrsdichte, urbane Landnutzungstypen und Bevölkerungsdichten erwartungsgemäß zum Land hin abnehmen. Haushaltsgößen, Wohnungsgößen und die Arbeitslosigkeit demonstrieren in Leipzig ein inkonsistentes Bild mit bemerkenswerten Fluktuationen. In Bezug auf die Relationen zwischen den Gradiententypen konnten interessante Korrelationen mit wechselnden regionalen Besonderheiten beobachtet werden. Abschließend wurde eine Methode zur Abschätzung der

Unsicherheiten bei der Ableitung räumlicher Gradienten entwickelt und exemplarisch angewendet

Das vierte Kapitel beschreibt die räumlichen Ausprägungen von Ökosystemleistungen und sozioökonomischen Variablen in einem stark gestörten Wassereinzugsgebiet Südwestchinas. Dabei werden zunächst 5 Ökosystemleistungen in 9 Wassereinzugsgebieten des Dianchi-Einzugsgebiets quantifiziert. Diese Arbeiten basieren auf Anwendungen des Modellsystems InVEST. Die Ergebnisse dieser Modellstudien werden einem Wechselwirkungs-orientierten Hypothesentest unterzogen, um die Auswirkungen gesellschaftlicher Faktoren auf die Ökosystemleistungen abschätzen zu können. Weiterhin werden anhand der Stickstoff-Modellierung die verschiedenen Ökosystemleistungskomponenten Potenzial, Fluss und Nachfrage differenziert, und die Modelloutputs werden einer Sensitivitätsanalyse unterzogen. Der Hypothesentest-Ansatz erbringt nur sehr wenige Übereinstimmungen zwischen den theoretischen Aussagen und den praktischen Korrelationen aus Ökosystemleistungen und sozio-ökonomischen Variablen. Die häufigsten Übereinstimmungen treten dabei in Bezug auf Landnutzungs-gesteuerte Parameter auf.

Das Kapitel 5 fasst die vielfältigen Quellen für Unsicherheiten bei der Landschaftsanalyse und der regionalen Abschätzung von Ökosystemleistungen zusammen und entwickelt eine Methodik zur Erfassung der Unsicherheiten. Diese basieren zum einen auf den methodischen Problemen der Landschaftsanalyse und der entsprechenden Datenquellen, zum zweiten auf der grundsätzlichen Komplexität der Mensch-Umwelt-Beziehungen und drittens auf vielen methodischen und technischen Problemfeldern. Um diese Unsicherheiten in Ökosystemleistungsabschätzungen zu integrieren, eignen sich insbesondere die Verfahren der Systemanalyse, Szenariotechniken und vergleichende methodische Ansätze. Zur Reduzierung der Unsicherheiten ist es erforderlich, die methodischen Entwicklungen eng zu verfolgen, die Datenbasis zu erweitern, die Zahl von befragten Experten zu erhöhen, Repräsentativitätskriterien zu beachten und zentrale Arbeitsschritte zu standardisieren.

Das sechste Kapitel enthält die wesentlichen Schlussfolgerungen dieser Arbeit. Die Forschungsfragen werden beantwortet, wobei die Verbindungen zwischen sozioökonomischen Variablen und Ökosystemleistungen hervorgehoben werden. Die Anwendbarkeit der Methoden wird mit Schwerpunkt auf Regionalisierungsverfahren diskutiert und die Notwendigkeit einer kritischen Unsicherheitsanalyse wird unterstrichen. Abschließend werden Anregungen für die Weiterentwicklung des Ökosystemleistungskonzepts formuliert.

Summary

Ecosystem services are strongly underpinned by ecological processes and functions and influenced by socioeconomics in human-environmental systems. As the prerequisites for human well-being, ecosystem services can reflect the interactions of human and environmental systems. Being pervading the process of ecosystem service assessments, uncertainties should be uncovered and preferably reduced before the assessing results are adopted for the decision-making of regional environmental management. This study explores the interrelationships between ecosystem services and socioeconomic variables at regional scales, develops a methodological framework of uncertainty analysis and applies it to investigate the uncertainties emerged in the assessments of ecosystem services of the study areas.

Chapter 1 provides a brief review of the fields concerning the ecosystem service issues addressed in this thesis. The introduction involves the basic concepts related to ecosystem services, the state of the art of ecosystem service quantification and mapping, the role of ecosystem services in human-environmental systems, ecosystem services' linkages with socioeconomics as well as the uncertainties in ecosystem service assessments. After uncovering the respective research gaps, this chapter identifies and elucidates the objectives of the study and raises the associated four research questions.

Chapter 2 explores the socioeconomic influences on biodiversity, ecosystem services and human well-being at the regional scale of Jiangsu, China on the basis of the DPSIR (Driver-Pressure-State-Impact-Response) conceptual model. Additionally, the study investigates the quantitative linkages between the five sectors of the DPSIR model. The results show that urbanization and industrialization in the urban areas can have positive influences on regional biodiversity, agricultural productivity, tourism services and rural residents' living standards. Besides, the knowledge, technology and finance inputs for agriculture have positive impacts on these system components. Concerning regional carbon storage, non-cropland vegetation cover obviously plays a significant positive role. Contrarily, the expansion of farming land and the increase of total food production are two important negative influential factors of biodiversity, ecosystems' food provisioning capacity, regional tourism income and the well-being of the rural population. Finally, the linkages of the DPSIR sectors in a network pattern are quantitatively evidenced.

Chapter 3 characterizes the urban-rural gradients of ecosystem services and socioeconomics of Leipzig, Germany and Kunming, China. It further quantifies the linkages between the gradients of ecosystem services and socioeconomics and conducts gradient comparisons between different gradient patterns in the two study areas. The chapter ends with the revelation of the uncertainties in creating the gradients. The results show some similar regularities in the spatial patterns of ecosystem services and socioeconomic dimensions in both study areas. Habitat quality and f-evapotranspiration of Leipzig and habitat quality of Kunming demonstrate apparent trends of increases along all gradient patterns. However, the other ecosystem services present divergent spatial variability in different gradient patterns. Road density, urban fabric and population density show identical declining trends in both study areas except for the soaring of population density around the center of Leipzig. Differently, household size, housing area and unemployment rate in Leipzig present inconsistent spatial dynamics with considerable fluctuations. Regarding the gradient interrelations, road density, urban fabric and population density are strongly correlated with most ecosystem service types in both case study areas. In contrast, the gradients of household size, housing area and unemployment rate of Leipzig show inconsistent correlations

with the ecosystem services gradients. The introduced uncertainty gradient method shows appropriateness to quantitatively capture the uncertainties in exploring ecosystem services and socioeconomic gradients in urban-rural areas.

Chapter 4 addresses the spatial characteristics of ecosystem services and the respective socioeconomic influences in a heavily human-disturbed watershed in Southwest China. It firstly quantifies and maps five ecosystem services of nine river basins of the Dianchi Lake Watershed. The quantification is based on the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model and the biophysical and socioeconomic data. Thereafter, a confirmatory research is conducted by using a hypothesis-test methodology to investigate the socioeconomic causes of the spatial changes of the five ecosystem services. On the basis of the modeling results of nitrogen retention and water yield, this chapter exemplifies the distinctions between ecosystem services potential, flow and demand and performs a sensitivity analysis to test the influences of input data and parameter uncertainties on the modeling results. The hypothesis-test analysis reveals only a small number of socioeconomic influential factors, most of which are related to land use structure. The hypothesis-test methodology provided in this study is applicable in the investigation of socioeconomic influences on ecosystem services in the situation of socioeconomic data uncertainty and scarcity.

Chapter 5 summarizes the sources of uncertainties in landscape analysis and ecosystem service assessments and proposes a methodology to analyze and reduce the uncertainties. The fundamental uncertainty origins of landscape analysis are landscape complexity and methodological uncertainties. The major uncertainty sources of ecosystem service assessments include the complexity of the natural system, respondents' preferences and technical problems. Among these uncertainty source categories, initial data uncertainty pervades the whole assessment process and the limited knowledge about the complexity of ecosystems is the focal uncertainty origin. To analyze the uncertainties in assessments, systems analysis, scenario simulation and the comparison method are promising strategies. Lastly, we assume that the actions to reduce uncertainties should integrate continuous learning, expanding respondent numbers and sources, considering representativeness, improving and standardizing assessment methods and optimizing spatial and geobiophysical data.

Chapter 6 reaches the general conclusions of this thesis. It firstly answers the four research questions asked in the introduction. In the answers, the close connections between ecosystem services and socioeconomics are confirmed, the applicability of the mainstreaming quantification methods is debated, the strength of ecosystem service mapping is illustrated and the necessity and possibility of uncertainty analysis are argued. In ending the entire thesis, chapter 6 further generally evaluates the ecosystem service approach and identifies and main obstacles and problems in the application of ecosystem services. Moreover, it proposes potential solutions to the overcome the impediments and finally calls for an optimistic attitude to propel ecosystem services research.

Acknowledgements

My path for earning my Ph.D. was not possible without the support of many colleagues, family members and friends.

I would first like to thank my first advisor, Professor Felix Müller, who has devoted limitless time to guiding me through research, writing and all the other academic things. I could not have asked for a better advisor. I have learned a lot from Felix, and particularly admire his productivity, knowledge, approachability and eagerness to help.

I would also like to thank my second advisor, Professor Nicola Fohrer, who has offered great help to improve my thesis. Besides, I'm thankful for the other two committee members, Professor Christian Henning and Professor Birgit Schulze-Ehlers who have provided me with insightful advises for my thesis.

I'm very appreciative to my collaborators on my work, including my master's advisor Professor Bo Li, Dr. Benjamin Burkhard, Professor Shudong Zhou and Dr. Franziska Kroll. Without them, much of the work would not been possible. I'm also indebted to my other colleagues, Claus Schimming, Wilhelm Windhorst, Marion Kruse, Liwei Ma, Anja Müller, Christian Hertz-Kleptow, Peter Wangai, Neicheng Wu and Song Song. I have learned a lot from them and enjoyed my work in the institute together with them. I would also like to thank the administrative and IT staffs, Britta Witt, Karen Grotkopp, Kerrin Frahm and Kay Adam. Without their help I could not have moved on with my work in the institute.

I would also like to thank my school fellows, including Wenjuan Li, Xiaohui Hu, Chanjuan Ren, Yang Wang, Cheng Xue, Li Wang, Yanjun Ren, Wanglin Ma, Kaihuai Liao and Hao Chang for lots of help and laughs during my stay in Kiel.

I could not have survived my PhD experience without the love and support from my family. My wife has been a constant rock during the past few years, always offering me encouragement and motivation far away from China. My parents have been very supportive, always making me feel better.

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Chapter 1. Introduction

1. The ecosystem service concept

“Ecosystem services are the contributions of ecosystem structure and function – in combination with other inputs – to human well-being” (Burkhard and de Groot et al., 2012). Ecosystems produce benefits for humans by providing goods, or maintaining a favourable living environment through the regulations resulting from ecosystem processes, or by supplying the conditions for humans to fulfil their cultural needs (Daily, 1997; MA, 2005). Accordingly, ecosystem services can be classified into three categories, i.e., provisioning services, regulating services and cultural services (Table 1). Since the origin as an interdisciplinary concept, the ecosystem service approach has attracted extensive attentions from scientists of different fields (Jax et al., 2013; MA, 2005; TEEB, 2010). The values of ecosystem services for supporting decision-making within environmental management have been increasingly recognized by policy makers in recent years (Bateman et al., 2013; TEEB, 2010; Vohland et al., 2011).

The generation of ecosystem services is believed to be underpinned by biodiversity, which is a key component of ecosystem structures and processes (Balvanera et al., 2006; Haines-Young and Potschin, 2010). The importance of biodiversity for the deliverance of ecosystem services is primarily attributed to the diversity-productivity relationship and the strong correlations between net primary productivity and ecosystem services (Costanza et al., 1998; Costanza et al., 2007; Díaz et al., 2005; Gaston, 2000; Richmond et al., 2007). Despite the attempts to reveal the dependence of ecosystem services on biodiversity, the mechanism of this reliance is lacking adequate explanations (Costanza et al., 2007; Turner et al., 2007). The “ecosystem service cascade” originally depicts the path with which ecosystem services emerge from ecosystem structures and processes and then influence human well-being (de Groot et al., 2010b; Haines-Young and Potschin, 2010; Kandziora et al., 2013, see Figure 1). The framework addresses the response from humans based on their perception and valuation of ecosystem services to adjust their influence on the environmental system in order to build a win-win relationship with nature (de Groot et al., 2010b; Kandziora et al., 2013; Müller and Fohrer, in press; Rounsevell et al., 2010; TEEB, 2010). In the “ecosystem service cascade” framework, ecosystem services plays the role of linking human well-being with biophysical properties and ecosystem functions (Haines-Young and Potschin, 2010; Kandziora et al., 2013). Without ecosystem services, human’s dependence on nature to sustain its survival and development will vanish. Although the “ecosystem services cascade” has captured the interactions between ecosystems and human systems, the causalities between each pair of the framework components have not been sufficiently revealed and the quantification of the mutual relationships is still in its infancy (Hou et al., 2014; Kandziora et al., 2013).

This thesis quantifies the linkages between biodiversity, ecosystem services and human well-being in an empirical study in order to provide evidence to the “ecosystem service cascade” framework.

Table 1. Ecosystem services classification and examples (Adapted from MA, 2005).

Category	Descriptions	Examples
Provisioning service	Goods produced or provided by ecosystems	Food, timber, freshwater, bioenergy, genetic resources
Regulating service	Benefits obtained from regulation of ecosystem processes	Climate regulation, gas regulation, water purification, waste treatment, natural hazards mitigation
Cultural service	Non-material benefits from ecosystems	Recreation, tourism, knowledge systems, landscape aesthetics, cultural heritage

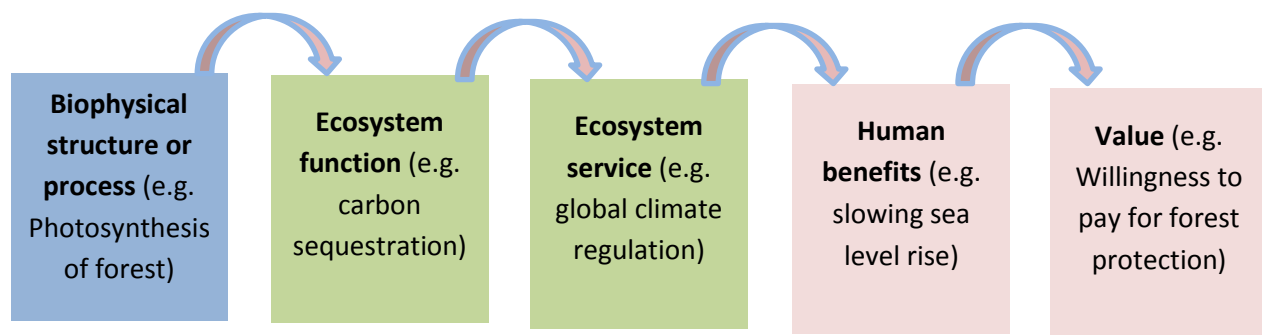


Figure 1. The “ecosystem service cascade” framework, illustrating the relationships between biophysical properties, ecosystem services and human benefits (adapted from Haines-Young and Potschin, 2010 and Kandziora et al., 2013).

2. Quantifying and mapping ecosystem services at regional scales

Ecosystem service quantification is a major concern for ecosystem service investigators. The mainstreaming methods involve biophysical measurements, modeling, expert judgments, monetization and value transference (Seppelt et al., 2012; TEEB, 2010). The researchers have performed a great number of ecosystem service quantifications at distinct scales in different areas around the world using various methods (Bateman et al., 2013; Costanza et al., 1997; Kienast et al., 2009; Larondelle and Haase, 2013; Leh et al., 2013; Liu et al., 2013; Xie et al., 2008) . Although the methods have been tested by extensive case studies, their applicability in more regions around the world needs to be further evidenced.

Ecosystem mapping can facilitate the communication of the spatial and temporal characters of ecosystem services among scientists and the public and policy makers (Crossman et al., 2013). Mapping ecosystem services has nearly become an indispensable section of an ecosystem service assessment report due to the facilitation provided by remote sensing and GIS techniques. Land use or land cover maps are usually necessary in mapping ecosystem services of most of the service types. Another essential data input is a matrix providing monetary or biophysical values of the visualized ecosystem services of different land cover or land use types (Burkhard et al., 2009). Newly developed quantification and mapping methods based on models have enabled the investigators to estimate ecosystem service values, present the value spatial variations as well as

predict ecosystem service quantities under different future scenarios without relying on the matrix of ecosystem service values (Tallis and Polasky, 2009; Villa et al., 2014). However, these modeling methods have a high demand of biophysical data and cost the researchers a relatively large amount of time to investigate literature and to get familiar with the tools (Bagstad et al., 2013).

Although mapping has shown strength in communicating ecosystem service information and aiding the policy making for regional environmental management, the potential inaccuracy and unreliability of the maps should not be overlooked. Dubious map information is due to the uncertainties pervading the entire process of ecosystem service assessments (Hou et al., 2013). The revelation and reduction of the uncertainties in ecosystem service quantification and mapping have already drawn researchers' attentions and associated theoretical and practical studies have been conducted (Bagstad et al., 2013b; Grêt-Regamey et al., 2013; Hou et al., 2013).

Ecosystem service quantification and mapping are two study focuses of this thesis. In this study, distinct methods are used to quantify multiple ecosystem services in different areas and respective maps are produced at different spatial scales.

3. Investigating ecosystem services in human-environmental systems

The interactions between human and environmental systems are an appealing issue for scientists and environmental managers. Connecting natural and human systems, ecosystem services are regarded as a promising concept to address human-environmental issues (de Groot et al., 2010; MA, 2005; TEEB, 2010). Ecosystem services can reflect the impacts of human systems on ecosystems. The types and intensities of human-induced influences on natural systems can be uncovered by investigating the magnitudes and the spatial and temporal variations of ecosystem services under the disturbance of human activities. The changes of ecosystem services can vice versa affect human well-being due to the inherent causalities (MA, 2005; Smith et al., 2013). The Millennium Ecosystem Assessment (MA) had built up a theoretical network of the causalities between ecosystem services and human well-being (MA, 2005). This framework connects the three ecosystem service categories to different constituents of human well-being. Besides, it differentiates the intensities and potentials for mediation by socioeconomic factors of the linkages. Some researchers have suggested distinctive classifications of human well-being by referring to the categorization of MA (Kandziora et al., 2013; Müller and Fohrer, in press; Smith et al., 2013; Summers et al., 2012). These classifications mainly involve economic, societal, environmental and personal well-being. Up to now, an influential number of studies have investigated the relations between ecosystem services and human well-being. However, most of the research still stays at the theoretical or conceptual stage (Duraiappah, 2011; Kandziora et al., 2013; Summers et al., 2012). Empirical studies and quantifications of the mutual connections are still limited, calling for more efforts from the scientists (Hou et al., 2014; Suneetha et al., 2011; Wang et al., 2014).

The complex relationships between ecosystem functions, services and human well-being require holistic approaches for comprehensively exploring the interactions to better aid the management of natural and social systems (Butler and Oluoch-Kosura, 2006; Summers et al., 2012). Accordingly, this thesis regards the relationships between ecosystem services and socioeconomics as a focal issue, which will appear in three articles.

4. Ecosystem services' linkages with socioeconomics

Ecosystem service changes are driven by multiple socioeconomic dimensions, mainly referring to population dynamics, economic activities and social factors (Bastian, 2013; Nelson et al., 2005). Indirect drivers derive from demography, economy, social policies, culture, religion, science and technology (MA, 2005; Nelson et al., 2005). Direct drivers (or pressures) mainly include land use change, overexploitation of natural resources and emissions of diverse pollutants (Alcamo et al., 2005; Bastian, 2013; Nelson et al., 2005), which have straightforward influences on ecosystem conditions and directly change the provision of ecosystem services. Although socioeconomic influences on ecosystem services have been extensively reported, empirical analyses of the quantitative causalities are lacking. The DPSIR (Driver-Pressure-State-Impact-Response) conceptual model is used to capture and organize the information of the interactions in human-environmental systems (Burkhard and Müller, 2008; Pinto et al., 2013). This adaptive management cycle can be coupled with the “ecosystem services cascade” to combine the information of ecosystem services and human well-being with the respective socioeconomic drivers and pressures (Kandziora et al., 2013).

This thesis tackles the applicability of the above described coupled framework in quantitatively investigating socioeconomic influences on biodiversity, ecosystem services and human well-being in empirical studies at regional scales.

Urban-rural areas have received increasing concerns from ecosystem service investigators due to the accompanying intensive human-environmental interactions (Kroll et al., 2012; Radford and James, 2013). To delineate the features of this significantly disturbed system, the principle of urban-rural gradients was proposed (McDonnell and Pickett, 1990). This principle has been proven to be applicable in capturing the spatial variation characters of ecosystem services in urban-rural area (Kroll et al., 2012; Radford and James, 2013). Additionally, socioeconomic gradients in urban-rural areas have been proven to be exiting, which has gained a growing popularity in the scientific communities of multiple disciplines (Dow, 2000; McDonnell and Hahs, 2008). However, the empirical evidence of urban-rural gradients of ecosystem services and socioeconomics are still limited and the relationships between them remain concealed.

This research gap provides a motivation to look for empirical evidence of ecosystem service and socioeconomic gradients and explore the mutual linkages in two urban-rural case areas.

The human-environmental interaction issue at watershed scales has drawn attentions from both environmental scientists and managers. Ecosystem service assessment has proven to be a promising approach to treat this issue (Bagstad et al., 2012; Nelson et al., 2009). As critical services generated by watershed, water-related services such as water yield, water purification and water regulation have drawn most interests from researchers (Bai et al., 2013; Leh et al., 2013; Notter et al., 2012). To quantify the ecosystem services of watershed regions, multiple methods, such as modeling, monetary valuation and public survey as well as multi-criteria decision analysis have been used (Bagstad et al., 2012; Bräuer, 2005; Liu et al., 2013; Tallis and Polasky, 2009). In human-disturbed watersheds, ecosystem services are interrelated with the socioeconomic status of human systems and influenced by human activities in different degrees. An extensive body of literature has reported the essential influences referring to human's land use activities, the growth of economy and population and the change of agricultural production patterns (Cai et al., 2013; Leh et al., 2013; Martinez et al., 2009; Reyers et al., 2009; Wu et al., 2013). Nevertheless, the investigations of the causes of ecosystem service changes virtually

addressed a limited number of socioeconomic dimensions, calling for comprehensive evaluations of human impacts on ecosystem services at watershed scales.

Aiming at narrowing this research gap, this thesis attempts to examine the influences of multiple socioeconomic attributes on the spatial variations of ecosystem services in a watershed case area.

5. Uncertainties in assessment of ecosystem services

Ecosystem service assessments contain outstanding degrees of generalization and uncertainty, owing to their high complexity and the integrative position of ecosystem services between human and environmental systems (Scolozzi et al., 2012). Quantitative reviews of ecosystem service studies show that a large number of the assessments did not provide sufficient information concerning their results' uncertainty (Seppelt et al., 2011, 2012). Insufficient information on uncertainty hinders the thorough understanding of the pattern- and process-based ecosystem services (Bateman et al., 2010; Díaz et al., 2005; Nicholson et al., 2009). Additionally, when investigating stakeholders' preferences, uncertainty may lead to systematically biased results and a false precision of ecosystem service valuation. This may further cause bias or even fault in decision making, which is based on the available information of ecosystem service status (Chavas, 2000; National Research Council, 2005). Therefore, the revelation of uncertainties is crucial in ecosystem service assessments and a methodological framework to treat the uncertainties is necessary.

Considering this research field, this thesis systematically summarizes the uncertainty sources in ecosystem service assessments and proposes a set of methods to analyze and reduce the respective uncertainties.

6. Objectives

This thesis assesses ecosystem services of several case study areas at distinct regional scales by using different quantifying and spatially explicit methods. Additionally, it explores the interactions between environmental and human systems by investigating the interrelationships between ecosystem services and socioeconomics. To cope with the uncertainties in ecosystem service assessments, the thesis develops a methodological framework for uncertainty analysis, which is used to evaluate the uncertainties emerged in the case studies.

The thesis has four main objectives:

1. Investigating the socioeconomic influences on biodiversity, ecosystem services and human well-being using the DPSIR model at the prefecture-level city scale of Jiangsu, China;
2. Analyzing the urban-rural gradients of ecosystem services and socioeconomics and quantifying the mutual relationships in the urban-rural areas of Leipzig and Kunming;
3. Exploring the impacts of the human system on the spatial variations of ecosystem services of the Dianchi Lake Watershed;
4. Systematically summarizing the uncertainty sources in ecosystem service assessments and suggesting the strategies to treat the uncertainties.

The goals of the study are related to four focal questions:

1. Are ecosystem services connected with socioeconomics in different human-environmental systems and how strong are the linkages?
2. What methods can be used to quantify ecosystem services and how is the applicability of different methods with respect to distinct ecosystem service types?
3. What types of maps can be used to visualize ecosystem services at different spatial scales and what are the strengths of the maps;
4. What uncertainties emerge in the processes of the assessments of ecosystem services in the case studies of this thesis and how can these uncertainties be treated?

The objectives are elucidated in four chapters:

Chapter 2 — **Socioeconomic influences on biodiversity, ecosystem services and human well-being: A quantitative application of the DPSIR model in Jiangsu, China.** This chapter focuses on the applicability of the DPSIR model in quantitatively investigating socioeconomic influences on biodiversity, ecosystem services and human well-being

Chapter 2 describes the theoretical linkages between biodiversity, ecosystem services and human well-being and their socioeconomic influential factors. It quantitatively investigates the socioeconomic influences on biodiversity, ecosystem services and rural human well-being at the prefecture-city level scale of Jiangsu, China using the DPSIR conceptual model and statistical methods. Furthermore, this chapter explores the quantitative linkages between the sectors of the DPSIR model and provides evidence to the network-form character of the model.

Chapter 2 is published in the journal *Science of the Total Environment*. It is co-authored by Shudong Zhou, Benjamin Burkhard and Felix Müller.

Chapter 3 — **Urban-rural gradients of ecosystem services and the linkages with socioeconomics.** This chapter concentrates on the spatial and quantitative characters of ecosystem services and socioeconomic gradients in urban-rural areas and the quantitative mutual linkages.

Chapter 3 quantifies and visualizes the important ecosystem services and socioeconomic dimensions of the urban-rural areas of Leipzig, Germany and Kunming, China. It further applies the urban-rural gradient principle to investigate the spatial characteristics of ecosystem services and socioeconomics of the two study areas. Based on these analyses, the paper compares the gradient patterns within and between the research areas and explores the linkages between the gradients of ecosystem services and socioeconomics. Chapter 3 ends with a quantitative analysis of the uncertainties in creating the gradients.

This chapter has been submitted to the journal *Environmental Monitoring and Assessment*. It is co-authored by Felix Müller, Bo Li, and Franziska Kroll.

Chapter 4 — **Ecosystem services of human-dominated watersheds and the influences from socioeconomics: A case study of the Dianchi Lake Watershed.** The focal issue of this chapter is the spatial variation of ecosystem services of the Dianchi Lake Watershed and the respective socioeconomic causes.

Chapter 4 firstly assesses five typical provisioning and regulating ecosystem services of the Dianchi Lake Watershed. It then investigates the impacts of the human system on ecosystem services at river basin scale of the watershed. The paper further illustrates the ecosystem services

potential, flow and demand concepts with the modeling results of nitrogen retention and analyzes the uncertainties emerging in ecosystem services modeling.

This chapter has been submitted to *Agriculture, Ecosystem & Environment*. It is co-authored by Bo Li and Felix Müller.

Chapter 5 — Uncertainties in landscape analysis and ecosystem service assessment. This chapter focuses on the sources of uncertainties in ecosystem services assessment and the respective uncertainty analyzing and reducing methods.

Chapter 5 summarizes the sources of uncertainty referring to general landscape systems analysis and ecosystem service assessments by literature investigation. It then analyzes the uncertainties in a land cover-based ecosystem service supply assessment which was conducted in a German case study region. Based on synthesizing the mainstreaming uncertainty analysis methods, this chapter proposes a methodological framework to systematically treat the uncertainties in landscape analysis and ecosystem service assessment. The framework consists of nine typical working steps targeting at the uncertainties emerging from the complexity of the studies system, the assessment methods, the bias in the preference-based surveys and uncertain data input.

This chapter has been published in the *Journal of Environmental Management*. It is co-authored by Benjamin Burkhard and Felix Müller.

Chapter 6 is the conclusions of this thesis, which echoes the focuses of chapter 2 to 5. It illustrates the implications of the primary findings of the former chapters concerning the “socioeconomic contexts for the spatial variations of ecosystem services and the associated uncertainties”.

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Chapter 2. Socioeconomic influences on biodiversity, ecosystem services and human well-being: A quantitative application of the DPSIR model in Jiangsu, China

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Science of the Total Environment, 2014, 490: 1012-1028

Abstract: One focus of ecosystem service research is the connection between biodiversity, ecosystem services and human well-being as well as the socioeconomic influences on them. Despite existing investigations, exact impacts from the human system on the dynamics of biodiversity, ecosystem services and human well-being are still uncertain because of the insufficiency of the respective quantitative analyses. Our research aims are discerning the socioeconomic influences on biodiversity, ecosystem services and human well-being and demonstrating mutual impacts between these items.

We propose a DPSIR framework coupling ecological integrity, ecosystem services as well as human well-being and suggest DPSIR indicators for the case study area Jiangsu, China. Based on available statistical and surveying data, we revealed the factors significantly impacting biodiversity, ecosystem services and human well-being in the research area through factor analysis and correlation analysis, using the 13 prefecture-level cities of Jiangsu as samples. The results show that urbanization and industrialization in the urban areas have predominant positive influences on regional biodiversity, agricultural productivity and tourism services as well as rural residents' living standards. Additionally, the knowledge, technology and finance inputs for agriculture also have generally positive impacts on these system components. Concerning regional carbon storage, non-cropland vegetation cover obviously plays a significant positive role. Contrarily, the expansion of farming land and the increase of total food production are two important negative influential factors of biodiversity, ecosystem's food provisioning service capacity, regional tourism income and the well-being of the rural population. Our study provides a promising approach based on the DPSIR model to quantitatively capture the socioeconomic influential factors of biodiversity, ecosystem services and human well-being for human-environmental systems at regional scales.

Keywords: DPSIR, ecosystem services cascade, indicators, statistics, factor analysis, human-environmental systems, agriculture, land use

1. Introduction

The interactions between human and environmental systems are appealing issues for scientists and environmental managers. Connecting natural and human systems, ecosystem services are regarded as a promising concept to address human environmental interaction issues (de Groot et al., 2010; MA, 2005; TEEB, 2010).

1.1. The Ecosystem Services Cascade

The “ecosystem service cascade” originally depicts the path with which ecosystem services emerge from ecosystem structures (e.g. biodiversity) and functions and then influence human well-being (de Groot et al., 2010; Haines-Young and Potschin, 2010; Kandziora et al., 2013; TEEB, 2010). This framework has been designed to address the response from humans to adjust their influence on the environmental system in order to build a win-win relationship with nature (Kandziora et al., 2013; Rounsevell et al., 2010). Biodiversity plays a significant role in the ecosystem service cascade (Balvanera et al., 2006; Díaz et al., 2005; Haberl et al., 2005; Haberl et al., 2004; Haines-Young and Potschin, 2010; Hooper et al., 2005), i.e. due to diversity-productivity relationships and the correlations between net primary productivity and ecosystem services (Costanza et al., 2007; Costanza et al., 1998; Díaz et al., 2005; Gaston, 2000; Richmond et al., 2007). Human well-being is assumed to have close causality with ecosystem services in the global context and at local scales (MA, 2005; Smith et al., 2013; Pereira et al., 2005; Shackleton et al., 2010a and 2010b; Suneetha et al., 2011). However, this linkage is proved to be not only a synergy but also a trade-off when diverse ecosystem services and human well-being items and different scales are considered (Duraiappah, 2011; MA, 2005; Raudsepp-Hearne et al., 2010). The complex connectedness between ecosystem functions, services and human well-being requires holistic approaches for comprehensive explorations of the interactions to better aid managing natural and social systems (Butler and Oluoch-Kosura, 2006; Summers et al., 2012).

1.2. Causes of changes in human-environmental systems¹

One research issue in the field of human-environmental interactions involves the human-induced causes of biodiversity and ecosystem service changes, which can affect human well-being due to prevailing inherent causalities (Nelson, 2005; Smith et al., 2013; Spangenberg et al., 2009; TEEB, 2010). These influences have been well documented and analyzed (Faggi et al., 2008; Hope et al., 2003; Spangenberg et al., 2009; Yu et al., 2014). Primary causes of biodiversity loss include population increase, land use change, biological invasions and pollutions caused by agricultural and rural development, whereof the influences from agriculture are regarded to be most influential (Kuldna et al., 2009; Maxim and Spangenberg, 2009; Pillsbury and Miller, 2008; Stratford and Robinson, 2005). Faggi et al. (2008) proved that local cultivation practices are the greatest threat to native plant and bird richness, turning out to be more dangerous than urban sprawl. The role of agriculture is also evidenced by (Hope et al., 2003) in an urban-pre-urban context.

¹ Human-environmental systems are coupled human and natural systems in which human and natural components interact (Liu et al., 2007).

Population dynamics, economic and social factors, which are named “drivers” are regarded as the three fundamental influence sources of ecosystem service on a global scale (Bastian, 2013; Huang et al., 2010; MA, 2005; Nelson, 2005). Drivers of ecosystem service are distinguished into direct and indirect ones. Indirect drivers derive from demography, economy, social policies, culture, religion, science and technology (MA, 2005; Nelson, 2005). Direct drivers have straightforward influences on ecosystem conditions and directly change the provision of ecosystem services. This type of drivers mainly includes land use change, overexploitation of natural resources and emissions of diverse pollutants (Alcamo et al., 2005; Bastian, 2013; Nelson, 2005). Couch and Karecha (2006) and NuiSSL and Rink (2005) specify social political influences as governmental regulations such as housing and tax policies. These regulations are considered to be more important than economic growth for land use change in the rural-urban regions facing economic and demographic decline. As a fundamental sector of economy, agriculture is considered to be an important driver for various ecosystem services (Bjorklund et al., 1999). Several case studies disclose the negative impacts of agricultural intensification on some ecosystem services, such as the loss of pollination (Klein et al., 2007), soil carbon decrease, water quality degradation, the decrease of nutrient retention and other services (Bjorklund et al., 1999; Matson et al., 1997). Besides the detrimental effects, socioeconomic drivers can also have positive influences on ecosystem service supplies, which are e.g., documented by Haase et al. (2012) and Kroll et al. (2012).

To sum up, the exiting investigations concerning the socioeconomic influences on biodiversity and ecosystem services are predominantly reporting and qualitatively describing the impacts. We found few studies quantitatively depicting the influential factors (Alcamo et al., 2005; Nelson, 2005) or revealing the quantitative linkages between the influences and biodiversity or ecosystem services (Huang et al., 2010; Klein et al., 2007). In this sense, quantification should be a promising issue for investigating the influences.

1.3. The DPSIR model of human-environmental systems

The Driver-Pressure-State-Impact-Response (DPSIR) model² originated from the Pressure- State-Response (PSR) framework, which was put forward by the Organization for Economic Cooperation and Development (OECD, 1993). Later it was adopted and elaborated by European Environmental Agency (Burkhard and Müller, 2008; European Environment Agency, 1995; Svarstad et al., 2008). As this model can capture the cause-effect relationships between the sectors of social, economic and environmental systems, it has been widely applied to analyze the interacting processes of human-environmental systems (Feld et al., 2010; Kristensen, 2004; Pinto et al., 2013). Up to now, the DPSIR framework has shown advantages, both conceptually and

² DPSIR definition (after Gabrielsen and Bosch, 2003):

Driver: Social, demographic and economic developments in societies and the corresponding changes in lifestyle, overall levels of consumption and production patterns – motivations for specific land use.

Pressure: Developments in release of substances (e.g. emissions), physical and biological agents, the use of resources and the use of land by human activities.

State: State of the environment, which refers to the quantifiable and quantitative physical, biological and chemical conditions in a defined area.

Impact: Impacts on natural and human systems (i.e. Impacts on the provision of ecosystem goods and services and the socio-economic system).

Response: The actions carried out by society and governments in efforts to minimize the negative impacts imposed on the environment, and feed back to the drivers or pressures influencing anthropogenic developments.

practically, in simplifying complex human-environmental systems, qualitatively describing their interrelationships and aiding in associated policy making (Agyemang et al., 2007; Burkhard and Müller, 2008; European Environment Agency, 2007; Feld et al., 2010; Kuldna et al., 2009; Potschin, 2009; Turner et al., 1998; Zhou et al., 2013).

Meanwhile, the shortages of the DPSIR approach have drawn increasing attentions. These weaknesses include not being able to capture the complexity of the real world, being inappropriate as an analytical tool, ignoring temporal and spatial scale issues, overlooking social or political aspects, being poorly connected to ecosystem service approaches and having difficulties in gathering complete and consistent information of the operation mechanism of the causality chain (Kohsaka, 2010; Maxim et al., 2009; Potschin, 2009; Spangenberg et al., 2009; Svarstad et al., 2008). In order to enhance the applicability of the DPSIR model, some scientists have proposed conceptual improvements, such as combining DPSIR with the so-called “tetrahedral” framework linking the four aspects of sustainability (environmental, economic, social and political, see e.g. Maxim et al., 2009; Spangenberg et al., 2009), coupling DPSIR with ecosystem services (Atkins et al., 2011; Kandziora et al., 2013; Rounsevell et al., 2010) and enhancing the DPSIR model by complexifying the causal chain to a causal network to tackle the complexities of the real world (Niemeijer and de Groot, 2008a). Although the DPSIR model has drawn remarkable effort from scientists to conceptually improve and practically apply it, we find only very few studies proceeding quantitative analyses. Further developments of the DPSIR model and applying it in combination with other approaches for quantitative studies will better exploit its potential and increase its effectiveness in treating human-environmental issues.

1.4. Case study: Jiangsu, China

China has experienced a remarkable economic growth and a rapid urbanization process in the past three decades. Started from the coastal areas in the east and south of China, this trend has also been spread to China’s middle and western areas. The fast economic growth, urbanization as well as industrialization have on the one hand, greatly increased the wealth of the whole society of China and generally raised Chinese people’s living standards and on the other hand, considerably altered China’s natural environment across the whole country (Zhang and Wen, 2008). Under the background of the whole country’s fast development, China’s rural areas have seen significant changes, including considerable land use conversions gradually sprawling from the east to the west and enormous labor migrations from the west to the big cities in the most developed areas (Ho and Lin, 2004). The noticeable socioeconomic changes in the rural areas of different regions have not only altered rural population’s production and living patterns, but also affected the local biodiversity, ecosystem’s capacity to provide services and local residents’ well-being (Zhou et al., 2013). Owing to the concordance or inconsistency between the regional socioeconomic dynamics and the ecological status or rural populations’ well-being, it is imperative to investigate the relationships between them through empirical studies. The evidence from China’s regional reality will provide reference for handling the human-environmental interaction issues in a broader context of the country. In our research, the issues of biodiversity, ecosystem services and human well-being are considered simultaneously. They are linked to socioeconomic aspects in the context of the Jiangsu province in the east of China, which comprises of an interesting regional divergence of socioeconomic features.

We aim at quantitatively exploring the socioeconomic influences on biodiversity, ecosystem services and human well-being. An integrated framework combining the ecosystem service

cascade and the DPSIR model was established as a theoretical basis of our analysis (Figure 1), allowing the linkage of socioeconomic factors with biodiversity, ecosystem services and human well-being. In this framework, ecological integrity³ including biodiversity equals the “State” sector of the DPSIR model and ecosystem services and human well-being equal the “Impact” component (Figure 1). By including the DPSIR model into our analysis, we can examine the quantitative applicability of this model as well as explore the quantitative linkages between the different sectors of the model, which may prove the causality between the sectors. Based on this, we propose try to answer the following two research questions of this study:

- 1) *Can the DPSIR model be used to quantitatively analyze the socioeconomic influences on biodiversity, ecosystem services and human well-being in human-environmental systems?*
- 2) *What are the quantitative linkages between the different sectors of the DPSIR model in the Jiangsu human-environmental system?*

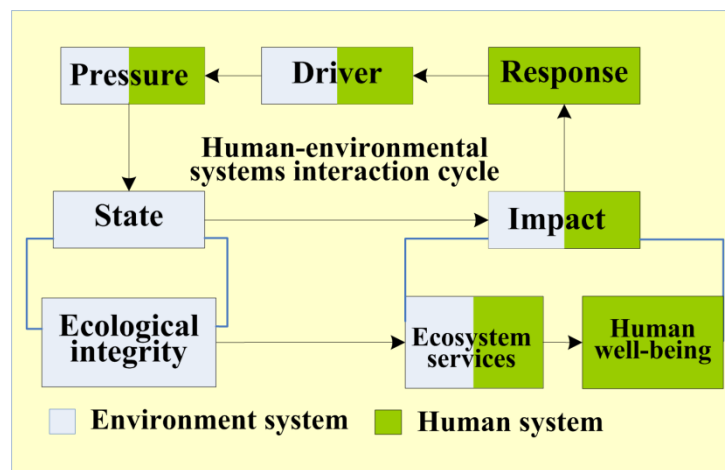


Figure 1. Linkage of DPSIR and ecological integrity/ecosystem services/human well-being in human-environmental systems (after Haines-Young and Potschin, 2010; Kandziora et al., 2013)

In this paper we firstly establish an indicator set matching our conceptual framework and then derive data from the research area to quantify the indicators. Afterwards we identify regional divergences in drivers, biodiversity, ecosystem services and human well-being in the research area. In order to find out the socioeconomic influences, we perform correlation analyses and factor analyses to investigate the relationships between the variables of the different DPSIR sectors. In a next step, we explore the quantitative linkages between each DPSIR sector by using correlation analysis and in the end discuss the results as well as the limitations of our work.

³ Ecological integrity denotes the “support and preservation of those processes and structures which are essential prerequisites of the ecological ability for self-organization” of ecosystems (Barkmann et al., 2001; p. 99).

2. Methods and data sources

2.1. Study area

Jiangsu province is composed of 13 prefecture-level cities⁴ (PLCs) located in the east of China (116°21'–121°56' E, 30°45'–35°07' N) with an overall area of $10.26 \times 10^4 \text{ km}^2$ and a population of 75.5 million in 2006 (SBJP, 2007) (Figure 2). About 70% of Jiangsu's total area is covered by plains and 18% by water bodies. Most of the area is not more than 50 meters above sea level. The climate types change from subtropical monsoon to temperate monsoon from the south to the north with the average annual precipitation ranging from 1200 mm in the southeast to 800 mm in the northwest. The average annual temperature ranges from 16 °C to 13°C, decreasing from the south to the north (Xu et al., 2011). Concerning the average annual solar radiation, the values range between 4450 and 4800 $\text{MJ} \cdot \text{m}^{-2}$ for most of the area with higher radiation in the northwest and lower radiation in the southwest (Wang et al., 2010).

Jiangsu is one of China's most developed regions, primarily due to the high urbanization and industrialization of the cities in the south of the Yangtze River. The domestic gross production (GDP) of Jiangsu was about 271.6 billion US dollars in 2006, which ranked third out of Chinese provinces (SBJP, 2007). The long agricultural productions periods, the rapid increasing industry and the expansion of human inhabitation have had significant impacts on the local environment. An apparent consequence is that the dominant land covers of Jiangsu are farmland and artificial area (arable land accounted for 45.4% and artificial area 15.9% of the total area in 2006, DLRJP, 2006).

⁴Prefecture-level city is a low level administrative division of province in China, including urban and rural areas.

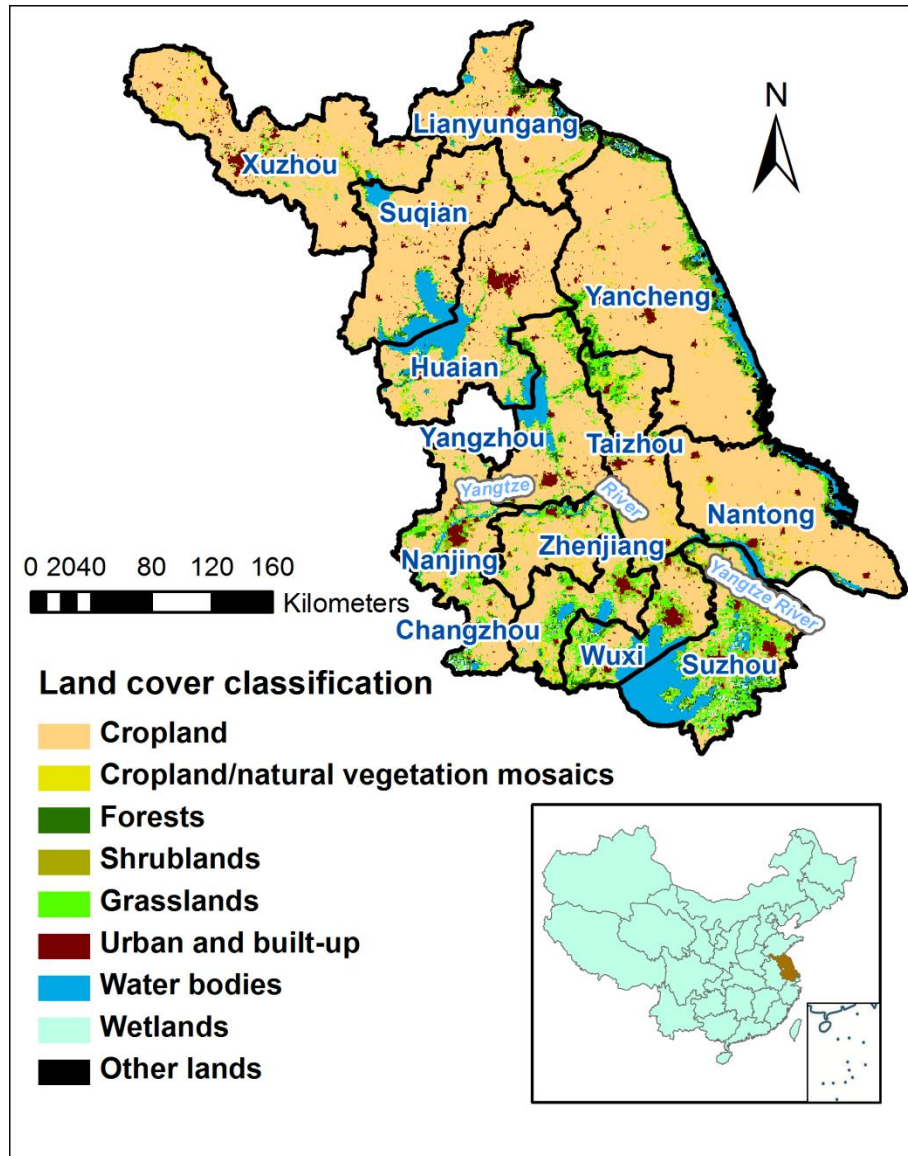


Figure 2. Land cover classification of the 13 prefecture-level cities of Jiangsu in 2006. The land cover classification is aggregated to nine types as shown in the map on the basis of the IGBP global vegetation classification scheme. Data sources: MODIS land cover type product⁵, National Fundamental Geographic Information System⁶.

2.2. Indicator set and data sources

A DPSIR indicator set for Jiangsu was derived by assigning several indicators to each sector of the DPSIR conceptual model (Table 1). The fundamental consideration for the indicator selection and assignment was the causal relationships between the DPSIR sectors (Maxim et al., 2009), i.e., the basic causal chain linkages from drivers to response and back to the drivers (Figure 1). In

⁵<http://webmap.ornl.gov/>[accessed at 04.30.2014]

⁶<http://nfgis.nsd.gov.cn/>[accessed at 04.30.2014]

other words, the indicators chosen for different DPSIR sectors should have potential impacts from one sector to another along the causal chain. We assumed the potential causal relationships between the indicators based on our understanding of the socioeconomic system dynamics and related information carried by the indicators and other researchers' arguments (e.g. in Burkhard and Müller, 2008; Gabrielsen and Bosch, 2003; Kristensen, 2004; Turner et al., 1998). For example, "population density", "Proportion of arable land area in total land area" and "richness of wild higher plants" were assigned to Driver, Pressure and State, respectively. The indicators represent urbanization levels, agricultural land use and regional biodiversity, respectively. These three attributes have been proven to have intrinsic causalities, i.e. the former ones have influential effects on the latter ones (see Turner et al., 1998; Zhou et al., 2013). Additionally, the selection was based on the aim of our research: exploring the socioeconomic influences on biodiversity, ecosystem services and human well-being and the research context of human-environmental systems. Therefore, the chosen indicators mainly relate to these components. Thirdly, we considered indicator representativeness, data availability, uniformity and completeness. For example, indicators for which data are lacking for the 13 samples (the 13 PLCs) were skipped. Consequently, the established indicator set is far less than comprehensive enough and some potentially important indicators are not included (e.g., State measures indicating water, soil and air qualities, many ecosystem service indicators as well as measures of human health). Our intention was to provide an example application of the DPSIR model to explore and quantify socioeconomic interrelations rather than depicting all conceivable influences on comprehensive sets of ecosystem services and human well-being.

The selected indicators were mainly intensity measures (ratios), except "total food crops output" and the four biodiversity indicators. Concerning the driver indicator "total food crops output", we assumed that it is more suitable to represent the actual food demand for the PLCs of Jiangsu than the mainstreaming indicator "food demand", which has traditionally been conceived as a key driver of land use change (Ericksen et al., 2009). Our consideration was based on the fact that the food demand of some prefectural-level cities was partly met by *ex-situ* supply, especially for the areas with relatively small amount of arable land south of the Yangtze River. As this part of food demand met by food import does not virtually drive the local land use change, using the alternative indicator measuring the local food output would more accurately indicate the drivers of the local land use change or other local agricultural activities.

The units, maximum, minimum and mean values, explanations and data sources of the indicators are given in Table 1. The data for the indicators were primarily derived from official sources such as the Statistical Bureau of Jiangsu Province and the Department of Land and Resources of Jiangsu Province. The data for the four biodiversity indicators were provided by the College of Economics and Management of Nanjing Agricultural University and the sample values of the indicator "organic carbon density" were calculated with the land use survey data and the organic carbon storage coefficients of different land use classes of Jiangsu given by Chuai et al. (2011).

Tables 1. Indicators of the DPSIR sectors, their units, values, explanations and data sources

Table 1.1. Indicators of Driver, their units, values, explanations and data sources

Indicators	Units	Maximum, minimum	Indicator explanations	Data sources
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		and mean		
		values		
Driver	Total food crops output	10000 t	535.33, 77.83, 248.38	Includes rice, maize, wheat, potato, bean and broomcorn Statistical Yearbook of Jiangsu (2007), from Statistical Bureau of Jiangsu Province (SBJP)
	Population density	persons/km ²	962, 474, 748	Calculation: total population/total area Statistical Yearbook of Jiangsu (2007), from SBJP
	Proportion of urban built-up area in total	%	8.73, 0.42, 2.09	Indicates urbanization level, calculation: area of urban built-up area in municipal districts/ total area Statistical Yearbook of Jiangsu (2007), from SBJP; China's Socio-Economic Development in municipal Statistical Database, from China National Knowledge Infrastructure(CNKI)
	Proportion of urban population in total	%	76.38, 32.35, 51.34	Indicates urbanization level, calculation: urban population/total population Statistical Yearbook of Jiangsu (2007), from SBJP

Industrial sector proportion in GDP	%	61.79, 34.32, 46.87	Indicates industrialization level, calculation: industrial share of GDP/GDP	Statistical Yearbook of Jiangsu (2007), from SBJP
GDP per person	yuan/capita	78801.05, 8618.43, 31440.79	Indicates industrialization level	Statistical Yearbook of Jiangsu (2007), from SBJP
The ratio of gross output value of agriculture to GDP	%	24.94, 1.38, 10.63	Indicates the relative size of agricultural economy, calculation: agricultural gross output/GDP	Statistical Yearbook of Jiangsu (2007), from SBJP

Table 1.2. Indicators of Pressure, their units, values, explanations and data sources

Indicators	Units	Maximum, minimum and mean values	Indicator explanations	Data sources
Pressure Arable land per capita	ha/capita	0.10, 0.03, 0.06	Indicates relative amount of arable land, calculation: arable land area/total	Statistical Yearbook of Jiangsu (2007), from SBJP; Land use change survey

			population (includes data of Jiangsu land for vegetables (2006), from and crops, not Department of Land includes land for and Resources of fruits) Jiangsu Province (DLRJP)
Proportion of arable land area in total land area	%	58.76, 28.23, 45.28	Indicates relative amount of arable land Statistical Yearbook of Jiangsu (2007), from SBJP; Land use change survey data of Jiangsu (2006),from DLRJP
Total power of agricultural machinery per unit sown area	W/ha	7984.07, 2264.47, 3851.16	Indicates agricultural technology level, calculation: total power of agricultural machinery/total sown area of crops Statistical Yearbook of Jiangsu (2007), from SBJP
Level of chemical fertilizer use	kg/ha	682.18, 311.25, 436.01	Calculation: amount of chemical fertilizer used in rural area (converted to pure substance)/arable land area Statistical Yearbook of Jiangsu (2007), from SBJP; Land use change survey data of Jiangsu (2006), from DLRJP

Level of pesticide use	kg/ha	32.98, 1.62, 7.79	Calculation: amount of pesticides used (converted to pure substance)/arable land area	China's Economic Development Statistical Database, from CNKI; Land use change survey data of Jiangsu (2006), from DLRJP
Irrigation rate	%	80.00, 42.75, 55.19	Indicates agricultural technology level, calculation: effectively irrigated area/total sown area of crops	Statistical Yearbook of Jiangsu (2007), from SBJP
Acid rain rate	%	76.40, 0, 31.52	Calculation: number of acid rains /number of rains, PH<5.6	Report on the State of Environment of the 13 PLCs of Jiangsu, from the Environmental Protection Bureaus of the 13 PLCs of Jiangsu

Table 1.3. Indicators of State, their units, values, explanations and data sources

Indicators	Units	Maximum,	Indicator	Data sources
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		minimum and mean values	explanations		
State	Richness of number wild higher plants	1437, 1072, 1221	Species number of wild higher plants, including monocotyledon, dicotyledon, gymnosperms and ferns	College Economics Management, Nanjing Agricultural University	of and
	Richness of number wild higher animals	398, 214, 320	Species number of wild higher animals, including, birds, fishes, amphibias, mammalias and reptilias	College Economics Management, Nanjing Agricultural University	of and
	Diversity of number ecosystem types	48, 28, 38	Number of ecosystem types (Wan et al., 2007), see (Wu, 1980) for classification system	College Economics Management, Nanjing Agricultural University	of and
	Number of number endemic	102, 53,	Includes endemic plant and animal	College Economics	of and

species		72	species	Management, Nanjing Agricultural University
Proportion of woodland, garden and grassland area in total	%	13.36, 0.95, 6.97	Includes the relative amount of non-cropland vegetation, including natural and artificial ones	Land use change survey data of Jiangsu (2006), from DLRJP

Table 1.4. Indicators of Impact, their units, values, explanations and data sources

Indicators	Units	Maximum, minimum and mean values	Indicator explanations	Data sources
Farming output value per chemical fertilizer use	yuan/kg	66.97, 30.23, 46.41	Calculation: gross farming value/rural chemical fertilizer uses(output does not include forestry, animal husbandry and fishing, the value is by the current year's	Statistical Yearbook of Jiangsu (2007), from SBJP

Impact I

(ecosystem services)				price)
	Food crops output per unit sown area	kg/ha	7256.85, 5837.88, 6478.51	Calculation: total food crops output/ total sown area of food crops Statistical Yearbook of Jiangsu (2007), from SBJP
	Agricultural labor productivity	yuan/capita	49421.46, 17693.91, 31238.34	Calculation: agricultural output value/agricultural labor population Statistical Yearbook of Jiangsu (2007), from SBJP (includes farming, forestry, animal husbandry and fishing, valued by the current-year price)
	Organic carbon density	kg/m ²	9.61, 8.77, 9.24	Calculation: organic carbon storage of all land uses/total land area (includes organic carbon stored in aboveground biomass, Land use change survey data of Jiangsu (2006), from DLRJP; (Chuai et al., 2011)

				belowground biomass and in soil)
	Ratio of %		16.68,	Calculation: total China's Socio-
	tourism		2.86,	tourism income/GDP Economic
	income to		8.47	(includes domestic Development
	GDP			income and foreign Statistical
				currency income, Database, from
				foreign currency is CNKI
				changed to RMB
				according to the
				current year's
				exchange rate)
Impact II	Rural	yuan/capita	9281.00,	Average net income Statistical
(human well-being)	residents' per		4228.00,	per capita in rural Yearbook of
	capita net		6213.38	area (for the total Jiangsu (2007),
	income			rural population) from SBJP
	Rural	yuan/capita	6811.00,	Average living Statistical
	residents' per		2797.00,	expenditure per Yearbook of
	capita		4469.23	capita for all the Jiangsu (2007),
	expenditure			rural population from SBJP
				(does not include
				producing
				expenditure)

Housing area per capita in rural area	m ² /person	65.30, 24.90, 42.28	Average housing area per capita for all the rural population	Statistical Yearbook of Jiangsu (2007), from SBJP
New rural cooperative medical insurance coverage	%	99.00, 90.60, 94.00	Proportion of rural population with new cooperative medical insurance (a new basic health social security system in China, started in 2003, combining insurance and social assistance, targeting all farmers and rural residents).	National Economic and Social Development Statistics Bulletins of the 13 PLCs of Jiangsu, from the statistics bureaus of the 13 PLCs of Jiangsu

Table 1.5. Indicators of Response, their units, values, explanations and data sources

Indicators	Units	Maximum, minimum and mean values	Indicator explanations	Data sources
Response Government agricultural expenditure	yuan/ha	7726.95, 523.20,	Calculation: fiscal expenditure on agriculture/total	Statistical Yearbook of Jiangsu (2007),

per unit sown area of crops		2229.38	sown area of crops	from SBJP; China's Socio-Economic Development Statistical Database, from CNKI
Agricultural loans per unit sown area of crops	yuan/ha	22320.98, 6588.21, 11522.40	Calculation: total amount of agricultural loan/total sown area of crops	Statistical Yearbook of Jiangsu (2007), from SBJP; China's Socio-Economic Development Statistical Database, from CNKI
Number of agricultural science and technology personnel	persons/per 10000 people	28, 8, 16	Number of employed people in agricultural science and technology	China's Socio-Economic Development Statistical Database, from CNKI
Years of rural education	years/person	8.82, 7.23, 7.95	Average schooling years of rural population	China's Socio-Economic Development Statistical

2.3. Identifying the divergence of drivers, state, ecosystem services and human well-being

There are socioeconomic state changes among the PLCs of Jiangsu (Liu et al., 2010), which can possibly cause the corresponding divergence of biodiversity, ecosystem services and human well-being (Zhou et al., 2013). Depicting these changes and generally identifying the correlations between the former and latter are the prerequisites to comprehensively explore the socioeconomic influential factors of the local biodiversity, ecosystem services and human well-being. To achieve this, we firstly performed a hierarchical cluster analysis using SPSS 17.0 to the 13 samples (PLCs) with the seven driver variables. The PLCs were grouped into different clusters based on their particularities reflected by these indicators. Then a descriptive statistical analysis was conducted with the identified clusters also with the driver indicators again using SPSS 17.0. This work aided in quantitatively sketching the differences of these indicators between the groups of PLCs. Afterwards we visualized the values of the indicators of State (biodiversity and non-crop land cover), ecosystem services and all human well-being variables for the 13 PLCs on maps using ArcGIS 10.0. These maps show the value distributions and comparisons among the PLCs. Meanwhile, they reveal the overall correlations between the differences of biodiversity, ecosystem services and human well-being and the divergence concerning the general driver status of the PLCs.

2.4. Statistical analysis of socioeconomic influences

We assumed that the socioeconomic influences could be quantitatively captured by the causal linkages between the indicators of Driver, Pressure, State, Impact and Response and the associated quantitative correlations (Bell, 2012). To quantitatively reveal these relationships, we carried out a Spearman correlation analysis between the indicators of the pair-wise DPSIR sectors. Spearman rank correlation is a nonparametric measure of statistical dependence between two variables, which assesses the monotonic degree of the relationship between two variables. As the results given in Online Appendix B showed high multicollinearities between the indicators of the same sector, it was feasible to reduce the dimensions of each sector to less underlying factors facilitating the demonstrations of the relationships between different sectors. In our research this dimension-reduction was achieved through factor analysis also using SPSS 17.0. Factor analysis is regarded as a typical method to examine the correlations between a large set of variables to see if a small number of potential factors can explain the variability in the original set of variables (Hinton et al., 2005). In the analyzing process, principal component analysis was chosen as the factor extraction method. It attempts to explain the maximum amount of variance with the minimum number of common factors (Hinton et al., 2005). The Kaiser-Meyer-Olkin (KMO) test and Bartlett's *test* of Sphericity were proceeded to examine the appropriateness of applying factor analysis to the data set of this research. The former test is a summary of how small the partial correlations between the variables are, relative to the original correlations. The latter one tests whether the correlation matrix is an identity matrix. The factors with an eigenvalue larger than 1 were retained. The factor loading matrix is given in the results section of this article, while the

other intermediate results including the KMO and Bartlett's test table, the total variance explained table, the component score coefficient matrices and the standardized scores of the factors of all samples are presented in Online Appendix A. Based on the results of the factor analysis, we correlated the principal components (factors) from one sector to another with Spearman correlation and found mutual quantitative relationships between the components. The causal linkages of the DPSIR model sectors led to the disclosure of the socioeconomic influences as described in the results part.

2.5. Additional influences: temperature and precipitation

Besides land cover and socioeconomic influences, temperature and precipitation also have impacts on biodiversity and ecosystem services (Gaston, 2000; Zhang et al., 2011), that possibly can impact human well-being. For example, temperature and precipitation are confirmed to be notably influential natural factors of crop production (Sun et al., 2012). Regarding Jiangsu, we also found high correlations between temperature or precipitation and some indicators of biodiversity, ecosystem services and human well-being. To eliminate the influences from these two environmental variables, we performed a partial correlation analysis between the indicators of biodiversity, ecosystem services and human well-being and the indicators of driver, pressure and response sectors by taking temperature and precipitation as control variables. Partial correlation measures the degree of association between two variables, with the effect of a set of controlling variables being removed. The results are presented in Tables B16-B18 of Online Appendix B. The socioeconomic influences that were modified by removing the impacts of the two environmental variables are depicted in the results section. The influences from temperature and precipitation on the results can be identified by comparing the findings before and after the partial correlation analyses.

2.6. Analyzing the DPSIR sectors' quantitative relationships

As referred to in the introduction, the DPSIR model describes not only internal causal chain relations but also potential causal network linkages between the different sectors. In order to quantitatively investigate these causal network linkages, we performed a Spearman correlation analysis with the DPSIR sectors. Before the correlation was conducted, the standardized scores of the sectors for the 13 samples were calculated by the equations combining the standardized scores of the sectors' components (shown in Table A4 of Online Appendix A) and the components' variance contribution rates in the extracted components of the associated sector. The equations are given in the results chapter and the standardized scores of the sectors are presented in Table A5 of Online Appendix A. Table 5 in the results section shows the correlation coefficients derived by correlating the DPSIR sectors' standardized scores.

3. Results

In this section the results produced by statistical methods and GIS visualizations are presented in the following sequence: 1) the differences of Jiangsu's prefecture-level cities concerning their Driver, State and Impact statuses, 2) the socioeconomic influences on Jiangsu's biodiversity, ecosystem services and human well-being and 3) the quantitative relationships between the DPSIR sectors.

3.1. Divergence of Jiangsu’s prefecture-level cities (PLCs) in Driver, State, ecosystem services and human well-being

The differentiations of the PLCs are based on a cluster analysis and descriptive statistics. The comparisons of the PLCs’ state and impact indicators are visually shown in this subchapter.

3.1.1. Driver clusters of Jiangsu’s PLCs

The result of the hierarchical cluster analysis of the 13 PLCs of Jiangsu showed that the cities can be separated into two groups according to the differences of the values of their driver indicators. The first cluster included 5 PLCs: Changzhou, Zhenjiang, Wuxi, Suzhou and Nanjing, whose areas are primarily in the south of Yangtze River. The second group included 8 PLCs in the north of Yangtze River: Huaian, Yangzhou, Yancheng, Xuzhou, Taizhou, Suqian, Nantong and Lianyungang (see Figure 2).

The descriptive statistical results are given in Table 2, which reveals the considerable differences of the driver indicator values between the two PLC clusters. Judging from the mean values, the first group generally had a lower total food crops output than the second cluster. The ratio of gross output value of agriculture to GDP of the first cluster was also much lower than that of the second one. Contrarily, the first group PLCs had larger mean values for the three urbanization indicators (population density, proportion of urban built-up area in total and proportion of urban population in total) than the second group. This comparison also applied to the indicators “industrial sector proportion in GDP” and “GDP per person”, which represent the level of industrialization. According to the comparison of the mean values of the driver indicators between the first and second PLC clusters, it can be seen that the first group of PLCs showed higher urbanization and industrialization levels, less agricultural production and a smaller agricultural economy size than the second group. In other words, the PLCs of the first cluster were more urbanized and industrialized than the ones of the second cluster.

Table 2. Comparison of the descriptive statistical results of the driver indicators with values for 2006 between the first and second cluster PLCs

		Units	First cluster			Second cluster		
			N	Mean	Std. Deviation	N	Mean	Std. Deviation
Total food crops output	10000 t	5	102.54	16.45	8	339.53	90.58	
Population density	persons/km ²	5	822.77	114.80	8	701.71	171.02	
Proportion of urban built-up area in	%	5	4.04	2.72	8	0.87	0.29	

total							
Proportion of urban population in total	%	5	65.65	6.81	8	42.40	5.54
Industrial sector proportion in GDP	%	5	54.26	7.04	8	42.25	6.14
GDP per person	yuan/capita	5	55986.40	18330.15	8	16099.63	5625.25
The ratio of gross output value of agriculture to GDP	%	5	2.76	1.31	8	15.56	6.94

3.1.2. Distributions and comparisons of ecological integrity, ecosystem services and human well-being of Jiangsu's PLCs

Figure 3 presents the distributions of the indicator values for the four biodiversity indicators and non-cropland vegetation cover (woodland, garden and grassland). Generally, the more urbanized and industrialized prefecture-level cities (PLCs) had higher values for the four biodiversity indicators than the less urbanized and industrialized ones. This phenomenon suggests that these areas had a better state of biodiversity as a whole. Meanwhile, these PLCs generally had a higher proportion of non-cropland vegetation cover. Concerning the performance of the more urbanized and industrialized PLCs in providing ecosystem services, they generally showed a higher agricultural productivity, a higher capacity for carbon storage and apparently more tourism service provision when compared to the less urbanized and industrialized PLCs (Figure 4). It should be noted that the PLCs with more total food crops output had an overall lower agricultural productivity (see the value distributions of the three indicators “total food crops output”, “food crops output per unit sown area” and “agricultural labor productivity” among the PLCs on the map). This trend is further analyzed in the following sections. Figure 5 indicates the comparison of human well-being for the rural population of Jiangsu's PLCs. Obviously, the rural residents of the more urbanized and industrialized PLCs enjoyed much better well-being, i.e., the rural inhabitants had much higher incomes, expenditures, housing and medical insurance levels.

The above results qualitatively indicate the correlations of urbanization and industrialization levels as well as agricultural production and economy sizes with regional biodiversity, general land cover, ecosystem services and rural human well-being. In the following, we will quantitatively and comprehensively explore the socioeconomic influences on biodiversity, ecosystem services and human well-being of Jiangsu by using factor and correlation analyses.

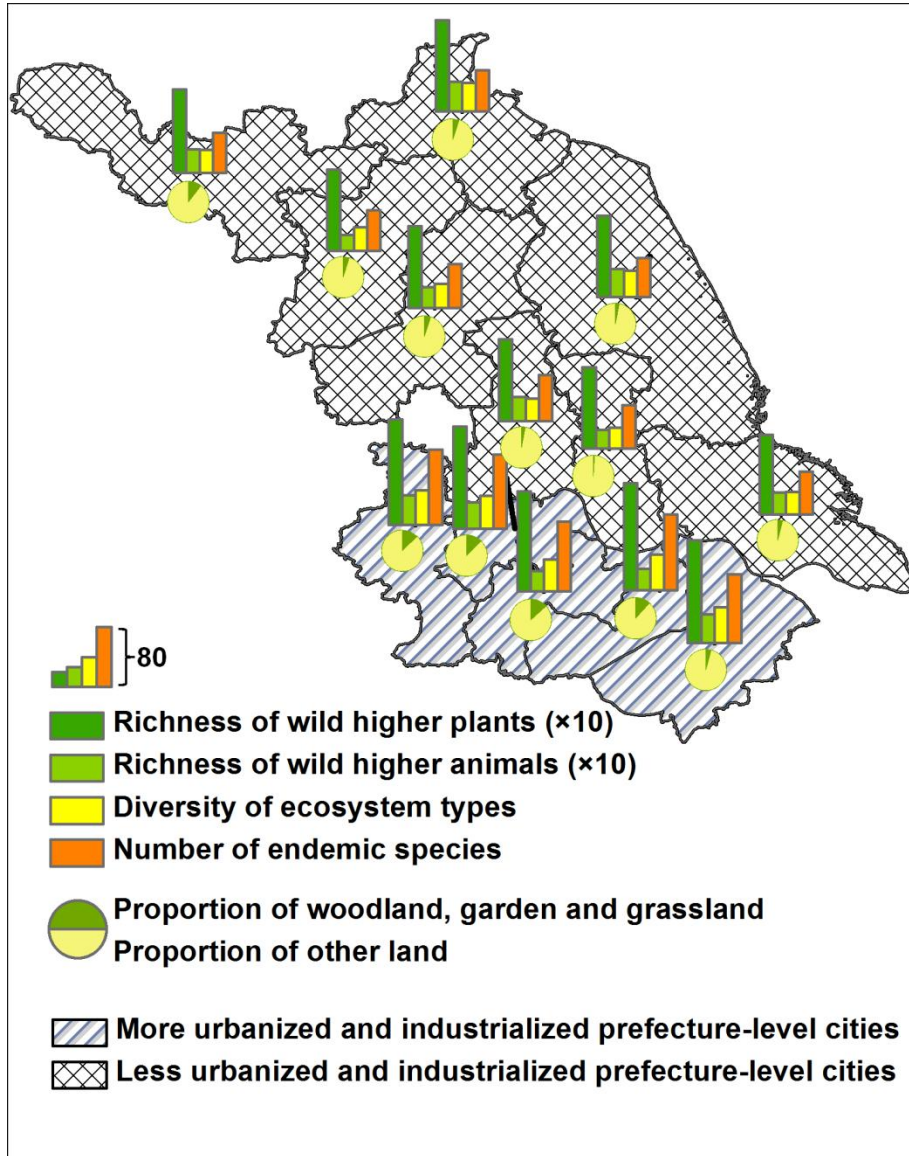


Figure 3. Biodiversity and non-cropland vegetation cover (woodland, garden and grassland) comparisons of the 13 PLCs of Jiangsu for 2006. The number “80” in the legend is a reference for the columns showing the biodiversity indicator values on the map.

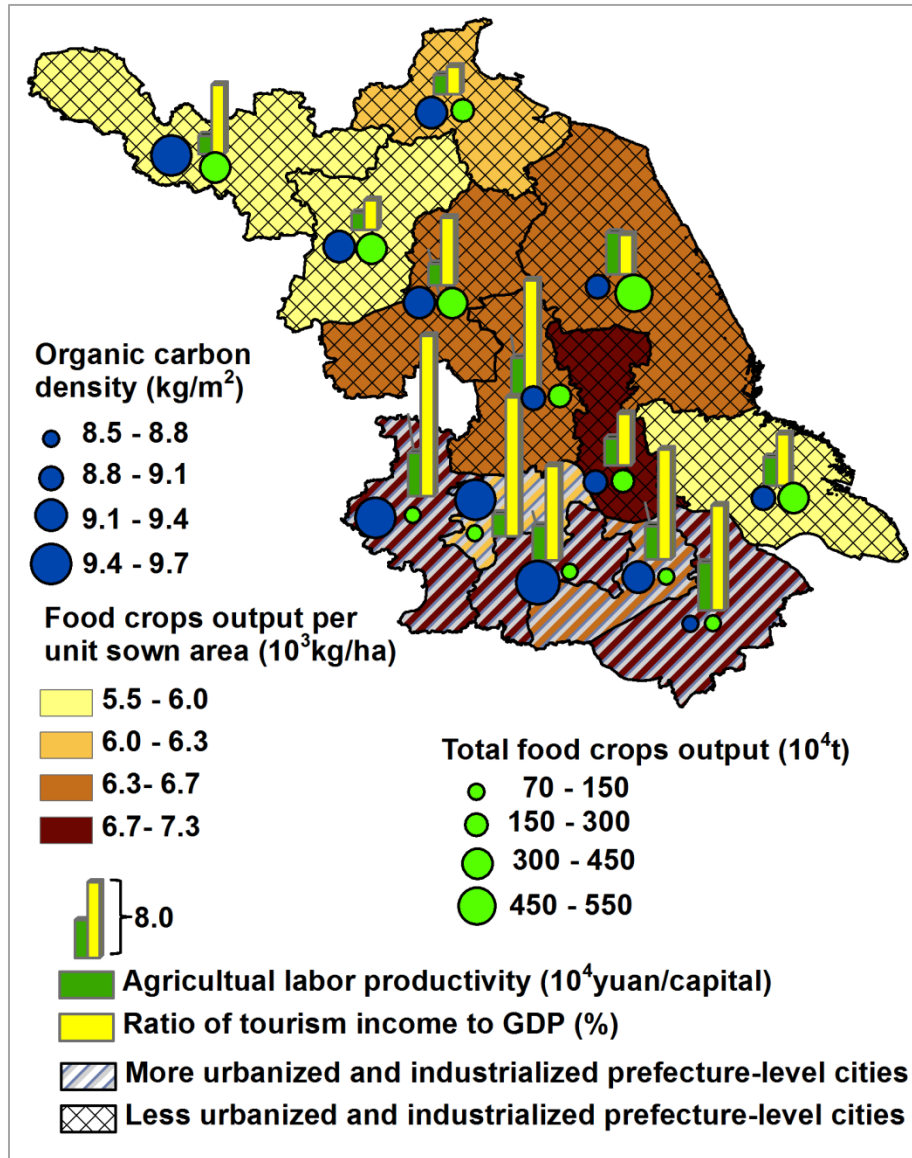


Figure 4. Ecosystem service comparison of the 13 PLCs of Jiangsu for 2006. The number “8.0” in the legend is a reference for the columns showing the values of “agricultural labor productivity” and “ratio of tourism income to GDP” on the map.

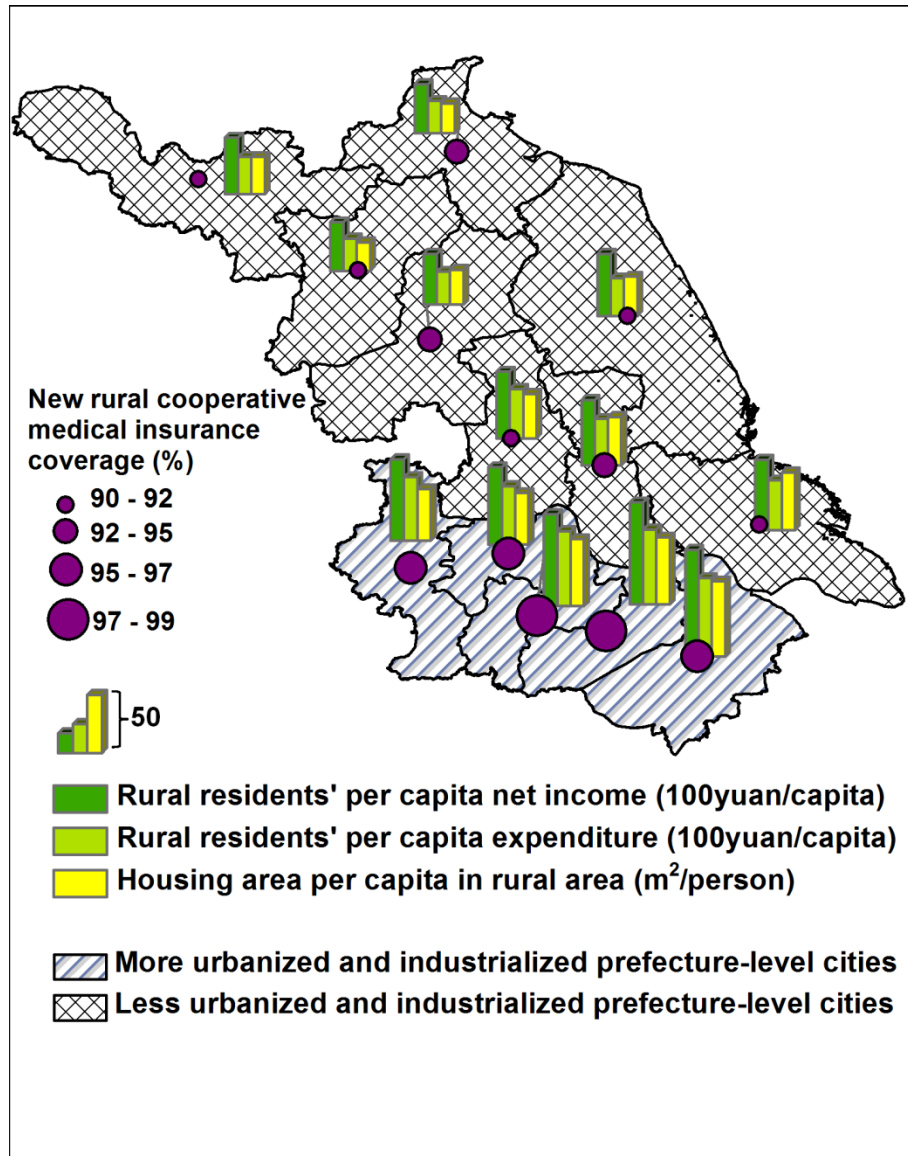


Figure 5. Rural human well-being comparison of the 13 PLCs of Jiangsu for 2006. The number “50” in the legend is a reference for the columns showing the associated three human well-being indicator values on the map.

3.2. Socioeconomic influences on biodiversity, ecosystem services and human well-being of Jiangsu

This section presents the findings derived by factor analysis, Spearman correlation and partial correlation analyses and demonstrates the socioeconomic influences.

3.2.1. Principal components of the D-P-S-I-R variables

Principal components for the indicators of each DPSIR sector were generated by factor analysis using SPSS 17.0 (Table 3). Meanwhile, the suitability of factor analysis concerning the data in this study was tested by KMO and Bartlett’s tests (see Table A1 of Online Appendix A). The test results indicate that a factor analysis is appropriate because the values of the KMO test for the

variables of all the DPSIR sectors exceeded 0.6 and the significance levels of the Bartlett's tests were all below 0.005. Factor rotation was not performed as a sole factor of the variables of each DPSIR sector highly loaded on the associated variable (except the loadings of the two factors on "level of pesticide use" of the Pressures and the factor's loading on "richness of wild higher animals" of State). This enables straightforward interpretations of the factors.

The component matrix of the driver sector shows that only one factor was derived, which was highly positively correlated with the three urbanization indicators and the two industrialization variables. In contrast, it was highly negatively correlated with "total food crops output" and "the ratio of gross output value of agriculture to GDP". Given this, this component can be interpreted as a positive index of urbanization and industrialization levels and a negative index of agricultural production and economy sizes.

Two factors were extracted for the variables of the pressure sector. The first one, explaining 58.3% of the total variance, was highly negatively associated with "arable land per capita" and "proportion of arable land area in total land area", highly positively associated with "total power of agricultural machinery", "irrigation rate" and "acid rain rate" as well as moderately positively correlated with "level of pesticide use". The second one had a highly positive loading on "level of chemical fertilizer use" and a moderate positive loading on "level of pesticide use". Note that the indicator "level of pesticides use" was moderately correlated with both of the two factors (even after the factor rotation was tested). The further use of it should be carefully considered (du Toit and Cilliers, 2011). Given that it had no significant correlations with the indicators of State and Impact (see Table B6-B8 of Online Appendix B), we excluded this indicator from the further analysis. In this sense, the first component of pressure indicators can be explained as a negative index of relative arable land amount and a positive index of agricultural technology level and pressure from industry (acid rain rate is normally closely related to the industrial level in developing countries, see its correlations with the industrialization level indicators in Table B1 of Online Appendix B, also see He et al., 2002); the second component indicates the fertilizer use.

The state indicators were reduced to one principal component, which was significantly positively correlated with "richness of wild higher plants", "diversity of ecosystem types", "number of endemic species" as well as "proportion of woodland, garden and grassland area in total". Obviously, this component was a positive index of plant species, ecosystem and endemic species diversity as well as non-cropland vegetation cover.

The ecosystem service variables were dimension-reduced to two factors. One had positive high loadings on "farming output value per chemical fertilizer use", "food crops output per unit sown area", "agricultural labor productivity" and "ratio of tourism income to GDP" and can be interpreted as a positive index of agricultural productivity and regional tourism service. The other one had a significantly positive loading solely on "organic carbon density" and can be named "positive index of carbon storage service".

Concerning the human well-being indicators, one principal component was derived. It was apparently a positive index of rural inhabitants' income, expenditure, housing and medical insurance level due to its highly positive correlations with the variables "rural residents' per capita net income", "rural residents' per capita expenditure", "housing area per capita in rural area" and "new rural cooperative medical insurance coverage".

Lastly, two factors were extracted from the indicators of Response. The first factor presented the agriculture financial and support and rural education levels as it was significantly positively

correlated with the associated variable; the second factor positively indicated agricultural science and technology supports.

Table 3. Principal components of the indicators of each DPSIR sector and their interpretations. Principal components were derived using factor analysis with SPSS 17.0. Components with an eigenvalue greater than 1 were retained in the components matrices (see Table A2 of Online Appendix A). The component matrices of each DPSIR sector were grouped in this table. All the components were interpreted according to the factor loadings they have on the associated indicators. The factor loadings with larger values mean stronger correlations of the components with the associated variables.

Indicators		Principal component 1		Principal component 2	
		Factor loading	Component interpretation	Factor loading	Component interpretation
Driver	Total food crops output	-0.900	Positive index of	--	
	Population density	0.704	urbanization and		
	Proportion of urban built-up area in total	0.724	industrialization levels and		
	Proportion of urban population in total	0.952	negative index of agricultural production and		
	Industrial sector proportion in GDP	0.809	economy sizes		
	GDP per person	0.900			
	The ratio of gross output value of agriculture to GDP	-0.959			
Pressure	Arable land per capita	-0.887	Negative index	0.094	Indicates
	Proportion of arable land area in total land area	-0.802	of relative arable land amount and	-0.021	chemical fertilizer uses
	Total power of agricultural	0.777	positive index of	0.174	level

	machinery per unit sown area		agricultural		
	Level of chemical fertilizer use	-0.222	technology level	0.926	
	Level of pesticides use	0.665	and pressure	0.565	
	Irrigation rate	0.943	from industry	0.082	
	Acid rain rate	0.815		-0.388	
State	Richness of wild higher plants	0.984	Positive index of		--
	Richness of wild higher animals	0.470	plant species,		
	Diversity of ecosystem types	0.966	ecosystem and		
	Number of endemic species	0.950	endemic species		
	Proportion of woodland, garden and grassland area in total	0.797	diversity as well		
			as non-cropland		
			vegetation cover		
Impact I (ecosystem service)	Farming output value per chemical fertilizer use	0.917	Positive index of	0.147	Positive index
	Food crops output per unit sown area	0.857	agricultural		of carbon
	Agricultural labor productivity	0.903	productivity and	-0.082	storage service
	Organic carbon density	-0.227	regional tourism		
	Ratio of tourism income to GDP	0.722	service	-0.288	
				0.936	
				0.565	
Impact II (human well-being)	Rural residents' per capita net income	0.983	Positive index of		--
	Rural residents' per capita	0.988	rural inhabitants'		
			income,		

	expenditure			expenditure,		
	Housing area per capita in rural area	0.957		housing and medical insurance level		
	New rural cooperative medical insurance coverage	0.831				
Response	Government agricultural expenditure per unit sown area of crops	0.902		Positive index of agriculture financial support level and rural education level	-0.166	Positive index of agricultural science and technology support level
	Agricultural loans per unit sown area of crops	0.949			-0.039	
	Number of agricultural science and technology personnel	0.118			0.989	
	Years of rural education	0.921			0.076	

3.2.2. Socioeconomic influences

The pair-wise Spearman correlation coefficients of the principal components of the DPSIR sectors' variables are presented by Table 4. The correlations were performed by the standardized scores of the components for the 13 PLCs (see Table A4 of Online Appendix A), based on the component score coefficient matrices and derived automatically by SPSS 17.0 (see Table A3 of Online Appendix A). The correlations of the components of State, ecosystem services and human well-being with the components of Driver, Pressure and Response revealed the mutual quantitative relationships. Combining these quantifiable linkages with the interpretations of the components, we can reveal the socioeconomic influences on biodiversity, ecosystem services and human-well-being of Jiangsu as described in the following paragraphs.

The positive socioeconomic influences on Jiangsu's plant species, ecosystem and endemic species diversity included urbanization and industrialization, the increases of agricultural technology and financial support levels as well as rural education level; the negative influences are primarily comprised of the growth of agricultural production and economy sizes and the expansion of arable land. We identified these positive and negative influences according to the results that the state principal component (SPC), representing the diversity of plant species, ecosystem and endemic species, was strongly positively correlated with the driver principal component (DPC), which is the positive index of urbanization and industrialization level and

negative index of agricultural production and economy sizes. Meanwhile, it also had strong positive correlations with the pressure principal component 1 (PPC1). This component was a negative index of relative arable land amount and a positive index of agricultural technology level and pressure from industry. Furthermore, SPC was significantly positively associated with the response principal component 1 (RPC1), which was a positive index of agriculture financial support and rural education.

Concerning the ecosystem services of Jiangsu, agricultural productivity and regional tourism service were probably positively driven by the increase of urbanization and industrialization levels, agricultural technology and financial support levels as well as rural education level. Contrarily, these two services were probably negatively influenced by the growth of agricultural production and economy sizes, the expansion of arable land area and the increase of chemical fertilizer use. These influences were found due to the strong positive correlations of the ecosystem service principal component 1 (ESPC1), which is a positive index of agricultural productivity and regional tourism service, with DPC, PPC1, RPC1, and additionally, due to its significant negative correlation with the pressure principal component 2 (PPC2) indicating the chemical fertilizer uses level (Table 4). The ecosystem service “carbon storage” (represented by the ecosystem service principal component 2 with the abbreviation ESPC2) was strongly positively impacted by the non-crop land vegetation cover as the associated indicator “organic carbon density” was highly significantly correlated with the state indicator “proportion of woodland, garden and grassland area in total” with the Spearman correlation coefficient of 0.801 (see Table B10 of Online Appendix B).

The positive influences on the rural population’s overall well-being from Jiangsu’s society and economy essentially consisted of the enhanced urbanization and industrialization, the boosting of agricultural technology and financial support levels, the rural education level and the agricultural productivity. In contrast, the increase of agricultural production and economy sizes and the expansion of arable land amount constituted the most important negative influences on the rural habitants’ income, expenditure, housing and medical insurance levels. We can achieve this revelation according to the evidence shown in Table 4, that the human well-being principal component (HWPC), which positively related to the rural inhabitants' income, expenditure, housing and medical insurance level, was significantly correlated with DPC, PPC1, ESPC1 and RPC1 (see the full names and interpretations of these components’ acronyms in the above paragraphs or in the notes of Table 4).

Table 4. Correlations between the principal components of the DPSIR sectors’ indicators

	DPC	PPC1	PPC2	SPC	ESPC1	ESPC2	HWPC	RPC1	RPC2
DPC	1.000	0.951**	n.s.	0.676*	0.720**	n.s.	0.907**	0.923**	n.s.
PPC1	0.951**	1.000	n.s.	0.626*	0.731**	n.s.	0.874**	0.934**	n.s.
PPC2	n.s.	n.s.	1.000	n.s.	-0.577*	n.s.	n.s.	n.s.	-0.665*
SPC	0.676*	0.626*	n.s.	1.000	n.s.	0.555*	0.582*	0.654*	n.s.
ESPC1	0.720**	0.731**	-0.577*	n.s.	1.000	n.s.	0.824**	0.769**	n.s.

ESPC2	n.s.	n.s.	n.s.	0.555*	n.s.	1.000	n.s.	n.s.	n.s.
HWPC	0.907**	0.874**	n.s.	0.582*	0.824**	n.s.	1.000	0.934**	n.s.
RPC1	0.923**	0.934**	n.s.	0.654*	0.769**	n.s.	0.934**	n.s.	n.s.
RPC2	n.s.	n.s.	-0.665*	n.s.	n.s.	n.s.	n.s.	n.s.	1.000

Spearman correlations, ** P< 0.01, *P<0.05, n.s. P≥0.05. Correlation coefficients in the table are based on the standardized scores of the principal components of the DPSIR sectors' indicators (see Table A4 of Online Appendix A). Full names of the principal components and their interpretations are as follows:

Driver principal component (DPC): positive index of urbanization and industrialization level and negative index of agricultural production and economy sizes;

Pressure principal component 1 (PPC1): negative index of relative arable land amount and positive index of agricultural technology level and pressure from industry;

Pressure principal component 2 (PPC2): indicates chemical fertilizer uses level;

State principal component (SPC): Positive index of plant species, ecosystem and endemic species diversity as well as non-cropland vegetation cover;

Ecosystem service principal component 1 (ESPC1): positive index of agricultural productivity and regional tourism service;

Ecosystem service principal component 2 (ESPC2): positive index of carbon storage service;

Human well-being principal component (HWPC): positive index of rural inhabitants' income, expenditure, housing and medical insurance level;

Response principal component 1 (RPC1): positive index of agriculture financial support level and rural education level;

Response principal component 2 (RPC2): positive index of agricultural science and technology support level.

3.2.3. Results modification by considering temperature and precipitation influences

Even after the results' modification considering temperature and precipitation effects, the partial correlations between the biodiversity indicators and those of Driver, Pressure and Response showed noticeable similarities to the results of the associated Spearman correlations (see Table B2, B6, B12 and B16 of Online Appendix B). Exceptions were "the ratio of gross output value of agriculture to GDP" that was no longer significantly correlated with the biodiversity indicators and the indicators representing agricultural technology and financial input levels that were less closely correlated with some biodiversity indicators. Hence, the findings of the socioeconomic influential factors of biodiversity can be modified as follow: urbanization and industrialization are significantly positive socioeconomic influential factors of Jiangsu's plant species, ecosystem and endemic species diversity; the increases of agricultural technology and financial support levels as well as rural education level have positive influences on Jiangsu's biodiversity to some extent; the negative influential factors primarily include the increase of total food production and the expanding of arable land.

The partial correlations between the indicators of ecosystem services and those of Driver, Pressure and Response indicated overall similarities to the results of the associated Spearman correlations, whereas, some differences can be seen by comparing these results (see Table B3, B7, B14 and B17 of Online Appendix B). The primary distinctions include that agricultural production and economy size indicators did not highly negatively correlate with agricultural productivity indicators any more. Additionally, agricultural technology and rural education levels showed loose correlations with agricultural productivity. As we presumed that the limited variations of temperature and precipitation of Jiangsu had little influence on the tourism service, we did not consider the partial correlation results of the indicator “ratio of tourism income to GDP” with the driver, pressure and response indicators. Hence, the revisions can be formulated as follows: agricultural productivity and regional tourism service were probably positively driven by the increase of urbanization and industrialization levels in some degree in Jiangsu. In the contrary, these two services were very likely negatively driven by the expansion of arable land. Additionally, agricultural financial support has some positive impact on agricultural productivity and non-crop land vegetation cover probably has significantly positive effects on the carbon storage service.

Referring to the correlations between the indicators of rural residents’ well-being and the indicators of Drivers, Pressure and Response, the partial correlation results were similar to the Spearman correlations (see Table B4, B8, B15 and B18 of Online Appendix B). The major changes involved the disappearance of the high correlations of “total food crops output” with the three indicators indicating rural population’s income, expenditure and housing levels and the high correlations of “the ratio of gross output value of agriculture to GDP” with all the four human well-being indicators. Besides, the urbanization level indicators lost a large part of significant correlations with the four human well-being indicators, except the moderate correlations of “proportion of urban population in total” with the indicators showing rural residents’ income, expenditure and medical insurance levels. Thus, the socioeconomic influential factors can be modified as follows: For the rural population’s overall well-being in Jiangsu, the significant socioeconomic positive influences consisted of the enhancement of industrialization, the increase of agricultural technology level and financial support as well as the rural education level. Comparatively, urbanization had moderately positive impacts on the income, expenditure and medical insurance levels of the rural population. Conversely, the growth of food production was related to the decrease of rural population’s medical insurance level and the increase of arable land probably caused the reduction of the income, expenditure and medical insurance levels of Jiangsu’s rural population.

3.3. Quantitative linkages between the DPSIR sectors of Jiangsu’s human-environmental system

The standardized scores of each DPSIR sector were calculated with the following equations, whereby the coefficient for each principal component is its variance contribution rate in the extracted components (see Table A2 and A4 of Online Appendix A and the results in Table A5):

$$\text{Driver score} = 1 \times \text{DPC score} \quad (1)$$

$$\text{Pressure score} = 0.748 \times \text{PPC1 score} + 0.252 \times \text{PPC2 score} \quad (2)$$

$$\text{State score} = 1 \times \text{SPC score} \quad (3)$$

$$\text{Ecosystem service score} = 0.694 \times \text{ESPC1 score} + 0.306 \times \text{ESPC2 score} \quad (4)$$

$$\text{Human well-being score} = 1 \times \text{HWPC score} \quad (5)$$

$$\text{Response score} = 0.718 \times \text{RPC1 score} + 0.282 \times \text{RPC2 score} \quad (6)$$

Table 5 presents the Spearman correlation coefficients between the DPSIR sectors, which were based on the standardized scores of each DPSIR sector for the 13 PLCs of Jiangsu (see also Table A5 of Online Appendix A). The correlation coefficients and the associated significance levels demonstrate the quantitative linkages between the DPSIR sectors as described in the following paragraph.

The Driver section was strongly correlated with human well-being, Response, Pressure and ecosystem services at the significant level of 0.01. Additionally, it correlated to State at the significance level of 0.05. Pressure was most significantly correlated with Driver and State at 0.01 level and comparatively weaker but still significant correlated with human well-being, Response and ecosystem services at 0.05 level. State was solely significantly correlated with Pressure at 0.01 level. As for the other sectors, the correlation coefficients were smaller but still significant at 0.05 level. Concerning ecosystem services, they were most closely linked with human well-being, followed by Response and Driver and least closely connected with State and Pressure. Human well-being also had apparent quantitative linkages with all of the other sectors. At the significance level of 0.01, it was highly correlated with Response, Driver and ecosystem services with the correlation coefficient values all greater than 0.8. Its correlations with Pressure and State were not as strong as with the other three sectors, but still significant at 0.05 level with the coefficient values both more than 0.55. Last but not least are the relationships of Response with other sectors. It was apparently closely linked to human well-being, Driver and ecosystem services at the significance level of 0.01. In addition, Response was correlated to State and Pressure with the correlation coefficients both greater than 0.6.

Table 5. Correlations between the DPSIR sectors

	Driver score	Pressure score	State score	Ecosystem service score	Human well-being score	Response score
Driver score	1.000	0.819**	0.676*	0.731**	0.907**	0.857**
Pressure score	0.819**	1.000	0.753**	0.555*	0.659*	0.604*
State score	0.676*	0.753**	1.000	0.676*	0.582*	0.648*
Ecosystem service score	0.731**	0.555*	0.676*	1.000	0.835**	0.819**
Human well-being score	0.907**	0.659*	0.582*	0.835**	1.000	0.945**
Response score	0.857**	0.604*	0.648*	0.819**	0.945**	1.000

Spearman correlations, ** P< 0.01, *P<0.05. The correlations are based on the standardized scores of the DPSIR sectors (see Table A5 of Online Appendix A).

4. Discussion

This article aims at providing an empirical study of quantifying the socioeconomic influences on biodiversity, ecosystem services and human well-being in human environmental systems and quantitatively examining the linkages between the different sectors of the DPSIR model. To achieve this purpose, we proposed a conceptual framework coupling the DPSIR model and a simplified form of the ecosystem service cascade framework and established an indicator set based on this framework. The results were produced by statistical analyses and mapping with the empirical data of the case study area. These findings proved the quantitative relationships between several socioeconomic aspects and biodiversity, ecosystem services and human well-being. The quantifiable connections between the sectors of the DPSIR model were also disclosed by statistics.

4.1. Socioeconomic influences

4.1.1. Effects on biodiversity

According to the adjusted socioeconomic influences based on the impacts from temperature and precipitation, urbanization and industrialization probably have significantly positive driving effects on plant species, ecosystem and endemic species diversity of the study area. This finding seems to be unexpected and unreasonable as it contradicts the mainstreaming arguments of

environmental management. For instance, Pillsbury and Miller (2008) find negative influence of urban density on overall anuran abundance and diversity in central Iowa, USA. Stratford and Robinson, (2005) also indicate strong negative effects of increased urban cover on the species richness of neotropical migratory birds across an urban-rural gradient in the southeastern United States. However, when considering the spatial scale issue and linking this result with some other findings, the positive role of urbanization and industrialization becomes reasonable. These findings relate to the obviously negative effect on biodiversity from increasing total food production and expanding arable land. Faggi et al., (2008) demonstrate the great danger of cultivation practices to the conservation of biodiversity, which was even more dangerous than urban sprawl. The authors found significant high plant and bird species diversity in urban reserves, peri-urban and semirural areas, which equals or exceeds those in rural areas. Additionally, moderate biodiversity was found in urban parks and suburban areas of their study area. The authors argued that agriculture practices would greatly alter natural habitats and consequently, threat biodiversity. On the contrary, the “intermediate disturbance hypothesis” may explain the increase of biodiversity at the moderate levels of urbanization (Blair, 1996; Connell, 1978). Another piece of evidence is related to Hope et al. (2003), who proved a higher diversity of perennial plant genera in urban areas than in agricultural areas in the Central Arizona–Phoenix region of the US. Agriculture’s great threats to biodiversity are furthermore proved by Ribeiro et al.’ (2009) study in Catalonia, Spain and also Kuldna et al. (2009) argue that the adverse effects of agriculture on pollinators can be a pressure on the remaining biodiversity.

Regarding the findings of this research, the more urbanized and industrialized PLCs in the south of the Yangtze River facilitated the migration of the rural population from rural to urban areas in a higher level compared to the PLCs in the north of the Yangtze River. Additionally, these areas had a higher transformation of agricultural production in the rural area to industrial production in the urban area (see Table 2 for the mean values of the driver indicators of the two PLCs clusters). Consequently, the former had a lower proportion of arable land and less agricultural activities in the rural area, which was more beneficial for habitat conservation or restoration than the latter (see Figure 2 for the land cover of the PLCs). Furthermore, the biodiversity of the (most likely) intermediately disturbed areas across the urban-rural gradient of Jiangsu could be relatively high according to the “intermediate disturbance hypothesis”. Two empirical studies respectively concerning Suzhou, a prefecture-level city in the south of Jiangsu and Shanghai bordering Jiangsu on the south provide evidence to this assumption. The former study proves that the species richness of artificial forests exceeds those of farm land and natural forests (Qi, 2008). The latter exploration in Shanghai finds that the districts with intermediate population and road densities have higher overall biodiversity than the most populated districts around the city center. Meanwhile, these areas also have higher biodiversity compared to the districts with least human activities most remote from the city center (Wang et al., 2012).

Another positive socioeconomic influential factor of biodiversity is the increase of agricultural technology and financial support levels as well as rural education level for the PLCs of Jiangsu. The beneficial role of this influence may also involve the reduction of agricultural area and the retention of non-cropland vegetation covers as these factors can lead to a growth of agricultural productivity, which demands less agricultural land for the same amount of production (see the correlations between pressure indicators and response indicators in Table B9 of on Online Appendix B).

4.1.2. Effects on ecosystem services

The revised findings show that urbanization, industrialization as well as agricultural financial support have positive influences on Jiangsu's agricultural productivity. Contrarily, the expansion of arable land had primarily negative effects. These results are consistent with other scientists' findings. For instance, Azadi et al. (2011) found significantly positive correlations between urbanization and productivity, Jiang and Wang (2012) proved the stimulating effect of industrialization on agricultural modernization including the increase of agricultural productivity and Alcamo et al. (2005) pointed out that investments on agricultural research would result in yield increases without large growing of new arable area. Chen et al. (2012) indicated the roles of urbanization and industrialization in the development of agricultural productivity. For example, urbanization and industrialization can both propel the migration of rural surplus workforce to urban areas and enhance the development of agricultural science and technology levels. Additionally, revenues from industrial production can be invested on agriculture by local fiscal policies.

Urbanization and industrialization were also proved to very likely have positive impacts on the tourism service of Jiangsu (in this study the tourism service primarily refers to urban tourism), which has also been found in some other empirical studies in China (Cao et al., 2012; Yu et al., 2012). This result can be explained by the supply and demand of tourism in industrialized urban areas. From this aspect, the improvement of the urban environment, infrastructure and the associated services in the post-industrial urban areas enhances the cities' capacity to provide tourism services (Luo and Tan, 2012). Besides, tourism's demand from urban residents grows with the increase of urban population and the rise of their income and consumption levels, which can be caused by the increase of industrial employment (Cao et al., 2012). Concerning our research area, the more urbanized and industrialized prefecture-level cities in the south of the Yangtze River provide more tourism service than most of the areas in the north of the Yangtze River. This is not only caused by the influences mentioned above, but also for a reasons that this area named "Jiangnan", which means the south of the lower reaches of the Yangtze River in China, has been rich in history and culture since ancient China and a famous tourism attraction for Chinese and foreigners.

In contrast to urbanization and industrialization, the expansion of arable land in the research area was evidenced to be probably negatively affecting tourism services. This finding is understandable as arable land expansion is often closely related to extensive agriculture, which usually occurs in the regions of China with comparatively low urbanization and industrialization levels and having relatively high dependency on the agricultural economy (Lu et al., 2011). Referring to the explanations in the former paragraph, the less urbanized and industrialized areas should have less tourism compared to the areas with higher urbanization and industrialization levels. Specifically considering Jiangsu's situation, the cities in the north of the Yangtze River are not conventional tourism attractions, when compared to the "Jiangnan" region in the south of the Yangtze River. This should be another important reason for their smaller urban tourism service supply.

For the ecosystem service of carbon storage, non-crop land vegetation cover was confirmed to probably have considerably positive effects, which consists with extensive body of evidence (Burkhard et al., 2009; Costanza et al., 1997; Li et al., 2010; Xie et al., 2008). This causality is apparent as the carbon storage capacity of non-crop land vegetation covers (e.g. forest, woodland,

shrubs and grassland) is much higher than that of cropland or other land cover classes (Burkhard et al., 2009).

4.1.3. Effects on human well-being

As indicated by the modified findings, industrialization, agricultural technology development and financial support as well as rural education were the dominant positive influences on rural residents' well-being in Jiangsu. Additionally, urbanization could to some extent enhance rural population's income, expenditure and medical insurance levels. These results generally conform to some published work concerning regions within or outside China (Bradshaw and Fraser, 1989; Liu et al., 2010; Robinson et al., 2012). By comparing human well-being and ecosystem services, it can be seen that they had similar positive socioeconomic influences. Given the strong causal linkages between these two sectors (e.g. provisioning services endow humans access to resources to earn income and gain livelihood, MA, 2005), it is not hard to understand that the positive socioeconomic influential factors of ecosystem services can also benefit human well-being. For example, industrialization and the enhancement of agricultural technology and financial support can improve agricultural productivity, which will increase rural residents' income. The revised results at the same time reveal the negative influence of total food production increase and arable land expansion on human well-being, which can be explained by their representations of extensive agriculture and comparatively low urbanization and industrialization levels.

4.2. Credibility of the quantitative relationships between the DPSIR sectors

The findings proved quantitative linkages between the DPSIR sectors (Impact is divided into ecosystem services and human well-being; see Table 5). Based on Table 5, a quantitative causal network of DPSIR interrelations can be derived (Figure 6). Table 5 and Figure 6 show that the causal relationships between the DPSIR sectors are more like a network than a sequent chain, which means one sector can "jump over" the closer sectors and link more strongly with more further ones (e.g. in this study Driver is more strongly correlated with human well-being than with Pressure or Response and Pressure has higher correlation with human well-being than ecosystem services). This result is not in agreement with the typical causal chain arguments of the DPSIR approach (Burkhard and Müller, 2008; European Environment Agency, 2007), but can in some degree be evidenced by the presumption of the developed causal network (Niemeijer and de Groot, 2008b). Maybe such result can exactly answer the criticism of the traditional DPSIR causal chain which is assumed to be too simple to really capture the interactions of the complex human-environmental systems (Svarstad et al., 2008). Meanwhile, it responds to the calling for a more sophisticated causal network to address this complicated issue.

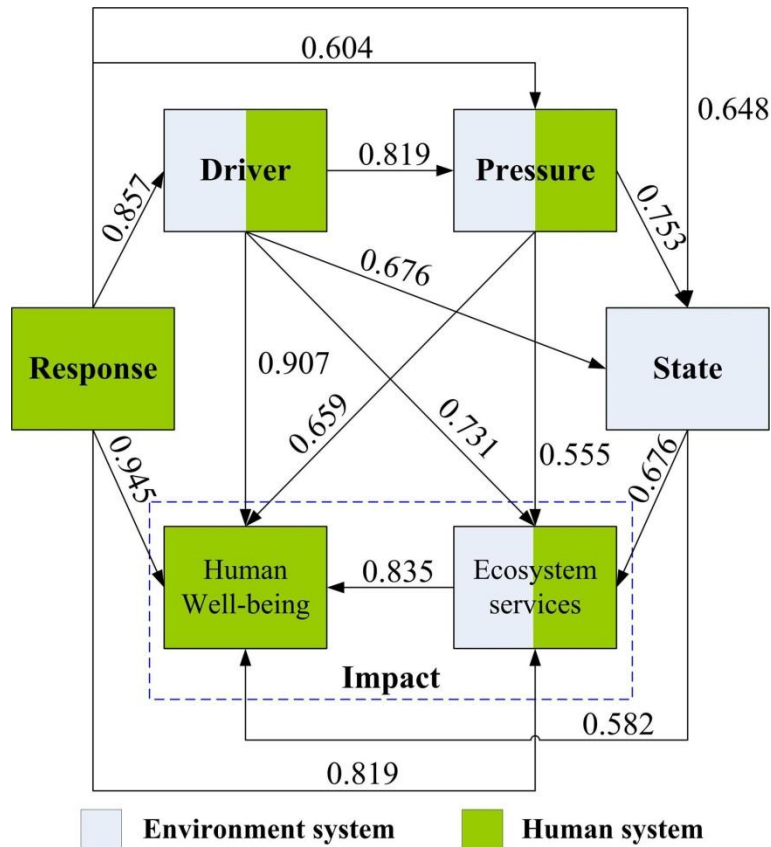


Figure 6. Quantitative linkages between the DPSIR sectors. Coefficients between every two sectors in the figure are Spearman correlation coefficients.

4.3. Limitations and implications

4.3.1. Limitations of the study

In spite of the interesting findings of our research, the noticeable limitations should not be neglected. These shortcomings primarily involve the inherent simplifications of reality, the incompleteness and bias of the indicator set, data unavailability, data uncertainty and uncertainties from aggregations. Simplification is an intrinsic feature of models. Although we made our effort to consider a high number of influential factors of the environmental and social systems of the study area, we had to compromise to some simplifications. One major limitation relates to the disregard of environmental influential factors. In the study we considered temperature and precipitation, which mostly affect biodiversity, ecosystem services and human well-being in the research area. Though the other factors did not alter considerably between different prefecture-level cities, they might still have some influence on the results. Another obvious simplification problem refers to the temporal hysteresis of the DPSIR sectors. As suggested by the DPSIR chain, time hysteresis exists between DPSIR sectors, i.e., changes of a certain sector (e.g., State or Impact) are caused by other sectors (e.g. Driver or Pressure) that work over a certain amount of time (Nelson, 2005). In this study, we did not consider this temporal hysteresis effect and solely limited the analysis to one year due to the availability of data for the indicators, which might result in inaccuracy of the result.

Incompleteness and bias are inevitable for the proposed indicator set in this study because of limited data availability and the uncertainties caused by the indicator assignment. One example relates to the tourism service indicator “tourism income”. As explained in the previous sections of discussions, this tourism service mainly involves urban tourism, which is primarily provided by highly human-shaped systems instead of ecosystems. Concerning the ecosystem service indicators, we only assigned “carbon storage density” as regulating service indicator and “tourism service” to represent cultural services. For human well-being, only four indicators were used. This limited number of indicators is far less than enough to address ecosystem services and human well-being realities in the research area. Besides, the assignment of the indicators to different DPSIR sectors could be inappropriate because of our knowledge limitations of the indicators themselves. It is probable that the quantification results and findings of the socioeconomic influences would be different if we use other indicators or would have a more complete indicator set. As emphasized in the section of “Indicator set and data sources”, the purpose of this study was to examine the applicability of the DPSIR model by quantitatively revealing socioeconomic influences on biodiversity, ecosystem services and human well-being taking Jiangsu as an example. More case studies applying the suggested methodology with more complete indicator sets and data would be helpful in order to further develop the method.

In addition to the limitations due to data unavailability described in the former paragraphs, extra threats to the credibility of the findings could occur. For example, the 13 prefecture-level cities of Jiangsu were taken as sample areas for the comprehensive statistical analyses in this study. This relatively small sample size might undermine the credibility of the statistic results and obstruct the extrapolation of the findings in Jiangsu to other areas. Furthermore, the uncertainties of the data sources, such as most of the data for the indicator values are secondary data and uncertainties with these data are unknown, could further reduce the accuracy of the results (Hou et al., 2013).

Last but not least, our analysis process was filled with aggregations, which can lead to a proliferation of uncertainties from data, methods or subjective judgments with each analytical step. This would further result in considerable uncertainties with the findings based on aggregation (Hou et al., 2013). The quantitative linkages between DPSIR sectors, generated by the integration of all the indicators and data in this study as well as several statistical methods (e.g. factor analysis, spearman correlations and standardizations of absolute values), are surely accompanied by notable uncertainties. This considerably hinders the results’ credibility and applications potential.

4.3.2. Implications of the study

In spite of the noticeable limitations, our work derived fruitful results to reveal the socioeconomic influences on biodiversity, ecosystem services and human well-being of the research area. These findings are virtually consistent with several of other peoples’ work. Furthermore, our attempt to quantify the influence from socioeconomics and examining the quantitative applicability of the DPSIR model can be regarded as promising. These new attempts can doubtlessly help to fill the lack of quantifications of socioeconomic influences on biodiversity, ecosystem services and human well-being and additionally, throw light on the quantitative usage of the DPSIR model for addressing human-environmental systems’ interaction issues. Concerning the finding of the quantitative linkages between the DPSIR sectors, despite the obvious uncertainties with it, we assume that it can at least quantitatively prove the network-shape causalities between the sectors. However, it should be noted that the findings of our case study are heavily dependent on the

specific environmental and socioeconomic contexts of the study area. Therefore, extrapolations of the results to other study areas will undoubtedly lead to uncertainties.

5. Conclusions

The aims of this paper were to quantitatively investigate the socioeconomic influences on biodiversity, ecosystem services and human well-being in Jiangsu, China and examine the quantitative applicability of the DPSIR model to address human-environmental interaction issues. Besides, it intends to test the causal relationships between the different DPSIR sectors from a quantitative perspective. With these goals we asked two research questions at the beginning of this paper, which can be now answered based on the results of this study:

1) Can the DPSIR model be used to quantitatively analyze the socioeconomic influences on biodiversity, ecosystem services and human well-being in human-environmental systems?

In our empirical study about the human-environmental system of Jiangsu, the DPSIR model was proved to be applicable to quantitatively capture the socioeconomic influences on biodiversity, ecosystem services and human well-being in combination with statistical methods.

2) What are the quantitative linkages between the different sectors of the DPSIR model in the Jiangsu human-environmental system?

Concerning the study area, factor analysis and spearman correlations revealed strong correlations between the different sectors of the DPSIR model. These correlations proved that the linkages between the sectors are forming a causal network more than a causal chain. This finding in some degree suggests the applicability of the developed DPSIR model in the form of a network to better address the complex human-environmental interactions than the conventional chain-form DPSIR causal model.

The encouraging findings of our work show that the attempt is promising to quantitatively explore the socioeconomic influences on biodiversity, ecosystem services and human well-being in human-environmental systems. This kind of work is informative for regional environmental management decision making. Meanwhile, the quantitative applicability of the DPSIR model proved in this paper suggests that further work to use it as a quantitative tool for empirical studies would be desirable.

Acknowledgments

The work of this paper is supported by the Key Projects of National Philosophy and Social Science Foundation of China (13&ZD160, 11&ZD046), The Key Projects of National Natural Science Foundation of China (70833001), China Agricultural Research System (CARS-14-10B), Doctoral Fund of Ministry of Education of China (20120097110034) as well as the Fundamental Research Funds for the Central Universities of China (XNC2012001). We are grateful for the time and efforts from the editors and the anonymous reviewers on improving our manuscript.

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Supplementary materials

Online appendix 1

Table A1. KMO and Bartlett's tests for the variables of the DPSIR sectors

Driver	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.633
	Bartlett's Test of Sphericity	Approx. Chi-Square	103.460
		df	21
		Sig.	0.000
Pressure	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.720
	Bartlett's Test of Sphericity	Approx. Chi-Square	47.288
		df	21
		Sig.	0.001
State	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.769
	Bartlett's Test of Sphericity	Approx. Chi-Square	63.466
		df	10
		Sig.	0.000
Ecosystem services	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.690
	Bartlett's Test of Sphericity	Approx. Chi-Square	29.767
		df	10
		Sig.	0.001
Human well-being	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.804
	Bartlett's Test of Sphericity	Approx. Chi-Square	69.986
		df	6
		Sig.	0.000
Response	Kaiser-Meyer-Olkin Measure of Sampling Adequacy.		0.719
	Bartlett's Test of Sphericity	Approx. Chi-Square	22.542
		df	6
		Sig.	0.001

The test result tables for each of the DPSIR sector produced in SPSS 17.0 are grouped into this table.

Table A2. Total variance explained by the components of the DPSIR sectors

	Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
		Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
Driver	1	5.119	73.123	73.123	5.119	73.123	73.123
	2	0.887	12.673	85.795			
	3	0.607	8.676	94.472			
	4	0.231	3.300	97.772			
	5	0.131	1.866	99.637			
	6	0.021	0.294	99.932			
	7	0.005	0.068	100.000			
Pressure	1	4.078	58.261	58.261	4.078	58.261	58.261
	2	1.374	19.626	77.887	1.374	19.626	77.887
	3	0.593	8.472	86.358			
	4	0.450	6.431	92.789			
	5	0.264	3.775	96.564			
	6	0.170	2.431	98.995			
	7	0.070	1.005	100.000			
State	1	3.659	73.171	73.171	3.659	73.171	73.171
	2	0.915	18.305	91.476			
	3	0.347	6.935	98.411			
	4	0.062	1.241	99.652			
	5	0.017	0.348	100.000			

	Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
		Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
Ecosystem services	1	2.964	59.287	59.287	2.964	59.287	59.287
	2	1.307	26.144	85.431	1.307	26.144	85.431
	3	0.371	7.430	92.860			
	4	0.218	4.368	97.228			
	5	0.139	2.772	100.000			
	Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
		Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
Human well-being	1	3.549	88.727	88.727	3.549	88.727	88.727
	2	0.397	9.925	98.652			
	3	0.039	0.981	99.633			
	4	0.015	0.367	100.000			
	Component	Initial Eigenvalues			Extraction Sums of Squared Loadings		
		Total	% of Variance	Cumulative %	Total	% of Variance	Cumulative %
Response	1	2.578	64.451	64.451	2.578	64.451	64.451
	2	1.013	25.335	89.786	1.013	25.335	89.786
	3	0.260	6.490	96.276			
	4	0.149	3.724	100.000			

Extraction method: principal components analysis. The result tables for each of the DPSIR sectors derived in SPSS 17.0 are grouped in this table

Table A3. Component score coefficient matrices of the DPSIR sectors

	Variables	Component 1	Component 2
Driver	Total food crops output	-0.176	
	Population density	0.138	
	Proportion of urban built-up area in total	0.141	

	Proportion of urban population in total	0.186	
	Industrial sector proportion in GDP	0.158	
	GDP per person	0.176	
	The ratio of gross output value of agriculture to GDP	-0.187	
Pressure	Arable land per capita	-0.217	0.068
	Proportion of arable land area in total land area	-0.197	-0.015
	Total power of agricultural machinery per unit sown area	0.190	0.127
	Level of chemical fertilizer use	-0.055	0.674
	Level of pesticides use	0.163	0.411
	Irrigation rate	0.231	0.060
	Acid rain rate	0.200	-0.283
State	Richness of wild higher plants	0.269	
	Richness of wild higher animals	0.128	
	Diversity of ecosystem types	0.264	
	Number of endemic species	0.260	
	Proportion of woodland, garden and grassland area in total	0.218	
Ecosystem services	Farming output value per chemical fertilizer use	0.309	0.113
	Food crops output per unit sown area	0.289	-0.063
	Agricultural labor productivity	0.305	-0.220
	Organic carbon density	-0.076	0.716
	Ratio of tourism income to GDP	0.244	0.432
Human well-being	Rural residents' per capita net income	0.277	
	Rural residents' per capita expenditure	0.278	
	Housing area per capita in rural area	0.270	
	New rural cooperative medical insurance coverage	0.234	
Response	Government agricultural expenditure per unit sown area of crops	0.350	-0.164
	Agricultural loans per unit sown area of crops	0.368	-0.038
	Number of agricultural science and technology personnel	0.046	0.976

	Years of rural education	0.357	0.075
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Extraction method: principal component analysis. The result tables for each of the DPSIR sector generated in SPSS 17.0 are grouped in this table. The factor score coefficients were estimated by regression method.

Table A4. Standardized scores of the principal components of the DPSIR sectors' indicators for the 13 prefecture-level cities of Jiangsu

Prefecture -level city names	DPC score	PPC1 score	PPC2 score	SPC score	ESPC1 score	ESPC2 score	HWPC score	RPC1 score	RPC2 score
Changzhou	0.8015	1.3172	-0.1479	0.9293	1.0191	0.8835	1.3149	0.8802	-0.1815
Huaian	-1.1330	-1.0329	0.0526	-0.7585	-0.5189	-0.0036	-0.7867	-1.0430	0.2638
Zhenjiang	0.6204	-0.0811	0.1422	1.2279	-0.1454	1.7134	0.4760	0.7046	0.7218
Yangzhou	-0.0718	-0.1262	-0.5352	-0.7617	0.4554	-0.4138	-0.4078	-0.2565	0.0374
Yancheng	-1.3545	-1.0938	-0.1083	-0.6230	0.0856	-1.1369	-0.8055	-0.7777	1.1418
Xuzhou	-0.4821	-0.6581	2.0841	-0.4731	-1.3459	1.1446	-0.8594	-0.8017	-1.2134
Wuxi	1.4307	2.1178	1.3462	1.2692	0.4741	0.2229	1.5546	1.9599	-0.7984
Taizhou	-0.0765	-0.3880	-0.5014	-1.1703	-0.1190	-0.9466	-0.2335	-0.3976	-0.4367
Suzhou	1.1976	1.3072	-0.9112	0.9068	1.5306	-1.4417	1.5883	1.5484	-0.4461
Suqian	-1.2848	-0.7877	0.3486	-0.9680	-1.3378	-0.1047	-1.2672	-1.1029	-1.4016
Nantong	-0.0620	-0.3922	-1.6600	-0.9308	-0.4594	-0.7288	-0.1026	-0.2118	2.0287
Nanjing	1.3002	0.3622	-0.9314	1.4949	1.5945	1.3042	0.5230	0.2704	0.9939
Lianyungang	-0.8858	-0.5444	0.8216	-0.1429	-1.2328	-0.4925	-0.9942	-0.7723	-0.7097

The standardized scores of the principal components are generated automatically through the factor analysis process with SPSS. The standardization method for the variable values is z-scores normalization. Full names of the principal components and their interpretations are as follows:

Driver principal component (DPC): positive index of urbanization and industrialization level and negative index of agricultural production and economy sizes;

Pressure principal component 1 (PPC1): negative index of relative arable land amount and positive index of agricultural technology level and pressure from industry;

Pressure principal component 2 (PPC2): indicates chemical fertilizer uses level;

State principal component (SPC): Positive index of plant species, ecosystem and endemic species diversity as well as non-cropland vegetation cover;

Ecosystem service principal component 1 (ESPC1): positive index of agricultural productivity and regional tourism service;

Ecosystem service principal component 2 (ESPC2): positive index of carbon storage service;

Human well-being principal component (HWPC): positive index of rural inhabitants' income, expenditure, housing and medical insurance level;

Response principal component 1 (RPC1): positive index of agriculture financial support level and rural education level;

Response principal component 2 (RPC2): positive index of agricultural science and technology support level.

Table A5. Standardized scores of the sectors of DPSIR for the 13 prefecture-level cities of Jiangsu

Prefecture-level city names	Driver score	Pressure score	State score	Ecosystem service score	Human well-being score	Response score
Changzhou	0.8015	0.9480	0.9293	0.9776	1.3149	0.5806
Huaian	-1.1330	-0.7593	-0.7585	-0.3612	-0.7867	-0.6743
Zhenjiang	0.6204	-0.0248	1.2279	0.4234	0.4760	0.7095
Yangzhou	-0.0718	-0.2292	-0.7617	0.1894	-0.4078	-0.1736
Yancheng	-1.3545	-0.8455	-0.6230	-0.2885	-0.8055	-0.2360
Xuzhou	-0.4821	0.0329	-0.4731	-0.5837	-0.8594	-0.9179
Wuxi	1.4307	1.9233	1.2692	0.3972	1.5546	1.1816
Taizhou	-0.0765	-0.4166	-1.1703	-0.3723	-0.2335	-0.4086
Suzhou	1.1976	0.7482	0.9068	0.6210	1.5883	0.9856
Suqian	-1.2848	-0.5014	-0.9680	-0.9604	-1.2672	-1.1872
Nantong	-0.0620	-0.7117	-0.9308	-0.5418	-0.1026	0.4204
Nanjing	1.3002	0.0363	1.4949	1.5057	0.5230	0.4746
Lianyungang	-0.8858	-0.2002	-0.1429	-1.0063	-0.9942	-0.7546

The standardized scores are derived from the standardized scores of the principal components of the DPSIR sector indicators and equations (1) to (6) in the text of the article (see table A4 in this appendix and equations (1) to (6) in the article text). The standardization method for the variable values is z-scores normalization.

Online appendix 2

Table B1. Correlations between the indicators of driver and pressure

Driver \ Pressure	Total food crops output	Population density	Proportion of urban built-up area in total	Proportion of urban population in total	Industrial sector proportion in GDP	GDP per person	The ratio of gross output value of agriculture to GDP
Arable land per capita	0.687**	-0.852**	-0.758**	-0.863**	-0.736**	-0.863**	0.896**
Proportion of arable land area in total land area	0.659*	0.000	-0.725**	-0.698**	-0.511	-0.725**	0.692**
Total Power of agricultural machinery per unit sown area	-0.808**	0.220	0.802**	0.560*	0.467	0.571*	-0.637*
Level of chemical fertilizer use	0.231	-0.412	-0.143	-0.538	-0.390	-0.544	0.500
Level of pesticides use	-0.544	0.291	0.440	0.258	0.379	0.220	-0.275
Irrigation rate	-0.923**	0.308	0.698**	0.698**	0.692**	0.714**	-0.780**
Acid rain rate	-0.638*	0.710**	0.490	0.747**	0.669*	0.749**	-0.788**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B2. Correlations between the indicators of driver and state

Driver \ State	Total food crops output	Population density	Proportion of urban built-up area in total	Proportion of urban population in total	Industrial sector proportion in GDP	GDP per person	The ratio of gross output value of agriculture to GDP
Richness of wild higher plants	-0.819**	0.286	0.956**	0.714**	0.489	0.687**	-0.753**
Richness of wild higher animals	-0.192	-0.016	0.412	0.385	0.055	0.313	-0.335
Diversity of ecosystem types	-0.632*	0.022	0.747**	0.582*	0.379	0.593*	-0.604*
Number of endemic species	-0.863**	0.451	0.775**	0.819**	0.643*	0.808**	-0.830**
Proportion of woodland, garden and grassland area in total	-0.527	0.209	0.698**	0.401	0.121	0.330	-0.374

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B3. Correlations between the indicators of driver and impact I (ecosystem services)

Driver \ Ecosystem services	Total food crops output	Population density	Proportion of urban built-up area in total	Proportion of urban population in total	Industrial sector proportion in GDP	GDP per person	The ratio of gross output value of agriculture to GDP
Farming output value per chemical fertilizer use	-0.588*	0.495	0.571*	0.868**	0.615*	0.863**	-0.797**
Food crops output per unit sown area	-0.511	0.121	0.484	0.632*	0.516	0.654*	-0.621*
Agricultural labor productivity	-0.319	0.203	0.374	0.714**	0.500	0.731**	-0.643*
Organic carbon density	-0.154	-0.066	0.294	-0.030	-0.215	-0.121	0.105
Ratio of tourism income to GDP	-0.665*	0.401	0.747**	0.824**	0.615*	0.808**	-0.786**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B4. Correlations between the indicators of driver and impact **II** (human well-being)

Driver Human Well-being	Total food crops output	Population density	Proportion of urban built-up area in total	Proportion of urban population in total	Industrial sector proportion in GDP	GDP per person	The ratio of gross output value of agriculture to GDP
Rural residents' per capita net income	-0.720**	0.588*	0.709**	0.945**	0.874**	0.984**	-0.951**
Rural residents' per capita expenditure	-0.710**	0.539	0.680*	0.927**	0.861**	0.971**	-0.927**
Housing area per capita in rural area	-0.627*	0.655*	0.605*	0.886**	0.864**	0.935**	-0.902**
New rural cooperative medical insurance coverage	-0.776**	0.396	0.839**	0.674*	0.641*	0.691**	-0.751**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B5. Correlations between the indicators of driver and response

Driver \ Response	Total food crops output	Population density	Proportion of urban built-up area in total	Proportion of urban population in total	Industrial sector proportion in GDP	GDP per person	The ratio of gross output value of agriculture to GDP
Government agricultural expenditure per unit sown area of crops	-0.890**	0.538	0.786**	0.885**	0.758**	0.896**	-0.940**
Agricultural loans per unit sown area of crops	-0.824**	0.670*	0.637*	0.813**	0.890**	0.846**	-0.879**
Number of agricultural science and technology personnel	0.049	0.115	-0.022	0.335	0.121	0.313	-0.225
Years of rural education	-0.830**	0.577*	0.731**	0.918**	0.874**	0.923**	-0.912**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B6. Correlations between the indicators of pressure and state

Pressure \ State	Arable land per capita	Proportion of arable land area in total land area	Total Power of agricultural machinery per unit sown area	Level of chemical fertilizer use	Level of pesticides use	Irrigation rate	Acid rain rate
Richness of wild higher plants	-0.626*	-0.802**	0.863**	0.027	0.456	0.758**	0.373
Richness of wild higher animals	-0.126	-0.434	0.066	-0.159	-0.308	0.203	0.075
Diversity of ecosystem types	-0.445	-0.912**	0.780**	-0.066	0.082	0.643*	0.262
Number of endemic species	-0.681*	-0.698**	0.698**	-0.401	0.352	0.709**	0.613*
Proportion of woodland, garden and grassland area in total	-0.346	-0.478	0.758**	0.055	0.412	0.341	0.095

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B7. Correlations between the indicators of pressure and impact I (ecosystem services)

Pressure Ecosystem services	Arable land per capita	Proportion of arable land area in total land area	Total Power of agricultural machinery per unit sown area	Level of chemical fertilizer use	Level of pesticides use	Irrigation rate	Acid rain rate
Farming output value per chemical fertilizer use	-0.709**	-0.687**	0.462	-0.747**	-0.049	0.511	0.680*
Food crops output per unit sown area	-0.473	-0.676*	0.407	-0.511	0.154	0.511	0.471
Agricultural labor productivity	-0.462	-0.665*	0.115	-0.643*	-0.258	0.363	0.535
Organic carbon density	0.113	-0.069	0.404	0.349	0.440	-0.047	-0.385
Ratio of tourism income to GDP	-0.560*	-0.632*	0.478	-0.368	0.264	0.505	0.407

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B8. Correlations between the indicators of pressure and impact **II** (human well-being)

Pressure Human Well-being	Arable land per capita	Proportion of arable land area in total land area	Total Power of agricultural machinery per unit sown area	Level of chemical fertilizer use	Level of pesticides use	Irrigation rate	Acid rain rate
Rural residents' per capita net income	-0.846**	-0.714**	0.555*	-0.593*	0.170	0.703**	0.783**
Rural residents' per capita expenditure	-0.814**	-0.715**	0.572*	-0.627*	0.179	0.696**	0.745**
Housing area per capita in rural area	-0.861**	-0.583*	0.470	-0.657*	0.113	0.627*	0.829**
New rural cooperative medical insurance coverage	-0.666*	-0.660*	0.806**	-0.160	0.473	0.666*	0.502

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B9. Correlations between the indicators of pressure and response

Pressure \ Response	Arable land per capita	Proportion of arable land area in total land area	Total Power of agricultural machinery per unit sown area	Level of chemical fertilizer use	Level of pesticides use	Irrigation rate	Acid rain rate
Government agricultural expenditure per unit sown area of crops	-0.841**	-0.698**	0.709**	-0.505	0.269	0.868**	0.780**
Agricultural loans per unit sown area of crops	-0.852**	-0.489	0.604*	-0.440	0.352	0.824**	0.813**
Number of agricultural science and technology personnel	-0.033	-0.165	-0.308	-0.709**	-0.593*	-0.192	0.198
Years of rural education	-0.786**	-0.648*	0.626*	-0.467	0.429	0.769**	0.752**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B10. Correlations between the indicators of state and impact I (ecosystem services)

State Ecosystem services	Richness of wild higher plants	Richness of wild higher animals	Diversity of ecosystem types	Number of endemic species	Proportion of woodland, garden and grassland area in total
Farming output value per chemical fertilizer use	0.473	0.253	0.560*	0.731**	0.368
Food crops output per unit sown area	0.434	0.137	0.440	0.599*	0.104
Agricultural labor productivity	0.297	0.495	0.434	0.440	-0.066
Organic carbon density	0.327	-0.030	0.193	0.110	0.801**
Ratio of tourism income to GDP	0.681*	0.319	0.467	0.819**	0.467

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B11. Correlations between the indicators of state and impact **II** (human well-being)

State Human Well-being	Richness of wild higher plants	Richness of wild higher animals	Diversity of ecosystem types	Number of endemic species	Proportion of woodland, garden and grassland area in total
Rural residents' per capita net income	0.615*	0.269	0.593*	0.775**	0.313
Rural residents' per capita expenditure	0.586*	0.209	0.564*	0.773**	0.319
Housing area per capita in rural area	0.479	0.179	0.481	0.685**	0.234
New rural cooperative medical insurance coverage	0.814**	0.118	0.721**	0.801**	0.702**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B12. Correlations between the indicators of state and response

State Response	Richness of wild higher plants	Richness of wild higher animals	Diversity of ecosystem types	Number of endemic species	Proportion of woodland, garden and grassland area in total
Government agricultural expenditure per unit sown area of crops	0.747**	0.225	0.621*	0.879**	0.330
Agricultural loans per unit sown area of crops	0.582*	0.121	0.478	0.714**	0.258
Number of agricultural science and technology personnel	-0.137	0.385	0.060	0.231	-0.038
Years of rural education	0.648*	0.203	0.527	0.791**	0.445

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B13. Correlations between the indicators of ecosystem services and human well-being

Ecosystem services Human Well-being	Farming output per value chemical fertilizer use	Food crops output per unit sown area	Agricultural labor productivity	Organic carbon density	Ratio of tourism income to GDP
Rural residents' per capita net income	0.901**	0.659*	0.747**	-0.165	0.725**
Rural residents' per capita expenditure	0.916**	0.702**	0.751**	-0.124	0.737**
Housing area per capita in rural area	0.886**	0.613*	0.713**	-0.252	0.600*
New rural cooperative medical insurance coverage	0.550	0.484	0.237	0.237	0.564*

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B14. Correlations between the indicators of impact I (ecosystem services) and response

Ecosystem services Response	Farming output per value chemical fertilizer use	Food crops output per unit sown area	Agricultural labor productivity	Organic carbon density	Ratio of tourism income to GDP
Government agricultural expenditure per unit sown area of crops	0.758**	0.681*	0.577*	-0.168	0.687**
Agricultural loans per unit sown area of crops	0.665*	0.429	0.412	-0.176	0.538
Number of agricultural science and technology personnel	0.555*	0.192	0.582*	-0.195	0.379
Years of rural education	0.808**	0.593*	0.582*	0.058	0.753**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B15. Correlations between the indicators of impact **II** (human well-being) and response

Human Well-being Response	Rural residents' per capita net income	Rural residents' per capita expenditure	Housing area per capita in rural area	New rural cooperative medical insurance coverage
Government agricultural expenditure per unit sown area of crops	0.874**	0.864**	0.820**	0.735**
Agricultural loans per unit sown area of crops	0.868**	0.834**	0.872**	0.707**
Number of agricultural science and technology personnel	0.357	0.344	0.369	0.011
Years of rural education	0.934**	0.933**	0.894**	0.740**

Spearman correlations

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

Table B16. Partial correlations of the indicators of biodiversity with the indicators of driver, pressure and response

Control Variables			Richness of wild higher plants	Richness of wild higher animals	Diversity of ecosystem types	Number of endemic species
Temperature & precipitation	Total food crops output	Correlation	-0.715	-0.102	-0.564	-0.815
		Significance (2-tailed)	0.013	0.765	0.071	0.002
	Population density	Correlation	-0.275	-0.317	-0.449	-0.247
		Significance (2-tailed)	0.414	0.342	0.165	0.463
	Proportion of urban built-up area in total	Correlation	0.690	0.386	0.559	0.663
		Significance (2-tailed)	0.019	0.241	0.074	0.026
	Proportion of urban population in total	Correlation	0.799	0.526	0.701	0.796
		Significance (2-tailed)	0.003	0.097	0.016	0.003
	Industrial sector proportion in GDP	Correlation	-0.083	-0.069	-0.031	0.062
		Significance (2-tailed)	0.809	0.841	0.928	0.856
	GDP per person	Correlation	0.667	0.298	0.775	0.570
		Significance (2-tailed)	0.025	0.373	0.005	0.067
	The ratio of gross output value of agriculture to GDP	Correlation	-0.238	-0.296	-0.059	-0.343
		Significance (2-tailed)	0.480	0.377	0.863	0.301
	Arable land per capita	Correlation	-0.343	-0.096	-0.205	-0.274
		Significance (2-tailed)	0.302	0.779	0.546	0.414
	Proportion of arable land	Correlation	-0.728	-0.401	-0.814	-0.621

	area in total land area	Significance (2-tailed)	0.011	0.221	0.002	0.042
	Total power of agricultural machinery per unit sown area	Correlation	0.479	-0.197	0.476	0.519
		Significance (2-tailed)	0.136	0.561	0.139	0.102
	Level of chemical fertilizer use	Correlation	0.081	0.201	0.080	-0.097
		Significance (2-tailed)	0.813	0.553	0.815	0.776
	Level of pesticides use	Correlation	0.197	-0.367	0.099	0.209
		Significance (2-tailed)	0.561	0.267	0.771	0.537
	Irrigation rate	Correlation	0.415	0.039	0.545	0.278
		Significance (2-tailed)	0.204	0.909	0.083	0.408
	Acid rain rate	Correlation	-0.376	-0.349	-0.354	-0.500
		Significance (2-tailed)	0.255	0.292	0.285	0.117
	Government agricultural expenditure per unit sown area of crops	Correlation	0.475	0.239	0.629	0.333
		Significance (2-tailed)	0.139	0.480	0.038	0.316
	Agricultural loans per unit sown area of crops	Correlation	0.421	-0.093	0.418	0.360
		Significance (2-tailed)	0.197	0.786	0.201	0.276
	Number of agricultural science and technology personnel	Correlation	0.147	0.500	0.127	0.141
		Significance (2-tailed)	0.666	0.117	0.710	0.679
	Years of rural education	Correlation	0.593	0.133	0.556	0.697
		Significance (2-tailed)	0.055	0.696	0.076	0.017

Table B17. Partial correlations of the indicators of ecosystem services with the indicators of driver, pressure and response

Control Variables			Farming output value per chemical fertilizer use	Food crops output per unit sown area	Agricultural labor productivity	Organic carbon density	Ratio of tourism income to GDP
Temperature & precipitation	Total food crops output	Correlation	-0.451	-0.314	0.107	-0.528	-0.615
		Significance (2-tailed)	0.164	0.348	0.754	0.095	0.044
	Population density	Correlation	-0.108	-0.349	-0.376	0.302	-0.157
		Significance (2-tailed)	0.751	0.292	0.254	0.367	0.646
	Proportion of urban built-up area in total	Correlation	0.702	0.445	0.477	0.486	0.746
		Significance (2-tailed)	0.016	0.170	0.138	0.129	0.008
	Proportion of urban population in total	Correlation	-0.116	0.315	0.133	-0.108	0.039
		Significance (2-tailed)	0.734	0.345	0.697	0.752	0.910
	Industrial sector proportion in GDP	Correlation	0.456	0.410	0.548	-0.139	0.173
		Significance (2-tailed)	0.159	0.211	0.081	0.683	0.611
	GDP per person	Correlation	-0.180	-0.412	-0.141	-0.349	-0.532
		Significance (2-tailed)	0.597	0.208	0.679	0.293	0.092
	The ratio of gross output value of agriculture to GDP	Correlation	-0.405	-0.138	-0.229	-0.319	-0.218
		Significance (2-tailed)	0.217	0.685	0.499	0.339	0.519
	Arable land per capita	Correlation	-0.487	-0.512	-0.615	0.053	-0.357
		Significance (2-tailed)	0.129	0.107	0.044	0.876	0.281

Proportion of arable land area in total land area	Correlation	0.472	0.445	0.023	0.477	0.031
	Significance (2-tailed)	0.143	0.171	0.946	0.138	0.927
Total power of agricultural machinery per unit sown area	Correlation	-0.468	-0.429	-0.211	0.286	0.019
	Significance (2-tailed)	0.146	0.187	0.534	0.394	0.957
Level of chemical fertilizer use	Correlation	-0.248	-0.085	-0.293	0.318	-0.037
	Significance (2-tailed)	0.462	0.803	0.382	0.341	0.913
Level of pesticides use	Correlation	0.039	0.275	0.280	-0.266	-0.228
	Significance (2-tailed)	0.910	0.414	0.405	0.430	0.500
Irrigation rate	Correlation	-0.118	0.183	0.104	-0.415	-0.686
	Significance (2-tailed)	0.730	0.590	0.762	0.205	0.020
Acid rain rate	Correlation	0.386	0.405	0.579	-0.384	-0.032
	Significance (2-tailed)	0.241	0.217	0.062	0.244	0.925
Government agricultural expenditure per unit sown area of crops	Correlation	0.386	0.405	0.579	-0.384	-0.032
	Significance (2-tailed)	0.241	0.217	0.062	0.244	0.925
Agricultural loans per unit sown area of crops	Correlation	-0.171	-0.091	-0.183	-0.003	-0.162
	Significance (2-tailed)	0.614	0.791	0.589	0.993	0.635
Number of agricultural science and technology personnel	Correlation	0.210	-0.174	-0.023	0.141	0.205
	Significance (2-tailed)	0.535	0.609	0.947	0.678	0.545
Years of rural	Correlation	0.325	0.406	0.154	0.544	0.432

	education	Significance (2-tailed)	0.330	0.216	0.651	0.084	0.185
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Table B18. Partial correlations of the indicators of human well-being with the indicators of driver, pressure and response

Control Variables			Rural residents' per capita net income	Rural residents' per capita expenditure	Housing area per capita in rural area	New rural cooperative medical insurance coverage
Temperature & precipitation	Total food crops output	Correlation	-0.089	-0.383	0.220	-0.682
		Significance (2-tailed)	0.794	0.246	0.515	0.021
	Population density	Correlation	-0.424	-0.364	-0.254	-0.228
		Significance (2-tailed)	0.194	0.271	0.451	0.501
	Proportion of urban built-up area in total	Correlation	0.152	0.300	-0.275	0.358
		Significance (2-tailed)	0.655	0.370	0.413	0.279
	Proportion of urban population in total	Correlation	0.475	0.593	0.049	0.475
		Significance (2-tailed)	0.140	0.055	0.887	0.140
	Industrial sector proportion in GDP	Correlation	0.463	0.395	0.623	0.163
		Significance (2-tailed)	0.152	0.230	0.040	0.632
	GDP per person	Correlation	0.873	0.695	0.492	0.523
		Significance (2-tailed)	0.000	0.018	0.125	0.099
	The ratio of gross output value of agriculture to GDP	Correlation	-0.061	-0.256	-0.028	-0.232
		Significance (2-tailed)	0.859	0.448	0.934	0.492
	Arable land per capita	Correlation	-0.171	-0.199	0.071	-0.153
		Significance (2-tailed)	0.614	0.557	0.835	0.654

Proportion of arable land area in total land area	Correlation	-0.754	-0.673	-0.266	-0.525
	Significance (2-tailed)	0.007	0.023	0.429	0.097
Total power of agricultural machinery per unit sown area	Correlation	0.501	0.708	0.468	0.637
	Significance (2-tailed)	0.116	0.015	0.147	0.035
Level of chemical fertilizer use	Correlation	-0.042	-0.231	-0.244	0.073
	Significance (2-tailed)	0.903	0.495	0.470	0.832
Level of pesticides use	Correlation	0.271	0.204	0.097	0.437
	Significance (2-tailed)	0.421	0.548	0.778	0.179
Irrigation rate	Correlation	0.711	0.536	0.456	0.456
	Significance (2-tailed)	0.014	0.089	0.159	0.158
Acid rain rate	Correlation	-0.072	-0.063	0.119	-0.244
	Significance (2-tailed)	0.834	0.855	0.729	0.469
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Government agricultural expenditure per unit sown area of crops	Correlation	0.765	0.548	0.466	0.369
	Significance (2-tailed)	0.006	0.081	0.149	0.263
Agricultural loans per unit sown area of crops	Correlation	0.471	0.310	0.261	0.530
	Significance (2-tailed)	0.144	0.353	0.438	0.094
Number of agricultural science and technology personnel	Correlation	-0.228	-0.141	-0.129	-0.141
	Significance (2-tailed)	0.501	0.679	0.706	0.679
Years of rural education	Correlation	0.633	0.770	0.447	0.600
	Significance (2-tailed)	0.037	0.006	0.168	0.051

Chapter 3. Urban-rural gradients of ecosystem services and the linkages with socioeconomics

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Submitted to Environmental Monitoring and Assessment

Abstract

The principle of urban-rural gradients is proven to be applicable to reveal the spatial variations of ecosystem services and socioeconomic dimensions. The interrelations between ecosystem services and socioeconomics have scarcely been considered in the context of urban-rural areas.

To narrow the respective research gaps, we investigated the spatial gradients and the mutual linkages of several ecosystem services and socioeconomic variables in the urban-rural areas of Leipzig, Germany, and Kunming, China. These two areas are affected by intensive human-environmental interactions and can represent the regions experiencing reurbanization in the east of Germany and rapid urbanization process in the southwest of China, respectively. The results showed some regularities in the spatial patterns of ecosystem services and socioeconomic dimensions in both study areas. Habitat quality and f-evapotranspiration of Leipzig and habitat quality of Kunming demonstrated apparent trends of increases along all gradient patterns. However, the other ecosystem service presented divergent spatial variabilities in different gradient patterns. Road density, urban fabric and population density showed identical declining trends in both case study areas except for the soaring of population density around the center of Leipzig. Differently, household size, housing area as well as unemployment rate in Leipzig presented inconsistent spatial dynamics with considerable fluctuations. Regarding the gradient interrelations, road density, urban fabric and population density were strongly correlated with most ecosystem services in both case study areas. In contrast, the gradients of household size, housing area and unemployment rate of Leipzig showed inconsistent correlations with the ecosystem services gradients.

Keywords: human-environmental systems, spatial dynamics, gradient patterns, quantitative, comparison, uncertainties

1. Introduction

1.1. The role of ecosystem services in tackling the issue of human-environment interactions

As a multidisciplinary principle, the concept of ecosystem services can be used to tackle the issue of human-environment interactions by linking natural sciences with social sciences. Being defined as the benefits people acquire from natural system, ecosystem services cannot stand without human perception and use (Costanza et al., 1997; TEEB, 2010). The relationship between ecosystem services and human well-being can be either synergies or trade-offs, depending on which types of ecosystem services and human well-beings and what scales are considered (Duraiappah, 2011; Haines-Young and Potschin, 2010; MA, 2005; Raudsepp-Hearne et al., 2010). Human activities can act as drivers for the changes of ecosystem services. These drivers primarily involve demography, economy growth, social policies, culture, religion as well as science and technology (MA, 2005; Nelson et al., 2005). Drivers induce pressures that directly affect ecological integrity and furthermore, influence the ecosystems' generation of services (Hou et al., 2014). Pressures are often referred to land use change, agriculture production activities or industrial activities (Alcamo et al., 2005; Bastian, 2013; Nelson et al., 2005). Apart from the impacts caused by drivers and pressures, the response, relating to social political influences, e.g. governmental regulations, can play a positive role in preserving ecological integrity and enhancing the supply of ecosystem services (Couch and Karecha, 2006; Nuissl and Rink, 2005; Schwarz et al., 2011). Owing to the encouraging role in addressing human-environment interactions, ecosystem services approach is promising in assisting decision-making processes of local and regional environmental management (de Groot et al., 2010; Polasky et al., 2011; TEEB, 2010).

1.2. Ecosystem services and socioeconomic gradients in urban-rural areas

Urban-rural areas have received increasing concerns from ecosystem service investigators due to the accompanying intensive human-environment interactions (Kroll et al., 2012; Radford and James, 2013). Among the associated surveys, the ones performed in European urban-rural areas constitute a primary part (e.g. Larondelle and Haase, 2013). These investigations involve food provisioning, regulating services such as local or global climate regulation and pollination as well as recreation (Haase et al., 2012; Larondelle and Haase, 2013; Lautenbach et al., 2011; Radford and James, 2013; Schwarz et al., 2011; Strohbach and Haase, 2012). A German research group has conducted comprehensive surveys of ecosystem services in Leipzig urban-rural gradient in eastern Germany. These surveys mainly relate to the issues such as ecosystem services synergies and trade-offs (Haase et al., 2012; Larondelle and Haase, 2013), ecosystem services value transference (Strohbach and Haase, 2012), ecosystem services' historical changes as well as the sensitivity and robustness of the evaluation results (Lautenbach et al., 2011).

The principle of urban-rural gradients was initially proposed to delineate the features of urbanization (McDonnell and Pickett, 1990). These characteristics primarily involve land cover structures, the biota of urban-rural area and socioeconomic dimensions (Hahs and McDonnell, 2006; McDonnell and Pickett, 1990). Based on the assessments of ecosystem services, some scientists further sketched urban-rural gradients of ecosystem services. The results concerning different provisioning, regulating and cultural service paint such a picture: Some services, such as food and water supply (Kroll et al., 2012), aesthetic services as well as spiritual service (Radford

and James, 2013), present an increasing trend along the urban-rural gradient in the associated study areas. Others such as above ground carbon storage or sequestration (Larondelle and Haase, 2013; Radford and James, 2013; Strohbach and Haase, 2012), energy supply (Kroll et al., 2012), water flow regulation, pollination potential as well as recreation (Radford and James, 2013), show fluctuations from city center to fringe. Noticeably, although apparent ecosystem services gradients have been found in different urban-rural areas, a typical model of ecosystem services gradient is evidenced not to be existing for different cities (Larondelle and Haase, 2013).

The issue of socioeconomic gradients in urban-rural areas has gained a growing popularity, not only in social science community, but also in some interdisciplinary subjects including environmental science (Dow, 2000; McDonnell and Hahs, 2008). Kroll and Kabisch (2012) and Kroll and Müller (2011) depicted the spatial gradients of human dimensions in urban-rural landscapes in several agglomeration in Europe (see Piorr et al., 2011). These human aspects primarily refer to population density, urban fabric, dwelling area, sealed surface, employment state, mean age as well as household size. The findings suggest the suitability of the gradient approach to delineate and understand the structural differences between urban areas and the surrounding environment. Another study performed in Australia reveals the demographic and employment gradients in urban-rural gradients in Melbourne (Hahs and McDonnell, 2006). In addition to the investigations describing the gradients of objective socioeconomic states, the gradient of subjective well-being expressed as “happiness” was explored in the US. This survey presents a rising trend of happiness from large central cities to small towns or the rural periphery (Berry and Okulicz-Kozaryn, 2011).

The literature investigations show that ecosystem services and socioeconomic gradients in urban-rural areas have been drawing increasing concerns from interdisciplinary scientists as these two approaches can uncover human-environmental interactions in urban-rural space. Whereas, studies on these two issues are still limited. Furthermore, we found little literature investigating the mutual linkages between the two issues. These shortages diminish the ability of scientists to understand how human and environmental systems are interplaying in urban-rural areas.

Consequently, our research aimed at bridging the revealed gap by spatially explicitly and quantitatively characterizing urban-rural gradients of ecosystem services and socioeconomics and exploring their mutual linkages. Accordingly, we propose the following three research questions:

- 1) What are the spatial and quantitative characteristics of ecosystem service and socioeconomic gradients in the investigated urban-rural areas?*
- 2) What are the similarities and differences of ecosystem service and socioeconomic gradients between different gradient patterns of distinct areas?*
- 3) Are there linkages between ecosystem service and socioeconomic gradients in urban-rural areas?*

In order to answer these questions, we performed empirical studies in the urban-rural areas of two cities, Leipzig, Germany, and Kunming, China. The former is a typical representative of the cities in the east of Germany witnessing their reurbanization, while the latter can be understood as a proxy of the cities in the southwest of China experiencing rapid urbanization process. These two research areas are both facing intensive human-environmental interactions and dynamics.

In the forthcoming text, we firstly provide a brief introduction of the natural and socioeconomic status of the two study areas. Secondly, the research methods are described. Thereafter, the findings are presented in the results section, which relates to the gradient characteristics, the comparisons between different gradient patterns and case areas and the linkages between ecosystem services and socioeconomics. In a fourth step, we discuss the quantitative relationships of ecosystem services and socioeconomics gradients, explain some findings revealed by the comparisons and quantitatively analyzed the uncertainties in characterizing the gradients. Lastly, the research questions are answered in the conclusions section and the implications of our study as well as future research problems are indicated.

2. Research areas

2.1. Leipzig urban-rural area

The research area is situated in Eastern Germany, covering Leipzig city and the adjacent municipalities. It is a circular area with the center of Leipzig city as the centroid and with a radius of 25 kilometers. The primary land covers of the entire research area are artificial areas, agricultural land, forests as well as semi-natural areas (Fig. 1). Located in a temperate transition zone, Leipzig has an oceanic-continental climate with an average annual temperature of 8.8 °C and 511 mm precipitation (Leipzig Amt für Statistik und Wahlen, 2009). Leipzig is surrounded by intensively cultivated agricultural land with large areas of commercial and industrial land. Eight percent of Leipzig is covered by green space and seven percent by forest (Leipzig Amt für Statistik und Wahlen, 2009). In the south of the city are located several lakes, some of which originate from former lignite open-cast mines (Strohbach and Haase, 2012).

Leipzig has a population of approximately 520,000 in 63 urban districts covering an area of 297 km² (Leipzig Amt für Statistik und Wahlen, 2009). The densest population can be found in districts with prefabricated multi-story housing estates. Surrounding the city are the municipalities of low density incorporated into Leipzig since 1999. Leipzig's population dropped by 12% between 1990 and 2000 due to suburbanization and out-migration, but grew by 2.7% between 2000 and 2006 (Saxon Office of Statistics, 2010). Historically, Leipzig was a center of commerce and trade. Lignite mining and chemical industry constituted two primary economic sectors from the 19th century to the German reunification in 1990. After the German reunification, the economy of Leipzig recessed facing national and international competition. Throughout these years new developmental directions tentatively emerged in the service sector and in fields such as education, transportation and tourism (Haase et al., 2012).

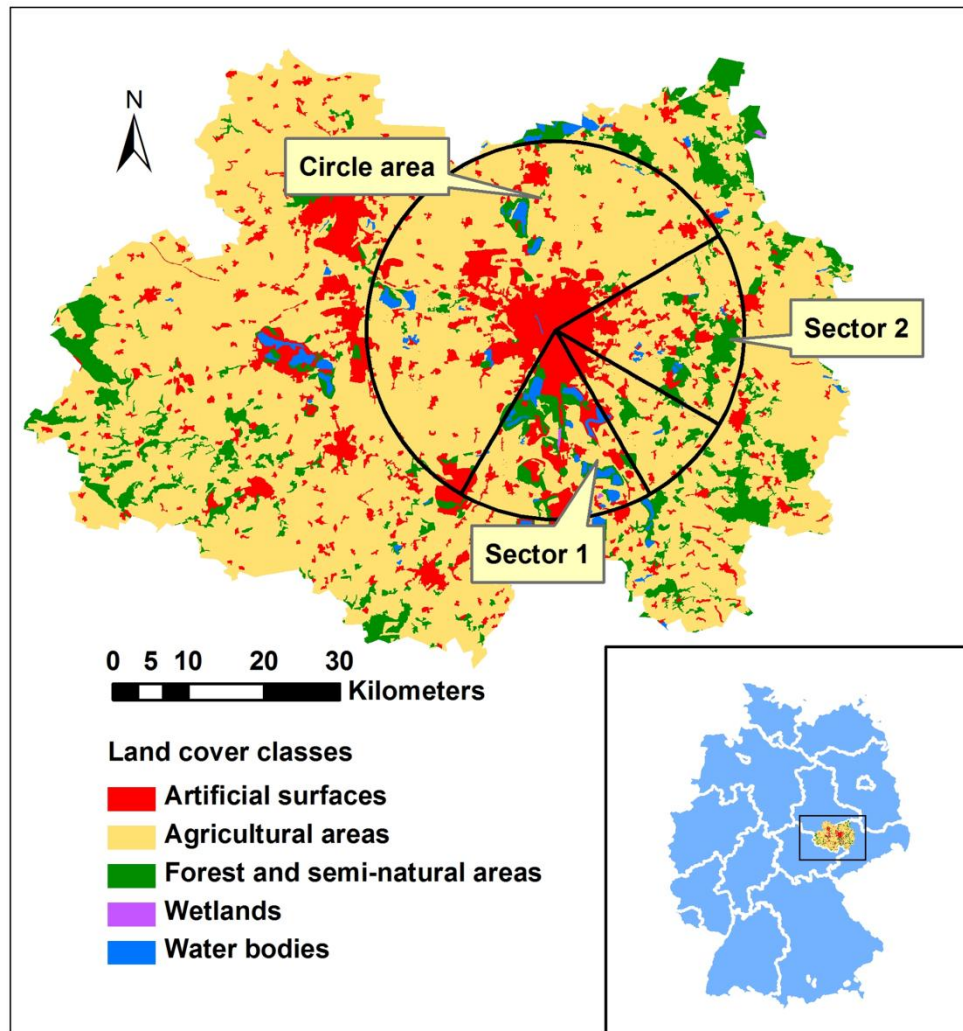


Fig. 1 Land cover map of the Leipzig urban-rural area for 2006. The five primary classes are aggregated from Corine land cover classes (German Federal Environmental Agency, 2014). The circle area, sector 1 and sector 2 are the investigation areas with the city center of Leipzig in the centers and radiuses of 25 kilometers for gradient analysis

2.2. Kunming urban-rural area

Kunming is the capital of Yunnan province in Southwestern China (Fig. 2). This research area consists of six districts within the Kunming region: Wuhua, Panlong, Guandu, Xishan, Chenggong and Jinning. The former four districts constitute the location of Kunming city. Situated in the Dianchi Lake basin, Kunming is surrounded by mountains in the north, east and west and connected with the Dianchi Lake in southern direction. It is situated at an elevation of around 1890m on the Yunnan-Guizhou Plateau. Kunming enjoys a subtropical, mild highland climate throughout the whole year with humid summers and dry winters. The mean annual precipitation and mean annual temperature are 1000 mm and 15 °C, respectively (Statistics Bureau of Kunming, 2011). The entire study area covers 4076.30 km² including Kunming city with an area of 212 km². Primary land covers of this area are urban and rural residential areas,

cropland in the southeast of the city and in the east and south of the Dianchi Lake, water bodies virtually consisting of the Dianchi Lake as well as forests mainly covering the peripheral mountains (Fig. 2).

The research area has a population of 3,870,000 with 3,000,000 belonging to Kunming city (Statistics Bureau of Kunming, 2011). Besides the city area with high population density, noticeable contributions to the total population are provided by the urban and rural residential areas of Chenggong as well as the rural dwelling areas around Jinning. Kunming has experienced rapid economic growth and urbanization during the last two decades. The gross domestic product (GDP) increased from 16.9 billion RMB Yuan in 1992 to 63.6 billion in 2000. From 2000 to 2010, the GDP more than tripled, reaching 212 billion RMB Yuan (Statistics Bureau of Kunming, 2011). The urbanization process of Kunming city had caused a great loss of green space in the suburban areas and an expansion of urbanized area from 184.4 km² in 1992 to 275 km² in 2010 (National Bureau of Statistics of China, 2011).

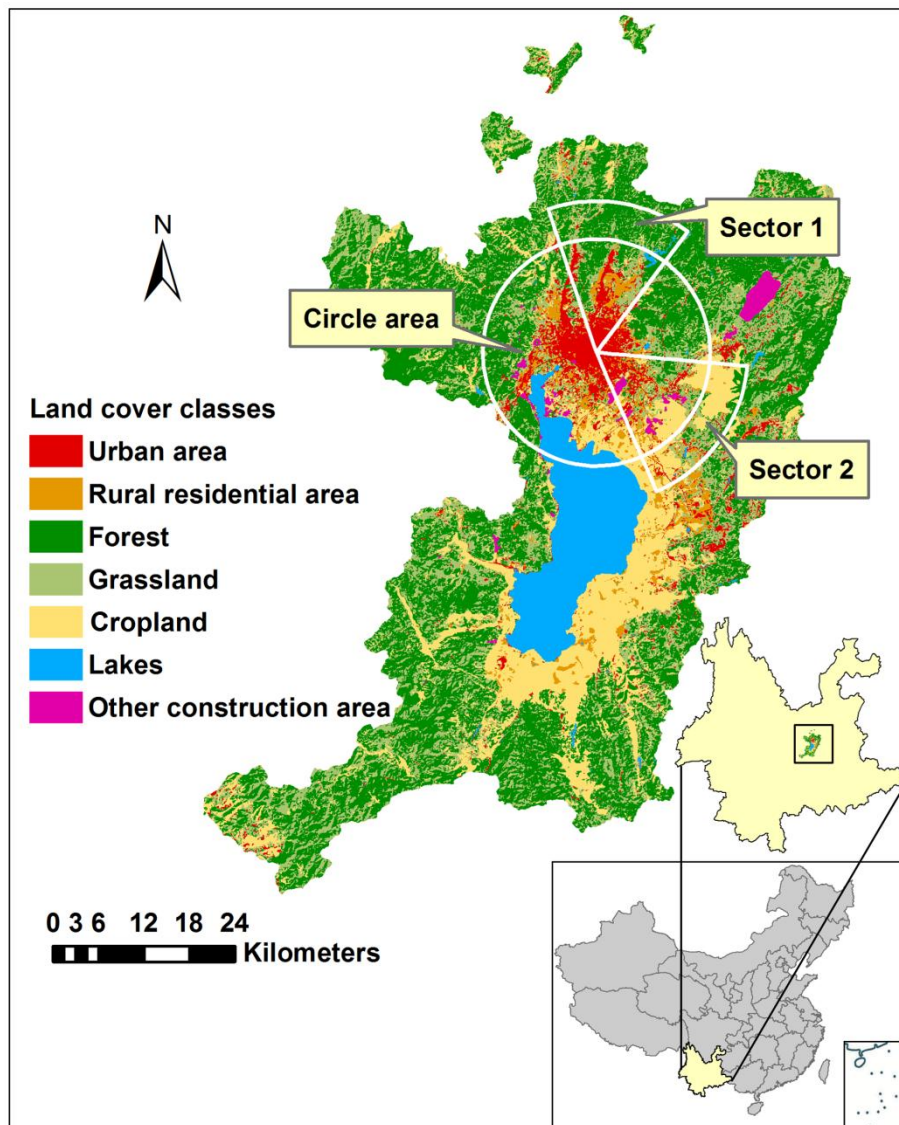


Fig. 2 Land cover map of the Kunming urban-rural area for 2010. The seven primary classes have been aggregated from the classes of the National Land Classification (applicable in transitional period) released by the Ministry of Land and Resources of China in 2002⁷. The circle area with a radius of 15 kilometers and sectors 1 and 2 with radiuses of 20 kilometers are the areas for gradient analysis

3. Methods and data sources

3.1. Data sources

In this research we mainly used biophysical data (land cover, climate, soil, etc.) and socioeconomic statistical and spatial data (see Table 4 and Table 5 in the appendix for the data, time span, scales and sources). The data were derived from associated regional or national governmental agencies through online or offline accessibilities, online open access data sources of scientific institutions as well as published work.

3.2. Urban-rural gradient analysis

The gradient approach is proven to be promising in revealing the spatial variations of environmental states, ecosystem processes and functions as well as socioeconomic characteristics in urban-rural areas (Kroll and Müller, 2011; McDonnell et al., 1997; McDonnell and Hahs, 2008; McDonnell and Pickett, 1990; Pouyat and McDonnell, 1991). These delineations can be either qualitative or quantitative (Kroll and Müller, 2011; McDonnell and Pickett, 1990). Moreover, by combining GIS methods with the urban-rural gradient approach, the changes can be explicitly visualized (Kroll et al., 2012; Larondelle and Haase, 2013). Up to now, two mainstreaming gradient patterns have been developed and applied, namely, concentric ring gradients dispersing from the city center to rural outskirts (Kroll et al., 2012) and transect gradient directionally crossing the city center and extending to rural areas with its two ends (Hahs and McDonnell, 2006; Zhou and Wang, 2011).

In our study, we used the concentric ring gradient method reported by Kroll et al. (2012) to spatially and quantitatively describe the distributions and variations of ecosystem services and socioeconomics of our research areas. Additionally, we modified this type of gradient and developed concentric sector ring gradients which can detailedly sketch specific sectors of the total circle area. For the Leipzig urban-rural area we created three concentric buffers to investigate gradients. One is a 25-km concentric ring area with 1-km intervals spreading out from Leipzig city center. The other two are two concentric sector ring areas with the same radius and intervals dispersing from the city center to the south and east respectively (Fig. 1). For the Kunming urban-rural area also three concentric gradients with 1-km intervals were created including a circle area and two sector areas. All starting from Kunming city center, the circle area extends 15 km and the two sectors disperse to the north and southeast with radiuses of 20 km (Fig. 2). When selecting the respective two concentric sector ring areas of the two study areas, we considered that the two sectors can capture the areas with apparently different land use structure and distribution compared to the rest of the entire circle area.

⁷<http://www.diji.com.cn/xl-8-12-5.html> (accessed on 4.17.2014)

Land covers have significant impact on ecosystem service supplies and the urban-rural distributions of socioeconomic elements (Burkhard et al., 2014; Jenerette et al., 2007; Luck et al., 2009). In this study land cover gradients were quantified and shown by area proportions of different land cover classes in each rings of all the gradient areas of the two study areas, respectively. With these results, we can to some extent explain the gradient characters of ecosystem services and socioeconomics.

In this study we used GIS techniques to delineate the features of several ecosystem services and socioeconomic dimensions across the urban-rural space in the two case areas. Quantitative changes of the gradients were further shown with the average values of each ecosystem service and socioeconomic indicators for each concentric ring area. The findings in the forms of maps and line charts are given in the results section. Considering the uncertainty propagation with spatial aggregation and value averaging (Hou et al., 2013), we quantitatively performed uncertainty analyses to the calculations of selected indicators. The findings shown in the discussion section can provide insights into the magnitudes and variation trends of uncertainties along the urban-rural gradients of the two research areas.

3.3. Indicator selections, calculations and gradient creating

Ecosystem services indicators can be proxies of different ecosystem service types. Calculating the indicators or measuring the values is a mainstreaming approach to quantify ecosystem services (Müller and Burkhard, 2012). Appropriate ecosystem service indicators should be quantifiable, sensitive to land cover changes, spatially and temporally explicit and scalable (Burkhard et al., 2012). The ecosystem service indicators chosen in this study are highly sensitive to land cover changes, so their spatial variation in the considerably dynamic urban-rural environments can be efficiently captured. The indicators were calculated by robust methods from published literature, or based on empirical or statistical data (see Table 1 for the detailed information on the indicators)

Intensively modified by humans, urban-rural systems noticeably present socioeconomic characters. Demonstrating the intensity and extensity of socioeconomic activities requires appropriate indicators which are sensitive to socioeconomic spatial and temporal dynamics. In this study we chose indicators involving the three primary human dimensions of urban-rural system: artificial environment, socio-demography and economy (Kroll and Müller, 2011). The calculations of the indicators were based on simple formulas with statistical or spatial data (Table 1).

Table 1 Overview of the quantified ecosystem service and socioeconomic indicators and the calculation methods

Case area	Indicator	Unit	Quantification method	Data sources and references
Leipzig ecosystem services	Habitat quality	relative value	InVEST biodiversity model	Tallis et al., 2013
	Above ground carbon storage	MgC/ha	Look-up table with above ground carbon storage per land cover type	Larondelle and Haase, 2013
	f-evapotranspiration	relative value	Look-up table with f-evapotranspiration value per land cover type	Larondelle and Haase, 2013
	Food supply	GJ/ha	Regression model for land cover and soil types (yield)	Kroll et al., 2012
	Energy supply	GJ/ha	Assignment of energy supplies of different energy types to associated land cover classes	Kroll et al., 2012
Leipzig socioeconomics	Road density	km/km ²	Road length divided by area of the associated ring	See data sources in Table 4 in the appendix
	Urban fabric	%	Urban fabric area divided by total area of the associated ring	See data sources in Table 4 in the appendix
	Population density	persons/km ²	Leipzig municipality and city district population statistical data	See data sources in Table 4 in the appendix
	Household size	persons/household	Leipzig municipality and city district population statistical data	See data sources in Table 4 in the appendix
	Housing area	m ² /person	Leipzig municipality and city district population statistical data	See data sources in Table 4 in the appendix
	Unemployment rate	%	Leipzig municipality and city district population statistical data	See data sources in Table 4 in the appendix
Kunming	Habitat quality	relative value	InVEST biodiversity model	Tallis et al., 2013

ecosystem services	NPP	gC/m ² /yr	MODIS GPP/NPP Project (MOD17)	Numerical Terradynamic Simulation Group, 2013; Lin et al., 2008; Sun et al., 2011; Chinnaiah and Madhu, 2010
	Food productivity	yuan/km ²	Assignment of food outputs of different food type to associated land cover classes	See data sources in Table 5 in the appendix
Kunming socioeconomics	Road density	km/km ²	Road length divided by area of the associated ring	See data sources in Table 5 in the appendix
	Urban fabric	%	Urban fabric area divided by total area of the associated ring	See data sources in Table 5 in the appendix
	Population density	persons/km ²	Kunming districts and counties population statistical data	See data sources in Table 5 in the appendix

3.3.1. Ecosystem service indicators

Habitat quality: Leipzig and Kunming

Biodiversity plays an essential role in ecosystem functioning and ecosystem service production. Although not usually regarded as a service, biodiversity is an important boundary condition in ecosystem service tradeoff analysis and decision-making (MA, 2005). Traditionally approaches to measure biodiversity relate to investigating richness and abundance of species, numbers of ecosystem types and community characteristics. Despite the universal and robust surveying methods, integrated methodology to fast and comprehensively quantify regional biodiversity is still rare. Habitat quality is a proxy of land's biodiversity support capability. It involves the capability of the ecosystem to provide conditions appropriate for individual and population persistence (Tallis et al., 2013). Habitat quality modeling can be an acceptable solution to reflect the overall potential of regional biodiversity. One novel and promising modeling tool is the InVEST biodiversity model, which combines information on LULC and threats to biodiversity to produce habitat quality maps (Tallis et al., 2013). For the details of the model introduction, working principles, data needs, results interpretations as well as model limitations, see the InVEST 2.5.5 User's Guide (Tallis et al., 2013).

This study employed the InVEST biodiversity model to quantify and visualize the habitat qualities of the two study areas. For the Leipzig urban-rural area, we used the Corine land cover spatial data of 2006 as the current land cover map and considered cropland, urban area, roads and railways as biodiversity threats. For the Kunming urban-rural area, a land cover map rasterized from a 1:100,000 shapefile map of 2010 was used and cropland, urban and rural residential area, construction area as well as roads were assigned as threats. The two biophysical tables of each case area were adapted from the sample data of the InVEST biodiversity model with changes based on the land cover classifications of the associated case area. The derived habitat quality raster maps were converted to polypoint shapefiles in order to calculate the average values of the concentric rings of the circle and sector areas. Based on this, the habitat quality gradients were generated in the form of line charts. The modeling results are relative values between 0 and 1. On the habitat quality maps areas with higher values enjoy better habitat quality compared to the rest of landscape. It is important to note that the habitat quality values of Leipzig and Kunming are not comparable as the model parameter inputs (e.g. the biophysical tables) were different from each other.

Above ground carbon storage: Leipzig

Urban-rural areas can be both carbon sinks and carbon sources. On the one hand, the intensive human activities in the cities release considerable amount of greenhouse gases into the atmosphere. On the other hand, vegetation covers in urban-rural areas store and sequester carbon in a noticeable amount (Strohbach and Haase, 2012). Being an important part of the global carbon cycle, terrestrial carbon storage consists of four layers: above ground biomass, deadwood and litters, below ground biomass and soil carbon storage. It is difficult to completely count the carbon storage of the four layers in terrestrial ecosystems. Whereas, measuring and estimating above ground carbon storage is proved to be feasible and the results are believed to be credible (Haase et al., 2012; Strohbach and Haase, 2012). In this study the indicator "above ground carbon storage" was chosen to represent urban-rural ecosystems' global climate regulation services. As above ground biomass is sensitive to land cover types, estimating above ground carbon storage of

different land cover classes can reveal the spatial dynamics of contributions from urban-rural ecosystems to mitigate climate change.

To calculate this indicator, empirical values from research proceeded in Leipzig city were used in this study (Larondelle and Haase, 2013; Strohbach and Haase, 2012). Above ground carbon storage values of different Corine land cover classes were derived by transferring the measured above ground carbon stock of the trees in the urban-rural gradient of Leipzig (Larondelle and Haase, 2013). By linking the empirical values to the associated land covers, we mapped the spatial distributions of the above ground carbon storage of the Leipzig urban-rural area. Furthermore, the quantitative changes of this service along the urban-rural gradient were characterized by plotting the average values in each concentric ring of the circle and the two sector areas.

Net primary production (NPP): Kunming

Representing the net carbon uptake from the atmosphere into biosphere, net primary production (NPP) is widely used to evaluate the patterns, processes and dynamics of carbon cycling in ecosystems, assess ecosystem status and monitor environmental changes (Ingraham and Foster, 2008; Melillo et al., 1993; Zhao and Zhou, 2005). Given the role NPP plays in carbon cycling, this study uses it to indicate the global climate regulation service of Kunming urban-rural area.

We distinguished NPP quantification into three methods based on different data sources and existing research concepts. For vegetation NPP, the MOD17A3 NPP product produced by the Numerical Terradynamic Simulation Group of the University of Montana was used (see Table 5 in the appendix for data sources). The NPP value of urban area was transferred from Lin et al. (2008). The NPP of the Dianchi Lake was assessed with reference to Sun et al. (2011) and Chinnaiah and Madhu (2010).

The MOD17A3 NPP product is generated from NASA MODIS data by the Numerical Terradynamic Simulation Group. It provides continuous estimates of NPP across the earth's entire vegetated land surface. The NPP is provided at 8-day, monthly, and annual time steps in GeoTIFF of 1km× 1km. For this research we used the annual global NPP of 2006 and extracted the Kunming urban-rural area from the global map. Lin et al. (2008) estimated the NPP of the urban area of Shanghai, China based on urban forest inventory data of 2006 and remote sensing. Considering the similarities of climates between Kunming and Shanghai, we transferred the NPP value of Shanghai city from Lin et al. (2008) to the Kunming urban area. Sun et al. (2011) measured the surface water gross primary production (GPP) of the northern, middle and southern the Dianchi Lake from winter of 2009 to the spring of 2010. Chinnaiah and Madhu (2010) estimated the NPP/GPP ratio from October 2009 to September 2010 for the Daromasagar Lake in India. Our study adapted the findings of these two studies to estimate the NPP values for the parts of the Dianchi Lake that belong to the six districts of the research area. For the calculation, GPP values of spring from Sun et al. (2011) were used to calculate the annual average GPP of the Dianchi Lake. The mean NPP/GPP ratio for the six spring months from Chinnaiah and Madhu (2010) was used to convert GPP to NPP.

By joining the NPP derived from the above described methods using the union tool of ArcGIS 10, we produced the NPP map for the whole Kunming urban-rural area. Furthermore, the map was resampled to grids of 200 m × 200 m and intersected with the concentric rings of the circle and two sector areas. Thereafter, the Kunming urban-rural gradients of NPP was generated by

averaging the values in each concentric ring and shown by line chart in the result section of this article.

f-evapotranspiration: Leipzig

The urban heat island is a typical urban environmental issue generated by the interactions between human systems and the environment. Owing to the heated urban atmosphere, urban residents particularly demand for local climate regulation service. Evapotranspiration plays a significant role in regulating local temperature, humidity and precipitation. Therefore, it is usually employed to indicate ecosystem's capacity to regulate local climate (Burkhard et al., 2009; Kandziora et al., 2013; Schwarz et al., 2011).

In our study "f-evapotranspiration" was used as a local climate regulation service indicator. Referring to latent heat flows, f-evapotranspiration is an approximation for the evapotranspiration potential of a land use class (Schwarz et al., 2011). According to Schwarz et al. (2011), f-evapotranspiration is calculated with the following equation:

$$f(i) = \frac{\max \text{evapotranspiration}(i)}{ET_{(0)}} \quad (1)$$

Here $\max \text{evapotranspiration}(i)$ is the maximal evapotranspiration of land cover type i and $ET_{(0)}$ is the reference evapotranspiration of 12 mm high grass in the local climate.

The f-evapotranspiration values of different Corine land cover classes were derived from a lookup table after Larondelle and Haase (2013). These values were transferred from the calculation and estimation (Schwarz et al., 2011) in the Leipzig urban-rural area. By linking the empirical values to the associated land covers, we showed the distribution of f-evapotranspiration across Leipzig urban-rural area. The quantitative dynamics of this service along the gradient were further presented by a line chart with the average values in each concentric ring of the circle and two sector areas.

Food supply of Leipzig and food productivity of Kunming

As one of the basic ecosystem goods, food is a fundamental material input of urban systems and crucial for the survival of the urban population (Jansson and Polasky, 2010). In this study we employed food supply and food productivity to indicate food provisioning service of Leipzig and Kunming urban-rural areas, respectively. For the details of the food supply calculation, see Kroll et al. (2012).

The food productivity of the Kunming urban-rural area was computed in RMB yuan/km² for each land cover type of the six districts. Statistical data of 2010 on crop types and outputs (cereals, oil crops, cane, tobacco, vegetables, fruits, nuts, spices and Chinese herbal medicine), sheep and caws, poultries as well as aquaculture products of different districts were used to calculate average values of food output per km² (food productivity) of cropland, forest, grassland, rural residential area and lakes of the associated district (for data sources see Table 5 in the appendix). Pig production was not considered as pig farms cannot clearly be classified into a certain land cover type.

Based on the above described calculations, we linked the food supply per hectare of Leipzig and the food productivity of Kunming to the associated land cover types of the spatial land cover data with ArcGIS, respectively. Accordingly, the maps of food provision of the Leipzig and Kunming

urban-rural areas were produced. Moreover, food provision values were averaged in each concentric ring of the circle and sector areas of Leipzig and Kunming.

Energy supply: Leipzig

Energy is another important good natural system provides for human well-being. Although some energy sources are not regarded to be strongly linked with ecosystem processes and functions (e.g. wind energy and solar energy), they can have synergies or tradeoffs with some ecosystem services and be considerably influential in the decision making procedures of environmental management (Burkhard et al., 2014). This study calculated energy supply of Leipzig urban-rural area for 2007 in the unit of GJ/ha and delineated its spatial distributions and gradients (for data sources, see Table 4 in the appendix). For details of the respective quantification methods, see Kroll et al. (2012).

3.3.2. Socioeconomic indicators

Road density and urban fabric: Leipzig and Kunming

To calculate the road density of the Leipzig urban-rural area, we used road network maps of 2006 for the entire Germany and the Leipzig city district level, respectively and a railway network map of 2008 for the Leipzig city district level (see Table 4 in the appendix for data sources). For Kunming, the road network map of 2007 at the China national level was used (see Table 5 in the appendix for data sources). Then we intersected these maps with the maps of the circle and sector areas of concentric rings of Leipzig and Kunming, respectively. Based on this, we calculated road densities of each concentric ring by dividing road length by area of the corresponding concentric ring. Urban fabric was computed in the unit of percent by dividing the urban fabric area by total area of each concentric ring. This method was applied both to the circle and sector areas of Leipzig and Kunming. To accomplish this calculation, urban fabric maps were extracted from the land cover maps of Leipzig for 2006 and Kunming for 2010, respectively (see Table 4 and Table 5 in the appendix for data sources).

Population density, household size, housing area and unemployment rate: Leipzig

To estimate the values of these four indicators, socio-demographic data from local and regional statistic offices were used (see Table 4 in the appendix for data sources). The calculation methods were developed and described by Kroll and Kabisch (2012).

Population density: Kunming

To estimate the population density of the Kunming urban-rural area, we used population and land cover data of the six districts constituting the research area (for data sources see Table 5 in the appendix). Urban population and rural population were assigned to urban area and rural residential area, respectively. The other five land cover classes, forest cropland, grassland, lakes and construction were assumed not to be populated. Then we calculated the population densities of the urban and rural residential areas of the six districts. By joining the results with the land cover map in ArcGIS, A population density map was produced. Afterward, we converted the map to a grid map with the cell size of 200 m ×200 m, intersected it this the concentric ring and concentric sector ring maps and computed the average values in each concentric ring. Finally, the population gradient line chart of Kunming urban-rural area was derived.

3.4. Comparisons of gradient patterns and case areas

Burgess (1967) developed a concentric zone model of residential differentiation for urban spatial analysis. This model led to the application of concentric rings around the city center as spatial units of analysis. The concentric ring method was more recently applied by e.g. Kroll and Kabisch (2012), Schneider and Woodcock (2008) and Solon (2009). The approach involves generalizations, as the spatial structure of a city area does not exhibit the same pattern in all directions from the center (Zheng, 1991).

In order to examine the appropriateness of the concentric ring and concentric sector ring gradients in this study, we compared the three gradients of these two patterns. The comparisons were performed with the land cover structure, each ecosystem service and socioeconomic item investigated in this study for Leipzig and Kunming urban-rural areas, respectively. To achieve this purpose, we presented the three gradients in the same line chart to reveal the mutual similarities and differences. These comparisons can on the one hand reveal the spatial heterogeneities of the compared objects and on the other hand, address the uncertainties of the concentric ring gradient pattern, which are generalized from the heterogeneous patches in the ring. The comparisons are described in the results section. Furthermore, we quantified and presented the uncertainties of the three gradients with four indicators in the discussions, namely above ground carbon storage and housing area of Leipzig and habitat quality and population density of Kunming.

The two case areas of this study, may exhibit different spatial dynamics and urban-rural gradients with ecosystem services and socioeconomic dimensions. These variations can be caused by their distinct physical conditions, urbanization processes and socio-political systems. To examine the applicability of the concentric ring approach in these two areas, we compared their ecosystem service and socioeconomic gradients and described the similarities and differences in the results section.

3.5. Analysis of ecosystem service-socioeconomics gradient linkages

Ecosystem service supply and demand in urban-rural systems are reported to be closely linked with regional human activities (Burkhard et al., 2012; Hou et al., 2014; Kroll et al., 2012). Accordingly, we assume that ecosystem service and socioeconomic gradients should correlate in the urban-rural contexts of the two research areas.

In this study we firstly observed the qualitative linkages between the investigated ecosystem services and socioeconomic items by comparing their gradients shown by the line charts. The observations are described in the results section. Furthermore, we applied Spearman correlation analyses between each pair of ecosystem service and socioeconomic indicators for the three concentric ring areas of the two case areas. Therefrom, the quantitative correlations between ecosystem service and socioeconomic gradients were revealed and these relationships are presented and discussed in the discussion section of this article.

4. Results

4.1. Land cover gradients in Leipzig and Kunming urban-rural areas

Fig. 3 shows sharp and opposite changes of artificial surfaces and agricultural areas along the urban-rural gradient in the circle area of Leipzig. Artificial surfaces share plummeted from 100% share of the total area in the city center to around 20% at the distance of 10 km from the city. In contrast, agriculture areas proportion soared from zero to more than 60% 12 km away from the city center. The other three primary land covers' shares: forest and semi-natural areas, wetlands and water bodies present slight fluctuations along the gradient in this area. In sector 1 of Leipzig artificial surfaces share also quickly dropped from 100% to around 20% along the gradient. The share of agricultural areas increased from zero in the city center to around 70% 17 km away and then showed fluctuations. The proportion of forest and semi-natural areas experienced noticeable fluctuations reaching the peak of about 50% at the 11th concentric ring and declined to nearly zero at the end of the gradient. The shares of wetlands and water bodies presented irregular changes at low levels along the gradient. In sector 2 the artificial surfaces proportion showed a similar plummeting trend to the circle and sector 2 areas. Agriculture areas' share soared to more than 80% at the 9th concentric ring and then went up and down till the 25th concentric ring. The other land covers showed inconsiderable fluctuations all along the gradient.

Fig. 4 presents the land cover gradients of the circle and the two sector areas of the Kunming urban-rural area. The urban area proportion showed apparent marked droppings from 100% in the city center to nearly zero 15km away from the city center in all the three concentric ring areas. The forest share kept growing from the city center to the end of the gradient in both the circle area and sector 1. In sector 2, it generally increased in fluctuations along the gradient and finally reached around 40%. The cropland shares showed notably different trends along the gradients in the three areas. In the circle area the cropland proportion gradually grew from zero in the city center to around 20% in the end of the gradient. In sector 1, it remained nearly zero all along the gradient. Differently, in sector 2, it experienced sharp going up and down between the 10th and 20th concentric rings and finally reached around 20%. The grassland proportions presented first rising and then declining trends below 40% in all the three concentric ring areas. The rural residential area shares, which were lower than 40% in all the three areas, showed variability similar to grassland along the gradients. The proportion of lakes stayed nearly zero for most of the three areas except in the last a few rings of the circle area, where it slightly grew to about 15%.

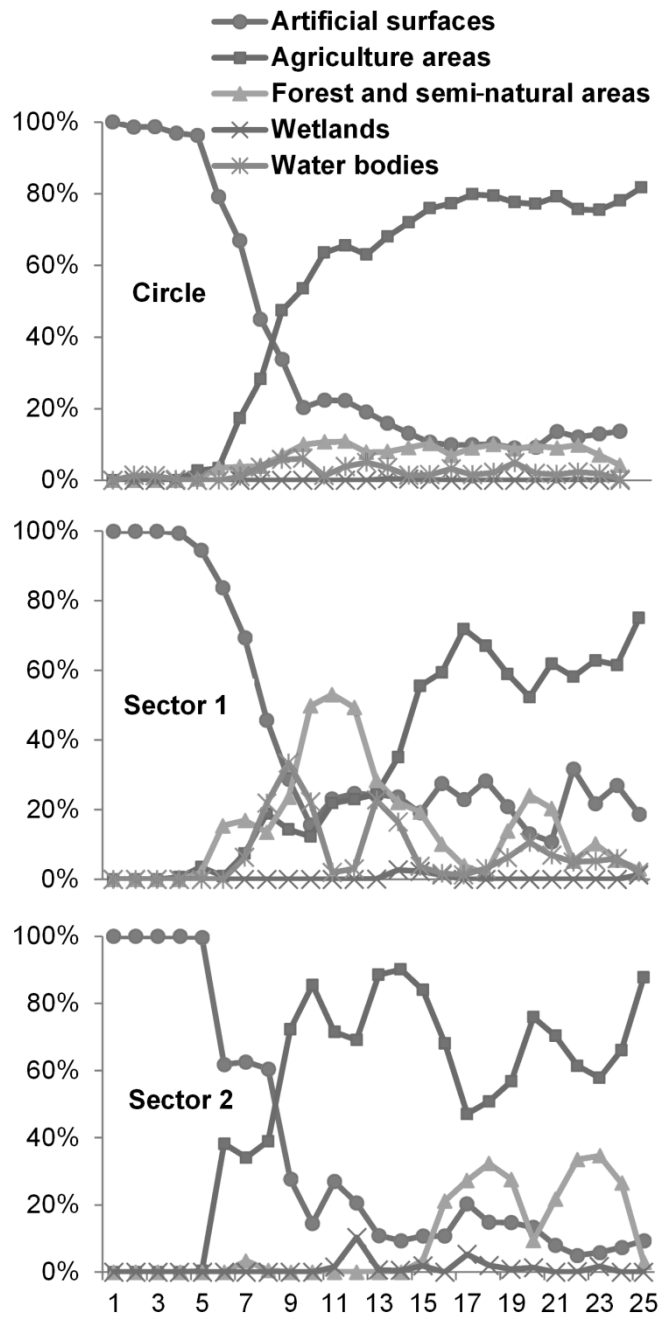


Fig. 3 Land cover gradients along the Leipzig urban-rural area. The X-axis is the distance in kilometers from the city center of Leipzig. The Y-axis is the area proportions of different land cover classes in the total area of the associated concentric ring or concentric sector ring

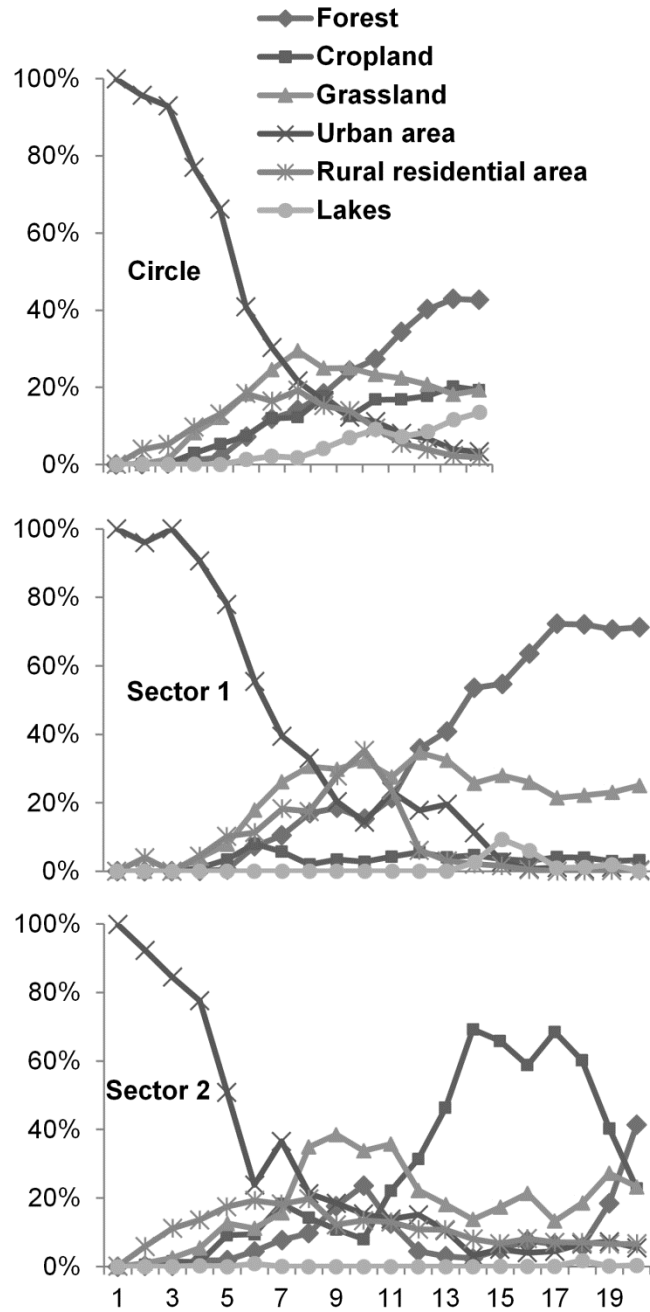


Fig. 4 Land cover gradients along the Kunming urban-rural area. The X-axis is the distance in kilometers from the city center of Kunming. The Y-axis is the area proportions of different land cover classes in the total area of the associated concentric ring or concentric sector ring

4.2. Ecosystem service and socioeconomic gradients in Leipzig urban-rural area

Fig. 5 shows the spatial distributions of the five ecosystem services along the urban-rural gradient of Leipzig. All the five ecosystem services generally demonstrated an increase along the urban-

rural gradient except the above ground carbon storage. This exception showed a reversed trend that the urban area roughly had higher above ground carbon storage than the rural area. Having said this, the urban-rural spatial variations of the five ecosystem services were not even in different parts of the circle area. Taking the sectors chosen in this study as examples, sector 1 revealed highly irregular distributions of all the five ecosystem services and sector 2 showed special irregularities with above carbon storage, food supply and energy supply.

Fig. 6 quantitatively illustrates the gradients of the five ecosystem services in Leipzig urban-rural area. The line chart of the habitat quality gradient shows that this service obviously kept increasing from the city center to the concentric ring 25 kilometers away in all the three investigated areas (circle, sector 1 and sector 2). It showed a smoother rising trend in the circle area than in sector 1 and sector 2. In the latter two areas this service showed slight fluctuations. As the value distribution map in Fig. 5 shows, above ground carbon storage generally declined along the urban-rural gradient of Leipzig. This tendency is further quantitatively proven by the line chart. It is noteworthy that in sector 2 this service showed two sudden surges in the regions around 17 and 23 kilometers away from the city center. The characteristic of the f-evapotranspiration gradient was similar to that of habitat quality, revealing an apparent value increase along the urban-rural gradient. Although food supply and energy supply appeared with higher values in the rural area than in the urban area, they enjoyed considerably irregular gradients than the other three services. Between them energy supply showed even more fluctuations along the gradient than food supply. By comparing the line charts of the five ecosystem services, a noteworthy phenomenon can be found that the services in the circle areas changed (increased or decreased) more smoothly than in the two sector areas for all the five ecosystem services.

The quantitative delineations of the six socioeconomic gradients are shown in Fig 7. Obviously, all the socioeconomic gradients of the Leipzig urban-rural area were much more regular than the ecosystem service gradients. Among the socioeconomic dimensions, road density and urban fabric showed generally declining trends along the gradient. Differently, population density, household size, housing area as well as unemployment rate appeared notable turning points in some concentric rings. Among them, population density reached the peak 3 kilometers away from the Leipzig city center and then kept decreasing. Household size grew to the peaks around 13 kilometers away from the city center and either remained stable or slightly decreased in the circle or the two sector areas. Housing area declined to the minimum around 12 kilometers away from the city center and then turned growing slowly. Unemployment rate experienced firstly increasing and then decreasing and finally rising trends for all the three areas.

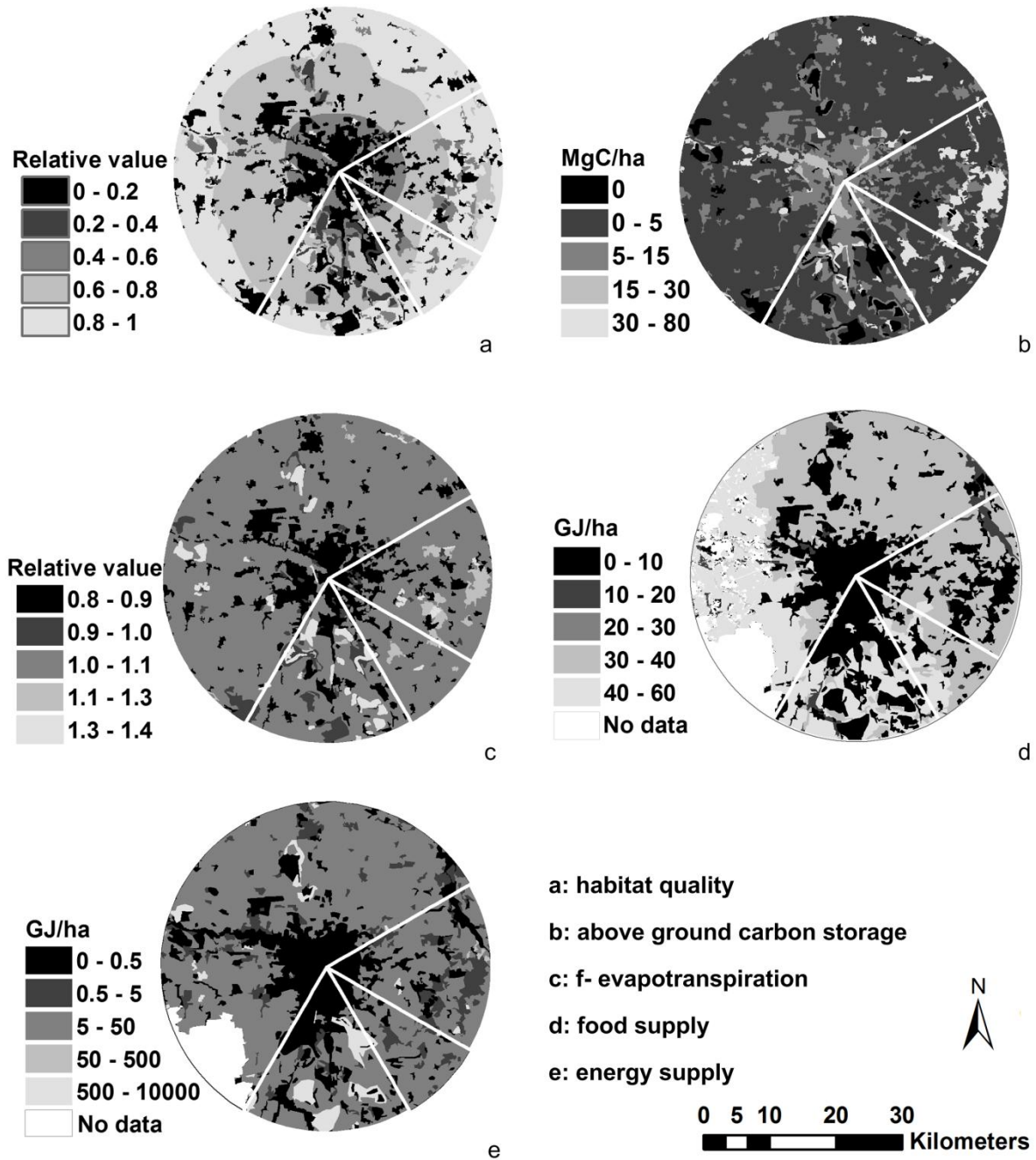


Fig. 5 Spatial distributions of the five ecosystem services in the circle and sector areas of the Leipzig urban-rural gradient. a, b, c items are for 2006 and d, e items are for 2007 (d and e are adapted from Kroll et al., 2012)

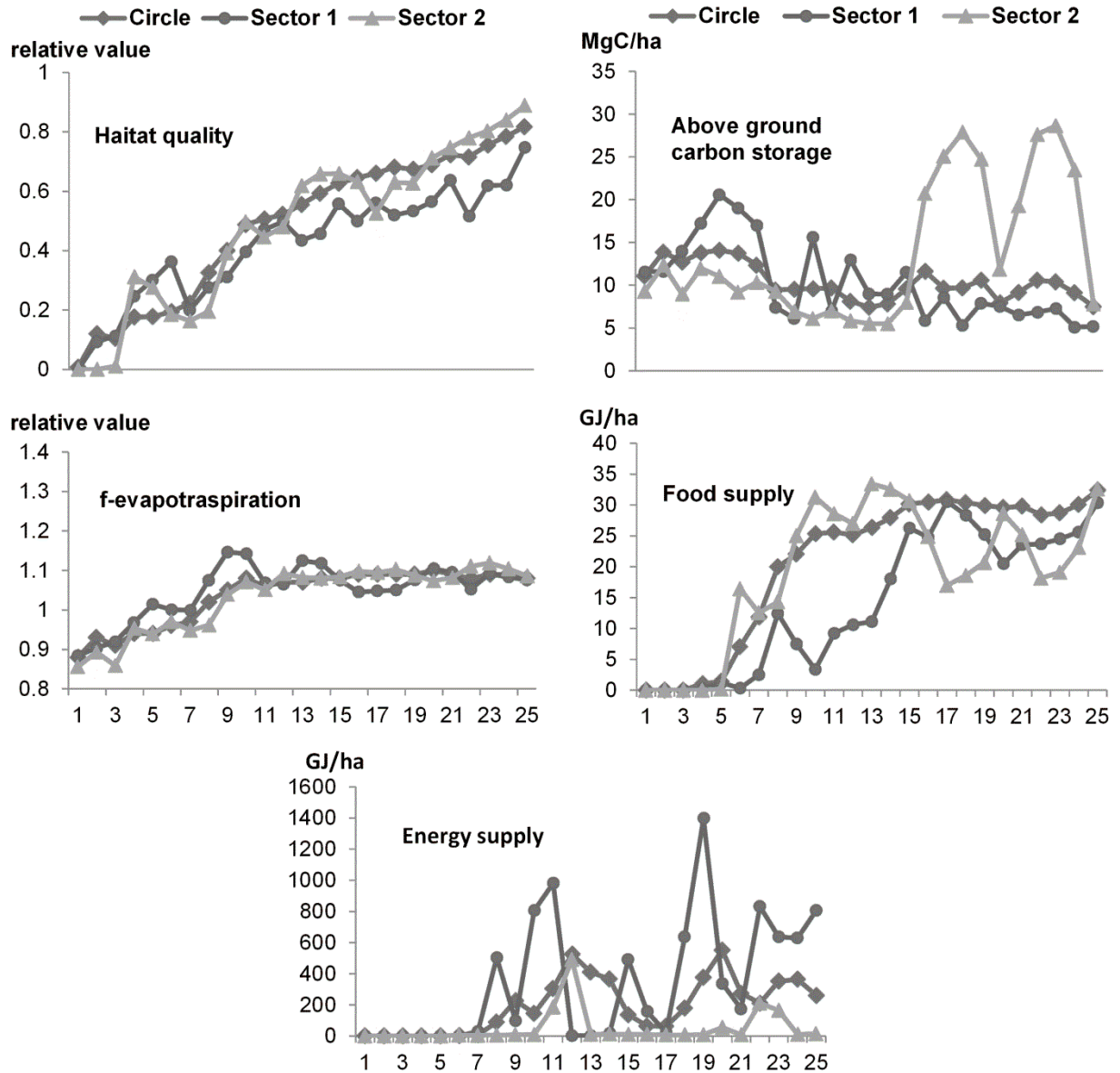


Fig. 6 Gradients of the five ecosystem services in the circle and sector areas of the Leipzig urban-rural area. The X-axis is the distance in kilometers from the city center of Leipzig. The Y-axis is the average ecosystem service value of the associated concentric ring or concentric sector ring of the circle or sectors (food supply and energy supply are adapted from Kroll et al., 2012)

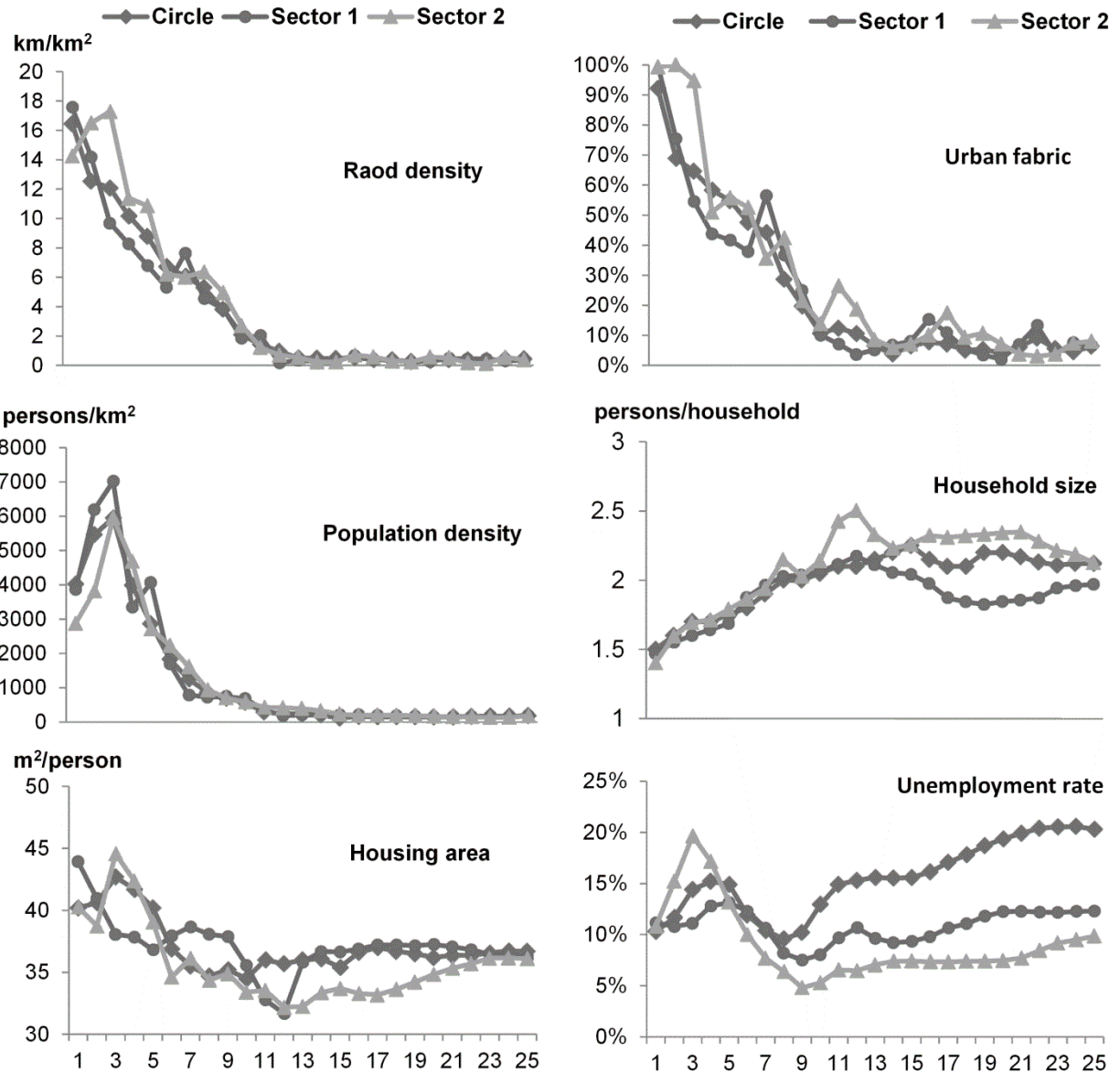


Fig. 7 Gradients of the six socioeconomic items in the circle and sector areas of the Leipzig urban-rural area. The X-axis is the distance in kilometers from the city center of Leipzig. The Y-axis is the average value of the socioeconomic indicators in the associated concentric ring or concentric sector ring of the circle or sectors. Road density and urban fabric are for 2006 and the other four indicators are for 2005 (after Kroll and Kabisch, 2012)

4.3. Ecosystem service and socioeconomic gradients in Kunming urban-rural area

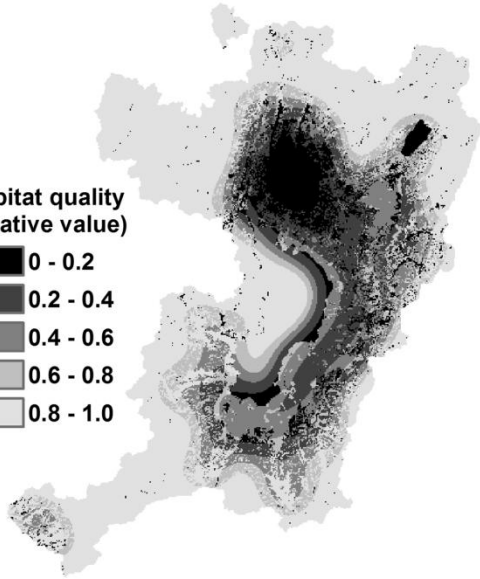
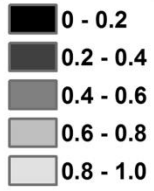
The spatial distributions of the three ecosystem services of the Kunming urban-rural area are showed in the left column of Fig. 8. Habitat quality appeared with an apparently increasing gradient from the city center to the northern and western directions. Along these two directions

the land covers changed from urban area to rural residential area and then quickly to forest and grassland (see Fig. 2). Along the eastern and southern directions the spatial changes of habitat quality were less regular than the other two directions. Having said this, three irregular positive gradients can be found. One was extending from the northern, western and southern lakeshores of the Dianchi Lake to the inner lake. Another one spread from the area, which was mainly covered by cropland and along the eastern part of the Dianchi Lake, to the lake area in the west and to the mountainous area in the east. The third one is located in the south of the Dianchi Lake and extends like branches in five directions. The NPP gradient was not as fine as habitat quality's, whereas apparent spatial changes, which were similar to that of habitat quality, can be identified. Concerning food productivity, cropland area had considerably higher values than the other areas. The southern part of the Dianchi Lake had moderate values higher than the northern part of the lake and most of the forest, urban and grassland areas. The five branch-shaped areas can be even more clearly seen. However, different from habitat quality and NPP, these five branches areas had higher values than the surrounding areas.

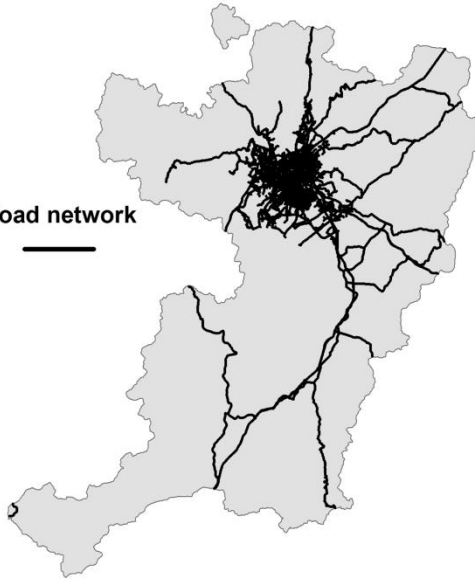
For the three investigated areas, the circle and the two sectors, quantitative changes of the three ecosystem services' gradients are demonstrated in the left column of Fig. 9. The top line chart shows that habitat quality virtually kept increasing along the urban-rural gradient of Kunming in all the three areas. Among them the growing line in the circle area was smoother than the two sectors'. NPP in the circle area monotonically rose from about 300 gC/m²/yr in the city center to nearly 900 gC/m²/yr in the concentric ring 15 kilometers away from the city center. In the two sectors, NPP roughly appeared growing trends. When looking into details, slightly fluctuations of NPP can be seen in sector 1, which stayed virtually stable in sector 2 from the 12th concentric sector ring and suddenly jumped in the last of the concentric sector rings. The food productivity gradients showed noticeable divergences in the three areas. In the circle area, food productivity appeared with a generally growing trend with trivial fluctuations. In sector 1 it remained below 100 × 10⁴ yuan/km² all along the gradient. In sector 2 neither an increasing nor a decreasing trend of this service can be found due to the significant value fluctuations.

The spatial distributions of the three socioeconomic dimensions are illustrated by the maps in the right column of Fig. 8. The gradients are further quantitatively revealed by the line charts in the right column of Fig. 9. All the three socioeconomic dimensions showed apparent high densities in the urban area and reductions in the areas spreading out from the city center. In addition, they extended to the southeast of the urban area and were distributed between the Dianchi Lake and the mountainous area in the east in moderate densities. Notably decreasing trends of the three socioeconomic indicators' values can be found along the urban-rural gradients in the circle and the 2 sector areas. Road density strictly monotonically kept reducing along the urban-rural gradient in all the three investigated areas. Urban fabric quickly dropped from 100% at the city center to nearly 0% at the end of the gradient. Moreover, population density showed identical gradients in the three areas decreasing from 20,000 persons/km² to nearly zero. The differences from urban fabric and population density to road density were the several slight fluctuations in sector 2.

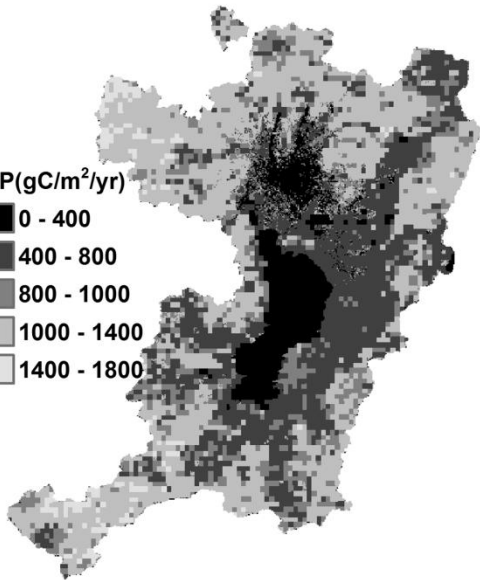
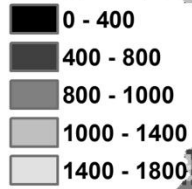
Habitat quality
(relative value)



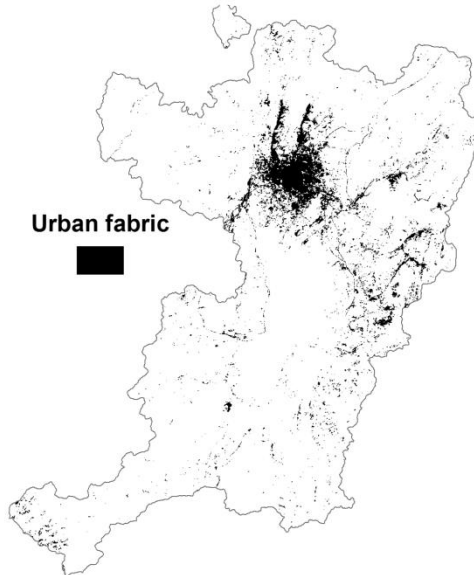
Road network



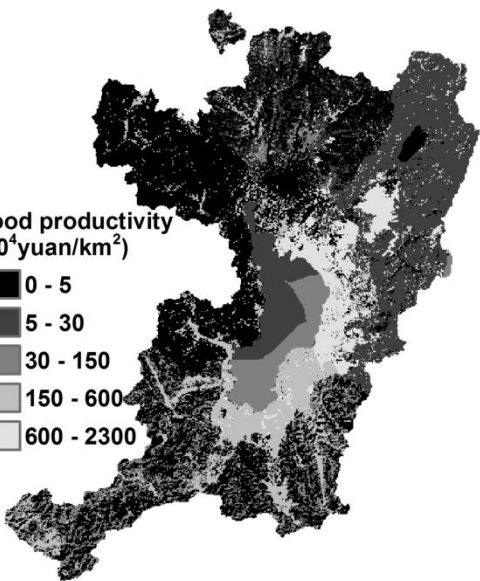
NPP(gC/m²/yr)



Urban fabric



Food productivity
(10⁴yuan/km²)



Population density
(10³persons/km²)

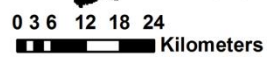
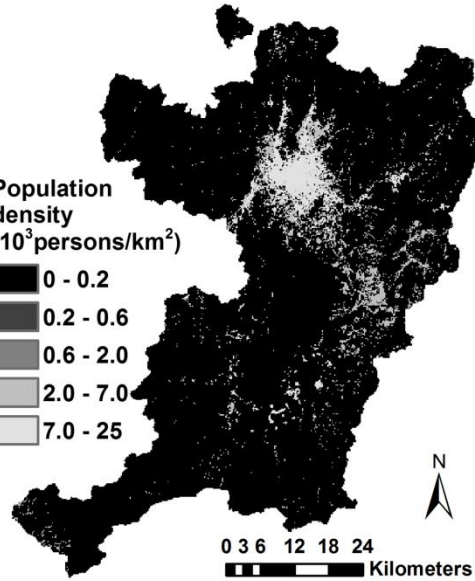
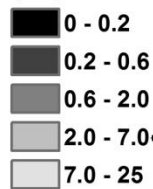


Fig. 8 Spatial distributions of the three ecosystem services (left column) and the three socioeconomic items (right column) in the Kunming urban-rural area. NPP data are for 2006, road network relates to 2007 and the other four item are for 2010

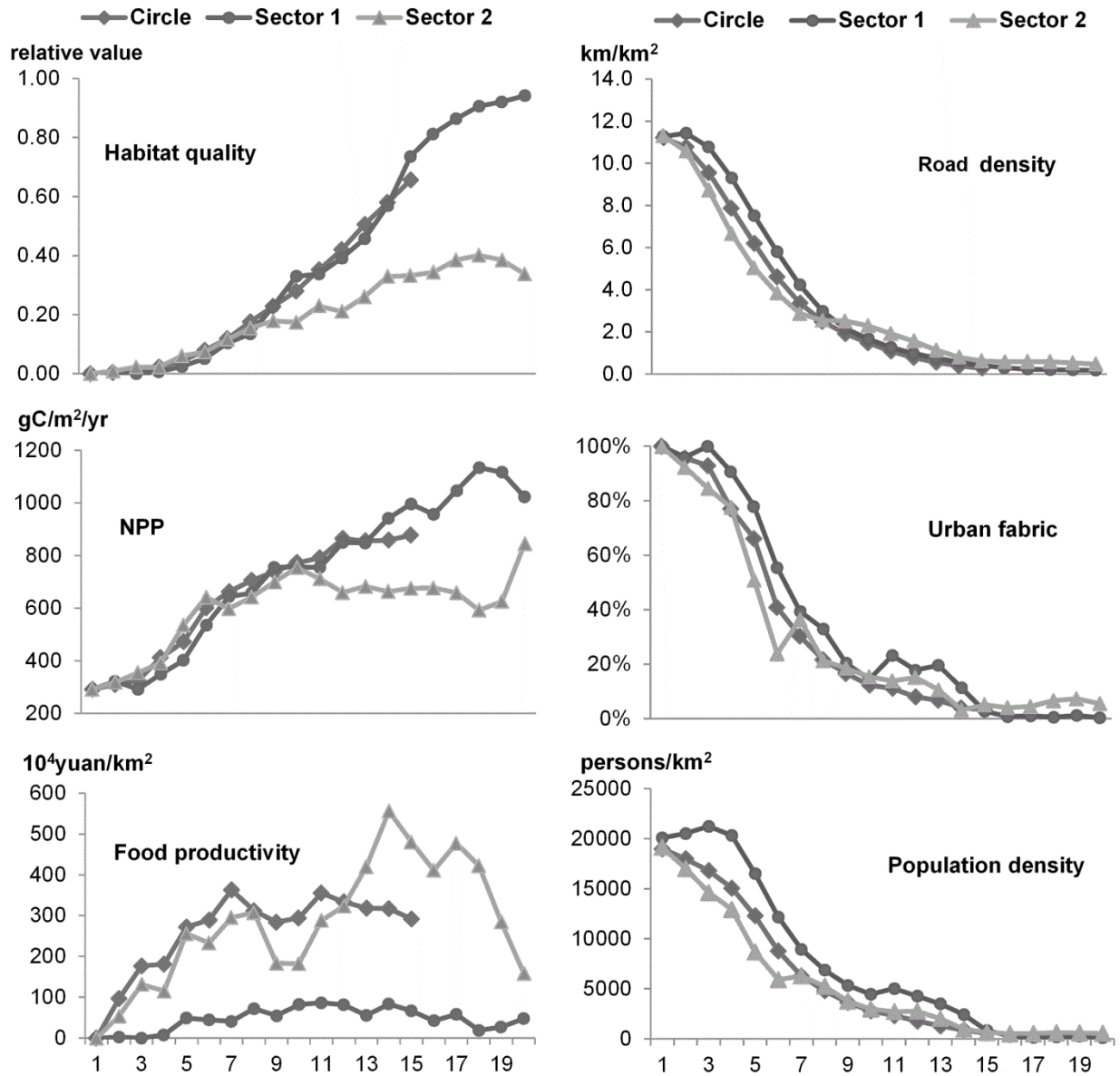


Fig. 9 Gradients of the three ecosystem services (left column) and the three socioeconomic dimensions (right column) in the circle and sector areas of the Kunming urban-rural area. The X-axis is the distance in kilometers from the city center of Kunming. The Y-axis is the average value of the indicators in the associated concentric ring or concentric sector ring of the circle or sectors

4.4. Gradient comparisons between Leipzig and Kunming

Notable similarities and differences emerge when ecosystem service gradients are compared between the urban-rural areas of Leipzig and Kunming. Concerning habitat quality, both research

areas enjoyed marked increases along the gradients in the circle and sector areas. The most significant difference was the much smoother changing trends of habitat quality in the three investigated areas of Kunming compared with that of Leipzig. Another comparison can be found between above ground carbon storage of Leipzig urban-rural area and NPP in the urban-rural gradient of Kunming. Although these two indicators both relate to ecosystem's production function based on photosynthesis and can in some degree represent climate regulation service, they demonstrated different gradient characteristics. As described in the preceding sections, above ground carbon storage roughly declined along the urban-rural gradient in the circle and sector 1 areas of Leipzig, despite of some fluctuations. Contrarily, NPP showed noticeably growing trends along the urban-rural gradients in the circle and sector 1 areas of Kunming. Food provisioning services of the two research areas showed similar variations along some gradient patterns and different spatial dynamics along others. In the circle area of Leipzig, food supply significantly appeared with a firstly increasing and then stable trend. This phenomenon was similar to that of food productivity in the circle area of Kunming. However, food productivity in sector 1 of Kunming remained stable at a low level along the gradient. This characteristic was considerably different from the food supplies along the two sector gradients of Leipzig, which can be characterized by marked fluctuations.

Concerning the socioeconomic gradients, the two research areas provided significant consistencies with the three identical indicators. Being apparently negative, the urban-rural gradients of road density and urban fabric of Kunming were considerably similar to those of Leipzig, respectively. These similarities occurred in all the three investigated areas. The most significant difference was that Kunming showed smoother trends of the indicator values than Leipzig. Population density also showed similarities between the two research areas by generally keeping dropping in the three analyzed areas respectively. The distinctions primarily existed in the first few concentric ring areas. In Leipzig, population density turned to drop quickly from the peak to zero from the 3rd concentric ring in all the three gradient patterns. Contrarily, in the three analyzed areas of Kunming, population densities did not show apparent turning points and the declining velocities were lower than that of Leipzig.

4.5. Linkages between ecosystem service and socioeconomic gradients

The comparisons between the ecosystem service and the socioeconomic gradients of Leipzig urban-rural area (Fig. 6 and Fig. 7) reveal mutual linkages. Some ecosystem service gradients showed similarities to that of socioeconomics; whereas, some others showed noticeable differences. Habitat quality in the circle and the two sector areas showed noticeably growing trends. Conversely, road density and urban fabric generally kept declining from the city center to outskirt. This opposition also applied to population density from the 3rd concentric ring. F-evapotranspiration gradient had similarities to that of household size in the three analyzed areas. All the six gradients demonstrated roughly increasing trends from the city center to the regions around 11 kilometers away. After that, f-evapotranspiration stayed stable till the end of the gradient in the three areas; Household size approximately kept steady in the circle and sector 2 areas and experienced a dropping and then rising process in sector 1. Another group of linkages can be found between food supply and four socioeconomic indicators, road density, urban fabric, population density as well as household size. Although the food supplies in the two sectors fluctuated considerably, the one in the circle area changed more regularly by keeping growing

from the fifth concentric ring till the fourteenth and then stayed steady. This trend was opposite to the gradients of the four socioeconomic dimensions in the circle areas.

Ecosystem service-socioeconomic linkages can also be identified along the urban-rural gradient of Kunming. The marked increases of habitat quality in the three analyzed areas were noticeably contrary to the declining trends of road density, urban fabric and population density in the three areas. The NPP gradient in the circle area also showed reversed correlations with the gradients of these three socioeconomic dimensions in the circle areas. The former one remained positive along the urban-rural gradient and latter three kept negative.

5. Discussion

5.1. Quantitative linkages between ecosystem services and socioeconomics

Leipzig

The linkages between ecosystem service and socioeconomic gradients delineated in the result section are quantitatively confirmed by Spearman correlation analyses (Table 2). In the Leipzig urban-rural area, the gradients of habitat quality, f-evapotranspiration, food supply and energy supply were significantly negatively correlated with the gradients of road density, urban fabric and population density in the circle and the two sector areas with $P < 0.01$. The sole exception was the weaker correlation between energy supply and road density in sector 1, which was still significant at the 0.05 level. These findings are logical as the share of artificial surfaces in the total area generally kept declining and the vegetation cover proportion (mainly including forest, grassland and agricultural areas) appeared with an increasing trend along the gradient (Fig. 3). In other words, it is the land cover variabilities in urban-rural areas that build the linkages between ecosystem services and socioeconomics. The close connections between ecosystem service supplies and land cover patterns in urban-rural areas have been evidenced by an extensive body of literature (Burkhard et al., 2012; Haase et al., 2012; Kroll et al., 2012; Larondelle and Haase, 2013). High habitat quality (or biodiversity) and evapotranspiration are usually due to intensive natural vegetation (Burkhard et al., 2009; Schwarz et al., 2011; Tallis et al., 2013). Food supply is undoubtedly attributed to agricultural land and energy, as estimated in this research, is mainly provided by peripheral areas of cities. Although not as much as ecosystem services, the spatial characteristics of socioeconomics have been investigated in the land cover context of urban-rural areas by some studies (Hope et al., 2003; Luck et al., 2009). Concerning the direct relationships between ecosystem services and socioeconomic elements, we found limited literature. Among these studies, one provided by Buyantuyev and Wu (2010) revealed the negative correlation between daytime surface temperature and median family income in the Phoenix metropolitan region. Another source from the same area additionally reported a positive relationship between mean temperature and population density (Jenerette et al., 2007). This finding can to some extent prove the correlation between f-evapotranspiration and population density in our research because evapotranspiration tends to go in an opposite direction of temperature in urban-rural gradients (Adler and Tanner, 2013).

The gradient of above ground carbon storage was significantly positively correlated with the gradients of road density, urban fabric and population density in the circle and sector 1 areas. However, in sector 2, the linkages were much looser. The reasons for these phenomena are obvious when looking into the above ground carbon storage gradients of the three areas (Fig. 6).

Shown by the gradient chart, the values generally remained decreasing along the gradient in the circle and sector 1 areas. Differently, in sector 2, the value showed notable fluctuations after the 14th concentric ring, which greatly disturbed the consistent trend with the three socioeconomic dimensions.

For the other three socioeconomic gradients, household size, housing area and unemployment rate, their correlations with the ecosystem service gradients were not as regular as road density, urban fabric or population density. For example, the correlations of household size with habitat quality, food supply and energy supply were not significant in sector 1. Furthermore, it was only negatively correlated with above ground carbon storage in the circle area. However, for its linkage with f-evapotranspiration, highly significant correlations were shown in all the three areas. Concerning housing area and unemployment rate, their correlations with the five ecosystem services showed even more divergences. For instance, housing area was significantly negatively correlated with habitat quality solely in sector 1 and positively correlated with above ground carbon storage solely in the circle area. Whereas, its correlation with energy supply showed significance in all the three gradient areas. Regarding unemployment rate, positive significant correlations were solely found with habitat quality, f-evapotranspiration, food supply and energy supply in the circle area. In contrast, it had no significant correlations with the five ecosystem services in sector 1. Even more unexpectedly, it was negatively correlated with food supply and energy supply in sector 2.

Looking into the possible reasons for the correlation irregularities described above, we assumed that the irregular spatial variations of household size, housing area and unemployment rate had done major contributions (see the irregular gradients of these three indicators in Fig. 7). These irregular spatial dynamics should originate from more complex socioeconomic determinants of these three elements compared to road density, urban fabric and population density.

Table 2 Spearman correlations between the ecosystem service and socioeconomic gradients of the circle and sector areas in the Leipzig urban-rural area. The sample values are the mean values of the indicators in the associated rings with all sample sizes of 25 (25 concentric rings in the circle area and 25 concentric sector rings in both of the two sectors). ** $P < 0.01$, * $P < 0.05$

		Habitat quality			Above ground carbon storage			f-evapotranspiration			Food supply			Energy supply		
		Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2
Road density	Circle	-0.927**			0.502*			-0.928**			-0.845**			-0.726**		
	Sector 1		-0.769**			0.425*			-0.640**			-0.744**			-0.472*	
	Sector 2			-0.876**			-0.244			-0.849**			-0.622**			-0.724**
Urban fabric	Circle	-0.879**			0.675**			-0.901**			-0.825**			-0.797**		
	Sector 1		-0.771**			0.454*			-0.698**			-0.651**			-0.548**	
	Sector 2			-0.940**			-0.205			-0.800**			-0.668**			-0.735**
Population density	Circle	-0.860**			0.513**			-0.919**			-0.897**			-0.616**		
	Sector 1		-0.878**			0.649**			-0.566**			-0.780**			-0.631**	
	Sector 2			-0.937**			-0.432*			-0.885**			-0.547**			-0.713**
Household size	Circle	0.791**			-0.631**			0.809**			0.792**			0.756**		
	Sector 1		0.225			-0.181			0.588**			0.253			0.300	
	Sector 2			0.567**			0.065			0.652**			0.634**			0.688**
Housing area	Circle	-0.197			0.574**			-0.242			-0.185			-0.528**		
	Sector 1		-0.563**			0.232			-0.555**			-0.426*			-0.421*	
	Sector 2			-0.259			0.282			-0.477*			-0.655**			-0.516**
Unemployment rate	Circle	0.903**			-0.351			0.800**			0.777**			0.552**		
	Sector 1		0.246			0.041			-0.327			.000			-0.157	
	Sector 2			-0.152			0.325			-0.320			-0.591**			-0.453*

Kunming

In the Kunming urban-rural area, the correlations between ecosystem service and socioeconomic gradients presented more consistency than in Leipzig (Table 3). Habitat quality and NPP both showed strongly negative correlations with road density, urban fabric and population density at the 0.01 significance level in the circle and the two sector areas. These findings are in agreement with the gradient characters of the five indicators. In other words, the three socioeconomic dimensions presented apparent decreasing trends and the two ecosystem services showed opposing trends along the gradient in all the three areas (Fig. 9). The determinants of these phenomena should be the spatial dynamic characteristics of land covers as the values of all these five indicators are strongly associated with the land cover status (Burkhard et al., 2009; Hawbaker et al., 2005; Helmut Haberl, 2001; Kroll and Kabisch, 2012). For the spatial variations of land covers, the gradients showed noticeable declines of urban area shares and general increases of forest and grassland shares in all the three areas (Fig. 4).

Unexpected finding refers to the linkages of food productivity gradients with those of road density, urban fabric and population density in sector 1. Unlike in the circle and sector 2 areas, where food productivity gradients were significantly negatively correlated with the three socioeconomic gradients, sector 1 did not exhibit close correlations between this service and the three socioeconomic dimensions. This exception can be easily explained by the ignorable proportion of cropland along the gradient of sector 1 (Fig. 4)

Table 3 Spearman correlations between the ecosystem services and socioeconomic gradients of the circle and sector areas in Kunming urban-rural area. The sample values are the mean values of the indicators in the associated rings. The sample sizes are 15 for the circle area and 20 for the sectors (15 concentric rings in the circle area and 20 concentric sector rings in both of the two sectors). ** $P < 0.01$, * $P < 0.05$

		Habitat quality			NPP			Food productivity		
		Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2	Circle	Sector 1	Sector 2
Road density	Circle	-1.000**			-0.989**			-0.736**		
	Sector 1		-0.995**			-0.977**			-0.375	
	Sector 2			-0.971**			-0.629**			-0.642**
Urban fabric	Circle	-1.000**			-0.989**			-0.736**		
	Sector 1		-0.976**			-0.973**			-0.384	
	Sector 2			-0.928**			-0.657**			-0.789**
Population density	Circle	-1.000**			-0.989**			-0.736**		
	Sector 1		-0.982**			-0.970**			-0.398	
	Sector 2			-0.972**			-0.642**			-0.687**

5.2. Comparisons between gradient patterns and case areas

As described in the results section, some ecosystem services showed similar spatial variations in the three gradient types (the circle and the two sectors areas) in the Leipzig and the Kunming urban-rural areas. Others, such as food supply, above ground carbon storage and energy supply of Leipzig and food productivity of Kunming presented different gradient patterns (Fig. 6 and 9). These variation divergences are understandable when looking into the land cover characters of the different gradient patterns. For example, forest and semi-natural areas intensively emerged in the southeast of sector 2 area of Leipzig (Fig. 1), causing the sharp increase and the two peaks of above ground carbon storage after the 15th concentric sector ring (Fig. 6). Another example relates to the spatial changes of food productivity of the Kunming urban-rural area, which showed noticeable divergences in the three gradient areas (Fig. 9). The causes of this phenomenon also originated from the differences of the land cover spatial patterns that cropland appeared mainly in the southeast of the circle area and in the last ten concentric rings of sector 2 (Fig. 2). The differences of the gradient patterns of ecosystem services in different areas imply that although the concentric ring gradient pattern can comprehensively capture the spatial variation information along the urban-rural gradient, it can be unsuitable in some cases. For instance, it is problematic when the land cover structure does not exhibit the same pattern in all directions from the city center as shown by our two case areas. In the regions with unique land cover patterns (e.g. sector 1 and 2 of Leipzig and sector 2 of Kunming in our study), refined gradient patterns that can more accurately capture details of the whole research area will be more preferable. These gradient patterns include the concentric sector ring gradient proposed in this study and the transect gradient used in some other research (Banzhaf et al., 2009; Hahs and McDonnell, 2006; Luck and Wu, 2002). Having said this, as the concentric sector ring and the transect gradients rely on directions, they can solely treat the chosen directions and lose information of the regions in other directions. Considering the advantages and disadvantages of different gradient patterns, a combination of them to both comprehensively and detailedly investigate ecosystem service gradients can be a promising solution.

Compared to ecosystem services, the investigated socioeconomic dimensions presented much more consistency between the three gradient areas of both Leipzig and Kunming. The major disagreements referred to household size, housing area and unemployment rate of Leipzig (Fig. 7). As discussed earlier, these three aspects are affected by more complex socioeconomic reasons compared to road density, urban fabric and population density, which are primarily based on urbanization levels (Hahs and McDonnell, 2006). Accordingly, we assume that the concentric ring gradient is appropriate to reveal the spatial dynamics of road density, urban fabric and population density in urban-rural gradients. Whereas to treat more complex socioeconomic dimensions, such as household size, housing area and unemployment rate, gradient patterns that can capture details (e.g. concentric sector ring and transect) are additionally required.

Comparing different case areas (the two research areas in this study and the ones from other research), we found limited ecosystem services presenting similar gradient patterns in urban-rural area. Different regions tend to exhibit divergent spatial variation characters of ecosystem services from the city center to outskirts. For example, the gradients of food provision showed considerable distinctions between the Leipzig and Kunming urban-rural areas in this study, especially in the sector areas. For f-evapotranspiration and above ground carbon storage, our findings in Leipzig are inconsistent with those from Larondelle and Haase (2013) in several other

European cities. Their results also showed divergent ecosystem service gradient patterns in urban-rural areas of different regions. According, we can induce that although ecosystem service gradients in certain patterns can be found in urban-rural areas, typical gradient patterns suiting different regions are scarcely existing. The possible reason, as we mentioned before, are the distinct land cover structures of different urban-rural regions.

Regarding socioeconomic dimensions, identical urban-rural gradients have been found between the two case areas in our study. Additionally, similar gradient patterns are identified in other studies. For example, Leipzig and Kunming have identical urban fabric gradient patterns (Fig. 7 and 9), which are similar to the ones of Manchester, Berlin, Hamburg as well as Munich delineated by Kroll and Müller (2011) and Kroll and Kabisch (2012). Concerning population density, consistency of urban-rural gradient patterns exist between Leipzig (this study) and Berlin, Hamburg and Munich (Kroll and Müller, 2011) and between Kunming (this study) and Warsaw (Kroll and Müller, 2011). Concerning the three more intricate socioeconomic dimensions, household size, housing area and unemployment rate, urban-rural gradient patterns primarily present disagreements between different regions (e.g. household size between Leipzig in this study and Hamburg and Munich in the research of Kroll and Kabisch (2012); housing area between Leipzig and Manchester and Hamburg presented by Kroll and Müller (2011)). To sum up, we can tentatively assume the existence of typical gradient patterns of some socioeconomic dimensions (e.g. urban fabric discussed above) in different urban-rural areas.

5.3. Uncertainty analysis

An ecosystem service assessment is always accompanied by uncertainties originating from the complexity of the ecological system, respondents' preferences and technical problems (Hou et al., 2013; Elmqvist et al., 2010; Scolozzi et al., 2011). In this research, the creation of gradients relied on averaging the values of heterogeneous ecosystem services and socioeconomic dimensions in each ring area, leading to uncertainties with the produced gradient patterns. These uncertainties may cause bias in assessment (Li and Wu, 2006), such as unreliable characterization of gradients and doubtful revelations of relationships between ecosystem services and socioeconomics. Therefore, it is imperative to obtain the information of uncertainties with the results. Considering the large number of ecosystem services and socioeconomic dimensions investigated in this study, we solely chose four of them as examples to quantify the uncertainties of gradients in the three gradient areas (circle and two sectors). The analyzed indicators are above ground carbon storage and housing area of Leipzig and habitat quality and population density of Kunming. Referring to the mainstreaming methods showing uncertainties with value averaging in ecosystem service studies (Larondelle and Haase, 2013; Yanai et al., 2010; Zhu and Carreiro, 2004), we used standard deviation (SD) to show the value ranges of the four indicators in each ring of the circle area and the two sectors, respectively. Accordingly, uncertainty gradients of the four examined indicators were created (Fig. 10).

In general, the established uncertainty gradients did not exhibit typical patterns. The magnitudes and changes of uncertainties were not depended on the magnitudes of the mean values or the distance from the city center. Taking the above ground carbon storage of Leipzig as an example, in sector 1 the greatest uncertainty (greatest SD value) appeared at the 12th ring, where the mean value was not the highest among all the rings (Fig. 10, b). In sector 2, the uncertainties remained at low levels before the 15th ring and soared between the distance of 15km and 24km from the city center (Fig. 10, c). Looking into the map of ecosystem services spatial distribution and the

land cover map (Fig. 1 and 5), we can attribute the considerable uncertainties in these rings to the divergences of the above carbon storage values and the heterogeneous land covers in these areas. For the other three indicators, we can also conveniently observe the uncertainty gradient patterns, compare the uncertainty magnitudes between different rings and find out the areas with most or least uncertainties. After testing this uncertainty gradient method, we assume that this method is appropriate to quantitatively capture the uncertainties in exploring ecosystem services and socioeconomic gradients in urban-rural areas.

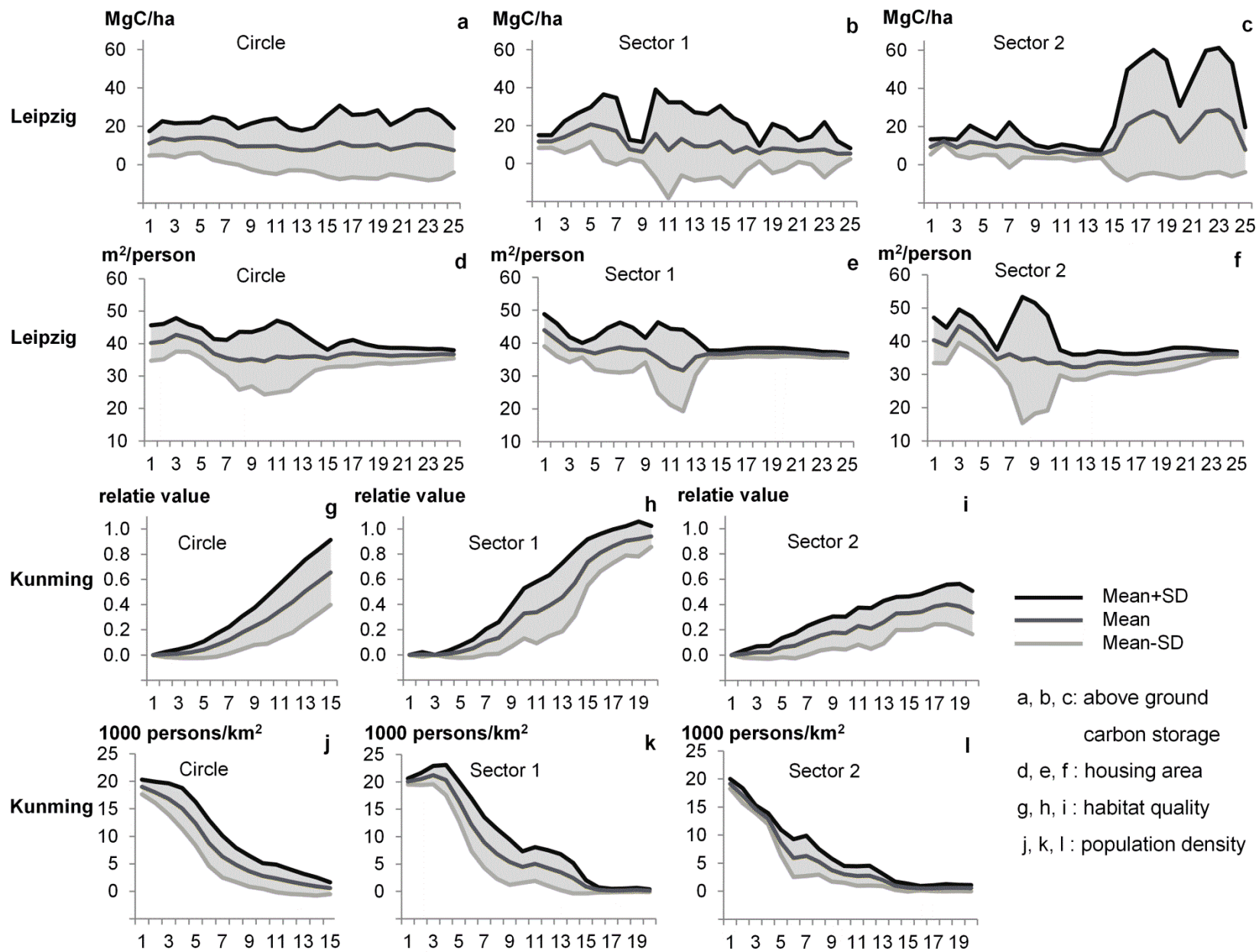


Fig. 10 Quantified uncertainties of ecosystem services and socioeconomic gradient calculations for the Leipzig and Kunming urban-rural areas. Above ground carbon storage and housing area were investigated for Leipzig. Habitat quality and population density were analyzed for Kunming. Uncertainties were quantified by the ranges between the values of mean plus standard deviation (SD), mean values and the values of mean minus SD

6. Conclusions

In this article we draw on the urban-rural gradient principle to offer a methodology to investigate the linkages between ecosystem services and socioeconomics in urban-rural areas. By performing empirical studies with the developed methods in two case urban-rural areas respectively in Germany and China, we derived interesting findings. With these results, we can now answer the three research questions asked at the beginning of this article as follows:

1) The investigated ecosystem services and socioeconomic dimensions of the two study areas presented regularities of spatial patterns in urban-rural gradients in different degrees. Some of them showed either generally increasing or decreasing trends along urban-rural gradients, others exhibited considerable fluctuations.

2) Ecosystem service gradient characters of the circle and the two sector gradient areas presented more divergences than that of socioeconomic gradients in both the two case areas. The concentric ring gradient pattern can more comprehensively capture the overall spatial dynamics of ecosystem services and socioeconomics along the gradients. In contrary, the concentric sector ring gradient pattern can better reveal the details of especially concerned areas in specific directions from the city center. For the two study areas, urban-rural gradients of identical ecosystem services did not necessarily present consistent patterns and gradients of the same socioeconomic dimensions showed similar patterns.

3) Strong linkages between some ecosystem service and socioeconomic gradients were observed and statistically proven in both the two case areas. The socioeconomic dimensions more based on land covers were found to have more close correlations with ecosystem services. Moreover, the correlations of a certain pair of ecosystem service and socioeconomic gradients could be inconsistent in different gradient areas of either case area.

In addition, the results of our study suggest another two important messages. By integrating our findings and the published work, we further proved that a typical model of ecosystem services gradient does not exist for different cities. The other message tells the necessity to be aware of uncertainties and effectively incorporate uncertainty analysis into the investigations of ecosystem service and socioeconomic gradients.

Novel innovations in the field of ecosystem services are emerging regularly. In this exiting time we call for future research to determine whether our arguments for the interrelationships between ecosystem services and socioeconomics in urban-rural areas are supported by more empirical evidence. We hope this paper will help draw more research interests to elevate our understanding of human-environmental interactions in urban-rural systems.

Appendices

Table 4 Date uses, time, scales and sources for the study of Leipzig urban-rural area

Data	Time	Scale	Publisher/source
Population number	2005	Municipality, city district	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Administrative borders of municipalities and city district	2001	Municipality, city district	German Federal Statistical Office (GENESIS)
Corine Land Cover	2006	1:100,000	German Federal Environmental Agency (2014)
Soil map of Saxony including usable field capacity and actual root depth	2007	1:200,000	Saxon State Department of the Environment and Geology
Soil survey map Germany No. CC 4734 Leipzig	2009	1:200,000	German Agency for Geology and Natural Resources Hanover
Soil fertility map of Saxony	2007	1:200,000	Saxon State Department of the Environment and Geology
Soil fertility map of Saxony-Anhalt	2008	1:200,000	Environmental Agency of Saxony-Anhalt
Evapotranspiration values linked to land cover	2011	Corine Land Cover classes	Larondelle and Haase (2013); Schwarz et al. (2011)

Above ground carbon storage values linked to land cover	2013	Corine Land Cover classes	Gibbs (2006); Larondelle and Haase (2013); Strohbach and Haase (2012)
(Continued)			
Total area of bioenergy crops (differentiated after crop types)	2007	Federal State	Fachagentur für Nach wachsende Rohstoffe e.V. (2014); Department of agriculture, forestry and horticulture Saxony-Anhalt; Ministry of Agriculture and Environment Saxony-Anhalt (2007); Saxon State Ministry of the Environment and Agriculture (2007)
Energy yield per ha crop type	2007	No scale	Institute for Energy and Environment (2007)
Wind, water and solar energy plants with installed power	2007	Exact locations	Vattenfall Europe AG (2009), URL: http://www.vattenfall.com/en/index.htm (accessed on 2.11.2014)
Lignite extraction	2007	Lignite extraction site	Saxon Upper Mining Authority (2007)
Crop yield	2007	District	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Crop composition	2007	District	Statistical Office of the Free State of Saxony; Statistical Office of Saxony-Anhalt
Fruit yield, livestock, game	2007	Federal State	Saxon State Ministry of the Environment and Agriculture; KTBL (2005)
Road network	2006	Germany, city district	Federal Agency for Cartography and Geodesy (2014); Geospatial Information and Surveying, Saxony (2014)
Railway network	2008	Leipzig city district	Geospatial Information and Surveying, Saxony (2014)
Urban fabric	2006	1:100,000	Corine land cover from German Federal Environmental Agency (2014)

Household size	2005	Municipality, district	city	Statistical Office Leipzig, Statistical Office of the Free State of Saxony,
Housing area	2005	Municipality, district	city	Statistical Office Leipzig; Statistical Office of the Free State of Saxony
Unemployment	2005	Municipality, district	city	Statistical Office Leipzig; Statistical Office of the Free State of Saxony

Table 5 Date uses, time, scales and sources for the study of Kunming urban-rural area

Data	Time	Scale	Publisher/source
Administrative borders of districts and counties	2000	1:4,000,000, China	Data Sharing Infrastructure of Earth System Science, URL: http://www.geodata.cn/Portal/?isCookieChecked=true (accessed on 2.11.2014)
Land cover map	2010	1:100,000, Kunming prefecture-level city	Produced by the interpretation of 30-meter Landsat thematic mapper (TM) data in combination of a 1:100,000 topographic map of the research area
Net primary production raster map for vegetation covers	2006	1 km × 1 km, global	Numerical Terradynamic Simulation Group (2013)
Net primary production of urban area	2006	Shanghai city district	Lin et al. (2008)
Net primary production of the Dianchi Lake	2009-2010	the Dianchi Lake	Sun et al. (2011); Chinnaiah and Madhu (2010)

Farming, animal husbandry and fishery yields	2010	Districts and counties, Kunming	Kunming Statistical Yearbook 2011, Statistics Bureau of Kunming
Road network	2007	China	National Administration of Surveying, Mapping and Geoinformation of China
Urban fabric	2010	1:100,000, Kunming prefecture-level city	Kunming Land and Resource Bureau
Population density	2010	Districts and counties, Kunming	Kunming Statistical Yearbook 2011, Statistics Bureau of Kunming

Acknowledgements

This work was financially supported by the EU Integrated Project PLUREL (contract No. 036921) and the project “Regulation of Integrative Land Use in Urban-Rural Areas With Ecological Principles in Lake Catchments of the Yungui Plateau” funded by the Department of Land and Resources of Yunnan Province, China. We are grateful for the time and efforts from the anonymous reviewers on improving this paper.

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Chapter 4. Ecosystem services of human-dominated watersheds and the influences from socioeconomics: A case study of the Dianchi Lake Watershed

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Submitted to Agriculture, Ecosystem & Environment

Abstract

Watersheds provide multiple ecosystem services. The spatial characters of ecosystem services are closely related with socioeconomic status if the watersheds are dominated by humans. In this study, we address the spatial characteristics of ecosystem services and the respective socioeconomic influences in a heavily human-disturbed watershed in Southwest China. We firstly quantify and map five ecosystem services of nine river basins of the Dianchi Lake Watershed. The quantification is based on the Integrated Valuation of Environmental Services and Tradeoffs (InVEST) model and biophysical as well as socioeconomic data. Thereafter, we conduct a confirmatory research by using a hypothesis-test methodology to investigate the socioeconomic causes of the spatial changes of the five ecosystem services. On the basis of the modeling results of nitrogen retention and water yield, we exemplify the distinctions between ecosystem services potential, flow and demand and perform a sensitivity analysis to test the influences of input data and parameter uncertainties on the modeling results. The hypothesis-test analysis reveals only a small number of socioeconomic influential factors, most of which are related to land use structure. We attribute the large number of invalid tests of the effects of demographical, agricultural production input and economic factors to the uncertain values of the respective indicators downscaled from larger districts or counties. The hypothesis-test methodology provided in this study is applicable in the investigation of socioeconomic influences on ecosystem services in the situation of socioeconomic data uncertainty and scarcity.

Keywords: human-environmental interactions, spatial variations, InVEST, hypothesis, ecosystem service potential, ecosystem service flow, sensitivity analysis

1. Introduction

By sustaining human activities, watersheds are usually under the interplay of natural and human systems. The human-environmental interaction issue at watershed scales has attracted attentions from both environmental scientists and managers. A promising approach to treat this issue is ecosystem service assessment, which has obtained advocacy not only from scientific community, but also from regional and national policy makers (Bateman et al., 2013; MA, 2005; Sanchez-Azofeifa et al., 2007; TEEB, 2010; Vohland et al., 2011). Ecosystem service assessment at watershed scales has witnessed a high popularity among ecosystem service assessors in recent years (Bagstad et al., 2012; Leh et al., 2013; Nelson et al., 2009). Water-related services such as water yield, water purification and water regulation are the critical services generated by a watershed due to its geographical, hydrological and ecological characters. These services have drawn most interests from researchers (Bai et al., 2013a; Bräuer and Marggraf, 2005; Jujnovsky et al., 2012; Leh et al., 2013; Notter et al., 2012). Studies of other ecosystem services produced by watersheds relate to carbon storage and sequestration, biodiversity, soil conservation, pollination, commodity production, storm peak management, erosion control and some culture services (Bagstad et al., 2012; Bai et al., 2011; Leh et al., 2013; Nelson et al., 2009; Reyers et al., 2009; Wang et al., 2009)

The assessments of watershed ecosystem services are achieved by different methods. One mainstreaming method is modeling by InVEST (Integrated Valuation of Environmental Services and Tradeoffs) (Bagstad et al., 2012; Bai et al., 2011; Leh et al., 2013; Nelson et al., 2009), a software aggregated by different ecosystem services models (Tallis and Polasky, 2009). Another modeling tool that can estimate and visualize multiple ecosystem services is ARIES (Artificial Intelligence for Ecosystem Services). Its applicability has been examined e.g., by modeling the ecosystem service supply of San Pedro river watershed, Arizona (Bagstad et al., 2012). Apart from these two tools, some other modeling methods have been newly reported (Johnston et al., 2011; Jujnovsky et al., 2012; Notter et al., 2012; Watanabe and Ortega, 2014). However, targeting at a small number of ecosystem service types, these tools are not as versatile as InVEST or ARIES. Besides modeling, other assessment methods involve monetary valuation based on biophysical measure (Bräuer, 2005), public survey and multi-criteria decision analysis for ecosystem services ranking (Liu et al., 2013) and the decision tree approach (Wang et al., 2009). Based on the assessments of ecosystem services provided by watersheds, some scientists further investigated ecosystem service variations of different land use scenarios (Bai et al., 2013a; Nelson et al., 2009; Watanabe and Ortega, 2014), ecosystem service trade-offs (Liu et al., 2013; Swallow et al., 2009), ecosystem services supply and demand (Jujnovsky et al., 2012) and the uncertainties in ecosystem service assessments (Leh et al., 2013; Notter et al., 2012). These attempts may aid the decision making processes to manage the interacting human-environmental systems of watersheds.

In human-disturbed watersheds, ecosystem services are interrelated with the socioeconomic status of the respective human systems and influenced by human activities in different degrees. As reported by an extensive body of literature, the essential influences origin from humans' land use activities (Bai et al., 2013a; Leh et al., 2013; Watanabe and Ortega, 2014). Conversions of forests or grassland to cropland or artificial areas were proven to greatly reduce the water quality of runoff flowing downstream (Bai et al., 2013b; Martinez et al., 2009; Zhang et al., 2011). However, forests were reported to retain a smaller amount of nutrients than agricultural land use by some research (Bai et al., 2013b; Leh et al., 2013). Water yield was found to increase in the areas where forests were replaced by agroforests, cropland or grassland (Bent, 2001; Leh et al., 2013; Watanabe and Ortega, 2014). In addition

to water-related services, some other ecosystem services were confirmed to be noticeably affected by land use changes. The change of forest to cropland apparently increases agricultural production (Bai et al., 2013a, b). In contrast, such land use change greatly undermines regulating services such as carbon storage, biodiversity, sediment retention and erosion control (Leh et al., 2013; Martinez et al., 2009; Watanabe and Ortega, 2014; Zhang et al., 2008). Concerning total ecosystem services supply, forest is regarded as the greatest contributor followed by grassland, cropland and urban area (Cai et al., 2013; Martinez et al., 2009; Pan et al., 2012; Wang et al., 2009). Besides land use changes, ecosystem service variations at watershed scales result from some other regional socioeconomic dimensions. Some research reported the negative influences of the growth of economy and population on the total value of ecosystem services (Cai et al., 2013; Huang et al., 2010; Li et al., 2010; Wu et al., 2013; Zhang et al., 2008). The change of agricultural production patterns is also evidenced to have impact on ecosystem services. Reyers et al. (2009) reported the declines in ecosystem services caused by the shift from a system of transhumance pastoralism and subsistence farming to ostrich farming and extensive sheep farming. Another critical socioeconomic driver in watershed regions is hydropower development, which was proven to be a major negative impact on biodiversity and water quality (Wang et al., 2010). Although human activities are usually found to be detrimental to ecosystem services supply in watersheds, some endeavors, such as ecological restorations and optimizations of agricultural production patterns, can have positive effects (Bräuer and Marggraf, 2005; Fennessy and Craft, 2011; Tanaka et al., 2013; Zhang et al., 2008).

Through literature investigation, we found an influential amount of research assessing ecosystem services at watersheds scale. Additionally, some studies have explored the effects from human activities and socioeconomics on ecosystem services. However, these explorations mostly looked at a limited number of human dimensions and comprehensive investigations of humans' impacts on ecosystem services at watershed scales are still lacking.

In order to address this research gap, we evaluated the ecosystem services supplied by the Dianchi Lake Watershed, which is dominated by humans in the Southwest of China. Additionally, we comprehensively investigated the influences on ecosystem services from human systems mainly referring to land uses and socioeconomics. As the estimations of some ecosystem services were achieved by modeling in our assessment, we further conducted an uncertainty analysis to some modeled results. Specifically, the objectives of this research were to:

1. Assess the typical provisioning and regulating ecosystem services of the Dianchi Lake Watershed
2. Investigate the impacts of the human system on ecosystem services at the Dianchi Lake Watershed scale
3. Analyze the uncertainties emerging in ecosystem services modeling.

2. Research area and methods

2.1. Research area

The Dianchi Lake Watershed is situated in the Kunming region on the Yunan Guizhou Plateau in Southwestern China. It is located in the drainage divide area of the Yangtze River, the Red River and the Pearl River (24°29' N ~ 25°28' N , 102°29' E ~ 103°01' E, Figure 1) with an average elevation of 1900m and an total area of 2920km². The Dianchi Lake Watershed has a subtropical, mild highland climate throughout the whole year with humid

summers and dry winters. The mean annual precipitation is 1000 mm and the mean annual temperature is 15 °C in this area (Statistics Bureau of Kunming, 2011). This area relates to seven districts or counties of the Kunming region, Wuhua, Panlong, Guandu, Xishan, Songming, Jinning and Chenggong (Gao et al., 2013).

The Dianchi Lake Watershed has been dominated and significantly affected by the human system with rapid population and economy growth and by a noticeable sprawl of urban areas. The population of this area had increased from around 1 million in 1960s to 3.44 million in 2009. The population density of the whole watershed was 1178 persons/km² in 2009, which was primarily contributed by Kunming city to the north of Dianchi Lake. With an annual average increasing rate of 18%, the Gross Domestic Product (GDP) of the Dianchi Lake Watershed had grown to 143 billion RMB in 2009, which accounted for 78% of the entire Kunming region. The built-up area of Kunming city has been rapidly expanding since 1988, resulting in the urbanization rate of the whole watershed of 72% in 2009 (Li et al., 2012; Pan and Gao, 2010; Statistics Bureau of Kunming, 2011). The Dianchi Lake Watershed has been facing worsening environmental problems, especially water quality degradation and the eutrophication of the Dianchi Lake, owing to the rapid economic growth and urbanization and the intensifying human activities (Li et al., 2012).

In this study we concentrate on nine major river basins in the Dianchi Lake Watershed with eight upstream of the Dianchi Lake and one downstream. The eight upstream river basins are located in the north, east and south of the Dianchi Lake and the downstream river basin is situated in the west of the lake. These river basins are named according to the respective main stem rivers (Figure 1). The nine investigated river basins have been extracted from the DEM map of the entire Kunming region. Land use distributions and structures show considerable divergences among the river basins (Figure 1 and Table 1). Urban areas are mainly distributed in the Xinyunliang River Basin, the Panlong River Basin, the Laobaoxiang River Basin and the Laoyu River Basin (river basins 1, 2, 3 and 5 in Figure 1). Forest and grassland are mainly located in the Laobaoxiang River Basin, the Laoyu River Basin, the Da River Basin, the Dongda River Basin and the Tanglang River Basin (3, 5, 6, 8 and 9 in Figure 1). Concerning cropland, five river basins which the Laobaoxiang River, the Maliao River, the Da River, the Chai River and the Dongda River run through, respectively, share the primary part (3, 4, 6, 7 and 8 in Figure 1).

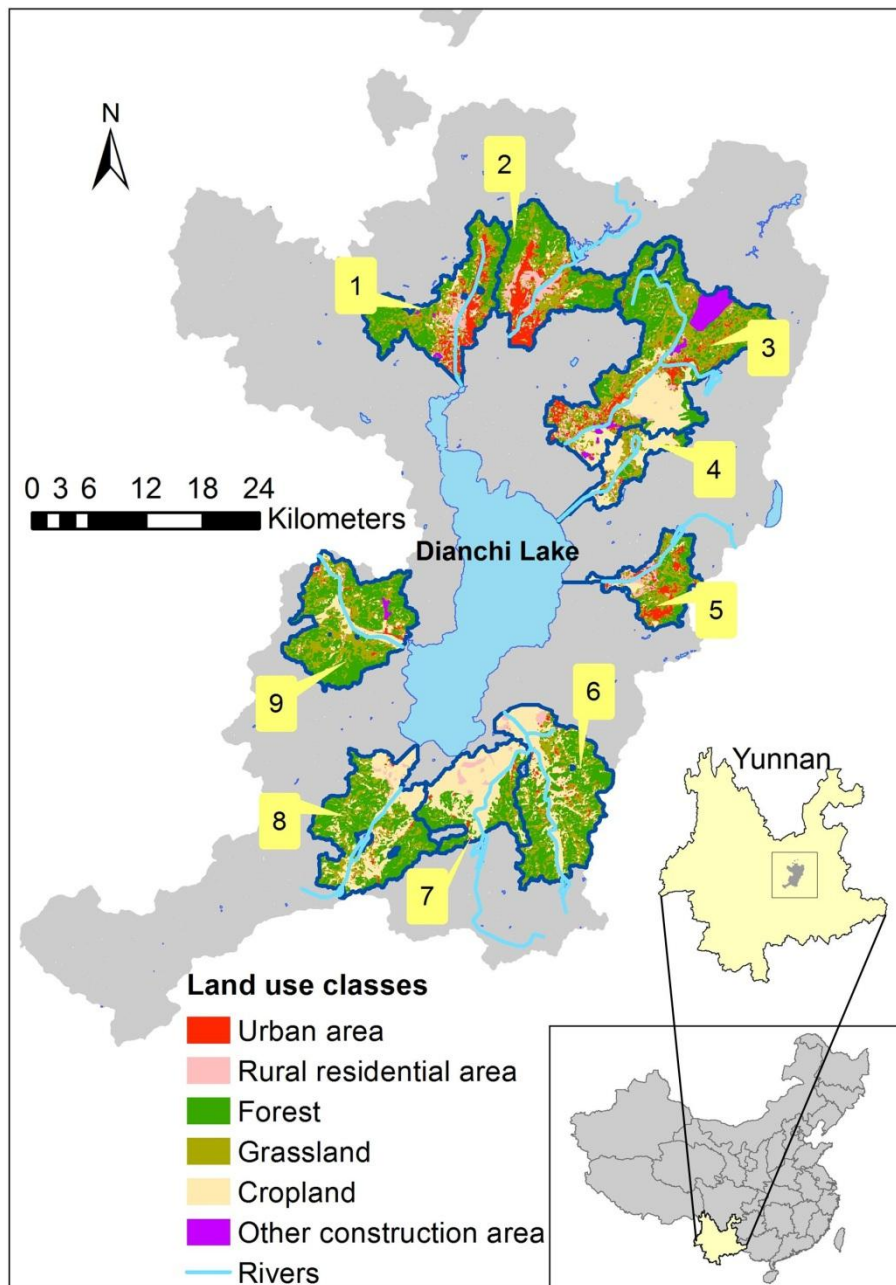


Figure1. Land uses of the nine investigated river basins of the Dianchi Lake Watershed in 2010. The shadow area covers the six district/counties of the Kunming region where the nine river basins are located. Other construction area mainly relates to the airport and its periphery in the northeast of the research area. The river basins are named according to the names of rivers flowing through: 1. Xinyunliang River Basin, 2. Panlong River Basin, 3. Laobaoxiang River Basin, 4. Maliao River Basin, 5. Laoyu River Basin, 6. Da River Basin, 7. Chai River Basin, 8. Dongda River Basin, 9. Tanglang River Basin. River basins 1–8 are upstream of the Dianchi Lake and the Tanglang River Basin is downstream. River basins 2, 3 and 6 are not complete defined by the DEM data because the respective river segments are artificially channeled.

Table 1. Proportions of different land use classes of the nine investigated river basins of the Dianchi Lake Watershed

Proportions of different land use classes	1 Xinyun River Basin	2 Panlong River Basin	3 Laoba- oxiang River Basin	4 Maliao River Basin	5 Laoyu River Basin	6 Da River Basin	7 Chai River Basin	8 Dongda River Basin	9 Tangla- ng River Basin
urban area	18.4%	19.6%	7.4%	3.7%	16.9%	2.3%	1.6%	1.4%	2.1%
rural residential area	8.5%	11.7%	3.5%	6.7%	12.0%	2.1%	4.0%	1.7%	1.1%
forest	37.8%	37.6%	31.8%	12.6%	28.8%	41.9%	31.0%	43.3%	49.3%
grassland	28.0%	26.4%	29.6%	26.4%	32.2%	22.7%	7.5%	20.7%	32.9%
cropland	6.4%	4.6%	20.9%	50.3%	10.1%	31.1%	56.0%	33.0%	13.7%
other construction	0.9%	0.0%	6.7%	0.3%	0.0%	0.0%	0.0%	0.0%	0.9%

2.2. Data sources

In this research we used biophysical data (land use, climate, soil, elevation, etc.) and socioeconomic statistical and spatial data (see Table A-2 in the appendix for the data, time span, scales and sources). The data were derived from respective regional or national governmental agencies though online or offline accessibilities, online open access data sources of scientific institutions and published work.

2.3. Methods

2.3.1. Ecosystem services quantification and mapping

In this study we investigated two provisioning services, food supply and water yield and quantified three regulating services, net primary production (NPP), habitat quality and nitrogen retention. According to the mainstreaming classification schemes of ecosystem services (Burkhard et al., 2009; de Groot et al., 2002; TEEB, 2010), NPP and habitat quality (a proxy of biodiversity, Tallis et al., 2013a) are ecosystem functions rather than services. We employed these two ecosystem functions in our study to indicate regulating services as they are critical for ecosystem integrity and essential for generating ecosystem services, especially regulating services (Burkhard et al., 2009; Kandziora et al., 2013). The chosen ecosystem services are highly sensitive to land use changes, so their spatial dynamics in the considerably varying environment of the Dianchi Lake Watershed can be effectively captured. The values of the services were for 2010 and derived by robust methods from published work, or based on empirical or statistical data (see Table 2 for the detailed information on the indicators). The details of the quantification methods for each ecosystem services are provided by the paragraphs below.

Table 2. Overview of the investigated ecosystem services and the respective quantification methods. The units in the form of per pixel of water yield and nitrogen retention are related to the ecosystem services maps. The per ha-formed units of these two services are used for the area-averaged values at the river basin scale. The values of the services are for 2010.

Ecosystem services	Units	Quantification methods	Data sources and references
Food supply	yuan/km ²	Assignment of food outputs of different food types to associated	Kunming Statistical Yearbook 2011, Statistics Bureau of

		land use types	Kunming, 2011
NPP	gC/m ² /yr	Aggregation of vegetation NPP and NPP of artificial areas	Lin et al., 2008 ; MODIS GPP/NPP Project (MOD17), Numerical Terradynamic Simulation Group, 2013)
Habitat quality	relative value	InVEST biodiversity model	Tallis et al., 2013a , see Table A-2 in the appendix for data sources
Water yield	mm/pixel, m ³ /ha	InVEST water yield model	Tallis et al., 2013a, see Table A-2 for data sources
Nitrogen retention	kg/pixel, kg/ha	InVEST water purification model	Tallis et al., 2013a , see Table A-2 for data sources

Food supply, NPP and habitat quality

Food is an essential ecosystem good to sustain human's survival. Food supply is a typical ecosystem provisioning service, which is often a subject of trade-offs with many other services (Kandziora et al., 2013). NPP can be used to indicate the global climate regulation service as it represents the net carbon uptake from the atmosphere into biosphere (Ingraham and Foster, 2008). Habitat quality represents land's capability to support biodiversity. It relates to the capacity of ecosystems to provide conditions suitable for individual and population persistence (Tallis et al., 2013a).

Hou et al. (under review) quantified and mapped the food productivity, NPP and habitat quality of the six districts or counties of Kunming region which covers the nine river basins in this study (for the area of these districts or counties, see Figure 1). To estimate food productivity, the statistical data of agricultural output were linked to the land use spatial data. To derive the NPP values of the study area, they used the MOD17A3 NPP product of 2006, which were produced by the Numerical Terradynamic Simulation Group of the University of Montana (see Table A-2 in the appendix for data sources) and referred to the work of Lin et al. (2008) for the NPP of urban areas. To quantify and map habitat quality, the biodiversity model of InVEST 2.5.5 was used (Tallis et al., 2013a). For the details of the above mentioned quantifications and methods, see the online appendix of this article. This study extracted the values of the food supply and habitat quality of the nine investigated river basins from the estimations and maps of these two ecosystem services from Hou et al. (under review). NPP of this study was quantified and mapped using the same method as the study of Hou et al. (under review) and the data was for 2010.

Water yield and nitrogen retention

Water yield is a key provisioning service for watershed areas. It is sensitive to the changes of landscapes, climate, hydrological and topographical statuses (Leh et al., 2013; Tallis et al., 2013a). Nitrogen retention relates to the amount of total nitrogen retained by landscapes. It can be a proxy of water purification, which is an important ecosystem regulating service, as total nitrogen concentration is a primary indicator of the water quality of water bodies (Leh et al., 2013; Tallis et al., 2013a).

In this study we estimated and mapped the water yield and the nitrogen retention of the nine river basins using the water yield and water purification models of InVEST 2.5.5, respectively. The water yield model computes annual average water yield by estimating the relative contributions of water from different parts of a landscape (precipitation minus water storage and evapotranspiration losses). The water purification model calculates the removal of

nutrient pollutants from runoffs contributed by vegetation and soil on every pixel and then sums and averages nutrients retention per sub-watershed. Nitrogen export is simultaneously estimated and mapped by the model. For the working principles of the two models, see the online appendix of this article. For greater details of the models, see the InVEST 2.5.5 users' guide (Tallis et al., 2013a, also see Leh et al., 2013). The data used for the modeling of these two services in this study are listed in Table A-2 in the appendix in this article.

Ecosystem services mapping and aggregation

The water yield and nitrogen retention raster maps were produced by the InVEST models in the form of grids. The five maps shown by Figure 2 in the results section demonstrate the spatial variations of the investigated ecosystem services in the nine river basins. In order to compare the general supplies of the five services between the nine river basins, the pixel values from the maps of the services were aggregated to area-averaged values for each river basin. The area-averaged values of water yield and nitrogen retention were generated by the two models, respectively. The average values of the other three services were derived using the Zonal statistics tool from ArcGIS 10. The ecosystem service average values of the river basins were further used to explore the interrelations between ecosystem services and socioeconomics. To facilitate the demonstration of the average values of the five services in different units on the same map, we standardized the values to a 0-10 scale with the following formula:

$$E_{s(i,a)} = \frac{E_{i,a}}{E_{max,i}} \times 10 \quad (1)$$

Where: $E_{s(i,a)}$ is the standardized average value of ecosystem service i of river basin a , $E_{i,a}$ is the original average value of ecosystem service i of river basin a , $E_{max,i}$ is the maximum average value of ecosystem service i among the nine river basins.

Moreover, we aggregated the pixel values of the five services to average values of different land use classes. Consequently, the quantity of ecosystem services provided by different land use classes can be revealed and compared.

2.3.2. Socioeconomic indicator selection and quantification

The ecosystem service supplies of human-dominated watersheds are significantly affected by socioeconomic activities (Reyers et al., 2009; Zhang et al., 2008). Indicating the extensity and intensity of socioeconomic activities needs appropriate measures that are sensitive to the spatial and temporal variations of the regional socioeconomic status. In this study we selected indicators relating to three primary socioeconomic dimensions of human-environmental systems: land use, demography and economic activities. Considering the extensive agricultural productions of the research area and the critical influences of agricultural activities on ecosystem services (Hou et al. 2014; Nelson et al., 2005), we especially chose the variables concerning chemical fertilizer use, rural electric power consumption, irrigation and agricultural machinery power, which can indicate agricultural production inputs. Accordingly, 22 indicators were selected to represent the socioeconomic state of the study area (Table 3).

The values of the socioeconomic indicators of the nine river basins were derived through two major approaches. Land use proportions and road density were calculated from the land use and road maps. Values of other indicators were mainly downscaled from the districts or counties where the river basins are situated. The indicator values of the districts or counties were obtained from the statistical yearbook of Kunming for 2011. The indicators, the respective units and values and the explanations and calculations of the indicators are listed in Table 3.

Table 3. Overview of the socioeconomic indicators, the values, the explanations and calculations. The road density value is for 2007 and the other indicator values are for 2010.

Category	Indicators	Units	Minimum, median and maximum values	Explanations and calculations	
Land use	Cropland proportion	%	4.6, 20.9, 56	Cropland area divided by total area	
	Forest proportion	%	12.6, 37.6, 49.3	Forest area divided by total area	
	Grassland proportion	%	7.5, 26.4, 32.9	Cropland area divided by total area	
	Rural residential area proportion	%	1.1, 4, 12	Rural residential area divided by total area	
	Urban area proportion	%	1.4, 3.7, 19.6	Urban area divided by total area	
	Road density	km/km ²	0.11, 0.30, 2.15	Road length divided by the area of the respective river basin	
Demography	Rural residential area population density	persons/km ²	2256, 4435, 14167	Rural population divided by rural residential area, downscaled from the district/county where the river basin is situated	
	Urban population density	persons/km ²	3020, 4320, 23506	Urban population divided by urban area, downscaled from the district/county where the river basin is situated	
	Total population density	persons/km ²	275, 608, 4876	Total population divided by total area of the respective river basin, total population is the sum of rural and urban populations	
Agricultural production	Chemical fertilizer use	tonnes/km ²	45.3, 73.1, 213.8	Total chemical fertilizer use (pure substance) divided by cropland area	The indicator values for the river basins were downscaled from the districts/counties
	Nitrogen fertilizer use	tonnes/km ²	25.2, 32.5, 116.2	Total nitrogen fertilizer use divided by cropland area	
	Phosphorus fertilizer use	tonnes/km ²	7.9, 10.1, 40	Total phosphorus fertilizer use divided by cropland area	
	Potassium fertilizer	tonnes/km ²	0.8, 5.3	Total potassium fertilizer use divided by cropland area	

	use		11.7			where the river basins are situated
	Compound fertilizer use	tonnes/km ²	5.0, 45.7	16.8,	Total compound fertilizer use divided by cropland area	
	Rural electric power consumption	10 ⁴ kWh/km ²	3.6, 11.3	6.0,	Total rural electric power consumption divided by total rural area	
	Cropland effective irrigation rate	%	14.4, 100	31.1,	The area of effectively irrigated cropland divided by total cropland area	
	Forest and garden irrigation rate	%	0.05, 4.3	0.85,	The area of irrigated forest and garden divided by total forest and garden area	
	Total agricultural machinery power	10 ⁴ W/km ²	5.3, 28.5	18.5,	Total agricultural machinery power divided by total rural area	
Economic output	GDP per person	10 ⁴ yuan/person	0.7, 1.7, 6.4		GDP divided by total population, GDP is downscaled from the respective district/county by area ratio of the river basin to the district/county, total population is the sum of rural and urban population	
	Primary sector output	10 ⁴ yuan/km ²	31.8, 143.5	92.4,	Primary sector output divided by the total area of non-urban area (cropland, forest, grassland, lakes, rural residential area)	The indicator values for the river basins were downscaled from the districts/counties where the river basins are situated
	Secondary sector output	10 ⁴ yuan/km ²	9032.7, 15594.1, 62741		Secondary sector output divided by the total area of urban area and other construction area	
	Tertiary sector output	10 ⁴ yuan/km ²	3751.1, 4463.9, 39156.1		Tertiary sector output divided by total urban and rural residential area	

2.3.3. Hypotheses and tests of socioeconomic influences on ecosystem services

Uncertainties are inevitable in the process of downscaling (Wu et al., 2006). The values of the socioeconomic indicators downscaled from the respective districts or counties to the river basins can be inaccurate. Consequently, the quantitative analysis of the relationships between socioeconomics and ecosystem services based on these values may produce incorrect results. To avoid the false findings of the socioeconomic influences in our case study, we employed the confirmatory research procedure, by proposing the hypotheses of the socioeconomic impacts on ecosystem services and testing the hypotheses by statistical analysis with the data derived in our case study.

We built the hypotheses on the basis of literature investigations (Bai et al., 2010; Pan et al., 2012; Polasky et al., 2011; Wu et al., 2013) and our own research experience in different study areas (Table 4). These qualitative hypotheses are based on an annual temporal scale and subject to a hypothetical typical watershed in Southwest China with a characteristic landscape pattern consisting of forest, grassland, cropland, water body as well as urban and rural areas. This assumed watershed has an area smaller than 300km². Undoubtedly, the relations between socioeconomics and ecosystem services might be different at other scales or in other areas.

Table 4 must be read from the vertical Y-axis to the horizontal X-axis. The intersections show the hypothetical influences of the socioeconomic attributes on the ecosystem services items. The hypotheses only considered significant direct effects in normal situations. However, it should be noted that indirect relations may be existing under some special conditions. The influences of socioeconomics were classified into four categories:

- + Socioeconomic attributes have positive influences on ecosystem services, i.e., the enhancement of a socioeconomic attribute increases the respective ecosystem service;
- Socioeconomic dimensions have negative influences on ecosystem services;
- + - The intensification of socioeconomic dimensions either enhances or undermines ecosystem services, depending on specific cases;
- Socioeconomic dimensions have no relevant direct influences on ecosystem services.

To test the hypotheses with our empirical study of the nine river basins, we conducted Spearman correlation analyses to quantify the connectedness between each pair of socioeconomic and ecosystem service indicators. As ecosystem services are additionally affected by environmental factors, we firstly calculated the Spearman correlation coefficients (SCC) between the primary environmental variables and the five ecosystem service items. The results showed that food supply was significantly negatively correlated with elevation (SCC was -0.633). Besides, water yield was considerably negatively affected by temperature and shortwave radiation (SCC were -0.767 and -0.700, respectively) and nitrogen retention was also significantly negatively correlated with these two factors (SCC were -0.850 and -0.800, respectively). Considering these environmental influences, we conducted Spearman partial correlation analyses between the socioeconomic variables and food supply by taking elevation as the control variable. For the Spearman partial correlations between socioeconomics and water yield as well as nitrogen retention, the spatial variability of temperature and shortwave radiation were controlled. The statistical analyses were achieved using SPSS 17.0 and Table A-1 in the appendix of this article provides the correlation coefficients.

Table 4. Hypotheses of socioeconomic influences on ecosystem services at watershed scale. The hypothetical scale is a watershed smaller than 300 km² in Southwestern China and the hypothesized value is annual mean value. All the indicators are intensive measures.

- + Socioeconomic dimensions have positive influences on ecosystem services.
- Socioeconomic dimensions have negative influences on ecosystem services.
- + - Socioeconomic dimensions have either positive or negative influences on ecosystem services, depending on specific cases.
- Socioeconomic dimensions have no relevant direct influences on ecosystem services.

Ecosystem service indicators		Food supply	NPP	Habitat quality	Water yield	Nitrogen retention
		yuan/km ²	gC/m ² /yr	relative value	m ³ /ha	kg/ha
Socioeconomic indicators						
Cropland proportion	%		+ -	+ -	+ -	+ -
Forest proportion	%		+	+	-	+
Grassland proportion	%		+ -	+ -	+ -	+ -
Rural residential area proportion	%		-	-	+	+ -
Urban area proportion	%		-	-	+	-
road density	km/km ²		-	-	+	-
Rural residential area population density	persons/km ²	+	-	-	+	-
Urban population density	persons/km ²		-	-		
Total population density	persons/km ²	+	-	-		-
Chemical fertilizer use	tonnes/km ²	+	+	-		+
Nitrogen fertilizer use	tonnes/km ²	+	+	-		+
Phosphorus fertilizer use	tonnes/km ²	+	+	-		
Potassium fertilizer use	tonnes/km ²	+	+	-		
Compound fertilizer use	tonnes/km ²	+	+	-		+
Rural electric power consumption	10 ⁴ kWh/km ²	+		-		
Cropland effective irrigation rate	%	+	+	+ -	+	-
Forest and garden irrigation rate	%	+	+	+ -	+	-
Total agricultural machinery power	10 ⁴ W/km ²	+	+	-		
GDP per person	10 ⁴ yuan/person	+	-	-		-
Primary sector output	10 ⁴ yuan/km ²	+	+ -	+ -	+ -	+ -

Secondary sector output	10^4 yuan/km ²	+	-	-	+	-
Tertiary sector output	10^4 yuan/km ²	+	-	-	+	-

3. Results

3.1. Ecosystem service spatial variations

Food supply appeared with striking spatial variations in the research area. The most food outputs in monetary values were supplied by the southern area of the Laobaoxiang River Basin (No. 3) and the northern area of Maliao River Basin (No. 4), followed by that provided by the northern parts of the Da River Basin, the Chai River Basin and the Dongda River Basin (No. 6, 7, 8). Comparatively, the other parts of these river basins and the other river basins supplied much less food per square kilometer (Figure 2, a). Comparing the land use map of Figure 1 and the food supply map indicates the notable overlap of the areas with high food supplies and cropland.

NPP and habitat quality showed similar spatial characteristics presenting high divergence in different parts of different river basins (Figure 2, b and c). High NPP and habitat quality can be found on the periphery of all the river basins except the Maliao River Basin (No 4). Low values of these two services were mainly observed in the south of three watersheds: the Xinyunliang River Basin, the Panlong River Basin and the Laobaoxiang River Basin (No. 1, 2, 3). The similar phenomenon occurred in most part of the Maliao River Basin and the Laoyu River Basin (No. 4, 5) as well as in the north of the Da River Basin, the Chai River Basin and the Dongda River Basin (No. 6, 7, 8). The most considerable distinction between the spatial distributions of NPP and habitat quality was that the Tanglang River Basin (No. 9) entirely showed high habitat quality while the middle part provided relatively low NPP.

Water yield generally demonstrated an opposite trend to habitat quality in spatial distribution (Figure 2, d). The highest water yield can be found in the south of the Xinyunliang River Basin and the Panlong River Basin (No. 1, 2), in the southwest and the airport area of the Laobaoxiang River Basin (No. 3) as well as in the east of the Laoyu River Basin (No. 5). Moderate water yield values mainly appeared in the areas covered by cropland (see Figure 1) and low values of this service can be found on the periphery of all the river basin.

Concerning nitrogen retention, the spatial characters were considerably different from the other four services. High nitrogen retention solely occurred in dispersed small patches, which were primarily located in the Xinyunliang River Basin, the Panlong River Basin, the Laobaoxiang River Basin and the Laoyu River Basin (No. 1, 2, 3, 5). Other parts of the research area retained negligible amounts of nitrogen (Figure 2, e).

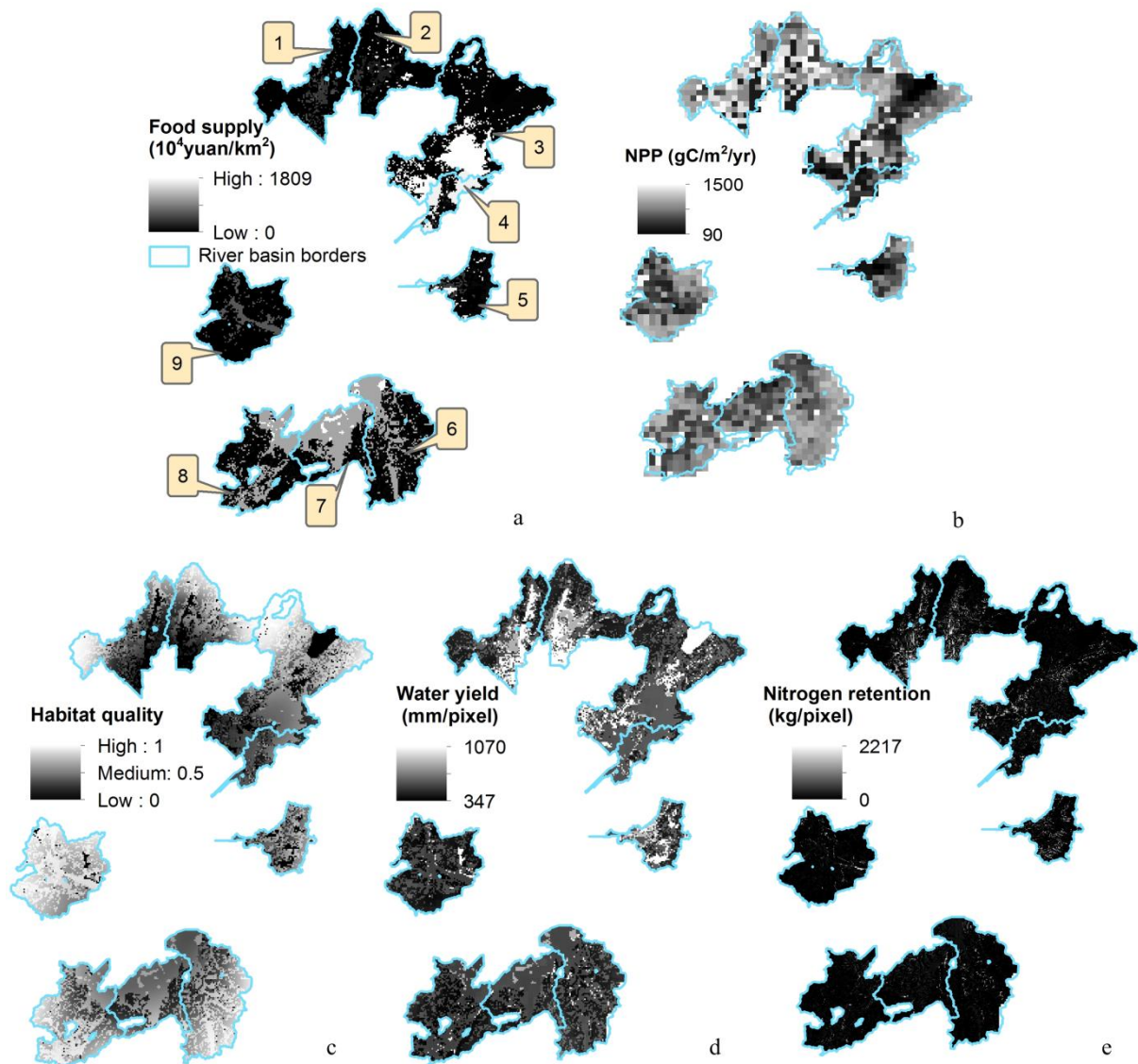


Figure 2. Spatial distributions of the five investigated ecosystem services in the nine river basins in 2010

3.2. Ecosystem service comparisons between river basins and land use types

Concerning the standardized scores of the ecosystem service average values, the comparisons between different river basins showed distinctions between different service types (Figure 3). Food supply and nitrogen retention both presented large gaps in quantity between different river basins. The extreme example relating to the Xinyunliang River Basin and the Maliao River Basin showed that food supply per unit area of the former was sole 1/10 of the latter. Concerning nitrogen retention, the three river basins in the south of the Dianchi Lake and the Tanglang River Basin in the west of the lake provided much lower values than the other five river basins. Habitat quality showed moderate differences between river basins with the largest gap by a factor of 0.4 between the Maliao River Basin and the Tanglang River Basin. The minor value gaps between different river basins involve NPP and water yield. It was the Laoyu River Basin that played both the roles to generate the most water yield and least NPP by unit area. Concerning water yield, the Laoyu River Basin had the value 1.3 times of that of

the Tanglang River Basin, which had the smallest value. With respect to NPP, the value of the Laoyu River Basin was 70% of the largest value from the Xinyunliang River Basin.

Another noticeable phenomenon was the emergence of ecosystem service bundles⁸ in some river basins. For example, the Xinyunliang River Basin and the Panlong River Basin both apparently presented high values of NPP, water yield and nitrogen retention; The Laoyu River Basin had an outstanding performance to provide both the service of water yield and nitrogen retention. Additionally, low values of NPP, habitat quality and water yield were simultaneously subject to the Da River Basin.

The area or pixel-averaged values of the five ecosystem services showed distinctions in different degrees between different land use types (Table 5). The most notable variations were subject to food supply and nitrogen retention. Concerning food supply, cropland played the most important role, followed by rural residential area. Grassland, urban area and forest provided much less food than cropland by unit area. Food supply from other construction areas was negligible as it was only 1/26 of that of cropland. With respect to nitrogen retention, urban area played the leading role by averagely retaining 6.92 kg nitrogen by each pixel, followed by rural residential area. Cropland and grassland retained less than half nitrogen per pixel of that of rural residential area. Unexpectedly, forest performed disappointingly in retaining nitrogen with the pixel value less than 1/10 of that of urban area. Concerning the provisions of the services of water yield and habitat quality, the rankings of land use types were nearly reversed. The highest water yields per pixel were subject to other construction area and urban area, which contrarily, enjoyed the lowest habitat quality. Forest and grassland had the best habitat quality but yielded the least water averagely by each pixel. Besides water yield, NPP appeared with slight changes between different land use classes. As expected, forest and grassland supplied the most and second most NPP by unit area, followed by cropland, rural residential area, urban area and other construction area in a descending order of the magnitudes of NPP values.

⁸ Ecosystem service bundles are a sets of ecosystem services that repeatedly appear together across space or through time (Raudsepp-Hearne et al., 2010)

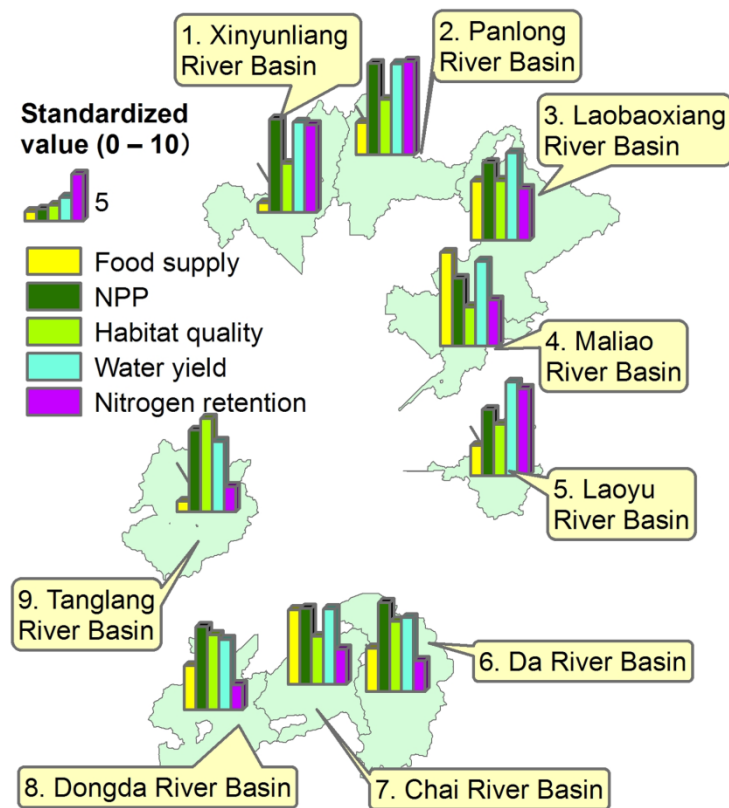


Figure 3. Standardized area-averaged values of the five investigated ecosystem services of the nine river basins in 2010. Different service types are not comparable as their original values have different units.

Table 5. Ecosystem service values of different land use classes of the study area in 2010. The pixel sizes of water yield and nitrogen retention are 200×200 m and 30×30 m, respectively.

Land use classes	Food supply (10 ⁴ yuan/km ²)	Water yield (mm/pixel)	NPP (gC/m ² /yr)	Habitat quality	Nitrogen retention (kg/pixel)
Urban area	53.7	1002.6	549.0	0.19	6.92
Rural residential area	241.1	725.7	565.9	0.43	3.57
Forest	35.0	403.1	699.5	0.62	0.48
Grassland	68.2	527.3	633.7	0.61	1.61
Cropland	413.7	511.2	569.8	0.54	1.45
Other construction area	15.6	1014.6	252.3	0.04	0.13

3.3. Tests of hypotheses

Table A-1 in the appendix gives the correlation coefficients of the hypotheses test results, which can be differentiated into five categories, i.e., significant positive or negative correlations with *P* smaller than 0.05 or between 0.05 and 0.1 and correlations not significant

with P greater than 0.1. Apparently, the number of the significant correlations is much smaller than that of the hypothetical significant direct influences of socioeconomics on the five ecosystem services (Table 4). Comparing the hypotheses and test results tables, we can find out verified, falsified and neither verified nor falsified (invalidly tested) hypotheses.

The verified hypotheses relate to 3 situations, 1) positive or negative socioeconomic influences as hypotheses and respective significant positive or negative correlations as test results, e.g., forest proportion–NPP and forest proportion–water yield, 2) either positive or negative hypothetical socioeconomic influences and respective significant positive or negative correlations as test results, e.g., rural residential area proportion–nitrogen retention and 3) no direct socioeconomic influences as hypotheses and correlations not significant in the test results table, e.g., urban area proportion–food supply.

Concerning the falsified hypotheses, 3 types of findings are included, i.e., 1) hypothetical positive socioeconomic influences with significant negative correlations in the test results table, e.g., rural residential area population density–water yield, 2) hypothetical negative socioeconomic influences with significant positive correlations, e.g., road density–NPP and 3) no direct socioeconomic influences in the hypotheses table but significant positive or negative correlations in the respective cells in the test results table, e.g., cropland proportion–food supply.

Invalid tests involve the hypotheses that socioeconomics have relevant direct influences on ecosystem services, which were neither verified nor falsified by significant correlations in the test results table, e.g., cropland proportion–NPP, urban area proportion–NPP and road density–water yield. By adding up the numbers of these three types of hypotheses, we can find that 31 hypotheses were verified by the statistical analysis with empirical data, 9 were falsified and 70 were invalidly tested.

Integrating the hypotheses and test results, we preliminarily considered the influential socioeconomic factors of ecosystem services as the ones hypothetically having direct influences which were statistically proved by the tests. Accordingly, the socioeconomic influences on ecosystem services at spatial scale in the study area can be interpreted as follows:

- 1) For food supply, rural electric power consumption and primary sector output had direct positive influences;
- 2) NPP was enhanced by the increases of forest proportion and cropland effective irrigation rate;
- 3) Habitat quality increased with the growth of forest proportion and the decline of rural residential area proportion;
- 4) The positive influences on water yield came from rural residential area proportion and urban area proportion and the negative influence related to forest proportion;
- 5) Nitrogen retention increased with more rural residential area proportion and chemical fertilizer use and decreased with higher GDP per person.

Other possible socioeconomic influences that were not confirmed by the hypotheses-tests conformity as described above are examined in the discussion section.

4. Discussion

This study has quantified and mapped five typical provisioning and regulating ecosystem services provided by the nine river basins of the Dianchi Lake Watershed. Besides, it

estimated and compared the average values of ecosystem services of different river basins and land use types to show the multi-functionality of the river basins and land use classes. Based on these evaluations, we further investigated the socioeconomic causes of the spatial variability of the five ecosystem services at the river basin scale using a hypothesis–test procedure. In this section, the findings of the hypothesis–test methodology are examined, the modeling results of nitrogen retention are combined to the “ecosystem services potential, flow and demand” concepts and the uncertainties in the modeling of ecosystem services are treated by a sensitivity analysis.

4.1. Socioeconomic influences on ecosystem services

4.1.1. Reasons for the unsatisfactory test results

The hypotheses suggested considerable direct socioeconomic influences on the five ecosystem services assessed in this study in a typical hypothetical watershed. Nevertheless, only a small part of the hypotheses were verified by the tests and even a smaller number of the socioeconomic–ecosystem services correlations were statistically significant. Looking into this phenomenon, we argue that the major reasons are the downscaling procedure in the assignment of the values of the socioeconomic variables, the small sample size for the correlation analysis and the uncertainties in the quantification, especially in modeling the ecosystem services.

As depicted in the methods section and denoted in the socioeconomic indicators table (Table 3), we downscaled the values of some socioeconomic indicators from the respective districts or counties to the river basins. This value transference undoubtedly caused major inaccuracy, although all the indicators are intensive measures (e.g., percentage or area averaged), as the socioeconomic phenomena were not evenly distributed across space in the respective districts or counties. In empirical studies, researchers should endeavor to match ecosystem services and socioeconomic in space and time when exploring their relationship in order to derive credible results (Hou et al., 2014; Pan et al., 2012). However, scale mismatches are sometimes inevitable and downscaling or upscaling of values is necessary, especially when ecosystem services and socioeconomic are considered simultaneously, as ecosystem services are often provided by environmental or ecological units (e.g., watershed in this study) and socioeconomic data are usually only available at administrative units (e.g., districts or counties in this study) (Rounsevell et al., 2010; Wu and Li, 2006; Yu et al., 2014). Concerning the second reason of the dissatisfactory test results, the small sample size, we used the nine river basins as the sample and the respective average values of the ecosystem services and socioeconomic variables for Spearman correlation analysis. Spearman correlation on the one hand is an appropriate method to cope with small sample sizes, non-linearity and outliers, on the other hand, leads to a loss of information and may conceal significant correlations. Taking one correlation of this study as an example, the coefficient of the correlation between urban area proportion and nitrogen retention is 0.645, which is not significant at the 0.1 level. Therefore, urban area proportion was excluded from influential factors. However, this variable was actually considerably correlated with nitrogen retention in a linear way shown by the scatter chart. Consequently, it should be regarded as a factor having direct positive influence on nitrogen retention. The third potential cause, the uncertainties in ecosystem services quantification, may also undermine the effectiveness of the tests, as the derived values of the five ecosystem services in this study may not well conform to the true values. This inaccuracy may be enhanced when modeling is used to replace measurement or observation (Hou et al., 2013).

4.1.2. Limitations of modeling

Limitations and simplifications of model assumptions significantly contribute to the uncertainties of modeling results (Hou et al., 2013). In this study, habitat quality, water yield and nitrogen retention are modeled by respective InVEST modules, which may cause biases in the results because of the respective limitations. For example, the habitat quality module assumes that all the threats are additive, without considering the cases that the collective impact of multiple threats is greater than the sum of individual threat levels. The water yield module is built on a number of limitations. An essential one relates to the limited modeling temporal scale, i.e., the input data are all based on annual averages, which neglect extremes and do not consider the temporal characters of water supply. Another one is the disregard of the differences of water consumptive demand between parcels of the same land use or land cover class. The nitrogen retention module is also subject to several limitations. For example, the saturation of the uptake is not considered, which may cause the overestimation of the results. Additionally, the model does not address any biological or chemical interactions of the pollutants. The InVEST users' guide detailedly demonstrates the limitations and simplifications of each module (Tallis et al., 2013a). The users should be aware of the constraints of the modules. This can help them in analyzing the uncertainties of the modeling results.

Model comparisons can examine the applicability of models and the reliability of the results (Hou et al., 2013). Up to now, some studies have compared InVEST to other modeling tools, very few of which refer to empirical investigations. Both being ecosystem service modeling tools, InVEST and ARIES have respective strengths and weaknesses (Vigerstol et al., 2011). Compared to ARIES, InVEST is much more constrained in temporal scales and can only cope with annual average values. Additionally, it demands more data and expertise. Bagstad et al. (2012) have compared the modeling results of these two tools in an empirical study in Arizona, which revealed that the divergence degrees between the results of the two tools are different in distinct modeling scenarios. Besides ARIES, InVEST is compared to SWAT (Soil and Water Assessment Tool) in modeling water-related ecosystem services both conceptually and in real-world studies. Compared to InVEST, SWAT is more sophisticated and data and expertise-demanding. However, when the modeling is based on the grid cell scale, it is InVEST, instead of SWAT that can come into play (Vigerstol et al., 2011). Concerning this study, we considered these respective advantages and disadvantages of the two tools and used InVEST to model the two water-related services. One empirical study has compared SWAT and InVEST in modeling water yield and nutrient export. The results showed good consistency between the two tools of some scenarios while disagreements of some other ones (Keeler et al. 2013).

Model validation is one of the most effective approaches to examine the applicability of the models. Nevertheless, validations are often difficult because of the shortages of data from observations or measurements. In this study, we didn't validate the modeling results owing to data insufficiency. However, we have found two empirical studies, which have succeeded in validating the modeling results of water yield and nitrogen retention from InVEST, respectively (Bai et al., 2013a; Marques et al., 2013). These two studies have increased our certainty of the modeling results in this study.

4.1.3. Influences of different land uses on nitrogen retention

As argued above, the increase of urban area proportion enhanced nitrogen retention in the study area, although this direct influence was not revealed by the Spearman partial correlation analysis. Some other socioeconomic influences, which were not supported by the hypotheses-tests conformity, can be further found out when looking into the reality of the study area.

Forest proportion was hypothesized to positively influence nitrogen retention per area but was found to be significantly negatively correlated with this service in the test. This contradiction excluded forest proportion from the influential factors in our study area. Nonetheless, when looking into the nitrogen retention value per pixel of different land use types (Table 5), this statistical result becomes reasonable. As can be seen in Table 5, forest retained the second least nitrogen per pixel among all the land use types in the study area (0.48 kg/pixel). This value was only about 1/7 and 1/14 of that of rural residential area and urban area, respectively. In other words, the increase of forest proportion and decrease of the share of most other land use classes, especially urban and rural residential areas, would reduce nitrogen retention in the study area. As described in earlier sections, urban and rural residential area proportions were significantly positively correlated with nitrogen retention. These phenomena seem to contradict the common sense that vegetation-covered landscapes have more potential or capacity in retaining nutrients than artificial areas with high impervious surface proportions (Burkhard et al., 2009). However, this disagreement is understandable when the reality of the influential factors of nitrogen retention in the research area is considered. As suggested by many studies, nitrogen retention of a landscape is essentially determined by nitrogen flux, the residence time of run-offs and the properties of soil and temperature (Groffman et al., 2004; Kaushal et al., 2011; Wollheim et al., 2005). Nitrogen flux is decided by the amount of nitrogen load on the land and the mass of nitrogen transported from the land upstream; the other three factors decide the nitrogen retention potential of the landscape (Maes et al., 2012; Wollheim et al., 2005). Vegetation-covered land should have more nitrogen retention potential than developed area when geomorphology and temperature are not considered. However, the effects of these environmental factors can narrow this potential gap, e.g., flat artificial area may have longer residence time than steep forest-covered land. Furthermore, when nitrogen fluxes are much higher in developed areas (e.g., urban and rural residential areas in this study) than in natural areas (e.g., forest and grassland), the actual amount of nitrogen retention of the former can exceed the latter, even though residential areas have lower retention potentials (Wollheim et al., 2005).

Concerning the nine river basins of this study, urban and rural residential areas had strikingly high nitrogen loads and were mainly located downstream the rivers with low slope. Conversely, forest and grassland had much lower nitrogen load and were primarily located in upstream areas with steeper slopes (see Table B-1 in the online appendix for nitrogen load and Figure 1 for the locations of different land use types). Consequently, urban and rural residential areas retained much more nitrogen per pixel than forest and grassland in this study area. This finding is consistent with several other studies (Groffman et al., 2004; Kaushal et al., 2011; Wollheim et al., 2005). Considering the reality of the study area described above and the respective arguments, we include forest proportion and urban area proportion as a negative and a positive influence on nitrogen retention, respectively.

4.1.4. Influences on habitat quality, water yield, NPP and food supply

Apart from nitrogen retention, land use structure of the study area had noticeably opposing influences on habitat quality and water yield. Forest proportion was confirmed to be positively influencing habitat quality while negatively influencing water yield. Contrarily, the increase of rural residential area proportion led to the decline of habitat quality and growth of water yield. Additionally, urban area enhanced water yield of the river basins. These findings are expected and reasonable due to the definitions and model assumptions of the two services. To be specific, a hospitable habitat of species is subject to vegetation-covered and intact landscape; Water yield, which is defined as the water volume exported downstream from an ecosystem, is reduced by evapotranspiration, especially in forest ecosystems. These effects of

different land use types agree with some empirical studies in other areas (Baral et al., 2014; Leh et al., 2013; Li et al., 2013).

Concerning the socioeconomic influences on NPP, one land use attribute, forest proportion and one agricultural production input factor, cropland effective irrigation rate were proved to be supportive. The former influence is apparent and the latter one should be attributed to this indicator's representing role of agricultural production efficiency and productivity. The boosting of agricultural production efficiency can save agricultural land for natural vegetation land which has relative high NPP (Hou et al., 2014, see the NPP values of different land use classes in Table 5). Food supply per unit area was confirmed to be only positively influenced by rural electric power consumption and primary sector output. These results are understandable as these two dimensions can represent agricultural productivity to some extent. Concerning the land use structure attributes, none of them was hypothesized to directly influence food supply and most of the hypotheses were verified by the tests. Such findings are rational as none of these attributes directly affect food productivity.

4.2. Ecosystem services potential, flow and demand

The distinction of ecosystem service potential, flow and demand is important in regional ecosystem services assessments, especially when the evaluators aim at supporting the policy-making of regional environmental management with the assessment outcomes (Burkhard et al., 2014)⁹. Up to now, some ecosystem service researchers have differentiated and demonstrated these three concepts and conducted empirical studies (Bagstad et al., 2013; Burkhard et al., 2014; Maes et al., 2012; Reyers et al., 2009; Schroter et al., 2014). In this section, we further exemplify the concepts on the basis of the modeling results of nitrogen retention in this study. We mapped the nitrogen retention, export, load as well as nitrogen retention potential of the nine river basins and revealed the quantitative relationships of these measures with scatter diagrams. The scattered plots indicate the average values of the measures of the nine river basins (Figure 4 and Figure 5). In this study, nitrogen retention, retention potential and load stand for ecosystem service flow, potential and demand, respectively and nitrogen export represents the balance of demand and flow. The quantification of ecosystem service potentials should be based on biophysical properties and ecosystem process and generate absolute values (Schroter et al., 2014). However, due to the unavailability of sufficient information of the investigated ecosystem, these two requirements are not always fulfilled and researchers may simplify their assumptions and use relative values (Burkhard et al., 2009). In this study we only considered land use type as a determinant of nitrogen retention potential and used the percentage of nitrogen flux that can be retained by a certain land use type as the potential value (see "N retention coefficient" in Table B-1 in the online appendix).

Figure 4 generally shows a spatial consistence of nitrogen retention, export and load and a spatial nonconformity of nitrogen retention potential to the other three variables. These phenomena are quantitatively demonstrated by the scatterplots in Figure 5. Figure 5 apparently shows high correlations between nitrogen retention and export, retention and load as well as export and load (a, b and d) in the nine river basins. Differently, the correlation

⁹ Ecosystem service potential is the hypothetical maximum yield of selected optimized ecosystem services. It can be understood as the capacity of ecosystem to produce ecosystem services (Burkhard et al., 2012; Maes et al., 2012); Ecosystem service flow refers to the actually used ecosystem services by humans, which fulfil human demand (Burkhard et al., 2014); Ecosystem service demand relates to the currently consumed or used ecosystem goods or services without considering where ecosystem services are actually provided (Burkhard et al., 2012).

between nitrogen retention and retention potential is not strong (Figure 5, c). Concerning the correlations and inconsistency in spatial extent and quantities of these four N-measures, some other studies derived similar results (Maes et al., 2012; Wollheim et al., 2005). Besides, Schroter et al. (2014) have found the spatial and quantity differences between potential and flow of some other ecosystem services.

The above delineated findings suggest the importance of the spatial mismatch issue of ecosystem service potential, demand and flow in regional ecosystem service management practices. Such as in our study, nitrogen retention demand (nitrogen load) occurred mainly in urban and rural residential areas and on croplands, which were primarily located downstream in the river networks of the river basins. In contrast, forest and grassland, although having relative high nitrogen retention potentials, were mainly distributed upstream the river networks and retained only a small amount of nitrogen (provided little ecosystem service flow). This spatial mismatch resulted in a large amount of nitrogen export, especially in river basins 1, 2 and 5, which had high proportions of urban and rural residential areas (Table 1). It is important to reduce the spatial mismatch of ecosystem service demand and potential through some approaches, e.g., adjusting the spatial distributions of high ecosystem service potential and demand landscapes. For example, in Dianchi Lake Watershed of this study, the strategies can be growing riparian forests along the rivers, restoring lakeshore wetlands as well as locating new construction areas upstream in the river basins.

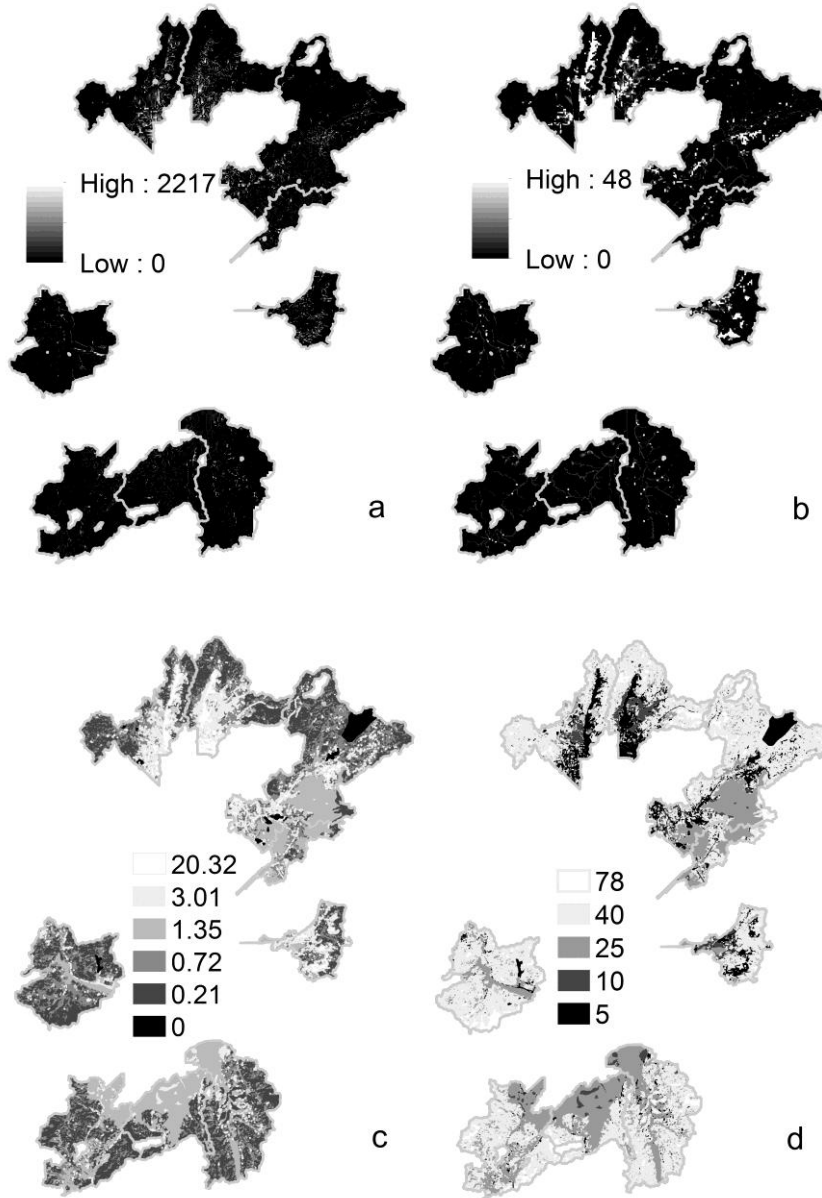


Figure 4. a, nitrogen retention (kg/pixel); b, nitrogen export (kg/pixel); c, nitrogen load (kg/pixel); d, nitrogen retention potential (%) of the nine river basins of the Dianchi Lake Watershed in 2010. Load is produced by the respective pixel, not including exported loads from upstream pixels.

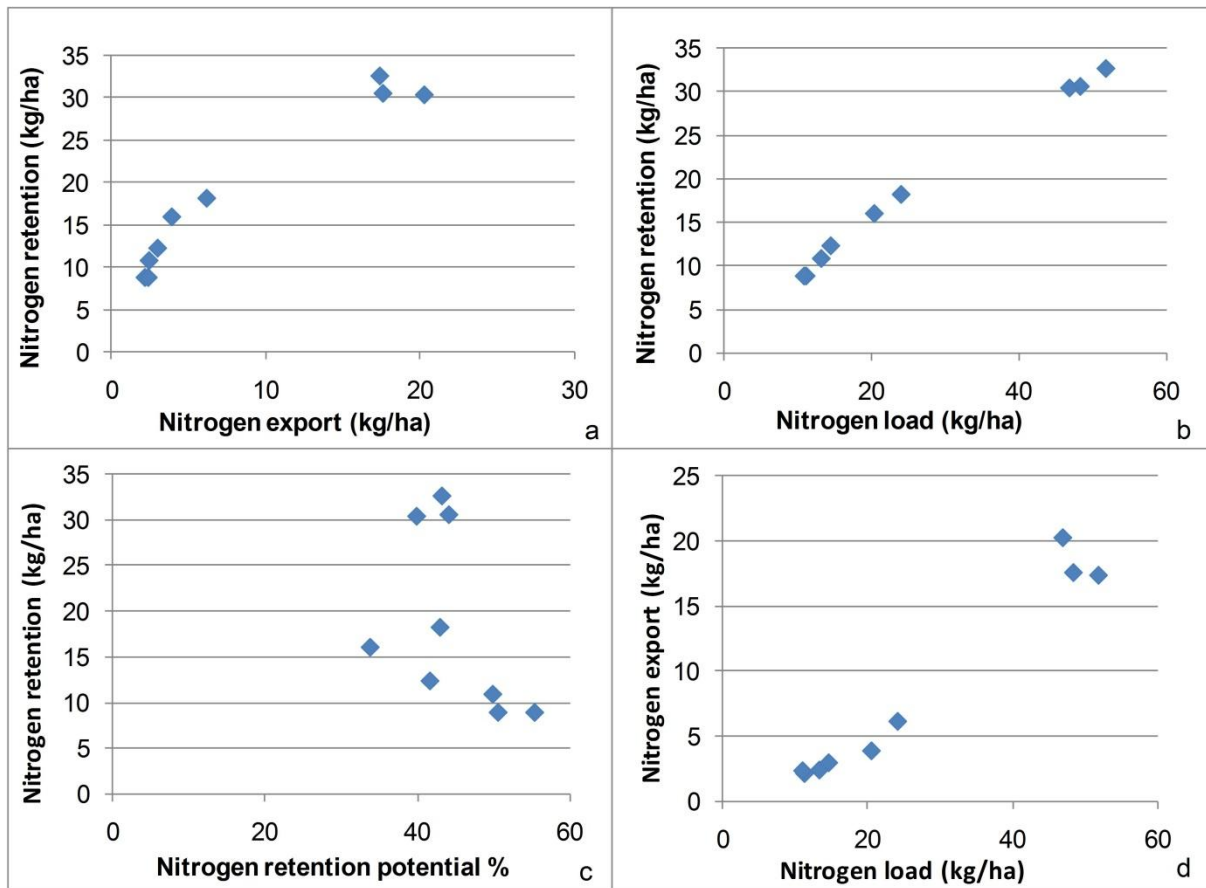


Figure 5. Quantitative relationships between the four nitrogen indicators concerning nitrogen retention potential, flow and demand. a, relationship between nitrogen export and retention; b, relationship between nitrogen load and retention; c, relationship between nitrogen retention and retention potential, percentage means the proportion of total nitrogen flux theoretically retained by a certain river basin; d, relationship between nitrogen load and nitrogen export.

4.3. Sensitivity analysis of ecosystem services modeling

4.3.1. Sensitivity analysis method

Sensitivity analysis is an important method to estimate the uncertainties of the results of ecosystem service assessments with respect to the used parameters or inputs. The method can test how influential the parameters or the input data are on the values of the evaluated ecosystem services by comparing the changes of the parameters or input data with the resulted changes of the assessment outcomes (Hou et al., 2013; Sanchez-Canales et al., 2012). If the results are sensitive to the parameters or input data, the ecosystem services assessors should be cautious about the uncertainties of these parameters or input data because they may lead to non-realistic estimates (Sanchez-Canales et al., 2012). Several studies have conducted sensitivity analyses to examine the influences of value coefficients of land use types on the estimated ecosystem service values by calculating the coefficient of sensitivity (CS) (Kreuter et al., 2001; Li et al., 2010a; Li et al., 2010b). This method is based on the standard economic concept of elasticity, i.e. the percentage change in the output resulted from a given percentage change in an input (Kreuter et al., 2001). Another work employed the Morris sensitivity index to test the sensitivity of water yield modeled by InVEST to three main coefficients: seasonal

precipitation distribution, annual precipitation and annual evapotranspiration (Sanchez-Canales et al., 2012). However, although this method is more sophisticated than the former, the calculation of CS, it is more time-consuming and more difficult to operate.

In this study, we used the CS calculation method to analyze the sensitivity of water yield and nitrogen retention to precipitation input data. Besides, we used the same method to test the sensitivity of the outcomes of nitrogen retention modeling to several model parameters, the nitrogen retention coefficients of urban and other construction areas as well as cropland. We chose these factors because we were least certain of the accuracy of the precipitation data and the nitrogen retention coefficients of these three land use types. By testing these factors, we can gain critical insight into the reliability of the estimated results (Kreuter et al., 2001). For the precipitation of the study area, we obtained data from two sources (see Table A-2 in the appendix). One is global average annual precipitation for 1950-2000 with a resolution of 1 km×1 km and the other one is the total precipitation of 2010 of China with a resolution of 10 km×10 km (the precipitations of each river basin generated by these two items of data are given by Table 6). The former was used as the initial data for the modeling which produced the results of this article as this data are more close to the reality of the research area. The latter, which had large variations between different river basins, was used as the alternative data for the sensitivity analysis. Additionally, we were uncertain about the nitrogen retention coefficients of urban areas, other construction areas (mainly airport area in this study) and cropland because different published work suggested distinct values of these coefficients (Leh et al., 2013; Tallis et al., 2013a; Wu et al., 2010). For the modeling in this study, we employed the more reliable coefficients, e.g. 5% for urban area and other construction area. The more doubtful coefficients were used as alternatives for the sensitivity analysis (Table 7 for the initial and alternative coefficients).

In each analysis, the coefficient of sensitivity (CS) of river basin “*k*” was calculated by:

$$CS_k = \frac{(ES_{jk} - ES_{ik}) / ES_{ik}}{(IP_{jk} - IP_{ik}) / IP_{ik}} \quad (2)$$

where ES is the modeled ecosystem service value (water yield or nitrogen retention in this study), IP is the input data or parameter (precipitation or nitrogen retention coefficient of urban and other construction area or cropland), “*i*” and “*j*” represent the initial and alternative values, respectively.

If CS is greater than 1, the estimated ecosystem service value is elastic with respect to that input data or parameter. If CS is smaller than 1, the modeled ecosystem service is considered to be inelastic. The greater the CS is, the more important is the use of an accurate parameter or data.

4.3.2. Sensitivity analysis results

Table 6 shows the effects of using alternative precipitation data as modeling input to the modeled values of water yield and nitrogen retention of the nine river basins. Among the river basins, the lowest sensitivity coefficient was 1.43 relating to the Laoyu River Basin (No. 5) with a 30.2% reduction of water yield caused by a 21.2% decrease of precipitation; the highest ratio of percentage water yield change to percentage precipitation change was 2.83, which occurred in the Maliao River Basin (No. 4). The considerably high values of the CSs of the river basins that were all greater than 1 indicate that water yield values were highly sensitive to precipitation. This phenomenon is consistent with the findings from Sanchez-Canales et al. (2012). It implies that inaccurate precipitation data can substantially affect the veracity of the modeled water yield values of the river basins in different degrees. In order to estimate realistic water yield with the model, a precipitation map best fitting the study area is

required. With respect to this study, the spatial changes of precipitation are not considerable in the research area (the initial precipitations of the river basins ranged between 1000 and 1025 mm/y, see Table 6). Accordingly, we presume that precipitation is not critical to the results that mainly characterized the spatial changes of water yield at pixel or river basin scales.

The CSs of nitrogen retention based on the alternative precipitation are also provided by Table 6. The CSs values of the river basins ranging between -0.01 and -0.11 indicate that precipitation had a negative effect on nitrogen retention and the impact was negligible.

Table 7 demonstrates the effects of using alternative nitrogen retention coefficients (NRCs) to the modeled values of nitrogen retention of the nine river basins. The adjustment of the NRCs of urban area and other construction area resulted in CSs of the river basins all smaller than 1 and ranging between 0.01 and 0.33. In other words, nitrogen retention was not elastic to the NRCs of urban area and other construction area. Among the CSs, the largest three, which were greater than 0.1, related to river basin 1, 2 and 5. This phenomenon was due to the largest urban area proportions of these three river basins (Table 1). Concerning the effects of the change of the cropland NRC, the calculated CSs of the river basins were all smaller than 0.25. This suggests the inelasticity of nitrogen retention to cropland NRC of all the river basins. The largest CS, which was 0.22 and subject to the Chai River Basin (No. 7) was owing to the highest cropland proportion of this river basin (56%, see Table 1). The low CSs values shown by Table 7 suggest that the estimates of nitrogen retention generated by the InVEST water purification model in this study were robust despite the uncertainties with the NRCs of urban area, other construction area and cropland.

Table 6. Sensitivity analyses of input data for water yield and nitrogen retention modeling. The table presents the modeling results of water yield and nitrogen retention with initial and alternative precipitation inputs, the magnitude of result changes following precipitation input changes and the coefficients of sensitivity (CSs) associated with the changes for the nine river basins of the Dianchi Lake Watershed.

River basin number	Precipitation (mm/y)			Water yield(m ³ /ha)				Nitrogen retention (kg/ha)			
	Initial	Alternative	Change percentage	Water yield with initial precipitation	Water yield with alternative precipitation	Change percentage	CS	Nitrogen retention with initial precipitation	Nitrogen retention with alternative precipitation	Change percentage	CS
1	1006	1455	44.6%	147.0	249.0	69.3%	1.55	30.5	30.2	-1.2%	-0.03
2	1010	1594	57.8%	148.5	274.9	85.2%	1.47	32.6	31.8	-2.4%	-0.04
3	1019	1263	23.9%	142.5	193.0	35.4%	1.48	18.2	17.8	-1.7%	-0.07
4	1016	1004	-1.2%	137.4	132.6	-3.5%	2.83	16.0	16.0	0.1%	-0.11
5	1025	808	-21.2%	152.1	106.1	-30.2%	1.43	30.4	30.8	1.4%	-0.07
6	1024	726	-29.0%	120.4	64.2	-46.6%	1.61	10.8	11.1	2.9%	-0.10
7	1007	869	-13.7%	122.9	91.6	-25.5%	1.87	12.3	12.4	0.7%	-0.05
8	1002	969	-3.3%	114.6	106.0	-7.5%	2.28	8.8	8.8	0.3%	-0.09
9	1000	1081	8.1%	114.3	129.5	13.2%	1.63	8.8	8.8	0.0%	-0.01

Table 7. Sensitivity analyses of model parameters for nitrogen retention modeling. The table provides the modeling results of nitrogen retention (kg/ha) with initial and alternative nitrogen retention coefficients (NRCs), the magnitude of result changes following the parameter adjustments and the coefficients of sensitivity (CSs) associated with the adjustments for the nine river basins of the Dianchi Lake Watershed. Nitrogen retention coefficient indicates the capacity of land use types to retain nitrogen, as a percentage of the amount of nitrogen flowing into a cell from upslope (Tallis et al., 2013a).

River basin number	Results with initial parameters	Sensitivity of modeling results to parameters					
	NRCs of urban area and other construction land is 5%, NRC of cropland is 25%	NRC of urban area and other construction area: 5% → 0 (-100%)			NRC of cropland: 25% → 10% (-60%)		
	Nitrogen retention	Nitrogen retention	Change percentage	CS	Nitrogen retention	Change percentage	CS
1	30.54	20.37	-33.3%	0.33	30.12	-1.4%	0.02
2	32.60	24.02	-26.3%	0.26	32.18	-1.3%	0.02
3	18.16	16.76	-7.7%	0.08	17.52	-3.5%	0.06
4	15.97	15.62	-2.2%	0.02	14.53	-9.0%	0.15
5	30.37	25.04	-17.5%	0.18	30.09	-0.9%	0.02
6	10.81	10.63	-1.6%	0.02	9.78	-9.5%	0.16
7	12.27	12.14	-1.1%	0.01	10.68	-12.9%	0.22
8	8.81	8.70	-1.2%	0.01	7.91	-10.2%	0.17
9	8.80	8.57	-2.7%	0.03	8.21	-6.7%	0.11






5. Conclusions

The objective of this study was to quantify and visualize multiple ecosystem services of the Dianchi Lake Watershed and to investigate the influences of human systems on ecosystem service flows. The study applied a hypothesis-test methodology, which was based on expert experience and statistical analyses with real data, to explore the influential factors. The test results were not satisfactory and the confirmed socioeconomic influences mainly related to land use structure. The essential reason for the large number of invalid tests was the inaccurate values of some socioeconomic measures, which were downscaled from the respective districts or counties.

Based on the modeling results of nitrogen retention and water yield, we provided empirical evidence to the differentiation of ecosystem service potential, flow and demand and conducted sensitivity analysis to treat the uncertainties in ecosystem service modeling. The spatial and quantity relationships of nitrogen retention, export, load and retention potential indicated the importance of the spatial match issue of ecosystem service demand, potential and flow in regional ecosystem service management. The sensitivity analyses suggested the relative reliability of the modeling results with respect to the tested input data and parameters for the models. The hypothesis-test methodology can facilitate the investigation of the influences of human systems on the spatial changes of regional ecosystem service flows in the face of socioeconomic data scarcity and uncertainty. Having said this, we suggest taking a high preference in the exact match of spatial scale of ecosystem services information and socioeconomic data when exploring human systems' influences in future empirical studies.

Appendices

Table A-1. Correlations of the five ecosystem services with socioeconomic indicators in the nine river basins of the Dianchi Lake watershed. The nine river basins are sample cases. No * means Spearman correlation; * denotes Spearman partial correlation with elevation as the control variable; ** denotes Spearman partial correlation with temperature and shortwave radiation as the control variables.

 Significant positive with $P < 0.05$	 Significant positive with $0.05 < P < 0.1$
 Significant negative with $P < 0.05$	 Significant negative with $0.05 < P < 0.1$
 $P > 0.1$	

Ecosystem service indicators		Food supply*	NPP	Habitat quality	Water yield**	Nitrogen retention**
		yuan/km ²	gC/m ² /yr	relative value	m ³ /ha	kg/ha
Cropland proportion	%	0.705	-0.483	-0.200	0.081	0.145
Forest proportion	%	-0.466	0.667	0.850	-0.792	-0.674
grassland proportion	%	-0.512	-0.151	0.234	-0.164	-0.522

Rural residential area proportion	%	-0.023	-0.133	-0.700	0.936	0.904
Urban area proportion	%	-0.163	0.250	-0.333	0.734	0.645
road density	km/km ²	0.119	0.617	-0.200	-0.288	0.122
Rural residential area population density	persons/km ²	0.132	0.162	0.366	-0.756	-0.541
Urban population density	persons/km ²	-0.491	0.247	0.145	-0.451	-0.492
Total population density	persons/km ²	-0.262	0.233	-0.317	0.367	0.469
Chemical fertilizer use	tonnes/km ²	0.072	0.281	-0.451	0.612	0.838
Nitrogen fertilizer use	tonnes/km ²	-0.252	0.264	-0.281	0.317	0.443
Phosphorus fertilizer use	tonnes/km ²	0.072	0.281	-0.451	0.612	0.838
Potassium fertilizer use	tonnes/km ²	0.575	-0.315	-0.264	0.595	0.473
Compound fertilizer use	tonnes/km ²	0.495	-0.247	-0.281	0.645	0.570
Rural electric power consumption	10 ⁴ kWh/km ²	0.685	-0.758*	-0.434	0.650	0.404
Cropland effective irrigation rate	%	-0.394	0.638	0.468	-0.638	-0.448
Forest and garden irrigation rate	%	0.048	-0.145	-0.230	0.581	0.209
Total agricultural machinery power	10 ⁴ W/km ²	0.465	0.145	0.128	0.189	0.365
GDP per person	10 ⁴ yuan/person	-0.353	0.283	0.317	-0.820	-0.830
Primary sector output	10 ⁴ yuan/km ²	0.637	-0.536	-0.349	0.634	0.416
Secondary sector output	10 ⁴ yuan/km ²	-0.034	0.809**	0.060	-0.383	0.000
Tertiary sector output	10 ⁴ yuan/km ²	-0.143	0.621	0.332	-0.608	-0.407

Table A-2. The time, scales and sources of the data used in this study

Data	Time	Scale	Publisher/source
Land use map	2010	1:100,000, Kunming	Produced by the interpretation of 30-meter Landsat thematic mapper (TM) data in combination of a 1:100,000 topographic map of the research area
DEM	—	30 m×30 m, Dianchi Lake Watershed	Geospatial Data Cloud, Chinese Academy of Sciences, http://www.gscloud.cn/ (accessed on 5.02.2014)
Precipitation	1950-2000 average annual; 2010 total	1 km×1 km, Global; 10 km×10 km, China	Chen et al. (2011); He and Yang (2011); Worldclim-Global Climate Date, http://worldclim.org/current (accessed on 5.02.2014)
Potential evapotranspiration	1950-2000 average annual	1 km×1 km, Global	1950-2000 annual average potential evapotranspiration, http://www.cgiar-csi.org/data (accessed on 5.02.2014)
Temperature	1950-2000 average annual	1 km×1 km, Global	Worldclim-Global Climate Date, http://worldclim.org/current (accessed on 5.02.2014)
Shortwave radiation	2010 total	10 km×10 km, China	Chen et al. (2011); He and Yang (2011)
Soil depth, plant available water content	—	1 km×1 km, China	Harmonized World Soil Database, http://webarchive.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/ (accessed on 5.02.2014)
Soil total Nitrogen content	—	1:1,000,000, China	Soil and Terrain Database Soil for China primary data (version 1.0), http://www.isric.org/data/soil-and-terrain-database-china (accessed on 5.02.2014)
Root depth and evapotranspiration coefficient	—	Different land use types	Bagstad et al., 2012, pp 73; Canadell et al., 1996; FAO, 2014; Jiang, 1980; Statistics Bureau of Kunming, 2011, crop types of Kunming; Tallis et al., 2013a, pp 275
Nitrogen load	—	Different land use types	China Pollution Source Census, 2009; Gao et al., 2013; Leh et al., 2013; Lin, 2004; Zhang et al., 2012; Zhang et al., 2013; Zhao et al., 2012
Vegetation nutrients filtering value	—	Different land use types	Leh et al., 2013; Tallis et al., 2013a; Tallis et al., 2013b; Wu et al., 2010

Net primary production raster map for vegetation covers	2010	1 km × 1 km, global	Numerical Terradynamic Simulation Group, 2013
Net primary production of urban area	2006	Shanghai city district	Lin et al., 2008
Food output	2010	Districts and counties, Kunming	Statistics Bureau of Kunming, 2011
Road network	2007	China	National Administration of Surveying, Mapping and Geoinformation of China
Data for other socioeconomic indicators used in this article	2010	Districts and counties, Kunming	Statistics Bureau of Kunming, 2011

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Supplementary materials

1. Quantifications and mappings of food productivity, NPP and habitat quality of the six district/counties of Kunming region

The six district or counties (Wuhua, Panlong, Guandu, Chenggong, Jinning, Xishan) cover the nine river basins in this study (see Figure 1 in this article). The methods to quantify and map food productivity, NPP and habitat quality are described below (after Hou et al., under review).

Food productivity

The food productivity of the six districts or counties of Kunming region was computed in RMB yuan/km² for each land use type. Statistical data of crop types and outputs (cereals, oil crops, cane, tobacco, vegetables fruits, nuts, spices and Chinese herbal medicine), sheep and caws, poultries as well as aquaculture products of different districts or counties for 2010 were used. Then we calculated the average values of food output per km² (food productivity) of cropland, forest, grassland, rural residential area and lakes of the associated district or county (for data sources see Table A-2 in the appendix in the article). Pig production was not considered as pig farms cannot clearly be classified into a certain land cover type.

Based on the above described calculations, we linked the food productivity values of different land use types to the land use spatial data with ArcGIS. Accordingly, the map of food productivity of the study area was produced.

NPP

The quantification NPP was divided to three processes based on different data sources and published work. For vegetation NPP, the MOD17A3 NPP product produced by the Numerical Terradynamic Simulation Group of the University of Montana was used (see Table A-2 in the appendix in the article for data sources). The NPP value of urban area was modified from Lin et al. (2008). The NPP of the Dianchi Lake was assessed referring to Sun et al. (2011) and Chinnaiah and Madhu (2010).

The MOD17A3 NPP product is generated from NASA MODIS data by the Numerical Terradynamic Simulation Group. It provides continuous estimates of NPP across the earth's entire vegetated land surface. The NPP is provided at 8-day, monthly, and annual time steps in GeoTIFF of 1km× 1km. For this research we used the annual global NPP of 2006 and extracted the six districts or counties of Kunming region from the global map. Lin et al. (2008) estimated the NPP of the urban area of Shanghai, China based on remote sensing and urban forest inventory data of 2006. As the climates of Kunming and Shanghai are similar to each other, we transferred the NPP value of Shanghai city from Lin et al. (2008) to the Kunming urban area. Sun et al. (2011) measured the surface water gross primary production (GPP) of the northern, middle and southern parts of the Dianchi Lake from the winter of 2009 to the spring of 2010. Chinnaiah and Madhu (2010) estimated the NPP/GPP ratio from October 2009 to September 2010 for the Darmaagar Lake in India. Our study adapted the results of these two studies to estimate the NPP values for the parts of the Dianchi Lake that belong to the six districts of the research area. For the calculation, GPP values of spring from Sun et al. (2011) were used to calculate the annual average GPP of the Dianchi Lake. The mean

NPP/GPP ratio for the six spring months from Chinnaiah and Madhu (2010) was used to convert GPP to NPP.

By combining the NPP maps derived from the above described methods using the union tool of ArcGIS 10, we derived the NPP map for the entire area of the six districts of Kunming region.

Habitat quality

Habitat quality can represent ecosystem's capability to support biodiversity. It involves the ability of the ecosystem to provide conditions appropriate for individual and population persistence (Tallis et al., 2013a). Habitat quality modeling can be a practical approach to reflect the overall potential of regional biodiversity. One promising modeling tool is the InVEST biodiversity model, which combines information on land use and land cover and threats to biodiversity to produce habitat quality maps (Tallis et al., 2013a). For the details of the model introduction, working principles, data needs, results interpretations as well as model limitations, see the InVEST 2.5.5 User's Guide (Tallis et al., 2013a).

The study of Hou et al. (under review) employed the InVEST biodiversity model to estimate and map the habitat quality of the six districts or counties of Kunming region. A land use map rasterized from a 1:100,000 shapefile map of 2010 was used and cropland, urban and rural residential area, construction area as well as roads were assigned as threats. The biophysical table was adapted from the sample data of the InVEST biodiversity model with changes based on the land use classifications of the research area. The modeling results are relative values between 0 and 1. On the habitat quality map areas with higher values enjoy better habitat quality compared to the rest of landscape.

2. Working principles of the InVEST water yield and water purification (nutrient retention) models (after Leh et al., 2013; Tallis et al., 2013a)

Water yield model

The model uses average annual precipitation P_x , annual reference evapotranspiration, soil depth, plant root depth, plant available water content and land use characteristics to calculate the average annual water yield (Y_{xj}) in each pixel as :

$$Y_{xj} = \left(1 - \frac{AET_{xj}}{P_x}\right) \cdot P_x \quad (1)$$

Where, AET_{xj} is the annual actual evapotranspiration on pixel x with LULC $_j$ and P_x is the annual precipitation on pixel x .

The evapotranspiration partition of the water balance, $\frac{AET_{xj}}{P_x}$, is an approximation of the Budyko curve developed by Zhang et al. (2001) as follows:

$$\frac{AET_{xj}}{P_x} = \frac{1 + w_x R_{xj}}{1 + w_x R_{xj} + \frac{1}{R_{xj}}} \quad (2)$$

Where, w_x is a modified dimensionless ratio of plant accessible water storage to expected precipitation during the year calculated as:

$$w_x = Z \frac{AWC_x}{P_x} \quad (3)$$

Where AWC_x is the volumetric (mm) plant available water content. The soil texture and effective soil depth define AWC_x . Z is a seasonality factor that presents the seasonal rainfall distribution and rainfall depths.

R_{xj} is the dimensionless Budyko Dryness index on pixel x with LULC j , defined as the ratio of potential evapotranspiration to precipitation (Budyko, 1974) as follows:

$$R_{xj} = \frac{k_{xj} \cdot ET_{o_x}}{P_x} \quad (4)$$

Where, ET_{o_x} is the reference evapotranspiration from pixel x and k_{xj} is the plant (vegetation) evapotranspiration coefficient associated with the LULC j on pixel x .

The water yield model script generates and outputs the total and average water yield at the sub-basin level.

Water purification (nutrient retention) model

The model estimates the quantity of nutrients retained for water purification from a landscape and the amount of nutrients exported from the landscape and reaches the streams in four steps.

It firstly calculates annual average runoff from each pixel using the same method as the water yield model. Then the model estimates the average annual amount of nutrients exported from each pixel based on nitrogen (N) and phosphorus (P) exports coefficients the user inputs for each LULC category with the following formula. A Hydrological Sensitivity Score is included to adjust the loading value of nutrients at each pixel.

$$ALV_x = HSS_x \cdot pol_x \quad (5)$$

where ALV_x is the Adjusted Loading Value of the nutrients at pixel x , pol_x is the nutrients export coefficient at pixel x , and HSS_x is the Hydrologic Sensitivity Score at pixel x which is calculated as:

$$HSS_x = \frac{\lambda_x}{\bar{\lambda}_W} \quad (6)$$

Where λ_x is the runoff index at pixel x , calculated using the following formula, and $\bar{\lambda}_W$ is the mean runoff index in the respective watershed.

$$\lambda_x = \log(\sum_U Y_u) \quad (7)$$

Where $(\sum_U Y_u)$ is the sum of the water yield of pixels along the flow paths above pixel x and the water yield of pixel x itself.

In the third step, the quantity of nutrient load retained by the landscape is calculated using the nutrient retaining potential of each LULC type as:

$$\text{Retention} = ALV_x \times \text{filtration} \quad (8)$$

where filtration is the nutrient retention potential of LULC_j (e.g., the nitrogen retention coefficient in percentage in Table B-1 for this study).

In the final step the model aggregates the loading that is retained by the landscape and that reaches the stream from each pixel to the sub-watershed then to the watershed level.

Table B-1. Biophysical table for the modeling of water yield and nitrogen retention. The plant evapotranspiration coefficient for each LULC class (ETcoeff) (Bagstad et al., 2012; FAO, 2014; Jiang, 1980; Statistics Bureau of Kunming, 2011; Tallis et al., 2013a; Tallis et al., 2013b), Root depth (Bagstad et al., 2012; Canadell et al., 1996; Tallis et al., 2013a; b), nitrogen load coefficient (China Pollution Source Census, 2009; Gao et al., 2013; Leh et al., 2013; Lin, 2004; Tallis et al., 2013b; Zhang et al., 2012; Zhang et al., 2013; Zhao et al., 2012) and nitrogen retention coefficient (Leh et al., 2013; Tallis et al., 2013a; b; Wu et al., 2010) of LULC classes.

LULC classes	ETcoeff	Root depth (mm)	Nitrogen load (kg/ha/yr)	N Retention coefficient (%)
Forest	1	7000	2.38	78
Cropland	0.65	2100	14.95	25
Grassland	0.65	2600	8.00	40
Urban area	0.001	300	225.77	5
Rural residential area	0.3	500	33.43	10
Other construction area	0.001	1	0.01	5

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Chapter 5. Uncertainties in landscape analysis and ecosystem service assessment

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Journal of Environment Management, 2013, 127, Supplement: S117-S131

Abstract

Landscape analysis and ecosystem service assessment have drawn increasing concern from research and application at the landscape scale. Thanks to the continuously emerging assessments as well as studies aiming at evaluation method improvement, policy makers and landscape managers have an increasing interest in integrating ecosystem services into their decisions. However, the plausible assessments carry numerous sources of uncertainties, which regrettably tend to be ignored or disregarded by the actors or researchers. In order to cope with uncertainties and make them more transparent for landscape managers, we demonstrate them by reviewing literatures, describing an example and proposing approaches for uncertainty analysis. Additionally, we conclude with potential actions to reduce the insecurities accompanying landscape analysis and ecosystem service assessments.

As for landscape analysis, the fundamental uncertainty origins are landscape complexity and methodological uncertainties. Concerning the uncertainty sources of ecosystem service assessments, the complexity of the natural system, respondents' preferences and technical problems play essential roles. By analyzing the assessment process, we find that initial data uncertainty pervades the whole assessment and argue that the limited knowledge about the complexity of ecosystems is the focal origin of uncertainties. For analyzing uncertainties in assessments, we propose systems analysis, scenario simulation and the comparison method as promising strategies. To reduce uncertainties, we assume that actions should integrate continuous learning, expanding respondent numbers and sources, considering representativeness, improving and standardizing assessment methods and optimizing spatial and geobiophysical data.

Keywords: Uncertainty, complexity, landscape management, ecosystem service assessment, model, scale

1. Introduction

Uncertainty is not only an unfriendly characteristic but also a universal feature of human-environmental systems. It is strongly related with insecurity and therefore provokes an enormous scientific motivation to make things more certain and thereby - hopefully - more secure. But regrettably this objective has some harsh limitations: Realizing that overall certainty can hardly be attained, a Chinese proverb says that ‘to be uncertain is to be uncomfortable, but to be certain is to be ridiculous.’ Underlining this significance of uncertainty, already Pliny the Elder (23 AD-79 AD) has formulated the frustrating wisdom that ‘the only certainty is that nothing is certain’. Bertrand Russell (1949/2011) has illuminated these circumstances with some more optimism: ‘When one admits that nothing is certain one must, I think, also admit that some things are much more nearly certain than others.’ That is an encouraging statement especially for scientists, because their basic vocation is to reduce uncertainty wherever this is possible.

In landscape ecology, several origins of uncertainty are cumulating: referring to the European Landscape Convention¹⁰, ‘landscape means an area, as perceived by people, whose character is the result of the action and interaction of natural and/or human factors’. Thus, besides the subjectivity of the perceiving persons, two very different origins of uncertainty are set into interaction. Their number even rises, when we take a look at the comprehension of landscape science: The International Association of Landscape Ecology IALE¹¹ defines landscape ecology as ‘the study of spatial variation in landscapes at a variety of scales. It includes the biophysical and societal causes and consequences of landscape heterogeneity’.... ‘Landscape ecology can be portrayed by several of its core themes:

- the spatial pattern or structure of landscapes, ranging from wilderness to cities,
- the relationship between pattern and process in landscapes,
- the relationship of human activity to landscape pattern, process and change,
- the effect of scale and disturbance on the landscape.’

All of these features contribute with distinct sources of uncertainty. And as we include the human dimension, all decision-based insecurities of societal and psychological processes have to be added. Therefore, it is not surprising that also resource management - the focal topic of this special issue - has to cope with a very high degree of uncertainty: major challenges of resource management are interrelated with extremely uncertain dynamics, such as globalization, demographic change and migration, climate change, strategies for sustainable provision of energy or motivations for conservation of natural resources and ecosystem services. Consequently, there is an overwhelming variety of insecurity correlated with all landscape analyses and landscape management activities, and in this context several questions can be asked:

- How do the sources of uncertainty influence landscape analysis and landscape modeling?
- Which sources of uncertainty can be assigned to the assessment of ecosystem services?
- How can we cope with these uncertainties in landscape science and management?

Consequently, the focal objective of this paper is to describe the origins and features of uncertainties in landscape analysis and ecosystem service assessment. After this introduction we

¹⁰ See http://www.coe.int/t/dg4/cultureheritage/heritage/landscape/default_en.asp

¹¹ See http://www.landscape-ecology.org/what_is.html

ask for the primary sources of uncertainty in landscape analysis and modeling. Thereafter we present an overview of special uncertainty sources referring to ecosystem service assessments from a general point-of-view and from the practical aspect of the land cover/land use- based assessment concept of Burkhard et al. (2009, 2012b). Following this demonstration we will discuss potential strategies to reduce uncertainty and to make it more transparent. Finally we will conclude with a summarizing outlook.

2. Key sources of uncertainty in landscape analysis and modeling

2.1. Introducing uncertainty

The concept of uncertainty has attracted several efforts especially in the field of philosophy of science (e.g. Halpern, 2003; Popper, 1979). From an environmental perspective, Funtowicz and Ravetz (1990) refer to uncertainty as the integration of error, inexactness, unreliability and ignorance. Li and Wu (2006) describe uncertainty as the inadequacy of people's knowledge to understand the system under investigation, and Breckling and Dong (2000) state that a subject is uncertain when it is either not exactly known or determined or reliably foreseen. Following these authors, predictability and reproducibility are the primary measures of uncertainty, but also irregularity, variability, or the appearance of singularities can be used to describe the uncertainty of ecological phenomena. In Petersen's distinction of uncertainty components (2000), inexactness implies significant digits and error bars, unreliability refers to levels of confidence and quality, and ignorance is a form of people's unawareness of what they do not know. In this article, we understand uncertainty as an analytical state of limited knowledge which aggravates the exact depiction of a system's current situation or the future outcomes of the system's development. In Online Appendix 1 we have summarized several general sources of uncertainty which are based on the principle conceptions of systems analysis and which therefore can appear in any sort of system.

In many elaborations, uncertainty is related to the concept of risk, as a consequence of action or inaction, in planning or in ecotoxicological valuations (Li and Wu, 2006). Risk is a measurable derivation of uncertainty, which possibly leads to undesired effects or significant loss (Knight, 1921). It is calculated from the magnitude of the potential loss and the probability that the loss will occur and it implies probabilities and consequences that can be uncertain themselves (Li and Wu, 2006).

2.2. Uncertainty in landscape analysis

Undoubtedly, the drivers of uncertainty listed in Online Appendix 1 also effect landscape analysis, because the study object is an extremely complex system (Hay et al., 2002), both in structure and process (Müller and Li, 2004). Resulting from spatial variations in the physical template, the landscape matrix generally presents a high heterogeneity, and the temporal variation of natural driving forces leads to a strong time-dependence of most landscape features. Moreover, disturbances may increase the uncertainty of landscape structures (Fürst et al., 2010). As many landscape features appear in continuous gradients, it is difficult to simplify and classify landscapes by distinguishing discrete features from one another (Bolliger and Mladenoff, 2005). Landscape processes tend to be complicated at the interfaces of different natural systems, due to

the interaction among contagious or neighboring processes at the connected areas (Hollenhorst et al., 2006; Peters et al., 2006), making an exact landscape assessment difficult.

Besides the origins listed above, landscape complexity is linked with methodological uncertainties that arise from two essential sources: i) scale mismatches between landscape structures or processes and the spatial data, ii) land cover classification uncertainty.

Owing to the fast development of remote sensing and GIS technology, the availability and accuracy of spatial data have been increasing significantly during the last decades. However, satellite-derived land cover data should be used carefully when addressing fine-scale landscape structures, especially when the scale of landscape processes is below the spatial resolution of the thematic spatial data (Hollenhorst et al. 2006). Insufficiency of accurate land cover data and uncertainty in the product of image classification are from landscape simplification and land cover misclassification (Álvarez-Martínez et al., 2010, 2010; Woodcock and Gopal, 2000). According to Zhu (1997) and Bolliger and Mladenoff (2005), land cover classification is associated with two kinds of uncertainty: (i) commission or exaggeration, which refers to the fuzzy assignment of a landscape entity to a set of classes, and (ii) omission or ignorance, which reflects the deviation of a certain landscape entity from the class to which it is prescribed. In the heterogeneous and changing landscape, the boundaries are uncertain because the transition from one landscape type to another is gradual. This leads to the loss of information and a variety of uncertainties while the aggregation process is carried out (Bolliger, 2005; Strand, 2011). Furthermore, when assigning a particular land cover category to an image pixel, this will also cause a certain degree of confusion (Álvarez-Martínez et al., 2010).

Uncertainty in landscape analysis affects the quality, sufficiency and reliability of landscape models' results. Some practical studies conclude that the land cover classification uncertainty substantially hinders the determination of landscape types, landscape mapping and landscape pattern analysis (Canters et al., 2002; Foody, 2002; Hollenhorst et al., 2006; Shao and Wu, 2008; Álvarez-Martínez et al., 2010). In addition, defective classification products result in misdetections of land cover dynamics, inaccuracy of landscape patterns and ambiguity of landscape processes (Álvarez-Martínez et al., 2010).

2.3. Uncertainty in landscape modeling

Landscape models are used as abstractions of reality for the survey of complex regional systems to understand and describe landscape dynamics and to represent landscape pattern-process relationships (Turner et al., 2001; Wainwright and Mulligan, 2004). Landscape modeling is regarded as an important method in landscape ecology and widely used in landscape analysis (Turner, et al., 2001; Paudel and Yuan, 2012). Nevertheless, the massive uncertainties accompanying model building and application should draw the full attentions of the modelers and users.

Uncertainties in landscape modeling essentially derive from the general limitations of models. According to thematic assumptions, model uncertainties are based on model structure, model input and natural variability (Heuvelink, 1998; Jansen, 1998; Katz, 2002; Klepper, 1997; Li and Wu, 2006). Structural errors constitute the basic uncertainty of models, resulting from assumptions, simplifications and formulations (Heuvelink, 1998; Li and Wu, 2006). Model input parameters are uncertain because of knowledge limitation about the parameters and estimations,

calibration errors (Li and Wu, 2006), unknown data quality, high natural variability of the system and missing data.

Besides the uncertainties due to the essential weakness of models, uncertainties pervading the landscape analysis process (landscape complexity-based and methodological uncertainties) also influence landscape modeling.

Moreover, uncertainties in landscape modeling are closely linked with spatial and temporal scaling. Scaling is the transfer of information across spatial, temporal or organizational scales (Bloschl and Sivapalan, 1995; Li and Wu, 2006). The basic theoretical source of scaling uncertainty derives from the comprehension of hierarchy theory (Allen and Starr, 1982; O'Neill et al., 1986) which proposes that each ecological process operates with individual spatial and temporal characteristics (Müller, 1992). Consequently, each scale transfer is linked with a loss of accuracy and this has strong effects on the reliability of the landscape model results.

Spatial scaling uncertainties in landscape modeling derive from two prominent sources: scale mismatches in the model or data, and the heterogeneity of the natural system in space (Katz, 2002; Li and Wu, 2006). When a landscape model is applied at a scale different from the one for which it has been designed, it may end up losing adequacy (Rykiel, 1996; Heuvelink, 1998; Katz, 2002; Li and Wu, 2006). One important spatial scaling method is spatial interpolation. Although it is widely used in landscape modeling and visualization, the uncertainties it brings should not be ignored. From technical perspective, spatial interpolation uncertainties are due to different basic data statistics (Tao et al., 2009) (e.g. Kriging interpolation, based on best local estimates, leads to approximation errors by smoothing out local details of the spatial variation, Xu et al., 2005). Additionally, landscape model predictions through time remain uncertain due to the temporal dynamics of landscape and the dynamics of the variables impacting land cover and land use (Álvarez-Martínez, 2011). Although the well-built landscape models are good at explaining landscape changes over the historical period, they are unable to conquer uncertainties when forecasting changes beyond the observed period for which they were built (Wear and Bolstad, 1998). Landscape changes are driven by multiple social-economic drivers. The complex human system and human-environment system interactions hinder the accurate drawing of land use scenarios for the long run (Wear and Bolstad, 1998; Price et al., 2012)

3. Uncertainty in ecosystem service assessments

Due to their high complexity and their integrative position between human and environmental systems, ecosystem service assessments contain outstanding degrees of uncertainty and generalization (Scolozzi et al., 2012). Seppelt et al. (2011, 2012) provide quantitative reviews of 153 ecosystem service studies taken from current scientific publications. About 45% of these studies did not give sufficient information concerning their results' uncertainty. Insufficient information on uncertainty constitutes serious problems (Bateman et al., 2010; Farley, 2008; Nicholson et al., 2009) because acknowledgments and explicit considerations of uncertainty are indispensable to understand the pattern- and process-based ecosystem services (Díaz et al., 2006; Nicholson et al., 2009). Additionally, when investigating stakeholders' preferences, uncertainty may lead to systematically biased results and a false precision of ecosystem valuation. As policy makers make their decisions based on the available information, it is crucial to evaluate and minimize uncertainty to avoid bias or even fault in decision making (Chavas, 2000; National Research Council, 2005).

We can distinguish three essential sources of uncertainty in ecosystem service assessments (e.g. Kumar, 2010). Firstly, uncertainty may occur while natural systems (with high degrees of uncertain functionalities and dynamics) deliver ecosystem services. Secondly, people's preferences about ecosystem services are uncertain, that is, their subjective judgments may differ with the knowledge and information they obtain. Thirdly, we may face uncertainties when evaluating ecosystem services by using certain tools. In the following section, we will discuss these three sources by integrating relevant assumptions from different literature sources.

- *Natural supply uncertainty*: Because of the complexity of ecosystem functions, the specific roles of many species in ecosystem service provision remain uncertain (Daily, 2000; Schneiders et al., 2012). This results in a lack of robust information and adequate understanding of what contributions biodiversity brings to ecological functions (Pascual et al., 2010). The uncertain relationship between ecological structures, function, and processes together cause uncertainties in the generation of ecosystem goods and services (National Research Council, 2005; Johnson et al., 2012).
- *Preference uncertainty* is associated with subjective decisions of respondents and the differentiation of levels of knowledge and experience about the ecosystem services to be assessed (Kumar et al., 2010; Spangenberg and Settele, 2010). A typical example of preference uncertainty involves the method of willingness to pay (WTP) to value ecosystem services (National Research Council, 2005; Pascual et al. 2010).

As different services are often valued with different methods, and consequently with different results, one question is how to conceive the compatibility of these techniques: Is it correct to add (monetary) values which have been based upon different preference identification techniques (Spangenberg and Settele, 2010). Another basic question can be asked for the degree of indirectness of valuation conceptions: Is it correct to value a direct use benefit with the same dimension and significance as the benefit of potential future beneficiaries or the good feeling about the existence of a valued item which the valuator will never experience?

- *Technical uncertainty* refers to the application of different methods to assess ecosystem services. The divergence of the methods used to calculate ecosystem service values leads to non-robust and ambiguous results. When these values are aggregated, they will result in a dubious total value (Spangenberg and Settele, 2010; Johnson et al., 2012). For example, applications of WTP (willingness to pay) and WTA (willingness to accept) to the same object of measurement will result in large disparity between the respective values (Kumar et al., 2010). Each specific method carries weaknesses, which are responsible for uncertain evaluation results. To take evaluation with models as an example, a lack of mechanistic understanding of system dynamics or processes may lead to ignorance of errors emerging in the models (Nicholson et al., 2009; Regan et al., 2002). Another example refers to economic discounting, which is conventionally applied to deal with uncertainties in climate change and biodiversity loss (Gowdy et al., 2010). Economic discounting is a promising technique to conduct trade-offs between the present and the future, by weighting the sequence of costs or benefits over time (Carpenter et al., 2006; Ludwig et al., 2005). In the discounting process, the choice of discount rates plays an important role for long-term decision making (TEEB 2008). However, the uncertainty of the process delivering future discounting rates causes uncertain discount rate models, which lead to diverging long run valuation results (Ludwig et al., 2005).

Being the predominant and inherent uncertainties for ecosystem services, natural supply uncertainties remain the origin for the uncertainties regarding preference and assessment methods. To have an insight into some components of this essential uncertainty, Table 1 presents a list of corresponding uncertainty sources for 12 major ecosystem services.

Table 1. Source examples for impact-based uncertainties in the generation of different ecosystem services, after Elmqvist et al. (2010).

Ecosystem services		Source examples
1	Food provision	<ul style="list-style-type: none"> – Uncertain outcome of interactions including an increase in agricultural production, shifting land use, climate change, etc. – Poor understanding of production effects to the ecosystems processes
2	Water provision, flows and purification	<ul style="list-style-type: none"> – Small changes of ecosystem capacities may cause sudden shifts for the services (e.g. eutrophication and water quality)
3	Fuels and fibers	<ul style="list-style-type: none"> – Fragmentation may cause a disproportional decline in the provision of timber, fibers and fuel wood – Climate change contributes to forest fire risk
4	Genetic resources	<ul style="list-style-type: none"> – Minor change in natural habitat, climate, or agricultural lands may cause incommensurate loss in genetic diversity
5	Medicinal and other biochemical resources	<ul style="list-style-type: none"> – In highly diverse regions, habitat destruction processes may result in an abrupt change in biochemical sources
6	Air quality regulation and other urban environmental quality regulation	<ul style="list-style-type: none"> – Isolation and fragmentation in the urban landscape influence the generation of air quality services – Rapid turnover of species affect important ecological functions for services
7	Climate regulation	<ul style="list-style-type: none"> – Large time lags of feedbacks between changes in ecosystem processes and the atmosphere – CO₂ absorbing capacity tends to be exceeded by CO₂ emissions
8	Moderation of extreme events, erosion prevention	<ul style="list-style-type: none"> – Effect of ecosystems on hazard mitigation is still poorly understood
9	Maintenance of soil quality	<ul style="list-style-type: none"> – Soil fertility declines, when expanded agricultural land is not specifically suitable for agriculture
10	Pollination services	<ul style="list-style-type: none"> – Habitat loss may lead to collapse of important pollinators – Climate change may provoke phenological shifts and asynchrony of plant-pollinator interactions

11	Biological control	<ul style="list-style-type: none"> – Non-linearity and the biological control services – Passing of inflection points in natural enemy diversity may lead to disproportionate decline of biological control functions
12	Cultural services (tourism services)	<ul style="list-style-type: none"> – Abrupt changes in the provision of tourism services, e.g. destroying of picturesque landscapes by fires – War, terrorism, socio-political disruption, natural disasters and health crises

4. Uncertainties in spatially explicit ecosystem service assessments

Besides the uncertainties in ecosystem service assessments mentioned above, additional and significant sources of uncertainty emerge when conducting spatially explicit ecosystem service assessments. The most prominent uncertainties in spatial ecosystem service assessments are related to the information of land cover/land use data (LCLU), which in most cases provide the basic spatial units. Further uncertainties are linked to missing data (Costanza et al., 1997; Koschke et al., 2012), scale issues, generalizations, specific methodological weaknesses, and imprecise function/service classifications or definitions (Scolozzi et al., 2012). Nevertheless, there is a clear demand for spatially explicit ecosystem service evaluations from landscape management and planning (Daily and Matson, 2008; Kienast et al., 2009) and several approaches have been developed on different scales, from regional (e.g. Nedkov and Burkhard, 2012; Vihervaara et al., 2010) to continental (Haines-Young et al., 2012; Maes et al., 2011), with different degrees of uncertainty.

Based on the example of land cover-based ecosystem service supply assessments (Burkhard et al., 2009, 2012b), technical as well as thematic uncertainties will be discussed in the following chapter, related to the subsequent steps of the respective analyses. The assessment approach starts with a matrix, linking different land cover types to ecosystem service supply capacities. Respective capacities are ranked on a relative scale, ranging from 0 (indicating no relevant capacity of the land cover type to provide this particular ecosystem service) up to 5 (maximum relevant capacity), referring the ecosystem goods and services provided in a particular area and actually used within a given time period (Burkhard et al., 2012b).

Based on the 0-5 rankings, ecosystem service supply capacities can be linked to spatially explicit units (e.g. land cover types) and finally, ecosystem services can be mapped. Table 2 gives an overview of the typical steps of land cover-based ecosystem service supply assessments and mentions related technical as well as thematic uncertainties, which are discussed in more detail in the following.

Table 2. Overview of typical steps of land cover-based ecosystem service supply assessments and the related uncertainties.

	Assessment step	Technical uncertainties	Thematic uncertainties
1	Selection of study areas	<ul style="list-style-type: none"> • Information availability and data access 	<ul style="list-style-type: none"> • Representativeness • Specific local natural and cultural settings • Definition of reference

			<p>(baseline) states</p> <ul style="list-style-type: none"> • Future scenario and management measure vagueness • Changing system conditions (e.g. climate change, social dynamics)
2	Selection of relevant land cover/land use classes (assessment matrix y-axis)	<ul style="list-style-type: none"> • Generalization and categorization of complex landscapes into a limited number of land cover/land use classes 	<ul style="list-style-type: none"> • Aggregation/differentiation of land cover vs. land use types, also depending on spatial data resolution and study area • Different landscape scientific approaches
3	Spatial land cover/land use and biophysical data acquisition	<ul style="list-style-type: none"> • Inaccuracies in spatial data • Suitability of spatial and temporal scales 	<ul style="list-style-type: none"> • Inaccuracies in thematic data
4	Selection of relevant ecosystem functions and services (assessment matrix x-axis) and respective indicators	<ul style="list-style-type: none"> • Appropriate indicandum-indicator relations • Indicators which are robust, scalable and sensitive to changes 	<ul style="list-style-type: none"> • Which ecosystem functions and services are really relevant in the case? • Various indicators are needed for service trade-off assessments
5	Quantification of ecosystem function and service indicators	<ul style="list-style-type: none"> • Mismatches of geobiophysical data and statistical data spatial units • Lack of appropriate data for quantifications • Model, measurement and statistical data quantification uncertainties 	<ul style="list-style-type: none"> • Limited knowledge about complex ecosystem functionalities • Indication of ecosystem functions, regulating and cultural (intangible) services is not well-developed
6	Interlinking land cover/land use classes and ecosystem functions/services in the assessment matrix using the relative 0-5 capacity scale	<ul style="list-style-type: none"> • Comparability of different data • Averaging and potential double-counting of ecosystem functions/services over space and time (i.e. weighting system needed) • Integration of data of varying quality and quantity as well as expert assessments • Spatially heterogeneous ecosystem service supply 	<ul style="list-style-type: none"> • Aggregation of data, models and indicators without losing relevant information • Ecosystem service supply potential vs. supply capacity, what to quantify? • Subjectivity in scoring procedures
7	Linkage of assessment values to spatial biophysical and/or	<ul style="list-style-type: none"> • Lack of appropriate biophysical data (soils, hydrology, vegetation, etc.) 	<ul style="list-style-type: none"> • Limited knowledge about complex human-environmental linkages,

	administrative units (mapping)	<ul style="list-style-type: none"> • Mismatches of spatial units • GIS software problems • Map layout issues might cause interpretation problems 	service providing units and ecosystem service flows <ul style="list-style-type: none"> • Difficulties in allocating services to land cover (e.g. for cultural services) • Multiple ecosystem services representation: 2D maps only allow the presentation of one service or averages/sums of services
8	Interpretation of results	<ul style="list-style-type: none"> • Data and mapping shortcomings • Model and map validation • Insufficient end-user interfaces • Data extrapolation to different or larger regions 	<ul style="list-style-type: none"> • Data and map misinterpretation, also due to lacking study area knowledge • Lack of expert knowledge, e.g. concerning interactions between landscape management and ecosystem service supply • Information too complex and too aggregated • Transferability of results to other regions

❖ Assessment step 1: Selection of areas and time spans

The selection of the study area should not only be based on information availability and data access, but also on the area's representativeness for a larger spatial, temporal and/or thematic context in order to allow extrapolations of locally obtained discrete data. Random, biased or wrong selection of study areas or sample sites can lead to non-representativeness (McCarthy et al., 2001). After Fränze et al. (2008), the term "representative" means to adequately reproduce the properties of different phenomena referring to characteristic frequency distributions and specific spatial patterns. Hence, also temporal aspects have to be considered carefully, especially in non-dynamic assessments of environmental states. Static singular surveys are not able to cover changing system conditions, for example land use/land cover changes, climate change effects or socio-economic dynamics. Assessments including historical dynamics as well as future scenario simulations are appropriate approaches to integrate temporal aspects, although they are again linked to uncertainties (Gómez-Sal et al., 2003).

Fig. 1 shows the example of a study site selection. The study area (on the right) was chosen to be representative for a larger area in a typical northern German agriculture-dominated rural landscape. The landscape consists of large agricultural fields, lakes, water courses, forest patches, small settlements and road infrastructure. Although the selected study area contains all of these characteristic elements, it is uncertain whether the area is really representative for all particular geobiophysical settings and the general landscape pattern in the total area, for example regarding

the spatial share of each land cover type. Geostatistical tools are suitable instruments for a respective uncertainty analysis.

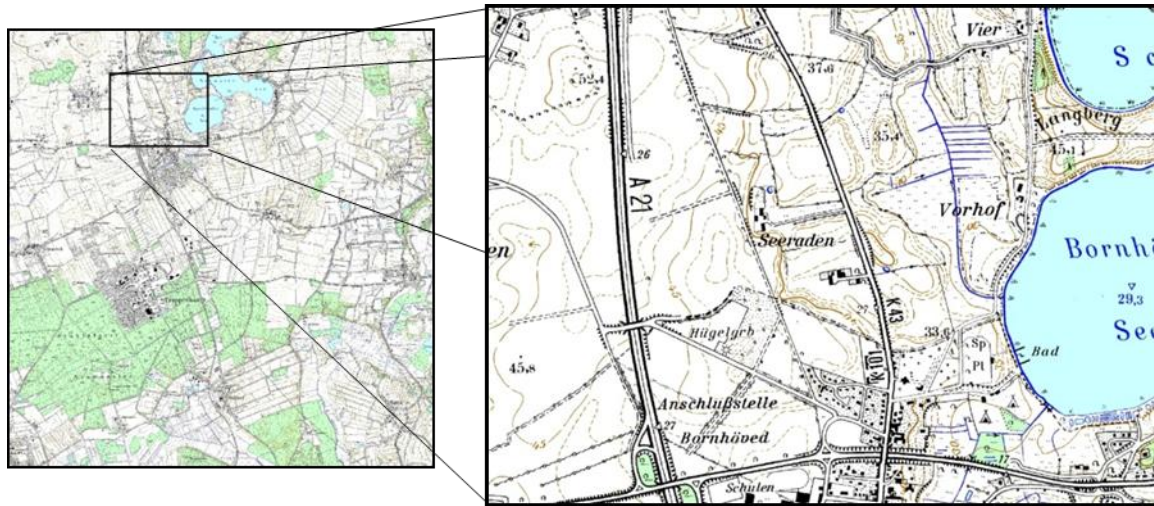


Fig. 1. Selection of a study area representative for a larger region, example from a northern German rural landscape

❖ Assessment step 2: Selection of relevant land cover/land use (LCLU) classes

LCLU have been acknowledged as a suitable framework for linking human and natural systems in ecosystem service assessments (Burkhard et al., 2009; Scolozzi et al., 2012). Land cover represents features on the land surface, whereas land use refers to human activities on land. Remote sensing-based classifications (such as the European CORINE Land Cover project; EEA 1994, Bossard et al., 2000) record both, land cover as well as land use (Burkhard et al., 2012b). Uncertainties are related to the generalizations and categorizations needed to reduce complex landscapes into a limited number of LCLU classes. In the CORINE nomenclature for example, altogether 44 land cover types are classified (EEA 1994), supposed to be representative for all European Union member states, from sub-arctic to semi-desert conditions. These generalizations remain a major challenge for working on LCLU-based assessments (Koschke et al., 2012), because logical, representative and research question-oriented categorizations have to be found, whose number does not exceed the demands of operational practicability.

Another crucial point is the selection of “relevant” LCLU types. It is generally desirable to consider all LCLU types present in a study region (Burkhard et al., 2012b; Koschke et al., 2012). Nevertheless, compromises have to be made between the accuracy of the landscape representation and technical feasibility. Different concepts might lead to additional spatial and thematic mismatches, depending for example on variations in landscape subsystem delineations, identification of processes and interrelations, scaling issues and spatial pattern development (Turner et al., 2001). Although it is desirable to carry out research question-based or management-related, many assessments follow data-driven approaches. Fig. 2 shows an example of the CORINE land cover data representative for the example region. In the CORINE data from 2000, the rather complex landscape has been reduced to five different classes. The forest patches, a characteristic landscape element of this area, are too small to be displayed in the data. It is

obvious that the low number of classes and related polygons as well as their coarse spatial resolution result in significant spatial and thematic uncertainties.

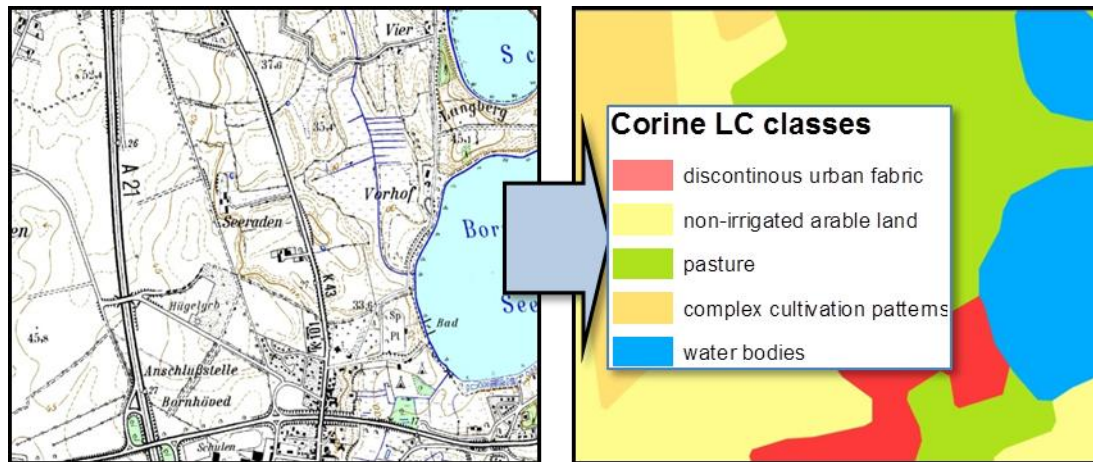


Fig. 2. Land cover map of the hypothetical study area, example with CORINE LC 2000 data.

❖ Assessment step 3: Spatial LCLU and biophysical data acquisition

The rather rough spatial and thematic resolution of the CORINE classification in Fig. 3 results in only ten different polygons for five different land cover classes. Thus, regional peculiarities will be lost, increasing the uncertainty of related assessments. CORINE's spatial resolution (minimum mapping units) of 25 ha might neglect smaller objects (or narrow linear objects such as rivers, roads, etc.), which can be extremely relevant for ecosystem service assessments. Misclassifications and inaccuracies in land cover type assignments can also occur due to spectral interferences within the satellite-borne remote sensing systems, resulting in technical uncertainty (Bossard et al., 2000). CORINE data are nowadays available for the three time steps 1990, 2000 and 2006. Therefore, CORINE is a suitable, free-available data base for large-scale assessments (e.g. on the continental scale; Haines-Young et al., 2012, Maes et al., 2011) and a suitable starting point for more detailed assessments, which then would integrate additional data with a better resolution (Burkhard et al. 2009; Vihervaara et al., 2010; Vihervaara et al., 2012). Fig. 3 illustrates the differences between CORINE 2000 land cover data and geo-topographic data from the German Authorative Topographic-Cartographic Information System (ATKIS¹²) managed by the German State Surveying Authorities. For the same area, ATKIS provides a much more detailed data set, consisting of a higher number of land cover classes with a higher spatial resolution. The use of higher resolution spatial data can help to reduce data uncertainties, certainly when working on smaller scales. But, the data processing and analyzing efforts as well as the costs for the data and their management are increasing at the same time.

¹² <http://www.atkis.de>



Fig. 3. Land cover maps from the hypothetical study area, CORINE LC 2000 (left) and ATKIS data (right).

Additionally to the land cover data, further geobiophysical data should be integrated to delineate spatially explicit units for ecosystem service assessments. Land use information (types and intensities), biotic information (supplementary vegetation data, fauna, habitats) and abiotic information (soil data, digital elevation models, climatic data, hydrological information) are relevant to determine functional units, which later can be classified into service providing areas (Burkhard et al., 2009). The different sources of uncertainty discussed in this article are relevant for all types of data mentioned here.

❖ Assessment step 4: Selection of relevant ecosystem functions and services

There are several recent ecosystem service studies and publications providing classifications and lists of ecosystem functions and services (Burkhard et al., 2009, 2012b; de Groot, 2010; TEEB 2010; Kandziora et al., 2012), providing good orientation to select relevant ecosystem functions and services. Uncertainties are related to the selection of functions and services themselves, and to the classification, definitions and indicators used. For example, there is still no consensus referring to the differentiation between ecosystem or landscape functions and services, potentials and capacities (Bastian et al., 2012). Ambiguously defined concepts and terminologies are a common source of uncertainty (McCarthy et al. 2001). Applied indicators have to represent sound indicandum-indicator relations and have to be quantifiable, sensitive to changes in land cover, temporarily and spatially explicit and scalable (Burkhard et al., 2009, 2012b; Müller and Burkhard, 2012; van Oudenhoven et al., 2012).

❖ Assessment step 5: Quantification of ecosystem function and service indicators

To assess the selected ecosystem functions and services, appropriate data for indicator quantification have to be acquired. Based on the integrative and interdisciplinary character of ecosystem service assessments, respective data are originating from a very broad range of sources of varying quantities and qualities. Therefore, quantifications are related to a broad range of uncertainties, depending on the data sources and on indicator-indicandum relations. The lack of appropriate data for quantifications or errors in the data sets cause high uncertainties when models are used (McCarthy et al., 2001). Another major issue is spatial or temporal mismatches of geobiophysical data and socio-economic statistical data, which often refer to different spatial and temporal units. Ecological data are based on environmental structures and processes, whereas

statistical data normally refer to distinct administrative units, such as counties or municipalities collected at statistical time intervals.

In general, there is still quite limited knowledge about complex ecosystem functionalities and their interactions with the human system. Especially indications of ecosystem functions, regulating and cultural (intangible) services are not really well-developed yet. This leads to significant uncertainties in modeling, measurement and statistical data quantification. Nevertheless, progress has been made during the last years of intensive ecosystem service research and assessments (Burkhard et al., 2012a).

❖ **Assessment step 6: Interlinking LCLU classes and ecosystem functions/services**

For the assessment of different LCLU classes' capacities to support ecosystem services, a matrix approach linking the different items has been developed (Burkhard et al., 2009, 2012b). At the intersections between LCLU types and ecosystem functions/services, the 0-5 capacity assessment values, originating from the respective quantifications, are entered. Uncertainties are related to the difficult differentiation between ecosystem service supply potentials and capacities. Especially with regard to consumptive ecosystem goods and services, such as most of the provisioning ecosystem services, the potential supply is often different from the actually used service capacity. One source of uncertainty specific for ecosystem functions and regulating services is their high risk for double-counting, which can occur when a clear delineation between function and service is lacking or where both are inseparably interconnected (e.g. in the case of pollination; Kandziora et al., 2012).

Averaging and interpolations of different ecosystem functions and services over space and time is necessary to assess the capacities in the matrix; both lead to uncertainties. In the scoring system, the function or service with the highest quantified indicator value in the study region should receive the maximum ranking, no matter what side-effects are emerging. To reduce related uncertainties, a weighting system, based for example on multi-criteria analysis (as applied in Haines-Young et al., 2012; Koschke et al., 2012), could be applied. However, such a system needs to be developed carefully and in a reproducible manner. Additionally, it would limit the advantages of the quick-scan character of the matrix approach, which is to produce reasonable overviews of a broad range of ecosystem functions and services assigned to spatially explicit landscape features.

Spatial interpolations and land cover generalizations contain uncertainties with regard to ecosystem service supply in reality. Ecosystem services are supplied spatially heterogeneously, and further generalizations are needed regarding temporal aspects of ecosystem functions and service supply. Of course there are seasonal and daily variations in natural processes which have to be considered when calculating average supply values. Thus, acceptable levels of uncertainty need to be defined, like in all ecosystem service assessments (Scolozzi et al., 2012). The integration of data of varying quality and quantity, originating from measurement, modeling, monitoring or statistics, is often the main source of uncertainty in comprehensive integrative assessments. Expert-based evaluations have been shown to be a suitable supplement for ecosystem service assessments (Burkhard et al., 2009, 2012b; Haines-Young et al., 2012; Kienast et al., 2009; Koschke et al., 2012). The number and selection of experts as well as the rather low degree of reproducibility are crucial sources of uncertainty in these approaches.

After collection, the data, models and indicators need to be aggregated and normalized into the relative capacity classes, without losing relevant information. The big advantage of this kind of

normalization to a relative scale is that data from various sources and with different units are made comparable, which might outweigh the disadvantages of information loss and uncertainty. Fig. 4 shows an exemplary ecosystem function and service supply assessment based on the land CORINE cover types from the study region.

	scale for assessing capacities:																									
	0 = no relevant capacity	1 = low relevant capacity	2 = relevant capacity	3 = medium relevant capacity	4 = high relevant capacity	5 = very high relevant capacity																				
	Ecological integrity					Regulating services					Provisioning services					Cultural services										
	Energy capture	Entropy production	Storage capacity	Cycling & Nutrient loss reduction	Biotic water flows	Metabolic efficiency	Heterogeneity	Biodiversity	Global climate regulation	Local climate regulation	Air quality Regulation	Water flow regulation	Water purification	Nutrient regulation	Crops	Biomass for energy	Fodder	Livestock (domestic)	Fish, seafood & edible algae	Freshwater	Recreation & Tourism	Landscape aesthetics & Inspiration	Knowledge systems	Religious & Spiritual experience	Cultural heritage & Cultural	Natural heritage & Natural diversity
Discontinuous urban fabric	1	1	1	1	1	1	1	1	0	0	0	0	0	0	1	1	0	0	0	0	3	2	1	5	1	0
Non-irrigated arable land	5	4	4	1	3	4	3	2	1	2	0	1	0	0	5	2	3	0	0	0	1	1	2	0	3	0
Pastures	5	5	4	2	4	5	2	2	1	1	0	1	0	0	0	1	5	5	0	0	3	2	2	0	3	0
Complex cultivation patterns	4	3	3	1	3	2	4	3	1	2	0	1	0	0	4	1	3	0	0	0	2	2	2	0	3	0
Water bodies	4	2	4	3	0	4	4	4	1	2	0	2	0	1	0	0	0	0	3	5	5	4	4	0	3	4

Fig. 4. Linking the land cover types in the study area to ecosystem functions and services, supply assessment example.

❖ **Assessment step 7: Linkage of assessment values to spatial biophysical and/or administrative units**

The 0-5 ecosystem function/service capacity values from Fig. 4 are then attributed to the spatial landscape units defined before. The respective spatial units should not solely be based on classified land cover types, but should also integrate further geobiophysical data such as spatial soil data, hydrological data, statistical land use data, more detailed vegetation and habitat data. A lack of data in quality and quantity, mismatches of spatial units (including spatial projection and coordinate system mismatches) or technical problems with the data or the GIS software are sources of uncertainty here.

Several ecosystem services, especially the cultural services, are difficult to assign to specific spatial units. For example the cultural service “landscape aesthetics” is, on the one hand, appreciated in a very subjective manner by humans and, on the other hand, related to landscape compositions rather than to single land cover types. Moreover, the landscape aesthetics service supply and benefit areas can be located at different places, as considered in the topographic concept of “viewsheds” (Boyd and Banzhaf, 2007). Maps basically allow the representation of one or two ecosystem services per map or averages/sums of different services are displayed, which again causes aggregation uncertainties.

Fig. 5 shows an example from the study area where land cover types are linked to selected ecosystem functions and services from the supply assessment. The example is based on CORINE land cover data and the hypothetical assessment matrix in Fig. 4. The obvious coarse spatial resolution of the CORINE-based assessment illustrates related spatial uncertainties.



Fig. 5. Linking the capacity assessment values to the spatial CORINE land cover units; exemplary hypothetical maps: biodiversity, water flow regulation, crop provision and recreation & tourism (from left to right).

❖ Assessment step 8: Interpretation of the results

The applicability of actually the best map with the best data and model sources can be reduced by limited map interpretation skills of the end-user. The lack of expert knowledge concerning the interactions between landscape management and ecosystem service supply cause data and map misinterpretations, leading to high uncertainty. One major problem is that ecosystem service assessments are often too complex and the information is too aggregated to be directly used in practical applications (Koschke et al., 2012). On the other hand, even the best map user with excellent expert knowledge cannot overcome significant data and mapping weaknesses, lacking model and map validation or incomplete information bases.

Data extrapolations as well as common ecosystem service value-transfer approaches reduce the reproducibility and the understandability of the results. Therefore, the applied approaches need to be transparent and the development of appropriate end-user interfaces is needed. In application-oriented ecosystem research, different contexts require different information with different degrees of precision (Scolozzi et al., 2012). It is the role of the scientific community to improve methods to quantify, measure, monitor, model, map and value ecosystem services. Finally, the data and information must be provided in an appropriate, transparent and reproducible format (Burkhard et al., 2012a). Based on that, different degrees of “acceptable” uncertainty in the results need to be defined (Scolozzi et al., 2012).

5. Consequences and discussions

5.1. Uncertainty sources

In the previous chapters, several sources of uncertainty have been listed, referring to general landscape systems analysis as well as ecosystem service assessments. The general origin of the corresponding methodological insecurities has been detected in the enormous complexity which human-environmental systems provide and the scientific attempts to cope with that complexity. The focal sources are the following:

- *Systems-based (unavoidable) sources* can arise from chaotic or catastrophe dynamics, stochasticity and non-linearity, which are implicit features of natural systems and which can hardly be avoided (see Online Appendix 1).
- *Systems-based methodological sources* of uncertainty emerge from indirect, cumulative and de-localized effects, which are often not considered systematically although their impact might be extremely high (see Online Appendix 1).

- *Basic sources of uncertainty in landscape analysis* come into existence due to landscape heterogeneity, hardly foreseeable landscape dynamics, spatio-temporal natural variabilities and due to scaling effects. The latter origin of uncertainty should not only be perceived with respect to the technical extents and resolutions of mapping procedures, but much more as a general organization principle of ecosystems and landscapes: All processes can be assigned to specific scales due to their individual spatial and temporal characteristics. Therefore, all attempts to depict them on other scales should in principle be failing and produce inexact results. *Basic sources of uncertainty in landscape modeling* can be attributed to model assumptions, model structures, uncertain or lacking model inputs and the consequences of spatial and temporal scaling.
- *Uncertainty sources referring to ecosystem service assessments* and valuations have been distinguished into the three parts: *Natural supply uncertainty* which is for example referring to the knowledge inadequacy of biodiversity's function in generating ecosystem goods and services; *Preference uncertainty* that considers for example the different levels of values, the development and integration of monetary valuation strategies and the respective methodological fundamentals; *Technical uncertainty* that is related to mapping and regionalization methods, service indication, quantification and valuation strategies, as well as the problem of a suitable data provision.

Focusing on the last groups, the uncertainty of natural service supply can be comprehended as the focal problem of spatially explicit ecosystem service assessments. Providing the essential spatial units for assessments, the highly aggregated information of land cover/land use data constitutes major uncertainties in quantifying and visualizing ecosystem services in space. As the spatially explicit ecosystem service assessments are highly demanded for landscape management and planning (Kienast et al., 2009), revealing and dealing with related uncertainties should be considered as an important issue.

To illustrate this problem, we have analyzed the uncertainties in the land cover-based ecosystem service supply assessments in a study area following the methods of Burkhard et al. (2009, 2012b). In the analysis, we identified eight typical working steps which are all accompanied by specific technical and thematic uncertainties. Among all the eight steps, the selections of the study area and the relevant land cover/land use classifications are the most crucial ones, which, are accompanied by significant uncertainties. No matter how carefully the area is selected, it is uncertain whether the area can really represent all particular geobiophysical settings and the general landscape pattern in the total area. As complex landscapes are always generalized and categorized into a limited number of land cover/land use classes for analysis, when we select the relevant ones for our study, we will inevitably lose information (Scolozzi et al., 2012).

Looking into the uncertainties pervading the whole assessment process, we found that initial data uncertainty is the most prevailing one. In addition to the land cover/land use data, geobiophysical data may also be too rough in resolution to adequately reveal the regional peculiarities. Acquired from different sources, the different types of data usually differ in quantity and quality and result in spatial and temporal mismatches. Furthermore, when they are used in extensive models or integrated into comprehensive assessment, the respective uncertainties may even be aggregated. Substantially, uncertainties in spatially explicit ecosystem service assessment originate from the limited knowledge we obtain about the complex ecosystem and its interactions with the economic and social systems, or in other words, from the interdisciplinary characteristic of ecosystem service assessment itself. This knowledge limitation results in the uncertainties related to the

definitions and classifications of ecosystem functions and services, the selection of relevant functions and services in the study area, and the differentiation between ecosystem service supply potentials and capacities. The origination of uncertainty in spatially explicit ecosystem service assessments supports the definition of uncertainty, which we discussed at the beginning of this paper as: “a state with limited knowledge”.

5.2. Uncertainty analyzing and reducing strategies

By summarizing the literature, we found several methods which seem to be promising to analyze and assess uncertainties in landscape analysis (see Online Appendix 2). Scenario simulation can integrate spatial and temporal dynamics of landscapes and reveal the possibly uncertain trends of landscape pattern and process (Katz, 2002). Another method has been developed by Strand (2011), illuminating and regarding statistical analysis. By building a landscape model consisting of multiple overlapping probability surfaces, the method indicates the uncertain affiliation of a particular pixel with different landscape categories. Additionally, it helps to reveal uncertainties caused by scale-mismatched data by comparing the results generated by different data sources. Several approaches can be used to improve ecosystem service assessments with respect to uncertainty. Some of these items have been listed in Figures 6 and 7. While Figure 6 depicts different classes of uncertainty sources from a general systems-dynamic approach, Figure 7 is related to the ecosystem service handling steps of the assessment methodology. In both cases the uncertainty origins are classified at the left side while potential measures to cope with the respective uncertainty are sketched at the right side. The numerated steps of uncertainty assessment (1-9) could be used as rough guidelines to qualify and document the uncertainties of ecosystem service studies.

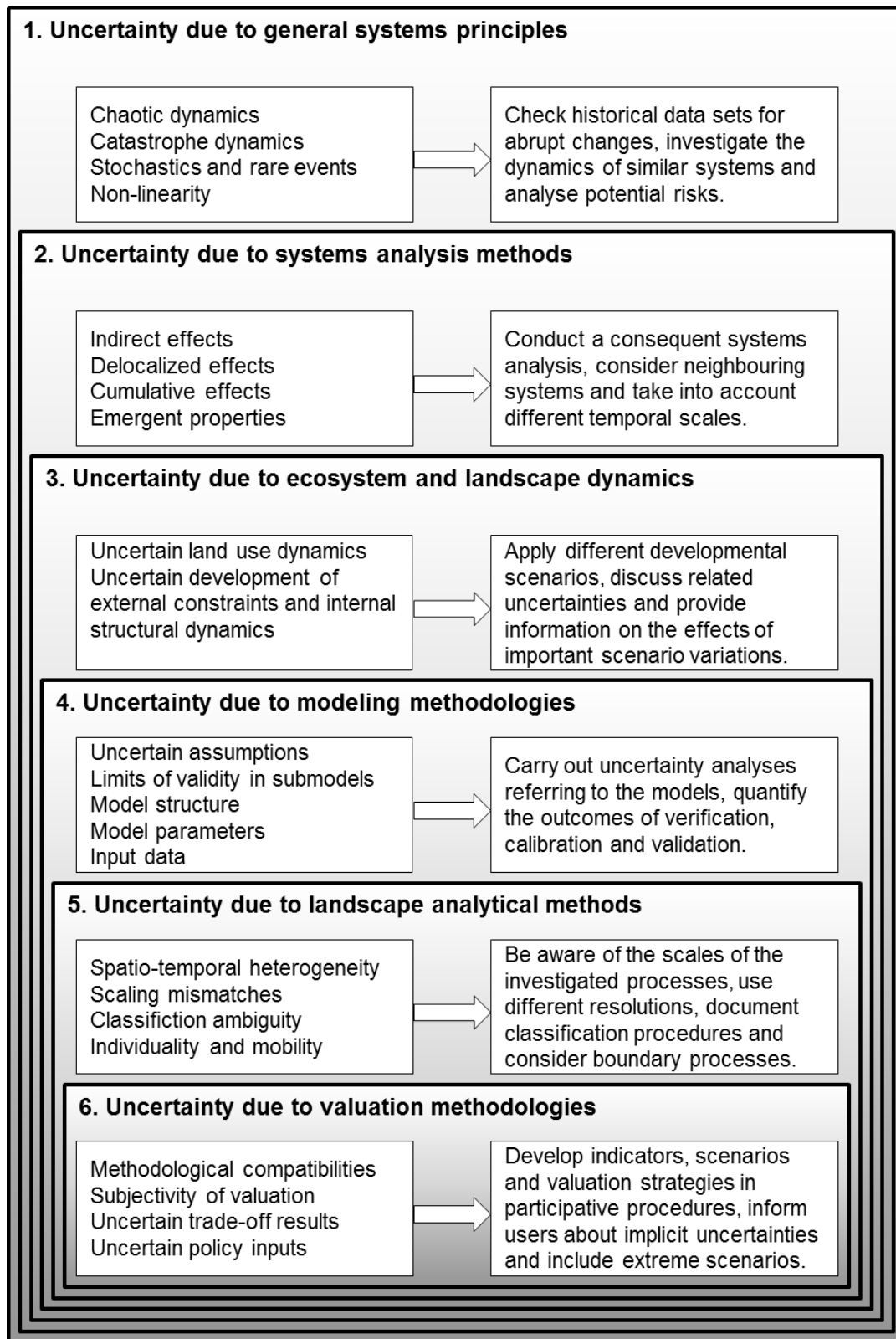


Fig. 6. General systems-based sources of uncertainty and potential actions for uncertainty reduction.

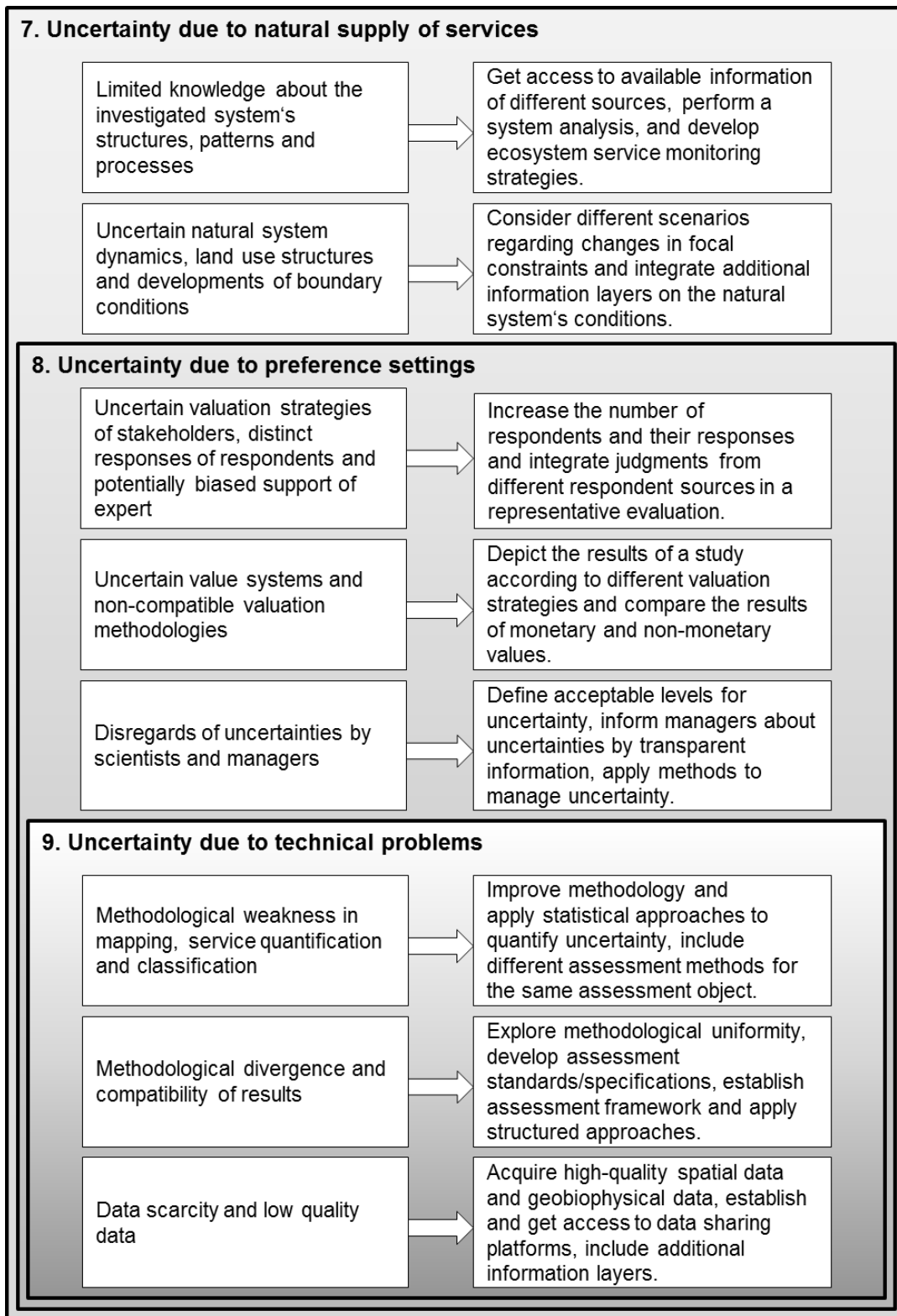


Fig. 7. Special sources of uncertainty in ecosystem service assessment and potential actions for uncertainty reduction.

From these instances we can derive approaches to deal with uncertainties in ecosystem service assessments: Uncertainty is pervading in the whole process of the assessments. We cannot avoid it, not only because it originates from our limited knowledge about the complexity of ecosystems and the transdisciplinary characteristic of ecosystem service assessment, but also because it is mostly communicated and accumulated with imperfect assessment techniques. However, we can find ways to deal with uncertainty and to increase the reliability and applicability of assessment results. To contribute to this objective, we propose the following approaches.

- *Being aware of uncertainties in ecosystem service assessments:* The most essential concern about uncertainty in ecosystem service assessments should be its location and the magnitude. This is a prerequisite for the identification of ways to reduce the effects of uncertainty. It is therefore indispensable to integrate uncertainty information into the assessment process and results (e.g. the accuracy of landscape classification, the resolution of spatial data and the reliability of the service values). As a guideline for uncertainty documentation, the components of Figures 6 and 7 might be applied. To measure uncertainties, probability assessment remains a main method, which especially improves the often ambiguous results of landscape classification. Regarding the measuring tools, geostatistical methods are promising instruments, because they can quantify the uncertainty of the unknown spatial characteristics (Zhang et al., 2009).
- *Using methodologies of systems analysis:* General systems analysis provides an elaborated methodology to cope with complex situations. The analytical steps include procedures such as a strict definition of the objectives and purposes, the spatial and temporal boundaries and the relevant scales of the investigation. Following these steps, the elements, subsystems and relations have to be defined, data requirements must be formulated and the most significant exogenous factors must be listed. These determinations are followed by a conceptual diagram (which is always helpful) of the investigated system. The subsequent steps refer to the modeling procedure including model verification, calibration and validation. Even if the methodology does not lead to the programming of a computer-based model, it is recommendable to use this procedure also for ecosystem service assessments due to their high complexity.
- *Learning continuously:* As uncertainty is substantially a state with limited knowledge, the most basic way to cope with it is increasing our knowledge about the objects to be assessed through continuous learning. By continuous investigations, we will hopefully increase our certainty about complex ecosystems and the complicated interrelations between ecological and socio-economic systems. In other words, the uncertainties within the ‘ecosystem service cascade’ (Haines-Young and Potschin, 2010) will be reduced. Taking acquiring and analyzing geobiophysical data as an example, long time, extensive sampling and increasingly developed statistical methods have increased our understanding of the distribution of spatio-temporal ecological variability and the underlying ecosystem processes (Nicholson et al., 2009).
- *Reducing preference uncertainty:* Preference uncertainty lies within the investigation of the respondents’ preferences about ecosystem services (e.g. respondents’ willingness to pay and professional judgments in ecosystem valuation). To reduce preference uncertainties, it is preferable to ask respondents for a range of values instead of only a specific one (e.g. conjoint analysis). To increase the certainties of the professional judgment results, it is also necessary to include methods that can reduce the judgment’s stochasticity and increase its representativeness (e.g. Delphi survey, Scolozzi et al., 2012).

- *Conducting scenario simulations:* Scenario simulation is a suitable approach in addressing the uncertainties generated by changes of ecosystem structures and processes across space and time. Affected uncertain natural driving forces (e.g. climate change), the socio-economic system (e.g. agriculture production and urbanization) and changes in land cover/land use carry along significant uncertainty. As scenario simulation can integrate different socio-economic conditions and reflect consequences of different policies (Swetnam et al., 2011), it can capture the potential trends of land cover/land use dynamics and reveal several relatively certain ecosystem service scenarios. We suggest that scenario simulation deserves more applications in ecosystem service assessments based on policies or for the purpose of decision making.
- *Applying the comparison method:* Comparison is the most effective method to reveal uncertainties in ecosystem service assessments, especially for the insecurities carried by data and assessment methods. By comparing the results generated by different data sources, uncertainties caused by data problems such as scale-mismatching and resolution-divergence can be addressed. In our case, we presented the uncertainties of different land cover classifications by comparing CORINE data and ATKIS data. Many assessment methods have been applied to ecosystem services, but to date there is no standard methodology. Each method carries strengths and shortcomings as well as respective uncertainties. Comparing the results of different methods can effectively indicate which approach is the most certain one for a specific study case.
- *Optimizing data quality:* Increasing data quality is the most straightforward way to reduce uncertainties in assessment studies. Approaches for this purpose include adopting better data sampling methods, enhancing measurement methods and tools and seeking high quality data. However, this plausible measure is actually not that easy, because the data processing and analyzing efforts as well as the costs for the data and their management are increasing at the same time. Nevertheless, increasing data quality should always be an essential requirement in ecosystem service assessments.

6. Conclusions

Working with landscape analysis and ecosystem service assessment, scientists and environmental managers should be aware that they are operating in a space of extreme insecurity and uncertainty. Consequently, respective assessments should be accompanied by *uncertainty assessments*, which use the prescribed statistical and analytical methods or at least list the different sources of uncertainty in the respective study (e.g. orienting at Figures 6 and 7) as well as their potential impacts on the results. It has been mentioned that a second step of such a strategy should be *uncertainty reduction* down to an acceptable level with the help of an adapted methodology. As ecosystem services are analyzed by practical reasons, the corresponding uncertainties of the study should be *documented and transferred* to the user in the subsequent step, making her or him aware of the (in)accuracy which she or he has to cope with in management. The practitioners' ability to include uncertainty into policy judgments and management processes would be enhanced by structured decision-making methods and an *adaptive management framework* (Nicholson et al., 2009). As this framework builds on learning processes, it is not necessary to wait for new comprehensions and knowledge, but to learn by doing. Carpenter et al. (2006, p. 9) have formulated this context in the following manner: "In a world in which ecological change occurs quickly, approaches that allow managers to consider uncertainty and to learn are more likely to be successful and to improve future management". An inevitable part of this

management strategy is continuous monitoring of the investigated system's development. The results of that long-term observation will be, on the one hand, to stepwise reduce the uncertainties formulated and, on the other hand, to help to improve the assessment methods in the future.

In the end, we can return to the initial questions of the paper and sketch some summarizing tentative statements:

- *How do the sources of uncertainty influence landscape analysis and modeling?* Several effects have been discussed in the preceding text. We are always in danger of classification errors, scaling mismatches and biased aggregations. Under these circumstances, landscape science has attached excellent quality of analytical methods and we can expect a continuous improvement of the analytical qualities.
- *Which special uncertainty sources can be assigned to the assessment of ecosystem services?* Ecosystem services are the benefits people obtain from ecosystem structures and functions in combination with other inputs. Therefore, all uncertainties listed for ecosystem and landscape analysis are also effective concerning ecosystem services. Additionally, the valuation component, the quantification of the related benefit size, brings about several uncertainties, biased viewpoints and trade-off preferences. Furthermore, the linkage of ecosystem structures, processes and functions with defined services is subject of uncertainties which are situated between system comprehension on the one side and system utility on the other.
- *How can we cope with these uncertainties in landscape science and management?* We can 'survive' with all those uncertainties discussed above astonishingly well. But undoubtedly, our studies could be more accurate by developing better methods for landscape analysis and ecosystem service quantification and by including an uncertainty analysis which might take the sources and actions as depicted in Figures 6 and 7 as a guideline.

After discussing so long about the frustrating reasons why we have problems in landscape and ecosystem service science and application, we might finally come back to the introductory citation of Bertrand Russel, saying that in fact nothing is certain, but that some things are more certain than others. With this knowledge, the long lists of uncertainties might not only be frustrating for scientists, but also challenging, because both, uncertainty assessment and uncertainty reduction are their basic tasks. Thus we can look into the future with full of optimism, because there is still so much work ahead of us.

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Supplementary materials

Online Appendix 1

Systemic and methodological uncertainties.

In general, several systems-based sources of uncertainty can be distinguished. Some of them are primarily basing on methodological or technical problems, thus they might be prevented if the methodology could be made more comprehensive and exact. Other sources can hardly be avoided due to their principle character. In the below list of uncertainty origins, we draw a gradient of these features, starting with hardly avoidable basements of uncertainty:

Uncertainty sources	Explanations
Chaotic dynamics	Chaotic behavior can arise when simple deterministic interactions provoke a broad potential of developmental directions in the context of self-reproducing processes. Therefore the behavior of the chaotic entity cannot be predicted, mainly due to a very high sensitivity of the system to small changes in the initial conditions (Jørgensen, 2000). In chaotic systems, parameter values may change unexpectedly, the behavior cannot be foreseen and it is impossible to produce generalizations on systems dynamics.
Catastrophe dynamics	In ecosystems, unexpected, fast and abrupt discontinuities enable different potential steady states and trajectories as functions of identical parameter values. Bendoriccio (2000) has applied the catastrophe theory of Thom (1972) and the different types of catastrophes to ecological systems and models. A landscape-related consequence is the appearance of hysteresis effects, e.g. leading to different behaviors of a variable whether a regulating parameter is increasing or decreasing.
Stochastic dynamics	Stochasticity emerges when the development of a state variable is influenced by stochastic events. The observer cannot conclude regularities because the system behavior is intrinsically non-deterministic due to random influences. Levin et al. (1997) have discussed stochastic behavior in the context of ecological models, which can hardly cope with these dynamics. In several cases, stochastic dynamics can lead to unexpected disturbances in ecosystems (e.g. Carpenter and Gunderson, 2001).
Rare events	Rare events are very often reasons for abrupt changes of ecosystem structures and functions. In these cases, the development of a variable cannot be described due to extraordinary external inputs that cannot be foreseen by the observer. Examples are floods, erosion events, or storms which can strongly attack the resilience of the investigated systems (Gunderson and Holling, 2000).
Non-linearity	Ecological relations are mostly non-linear (May, 1980): An increase in a parameter value is not always correlated with an increase in the values of a corresponding state variable. Many physiological interrelations can be taken as examples (e.g. temperature regulation), and also landscape dynamics can be observed from the viewpoint of non-linearity (Koch et al., 2009).

Individuality	Another source of uncertainty in ecology can be assigned to the fact that single organisms differ genetically, due to their adaptability, vulnerability and their sensitivities. Thus, they can react in a broad spectrum of possibilities, making generalizations rather fuzzy. Furthermore, organisms comprise a specific behavior which can vary strongly. This diversity of reactions may have effects on landscape structures and functions which can hardly be foreseen. For example, organisms can evade from threats and thus their total abundance may be changing steadily.
Irreversibility and emergence	All self-organized processes in open systems are irreversible processes (Jørgensen et al., 2007; Prigogine, 1980). The initial state will never be reached again; the systems change constantly, even if the dynamic states seem to provide rather similar impressions. Furthermore, during the development of a system, new features can emerge, and this emergence can appear very fast and effective (Nielsen and Müller, 2000). With these new or modified elements, the general behavior may change, and thus a prediction of future states will be difficult.
Indirect effects	In ecosystems and landscapes, indirect effects are mostly more significant than direct relations (Fath and Patten, 2000; Jørgensen et al., 2007). Regrettably, the analysis of indirect effects often demands for a very comprehensive systems analysis. If the resulting indirect, cumulative, delocalized and delayed long-term influences are not taken into account, the results of analyses and assessments can easily tend to be inaccurate.
Boundary/edge effect	Systems usually have uncertain or ambiguous boundaries/edges, as changes from one system to another are seldom abrupt, but normally gradual. The conditions at the boundary/edge may be different from either of the connected systems. They are usually difficult to predict due to insufficient information (Breckling and Dong, 2000). For example, in landscape classifications from remote sensing images, the uncertain boundaries between regions will lead to ambiguous and inaccurate results (Strand, 2011). The edge between two natural habitats may have emerging species with different abundance and distributions from the neighbored ecosystems.
Uncertain methods	The available methodologies are not always perfect and could often be improved. The results' certainty varies broadly dependent on the distinct methodologies and techniques of the disciplines involved. For example, due to organismic mobilities and heterogeneities, failures up to 100 % can occur quite frequently in ecological data sets, i.e. concerning animal abundances.
Limits of validity	Models or regressions are valid within a specific range only. Inside that validation area interpolation is possible, but outside this space the model should not be applied. Nevertheless, such extrapolations are carried out in ecological studies, thereby enhancing the uncertainty of the results.
Spatial heterogeneity	Ecological systems can be extremely heterogeneous. Therefore, the resolution of an investigation strongly influences the results. At different spatio-temporal scales measurements are conducted with distinct extents and resolutions. Therefore, the outcomes can provide strong differences, depending on the investigation's hierarchical level.

Temporal heterogeneity	Ecological state variables develop with a very high temporal variability. Therefore, measurement durations, periods and effective sampling times can strongly influence the results.
Scale mismatches	At different spatio-temporal scales, measurements are conducted with distinct extents and resolutions. Therefore, the results of an analysis can provide very strong differences, if different resolutions and extents are utilized. In this context it has to be considered that scales are interrelated spatial and temporal characteristics, that each process operates at a distinct scale, and that in several instances the selected methodology has been developed for and is focusing on one special scale only.
Human influence uncertainties	In comparison with the mostly nature based uncertainties mentioned above, a very high insecurity is introduced by including human decisions, valuations and dynamics. Developing an inter-subjective methodology for the analysis of human influences additionally is influenced by several societal processes and normative loadings, which provide a high degree of uncertainty.

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Online Appendix 2

Uncertainty analysis methods for modeling and landscape analysis.

Uncertainty analysis aims at identifying major uncertainty sources, quantifying their magnitudes and relative contributions, testing their effects on model outputs, and determining prediction accuracy (Jansen 1998; Katz 2002; Li and Wu 2006). Uncertainty analysis is indispensable in modeling as it illuminates the adequacy of models, determines the most critical factors to sensitivity and uncertainty and reveals the reliability of the modeling results (Li and McNulty, 2007; Wu et al., 2006). Among the methods of uncertainty analysis, the typical ones are sensitivity analysis, scenarios simulation, Monte Carlo simulation, statistical techniques, model comparison and uncertainty visualization (Heuvelink 1998; Katz, 2002; Li and Wu, 2006; Álvarez-Martínez et al., 2010). As landscape modeling is closely linked with landscape analysis, these methods are also applied in analyzing the uncertainties emerging in landscape patterns and process analysis. These methods are sketched as follows:

Methods	Explanations
Sensitivity analysis	Traditionally, sensitivity analysis tests the rate of change in the model output, which is caused by an infinitesimal amount of variation of a single input variable. As sensitivity analysis can make comparisons between different inputs, it helps to determine the most critical factor for the model results from different uncertainty sources (Katz, 2002). Concerning landscape modeling and analysis, sensitivity analysis is used as an important method to analyze the result variations caused by the uncertainty of spatial data input (Baldwin al., 2004; Chen et al., 2010; Mas et al., 2010; Tenerelli and Carver, 2012).
Scenario stimulation	Scenario simulation can be applied as a special sensitivity analysis, taking changed possible future boundary conditions as model inputs (Clark et al., 2001; Katz, 2002; Li and Wu, 2006). Usually, it reveals realistic future conditions caused by policies by designing a combination of a best, a most likely and a worst scenario (Katz, 2002; Li and Wu, 2006). As a promising tool for dealing with uncertainty, scenario simulation is widely used in modeling possible alternative paths for the economic, social, climatic and political environment (Bohensky et al., 2006; Gómez-Sal et al., 2003). It is also applied in integrating spatial and temporal dynamics uncertainties of ecosystem and landscapes (Carpenter et al., 2006; Swetnam et al., 2011)
Monte Carlo	Monte Carlo simulation leads to a common quantitative method to assess

simulation	<p>uncertainty in model prediction by running the model under perturbations of input values, estimating the range of possible outputs and the probability distribution of the model parameters (Li and McNulty, 2007; Li and Wu, 2006; National Research Council, 2005). Usually, it is carried out by drawing samples randomly from the defined input distribution, repeating simulation with the stochastic samples and determining a probability distribution for the outcome (Jansen, 1998; Katz, 2002; Li and Wu, 2006). Regarding space-based modeling simulation, the Monte Carlo method is used to analyze the uncertainties in simulation processes and results (Xu et al., 2005; Wechsler, 1999)</p>
Statistical techniques	<p>The uncertainty analysis results are usually formulated as probability distributions or statistical characteristics of model predictions (Li and Wu, 2006; Wu et al., 2006). Wu et al. (2006) and Jansen (1998) point out the necessity of statistical uncertainty measures for the reliability of model results. Multivariate statistical techniques are beneficial for examining model parameter properties, quantifying models and exploring possible concepts to improve them (Klepper, 1997). Among the relevant methods, Bayesian statistics hold a promising role. Different from the traditional probabilistic methods, it assigns prior probability distributions to the model parameters through experience and subjective judgment and builds a likelihood function with the observed data. Today, the increasing computer technology pushes the Bayesian method into large uncertain problems and space-based ones, such as estimating climate sensitivity, determining uncertainties in hydrologic models (Najafi et al., 2011; Raje and Mujumdar, 2010; Renard et al., 2010) and addressing the uncertainties in landscape assessments (Stelzenmüller et al., 2010; Grêt-Regamey et al., 2012).</p>
Comparison method	<p>Models always carry structural errors, input uncertainties and sometimes poor observation data for validation. If uncertainty analysis is not adequate, model comparison remains indispensable (Heuvelink, 1998; Jansen, 1998). Comparing the prediction results of a model with the ones of other relevant models helps to assess model quality and test the accuracy of the prediction results (Callaghan et al., 2004; Heuvelink, 1998). Examining how well model predictions of the future match actual observations is crucial to determine whether a model is reliable and applicable.</p> <p>Model comparison contributes to revealing model uncertainties for two aspects. One is known as inter-model comparison, which involves comparing one model with another one and is promising to identify model structural uncertainty and evaluate model adequacy, especially under the condition of data scarcity, (Li and Wu, 2006). The other approach is inter-data comparison, where the results of the same model predicted from data at different quality levels are compared (Heuvelink, 1998; King et al., 1998).</p>
Uncertainty visualization	<p>Owing to the development of spatial analysis methods based on GIS, uncertainties in landscape analysis and modeling are becoming more and more visually explicit to researchers. Confusion matrix (named also fuzzy</p>

	<p>classification) is a promising method to quantify and show the uncertainties in mapping landscapes (Crosetto et al., 2000; Álvarez-Martínez et al., 2010). With the help of GIS techniques, it can measure the magnitude of errors and locate them in maps by comparing different map categories (Woodcock and Gopal 2000; Tenerelli and Carver 2012). Another method to model and visualize spatial data errors is spatial autocorrelation analysis, which is useful to quantify and map digital elevation model (DEM) error propagations (Hengl et al., 2010; Gongga-Saholiariliva et al., 2011). Besides, DEM errors can also be spatially quantified by the methods such as geostatistical simulations (Borisov et al., 2009; Gongga-Saholiariliva et al., 2011) and geographically weighted regression (Erdoğan, 2010; Gallay et al., 2010).</p>
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Chapter 6. General conclusions

This thesis investigates the quantitative linkages of ecosystem services with socioeconomics at regional scales and analyzes the respective uncertainties of the respective quantifications. Within several findings, the four objectives identified in the introduction of this thesis have been achieved. Basing upon this research, the following conclusions can be drawn, which also constitute the answers of the four research questions raised in the introduction:

Ecosystem services are closely connected with socioeconomics in human-environmental systems.

1) The quantitative linkages between biodiversity, ecosystem services and human well-being at the prefecture-level city scale of Jiangsu have been evidenced using the DPSIR model and statistical methods. All of these three components of the investigated human-environmental system are influenced by the regional socioeconomic dimensions. The DPSIR model shows strength in integrating the information of different components of human-environmental systems into a causal network. It can be combined with quantification methods such as statistics to estimate the strengths of the connections between different components and calculate the magnitudes of the socioeconomic influences.

2) Ecosystem services and socioeconomics both show gradient characters across the space of the urban-rural areas of Leipzig and Kunming. The analyzed ecosystem service gradients have significant correlations with different socioeconomic gradients in different degrees in both of the two study areas. The spatial characters of land covers can explain the strong linkages of some ecosystem service gradients with some socioeconomic gradients. Some socioeconomic dimensions affected by complex factors of the human system present idiographic gradient patterns compared to the ecosystem services or socioeconomic items with regular gradients.

3) The spatial characteristics of ecosystem services are hypothesized to be affected by socioeconomic status at river basin scale in lake watersheds. The hypotheses have partly been proven by the empirical study of the Dianchi Lake Watershed. The evidenced influences primarily relate to the attributes of land use structure. The invalid tests of the hypotheses are mainly due to socioeconomic data scarcity and uncertainty of the study area. The hypothesis-test methodology used in the study can be applied to investigate socioeconomic effects on ecosystem services in the face of data insufficiency and uncertainty. Therefore, researchers should prioritize the scale match of biophysical and socioeconomic data when investigating the relationships between ecosystem services and socioeconomics.

Different methods are applicable to quantify distinct ecosystem services.

The quantification methods used in this thesis relate to ecosystem service modeling, value transfer and the use of biophysical and socioeconomic data. I used a mainstreaming and regional-targeting model, which is named Integrated Valuation of Environmental Services and Tradeoffs (InVEST), to estimate the ecosystem services whose values are not directly available (habitat quality in chapter 3 and 4, and water yield and nitrogen retention in chapter 4). Despite the uncertainties in the modeling procedure (e.g., the limitations in the assumptions of different modules of InVEST), the results are basically credible to show the spatial variations of the modeled ecosystem services.

The modeling results are not validated in the study due to the lack of data from observations or measurements or the results produced by other models. However, some other studies have provided validations of the modeling results of InVEST, which suggest the essential reliability of the tool in estimating the biophysical values of ecosystem services (Bai et al., 2013; Marques et al., 2013). Considering the use of InVEST in this thesis and other research, I suggest its application in quantifying ecosystem services in the form of biophysical values at regional scales when the investigators have difficulties in deriving the ecosystem service values from field work. Thus, the monetary value-based modeling by InVEST, which is not conducted in this thesis but involved in many other studies, should be applied with more cautions. As argued by many researchers, monetization adds additional uncertainties to ecosystem service quantification results (Hou et al., 2013; Spangenberg and Settele, 2010; TEEB, 2010). The uncertainties will proliferate when aggregating the values of different service types calculated by divergent monetization methods. This is one of the reasons why this thesis does not monetize ecosystem services.

The value transfer used in this thesis should be reliable as I derived the ecosystem services values from the existing empirical studies of the same or similar study areas as this thesis (e.g., carbon storage in chapter 2 and 3). Value transfers have been a widely used method in the field of ecosystem service research since the release of the report of the world's ecosystem service values (Costanza et al., 1997). This article provides the area-averaged monetary values of different ecosystem service types of distinctive ecosystem classes at the global scale. Nevertheless, the investigators in some cases virtually abused these values when transferring these values to their case areas at other spatial scales, intentionally or unintentionally overlooking the tremendous uncertainties with the global-averaged values. Such critical value transfers probably had caused considerable biases of the results. Fortunately, more and more researchers became aware of this issue and attempted to adjust the value coefficients to fit other spatial scales (Li et al., 2010; Xie et al., 2008). Concerning the value transfer method, this thesis presumes that biophysical values should be preferred to monetary values. Besides, the researchers should prioritize the values from similar study areas and similar spatial scales.

Maps show strengths in visualizing ecosystem services and communicating the respective spatial information.

The ecosystem services maps in this thesis involve two primary patterns, pixel-based maps and maps at administrative units or river basin scales. The former can present the details of spatial characters of ecosystem services and the latter can facilitate the comparisons of ecosystem services between different regions or river basins. Ecosystem service maps can be further differentiated into more specific ones, such as maps of ecosystem service potential, flow and demand (chapter 4, also see Burkhard et al., 2012b; Maes et al., 2012). Comparing these maps can reveal the spatial match status of these ecosystem service attributes, which may draw the awareness of the environmental managers and planners to enhance the spatial balance of these three items. The achievement of such spatial balances can increase the sustainable use of ecosystem services by the regional residents. Furthermore, the spatial relationships between ecosystem services and socioeconomic attributes (e.g., land use, population density and urban fabric in chapter 3) can be uncovered when comparing the respective maps.

Uncertainties accompanying ecosystem service assessments can be revealed, quantified and visualized.

This thesis points out the importance of being aware of, revealing and reducing uncertainties in ecosystem service assessments and proposes a methodology to achieve these purposes. This methodological framework is applied to treat the uncertainties of the case studies. Concerning the case study in chapter 2, uncertainty sources are comprehensively uncovered and described. Chapter 3 indicates that the built uncertainty gradients can explicitly show the magnitude and changes of uncertainties generated by value averaging in the process of building ecosystem services and socioeconomic gradient. In chapter 4 the sensitivity analysis results indicate different sensitivities of the modeled ecosystem services to the input data and parameters for modeling. As demonstrated by the empirical analyses in this thesis and many other studies, uncertainties are inevitable in the assessment of ecosystem services (Grêt-Regamey et al., 2013; Spangenberg and Settele, 2010). The production of ecosystem service values and maps which appear sound and undebatable is far less than enough. The investigators should include uncertainty analysis as part of the report of ecosystem service evaluations and indicate the credibility and application limitations of the assessment outcomes. The uncertainty analysis methodology established in this thesis can aid the researchers when they confront the uncertainty issues.

The past 20 years have witnessed “many flowers blooming” of ecosystem service studies. The extensive attentions from the scientific community on the concepts and approaches of ecosystem services are due to the great concern of human-environmental issues from the human society. Ecologists, economists as well as environmental and social scientists have all shown interests in ecosystem services as it is promising in addressing the interactions between environmental and human systems. Moreover, the researchers have seen the hopes that the ecosystem services approach can benefit the policy making of sustainable environmental management and help to build a win-win relationship between humans and the nature. However, these hopes have not yet been fulfilled as the concepts and approaches are still under development. Although research findings are fruitful, very few are transparent enough for policy makers and supportive enough in the decision making of regional environmental management practices.

Due to the young state of the approach, many methodological problems can be listed. The obstacles for the application of ecosystem service studies mainly result from the unrefined concepts, the divergent quantification methods of different robustness, the data shortages as well as the pervading uncertainties throughout the entire assessment process. The concepts of ecosystem services and ecosystem functions had not been clearly differentiated for a long time. Investigators had not considered the differences between ecosystem service supply potential, the actually used ecosystem services by humans and humans’ demand until recent years. Moreover, scientists have been struggling to reach an agreement of a common and widely accepted classification of ecosystem services and to find appropriate measures to indicate different service types. Concerning the quantification of ecosystem services, different methods have their own strengths and weaknesses and none of them are versatile enough to estimate the values of all different services. Consequently, researchers face difficulties when aggregating the values of different ecosystem services estimated by distinct methods, even if the values have the same unit, e.g., all in monetary term. As often a thorny issue for ecosystem service investigators, data scarcity and unavailability constitute another hindrance in the path where the assessment results transfer from academic publications to policy makers. Many researchers have to compromise between the large number of ecosystem services they are interested in and the small database on

which they can base the research. Uncertainties are inseparable from ecosystem service research and an impediment to applying the findings in policy making. Whereas, not until recently were the uncertainty sources in ecosystem service assessments systematically summarized and was a comprehensive methodology proposed to cope with the uncertainties.

The mentioned obstacles provoke great efforts from the researchers in future studies on ecosystem services. The refinement of the concept calls for the differentiations between ecosystem service potential, flow and demand and the assignment of appropriate indicators. Besides, the concept refinement can be the revelation of the linkages between ecosystem functions, ecosystem services and human well-being and the seeking of respective empirical evidence. Concerning the development of the assessment methodologies, more versatile, robust and adaptable tools to model ecosystem services are needed. Such tools should be easily accessible, easy to use and able to capture the uncertainties in the modeling. Additionally, common assessment frameworks that can integrate the opinions of different stakeholders, e.g., scientists, local residents, NGOs and policy makers, are desirable to facilitate the application of the assessment results. In order to improve situations of data shortages many researchers are facing, data sharing platforms will be necessary. The platforms can be based on national levels which include sub-platforms of different regions and are linked with the ones of other nations by international collaborations. The reduction of uncertainties in ecosystem service assessments can be achieved by the improvement of assessment methods and the enrichment of data. Other potential approaches include comparing the evaluation results derived by different methods with different data sources, developing assessment standards and specifications, applying structured approaches as well as integrating more local experts' opinions on the evaluation processes and results.

Being surrounded by those demands, this is an exciting time for the area of ecosystem service research with novel innovations emerging regularly. The potential for ecosystem service approaches to propel decision makers towards more sustainable policies is tremendous. Therefore, we should keep on working on this desirable issue with full optimism in spite of the thorns on the path, with the hope that we can increase the understanding and acceptance of ecosystem services for the public and the decision makers in the future not far away.

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Appendix

Curriculum Vitae

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Education

- Ph.D. candidate in Ecology, Christian Albrechts University of Kiel, Germany
PhD finalized by August 2014
Dissertation: “Socioeconomic contexts for the spatial variations of ecosystem services and the associated uncertainties” Advisor: Prof. Dr. Felix Müller
Explored the interrelations between ecosystem services and socioeconomics in human-environmental systems. Theoretically and practically analyzed the uncertainties in ecosystem service assessments
- M.Sc. in Land Resource Management, Beijing Normal University, China July 2011
Thesis: “The Benefits of the Conversion of Cropland to Forest and Grassland Project in Zhungeer County, Inner Mongolia.” Advisor: Prof. Dr. Bo Li
Investigated environmental and ecological effects of land use changes by field survey, sample physical and chemical analyses as well as statistics
- B.E. in Environmental Engineering, University of Science and Technology, Beijing
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Awards and honors

- Excellent Graduate of Beijing Normal University, 2011.
- Bayer Young Environmental Envoy, Chinese delegate of a study trip to Germany. Bayer AG, Germany in 2008.

Research focuses and interests

- Modeling and quantifying ecosystem services
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- Interactions between human and environmental systems