Spatially explicit environmental assessment of global biomass production for bioenergy at the regional scale

Dissertation
zur Erlangung des
Doktorgrades der Naturwissenschaften (Dr. rer. nat.)

der

Naturwissenschaftlichen Fakultät III
Agrar- und Ernährungswissenschaften,
Geowissenschaften und Informatik

vorgelegt von
Herrn Markus Andreas Meyer
geb. am 17.04.1987 in Salzgitter

1. Gutachter: Prof. Dr. Ralf Seppelt

2. Gutachter: Prof. Dr. Armin Grunwald

Halle (Saale), den 14.04.2016 (Tag der Verteidigung)

Summary

Bioenergy is promoted for its potential to lower greenhouse gas (GHG) emissions, to guarantee energy provisions and to encourage development in biomass sourcing regions. However, several negative social and environmental effects of bioenergy production have been revealed. Therefore, legislation such as the EU Renewable Energy Directive 2009/28/EC (EU RED) includes sustainability requirements. For the practical sustainability assessment, the EU approved certification schemes that biomass producers can chose from to demonstrate compliance with the EU RED. Life-cycle assessments (LCAs) are a major component of certification schemes for the environmental assessment of bioenergy production. LCAs may easily comply with the processing steps of (bioenergy) supply chains concerning global environmental impacts such as GHG emissions. However, local/regional environmental impacts (e.g., water quality or biodiversity) require site-specific and flexible assessments in order to consider the regional capacity of the environment, which LCAs hardly provide.

Against this background, this dissertation aims at answering the question (i) whether certification schemes as major governance mechanism of local/regional environmental impacts allow for a reliable and feasible assessment of biomass production for bioenergy. Beyond, it is the aim to tackle the broader question (ii) whether certification schemes would be able to detect changes of ecological processes caused by biomass production for bioenergy. Certification schemes predominantly used feasible causal indicators (e.g., on management practices) instead of more reliable and less feasible effect indicators (e.g., on water quality changes). Certification schemes rather demonstrated compliance with the EU RED, and lacked an assessment of causal links and interactions in ecological processes affected by increased biomass production. They also did not allow for comparing impacts between different bioenergy feedstock production regions. In addition, their narrow focus on indicators at the plot scale (i.e., land management) may overlook landscape scale impacts of bioenergy production. The developed analytical framework could be used in a revised form as a starting point to assess the quality and comprehensiveness of other environmental monitoring schemes, for instance, in water planning.

For different solid biomass production regions, this dissertation aims at answering the questions (i) how environmental and ecosystem service (ESS) impacts differ for different feedstocks and (ii) which plot and landscape factors drive the supply of multiple ESS. The selected production regions consisted of a current and a future major global solid biomass supply region (Southeastern US and Tanzania) and one major consumer and producer region (Central Germany). The investigated feedstocks were forest biomass, agricultural residues, and biomass from short rotation coppices (SRCs). In the Southeastern US, a major solid biomass production region, this dissertation compared two intensively managed land-use systems under comparable climate conditions: Satilla watershed for pine plantation forestry and Big Sunflower watershed for intensive agriculture for corn and winter wheat production. Plantation forestry showed less distinct ESS tradeoffs compared with natural or semi-natural forest as remnants of potential natural vegetation (PNV). Corn and wheat production provided less carbon storage, phosphorous and sediment retention, and biodiversity, and higher groundwater recharge rates compared with natural or semi-natural forests. ESS were simulated with the Integrated Valuation of Ecosystem Services and Tradeoffs and the Soil-Water Balance model. More than 30 % of the variance in ESS supply depended on landscape structure and naturalness as shown by redundancy analyses. A focus on site conditions and individual feedstock producers at the

plot scale may be insufficient to assess environmental impacts. Certification schemes should be complemented by assessments at the regional/watershed scale. In Central Germany, SRCs as a new solid biomass production option were modeled at the regional scale based on agent-based land-use decisions. Their environmental impact was assessed at the regional scale for different economic and policy scenarios. Only a substantial increase in SRCs' production area (from 0% to 14-24 % of the Mulde watershed in Central Germany) had considerably negative effects on crop production and positive effects on biodiversity and regulating ESS. The increased SRC production considerably exceeded regional demand of solid biomass of combined heat and power plants; the rather low efficiency of SRCs to raise regulating ESS may be explained by the fact that SRCs in the market simulation did not establish at the most efficient locations with respect to regulating ESS supply; instead, they rather established at locations with low soil quality, which were less beneficial for annual crops.

Environmental heterogeneity makes local/regional environmental impacts hardly comparable between world regions. However, it is necessary to compare bioenergy production options between world regions, e.g., for certification schemes. Therefore, this dissertation aims to answer the question (i) how environmental and ESS impacts could be compared despite environmental heterogeneity between different biomass production regions (Southeastern US, Tanzania, and Central Germany). Equally, it should answer the question (ii) how regional environmental thresholds, PNV, and environmental stratification perform in respect to reliability, feasibility, and relevance. The different approaches ranked environmental impacts within a production region according to their naturalness (managed forest, plantation forestry, cropland). When comparing plantation forestry (Southeastern US and Tanzania) and managed forests (Central Germany) between the case studies, managed forests had the most positive environmental impact values similarly identified in the PNV and stratification approach. The PNV approach was the most reliable, although there were major deficiencies concerning feasibility and relevance. The PNV approach explicitly included most of the factors that drive environmental heterogeneity in contrast to the stratification and threshold approach. The stratification approach was more feasible due to the possibility of a concise global application and hardly any additional data needed beyond those to model current environmental impacts. PNV and environmental stratification allowed for a comprehensive application, whereas environmental thresholds did not. Nevertheless, certification schemes or environmental agencies regularly recommend and use environmental thresholds in sustainability assessments.

All assessments approaches and techniques could be applied not only for biomass production systems for bioenergy but also for the assessments of environmental and ESS impacts of other land-use systems. These results of globally distributed environmental assessments of solid biomass for bioenergy and their major governance mechanism, certification schemes, hint to a general challenge that future research may analyze. For increasing global trade of agricultural or forestry products, future research could assess and compare local/regional environmental and ESS impacts with the tested approaches between world regions. In that respect, it would be interesting to link local/regional environmental or ESS assessments with major global biomass trade patterns. It could help to assess how importers use ESS in the exporting location, and thereby affect regional ESS supply and demand relationships.

Zusammenfassung

Bioenergie wird gefördert, um Treibhausgasemissionen (GHG) zu senken, die energetische Versorgungssicherheit zu erhöhen und die regionale Entwicklung in Biomasseproduktionsregionen zu stärken. Jedoch können negative soziale und ökologische Effekte der Bioenergieproduktion auftreten. Aus diesem Grund umfasst die Gesetzgebung, wie die Erneuerbare-Energien-Richtlinie der EU 2009/28/EC (EU RED), Nachhaltigkeitskriterien. Für die praktische Um- und Durchsetzung der Nachhaltigkeitsbewertung hat die EU Zertifizierungssysteme anerkannt, unter denen Biomasseproduzenten wählen können, um die Einhaltung der EU RED nachzuweisen. Ökobilanzen (LCAs) sind ein wesentlicher Bestandteil von Zertifizierungssystemen zur Bewertung der ökologischen Nachhaltigkeit der Bioenergieproduktion. LCAs dienen vor allem der Bewertung weltweiter Umweltfolgen wie Treibhausgasemissionen und sind leicht mit den Verarbeitungsschritten von (Bioenergie-)Wertschöpfungsketten in Einklang zu bringen. Jedoch bedürfen lokale/regionale Umweltfolgen, z. B. im Bereich Wasserqualität oder Biodiversität, Raumbezug und Flexibilität beim Bewertungsansatz, um die regionale ökologische Tragfähigkeit zu berücksichtigen. Raumbezug und methodische Flexibilität sind in LCAs kaum gegeben.

Vor diesem Hintergrund stellt sich die Frage, (i) ob Zertifizierungssysteme als dominierendes Governanceinstrument lokaler/regionaler Umweltfolgen der Biomasseproduktion für Bioenergie eine verlässliche und leicht umsetzbare Umweltbewertung zulassen. Darüber hinaus besteht die Frage, (ii) ob Zertifizierungssysteme Veränderungen ökologischer Prozesse aufgrund von Biomasseerzeugung für Bioenergie aufdecken können. Zertifizierungssysteme verwenden vorrangig leicht umsetzbare Indikatoren zur Messung der Ursachen von Umweltveränderungen, z. B. zu land- und forstwirtschaftlicher Bewirtschaftungspraxis. Kaum verwendet werden hingegen Wirkungsindikatoren, z. B. zur Bestimmung der Wasserqualität. Zertifizierungssysteme zeigen vornehmlich die Einhaltung der EU RED. Sie ermöglichen nur unzureichend eine Analyse der möglichen Veränderungen ökologischer Prozesse aufgrund erhöhter Biomasseerzeugung. Darüber hinaus lassen es Zertifizierungssysteme nicht zu, Umweltfolgen verschiedener Bioenergieproduktionsregionen zu vergleichen. Zusätzlich können aufgrund des beschränkten räumlichen Fokus der Indikatoren auf Schlagebene, d.h. Landbewirtschaftung, Folgen der Bioenergieproduktion auf Landschaftsebene übersehen werden. Der entwickelte analytische Bewertungsrahmen könnte in einer angepassten Form als Ausgangspunkt für die Bewertung der Vollständigkeit und der Qualität von Umweltmonitoringschemata, z. B. in der Gewässerplanung, verwendet werden.

Für verschiedene Produktionsregionen fester Biomasse möchte diese Dissertation, die Fragen beantworten, (i) inwiefern sich verschiedene Biomassesubstrate unterschiedlich auf die Umwelt und auf Ökosystemdienstleistungen (ESS) auswirken und (ii) welche Faktoren auf Schlag- und Landschaftsebene die Bereitstellung mehrerer ESS beeinflussen. Die betrachteten Regionen sind eine aktuelle bzw. eine zukünftige Produktions- und Exportregion (Südosten der USA und Tansania) sowie eine Produktions- und Importregion ((Mittel-)Deutschland) fester Biomasse. Die zu bewertenden Substrate sind Waldbiomasse, Agrarreststoffe und Biomasse aus Kurzumtriebsplantagen (SRCs). Im Südosten der USA als eine bedeutende Exportregion fester Biomasse vergleicht diese Dissertation zwei intensiv bewirtschaftete Landnutzungssysteme mit vergleichbaren klimatischen Bedingungen (Satilla als ein Wassereinzugsgebiet für Kiefernplantagen und Big Sunflower als ein Wassereinzugsgebiet für intensiven Ackerbau für Mais und Winterweizen). Plantagenwälder weisen

im Vergleich zu natürlichen oder naturnahen Wäldern, ihrem heutigen Pendant zu potentieller natürlicher Vegetation (PNV), kaum Unterschiede in Bezug auf die Bereitstellung von ESS auf. Mais und Winterweizen weisen eine deutlich geringere Kohlenstoffspeicherung, Phosphor- und Sedimentrückhaltung sowie Biodiversität im Vergleich zu natürlichen oder naturnahen Wäldern auf. Mais und Winterweizen weisen jedoch eine höhere Grundwasserneubildungsrate als natürliche oder naturnahe Wälder auf. ESS wurden mit dem „Integrated Valuation of Ecosystem Services and Tradeoffs“ und dem „Soil-Water Balance“ Model simuliert. Mehr als 30 % der Varianz in der Bereitstellung von ESS wurde durch die Struktur und Natürlichkeit der Landschaft erklärt, wie Redundanzanalysen gezeigt haben. Der Fokus auf Standortbedingungen und einzelne Biomasseerzeuger auf Schlagebene kann unzureichend sein, um Umweltfolgen zu bewerten. Zertifizierungssysteme sollten Umweltbewertungen auch auf regionaler Ebene durchführen. In Mitteldeutschland wurde die Verbreitung von SRCs als neue Möglichkeit der Biomasseerzeugung mit einem agentenbasierten Modell simuliert. Regionale Umweltfolgen wurden für verschiedene ökonomische und politikgetriebene Szenarien analysiert. Nur ein deutlicher Anstieg der Produktionsfläche von SRCs (von 0 % auf 14-24 % des Mulde Wassereinzugsgebiets in Mitteldeutschland) hat merkliche Effekte auf die Produktion von Feldfrüchten (negativ) und auf die Biodiversität und regulierende ESS (positiv). Der damit verbundene Anbau von SRCs übersteigt die regionale Nachfrage von Kraft-Wärme-Kopplungskraftwerken nach fester Biomasse deutlich. Ein Grund für die geringe Effizienz der SRCs bei der Erhöhung der regulierenden ESS lässt sich dadurch erklären, dass SRCs sich in der Marktsimulation nicht an den Orten der höchsten Effizienz in Bezug auf die Bereitstellung von regulierenden ESS ansiedeln. Stattdessen werden SRCs vornehmlich an den für den Feldfruchtanbau weniger förderlichen Standorten mit geringerer Bodenqualität angebaut.

Naturräumliche Heterogenität lässt einen Vergleich lokaler/regionaler Umweltfolgen zwischen Weltregionen kaum zu. Dennoch ist es notwendig, Möglichkeiten der Bioenergieproduktion zwischen Weltregionen zu vergleichen, z. B. für die Verwendung in Zertifizierungssystemen. Daher hat diese Dissertation das Ziel, die Frage zu beantworten, (i) wie Folgen für die Umwelt und ESS zwischen verschiedenen Biomasseproduktionsregionen verglichen werden können (Südosten der USA, Tansania und Mitteldeutschland). Ebenso soll die Frage beantwortet werden, (ii) wie drei Ansätze, regionale Umweltgrenzwerte, PNV und Stratifizierung anhand von Umweltparametern, in Bezug auf Verlässlichkeit, Umsetzbarkeit und Relevanz zu bewerten sind. Diese drei Ansätze ordnen die verschiedenen Umweltfolgen in einer Produktionsregion nach dem Grad der Natürlichkeit (natürlicher oder naturnaher Wald, Plantagenwald und Ackerland). Im Vergleich zwischen den Plantagenwäldern (Südosten der USA und Tansania) und dem Wirtschaftswald (Mitteldeutschland) wird der Wirtschaftswald mit dem PNV- und Stratifizierungsansatz jeweils als Option mit den höchsten positiven Werten identifiziert. Der PNV-Ansatz ist der verlässlichste mit Nachteilen in Bezug auf Umsetzbarkeit und Relevanz. Im Gegensatz zum Stratifizierungs- und Umweltgrenzwertansatz berücksichtigt der PNV-Ansatz wesentlich mehr Faktoren, die die naturräumliche Heterogenität beeinflussen. Der Stratifizierungsansatz ist aufgrund eines einheitlich zugrundeliegenden Datensatzes und geringem zusätzlichem Datenbedarf leichter umsetzbar. Der PNV-Ansatz und der Stratifizierungsansatz lassen eine umfassende Anwendung im Gegensatz zum Umweltgrenzwertansatz zu. Nichtsdestotrotz empfehlen und verwenden Zertifizierungssysteme oder Umweltbehörden regelmäßig Umweltgrenzwerte zur Nachhaltigkeitsbewertung.

Alle Umweltbewertungsansätze können nicht nur für die Biomasseerzeugung sondern auch für die Bewertung von Umweltfolgen und ESS in anderen Landnutzungssystemen verwendet werden.

Die Ergebnisse der Umweltbewertungen fester Biomasse in weltweit verteilten Fallstudien und zu Zertifizierungssystemen als zugehörigem Governanceinstrument deuten auf eine grundsätzliche Herausforderung für die zukünftige Forschung hin. Der zunehmende weltweite Handel mit land- und forstwirtschaftlichen Produkten erfordert eine Bewertung und einen Vergleich der lokalen/regionalen Umweltfolgen und ESS zwischen Weltregionen. Vor diesem Hintergrund besteht zukünftiger Forschungsbedarf, wie Bewertungsansätze für lokale/regionale Umweltfolgen oder ESS mit weltweiten Handelsströmen verknüpft werden können. Dadurch könnte beispielsweise eine Analyse ermöglicht werden, wie Importeure ESS in der Exportregion nutzen und dadurch die regionale Beziehung von Angebot und Nachfrage für ESS beeinflussen.

Table of contents

List of abbreviations

ABM	Agent-Based Model
AIC	Akaike Information Criterion
ANN	Annual Agricultural Crops
AWC	Available Water Holding Capacity
CAP	Common Agricultural Policy
CHP	Combined Heat and Power
CSBP	Council on Sustainable Biomass Production
C&Is	Criteria and Indicators
DEM	Digital Elevation Model
DM	Dry Matter
DPSIR	Driving Forces – Pressures – States – Impacts – Responses
EI	Environmental Impact
EFA	Ecological Focus Area
ESS	Ecosystem Services
ESTIMAP	Ecosystem Services Mapping at European scale
EU	European Union
EU RED	EU Renewable Energy Directive
FSC	Forest Stewardship Council
GBEP	Global Bioenergy Partnership
GGL	Green Gold Label
GHG	Greenhouse Ggas
GlobalGAP	Global Good Agricultural Practice
HHV	Higher Heating Value
iLUC	Indirect Land-Use Change
InVEST	Integrated Valuation of Environmental Services and Tradeoffs
ISCC	International Sustainability and Carbon Certification
IWPB	Initiative Wood Pellet Buyers

LCA	Life-Cycle Assessment
LUC	Land-Use Change
LU/LC	Land Use/Land Cover
LU/LUC	Land-Use/Land-Use Change
NoC	No Cultivation
NTA	Netherlands Technical Agreement
ODD	Overview, Design Concepts and Detail
OECD	Organisation for Economic Co-operation and Development
PEFC	Programme for the Endorsement of Forest Certification
PNV	Potential Natural Vegetation
RDA	Redundancy Analysis
RSB	Roundtable on Sustainable Biomaterials
SAN	Sustainable Agricultural Network
SD	Standard Deviation
SEM	Standard Error of the Mean
SFI	Sustainable Forestry Initiative
SOC	Soil Organic Carbon
SRC	Short Rotation Coppice
SWB	Soil-Water Balance Model
UN	United Nations
USLE	Universal Soil Loss Equation
2G	Second Generation

1 Introduction

1.1 Motivation

Bioenergy is subsidized and promoted to save greenhouse gas (GHG) emissions, to diversify, secure and create energy supplies, and to create jobs [1-4]. Arguments in favor of bioenergy can be summarized under the concept of sustainability with its three dimensions: economic, social, and environmental. A widely accepted definition for sustainability by the Brundtland Commission was recently updated by Griggs et al. [5] "development that meets the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generations depends". Market mechanisms typically ensure economic sustainability if following the neoclassical theory [6]. However, several negative social and environmental externalities of bioenergy production have been discovered, but are not included in market prices. Therefore, legislation such as the EU Renewable Energy Directive 2009/28/EC (EU RED) [7] is required if sustainability goals should be met to reduce negative externalities.

Historically, environmental assessments of bioenergy production are especially important against the background of the European history of traditional wood and bioenergy use from forests. The historical cycle of forest decline as demonstrated for the Menzer Heide in Eastern Germany, but characteristic for Central Europe [8] started with a forest decline due to overexploitation of roundwood and residues. This was followed by afforestation with fast growing tree species such as *Picea abies* and *Pinus silvestris* comparable to those of plantations. Currently, forest practice in Central Europe is in a state of conservation with measures to re-approach the natural situation with the introduction of broadleaved tree species [9, 8]. Increasing imports might induce or enhance such a cycle in developing, rather than developed countries, with prevailing primary or secondary forests and the economic need for export goods. To avoid or abate such a cycle, a set of rules or requirements, as operationalized in sustainability criteria and indicators (C&Is) in certification schemes, is necessary. Certifications schemes as governance tools have the purpose to avoid negative human alterations in ecological and social systems worldwide [10]. If most stakeholders agree on C&Is at an international level to cover the large imports of solid biomass [11], the trade of unsustainably produced biomass will be more likely delimited.

Considering global biomass production for bioenergy, more than 80 percent of global bioenergy supply originates from forest biomass, whereas only three percent of the global bioenergy crops was obtained from dedicated energy crops in 2008 [12]. A projection for energy crop production for 2050 that considers the competition for land with food crops expects that the potential of bioenergy crops may amount to 77 EJ a^{-1} [13], which is far below the total biomass potential for bioenergy of up to 300 EJ a^{-1} [12]. In addition, Haberl et al. [14] argue that a further significant increase of purposefully grown bioenergy crops will not be economically viable due to the low photosynthetic conversion efficiency. They also question the environmental sustainability of increased bioenergy feedstock production and the associated direct or indirect land-use change and see this as especially problematic for natural lands with high carbon stocks and biodiversity. Considering global bioenergy trade, Matzenberger et al. [15] expect solid biomass to be the predominately traded bioenergy resource ranging from 2,500 to 7000 Mt by 2030. Liquid biofuels are expected to range between 65 to 360 Mt. The case of solid biomass for bioenergy is to be seen exemplary for increasing global trade of biomass and agricultural products and the consequently induced land-use change and its associated environmental, economic, and social impacts, see Meyfroidt et al. [16] for an overview beyond bioenergy. For the EU, Lamers et al. [17] expect an increase in solid biomass imports from 2010 until 2020 by about 300 percent to 236 PJ as a net importer even under potentially implemented EU RED sustainability requirements.

To fulfill the normative dimension of sustainability assessments [18], it is necessary to assess whether biomass production for bioenergy is sustainable. Common tools to assess the environmental sustainability of bioenergy production are life-cycle assessments (LCAs) that focus on GHG emissions and air pollutants [19, 20]. Bioenergy production will be seen as beneficial if GHG emissions are reduced compared with a fossil fuel energy production systems [21]. Contrastingly, there is no constantly applied assessment methodology for site-specific local/regional environmental impacts. There are studies aiming at regionalized LCAs, but they hardly overcome the mismatch of available input data and impact scales [22]. For example, resource flows and life-cycle inventory data on emissions are available for nations, but not for environmental impact scales [22] (e.g., data on water quality impacts at the watershed scale such nutrient export). To solve this deficiency of generic, global LCAs, Koellner and Geyer [23] suggest to rely on and to integrate the results of site-specific and -dependent environmental assessments of production system impacts (e.g., for biodiversity and ecosystem services (ESS)). Koellner et al. [24] especially criticize that LCAs do not consider interactions between different ESS and biodiversity or the influence of landscape configuration. For bioenergy used in the EU, the EU RED is in force for liquid biofuels and in a draft form for solid and gaseous bioenergy production [25]. It requires amongst others to assess the environmental impact of biomass production on biodiversity, and on ESS. However, site-specific and regional environmental impacts or ESS miss a consistently applicable reference system or reference conditions for a comparison between worldwide biomass production regions for a sustainability assessment.

In recent years, governments in more than 60 countries, including the US, China, and several countries from the EU, have started to implement or use the ESS concept in legislative acts or in national assessments [26]. ESS are defined as human benefits provided by ecosystems [27]. ESS are classified in three major groups: provisioning (e.g., biomass for bioenergy), regulating (e.g., sediment retention), and cultural (e.g., scenic beauty) [28]. ESS have been modeled or mapped in various studies within (e.g., Meehan et al. [29]) and beyond the scope of bioenergy (e.g., Raudsepp-Hearne et al. [30]). Often, studies map ESS and aim at identifying synergies and tradeoffs such as Qiu and Turner [31]. A major rational of ESS assessments instead of (common) environmental assessments is that the impact of a policy or human action on the ESS supply for the beneficiaries needs to be considered [32]. A policy such as the EU RED needs to be assessed not only for biophysical impacts but also for its social impacts. Following Baker et al. [33], the focus is often on immediately lost economic benefits. For example, in the context of bioenergy, short rotation coppices (SRCs) planted as buffer strips along rivers do not only have economic impacts, e.g., biomass yields instead of crop yields, but also other non-monetary environmental such as nutrient retention and social impacts such as loss of food production [34]. Initially, ESS impacts of bioenergy have been mostly analyzed in the context of first generation liquid biofuels such as maize in the US or rapeseed in Germany [35], although solid biomass is dominant in trade and production as shown above.

1.2 Conceptual background and main research questions

Environmental and social sustainability of bioenergy, in contrast to economic sustainability, are often not ensured through market mechanisms. To avoid market failures such as negative environmental and social externalities, government interventions in the form of quotas for biofuels or subsidies, or both, are required [36]. Voluntary certification schemes to reduce or avoid negative environmental and social externalities have a long history of application in the forestry sector to solve the asymmetry of information between feedstock producers and users [37]. Certification schemes reduce governmental transaction costs by shifting the audit effort to the scheme issuer. However, strongly relying on a market-driven approach without actual government-set minimum standards or the risk to lose government control, especially in third countries, is debatable. Both community pressure and likely state intervention for unsustainable practices are largely missing especially in developing countries exporting biomass [38]. The relevant criteria might exceed the local requirements for bioenergy sustainability or set other foci than locally intended. According to Van Dam et al. [10], developing countries have food security rather than environmental sustainability in focus. Beyond the regional aspect, the goals for certification schemes vary between stakeholder groups [39]. For example, NGOs expectedly prefer reliable and comprehensive certifications schemes for a high level of social and environmental sustainability. Contrastingly, feedstock producers are more likely interested in easily applicable certification schemes that allow for a higher market penetration to comply with consumer needs, or to gain access to subsidies within the EU RED [40, 2, 41]. Bioenergy certifications schemes, approved for compliance with the EU RED [7], combine government defined minimum standards for environmental protection, and fuzzy generic environmental and social requirements. The scheme issuer has the freedom to translate and refine fuzzy generic requirements to C&Is. The need to obtain good agricultural and environmental conditions or to fulfill the Conventions of the International Labour Organisation, are examples of fuzzy requirements in the EU RED. Due to this freedom for the certification scheme issuer, an increasing number of certification schemes likely raises the number of (diverging) levels of comprehensiveness and quality [42].

The assessment and governance of social (human-human interactions) and environmental (human-environment interactions) sustainability issues through certification can be conceptualized in socio-ecological systems. According to Berkes et al. [43], socio-ecological systems are linked, multilayered systems with a social and ecological entity or component. They supply crucial services for society such as nutrition or biomass for various purposes. Binder et al. [44] categorized existing frameworks to analyze socio-ecological systems in four groups: ecocentric, integrative, policy, and vulnerability frameworks. Ecocentric frameworks suit to analyze the impact of human activities on ecosystems. Integrative frameworks focus on the dynamics in the social system and the bidirectional interaction of the social and ecological system. Policy frameworks explore options to lower the human impact on ecosystems. Vulnerability frameworks identify how susceptible humans are to environmental change. The Driving force-Pressure-State-Impact-Response (DPSIR) framework [45] as policy framework should allow to assess whether certification schemes can contribute to lowering environmental impacts of biomass production. Specifically, the DPSIR may distinguish whether indicators assess causes or effects of biomass production on the environment and thereby determine the reliability of indicators. Beyond this, rating scales to determine the feasibility of indicators, i.e., measurability and practicality [46], are needed. The ecocentric ESS concept may be used to determine considered and missing environmental processes in biomass production systems. In that respect, this dissertation may determine whether certification schemes well-represent human induced changes of biomass production. It is necessary to test whether certifications schemes properly represent the ecological system and its human modification. Therefore, ESS frameworks typically link ecosystem structures and processes with an ecosystem's capacity to supply ESS, human benefits and values, see Villamagna et al. [47] for a synthesis of concepts. These frameworks could help to operationalize the use of the ESS concepts [48] to evaluate certification schemes. In this dissertation, the DPSIR and ESS frameworks are applied to assess reliability and comprehensiveness of certification schemes for an environmental assessment:

Do certification schemes allow for a reliable and feasible assessment of local/regional environmental impacts of bioenergy feedstock production? Do individual indicators and the entire set of a certification scheme well represent the human impact on the environment? Could certification schemes detect changes of ecological processes caused by biomass production for bioenergy?

Biomass production for bioenergy may affect ecosystems at a range of spatial scales, from plants through ecosystems, landscapes and biomes to the globe as listed in Levin [49]. The underlying ecosystem structures and processes to determine the actual capacity of ESS supply act at various spatial and temporal scales [50]. The spatial scales in ESS assessments often do not match the actual affected processes that determine the capacity of ESS supply [51]. For example, water use for different bioenergy feedstocks (e.g., forest plantations or crop production) may be assessed at the plot scale, but effects on the water balance are felt at the regional or watershed scale. In that respect, both the regional impact of water use and the regional availability, or capacity, need to be included in the assessment. Therefore, it might be reasonable to assess ESS at a regional or landscape scale as recommended for certification schemes in forestry [52]. The demand for regulating ESS often also arises at the regional or landscape scale [53]. This is due to the fact that most of the environmental processes such as runoff, soil erosion, or nutrient export, and their associated ESS occur at that scale. In that respect, Dale et al. [54] emphasize the need for assessments considering the impact of bioenergy on the environment at the regional scale, for instance, with water quality impacts at the watershed level.

Typically, land use such as biomass production for bioenergy is often associated with trade-offs towards the supply of other ESS and therewith affects the multifunctionality of landscapes [55]. For example, few beneficiaries maximize one provisioning service such as biomass for bioenergy. These and others benefit less from ESS with public good character without immediate private benefit such as erosion protection. Private interests are typically equally fulfilled at the local scale, whereas the losers, beneficiaries of ESS not using the maximized provisioning service, might prefer a bundle of ESS at different scales such as a watershed [56] or a region. Therefore, a regional analysis of ESS supply in a bioenergy feedstock production region may consider spatially explicit interactions with ecosystems and remaining land-use activities in a landscape [50, 57].

Under natural conditions, various ecological processes shape landscape composition and configuration [58]. Land-use activities such as plantation forestry or crop production do not only modify site conditions and ESS supply at plot scale, but also have impacts at landscape scale. Land-use activities may enhance the fragmentation connectivity of natural or semi-natural landscapes [50]. Modifying landscape structure affects ecosystem functioning such as energy and matter transport [59] and ESS [60, 31] (e.g., sediment retention). Identifying major landscape characteristics of balanced bundles of high ESS supply in bioenergy landscapes may allow to derive indicators and management recommendations to ensure efficient landscape planning. Existing research hardly analyzes driving factors determining ESS bundles [61].

Advances in processing technologies for biomass for bioenergy may enhance the interchangeable use of different biomass sources. Beyond the largely debated competition between food, fuel and fiber, bioenergy producers may also compete for forestry and agricultural biomass interchangeably [62]. Existing research regularly addresses environmental and ESS impacts of bioenergy from forestry, agriculture or second generation bioenergy feedstocks such as SRCs separately, see Cademus et al. [63] (forestry), Power [64] (agriculture), or Holland et al. [65] (SRCs). A common market for these different biomass sources urges for a holistic approach and tools for integrative environmental assessment. In that respect, it is necessary to apply a methodology suitable to compare agricultural and forestry dominated production regions and to test how an increase of second generation bioenergy feedstocks would affect multiple ESSs:

How do impacts on ESS differ between solid bioenergy feedstock, i.e., from forestry, agriculture, and SRCs, at the regional scale? Which landscape scale factors affect ESS synergies and trade-offs?

Environmental and ESS impacts at the local/regional scale may affect a broad variety of ecosystems, which are differently affected by human-induced changes such as land-use change due to increased biomass trade. The variety of ecosystems is mostly driven by the spatial variation of biotic and abiotic environmental factors, namely environmental heterogeneity. Environmental heterogeneity is a major driver of species richness, and is commonly considered in species distribution modeling in biodiversity research [66]; it equally affects the occurrence of positive and negative environmental impacts [67] and their respective reference conditions. With respect to environmental impact assessments, no generally applicable methodology has been identified [68]. In that respect, it is necessary to develop and evaluate potential options to enable comparative environmental impact assessments. It is possible to compare environmental impact assessments and to determine the environmental sustainability for different biomass production regions by standardizing the environmental impacts, by having a reference state, or by defining threshold or target values. There are some approaches such as environmental stratification proposed to make biodiversity impacts comparable for worldwide biodiversity monitoring networks [69]. Potential natural vegetation (PNV) can be used as reference condition to compare carbon sequestration worldwide [70] or for LCAs [71]. Equally, threshold or target values to determine critical nutrient loads could be used [18].

For ESS, there are several conceptual frameworks, which propose different ways to quantify an ecosystem’s capacity or reference conditions to provide ESS [72, 47]. Both studies, synthesizing various frameworks and approaches to operationalize the question of sustainable ESS provision and use, could not identify a generally applicable methodology for an entire bundle of ESS. However, the authors agree on the statement that a sustainable use of ESS is ensured if it is lower than the capacity to provide ESS. Generally, it is possible to model ESS capacity and use or supply with different indicators, e.g., Schröter et al. [73], or to use environmental thresholds or target values, e.g., for water purification [47]. However, all options need to control for environmental heterogeneity to determine how and to which extent land use affects the ESS capacity relative to the actual ESS demand or supply, see Villamagna et al. [47]. In practice, a broader application, for instance, in certification schemes, would require a simplified or at least a well-guided approach [53]. These issues lead to the following questions:

How can local/regional environmental or ESS impacts be compared despite environmental heterogeneity? What would be the outcome for exemplary biomass production regions? Which criteria determine the reliability, feasibility and relevance of the applied methodologies to compare environmental and ESS impacts under environmental heterogeneity? Which approach is preferable with respect to reliability, feasibility, and relevance?

1.3 Overview of dissertation structure

This dissertation comprises six chapters. Chapters 2, 3, 4 and 5 represent the manuscripts for the journal articles for this cumulative dissertation.

Chapter 2 analyzes 12 certification schemes and indicator sets as common tools for the assessment of (environmental) sustainability for bioenergy feedstock production in practice. It assesses the quality and comprehensiveness of C&Is in certification schemes for local/regional environmental impacts, which globally targeted and well-developed LCAs often disregard. First, rating scales to assess the reliability and feasibility are developed and applied for each environmental indicator. Secondly, an analytical framework based on the ESS cascade is used to analyze how each indicator set represents potentially affected environmental systems. This second analysis tests how well certification schemes represent tradeoffs of biomass use and other ESS respectively [46].

Chapter 3 assesses potential ESS supply and its tradeoffs and synergies in solid biomass production systems in the Southeastern US, a major global solid bioenergy production region for EU imports. First, ESS tradeoffs and synergies of plantation forestry (i.e., pine poles) and agricultural production (i.e., wheat straw and corn stover) with the counterfactual natural or semi-natural forest are analyzed in two representative watersheds. Secondly, environmental factors at the plot scale and landscape composition, configuration, and naturalness are tested for their influence on ESS supply and biodiversity. Thirdly, the role of landscape factors to meet partly socially accepted environmental thresholds as available for water quality within the two contrasting case studies is assessed [74].

Chapter 4 places SRCs in the landscape of the Mulde watershed in Central Germany by modeling farmers' land-use decisions under different economic and policy-driven scenarios using an agent-based model (ABM). Based on the simulated deployment of SRCs, regional-scale impacts on multiple ESS despite missing large-scale, commercial implementation of SRCs can be assessed. This chapter also tests whether SRCs enhance the number of balanced ESS bundles [75].

Chapter 5 compares environmental impacts between biomass production systems in different parts of the world: the Southeastern US, Tanzania and Central Germany. This chapter tests three approaches to make environmental impacts comparable despite environmental heterogeneity: environmental stratification, PNV, and environmental thresholds for an environmental assessment. It develops criteria for reliability, feasibility, and relevance, and evaluates the three approaches according to the developed criteria [76].

Chapter 6 discusses the findings from chapters 2 to 5 in a broader context and shows limitations. Conclusions with respect to future research needs are also drawn.

1.4 References

[1] Demirbas A. Political, economic and environmental impacts of biofuels: A review. Applied Energy 2009;86(Suppl 1):S108-S17.
[2] Lewandowski I, Faaij APC. Steps towards the development of a certification system for sustainable bio-energy trade. Biomass and Bioenergy 2006;30(2):83-104.
[3] Ragauskas AJ, Williams CK, Davison BH, Britovsek G, Cairney J, Eckert CA, et al. The path forward for biofuels and biomaterials. Science 2006;311(5760):484-9.
[4] WG III. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, et al., editors. Cambridge and New York: Cambridge University Press; 2012.
[5] Griggs D, Stafford-Smith M, Gaffney O, Rockstrom J, Ohman MC, Shyamsundar P, et al. Policy: Sustainable development goals for people and planet. Nature 2013;495(7441):305-7.
[6] Cohen B, Winn MI. Market imperfections, opportunity and sustainable entrepreneurship. Journal of Business Venturing 2007;22(1):29-49.
[7] Directive 2009/28/EC. On the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.2009.04.23. Off J EU 2009. L 140: 16-62.
[8] Zerbe S. Restoration of natural broad-leaved woodland in Central Europe on sites with coniferous forest plantations. Forest Ecology and Management 2002;167(1-3):27-42.
[9] Farrell EP, Führer E, Ryan D, Andersson F, Hüttl R, Piussi P. European forest ecosystems: building the future on the legacy of the past. Forest Ecology and Management 2000;132(1):5-20.
[10] Van Dam J, Junginger M, Faaij APC. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renewable and Sustainable Energy Reviews 2010;14(9):2445-72.
[11] Sikkema R, Steiner M, Junginger M, Hiegl W, Hansen MT, Faaij A. The European wood pellet markets: current status and prospects for 2020. Biofuels, Bioproducts and Biorefining 2011;5(3):250-78.
[12] Chum H, Faaij A, Moreira J, Berndes G, Dhamija P, Dong H, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, et al., editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press; 2011.
[13] Erb K-H, Haberl H, Plutzar C. Dependency of global primary bioenergy crop potentials in 2050 on food systems, yields, biodiversity conservation and political stability. Energy Policy 2012;47(0):260-9.
[14] Haberl H, Erb K-H, Krausmann F, Running S, Searchinger TD, Smith WK. Bioenergy: how much can we expect for 2050? Environmental Research Letters 2013;8(3):031004.
[15] Matzenberger J, Kranzl L, Tromborg E, Junginger M, Daioglou V, Sheng Goh C, et al. Future perspectives of international bioenergy trade. Renewable and Sustainable Energy Reviews 2015;43(0):926-41.
[16] Meyfroidt P, Lambin EF, Erb K-H, Hertel TW. Globalization of land use: distant drivers of land change and geographic displacement of land use. Current Opinion in Environmental Sustainability 2013;5(5):438-44.
[17] Lamers P, Hoefnagels R, Junginger M, Hamelinck C, Faaij A. Global solid biomass trade for energy by 2020: an assessment of potential import streams and supply costs to North-West Europe under different sustainability constraints. GCB Bioenergy 2015;7(4):618-34.
[18] Binder CR, Feola G, Steinberger JK. Considering the normative, systemic and procedural dimensions in indicator-based sustainability assessments in agriculture. Environmental Impact Assessment Review 2010;30(2):71-81.

[19] Efroymson RA, Dale VH, Kline KL, McBride AC, Bielicki JM, Smith RL, et al. Environmental indicators of biofuel sustainability: What about context? Environmental Management 2013;51(2):291-306.
[20] Muench S, Guenther E. A systematic review of bioenergy life cycle assessments. Applied Energy 2013;112(0):257-73.
[21] Cherubini F, Bird ND, Cowie A, Jungmeier G, Schlamadinger B, Woess-Gallasch S. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. Resources, Conservation and Recycling 2009;53(8):434-47.
[22] Hellweg S, Milà i Canals L. Emerging approaches, challenges and opportunities in life cycle assessment. Science 2014;344(6188):1109-13.
[23] Koellner T, Geyer R. Global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 2013;18(6):1185-7.
[24] Koellner T, de Baan L, Beck T, Brandão M, Civit B, Margni M, et al. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 2013;18(6):1188-202.
[25] Directive 2013/draft/EU Directive of the European parliament and of the council on sustainability criteria for solid and gaseous biomass in electricity and/or heating and cooling and biomethane injected in the natural gas network. 2013.
[26] Wong CP, Jiang B, Kinzig AP, Lee KN, Ouyang Z. Linking ecosystem characteristics to final ecosystem services for public policy. Ecology Letters 2015;18(1):108-18.
[27] MEA Board. Ecosystems and human well-being : current state and trends : findings of the condition and trends working group. Hassan R, Scholes R, Ash N, editors. Washington, DC: Island Press; 2005.
[28] CICES. Common International Classification of Ecosystem Services. 2013 [cited 2013 15 January]; Available from: http://cices.eu/.
[29] Meehan TD, Gratton C, Diehl E, Hunt ND, Mooney DF, Ventura SJ, et al. Ecosystem-Service Tradeoffs Associated with Switching from Annual to Perennial Energy Crops in Riparian Zones of the US Midwest. PLoS ONE 2013;8(11):e80093.
[30] Raudsepp-Hearne C, Peterson GD, Bennett EM. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences 2010;107(11):5242-7.
[31] Qiu J, Turner MG. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proceedings of the National Academy of Sciences 2013;110(29):12149-54.
[32] Helming K, Diehl K, Geneletti D, Wiggering H. Mainstreaming ecosystem services in European policy impact assessment. Environmental Impact Assessment Review 2013;40(0):82-7.
[33] Baker J, Sheate WR, Phillips P, Eales R. Ecosystem services in environmental assessment — Help or hindrance? Environmental Impact Assessment Review 2013;40(0):3-13.
[34] Manning P, Taylor G, E. Hanley M. Bioenergy, Food Production and Biodiversity – An Unlikely Alliance? GCB Bioenergy 2015;7(4):570-6.
[35] Gasparatos A, Stromberg P, Takeuchi K. Biofuels, ecosystem services and human wellbeing: Putting biofuels in the ecosystem services narrative. Agriculture, Ecosystems & Environment 2011;142(3–4):111-28.
[36] Hill J, Nelson E, Tilman D, Polasky S, Tiffany D. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. Proceedings of the National Academy of Sciences 2006;103(30):11206-10.
[37] Rametsteiner E. The role of governments in forest certification—a normative analysis based on new institutional economics theories. Forest Policy and Economics 2002;4(3):163-73.
[38] van Kooten GC, Nelson HW, Vertinsky I. Certification of sustainable forest management practices: a global perspective on why countries certify. Forest Policy and Economics 2005;7(6):857-67.

[39] van Dam J, Junginger M. Striving to further harmonization of sustainability criteria for bioenergy in Europe: Recommendations from a stakeholder questionnaire. Energy Policy 2011;39(7):4051-66.
[40] Junginger M, van Dam J, Zarrilli S, Ali Mohamed F, Marchal D, Faaij A. Opportunities and barriers for international bioenergy trade. Energy Policy 2011;39(4):2028-42.
[41] Van Dam J, Junginger M, Faaij A, Jürgens I, Best G, Fritsche U. Overview of recent developments in sustainable biomass certification. Biomass and Bioenergy 2008;32(8):749-80.
[42] Scarlat N, Dallemand J-F. Recent developments of biofuels/bioenergy sustainability certification: A global overview. Energy Policy 2011;39(3):1630-46.
[43] Berkes F, Folke C, Colding J. Linking social and ecological systems: management practices and social mechanisms for building resilience: Cambridge University Press; 1998.
[44] Binder CR, Hinkel J, Bots PWG, Pahl-Wostl C. Comparison of Frameworks for Analyzing Social-ecological Systems. Ecology and Society 2013;18(4).
[45] Niemeijer D, de Groot R. Framing environmental indicators: moving from causal chains to causal networks. Environment, Development and Sustainability 2008;10(1):89-106.
[46] Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65:151-69.
[47] Villamagna AM, Angermeier PL, Bennett EM. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity 2013;15(0):114-21.
[48] Reyers B, Biggs R, Cumming GS, Elmqvist T, Hejnowicz AP, Polasky S. Getting the measure of ecosystem services: a social–ecological approach. Frontiers in Ecology and the Environment 2013;11(5):268-73.
[49] Levin SA. The problem of pattern and scale in ecology. Ecology 1992;73(6):1943-67.
[50] Birkhofer K, Diehl E, Andersson J, Ekroos J, Früh-Müller A, Machnikowski F, et al. Ecosystem services – current challenges and opportunities for ecological research. Frontiers in Ecology and Evolution 2015;2.
[51] de Groot RS, Alkemade R, Braat L, Hein L, Willemen L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 2010;7(3):260-72.
[52] Meijaard E, Wunder S, Guariguata MR, Sheil D. What scope for certifying forest ecosystem services? Ecosystem Services 2014;7(0):160-6.
[53] Geijzendorffer IR, Roche PK. The relevant scales of ecosystem services demand. Ecosystem Services 2014;10(0):49-51.
[54] Dale VH, Kline KL, Wright LL, Perlack RD, Downing M, Graham RL. Interactions among bioenergy feedstock choices, landscape dynamics, and land use. Ecological Applications 2011;21(4):1039-54.
[55] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global Consequences of Land Use. Science 2005;309(5734):570-4.
[56] Howe C, Suich H, Vira B, Mace GM. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. Global Environmental Change 2014;28(0):263-75.
[57] Howard DC, Burgess PJ, Butler SJ, Carver SJ, Cockerill T, Coleby AM, et al. Energyscapes: Linking the energy system and ecosystem services in real landscapes. Biomass & Bioenergy 2013;55:17-26.
[58] Turner MG. Landscape Ecology: The Effect of Pattern on Process. Annual Review of Ecology and Systematics 1989;20(1):171-97.
[59] Blaschke T. The role of the spatial dimension within the framework of sustainable landscapes and natural capital. Landscape and Urban Planning 2006;75(3–4):198-226.

[60] Frank S, Fürst C, Witt A, Koschke L, Makeschin F. Making use of the ecosystem services concept in regional planning—trade-offs from reducing water erosion. Landscape Ecology 2014:1-15.
[61] Mouchet MA, Lamarque P, Martín-López B, Crouzat E, Gos P, Byczek C, et al. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change 2014;28(0):298-308.
[62] Scarlat N, Dallemand J-Fo, Banja M. Possible impact of 2020 bioenergy targets on European Union land use. A scenario-based assessment from national renewable energy action plans proposals. Renewable and Sustainable Energy Reviews 2013;18(0):595-606.
[63] Cademus R, Escobedo F, McLaughlin D, Abd-Elrahman A. Analyzing Trade-Offs, Synergies, and Drivers among Timber Production, Carbon Sequestration, and Water Yield in Pinus elliotii Forests in Southeastern USA. Forests 2014;5(6):1409-31.
[64] Power AG. Ecosystem services and agriculture: tradeoffs and synergies; 2010.
[65] Holland RA, Eigenbrod F, Muggeridge A, Brown G, Clarke D, Taylor G. A synthesis of the ecosystem services impact of second generation bioenergy crop production. Renewable and Sustainable Energy Reviews 2015;46:30-40.
[66] Stein A, Kreft H. Terminology and quantification of environmental heterogeneity in species-richness research. Biological Reviews 2015;90(3):815-36.
[67] Kienast F, Bolliger J, Potschin M, de Groot R, Verburg P, Heller I, et al. Assessing Landscape Functions with Broad-Scale Environmental Data: Insights Gained from a Prototype Development for Europe. Environmental Management 2009;44(6):1099-120.
[68] Acosta-Alba I, Van der Werf H. The Use of Reference Values in Indicator-Based Methods for the Environmental Assessment of Agricultural Systems. Sustainability 2011;3(2):424-42.
[69] Metzger MJ, Brus DJ, Bunce RGH, Carey PD, Gonçalves J, Honrado JP, et al. Environmental stratifications as the basis for national, European and global ecological monitoring. Ecological Indicators 2013;33(0):26-35.
[70] West PC, Gibbs HK, Monfreda C, Wagner J, Barford CC, Carpenter SR, et al. Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. Proceedings of the National Academy of Sciences 2010;107(46):19645-8.
[71] Alvarenga RF, Erb K-H, Haberl H, Soares S, van Zelm R, Dewulf J. Global land use impacts on biomass production—a spatial-differentiated resource-related life cycle impact assessment method. The International Journal of Life Cycle Assessment 2015;20(4):440-50.
[72] Geijzendorffer IR, Martín-López B, Roche PK. Improving the identification of mismatches in ecosystem services assessments. Ecological Indicators 2015;52(0):320-31.
[73] Schröter M, Barton DN, Remme RP, Hein L. Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway. Ecological Indicators 2014;36(0):539-51.
[74] Meyer MA, Chand T, Priess JA. Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLoS ONE 2015;10(3):e0116336.
[75] Schulze J, Frank K, Priess JA, Meyer MA. Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLoS ONE under review.
[76] Meyer MA, Seppelt R, Witing F, Priess JA. How can we compare environmental and ecosystem service assessments for biomass production systems in different parts of the world? Environmental Research Letters under review.

2 Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts

This chapter is published by Elsevier:

Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014; 65:151-69.

The definitive version is available at:
http://www.sciencedirect.com/science/article/pii/S0961953414001706.

2.1 Abstract

Bioenergy is receiving increasing attention because it may reduce greenhouse gas emissions, secure and diversify energy supplies and stimulate rural development. The environmental sustainability of bioenergy production systems is often determined through life-cycle assessments that focus on global environmental effects, such as the emission of greenhouse gases or air pollutants. Local/regional environmental impacts, e.g., the impacts on soils or on biodiversity, require site-specific and flexible options for the assessment of environmental sustainability, such as the criteria and indicators used in bioenergy certification schemes.

In this study, we compared certification schemes and assessed the indicator quality through the environmental impact categories, using a standardized rating scale to evaluate the indicators. Current certification schemes have limitations in their representation of the environmental systems affected by feedstock production. For example, these schemes predominantly use feasible causal indicators, instead of more reliable but less feasible effect indicators. Furthermore, the comprehensiveness of the depicted environmental systems and the causal links between human land use activities and biophysical processes in these systems have been assessed. Bioenergy certification schemes seem to demonstrate compliance with underlying legislation, such as the EU Renewable Energy Directive, rather than ensure environmental sustainability. Beyond, certification schemes often lack a methodology or thresholds for sustainable biomass use. Lacking thresholds, imprecise causal links and incomplete indicator sets may hamper comparisons of the environmental performances of different feedstocks. To enhance existing certification schemes, we propose combining the strengths of several certification schemes with research-based indicators, to increase the reliability of environmental assessments.

2.2 Introduction

Bioenergy is receiving increasing attention because it is assumed to be associated with the following major advantages over fossil fuels [1-4]:

- Reduction of greenhouse gas (GHG) emissions and strengthening of the environmental sustainability of energy provision
- Securing and diversifying the energy supply
- Positive socioeconomic impacts such as increased energy access in developing and jobs in developed countries

The arguments in favor of bioenergy can be summarized under the concept of sustainability as defined by the Brundtland Commission [5]. The aspects listed above show that several dimensions of sustainability are of importance, namely the economic, environmental and social dimensions [6]. According to neoclassical theory, economic sustainability is ensured through market mechanisms [7]. Environmental and social sustainability are often not ensured through these mechanisms and require government interventions, for example, quotas for bioenergy or subsidies to overcome market failures [8]. Even if environmental and social sustainability are considered for bioenergy, Robbins [9] stated that it is currently unclear how to assess the sustainability of bioenergy from both environmental and socioeconomic perspectives.

The major environmental impact categories of bioenergy feedstock production have been summarized to GHG emissions, air pollutants, soil quality, water quality, water availability or quantity, biodiversity and land-use and land-use change (LU/LUC) based on scientific [10-13] and broader stakeholder panels [14].To a great extent, the environmental sustainability of bioenergy production systems is evaluated with well established life-cycle assessments (LCAs), assessing large-scale or globally occurring environmental effects, such as GHG emissions or air pollutants, along the major steps of the supply chain [10, 15]. The highly site-specific and locally/regionally occurring environmental impacts of feedstock production in the first step of most of the bioenergy supply chains are difficult to assess in LCAs. Impacts on soil quality, biodiversity and land use change, water availability and water quality [16, 17] are often insufficiently covered. These limitations comprise necessary but missing regional thresholds to ensure the stability of the ecological system. Such thresholds are not easily integrated into highly standardized LCAs. Existing LCAs assessing environmental impacts often disregard the interaction for example between different regulating ecosystem services (ESS) and biodiversity, such as the buffering capacity of environmental impacts of agriculture or forestry [18, 19]. In the context of bioenergy feedstocks and sustainability, this type of assessment of interactions is supposed to extend the EU RED, i.e. the provision of “basic ecosystem services” such as erosion control should be accounted for if biomass is produced for bioenergy [20]. Dale et al. [21] recommend to determine water quality and soil quality impacts of bioenergy feedstock production in addition to LCAs, e.g. nutrient export to water bodies or soil loss. The regional water quality assessment may will more likely allow to determine, whether the regional thresholds of nutrient exports to maintain good ecological status of water bodies are met.

Site-specific and flexible options for the assessment of local/regional environmental impacts and other aspects of sustainability could be sets of criteria and indicators (C&Is) as used in certification schemes. Such a site-dependent audit approach allows assessing the environmental impacts and their interactions mentioned above. C&Is are currently under development or are in an early stage of implementation for bioenergy but have been extensively applied for a longer period to other products from forestry or agriculture. Examples of C&Is are the Forest Stewardship Council (FSC) for timber or the Sustainable Agriculture Network (SAN) as a label for Good Agricultural Practices [2]. Especially FSC provides nationally or regionally adapted indicator sets [22]. Several bioenergy certification schemes are used to demonstrate compliance with the EU Renewable Energy Directive 2009/28/EC (EU RED) [23].

Despite the common aim of EU RED compliance for most of the bioenergy schemes, an increasing number of alternative schemes may contribute to confuse stakeholders and decrease the acceptance of certification schemes in general [24, 12]. On the one hand, comprehensive and clearly defined requirements may exclude producer groups [2], e.g. in developing countries, and augment certification costs due to increasing effort, such as audits. On the other hand, vaguely defined and less comprehensive schemes may allow for a higher market penetration, but more likely disregard major environmental or social impacts and are not acknowledged by NGOs [25, 26]. An increase in EU imports of biomass for bioenergy might induce or enhance deforestation in countries with prevailing primary forests [27] and the need to export goods. Thus, overexploitation is more likely to occur in developing countries than in developed countries. To avoid or abate e.g. deforestation, a set of C&Is must be agreed upon internationally to cover international biomass trade [28]. International criteria might exceed the local requirements for bioenergy sustainability or might set foci other than the locally intended ones [29]; e.g., criteria might focus on environmental aspects in developed countries, such as sequestering carbon or halting biodiversity loss instead of ensuring food security in developing countries [13]. Such potential discrepancies may provide additional obstacles for implementation.

Beyond existing reviews [29, 2, 12, 26, 13], this paper, assesses the comprehensiveness and quality of indicators used by bioenergy, forestry and agricultural certification schemes. Against the background of conflicting goals for bioenergy certification discussed above, we develop and apply standardized rating scales for indicators grouped into six environmental impact categories to identify their reliability and feasibility. We focus on local/regional environmental impacts, which require site-specific information, affect predominately the local/regional environment and are usually not covered by LCAs. Beyond rating the individual indicators, certification schemes are evaluated at the scheme level based on the ESS cascade [30] to analyze their comprehensiveness and the quality of the representation of the potentially affected environmental system. The aim is to test whether certification schemes are able to show trade-offs between biomass use and other ecosystem services.

2.3 Material and methods

2.3.1 Selection of certification schemes and indicator sets

In this paper, indicator sets for certification have been selected for evaluation. We used sets from bioenergy, agriculture and forestry. The latter two have the advantage of a much longer lasting application of C&Is. Concentrating on the currently rather limited number of specific schemes for bioenergy would have led to a very small set of C&Is, ignoring relevant and important C&Is applied in related sectors.

First, the EU might consider the extension of bioenergy specific with forestry schemes as a relevant policy option for solid biomass for bioenergy in the EU, e.g., by using additional forestry indicators for sustainability certification [31]. Therefore, an evaluation of studies is conducted, assessing the environmental impacts of forest management with a focus on bioenergy production. To identify major characteristics of forestry certification schemes, we selected the FSC and the Sustainable Forestry Initiative (SFI), a major scheme of the meta-standard “Programme for the Endorsement of Forest Certification” (PEFC), which are globally dominating and largely applied certification schemes in forestry [2, 32]. We avoided meta-standards since they typically do not have indicators sets for the actual environmental assessment.

Secondly, new technologies to enhance the transport, storage and co-firing characteristics, such as torrefaction, are under development. These technologies might create additional feedstock options, for instance agricultural residues , such as straw, shells and others, which currently may be used to a limited extent [33]. Therefore, overarching and globally applied agricultural certification schemes, i.e. SAN and Global Good Agricultural Practice (GlobalGAP), are needed to cover feedstocks not targeted by bioenergy certification schemes, predominately aiming at selected bioenergy crops. The relevance of agricultural certification schemes show NTA8080 and other bioenergy certification schemes as they use agricultural certification schemes, which we also selected in this paper, to ensure compliance with environmental sustainability requirements [13]. Despite the fact that GBEP is no operational certification scheme, we included it in our assessment since its indicator set reflects the consensus of numerous governments and international institutions and because it is a framework to assess bioenergy sustainability [12].

2.3.2 Requirements and rating scale for indicator evaluation

The major requirements for indicators are reliability and conceptual soundness, feasibility, i.e., measurability and practicality, and relevance for the end user [34, 35, 2, 36]. The requirements for an indicator discussed in this section are rated on a five step scale. Bockstaller et al. [34] have demonstrated the methodological suitability of such an approach at the indicator level by evaluating sets of agri-environmental indicators for crop production and farming systems, which are methodologically comparable to the certification scheme indicators evaluated in this paper.

We rate the individual indicators for feasibility in three requirement subcategories and for reliability in four requirement subcategories, two exemplary requirement subcategories each are listed in Table 2.1 and the remaining ones in Appendix A. The first rated subcategory for reliability is the *Indicator type* [34, 37]. For practical implementation, we followed the logic of the **D**riving forces - **P**ressures - **S**tates - **I**mpacts - **R**esponses (DPSIR) framework of the European Environment Agency [36], extending preceding frameworks, such as the **P**ressure-**S**tate-**R**esponse framework, applied by the OECD and the UN [38]. We present an application example for the DPSIR framework for rising wood pellet demand, conceptually based on Bockstaller et al. [39] and Svarstad et al. [38]. A rising demand of wood pellets may require to apply more fertilizer for shorter rotation cycles of forest plantations, e.g. *Pinus* spp., (Driving force). Consequently, increased fertilizer application may increase the nutrient runoff to surface water bodies (Pressure), which may lead to higher nutrient concentrations (State), i.e. possibly eutrophication, which may change e.g. the species composition (Response). Thus, an indicator of an environmental pressure such as the nutrient load from pine plantations on a water body would be rated as “three” on the five step scale, and a state indicator such as the nutrient concentration in a river would be rated as “four” or the nutrient application rate in the driver category as “one”. The closer the assessment is to the environmental impact, the more information on the environmental impact is expected to be considered.

Table 2.1: *(upper part)* Rating scale for the reliability of indicators, subcategory *Indicator type* adapted from Bockstaller et al. [34]; *(lower part)* rating scale for the feasibility of indicators, subcategory *Required resources (assessment interval)*

Indicator type (cause vs. effect-related)	
1 Driver	management practice
2 Driver	management practices related to state or impact
3 Pressure	release of pollutants or sediment
4 State	concentration of pollutant in environmental compartment
5 Impact	environmental changes attributable to pollutants or sediments

Required resources (assessment interval)
1 daily assessment/measurements required
2 seasonal assessment/measurements required
3 annual assessment/measurements required
4 less than annual measurements
5 no measurement, only completing a survey

The second subcategory for reliability is the *Validity of indicators*. We rate the validity, according to a rating scale, see Table 2.A1 in Appendix A, modified from Bockstaller et al. [34], which has been developed by Bockstaller and Girardin [40]. We rate the indicators (i) based on scientific literature, i.e. whether peer-reviewed articles use and confirm the exact indicator (value 4), whether the indicator is under debate in the scientific literature (value 3), or only confirm the calculation method of the indicator or even reject the indicator (value 2). (ii) If the indicator needs to agree with locally collected data (value 5), is typically gained from a validated model (value 4), a partly or only regionally validated model (value 2). If no validation is possible due to the rating in the subcategory *Indicator type* rated as given for indicators on management practices (value 1 or 2), we rate the indicator with a value of “three”. The third subcategory for reliability is the *Response time* since an immediate response or at least in the time frame of political decision making [10, 36] enable timely detection and counteraction to the expected or observed environmental problems. We rate the response time of indicators based on peer-reviewed publications.

The first subcategory for feasibility is the *Data requirement*, assessing the ease of data access [39, 34, 2, 36]. We rate indicators based on (i) the nature of the data, i.e. whether it can be obtained from authorities or other data sources (value 5), requires questioning the feedstock producer (value 4) or measurements are required (value 1 to 3). (ii) The measurement scale is additionally used for the rating [41], i.e. whether indicator data has to be measured at each field or farm individually (value 1) or whether one regional assessment is sufficient for the indicator (value 3). In addition, indicators may be attributed to the field/farm or the regional scale depending on the individual case (value 2), e.g. influenced by farm size (group certification) or an imprecise definition of the indicator in the certification scheme. The second subcategory for feasibility is the *Qualification requirement* [39, 34, 2] covering the ease or difficulty to assess an indicator due to its specificity or the required expert knowledge (requirements defined in Appendix A). High qualification requirements may be an obstacle for small scale producers, especially in developing countries [24]. The third subcategory for feasibility is the *Required resources (assessment interval)*, i.e. the frequency of possible measurements influences the effort and costs for certification. The fourth subcategory for feasibility is *Clearly defined thresholds*. We rate the existence of target values, reference conditions or thresholds because their availability influences the measurability [11]. A threshold or a possible source to derive it provided by the scheme, facilitates the interpretation of feedstock impacts regarding sustainability during the auditing process [41].

The relevance of an indicator first depends on its acceptance by stakeholders, i.e., whether the indicator is suitable to address a certain environmental impact category [36], and secondly on the degree to which stakeholders are involved in the selection process [26]. Data on the preferences of stakeholders is only available for criteria or for the even higher aggregation level of environmental impact categories, but is not available for the corresponding indicators (c.f. Buchholz et al. [35]). The lack of data might also be due to the fact that the development and choice of the rather technical indicators are related to the expertise of the practitioners or scientists. Therefore, the relevance of the indicators cannot be rated but will be checked indirectly by its fit to the relevant environmental impact categories.

We rate indicators that provide direct information about the occurrence or avoidance of environmental impacts. The indicators are aggregated by local/regional environmental impact category on a composite scale. In this context, a composite scale is the combination of several indicators into a thematic category, i.e. we compute the arithmetic mean of all indicators per certification scheme per environmental impact category and the indicator subcategories respectively. Similarly, the standard error of the mean (SEM) is calculated to assess the uncertainty of the arithmetic mean. We assess the indicator sets for the environmental impact categories soil quality, water quality, water availability or quantity, biodiversity and LU/LUC. Soil quality indicators cover indicators on both the management of soils and soil properties. Water quality and availability indicators assess both management activities with an impact on water bodies as well as state indicators of water bodies. Biodiversity indicators may assess the state of conservation areas, species composition or management activities for biodiversity. LU/LUC indicators give information on characteristics of a land use, e.g. carbon payback time, or assess whether no-go areas according to the EU RED definition have been converted for bioenergy feedstocks. The composite scale *Other* comprises indicators without a link to the listed environmental impact categories, which are related to the environmental stability of a system such as indicators on sustainable harvest levels. If applicable, indicators are attributed to two composite scales if a clear link to both is given, e.g. “no conversion of areas of high conservation value” to biodiversity and LU/LUC or “no removal of coarse woody debris” to soil quality and biodiversity.

Internal consistency is ensured by excluding indicators that do not directly measure environmental impacts, i.e. contextual knowledge is used according to Coste et al. [42]. Background knowledge on the environmental indicators, e.g. given by the certification scheme, allows to categorize the indicators. Internal consistency is required since the arithmetic mean should only be calculated for indicators that measure the same latent variable, i.e. environmental impact category. We exclude indicators, for example, if they assess whether legislation is covering environmental impacts, e.g. on water quality. In this case, certification schemes assume that environmental impacts are avoided (complying with existing regulations).

We list the indicators we included and excluded for each scheme in Table 2.2.

Table 2.2: Number of indicators analyzed for each scheme and each environmental impact category (=composite scale) and abundance of aspects in certification schemes excluded from evaluation to ensure internal consistency of composite scales; these results are based on CSBP [43], GBEP Task Force [14], GGL [44], GlobalGAP [45], ISCC [46], IWPB [47], Netherlands Standardization Institute [48, 49], REDcert [50], RSB [51], SAN [52] and forestry, [29, 53-56, 32]. For GGL, the *Agricultural source criteria* (GGL2) are assessed.

Composite scales		**GBEP**	**NTA8080**	**ISCC**	**REDcert**	**GGL**	**RSB**	**CSBP**	**IWPB**	**SAN**	**GlobalGAP**	**Forestry**
Total	*87*											
Soil quality	31	1	9	11	2	0	5	5	8	5	2	30
Water quality	17	2	4	6	7	4	3	6	3	4	7	6
Water availability	9	3	2	2	0	1	1	2	3	1	2	6
Biodiversity	18	3	5	1	1	1	7	3	1	10	2	13
LU/LUC	9	5	3	2	1	0	1	0	3	1	1	1
Others	3	2	0	0	0	0	0	0	1	0	0	3
Abundance of excluded aspects												
Off-site handling rules and machinery maintenance (e.g. disposal of plant protection product containers)	46	1	1	21	7	1	0	0	0	3	12	0
Demonstration of compliance with existing legislation or other rules such as certification schemes, manuals or rules (e.g. registration of product use)	32	0	3	5	3	0	4	3	6	5	3	0
Management plan or other unspecified action or goal required	33	0	0	2	1	2	19	4	0	2	3	0
Qualification and training of staff	10	0	0	2	1	0	0	1	3	1	2	0
Generic monitoring (e.g. soil quality has to be assessed)	6	0	0	0	0	2	2	0	0	2	0	0

2.3.3 The ecosystem service cascade for evaluation of certification schemes

Assessing certification schemes by only looking at indicators individually would disregard the schemes' quality and comprehensiveness concerning the use of environmental systems and the services/disservices derived thereof. A widely accepted concept to determine and quantify the human use of the environment is ESS [57, 58].

The ESS cascade [30] is a conceptual framework used to connect ESS to the underlying ecosystem structures and processes and to the human benefits derived from the use of the ecosystem. Ecosystem structures and processes are the basis to derive thresholds for the sustainable provision of an ESS [57, 30], i.e., the ecosystem capacity. For example, the ecosystem capacity can be used to answer questions about the critical limits or thresholds [59] for e.g. the extraction of tree biomass to sustain forest stocks. Because this evaluation focuses on local/regional environmental impacts, it is beyond our scope to depict the socioeconomic components of the ESS cascade, i.e., the human benefits and (monetary) values. We focus on biophysical and ecological structures and functions and their alteration due to the use of ESS. The ecological and the socioeconomic systems are linked by the use of ESS [60], e.g., biomass use. In practice, the ESS cascade has been used as a conceptual framework to embed indicators of different provisioning services, e.g., biomass production [61, 62], and regulating services, e.g., water purification [63], of the underlying environmental systems. In addition, the ESS cascade has also been used to visualize the interaction of indicators within and between the different components of the ESS cascade [64, 62]. Maes et al. [63] and Van Oudenhoven et al. [62] add land management to the beforehand mentioned components of the ESS cascade. The necessity of including land management was previously stated by Haines-Young and Potschin [30] but was not implemented. Like Ojima et al. [65], we included land management aspects because indicators of ESS describe the use of natural capital but do not provide insight into the extent that the use of ESS is altered by human land use activities, i.e. agricultural practices such as irrigation or fertilization or conservation measures such as field margins for biodiversity.

In this study, we use the term "human land use activity" because this term includes land management, land conversion and changes in the structure of the landscape [66]. Therefore, indicators of human land use activities enable the assessment of the intensity of land use associated with different types of and options for biomass provision. For example, changes in production practices or landscape planning are likely to affect ecosystems, i.e., the structures, processes and capacity. A better representation of the interaction of human land use activities, ecosystems and ESS use might help to identify environmentally especially harmful biomass use and land management practices. More reliable results could allow decision makers to better target, e.g., mitigation activities.

In this study, the ESS cascade is extended from a conceptual to an analytical framework for bioenergy feedstock production (Fig. 2.1). The ESS cascade is converted and expanded into an analytical tool to assess the quality of certification schemes. The latter are implemented within the framework to assess the sustainability of feedstock provision with environmental C&Is; i.e., the adverse environmental impacts should be revealed to facilitate mitigation or avoidance [13]. Thus, the extended ESS cascade is applied to investigate whether certification schemes represent biophysical processes for feedstock production in a qualitatively and quantitatively useful manner. We apply the widely used "Common International Classification of ESS – CICES" v4.3 [67], which has undergone several rounds of international review and consultation, to ensure assessing all major ESS, which may be affected by bioenergy feedstock production.

The mapping used for the certification scheme indicators is presented in Fig. 2.1. For the different certification schemes we analyzed, we focused especially on the representation of causal links and the coverage of ecosystem structures and functions represented in the extended ESS cascade, i.e., the quality of the representation of the environmental system. For example, does a certification scheme include indicators that would reveal if biomass use affected other ecosystem services such as surface or groundwater provision? Does a certification scheme include the link from fertilized pine plantations to a possible ground- or surface water pollution and does it provide the relevant indicators on, e.g. water quality and fertilization practices? We took the individual indicators per certification scheme, related them to the environmental system and indicated the causal links and components covered.

For an overview, we counted the actual number of indicators for each of the four components of the ESS cascade displayed in Fig. 2.1 and rated them on a three step scale based on thirds. For causal links, the certification schemes are compared with their peers. The certification scheme with the highest number of causal links has the best rating, i.e. 100 percent, and is used as a benchmark and rated as done for the indicators. The indicators and causal links for each scheme are displayed in Appendix A.

The following three types of common causal links and links without cause-effect relationships are found in the evaluated certification schemes and indicator sets:

a. **Positive causal link (<u>In</u>crease in X causes an <u>in</u>crease in Y):**
   Example: "The participating operator provides objective evidence demonstrating that her/his/its biomass/biofuels operation(s) does/do not contribute to exceeding the replenishment capacity of the water table(s) […]," RSB [51]. This statement implies that the maximal sustainable water use does not negatively affect the groundwater table and is adapted to the local level of precipitation. Therefore, both a higher precipitation and a higher change of the groundwater table, i.e. a lower decline, may result in a higher maximal sustainable water use.
b. **Negative causal link (<u>In</u>crease in X causes a <u>de</u>crease in Y):**
   Example: The feedstock provider measures the water use per area and uses irrigation techniques that conserve water most, e.g., CSBP [43]. In other words, if more irrigation techniques with low water use are applied (replacing inefficient technologies), the use of water units per unit bioenergy feedstock will decrease per ha.
c. **Varying causal link (<u>In</u>crease in X causes an <u>in</u>crease OR <u>de</u>crease in Y)**:
   Example: "Have systematic methods of prediction been used to calculate the water requirement of the crop?" GlobalGAP [45]. Options for actions are suggested in the explanation of the indicator. The actions may be operationalized as follows: The amount of water used varies with the crop type. Hydrologically, the upward flux of water via plants and soil is termed evapotranspiration. The choice of a crop may increase or decrease evapotranspiration. Because this biophysical flux is not named in the indicator, but is only implicitly considered, it is highlighted in yellow.
d. **No cause-effect relationship:**
   The soil organic carbon content is maintained or improved, e.g., GBEP Task Force [14].

The definition of the indicator specifies both the ecosystem capacity and the parameter to be measured to determine the ESS use, i.e., mediation of mass flows. Here, a thematic link between ecosystem capacity and ESS is given instead of a cause-effect relationship.

Additionally, we need to assess how certification schemes are able to overcome the challenge of the necessity of assessing (i) environmental impacts at scales beyond the field/farm level [12] and (ii) the interaction and accumulation of environmental impacts beyond different spatial scales [10, 37] and how to distribute target values or thresholds [74, 75]. Within this study, the relevant spatial scales from both the literature on actual indicators and from specific studies on scales to determine specific environmental parameters are shown in Fig. 2.1. Because this study focusses on local/regional scale environmental impacts, there are no indicators included beyond those scales. Local scale, also plot or field scale, is typically areas less than one km^2 and regional, also landscape or watershed scale ranges from 1 to 10,000 km^2 [57, 37]. There are some indicators that are more flexible and provide reasonable results at both of the considered scales. For example, the sustained yield and the underlying primary productivity can be scaled up or down for largely homogenous ecosystems, such as those in forestry, where sustainable harvest levels or wood resources and residues are common indicators [32].

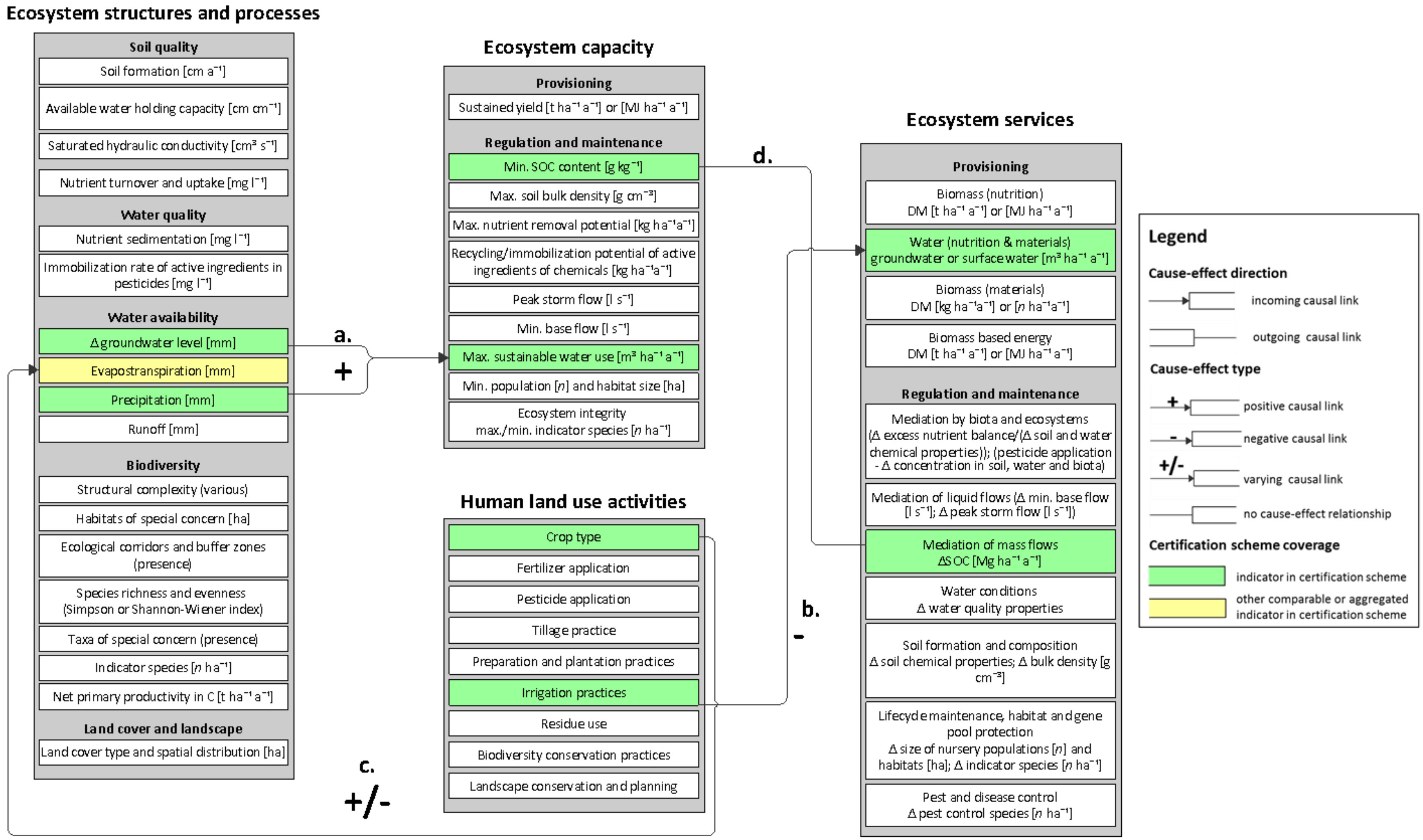

Fig. 2.1: ***(upper part)*** **ESS cascade (modified from CICES [67], Maes et al. [63], Potschin and Haines-Young [60], Van Oudenhoven et al. [62]) as an analytical framework to evaluate certification schemes for bioenergy feedstock production; the components shown are ecosystem structures and processes (underlying biophysical mechanisms), ecosystem capacity (sustainability thresholds for ESS use) and ESS (actual use of ESS or creation of disservices). The arrows indicate a. positive, b. negative, c. varying and d. no causal link. The selected indicators are adapted to the major impacts of bioenergy production identified from Dale and Beyeler [68], De Groot et al. [57], Haines-Young and Potschin [30], Kandziora et al. [64], Kienast et al. [69], Lattimore et al. [53], McBride et al. [11], McElhinny et al. [70], Schoenholtz et al. [55], Wascher [71].**

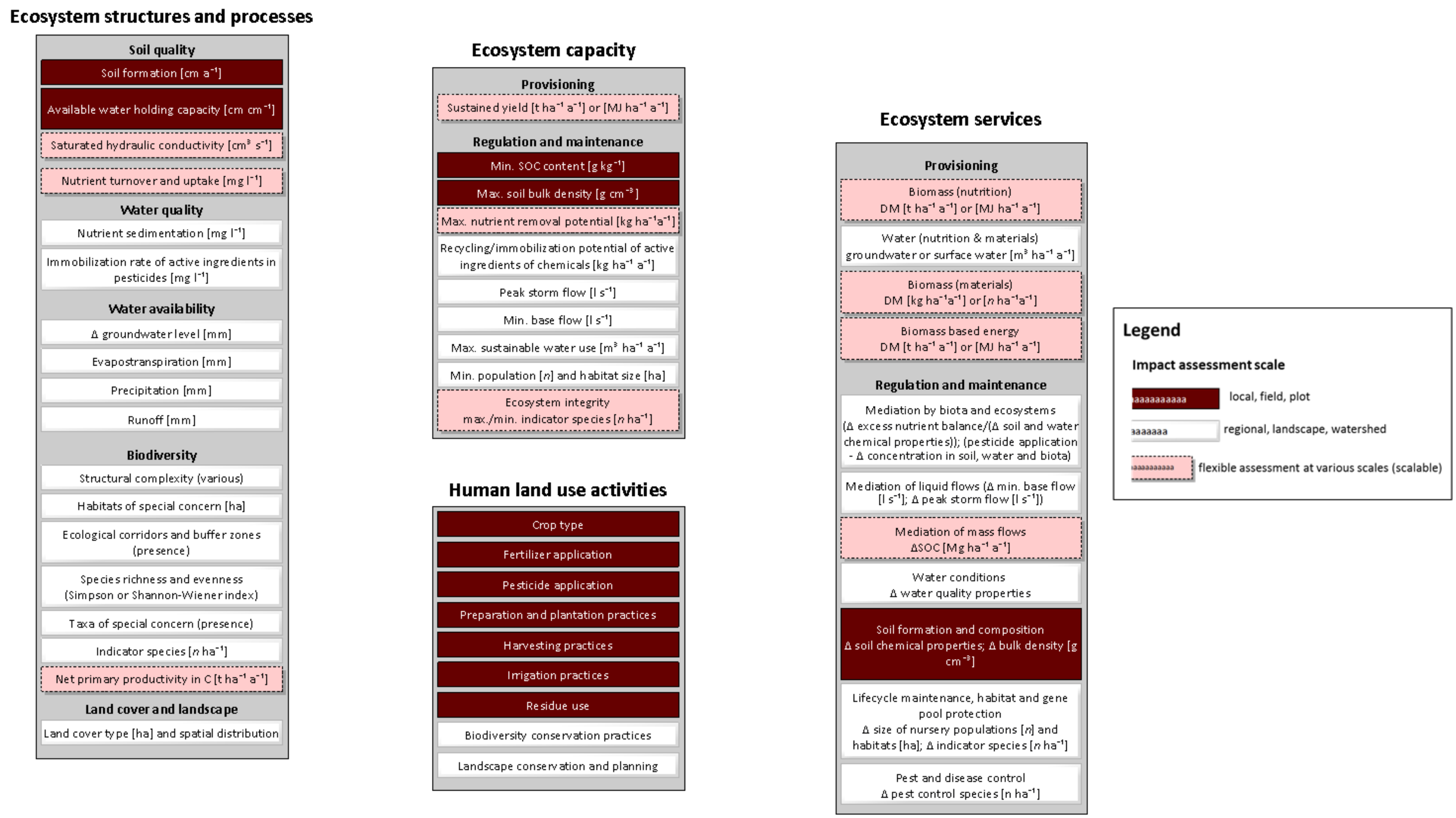

Fig. 2.1: ***(lower part)*** **Spatial impact assessment scales of the ESS cascade adapted for bioenergy feedstock production. The impact assessment scales are generally based on De Groot et al. [57] and Efroymson et al. [10] and are specifically based on Sposito [72] for hydrology and Turner et al. [73] for landscape patterns.**

2.4 Results and discussion

2.4.1 Major characteristics of certification schemes

The major characteristics evaluated in this study are those identified as relevant by existing reviews [76, 10, 13, 77], and the evaluated certification schemes and their indicators are introduced in the following sections. Table 2.3 shows that only GBEP, NTA 8080, GGL and CSBP target all types of bioenergy. CSBP intends to certify any type of bioenergy from ligno-cellulosic biomass. ISCC, REDcert and RSB originally were developed to demonstrate compliance with national or supra-national legislation, i.e., the EU RED, which primarily cover biofuels and bioliquids [13]. Currently, these schemes are being partially extended and revised to certify solid and gaseous bioenergy to ensure compliance with regulations in potential new versions of the EU RED. NTA 8080 is also used to demonstrate EU RED compliance for biofuels and bioliquids but is the implementation of the ‘‘Testing framework for sustainable biomass,’’ the so-called Cramer Criteria, which originally focused on any type and use of sustainable biofuels and other products from biomass [12]. The remaining certification schemes have been developed to ensure sustainable production of agricultural or timber products. To ensure cost-effectiveness, the EU might consider forest certification schemes to be a proof of sustainable production of solid biomass [31]. Table 2.3 shows that certification schemes for bioenergy attempt to assess the entire supply chain of a product to demonstrate, for example, the higher environmental sustainability than that of fossil energy carriers. The agricultural or forestry certification schemes are rather purpose specific; for example, the schemes demonstrate low-impact cultivation techniques or sustainable forest management [12] and thus focus on feedstock production rather than on the final product. In the latter aspect they differ from bioenergy certification schemes.

Table 2.3: Major characteristics of certification schemes based on BEFSCI [76], CSBP [43], EC [78], FSC [79], GBEP Task Force [14], GGL [44], GlobalGAP [45], ISCC [46], IWPB [47], Netherlands Standardization Institute [48], 49], REDcert [50], RSB [51], SAN [52], SFI [80].

	Bioenergy								*Agriculture*		*Forestry*	
	GBEP	**NTA8080**	**ISCC**	**REDcert**	**GGL****	**RSB**	**CSBP**	**IWPB**	**SAN**	**GlobalGAP**	**FSC**	**SFI**
Major characteristics:												
Applicable biofuel type												
solid	x	x	(x)	(x)	x	(x)	x	x			(x)	(x)
liquid	x	x	x	x	x	x	x					
gaseous	x	x	(x)	(x)	x	(x)	x					
Spatial scope for application	Global	Global	Global	EU*	Global	Global	US	Global	Global	Global	Regional	US/Canada
EU RED recognition		x	x	x		x						
Degree supply chain coverage***	FTPD	FTPD	FTPD	FTPD	FTPD	FTPD	FT	FTPD	F	F	F	F

*few third countries (e.g. Belarus, Ukraine)

** agricultural and forestry source criteria

***supply chain coverage: Feedstock (F), Transport (T), Processing (P), Distribution (D)

2.4.2 Indicator evaluation

2.4.2.1 Overview

For the requirements for indicators, the mean of the indicators for certification schemes in Fig. 2.2 shows that most of the certification schemes are rated at the center of the scale at this aggregation level. The mean for the *Required resources (assessment interval)* with an above-average rating and the mean for the *Indicator type* with a below-average rating for most of the schemes deviates from the general tendency toward a centered rating.

The pattern of the *Required resources (assessment interval)* and *Indicator type* may be interpreted as the common trade-off between the feasibility and the reliability of indicators (c.f. Payraudeau and van der Werf [37]).

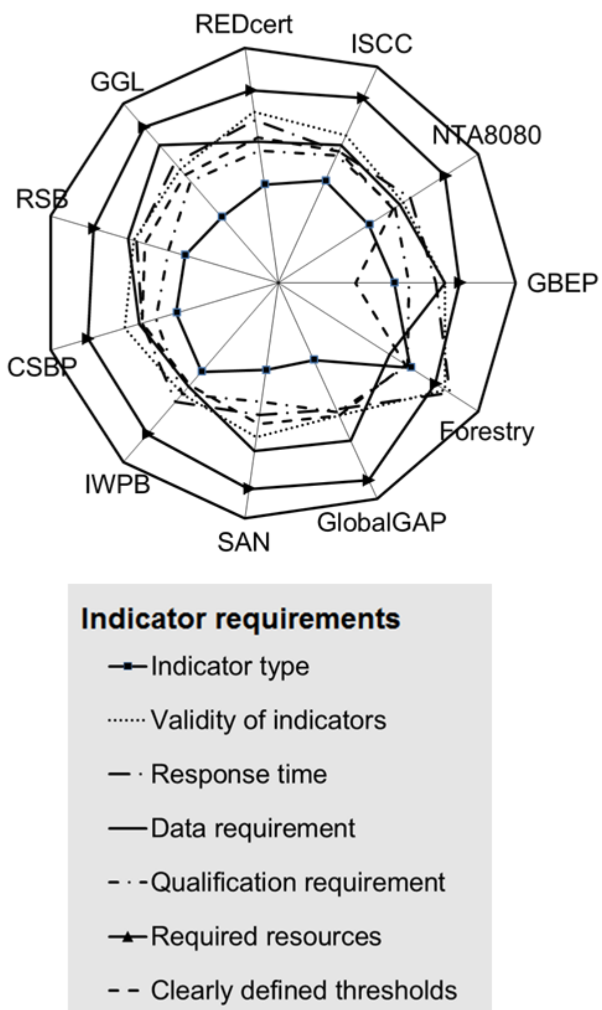

Fig. 2.2: Arithmetic mean of the ratings by subcategory of the indicator requirements for the evaluated certification schemes and indicator sets, CSBP [43], GBEP Task Force [14], GGL [44], GlobalGAP [45], ISCC [46], IWPB [47], Netherlands Standardization Institute [48], 49], REDcert [50], RSB [51], SAN [52] and forestry, [29, 53-56, 32]; a detailed explanation of the meaning of each step of the rating scales per indicator requirement is discussed in section 2.3.2, and the SEM can be found in Appendix A.

The thematic abundance of indicators not suitable for a direct environmental assessment and therefore excluded for internal consistency of the composite scales has been shown in section 2.3.2 in Table 2.2. Analyzing such excluded indicators gives insight into how certification schemes aim to demonstrate environmental sustainability without an environmental assessment. The majority of the aspects excluded are those not directly related to biomass cultivation or harvesting but are instead related to the handling of equipment and post-production waste or to the documentation of farming activities. The evaluated certification schemes build on cross compliance or are at least partly set up as a meta-standard. Indicators assess whether legislation or other certification schemes are fulfilled but do not assess whether the environmental impacts of bioenergy production are addressed. Indicators that require the establishment of management plans or actions to achieve a target, such as maintaining water quality, are equally abundant. In minor abundance is the qualification of staff members conducting different tasks in biomass cultivation and processing and generic monitoring activities, such as those related to soil quality.

This overview may provide the impression that the selection of most of the indicators is predominately driven by the aim to allow for highly feasible or practical and probably cost-effective assessment, e.g., leading to assessments that do not require (on-site) measurements, such as demonstrated compliance with local legislation or the review of existing documentation. The named indirect assessment approaches not only consume less time and fewer resources but also do not require an understanding of environmental processes or measurement techniques for an on-site assessment for either the certified party or for the auditor. Certification schemes that require the establishment of generic management plans or monitoring without any consideration of local environmental conditions and processes may facilitate a worldwide sustainability assessment.

2.4.2.2 Evaluation of indicators by requirements and by composite scales

The overview in section 2.4.2.1 revealed that a high aggregation level does not reveal significant differences between certification schemes. Therefore, the results for the ratings of certification scheme indicators are analyzed at the less aggregated level of composite scales and are grouped by the indicator requirements and their subcategories, see Fig. 2.3.

Based on reliability and conceptual soundness, the *Indicator type* has a nearly universal low rating (value 1-2); i.e., driver indicators on management practices are used, especially for water quality and water availability. Biodiversity and LU/LUC indicators are partially state or impact indicators (value 4-5). These indicators determine whether land use types are converted for biomass production for bioenergy. An example of such state indicators are spatial biodiversity indicators; e.g., there is no bioenergy feedstock production in areas of high conservation value (ecosystems, species) that demonstrate or intend to demonstrate compliance with EU RED (ISCC, REDcert, IWPB, GGL). For example, the certification schemes named above assess whether areas of high conservation value or of specific land use types with high carbon stocks, such as peatland, are converted for bioenergy feedstock production. Other EU RED compliance demonstrating schemes (NTA8080, RSB) without such a pattern have indicators other than spatial indicators that address the protection or restoration of ecological corridors or buffer zones. The *Validity of indicators*, with the exceptions of the composite scale for water availability and, more significantly, the composite scale *other* for the non-attributable indicators, could be largely characterized as being validated by the models or by agreement in the scientific literature (value 4). The *Response time*, see Fig. 2.3, of the chosen indicators is typically one to five years or is not measured, as for causal indicators (value 3), i.e. *Indicator type* (value 1 or 2). The latter option is more likely because Fig. 2.3 shows that most of the indicators are causal. Biodiversity and LU/LUC indicators partially show immediate responses (value 5). The rating pattern for Biodiversity and LU/LUC is comparable to the requirements for *Indicator type* and for the described indicators; see Fig. 2.3; i.e., the chosen impact indicators are associated with short response times.

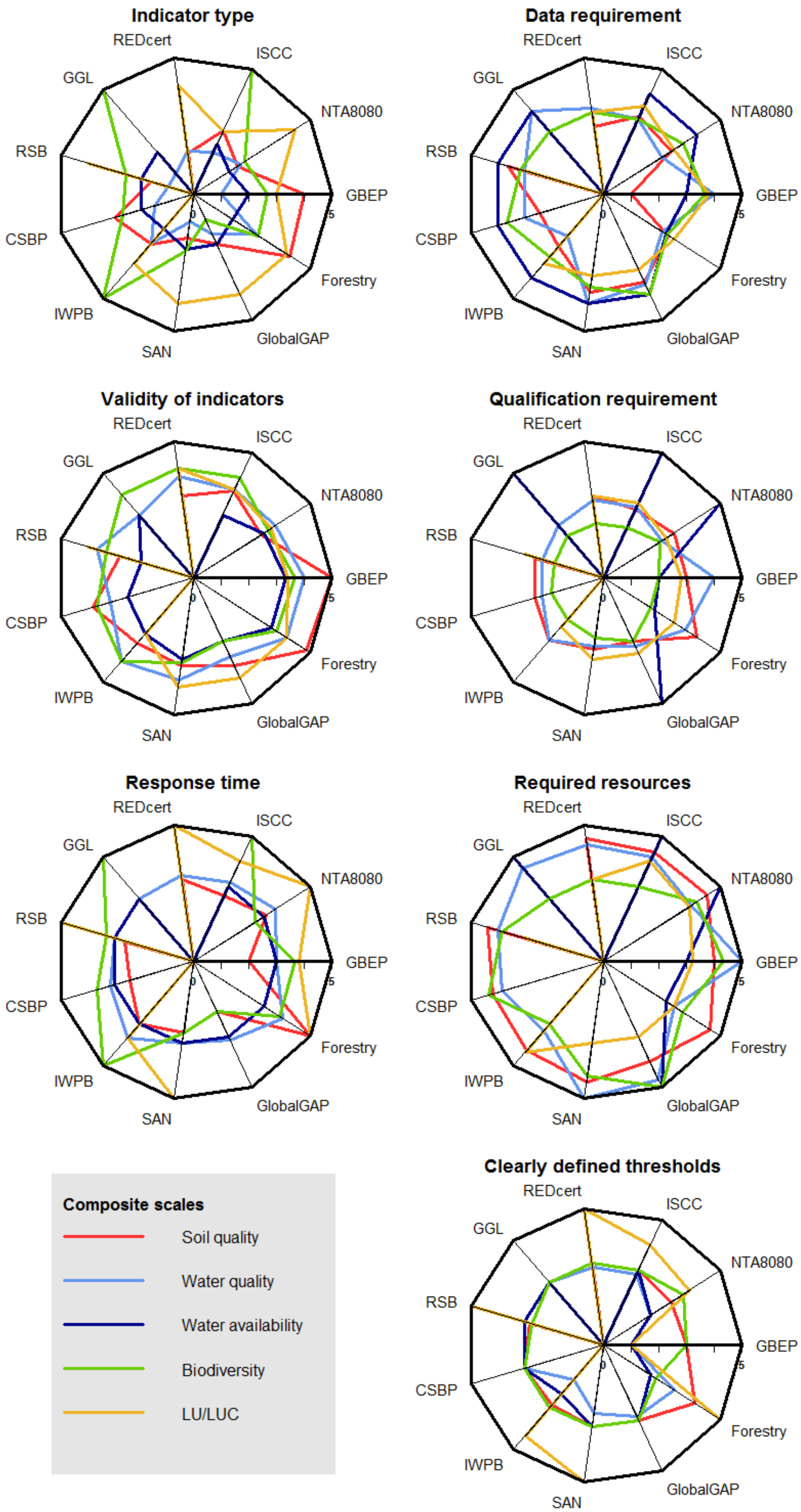

Fig. 2.3: Arithmetic mean for each indicator requirement subcategory disaggregated by composite scale and certification scheme/indicator set. Five is the best rating; zero indicates a lack of direct environmental assessment indicators for the composite scale and certification scheme. The SEM can be found in Appendix A.

Based on the results for feasibility, the *Data requirement* for the evaluated certification schemes shows that indicators for which data are available at other scales (value 3) or which require data from field observations and questionnaires but measurements (value 4) are not predominately used. The *Qualification requirement* greatly varies for the different composite scales. The biodiversity indicators are difficult to assess or require prior knowledge. At the least, general higher education, a university degree in agricultural science, or vocational training is required for the assessment (value 2-3). In contrast, the indicators chosen for water availability, e.g., water use per area, require no education or at least no more than a short introduction (value 4-5). The *Required resources (assessment interval),* soil quality, water quality and availability and other indicators are assessed predominately at intervals longer than one year (value 4) or do not even require field assessment (value 5). Biodiversity and LU/LUC impacts need to be assessed with a higher frequency; some must be assessed annually (value 3). The comparable patterns for *Data requirement* and *Required resources (assessment interval)* show that the data type and collection mode and the required resources seem to be correlated, i.e., the more effort that data collection for an indicator requires, the higher the frequency of assessment and vice versa. With respect to the requirement *Clearly defined thresholds*, certification schemes mostly only indicate (value 3) how to derive target values/thresholds or use causal indicators. Causal indicators do not require an actual threshold. Instead, the question is whether a (sustainable) management practices is applied or not, i.e., an assessment of compliance or non-compliance. LU/LUC indicators are an exception; for these indicators a threshold is typically given because their formulation implies that there must not be any land conversion for bioenergy feedstock production.

Trade-offs between feasibility (*Data requirement, Required resources (assessment interval)*) and reliability (*Indicator type, Response time*), mentioned in 2.4.2.1, are especially pronounced for the composite scale for water availability but are also pronounced for soil and water quality. For water availability, the requirements characterizing feasibility, *Data requirement* and *Required resources (assessment interval)*, are highly rated (value 4 or 5). The *Data requirement* can be met with field observations or questionnaires (value 4). The *Required resources (assessment interval)* are minimal because only surveys and no measurements need to be conducted (value 5). Because it is only necessary to complete a survey without measurements and this process requires even less assessment effort than the least frequent measurement, personnel resources and equipment can be saved relative to indicators that are regularly measured.

The indicator requirements for reliability are rated low. Driver indicators (management practices) that measure no response for the *Indicator type* (value 1 to 2) and *Response time* (value 3) are used. Such a trade-off is not pronounced for the *Validity of indicators* and their feasibility (*Data requirement, Required resources (assessment interval)*) because both are often highly rated (value 4). I.e., many driver indicators are either validated by models or are widely accepted in the scientific literature. The latter explanation applies to many of the indicators in this study. The comparable high ratings for the *Data requirement* and *Required resources (assessment interval)* reveal that certification schemes preferably use feasible indicators.

In Fig. 2.4, the results for the rating of certification scheme indicators are grouped by composite scale to reveal possible further patterns.

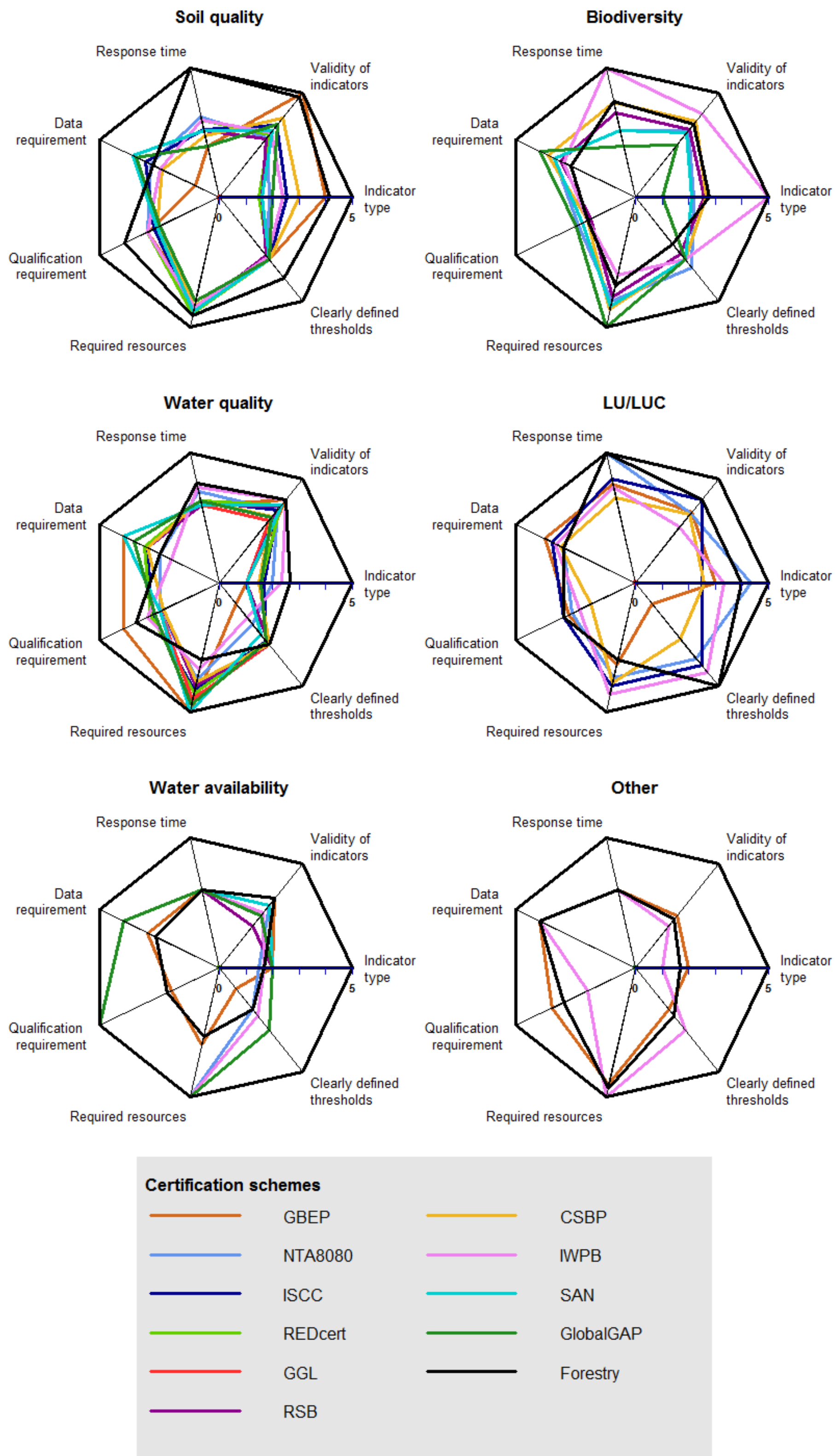

Fig. 2.4: Arithmetic mean for each composite scale disaggregated by indicator requirement subcategory and certification scheme/indicator set. Five is the best rating; zero indicates a lack of direct environmental assessment indicators for the indicator requirements and certification scheme. The SEM can be found in Appendix A.

Soil quality

With the exception of the *Data requirement*, soil quality indicators are especially high rated in the forestry indicator set. For the *Data requirement*, the forestry indicator set still performs as well as most of the other certification schemes. The higher rating of the forestry indicator set might reveal some potential for improvement in existing bioenergy certification schemes.

Water quality

With respect to water quality, most of the certification schemes perform equally well, with the exception of the *Data requirement*. Here, the low rating of the *Indicator type* is very apparent and reflects the dominant use of indicators that assess management practices and not the actual changes in the environmental compartment, i.e., water bodies.

Water availability

Water availability could be characterized as highly feasible (*Required resources (assessment interval)*, *Qualification requirement*, *Data requirement*) for most of the certification schemes, with the exception of the forestry schemes and GGL, which had low ratings for all of the requirements. This composite scale shows the differences in how well certification schemes chose indicators that optimized the trade-off between characteristics, e.g., reliability and feasibility. I.e., a comparable level of reliability and conceptual soundness (*Indicator type, Validity of indicators, Response time*) may be achieved with a high or low resource use (*Required resources (assessment interval)*, *Qualification requirement, Data requirement*).

Biodiversity and LU/LUC

Biodiversity is rated very homogenously by ISCC, REDcert, GGL and IWPB and LU/LUC by REDcert, RSB, SAN, GlobalGAP and forestry indicators. Both groups of certification schemes only use one environmental assessment indicator for biodiversity and for LU/LUC; respectively, this indicator is no production of bioenergy feedstocks in areas of high conservational value (ecosystems, species) and no conversion of land use types equivalent to those in the EU RED.

The rather high rating observed, especially for biodiversity, can be explained by the nature of the change because the coupling of biodiversity loss to land-use change facilitates the assessment for most of the requirements. Biodiversity gains higher indicator feasibility and reliability and conceptual soundness from land-use change indicators. Both the biodiversity and LU/LUC indicators also show the extent to which certification schemes exclusively fulfill and go beyond the underlying legislation. Here, the question is how detailed legislation should define environmental impacts that are to be avoided. Assuming that a large abundance of an indicator in the schemes is equal to the relevance, it can be said that the clear indicator definition by EU RED is suitable. This indicator is also used by other certification schemes than EU RED. However, this indicator is most likely not sufficient to comprehensively cover the major environmental impacts if only this legal minimum is assessed by certification schemes. Such clearly defined legislation might even hinder the competition among certification schemes to find an optimal solution for comprehensive detection of environmental impacts.

Other

The following composite scales are not completely assessed by the respective scheme. These certification schemes lack direct environmental assessment indicators for some of the composite scales: soil quality (GGL), water availability (REDcert) and LU/LUC (GGL, CSBP) (value 0). Indicators that do not belong to any composite, i.e., indicators grouped under *Other*, are largely missing. *Other* indicators only occur in the GBEP, IWPB and forestry schemes, as shown in Fig. 2.4, and contain only three indicators on sustainable harvest levels, which are predominately related to forestry. If indicators for the different composite scales are missing for a certification scheme, they are either neglected by the respective certification scheme or the scheme uses no direct environmental impact assessment indicators, as described in 2.4.2.1.

2.4.3 Comprehensiveness and quality of environmental indicator sets

The certification schemes and indicator sets for bioenergy production are mapped to the ESS cascade as described in section 2.3.3 and as displayed in Appendix A.

2.4.3.1 Comprehensiveness of indicators and causal links for system representation

The comprehensiveness of the system representation in these schemes is shown in Table 2.4.

Table 2.4: Comprehensiveness of system representation in certification schemes and indicator sets; better ratings mean that more indicators are covered for the different components of the ESS cascade (Fig. 2.1. in section 2.3.3), i.e., the representation of the function of the affected ecosystem and the used ESS. For causal links, the certification schemes are compared with their peers. The certification scheme with the highest number of causal links has the best rating and is used as a benchmark.

Certification schemes	**Indicators**				**Causal links**
	Ecosystem structures and processes	Ecosystem capacity	Ecosystem services	Human land use activities	
GBEP	-	-	+/-	+/-	-
NTA8080	-	-	+/-	+	+/-
ISCC	-	-	-	+	-
REDcert	-	-	-	+	+/-
GGLS2	+/-	-	+/-	+	+/-
RSB	+/-	+	+/-	+	+
CSBP	-	-	+/-	+/-	+/-
IWPB	-	-	+/-	+	+/-
SAN	+/-	-	+/-	+	+
GlobalGAP	-	-	-	+	-
Forestry	+/-	+/-	+/-	+	+/-

Coverage of indicators: >66.6%: +, 33.4-66.5%: +/-, <33.3 %: -

Human land use activities can be identified as the most comprehensively covered component of the ESS cascade for most of the schemes reviewed, except for GBEP and ISCC.

This pattern might be explained by the greater feasibility of assessment rather than the relevance of the biophysical processes; see the less comprehensive coverage of ecosystem structures and processes and ESS and the necessity that certification schemes demonstrate sustainability at a local scale instead of the required assessment at a regional scale for other indicators, and see Fig. 2.1 in section 2.3.3. In contrast, the disproportionately small number of indicators to be assessed at a regional scale renders it very likely that certification schemes miss cumulative effects. Cumulative effects are only harmful if a farming practice is applied throughout a region. For example, a crop and the respective fertilizer and pesticide application might only cause significant impacts on water quality if repeatedly applied within a catchment. This problem is addressed by NTA8080 and IWPB, which both include indicators for off-site impact, such as the Biological Oxygen Demand. GBEP has a large share of indicators that are beyond the local scale, but this share can very likely be attributed to its difference in purpose. GBEP indicators have been developed for national assessments [14] rather than for certifying single producers.

Ecosystem capacity is considered in most of the certification schemes; however, in RSB ecosystem capacity is not explicitly considered (yellow color) or is not considered (white color), as shown in Appendix A.

An explanation for the lack of thresholds or target values might be the flexibility required to consider the applicability globally and for multiple feedstocks. The indicators need to be equally applicable to different feedstocks that are grown under various environmental conditions and alongside various ecosystems associated with a large variability in ecosystem capacity. Here, clear target values are neither feasible nor practical. However, a methodology for the derivation of the ecosystem capacity can be given. A positive example is the RSB; see Fig. 2.5. Usually, a threshold is set for the SOC content for several certification schemes. However, the SOC content is only expected to reveal significant changes from changes in management practices, e.g., tillage regime, after a long time lag of at least five to ten years [81]. Because the reviewed certification schemes do not consider such a time lag in their certificate, such a threshold for SOC will be unlikely to have an impact on the certification decision. Only severe changes of the SOC content over the respective time frame might have an impact.

2.4.3.2 Quality of indicators and causal links for system representation: exemplary cases

The quality of the system representation is analyzed in the examples in Fig. 2.5; i.e., how certification schemes translate the human-environment interactions and the biophysical cause-effect relationships. As mapped in Fig. 2.5, the water availability indicators from GGL show that the central aspect of the certification schemes is often driver indicators for management practices, and these indicators should partly consider biophysical processes (2.). These biophysical processes are usually not specified. As an example, indicators are defined as follows: “Data about: climate, water […] are collected on a regular basis.” [44]. In addition, it is required that practices are applied to enhance the use of scarce water resources: “4.1 Efficiency and productivity of agricultural water use for better utilization of limited water resources has to increase” [44]. Neither the practices (3.) nor the ecosystem capacity of a scarce water resource (4.) are defined. Missing indicators and open formulations for indicators often result in imprecisely formulated causal links (5.). In contrast to the previous examples, for GBEP, shown in Appendix A, clearly defined indicators, which result in equally clear causal links, can be found.

A higher accuracy of the defined causal links facilitates environmental performance measurements and the determination of options for improvement. Predictions for the alteration of one parameter allow the direction of the change in another indicator to be determined qualitatively or even quantitatively. For example, excluding land cover types such as peatlands from feedstock production reduces the sustainable yield of a region by the theoretical biomass yield of peatland. As shown in Fig. 2.5, compared with RSB, a deficiency of both GBEP and GGL is the incomprehensive coverage of most of the components of the ESS cascade.

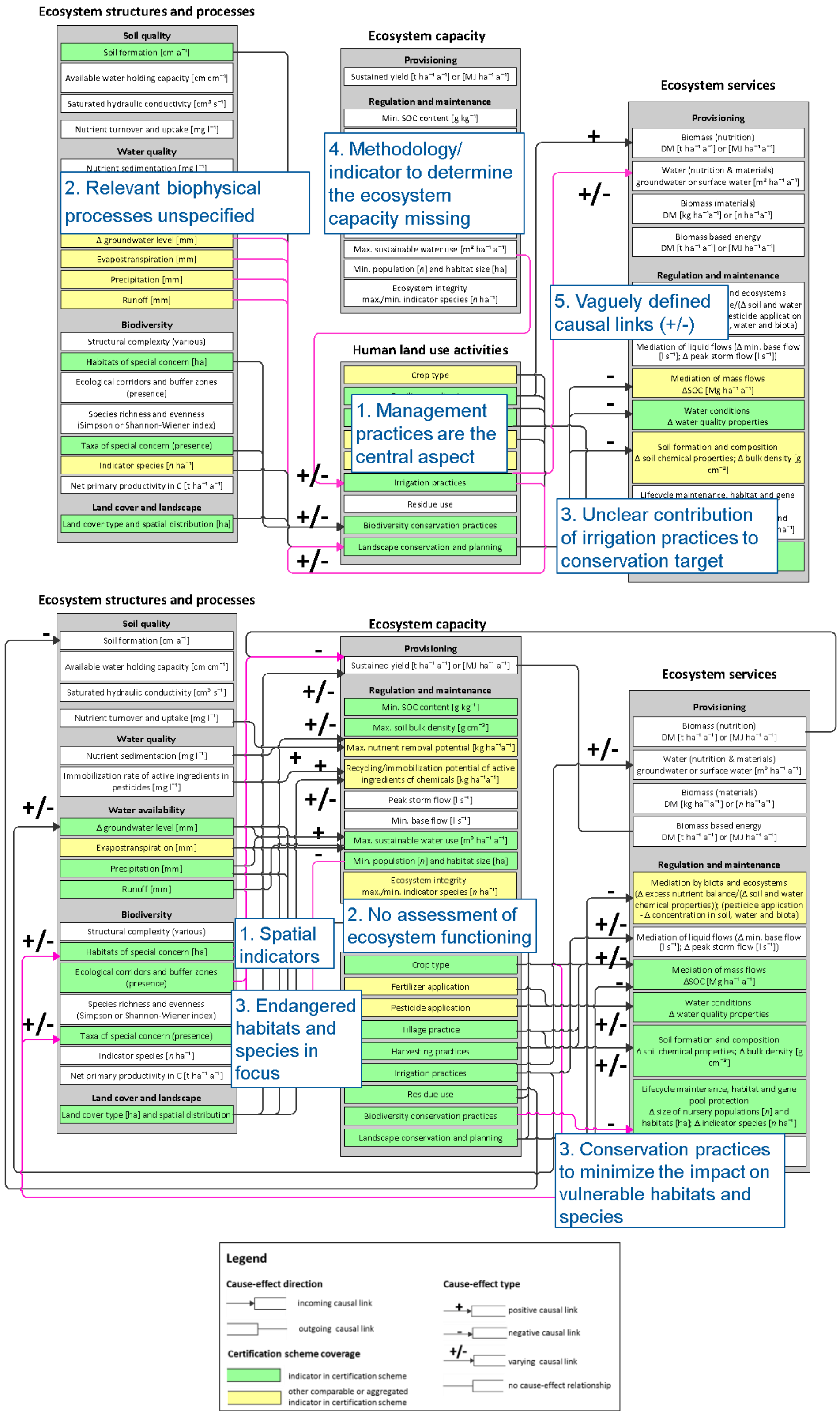

Fig. 2.5: (*upper part*) Water availability indicators from GGL mapped onto the ESS cascade. *(lower part)* Biodiversity indicators from RSB mapped onto the ESS cascade; common characteristics and deficiencies are indicated in the numbered boxes.

In contrast to GGL, RSB more comprehensively covers the ESS cascade. Despite the greater comprehensiveness, qualitative deficiencies can be shown for examples of the biodiversity indicators from RSB. Preferably, the indicators used are spatial indicators of biodiversity (1.) and not indicators that directly demonstrate ecosystem functioning, such as species richness and evenness indices, e.g., Shannon index, or the abundance of indicator species (2.). The typically chosen spatial indicators and indicators on conservation practices focus on endangered or protected species and habitats (3.).

Possible explanations for the prevailing indicator choice might be:

a. The requirements of the underlying legislations, i.e., the EU RED, govern the indicator choice.

b. Because of their higher risk of extinction, highly vulnerable species and habitats have greater importance for the public or for nature enthusiasts [82].

c. The availability of data for endangered species and habitats is widely available for many parts of the world. Data on species and habitats of less concern is not collected as extensively [68]. Therefore, data availability seems to be better for indicators on endangered species.

d. Indicators on ecosystem function must be adapted to the local context, i.e., indicator species, other indicanda of ecosystem functioning and species richness greatly vary by both location and ecosystem.

The most common case in which causal links in certification schemes are defined is when management practices are to be applied to minimize the use of ESS and the creation of disservices is respectively compared with an uncertified alternative in feedstock production. This case is revealed for RSB (3.) and GGL in Fig. 2.5.

Such an approach neglects the underlying ecosystem structures and processes in the indicator definition. Certification schemes assume a shortened causal link from human land use activities to the ESS and ignore the often directly affected ecosystem structures and processes. Currently, certification schemes are unlikely to allow the measurement and comparison of the environmental performance of bioenergy feedstocks. First, certification schemes, as shown for the example in Fig. 2.5, partially do not cover the obviously affected ESS. For example, biomass (use) is neglected as an indicator although this indicator could easily be determined. Missing indicators are not only those indicators obtained with more effort or technical skills, such as the impact on the minimum and peak flow of surface waters. Secondly, a large proportion of causal links that are represented by the reviewed certification schemes map the interactions between but not within the different components of the ESS cascade. Therefore, it is not possible to determine trade-offs and synergetic interactions between different ecosystem services. Thirdly, feedbacks from the use of ESS on ecosystem structures, processes and capacities are mostly not determined, as shown in the mapped certification schemes. Such less comprehensive coverage of the ESS and the causal links renders it impossible to compare the uses and consequently, the environmental impacts of different feedstocks. This deficiency might be because of the nature of the certification schemes to demonstrate compliance with legislation, such as the EU RED or other non-prescriptive rules. The schemes were not originally developed to assess the environmental performances of different feedstocks. Despite this focus, other ESS affected by biomass use could be theoretically used as a multidimensional unit for normalization to allow comparisons of different pathways for biomass provision; this unit would be comparable to the functional unit, e.g., the biomass, in LCAs for energy use or GHG emissions.

2.4.4 Limitations of this approach

One may argue that there is an assessor bias inherent to both the development and application of the rating scales for the indicator and scheme evaluation. Nevertheless, several measures to reduce and reveal such an assessor bias have been taken:

a. The use of empirically applied and peer-reviewed rating scales for agri-environmental indicator systems;
b. The determination of missing rating scales from the range of weak to strong implementation options for bioenergy certification schemes and existing reviews;
c. Ensuring the transparency of the rating by providing detailed descriptions of each rating scale.

Using the mean to aggregate indicators by composite scale, it was necessary to account for the uncertainty of the mean by the SEM, as shown in Appendix A. There are only a few cases in which the arithmetic mean does not well represent the composite scale. Therefore, the enhanced clarity of the composite scales for each indicator individually should be valued higher. There may be more accurate clustering options than the arithmetic mean, but those options would require complete data sets. Because they do not include indicators for all composite scales, several certification schemes, namely REDcert, GGL, and CSBP, would have had to be excluded. The same problem applies to tests for the internal consistency of the composite scales, such as Cronbach's alpha test, which could not be used because the data sets were incomplete. Because only three of 87 indicators could not be grouped to the chosen composite scales, as given by the environmental impact categories, the expert-based approach seems to be sufficient.

Empirically, the ESS cascade has been used to assess the impact of human appropriation for purely scientific purposes in a number of cases already, e.g., the studies by Kandziora et al. [64], Maes et al. [63], Petz and van Oudenhoven [61]. Such science-focused studies partially may not reflect practical needs. For example, indicators at the catchment scale are not necessarily suitable to certify individual farmers although these indicators are scientifically more appropriate. In addition, the scope of this study on local/regional environmental impacts required the exclusion of global environmental impacts (e.g., air quality). Therefore, a smaller number of interactions with the related ESS, e.g., the atmospheric composition and climate regulation, are missing. Nevertheless, it is unlikely that a few additional ESS would significantly change the relatively clear patterns shown for the included ESS.

2.4.5 Results in the context of existing and possible future research

This section sets the findings of this study in relation to existing research and outlines future research needs.

2.4.5.1 Usefulness of precise and harmonized legislation on environmental impacts as baseline for certification schemes

Biodiversity and LU/LUC, as composite scales, demonstrate that there is a convergence of certification schemes. The results by Van Dam et al. [13] noting the abundance of spatial biodiversity indicators for endangered habitats and species can be confirmed. The actual change in biodiversity is typically not assessed in the evaluated certification schemes, but stated by as hardly possible by current schemes and requiring beyond farm scale assessments [12]. For biodiversity, the hypothesis that precise definitions of the underlying legislation such as the EU RED might hinder the use of more reliable impact indicators seems relevant. In particular, other composite scales with less precise definitions, e.g., the Water Framework Directive in the EU, or with no underlying legislation, such as the scale for water quality, show a larger variety of indicators. Such convergence caused by precisely defined legislation indicates that exclusive peer comparison in existing review papers (e.g., Van Dam et al. [26]) does not completely reveal the limitations and potential improvements.

An additional research-based indicator set, such as the analytical framework developed in this study, revealed further limitations and potential improvements. Based on this analytical framework, limitations in the qualitative and quantitative representations of environmental impacts and the use of ESS in certification schemes could be shown. Some certification schemes are good examples for selected aspects of the assessment of environmental sustainability. Improvements may be achieved by combining the comprehensiveness of RSB with the quality of GBEP, for example. The focus on human land use activity indicators and the largely incomplete assessment of other key functional relationships show that the selection of indicators for certification schemes is driven by feasibility rather than by relevance or reliability. With respect to feasibility, Scarlat and Dallemand [12] recommend striving for a further harmonization of certification schemes through a meta-standard approach or through internationally harmonized minimum sustainability requirements. Their approach might contribute to reduced certification costs, increased feasibility or increased international acceptance of bioenergy certification schemes; these effects are comparable to the developments in forestry certification schemes (e.g., FSC and PEFC). However, enhanced reliability and conceptual soundness of certification schemes requires empirical tests or comparisons with a research-based indicator set. The converging biodiversity and LU/LUC indicators have shown some limitations of peer comparison for certification schemes and missing improvement options from academia.

2.4.5.2 Trade-off between a reliable sustainability assessment and securing feasible compliance with legislation

The focus on feasibility has been apparent in the indicator evaluation in section 2.4.2. Existing studies (e.g., Van Dam et al. [13] or Lewandowski and Faaij [2]) identifying the predominant use of feasible causal indicators can be confirmed. Additionally, recent versions of certification schemes, such as the draft from IWPB issued after the findings of former studies, have not been improved in this respect. In addition, the necessity of linking different spatial assessment scales in a proper consideration of environmental impacts has been identified by Van Dam et al. [13]. Nevertheless, this requirement is still only rarely overcome, e.g., by GBEP. With respect to feasibility, *Data requirement* and *Required resources* could be observed to be drivers for indicator selection. Similarly, the weak inclusion of ecosystem capacities, i.e., thresholds or target values, or the use of causal indicators without thresholds is deficient with respect to both feasibility and conceptual soundness.

2.4.5.3 Options to improve current certification schemes

The interactions (causal links) between and within the different components of the environmental systems mapped to the ESS cascade often seem to be incomplete and/or only weakly specified; this incompleteness makes quantification of the interactions difficult or even impossible. This limitation could be improved after specification of the causal links. Incomplete indicator sets do not favor the reliable (environmental) performance measurement of feedstocks. Bioenergy certification schemes have been developed to demonstrate compliance rather than to measure and compare the environmental performances of different feedstocks, confirming Diaz-Chavez [29]. In addition, only the compliance or non-compliance with the certification scheme is of interest not the variable degrees of under-/over-compliance of different feedstocks and producers under different environmental conditions. Mostly likely, future certification schemes could consider different degrees of compliance, e.g., different threshold levels, since too high requirements for producers with low financial means may hinder them to participate [2]. Implementation options could be an extension to the current differentiation of mandatory and facultative requirements used in several certification schemes, e.g., NTA8080. This approach might (i.) raise the information content of certification schemes by visualizing different degrees of environmental performance. (ii.) This approach also facilitates access for small shareholders in developing countries if they initially only need to comply with less strict thresholds. (iii.) This approach could also be used as a strong marketing tool.

2.5 Conclusions

In this study, we evaluated existing indicator sets and certification schemes to assess the environmental sustainability of different feedstocks for bioenergy. No outstanding certification scheme could be identified. Nevertheless, certain available schemes are better than others for assessing the selected environmental impact categories. To date, the proliferation of schemes, which was noted by several authors [12, 26, 13], has not led to significant changes in the use of reliable and conceptually sound indicators. Instead, schemes strive for feasibility in the indicator choice by complying with existing legislation or consumer expectations. For legislators, potential conclusions could be (i) to require certification schemes and academia to develop more reliable, but still feasible and cost-effective indicator sets, which at least cover the major underlying ecosystem structures and processes, and/or (ii) to consider a methodology to assess the capacity of an ecosystem, i.e., a methodology to determine threshold values for sustainable production. As a second step, certification schemes could assess well-defined causal links and feedbacks for biomass production; for example, schemes could use the adapted versions of the ESS cascade as an analytical framework. The suggested improvements would contribute to increased reliability in the identification of the environmental impacts of bioenergy feedstocks. As an additional benefit, the improved representation of ecosystem functions and feedback mechanisms will facilitate assessments of the interaction between different ESS, such as biomass use, water use or regulating ESS. In further empirical studies, it will be especially interesting to find out, under which conditions cause-related indicators reliably identify sustainable production and for which cases such indicators do not reveal sustainability deficiencies. Beyond the environmental impacts targeted in this study, further social or economic impacts must be considered in bioenergy certification to enable a more comprehensive comparison of alternative feedstocks.

2.6 Acknowledgments

This work was partly funded by the by EU FP7 project SECTOR (Production of Solid Sustainable Energy Carriers by Means of Torrefaction). The authors thank Ameur Manceur for advice on statistical methods and five anonymous reviewers for their comprehensive and helpful comments.

2.7 References

[1] Demirbas A. Political, economic and environmental impacts of biofuels: A review. Applied Energy 2009;86(Suppl 1):S108-S17.

[2] Lewandowski I, Faaij APC. Steps towards the development of a certification system for sustainable bio-energy trade. Biomass and Bioenergy 2006;30(2):83-104.

[3] Ragauskas AJ, Williams CK, Davison BH, Britovsek G, Cairney J, Eckert CA, et al. The path forward for biofuels and biomaterials. Science 2006;311(5760):484-9.

[4] WG III. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, et al., editors. Cambridge and New York: Cambridge University Press; 2012.

[5] World Commission on Environment and Development. Our common future. Oxford: Oxford University Press; 1987.

[6] UN. Johannesburg declaration of sustainable development. World Summit on Sustainable Development; 2002 August 26 - September 4; Johannesburg.

[7] Cohen B, Winn MI. Market imperfections, opportunity and sustainable entrepreneurship. Journal of Business Venturing 2007;22(1):29-49.

[8] Hill J, Nelson E, Tilman D, Polasky S, Tiffany D. Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. Proceedings of the National Academy of Sciences 2006;103(30):11206-10.

[9] Robbins M. Policy: Fuelling politics. Nature 2011;474(Suppl 7352):S22-S4.

[10] Efroymson RA, Dale VH, Kline KL, McBride AC, Bielicki JM, Smith RL, et al. Environmental indicators of biofuel sustainability: What about context? Environmental Management 2013;51(2):291-306.

[11] McBride AC, Dale VH, Baskaran LM, Downing ME, Eaton LM, Efroymson RA, et al. Indicators to support environmental sustainability of bioenergy systems. Ecological Indicators 2011;11(5):1277–89.

[12] Scarlat N, Dallemand JF. Recent developments of biofuels/bioenergy sustainability certification: A global overview. Energy Policy 2011;39(3):1630–46.

[13] Van Dam J, Junginger M, Faaij APC. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renewable and Sustainable Energy Reviews2010;14(9):2445-72.

[14] GBEP Task Force. The Global Bioenergy Partnership sustainability indicators for bioenergy. Rome: GBEP Secretariat - FAO; 2011.

[15] Von Blottnitz H, Curran MA. A review of assessments conducted on bio-ethanol as a transportation fuel from a net energy, greenhouse gas, and environmental life cycle perspective. Journal of Cleaner Production 2007;15(7):607-19.

[16] Haes HAU, Heijungs R, Suh S, Huppes G. Three strategies to overcome the limitations of Life-Cycle Assessment. Journal of Industrial Ecology 2004;8(3):19-32.

[17] Koellner T, Geyer R. Global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 2013;18(6):1185-7.

[18] Koellner T, Baan L, Beck T, Brandão M, Civit B, Margni M, et al. UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 2013;18(6):1188-202.

[19] Milà i Canals L, Rigarlsford G, Sim S. Land use impact assessment of margarine. The International Journal of Life Cycle Assessment 2013;18(6):1265-77.

[20] Directive 2013/draft/EU Directive of the European parliament and of the council on sustainability criteria for solid and gaseous biomass in electricity and/or heating and cooling and biomethane injected in the natural ags network. 2013.

[21] Dale VH, Lowrance R, Mulholland P, Robertson GP. Bioenergy sustainability at the regional scale. Ecology and Society 2010;15(4):23.

[22] Rametsteiner E, Simula M. Forest certification—an instrument to promote sustainable forest management? Journal of Environmental Management 2003;67(1):87-98.
[23] Directive 2009/28/EC. On the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.2009.04.23. Off J EU 2009. L 140: 16-62.
[24] Kaphengst T, Ma MS, Schlegel S. At a tipping point? How the debate on biofuel standards sparks innovative ideas for the general future of standardisation and certification schemes. Journal of Cleaner Production 2009;17(Suppl 1):S99-S101.
[25] Junginger M, van Dam J, Zarrilli S, Ali Mohamed F, Marchal D, Faaij A. Opportunities and barriers for international bioenergy trade. Energy Policy 2011;39(4):2028-42.
[26] Van Dam J, Junginger M, Faaij A, Jürgens I, Best G, Fritsche U. Overview of recent developments in sustainable biomass certification. Biomass and Bioenergy 2008;32(8):749-80.
[27] Searchinger T, Heimlich R, Houghton RA, Dong F, Elobeid A, Fabiosa J, et al. Use of US croplands for biofuels increases greenhouse gases through emissions from land-use change. Science 2008;319(5867):1238-40.
[28] Sikkema R, Steiner M, Junginger M, Hiegl W, Hansen MT, Faaij A. The European wood pellet markets: current status and prospects for 2020. Biofuels, Bioproducts and Biorefining 2011;5(3):250-78.
[29] Diaz-Chavez RA. Assessing biofuels: Aiming for sustainable development or complying with the market? Energy Policy 2011;39(10):5763–9.
[30] Haines-Young R, Potschin M. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli D, Frid C, editors. Ecosystem Ecology: a new synthesis. BES Ecological Reviews Series. C. Cambridge: Cambridge University Press; 2010.
[31] Volpi G. EU policy on bioenergy sustainability [Presentation on the Internet from 5th of December 2012]. Brussels: European Commission [cited 2013 January 10]; Available from: http://ec.europa.eu/agriculture/consultations/advisory-groups/forestry-cork/
[32] Stupak I, Lattimore B, Titus BD, Tattersall Smith C. Criteria and indicators for sustainable forest fuel production and harvesting: A review of current standards for sustainable forest management. Biomass and Bioenergy 2011;35(8):3287-308.
[33] Cocchi M, Nikolaisen L, Junginger M, Sheng Goh C, Heinimö J, Bradley D, et al. Global Wood Pellet Industry - Market and Trade Study. Task 40: Sustainable international bioenergy trade: IEA Bioenergy; 2011.
[34] Bockstaller C, Guichard L, Keichinger O, Girardin P, Galan M-B, Gaillard G. Comparison of methods to assess the sustainability of agricultural systems. A review. Agronomy for Sustainable Development 2009;29(1):223-35.
[35] Buchholz T, Luzadis VA, Volk TA. Sustainability criteria for bioenergy systems: results from an expert survey. Journal of Cleaner Production 2009;17(Suppl 1):S86-S98.
[36] Niemeijer D, de Groot RS. A conceptual framework for selecting environmental indicator sets. Ecological Indicators 2008;8(1):14-25.
[37] Payraudeau S, van der Werf HMG. Environmental impact assessment for a farming region: a review of methods. Agriculture, Ecosystems & Environment 2005;107(1):1-19.
[38] Svarstad H, Petersen LK, Rothman D, Siepel H, Wätzold F. Discursive biases of the environmental research framework DPSIR. Land Use Policy 2008;25(1):116-25.
[39] Bockstaller C, Gaillard G, Baumgartner D, Freiermuth Knuchel R, Reinsch M, Brauner R, et al. Betriebliches Umweltmanagement in der Landwirtschaft: Vergleich der Methoden INDIGO, KUL/USL, REPRO und SALCA Colmar: Grenzüberschreitendes Institut zur rentablen umweltgerechten Landbewirtschaftung; 2006.
[40] Bockstaller C, Girardin P. How to validate environmental indicators. Agricultural Systems 2003;76(2):639-53.

[41] Binder CR, Feola G, Steinberger JK. Considering the normative, systemic and procedural dimensions in indicator-based sustainability assessments in agriculture. Environmental Impact Assessment Review 2010;30(2):71-81.
[42] Coste J, Guillemin F, Pouchot J, Fermanian J. Methodological approaches to shortening composite measurement scales. Journal of Clinical Epidemiology 1997;50(3):247-52.
[43] CSBP. Standard for sustainable production of agricultural biomass. [Monograph on the Internet updated 2012]. Council on Sustainable Biomass Production [cited 2012 December 21]; [about 1 screen]. Available from: http://www.csbp.org/
[44] GGL. Certification Requirements Green Gold Label. [Monograph on the Internet updated 2011]. Zwolle: Green Gold Label Foundation [cited 2012 December 6]; [about 1 screen]. Available from: http://www.greengoldcertified.org
[45] GlobalGAP. Integrated Farm Assurance: All farm base/crops base/combinable crops. [Monographs on the Internet updated 2012]. Cologne: FoodPLUS GmbH [cited 2012 December 8]; [about 4 screens]. Available from: http://www.globalgap.org/uk_en/
[46] ISCC. ISCC EU System Documents. [Monographs on the Internet updated 2011]. Cologne: ISCC System GmbH. [cited 2012 December 4]; [about 5 screens]. Available from: http://www.iscc-system.org/en/
[47] IWPB. IWPB Sustainability principles/IWPB Sustainability criteria and indicators. [Monographs on the Internet updated 2012]. Linkebeek: Laborelec. [cited 2012 December 21]; [about 3 screens]. Available from: www.laborelec.be/ENG/initiative-wood-pellet-buyers-iwpb/
[48] NTA 8080:2009 NTA 8080 - Sustainability criteria for biomass for energy purposes. 2009. 1-41.
[49] NTA 8081:2012-04 NTA 8081 - Certification scheme for sustainably produced biomass for energy purposes. 2012. 1-27.
[50] REDcert. REDcert-EU system documents. [Monographs on the Internet updated 2012]. Bonn: REDcert GmbH [cited 2012 December 12]; [about 3 screens]. Available from: http://www.redcert.org
[51] RSB. RSB Sustainability Standards – EU RED. [Monographs on the Internet updated 2011]. Chatelaine: Roundtable on Sustainable Biomaterials [cited 2012 December 8]; [about 5 screens]. Available from: http://rsb.org/sustainability/rsb-sustainability-standards/
[52] SAN. Farm Standards/Group Standards. [Monographs on the Internet updated 2011]. San José: Sustainable Agriculture Network [cited 2012 December 6]; [About 2 screens]. Available from: http://sanstandards.org/sitio/
[53] Lattimore B, Smith C, Titus B, Stupak I, Egnell G. Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. Biomass and Bioenergy 2009;33(10):1321-42.
[54] Newton AC, Kapos V. Biodiversity indicators in national forest inventories. Unasylva 2002;53(210):56-64.
[55] Schoenholtz SH, Miegroet HV, Burger J. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. Forest Ecology and Management 2000;138(1):335-56.
[56] Smith GF, Gittings T, Wilson M, French L, Oxbrough A, O'donoghue S, et al. Identifying practical indicators of biodiversity for stand-level management of plantation forests. Biodiversity and Conservation 2008;17(5):991-1015.
[57] De Groot RS, Alkemade R, Braat L, Hein L, Willemen L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 2010;7(3):260-72.
[58] MEA Board. Ecosystems and human well-being : current state and trends : findings of the condition and trends working group. Hassan R, Scholes R, Ash N, editors. Washington, DC: Island Press; 2005.

[59] Costanza R, d'Arge R, De Groot R, Farber S, Grasso M, Hannon B, et al. The value of the world's ecosystem services and natural capital. Nature 1997;387(6630):253-60.
[60] Potschin MB, Haines-Young RH. Ecosystem services: exploring a geographical perspective. Progress in Physical Geography 2011;35(5):575-94.
[61] Petz K, van Oudenhoven APE. Modelling land management effect on ecosystem functions and services: a study in the Netherlands. International Journal of Biodiversity Science, Ecosystem Services & Management 2012;8(1-2):135-55.
[62] Van Oudenhoven APE, Petz K, Alkemade R, Hein L, de Groot RS. Framework for systematic indicator selection to assess effects of land management on ecosystem services. Ecological Indicators 2012;21:110-22.
[63] Maes J, Teller A, Erhard M, Liquete C, Braat L, Berry P, et al. Mapping and assessment of ecosystems and their services: An analytical framework for ecosystem assessments under Action 5 of the EU Biodiversity Strategy to 2020. Luxemburg: Publications office of the European Union; 2013.
[64] Kandziora M, Burkhard B, Müller F. Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators—A theoretical matrix exercise. Ecological Indicators 2013;28:54-78.
[65] Ojima D, Moran E, McConnell W, Stafford Smith M, Laumann G, Morais J, et al. Science Plan and Implementation Strategy. Young B, editor. Stockholm: IGBP Secretariat; 2005.
[66] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global consequences of land use. Science 2005;309(5734):570-4.
[67] CICES. CICES V 4.3. [Spreadsheet on the Internet updated 2013]. University of Nottingham: Nottingham [cited 2013 January 15]; Available from: http://cices.eu/
[68] Dale VH, Beyeler SC. Challenges in the development and use of ecological indicators. Ecological Indicators 2001;1(1):3-10.
[69] Kienast F, Bolliger J, Potschin M, de Groot RS, Verburg PH, Heller I, et al. Assessing landscape functions with broad-scale environmental data: insights gained from a prototype development for Europe. Environmental Management 2009;44(6):1099-120.
[70] McElhinny C, Gibbons P, Brack C, Bauhus J. Forest and woodland stand structural complexity: its definition and measurement. Forest Ecology and Management 2005;218(1):1-24.
[71] Wascher DM. Agri-environmental Indicators: For Sustainable Agriculture in Europe. Tilburg: European Centre for Nature Conservation; 2000.
[72] Sposito GE. Scale dependence and scale invariance in hydrology: Cambridge University Press; 1998.
[73] Turner M, O'Neill R, Gardner R, Milne B. Effects of changing spatial scale on the analysis of landscape pattern. Landscape Ecololgy 1989;3(3-4):153-62.
[74] João E. How scale affects environmental impact assessment. Environmental Impact Assessment Review 2002;22(4):289-310.
[75] João E. A research agenda for data and scale issues in Strategic Environmental Assessment (SEA). Environmental Impact Assessment Review 2007;27(5):479-91.
[76] BEFSCI. A compilation of bioenergy sustainability initiatives. [Monographs (each certification scheme) on the Internet updated 2010-11]. Rome: FAO [cited 2012 December 10]; [about 8 screens]. Available from: http://www.fao.org/bioenergy/foodsecurity/befsci/compilation/en/
[77] Vissers P, Paz A, Hanekamp E. How to select a biomass certification scheme? . In: Lammers E, Op de Coul M, editors. Netherlands Programmes Sustainable Biomass Utrecht: NL Agency; 2011.
[78] EC. Biofuels – Sustainability Schemes [Internet]. Brussels: European Commission [cited 2012 December 20]; [about 5 screens]. Available from: http://ec.europa.eu/energy/renewables/biofuels/sustainability_schemes_en.htm
[79] FSC. Forest Stewardship Council Certification - Ensuring environmental, social and economic benefits [Internet]. Bonn: FSC International Center gemeinnützige Gesellschaft mbH [cited 2012 December 12]. [about 9 screens]. Available from: https://ic.fsc.org/certification.4.htm

[80] SFI. Requirements for the SFI 2010-2014 Program: Standards, Rules for Label Use, Procedures and Guidance. [Monograph on the Internet updated 2010 January]. Washington DC: Sustainable Forestry Initiative Inc. [cited 2012 December 7]; [About 2 screens]. Available from: www.sfiprogram.org

[81] West TO, Post WM. Soil organic carbon sequestration rates by tillage and crop rotation. Soil Science Society of America Journal 2002;66(6):1930-46.

[82] Raudsepp-Hearne C, Peterson G, Bennett E. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences 2010;107(11):5242-7.

2.8 Appendix A

Further rating scales for the indicator requirements

Table 2.A1: Further rating scales for the reliability of indicators, *Validity of indicators* adapted from Bockstaller et al. [1] *(upper part)* and *Response time (lower part)*.

Validity of indicators
1 indicator not coherent with measurements for the majority of cases or not testable
2 indicator rejected by scientific literature or only calculation methodology confirmed
2 indicator coherent with data in very few cases (=partly validated model)
3 indicator partly rejected and partly recognized by scientific literature ("debated") or hardly testable (*Indicator type* = 1 or 2)
4 indicator coherent with validated model
4 indicator recognized by scientific literature
5 indicator coherent with observed data

Response time
1 more than 10 years
2 more than 5 years
3 more than one year but less than 5 years
3 no response measured (*Indicator type* = 1 or 2)
4 less than one year
5 immediate

Table 2.A2: Further rating scales for the feasibility of indicators, *Data requirement* developed based on Bockstaller et al. [2] *(upper part)*, *Qualification requirement* adapted from Bockstaller et al. [2] *(middle part)*, and *Clearly defined thresholds (lower part)*.

Data requirement	
1	assessment/measurement at field/farm scale required
2	1 or 3 possible, but variable
3	data available at other scales, but encompassing site
4	only field observations or questionnaires, but no measurement required
5	data available at relevant scale (e.g. authorities, literature, etc.)

Qualification requirement	
1	higher education in specific subject and specialization such as taxonomy (e.g. for indicator species), hydrology or soil chemistry
2	higher education in specific subject (e.g. Forestry, agricultural sciences) necessary
3	vocational training in e.g. forestry or agriculture necessary
4	no education required, but short introduction to indicator sufficient
5	no specific agricultural education (=subsistence farmer) and no introduction for assessor

Clearly defined thresholds	
1	no target value/threshold or link to derive it
3	source or clear description how to derive a target value/threshold indicated
3	only for process indicators: Target value=application of requested management practice
5	target value/thresholds indicated

Standard error of the mean for indicator requirements

The standard error of the mean (SEM) shows comparable high values for the requirements for *Indicator type* and the forestry indicator set. For the indicator requirements, the rating is quite low throughout the certification schemes. For the forestry indicator set, this higher SEM seems reasonable because the evaluated indicators originate from different studies on woodfuel indicators whereas the remaining indicators belong to a single certification scheme.

At the lower aggregation level per composite scale, i.e., the environmental impact category, a large number of certification schemes only use one[1] or no indicator (*#NV*) per composite scale and certification scheme; see Table 2.A4 to Table 2.A10. A slightly smaller number of certification

[1] Indicators that neither have a conceptual or informational difference, e.g. "4.1.1 Biomass is not produced on land with high biodiversity value […]" and "4.1.2 Biomass is not produced on highly biodiverse grassland", [7], are considered to be a single indicator for the rating.

schemes apply several indicators of similar types; this situation results in an equal rating, denoted as 0% in Table 2.A4. For example, GBEP uses Driver indicators (value 2) on water availability; i.e., the management practices are related to a state or impact as shown in Table 2.1. I.e., water abstraction is measured per unit biomass and per watershed and is related to the local amount of renewable and non-renewable water.

A high SEM, which is especially abundant for the requirement *Indicator type* in Table 2.A4, shows that a certification scheme contains indicators with great differences. For example, water quality in the draft by the IWPB consists of three indicators, of which two are driver indicators (value 1-2) that assess the amount of N, P and active pesticide ingredients applied per ha per year and the practices to avoid fertilizer runoff to groundwater and surface water bodies during application; the third is an impact indicator (value 5) that assess the biological oxygen demands on and near the production unit.

Table 2.A3: SEM for the indicators by indicator requirement for the evaluated certification schemes and indicator sets: CSBP [3], GBEP [4], GGL [5], GlobalGAP [6], ISCC [7], IWPB [8], Netherlands Standardization Institute [9], 10], REDcert [11], RSB [12], SAN [13] and forestry [14-19]

	Indicator type	Validity of indicators	Response time	Data requirement	Qualification requirement	Required resources	Clearly defined thresholds
GBEP	0.36	0.20	0.22	0.20	0.19	0.23	0.24
NTA8080	0.32	0.17	0.23	0.23	0.21	0.21	0.24
ISCC	0.29	0.17	0.18	0.28	0.23	0.19	0.14
REDcert	0.46	0.20	0.25	0.36	0.23	0.34	0.28
GGL	0.65	0.31	0.33	0.17	0.54	0.42	0.00
RSB	0.34	0.25	0.21	0.27	0.23	0.26	0.18
CSBP	0.37	0.20	0.20	0.35	0.27	0.25	0.00
IWPB	0.37	0.23	0.24	0.30	0.26	0.22	0.24
SAN	0.29	0.16	0.24	0.19	0.19	0.20	0.20
GlobalGAP	0.30	0.20	0.27	0.27	0.30	0.20	0.22
Forestry	0.72	0.69	0.88	0.23	0.53	0.52	0.53

Table 2.A4: SEM for the rating requirement *Indicator type* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	1.20	0.84	0.00
NTA8080	0.42	1.00	0.50	0.73	0.33	#NV
ISCC	0.39	0.49	0.00	#NV	1.50	#NV
REDcert	0.50	0.43	#NV	#NV	#NV	#NV
GGL	#NV	0.00	#NV	#NV	#NV	#NV
RSB	0.40	0.00	#NV	0.65	#NV	#NV
CSBP	0.71	0.50	0.00	1.20	#NV	#NV
IWPB	0.60	1.33	0.33	#NV	0.67	#NV
SAN	0.40	0.00	#NV	0.50	#NV	#NV
GlobalGAP	1.00	0.43	0.00	0.00	#NV	#NV
Forestry	1.40	0.76	0.33	0.41	#NV	0.33

Table 2.A5: SEM for the rating requirement *Validity of indicators* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.33	0.33	0.40	0.50
NTA8080	0.26	0.50	0.00	0.37	0.67	#NV
ISCC	0.28	0.22	0.50	#NV	#NV	#NV
REDcert	0.00	0.29	#NV	#NV	#NV	#NV
GGL	#NV	0.41	#NV	#NV	#NV	#NV
RSB	0.49	0.88	#NV	0.29	#NV	#NV
CSBP	0.37	0.31	0.50	0.33	#NV	#NV
IWPB	0.40	0.00	0.33	#NV	0.67	#NV
SAN	0.20	0.48	#NV	0.23	#NV	#NV
GlobalGAP	0.50	0.26	0.50	0.50	#NV	#NV
Forestry	1.36	0.26	0.21	0.18	#NV	0.33

Table 2.A6: SEM for the rating requirement *Response time* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	0.67	0.49	0.00
NTA8080	0.11	0.50	0.00	0.75	0.00	#NV
ISCC	0.20	0.17	0.00	#NV	1.00	#NV
REDcert	0.00	0.14	#NV	#NV	#NV	#NV
GGL	#NV	0.00	#NV	#NV	#NV	#NV
RSB	0.40	0.00	#NV	0.29	#NV	#NV
CSBP	0.40	0.17	0.00	0.67	#NV	#NV
IWPB	0.19	0.67	0.00	#NV	1.33	#NV
SAN	0.40	0.00	#NV	0.40	#NV	#NV
GlobalGAP	1.00	0.14	0.00	1.00	#NV	#NV
Forestry	1.73	0.40	0.00	0.26	#NV	0.00

Table 2.A7: SEM for the rating requirement *Data requirement* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	0.33	0.20	0.00
NTA8080	0.42	0.87	0.00	0.24	0.00	#NV
ISCC	0.44	0.63	0.00	#NV	0.50	#NV
REDcert	0.50	0.55	#NV	#NV	#NV	#NV
GGL	#NV	0.00	#NV	#NV	#NV	#NV
RSB	0.51	1.00	#NV	0.40	#NV	#NV
CSBP	0.75	0.63	0.00	0.33	#NV	#NV
IWPB	0.50	1.00	0.00	#NV	0.33	#NV
SAN	0.51	0.00	#NV	0.31	#NV	#NV
GlobalGAP	1.50	0.43	0.00	0.00	#NV	#NV
Forestry	0.39	0.67	0.56	0.35	#NV	0.00

Table 2.A8: SEM for the rating requirement *Qualification requirement* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	0.00	0.20	0.50
NTA8080	0.29	0.29	0.00	0.40	0.33	#NV
ISCC	0.30	0.40	0.00	#NV	0.00	#NV
REDcert	0.00	0.34	#NV	#NV	#NV	#NV
GGL	#NV	0.50	#NV	#NV	#NV	#NV
RSB	0.24	0.33	#NV	0.26	#NV	#NV
CSBP	0.24	0.33	0.00	0.00	#NV	#NV
IWPB	0.33	0.58	0.00	#NV	0.33	#NV
SAN	0.24	0.29	#NV	0.25	#NV	#NV
GlobalGAP	0.50	0.36	0.00	0.50	#NV	#NV
Forestry	1.01	0.34	0.60	0.16	#NV	0.58

Table 2.A9: SEM for the rating requirement *Required resources (assessment interval)* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	0.67	0.20	0.50
NTA8080	0.24	0.75	0.00	0.45	0.67	#NV
ISCC	0.15	0.54	0.00	#NV	1.00	#NV
REDcert	0.50	0.47	#NV	#NV	#NV	#NV
GGL	#NV	0.50	#NV	#NV	#NV	#NV
RSB	0.24	1.00	#NV	0.46	#NV	#NV
CSBP	0.20	0.54	0.00	0.67	#NV	#NV
IWPB	0.25	0.88	0.00	#NV	0.67	#NV
SAN	0.24	0.00	#NV	0.36	#NV	#NV
GlobalGAP	0.00	0.29	0.00	0.00	#NV	#NV
Forestry	0.99	0.45	0.56	0.29	#NV	0.33

Table 2.A10: SEM for the rating requirement *Clearly defined thresholds* by composite scale.

	Soil quality	Water quality	Water availability	Biodiversity	LU/LUC	Other
GBEP	#NV	0.00	0.00	0.00	0.00	1.00
NTA8080	0.11	0.58	1.00	0.40	1.33	#NV
ISCC	0.00	0.40	0.00	#NV	1.00	#NV
REDcert	0.00	0.34	#NV	#NV	#NV	#NV
GGL	#NV	0.00	#NV	#NV	#NV	#NV
RSB	0.20	0.00	#NV	0.29	#NV	#NV
CSBP	0.00	0.00	0.00	0.00	#NV	#NV
IWPB	0.13	0.67	0.67	#NV	0.67	#NV
SAN	0.00	0.50	#NV	0.30	#NV	#NV
GlobalGAP	0.00	0.34	0.00	0.00	#NV	#NV
Forestry	1.02	0.52	0.45	0.28	#NV	0.67

Evaluated certification schemes mapped on the ESS cascade

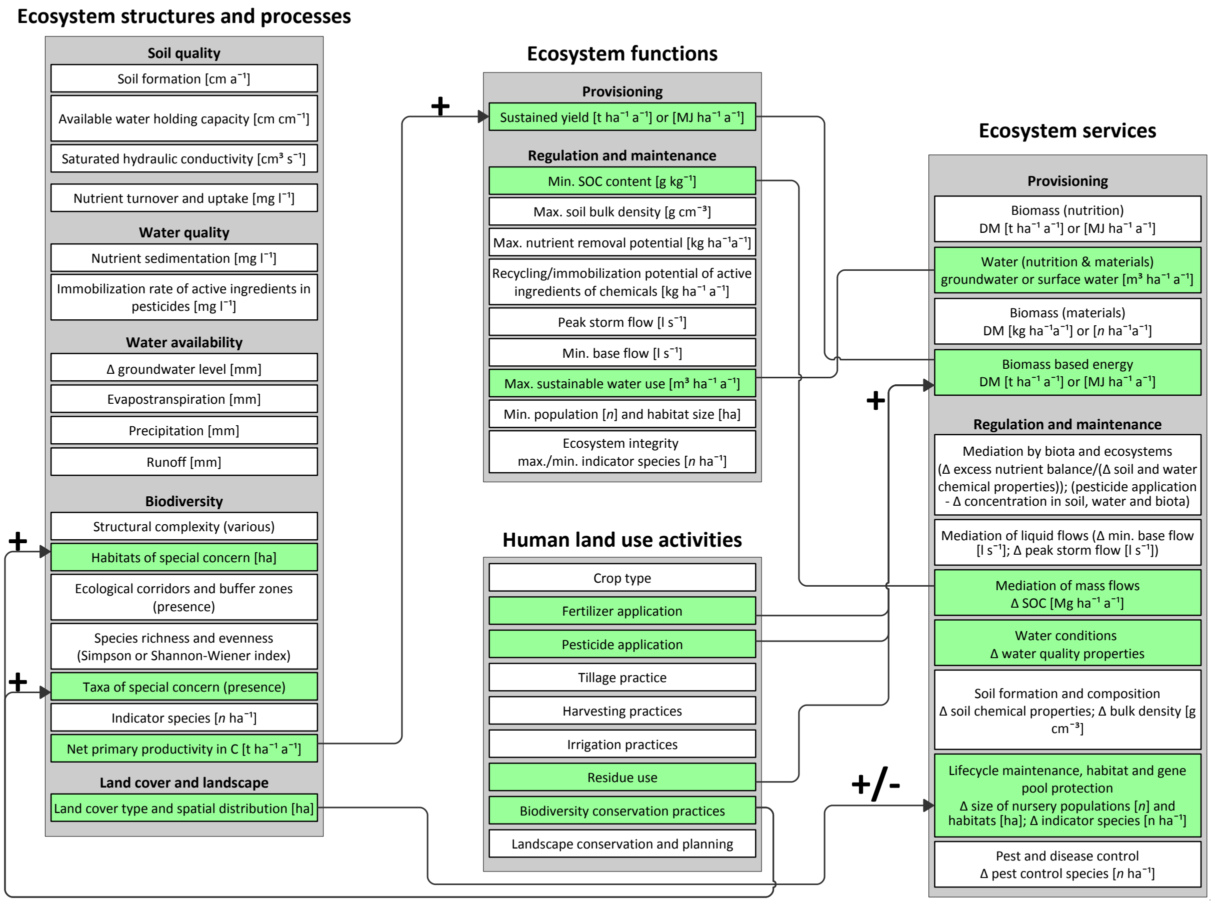

Fig. 2.A1: Indicators from GBEP [4] mapped on the ESS cascade

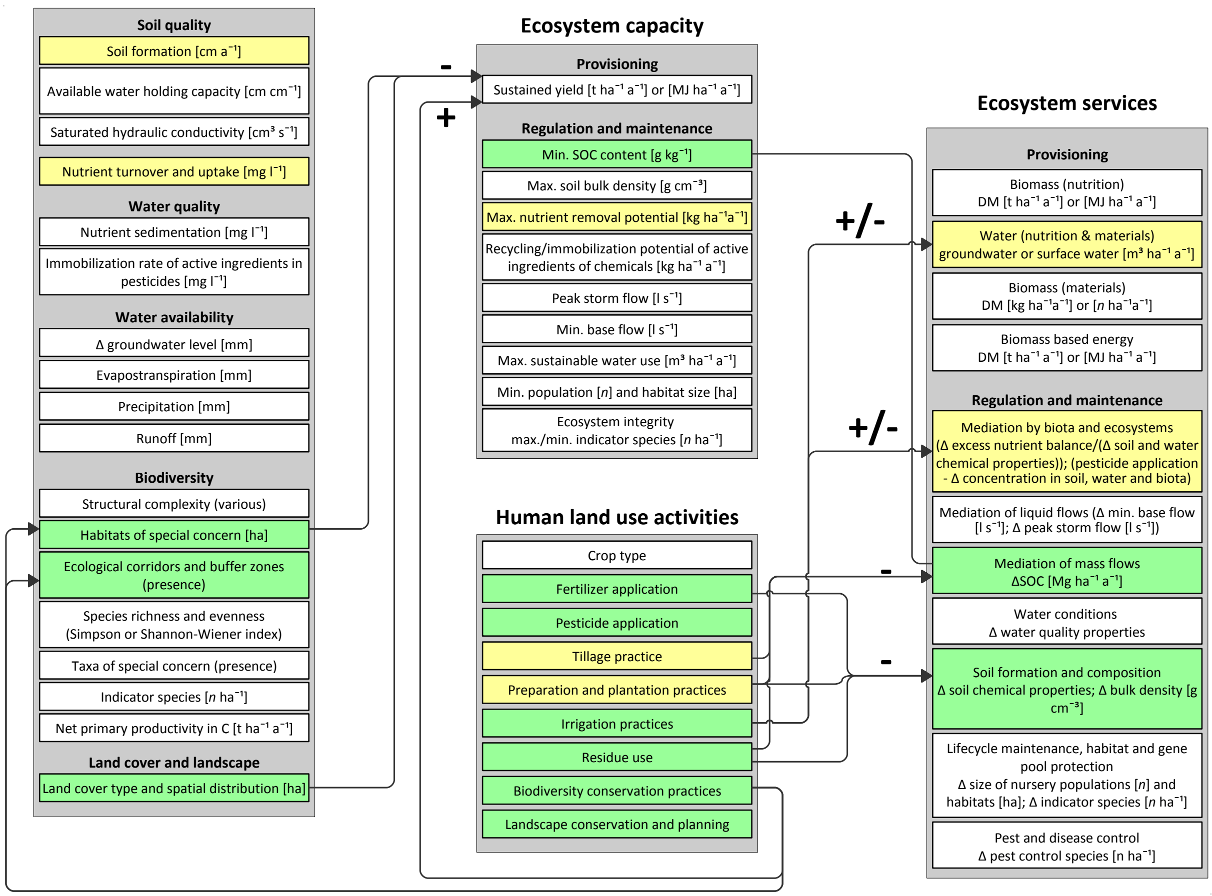

Fig. 2.A2: Indicators from NTA8080 [9, 10] mapped on the ESS cascade

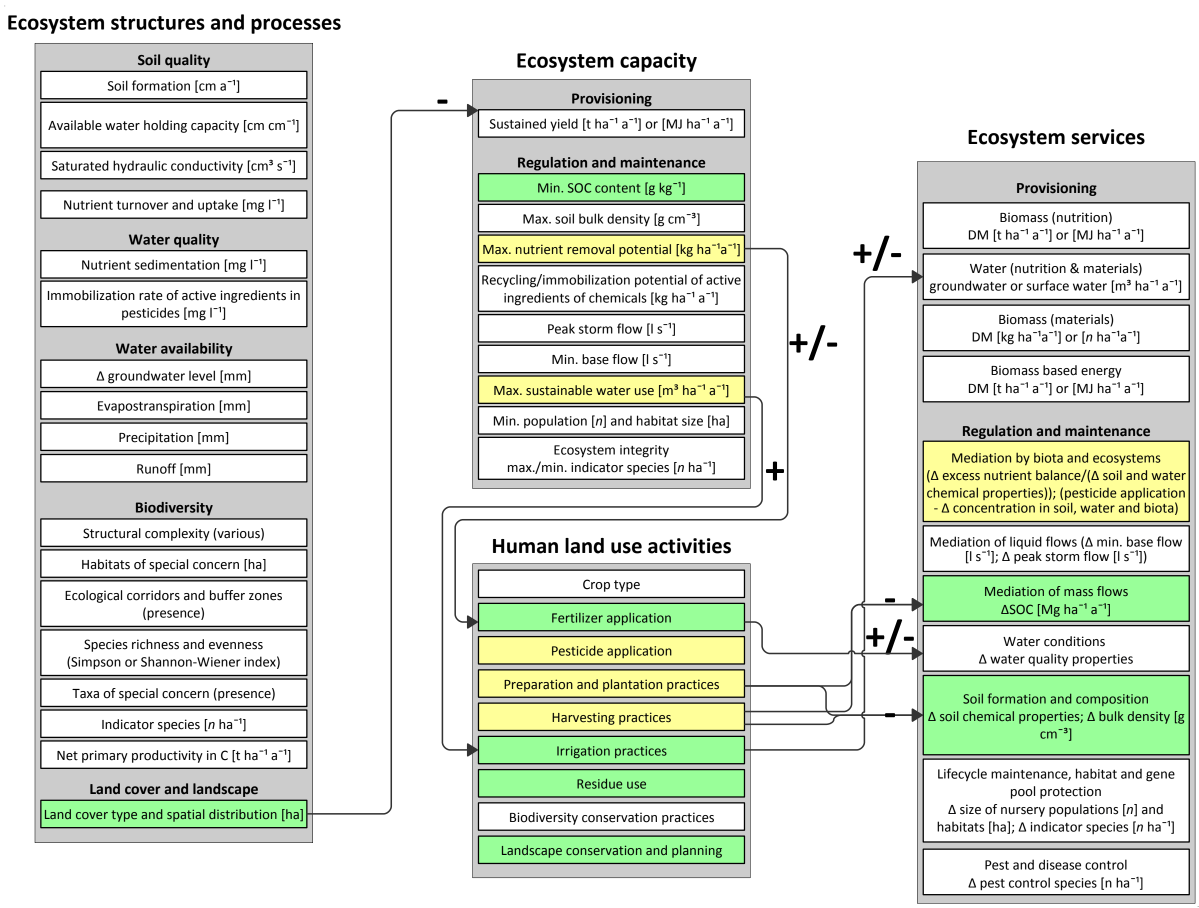

Fig. 2.A3: Indicators from ISCC [7] mapped on the ESS cascade

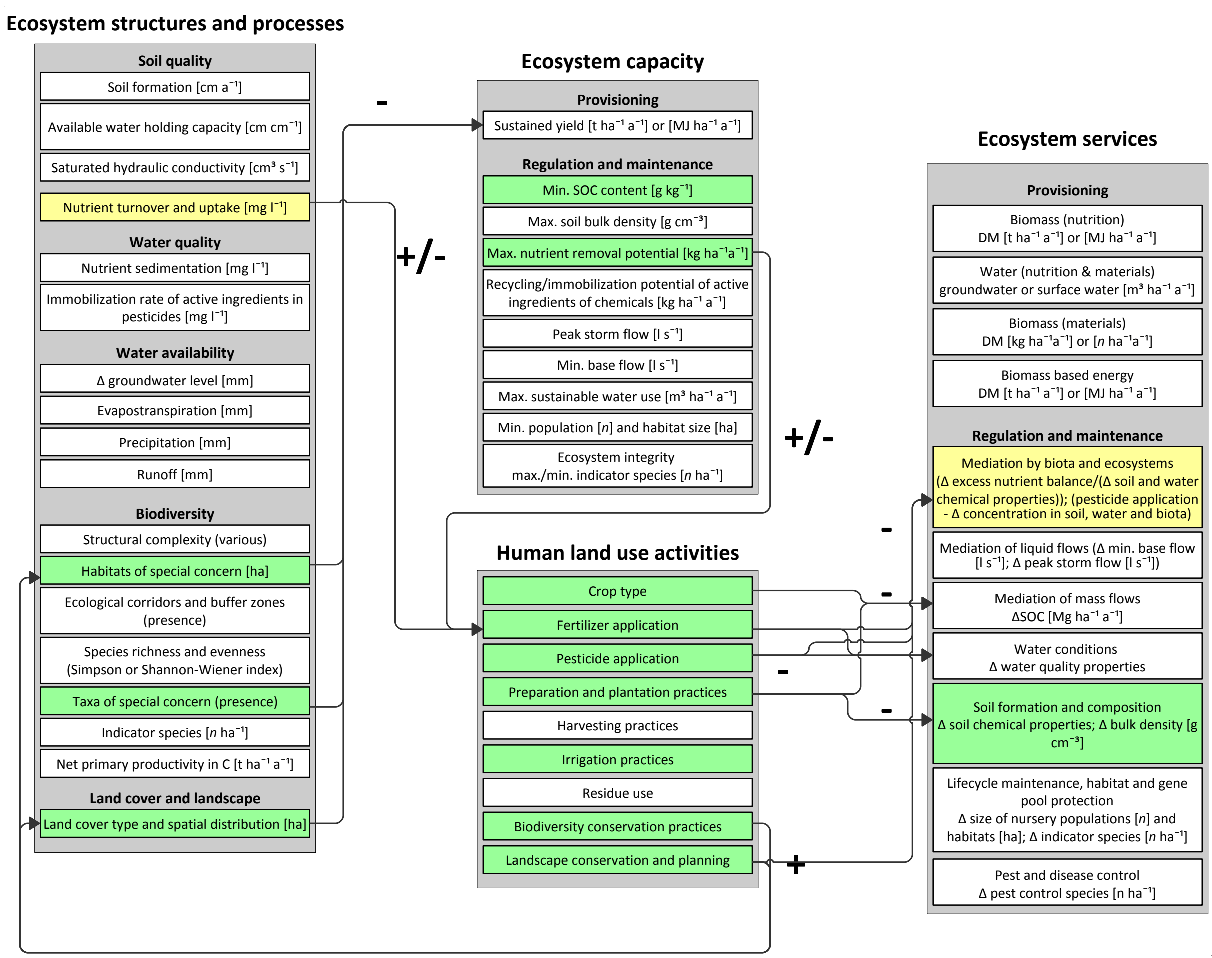

Fig. 2.A4: Indicators from REDcert [11] mapped on the ESS cascade

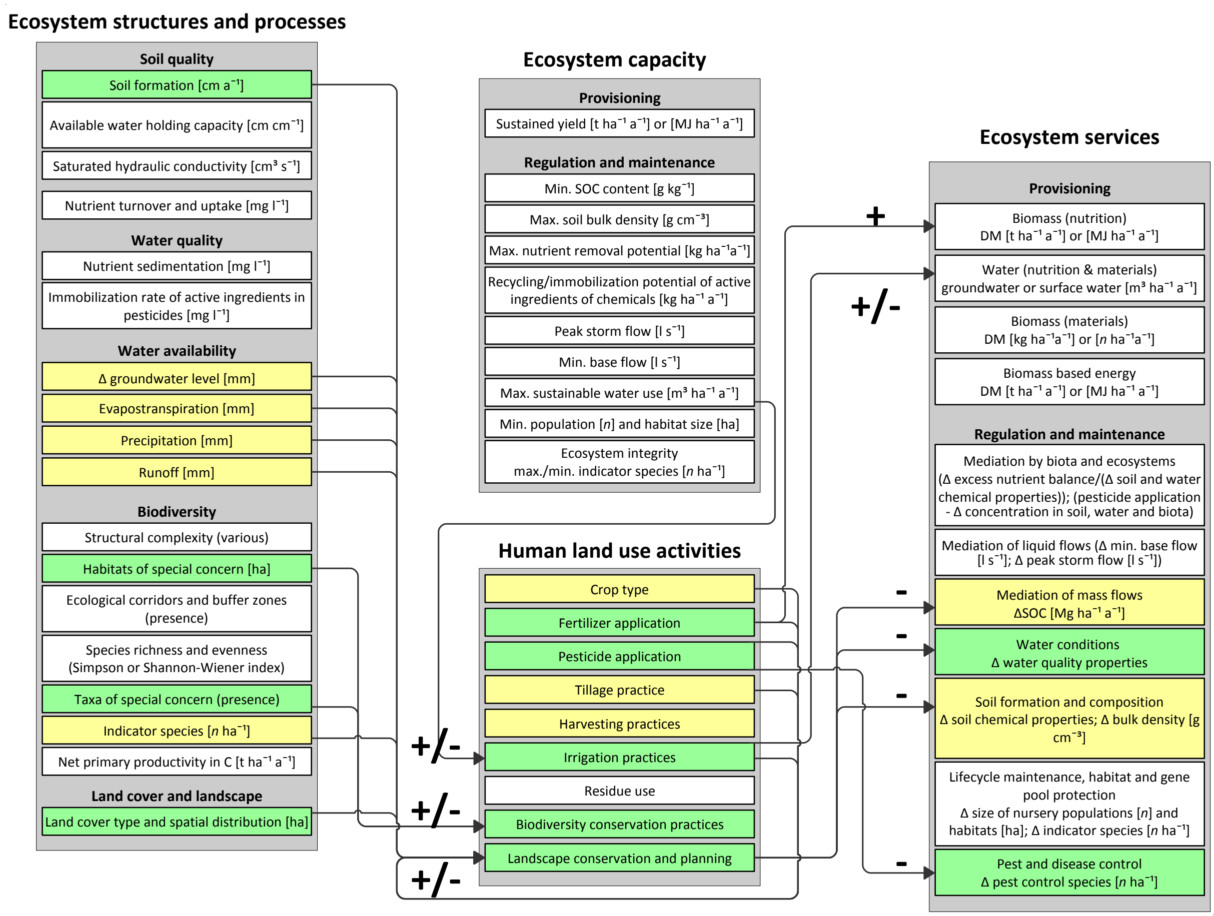

Fig. 2.A5: Indicators from GGL, *Agricultural Source criteria*, [5] mapped on the ESS cascade

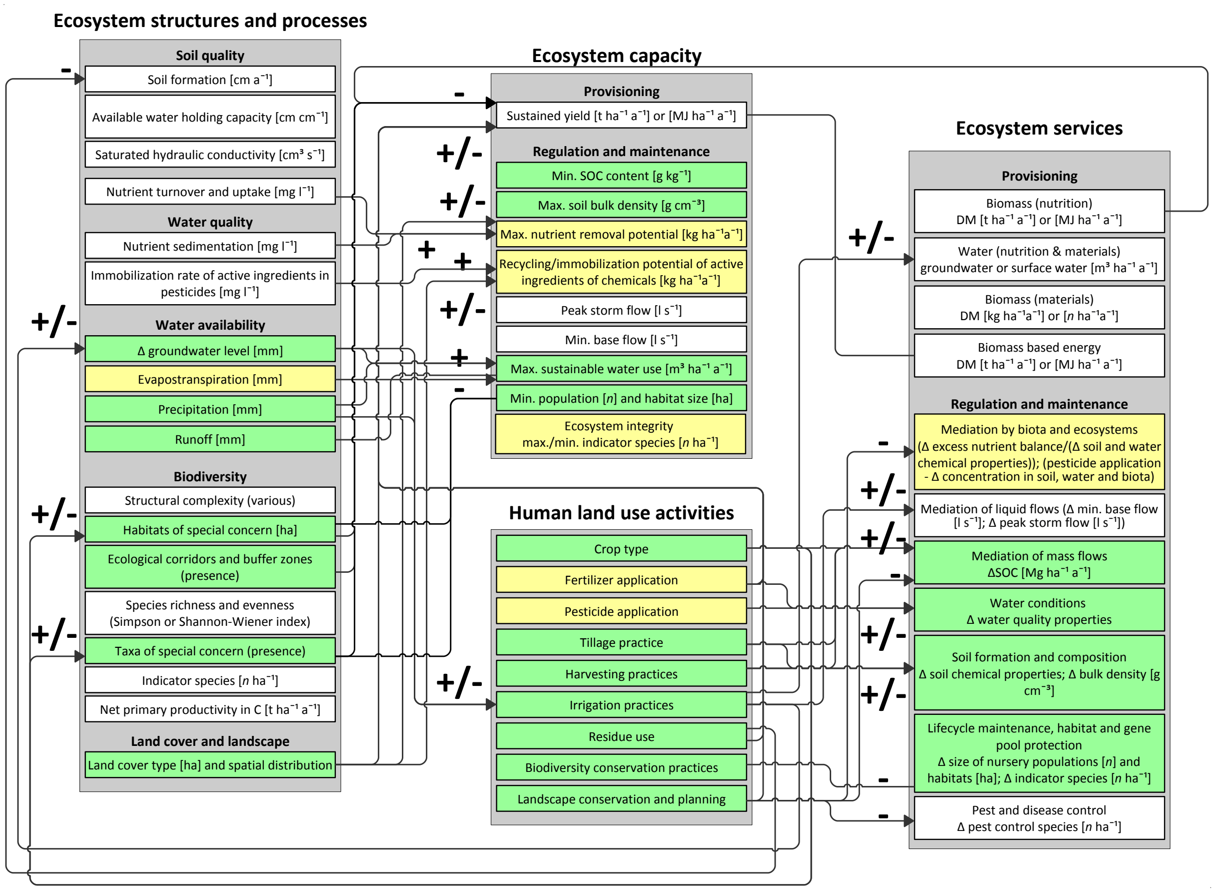

Fig. 2.A6: Indicators from RSB [12] mapped on the ESS cascade

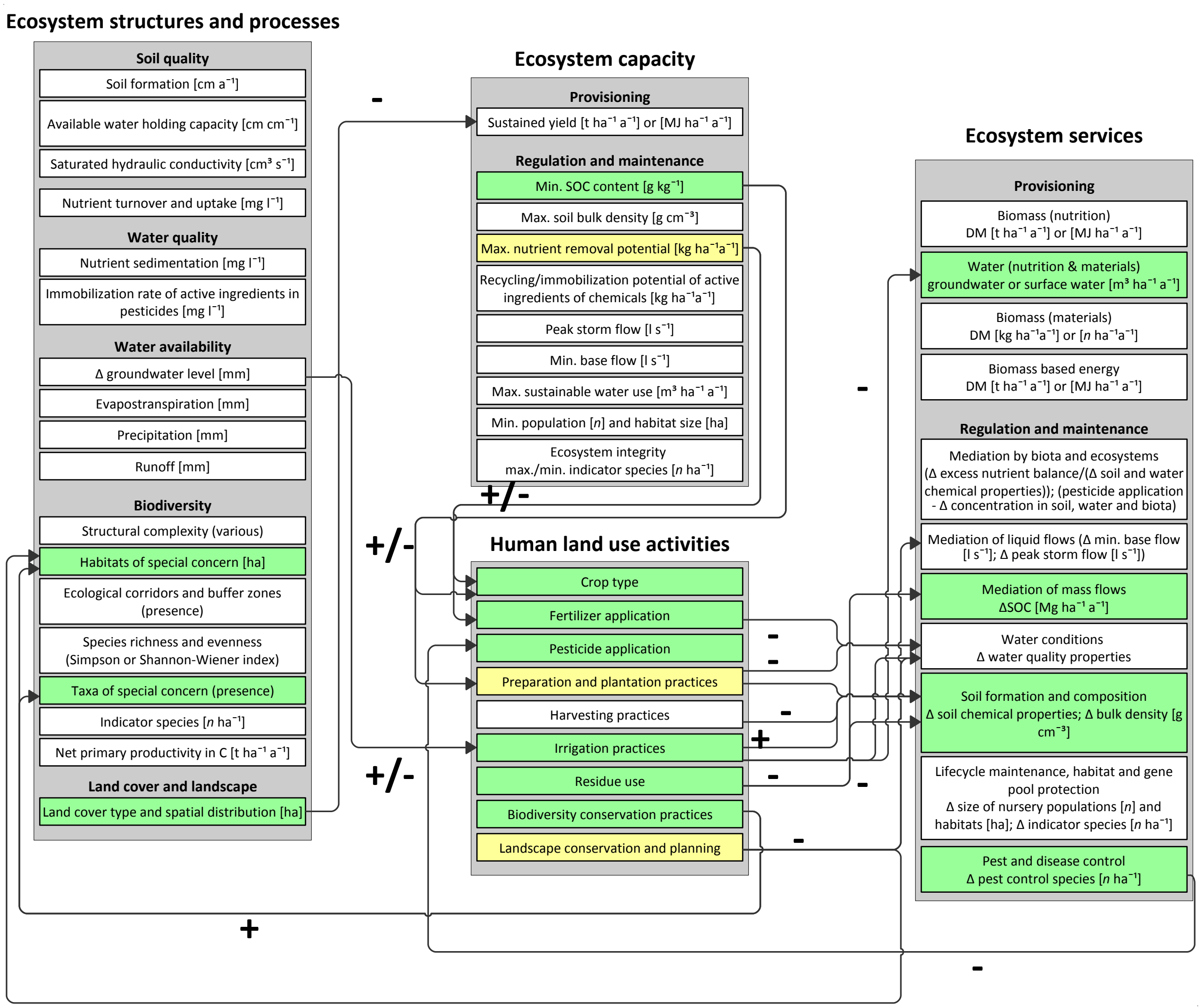

Fig. 2.A7: Indicators from CSBP [3] mapped on the ESS cascade

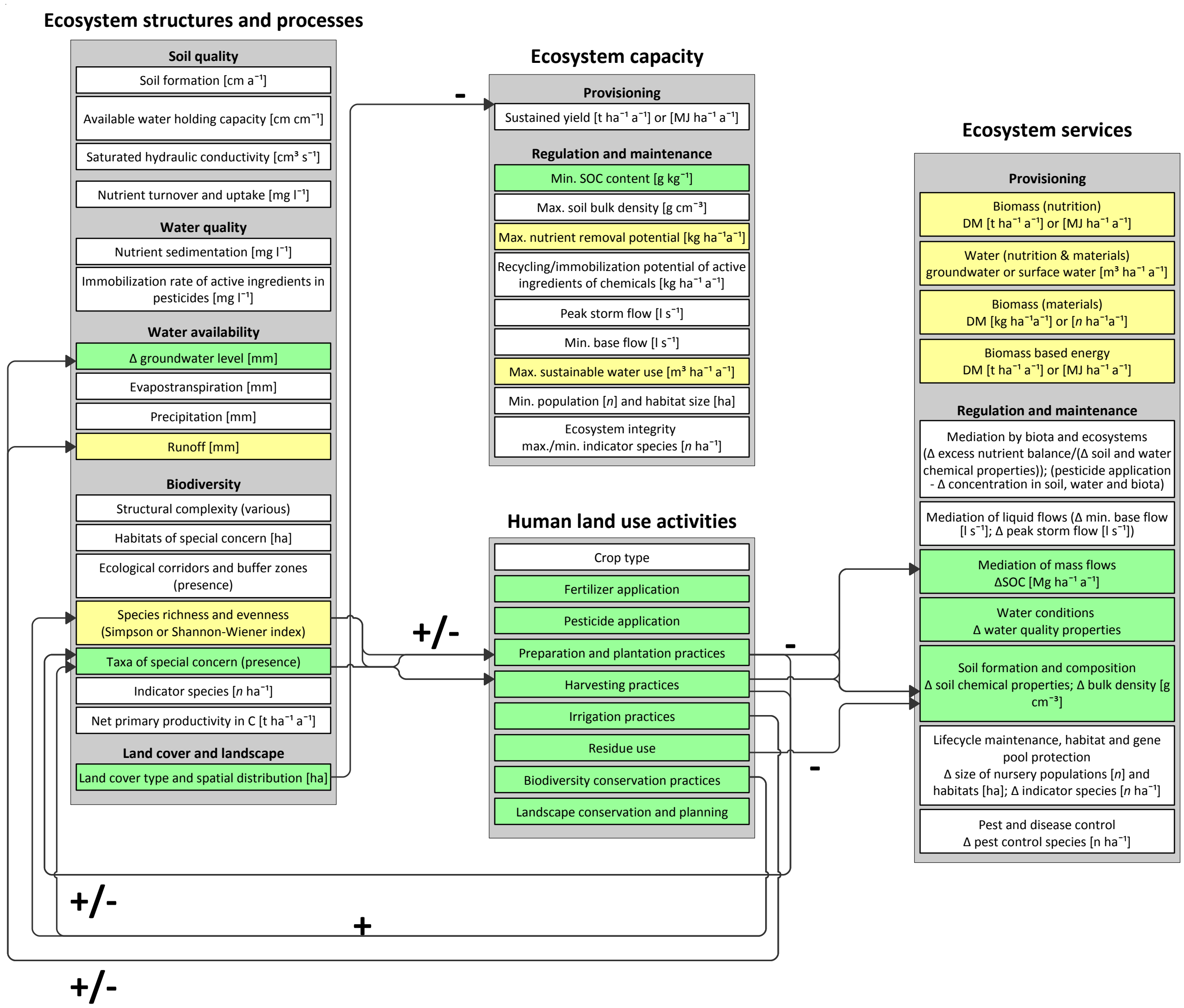

Fig. 2.A8: Indicators from IWPB [8] mapped on the ESS cascade

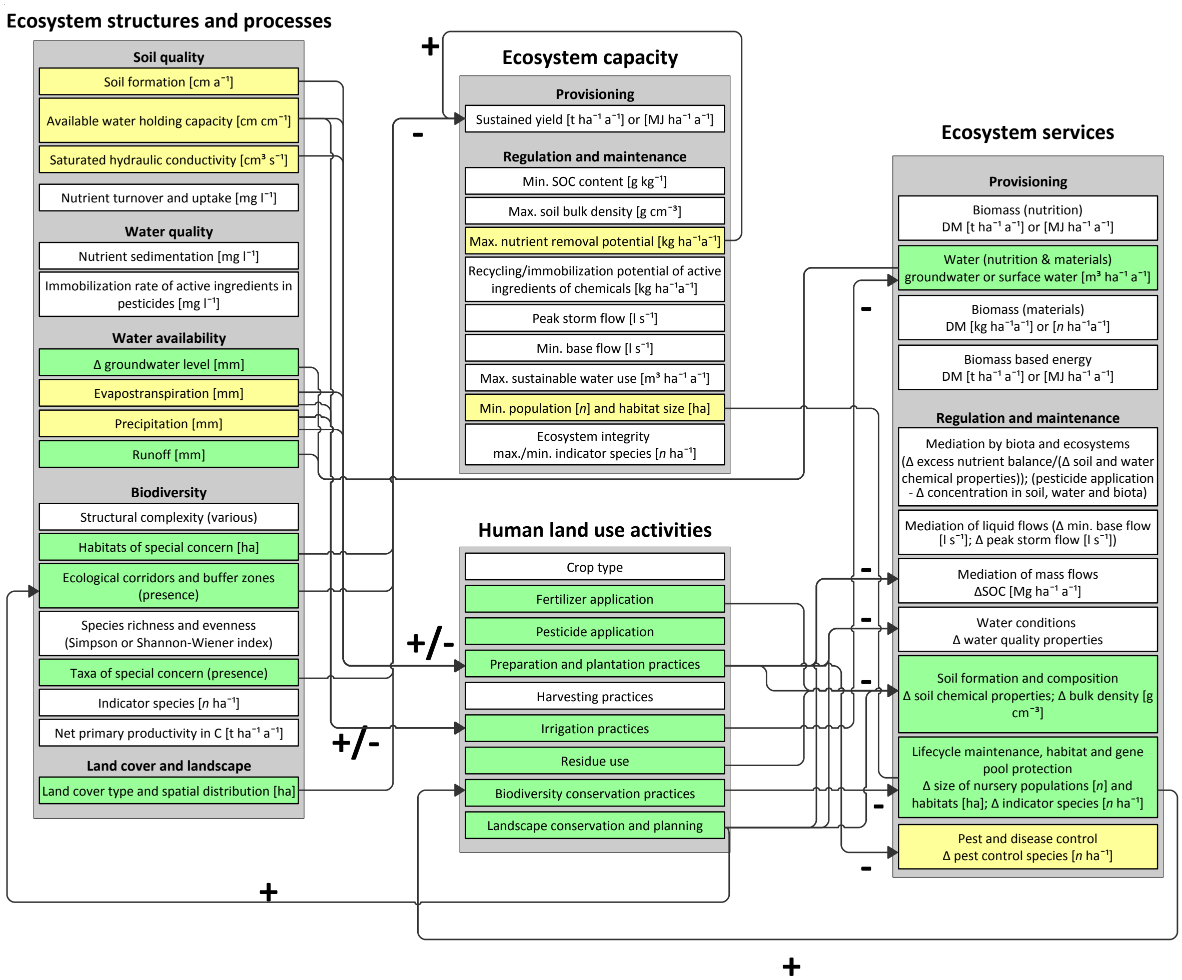

Fig. 2.A9: Indicators from SAN [13] mapped on the ESS cascade

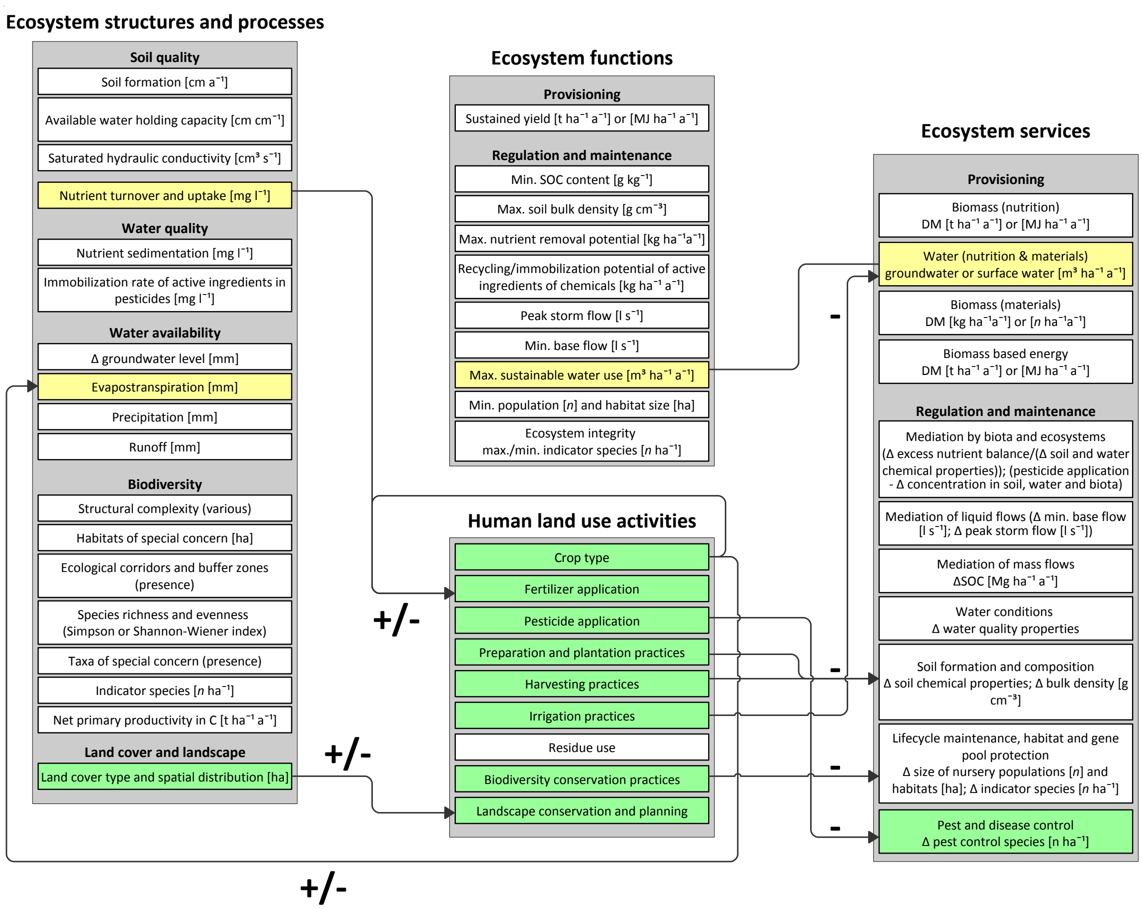

Fig. 2.A10: Indicators from GlobalGAP [6] mapped on the ESS cascade

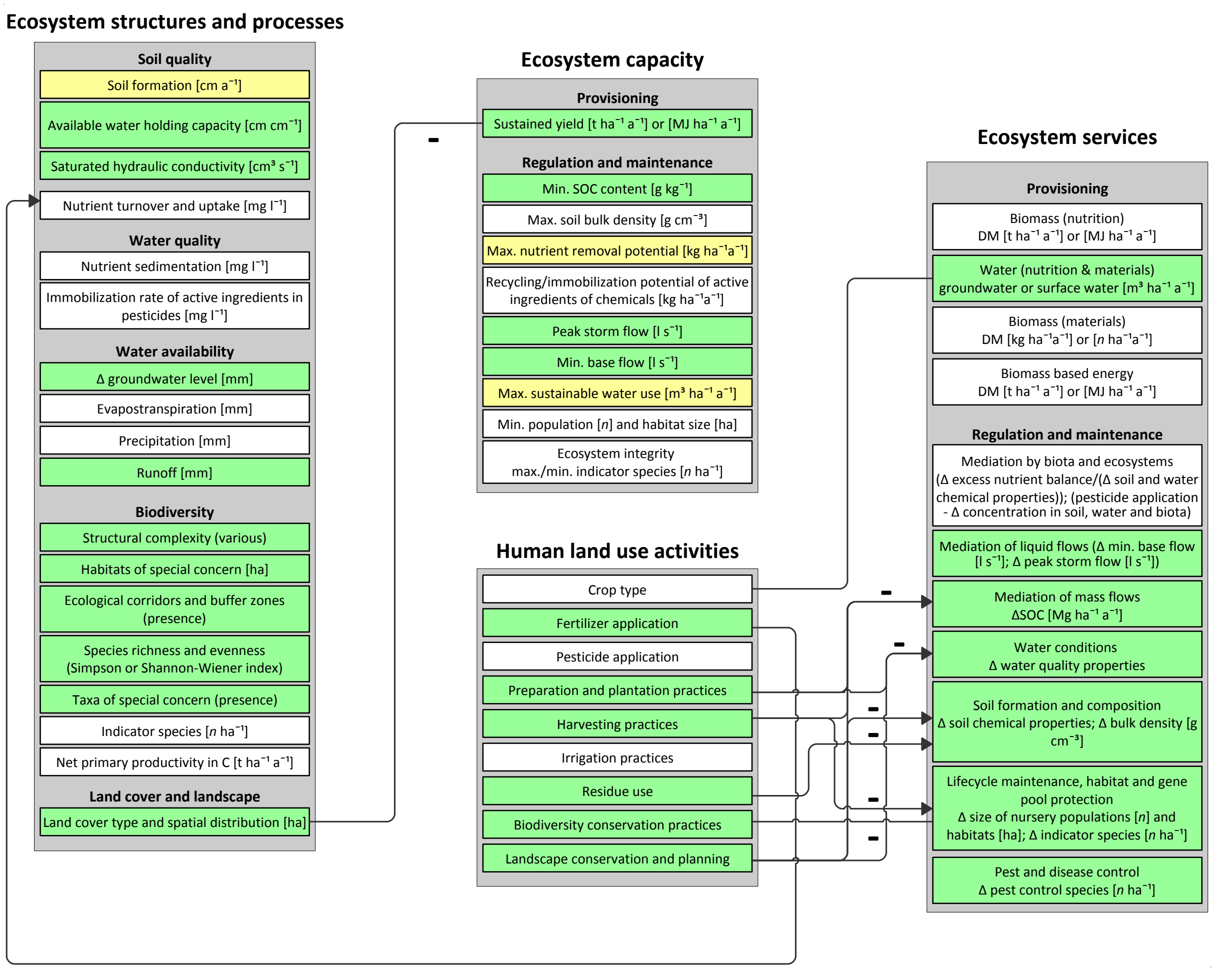

Fig. 2.A11: Forestry indicators [14-19] mapped on the ESS cascade

References

[1] Bockstaller C, Guichard L, Keichinger O, Girardin P, Galan M-B, Gaillard G. Comparison of methods to assess the sustainability of agricultural systems. A review. Agronomy for Sustainable Development 2009;29(1):223-35.

[2] Bockstaller C, Gaillard G, Baumgartner D, Freiermuth Knuchel R, Reinsch M, Brauner R, et al. Betriebliches Umweltmanagement in der Landwirtschaft: Vergleich der Methoden INDIGO, KUL/USL, REPRO und SALCA Colmar: Grenzüberschreitendes Institut zur rentablen umweltgerechten Landbewirtschaftung; 2006.

[3] CSBP. Standard for sustainable production of agricultural biomass. [Monograph on the Internet updated 2012]. Council on Sustainable Biomass Production [cited 2012 December 21]; [about 1 screen]. Available from: http://www.csbp.org/

[4] GBEP. The Global Bioenergy Partnership Sustainability Indicators for Bioenergy. Rome: GBEP Secretariat; FAO; 2011.

[5] GGL. Certification Requirements Green Gold Label. [Monograph on the Internet updated 2011]. Zwolle: Green Gold Label Foundation [cited 2012 December 6]; [about 1 screen]. Available from: http://www.greengoldcertified.org

[6] GlobalGAP. Integrated Farm Assurance: All farm base/crops base/combinable crops. [Monographs on the Internet updated 2012]. Cologne: FoodPLUS GmbH [cited 2012 December 8]; [about 4 screens]. Available from: http://www.globalgap.org/uk_en/

[7] ISCC. ISCC EU System Documents. [Monographs on the Internet updated 2011]. Cologne: ISCC System GmbH. [cited 2012 December 4]; [about 5 screens]. Available from: http://www.iscc-system.org/en/

[8] IWPB. IWPB Sustainability principles/IWPB Sustainability criteria and indicators. [Monographs on the Internet updated 2012]. Linkebeek: Laborelec. [cited 2012 December 21]; [about 3 screens]. Available from: www.laborelec.be/ENG/initiative-wood-pellet-buyers-iwpb/

[9] NTA 8080:2009 NTA 8080 - Sustainability criteria for biomass for energy purposes. 2009. 1-41.

[10] NTA 8081:2012-04 NTA 8081 - Certification scheme for sustainably produced biomass for energy purposes. 2012. 1-27.

[11] REDcert. REDcert-EU system documents. [Monographs on the Internet updated 2012]. Bonn: REDcert GmbH [cited 2012 December 12]; [about 3 screens]. Available from: http://www.redcert.org

[12] RSB. RSB Sustainability Standards – EU RED. [Monographs on the Internet updated 2011]. Chatelaine: Roundtable on Sustainable Biomaterials [cited 2012 December 8]; [about 5 screens]. Available from: http://rsb.org/sustainability/rsb-sustainability-standards/

[13] SAN. Farm Standards/Group Standards. [Monographs on the Internet updated 2011]. San José: Sustainable Agriculture Network [cited 2012 December 6]; [About 2 screens]. Available from: http://sanstandards.org/sitio/

[14] Diaz-Chavez RA. Assessing biofuels: Aiming for sustainable development or complying with the market? Energy Policy 2011;39(10):5763–9.

[15] Lattimore B, Smith C, Titus B, Stupak I, Egnell G. Environmental factors in woodfuel production: Opportunities, risks, and criteria and indicators for sustainable practices. Biomass and Bioenergy 2009;33(10):1321-42.

[16] Newton AC, Kapos V. Biodiversity indicators in national forest inventories. Unasylva 2002;53(210):56-64.

[17] Schoenholtz SH, Miegroet HV, Burger J. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. Forest Ecology and Management 2000;138(1):335-56.

[18] Smith GF, Gittings T, Wilson M, French L, Oxbrough A, O'donoghue S, et al. Identifying practical indicators of biodiversity for stand-level management of plantation forests. Biodiversity and Conservation 2008;17(5):991-1015.

[19] Stupak I, Lattimore B, Titus BD, Tattersall Smith C. Criteria and indicators for sustainable forest fuel production and harvesting: A review of current standards for sustainable forest management. Biomass and Bioenergy 2011;35(8):3287-308.

3 Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand

This chapter is published by PLOS ONE:

Meyer MA, Chand T, Priess JA Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLOS ONE 2015;10(3):e0116336.

The definitive version is available at:

http://journals.plos.org/plosone/article?id=10.1371/journal.pone.0116336.

3.1 Abstract

Biomass for bioenergy is debated for its potential synergies or tradeoffs with other provisioning and regulating ecosystem services (ESS). This biomass may originate from different production systems and may be purposefully grown or obtained from residues. Increased concerns globally about the sustainable production of biomass for bioenergy has resulted in numerous certification schemes focusing on best management practices, mostly operating at the plot/field scale. In this study, we compare the ESS of two watersheds in the southeastern US. We show the ESS tradeoffs and synergies of plantation forestry, i.e., pine poles, and agricultural production, i.e., wheat straw and corn stover, with the counterfactual natural or semi-natural forest in both watersheds. The plantation forestry showed less distinct tradeoffs than did corn and wheat production, i.e., for carbon storage, P and sediment retention, groundwater recharge, and biodiversity. Using indicators of landscape composition and configuration, we showed that landscape planning can affect the overall ESS supply and can partly determine if locally set environmental thresholds are being met. Indicators on landscape composition, configuration and naturalness explained more than 30 % of the variation in ESS supply. Landscape elements such as largely connected forest patches or more complex agricultural patches, e.g., mosaics with shrub and grassland patches, may enhance ESS supply in both of the bioenergy production systems. If tradeoffs between biomass production and other ESS are not addressed by landscape planning, it may be reasonable to include rules in certification schemes that require, e.g., the connectivity of natural or semi-natural forest patches in plantation forestry or semi-natural landscape elements in agricultural production systems. Integrating indicators on landscape configuration and composition into certification schemes is particularly relevant considering that certification schemes are governance tools used to ensure comparable sustainability standards for biomass produced in countries with variable or absent legal frameworks for landscape planning.

3.2 Introduction

Research in the context of bioenergy and ecosystem services (ESS), the perceived human benefits from ecological systems [1], often focuses on largely debated 1st generation liquid biofuel feedstocks such as maize in the US, sugarcane or soybeans in Brazil, or rapeseed in Europe [2]. Some papers address scenarios with a shift to 2nd generation liquid biofuel feedstocks, such as grasses or other perennial bioenergy feedstocks [3, 4]. Research in this area only partly reflects the fact that only 3 % of the global bioenergy supply was obtained from dedicated energy crops in 2008. More than 80 % of the global bioenergy supply originates from forest biomass [5]. With respect to modern solid bioenergy carriers, wood pellets have experienced an increased global trade volume, accounting for 120 PJ (~660 Mt) of the total global solid bioenergy carrier trade of 300 PJ (~1640 Mt) as of 2010 [6]. For trade between EU and non-EU countries in 2010, the wood pellet trade volume of 45 PJ (~250 Mt) is comparable to those of biodiesel and bioethanol [7].

Increasing forest biomass use and trade may also affect the supply of other ESS, e.g., carbon storage or groundwater recharge [8, 9], or create environmental impacts exceeding the capacity of regulating ESS; e.g., increasing biomass may affect sediment retention due to increased plantation forestry [10]. The expansion of bioenergy production is limited by and competing with the demand for land for other bio-based commodities (food, feed and fiber) [11]. In that respect, a current draft of new sustainability requirements of the EU Renewable Energy Directive (RED) emphasizes the consideration and quantification of tradeoffs of feedstock production for liquid, gaseous and solid bioenergy and other ESS, such as carbon storage or sediment retention [12]. Further research in this context may examine the ability to avoid negative impacts, such as the effects on erosion, carbon storage or biodiversity [2, 7, 13].

Existing studies on ESS supply typically model a single case study area with a mostly heterogeneous or contrasting land use/land cover composition and partly model synergies and tradeoffs in ESS supply, e.g., [14, 15]. Rather homogenous and more intensively managed land use/land cover systems for ESS supply, e.g., systems specialized in forest plantations or agriculture, may require more significant tradeoffs regarding other ESS compared to heterogeneous production systems for biomass. For example, they are more likely to exceed critical environmental thresholds such as erosion control, water purification or recreation due to underrepresented natural or semi-natural vegetation [16, 17]. Increased landscape heterogeneity helps to ensure a balanced supply of biodiversity and regulates ESS, such as the higher nutrient retention efficiency of riparian buffer zones in agricultural landscapes [18]. Larger quantities of bioenergy feedstocks may generate economies of scale for processing and logistics [19], contributing to more homogenous landscapes. Considering that different biomass provision options may become even more important in the near future, we analyze ESS supply both in forest plantations and agricultural systems, which may be used interchangeably. For example, agricultural residues, such as cereal straw or corn stover, currently amount to 4 % of the global bioenergy supply [5]. In the US, cereal straw and corn stover comprise 97 % of the estimated available agricultural residues in the US (2011) [20]. Using residues contributes to reducing or completely avoiding the food versus fuel conflict compared with dedicated energy crops [21]. Direct land use change (LUC) is when biomass production replaces other crops, forests or natural grasslands. Indirect land use change (iLUC) is the clearing of land not specifically for biomass but to meet the demands for other commodities, such as food and fiber, and may occur not only nearby but also in different parts of the region or even different parts of the world [22].

A wide range of factors may influence ESS supply, such as environmental conditions, including the topography, soil characteristics and climate. In contrast, land management may affect ESS supply [23-25]. In the context of bioenergy production, certification schemes are used as a governance tool to ensure sustainable production. They focus on indicators and prescribed management practices mostly applicable at the plot scale [26, 27]. However, certification schemes rarely require indicators at the regional or landscape scale in the context of both bioenergy [12] and agricultural products [28] and payment schemes for ESS [29]. At the landscape/regional scale, i.e., the typical scale of landscape planning, the influence of landscape composition and configuration has been argued [30, 31] and exemplarily demonstrated for single ESS, i.e., soil protection and retention [32] and biodiversity [33].

In this paper, we first assess ESS supply in subtropical watersheds mostly used for (i) forest plantations, *Pinus* spp., and (ii) agricultural production as bioenergy sourcing regions in the southeastern US. Following [34, 17], we expect that the tradeoffs between forest plantations and natural or semi-natural forest as a counterfactual are smaller than between corn and wheat production and natural or semi-natural forest. The remnants of the existing forests reflect the potential natural vegetation in both watersheds [35]. Second, we hypothesize that not only environmental or management factors at the plot scale but also landscape composition and configuration and naturalness assessed at the landscape scale influence ESS supply and biodiversity. Third, we assume that these landscape factors play a role in whether socially accepted environmental thresholds, e.g., water quality, are met within the two contrasting case studies. For example, the connectivity or dominance of patches of natural land cover, which may serve a buffering function, strengthens nutrient or sediment retention.

3.3 Materials and methods

3.3.1 Study sites

The decline in pulpwood demand in the pulp and paper industry released capacities of existing pine plantations for wood pellets in the southeastern US [36]. The 2008/09 recession and decline of the housing market released round wood from the timber market for solid bioenergy production [6]. A large share of up to 80 PJ (~440 Mt) of the produced pellets is expected to be exported to the EU by 2020 [7, 6]. The Big Satilla and Little Satilla watersheds, addressed as the Satilla watershed throughout the paper, are representative examples of such pine plantation production systems in a humid subtropical climate. The Satilla watershed includes an area of 8,760 km² (hereof: 28 % forest plantations in 2006, see Fig. 3.1) and is located in southeast Georgia, US.

For agricultural residues as an alternative feedstock option, the Mississippi Delta in humid subtropical western Mississippi, US is one of the major agricultural production areas in the subtropical southeastern US due to its alluvial fertile soils. The commodities include corn and wheat [37], with the area producing 68 % of the winter wheat and 79 % of the corn in Mississippi in 2013 [38]. A common practice for residues is to burn them completely onsite. Alternatively, a certain share of residues may be used for bioenergy without negatively affecting the nutrient and carbon balances [39]. The Big Sunflower watershed covers 8,170 km² (hereof: 80 % agricultural land (four percentage points corn and winter wheat production) in 2006), which represents most of the Mississippi Delta. The Big Sunflower River is a major river in the Yazoo River basin; the latter is a tributary of the Mississippi river.

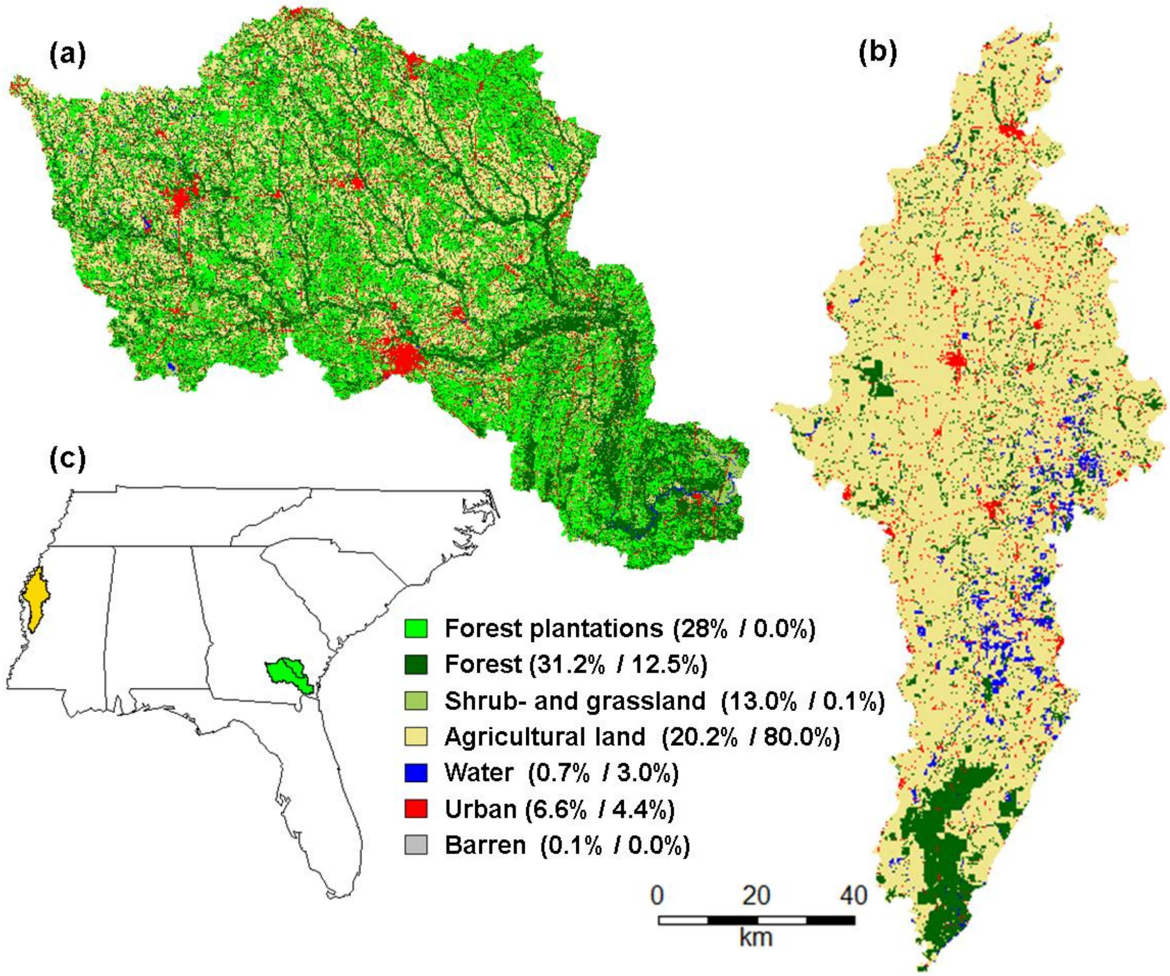

Fig. 3.1: Land use/land cover in the Satilla (a) and Big Sunflower (b) watersheds and their location in the southeastern US (c) [61, 62].

We model ESS supply for 2006, for which land cover information differentiating between natural and semi-natural forest and plantation forestry is available. Climatically, the precipitation in 2006 in the Big Sunflower watershed (1276.9 mm, SD: 31.0 mm) was 7 % lower than the normal climate conditions for the period from 1981 to 2010 [40]; the precipitation in the Satilla watershed (950.7 mm, SD: 63. 9 mm) was 22 % lower than the normal climate conditions. The minimum and maximum average temperatures in 2006 in the Big Sunflower watershed (Tmin: 11.9 °C, SD: 0.3 °C, Tmax: 24.2 °C, SD: 0.5 °C) and in the Satilla watershed (Tmin: 11.9 °C, SD: 0.3 °C, Tmax: 26.4 °C, SD: 0.2 °C) deviated less than 1 degree Celsius from the normal climate conditions for the period from 1981 to 2010 [40].

3.3.2 Ecosystem services

ESS are classified as provisioning, regulating and cultural ecosystem services [41]. We identified the following potentially affected provisioning and regulating ESS based on the existing literature on environmental impacts and ESS for bioenergy [42, 13, 43, 44, 27] and in general [45, 46, 18, 14, 25]: carbon storage, nutrient retention, sediment retention, and groundwater recharge. In addition, we assess the impact on biodiversity, which may be a recreational ESS in itself but largely supports the supply of other ESS [47, 48]. We focus on phosphorous (P) retention because (i) the nutrient retention efficiency is higher than that for nitrogen, (ii) agricultural sources are responsible for approximately 80 % of the P input in the Gulf of Mexico, and (iii) P has been underestimated in its contribution to the eutrophication of the Gulf of Mexico for the Big Sunflower watershed region [45].

Sustainable production of bioenergy will only be possible if a feedstock is available without major negative direct and indirect impacts on ESS and biodiversity. In contrast to the plantation forestry system with alternative uses of timber, e.g., as construction wood, corn stover and wheat straw have no competing use but are burnt onsite in the Mississippi Delta and are therefore unlikely to cause iLUC risks. Therefore, residue use for bioenergy may save GHG emissions, which are not covered in the ESS assessment. We calculate the amount of sustainably available agricultural residues, i.e., from corn and winter wheat, based on 2006 production data [38], on ranges of sustainable harvest residue removal rates [49, 50, 39], and average CO_2 and CH_4 emission factors for onsite burning practices for agricultural residues in the US [51].

3.3.2.1 Carbon storage

The amount of carbon stored was modeled with InVEST (Integrated Valuation of Environmental Services and Tradeoffs) [52, 53] by refining standard assumptions for aboveground [54-57], belowground [58, 14, 55, 56], soil [59, 54, 58, 60, 57] and dead organic carbon [59, 14, 57] for the land use/land cover data for 2006 [61, 62].

3.3.2.2 Phosphorous retention

The amount of retained P was modeled with InVEST. The model estimates P export and retention to surface water bodies based on the land use/land cover specific P input and retention capacity as well as the water yield. The major spatial inputs are the land use/land cover data for 2006 [61, 62], a digital elevation model (DEM) [63], annual precipitation data for 2006 [40], and the long-term annual average reference evapotranspiration [64] as well as the depth to any root restrictive layer and the available water holding capacity (AWC) from the Soil Survey Geographic (SSURGO) database [65]. We modified the InVEST default assumptions for evapotranspiration coefficients [66, 67], rooting depth [68-72], P export rates and P retention efficiencies [73-75]. We validated the modeled P export against the corresponding phosphorous concentration measurements of the stations Little Satilla near Offerman (USGS 02227500), Satilla River at Atkinson (USGS 2228000) and Big Sunflower River at Sunflower (USGS 07288500).

3.3.2.3 Sediment retention

The amount of retained sediment was modeled with InVEST. The model estimates the sediment retention and export based on the modeled soil loss from the universal soil loss equation (USLE) [76, 77], i.e., sheet erosion, and the land use/land cover specific sediment removal efficiencies. The same land use/land cover and DEM datasets used for modeling P retention were used. We obtained the k factor (soil erodibility) from the SSURGO database [65]. We calculated the r factor (rainfall erosivity) based on the following formula [78]:

$$R = (210 + 89 * log_{10} I_{30}) * I_{30} \tag{1}$$

where *I30* is the maximum rainfall intensity in 30 minutes obtained from [79]. We refined the required cover and management factor C [80-85, 77] and the support practice factor P [81, 82, 86, 85]. We validated the model outcome against the suspended sediment concentration for the same stations as for P export.

3.3.2.4 Groundwater recharge

The net infiltration was modeled with the soil-water balance model (SWB) from the USGS [87, 88]. We simulated the groundwater recharge on a daily basis with the Thornthwaite-Mater evapotranspiration calculation method. We used the same land use/land cover, DEM and AWC datasets as when modeling P retention. The hydrologic soil groups were obtained from the SSURGO database [65]. The daily average temperature and precipitation data were obtained for the Waycross 4 NE (USC00099186) (Satilla watershed) and Cleveland (USC00221738) (Big Sunflower watershed) stations [89]. We validated the results with a spatially explicit study on groundwater recharge, a modeled average for the period from 1951 to 1980, for the conterminous US with a resolution of one km [90] and with two other studies [91, 92] with more recent indicative ranges of groundwater recharge for larger regions including the targeted watersheds.

3.3.2.5 Biodiversity

A spatially explicit dataset for biodiversity was used to model terrestrial vertebrate species richness resulting from the GAP Analysis program from the USGS for Georgia [93] and Mississippi [94].

3.3.3 Tradeoff analysis

To identify tradeoffs in ESS supply, we distinguished the following major land use/land cover classes, as adapted from [95-97]:

1. Natural or semi-natural forest (counterfactual)
2. Plantation forestry (only the Satilla watershed)
3. Corn and winter wheat (only the Big Sunflower watershed)
4. Agricultural land (other)

We calculated the arithmetic mean ESS supply for these major land use/land cover classes and normalized them to the maximum value in each ESS category for each watershed. We conducted this analysis to assess the differences between targeted land use/land cover classes. In addition, the paired Pearson correlation coefficients between the ESS, selected from the list of methods in Mouchet et al. [98], were calculated for the entire watershed with the statistical software package R [99] to assess general ESS and biodiversity trade-offs in current production systems specialized in plantation forestry and agriculture respectively.

Because we considered two significantly different land use systems, it was reasonable to have a counterfactual, which served as a baseline to compare several alternatives of natural or semi-natural forest. This approach is recommended to test the suitability of bioenergy feedstock production options in the local hydrological context [100] or for biodiversity [101].

3.3.4 Indicators at the plot and landscape scale that potentially explain variation in ecosystem service supply

In this study, we tested indicators of landscape composition, configuration and naturalness for their influence on ESS supply in both watersheds (see Table 3.1). Therefore, we calculated the landscape composition and configuration indicators in a moving window approach for a buffer of 300 m, which was ten times the minimum pixel size [14]. At the plot scale, potential explanatory variables of topography and soil properties were used to set the explanatory value of landscape scale variables in the context of other groups of variables driving ESS supply (see Table 3.1). Landscape composition was defined as the quantity, and landscape configuration was defined as the relevant shape or form of different land use/land cover classes [89, 102]. Landscape naturalness was defined as the degree of human influence or impact on a natural system [103]. In addition to the selected explanatory variables used by others for landscape naturalness, we rated the land use intensity partly based on Brockerhoff et al. [34] as follows: urban (5), agricultural land (4), plantation forestry (3), open water (2) and primary or secondary natural vegetation (1). We calculated landscape metrics using Fragstats 4.1 [104] and R [99] based on the land use/land cover classes indicated in the following formula and the data source in Table 3.1. We modified the urbanity indicator from Wrbka et al. [103]:

$$Urbanity = log_{10}\left(\frac{U+A+P+1}{F+SG+W+B+1}\right) \tag{2}$$

where *U* is urban, *A* is agricultural land, *P* is plantation forestry, *F* is forestry, *SG* is shrub and grassland, *W* is open water, and *B* is barren land.

We applied a redundancy analysis (RDA) to identify explanatory values of plot and landscape factors on the variability of ESS supply in the selected case study regions. We chose the RDA because it allows us to (i) estimate the impact of the explanatory variable on ESS supply and vertebrate species richness simultaneously. A more complex alternative, machine learning methods, e.g., boosted regression trees, may only be applied to one response variable [98]. (ii) RDA allows to control for multicollinearity among explanatory variables [95, 105]. To consider nonlinear relationships between explanatory variables, we tested also the second degree terms of the potential explanatory variables as recommended by Borcard et al. [106]. We reduced the number of explanatory variables based on the permutation of p-values ($p<0.05$; 1000 permutations per step) as described by Blanchet et al. [107], which is used instead of the Akaike information criterion (AIC) to select explanatory variables. We used the former method as it delimits the type I error and provides reliable results for non-orthogonal and non-independent explanatory variables [107]. We partitioned the variation into the following groups: plot indicators (topography and soil properties), indicators on landscape composition, landscape configuration and naturalness. To test for spatial autocorrelation, we added the latitudinal and longitudinal coordinates and their interaction as an additional group [95].

Table 3.1: Potential variables explaining ESS supply.

Independent variable	Unit	Methodological reference (data source)
Landscape composition		([61, 62])
Shannon's diversity of land use/land cover	[score]	[128]
Largest patch index	[%]	[129, 105, 104]
Edge density	[m ha^{-1}]	[129, 125, 104, 102]
Share of land use types in the neighborhood:		
Forest	[%]	[97, 14]
Agricultural land	[%]	[97, 14]
Pine plantation share (only Satilla watershed)	[%]	
Corn and winter wheat (only Big Sunflower watershed)	[%]	
Wetlands	[%]	[97, 14]
Landscape configuration		([61, 62])
Connectance index	[%]	[104]
Effective mesh size	[ha]	[125, 104, 102, 103]
Landscape shape index	[score]	[129, 125, 104, 103]
Distance to stream	[m]	[14]
Topography		
Elevation	[m]	[103] ([63])
Slope	[%]	[97, 14, 103] ([130])
Curvature	[score]	[103] ([63])
Aspect	[°]	([131])
Soil parameters		[14] ([65])
Saturated hydraulic conductivity	[µm s^{-1}]	
Depth to water table	[mm]	
Available water holding capacity	[cm cm^{-1}]	
Silt content	[%]	
Soil erodibility	[Mg ha MJ^{-1} mm^{-1}]	
Naturalness		
Land use intensity	[score]	Based on Brockerhoff et al. [34] ([61, 62])
Urbanity	[score]	Modified from Wrbka et al. [103] ([61, 62])
Hemeroby index (human impact)	[score]	[125] ([132])

3.3.5 Sites of sufficient and insufficient ecosystem service supply

Villa et al. [108] argued that the benefits from ESS to society are particularly relevant if thresholds or target values, e.g., regarding drinking water quality or good ecological status, are closely met or exceeded. Therefore, we use these thresholds, if available, to distinguish sites of sufficient and insufficient ESS supply. If thresholds are set following representative stakeholder consultation, it may be assumed that they reflect the demand for regulating ESS. In the context of bioenergy, it has been shown that common tools for assessing environmental sustainability, i.e., certification schemes, largely miss such thresholds [12].

The thresholds must be identified at the impact scale of the beneficiaries of ESS, which is the global scale for carbon storage as a factor influencing the global climate. By contrast, the regional or watershed level is relevant for P and sediment retention and groundwater recharge [108]. Typically, P and sediment loading thresholds are set as the Total Maximum Daily Loadings for most of the surface water pollutant and are translated to land-based thresholds, e.g., sediment yields/soil erosion rates and P export rates, as indicated in Table 3.2. Because sediment loadings do not have thresholds in the Satilla watershed as a minor environmental concern, we did not include sediment export as an indicator for identifying sites of sufficient and insufficient ESS supply in the Satilla watershed.

Carbon storage of the current land use/land cover was compared with potential natural vegetation [35], following West et al. [109], i.e., sites of sufficient ESS supply are those with a gain in carbon storage for the current land use/land cover toward potential natural vegetation. Such a conservative classification of sites of sufficient and insufficient carbon storage should avoid that the conversion of naturally high carbon stocked land cover types, such as native forests, is viewed as beneficial.

In contrast to the investigated regulating ESS, P and sediment retention, groundwater recharge is an ESS that requires longer time scales to be generated. Therefore, land use activities may have longer lag phases before the consequences become apparent. Groundwater resources are declining due to human groundwater abstraction at both study sites [110-112]. Therefore, higher recharge rates are beneficial. Biodiversity may support other ESS or may be an ESS itself, as discussed in the materials and methods section. Biodiversity as a cultural ESS is highly subjective and strongly varies between stakeholder groups, i.e., among farmers, nature conservation activists, other citizens [113], species or species groups [114]. Facing these limitations, we use the arithmetic mean for both groundwater recharge and biodiversity as the indicative threshold between sufficient and insufficient supply, i.e., assuming that a higher supply is more beneficial.

To explain differences between sites of sufficient and insufficient ESS supply as defined in the materials and methods section, we followed Qiu and Turner [14] and set beneficial sites to one and non-beneficial sites to zero and applied a binomial logistic regression model. We conducted a binomial logistic regression in addition to the RDA since it (i) reflects the case of ESS supply and biodiversity supply relevant in practice, i.e., above and below a threshold or target value and (ii) allows to identify the direction of the impact, i.e., positive or negative on ESS supply. (iii) It is computationally more feasible for larger datasets as in this study than machine learning techniques, e.g., boosted regression trees, and (iv) commonly used in ESS research [98]. We focused on maximizing the bundle of relevant ESS instead of a single ESS. Maximizing bundles of ESS, particularly if regulating services are included, is more likely to ensure the stability of ESS supply, e.g., during sudden changes in environmental conditions. Maximized bundles may also avoid strong tradeoffs toward maximizing single ESS [115]. We removed non-significant explanatory ($p<0.05$)

variables in a backward stepwise manner based on the AIC. Next, variables with variance inflation factors >10 were removed to reduce multicollinearity. The significance of the final model was tested against a null model using a likelihood ratio test. We used the same indicators as those for the RDA to differentiate between sites of sufficient and insufficient ESS supply.

Table 3.2: Sustainability thresholds for P and sediment export set by environmental protection agencies in Georgia and Mississippi with public consultation.

Thresholds	Value	Unit	Data source
P export (Satilla Watershed)	917,627	[lbs a^{-1}]	[133]
	2.31	[kg ha^{-1} a^{-1}]	
P export (Big Sunflower watershed)	17,759.7	[lbs d^{-1}]	[134]
	7.56	[kg ha^{-1} a^{-1}]	
Sediment yield (Big Sunflower watershed)	0.6-1.6	[t km^{-2} d^{-1}]	[135]
	2.19	[Mg ha^{-1} a^{-1}]	

3.4 Results

3.4.1 Ecosystem service supply in the Satilla and Big Sunflower watersheds

Examining the plantation forestry system (Satilla watershed) (left) and the agricultural production system (Big Sunflower watershed) (right); c.f. Fig. 3.2 a, e, f, j, k, and o., we observed that carbon storage and vertebrate diversity were much higher in the Satilla watershed. In contrast, groundwater recharge and sediment retention were mostly higher in the Big Sunflower watershed; c.f. Fig. 3.2c, d, h, i, m, and n. P retention was only slightly higher but varied more in the Big Sunflower watershed.

If burning was avoided, the potential GHG emission would be reduced by up to 34,000 t for winter wheat and 130,000 t for corn (CO_2-equivalents (2006); see Table 3.3.

Table 3.3: Potential sustainable biomass availability (calculations based on [136, 49, 50, 137, 39, 38]) and emission savings in t CO_2 equivalent (emissions factors (CO_2 and CH_4, [51]) for the Mississippi Delta in 2006. Wheat and corn residues in the Mississippi Delta may contribute up to 0.4 % of the potentially available residues of 27 million t dry matter in the entire US in 2012 [39].

	Sustainable residue removal rates	Potentially available residues				Potential GHG emission savings (residue burning)	
		lower estimate		*upper estimate*		*lower estimate*	*upper estimate*
	[%]	DM [t]	HHV [GJ]	DM [t]	HHV [GJ]	CO_2 eq. [t]	CO_2 eq. [t]
Winter wheat	15-50	,900	110,000	20,000	400,000	34,000	20,000
Corn	40-50	0,000	2,000,000	100,000	2,000,000	130,000	110,000

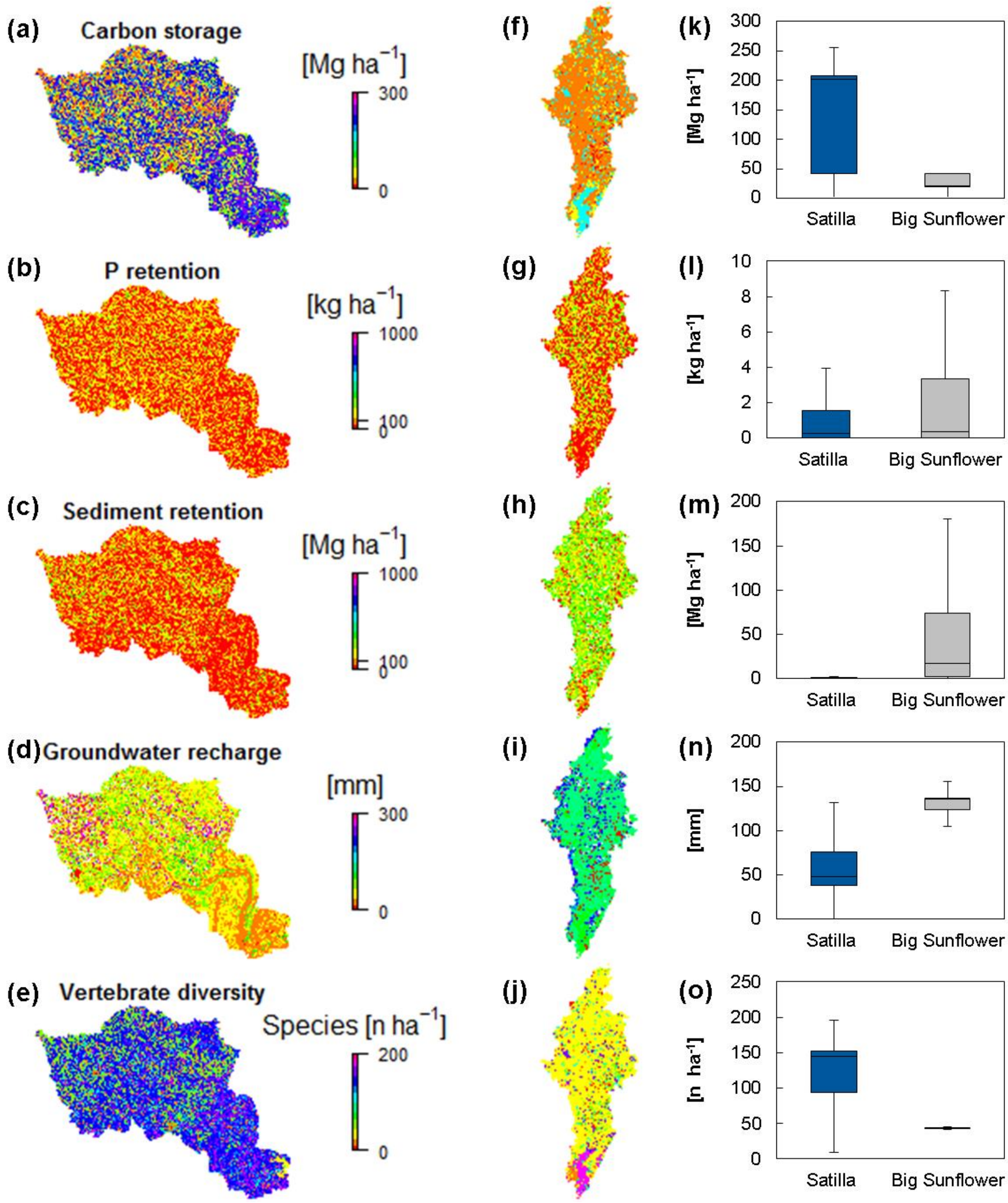

Fig. 3.2: Mapped ESS supply in the Satilla (a-e) and Big Sunflower (f-j) watersheds, also shown as boxplots (k-o). Mapped P and Sediment retention are plotted with breaks at 0.1, 1, 10 and 100 for better visualization.

The modeled annual P export rates explained approximately 90 % of the average P concentrations from the empirical data (Fig. 3.3a). The modeled sediment export rates explained between 78 and 90 % of the total suspended solid concentration (Fig. 3.3b). The modeled groundwater recharge rates for both watersheds were in the range of the existing studies, as shown in Fig. 3.3c, d.

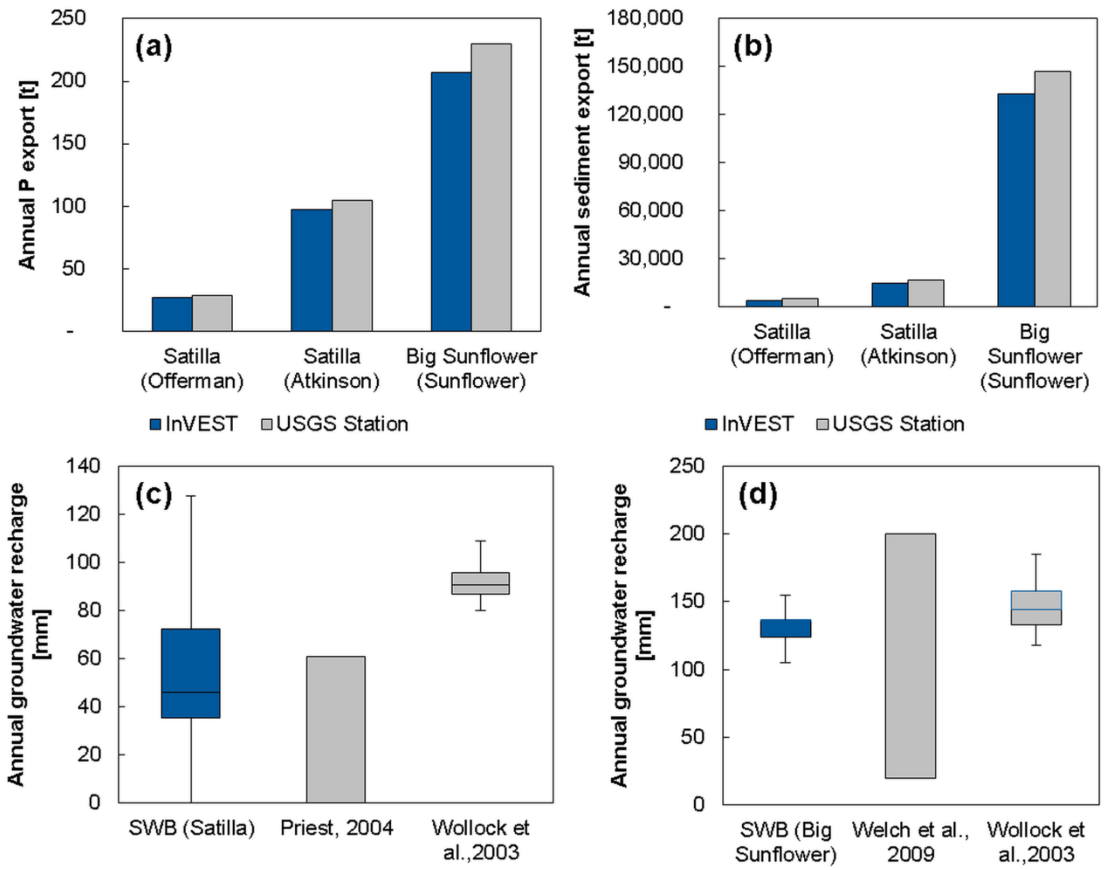

Fig. 3.3: Validation of modeled annual P (a) and sediment export (b) with measured water quality parameters, total P and total suspended solids, and annual groundwater recharge rates with existing studies (c-d). A Turkey boxplot is used for the groundwater recharge rates from an existing model [90] and a range is indicated for existing studies [91, 92]. The station and watershed names are listed in brackets.

3.4.2 Tradeoffs of ecosystem service supply

In the Satilla watershed, plantation forestry had a slightly lower mean carbon storage and vertebrate diversity than did natural- or semi-natural forests and the counterfactual; c.f. Fig. 3.4b, c. The plantation forestry had a higher vertebrate diversity and carbon storage than did the agricultural land and watershed average; c.f. Fig. 3.4a, d. By contrast, groundwater recharge was higher for plantation forestry than for forests. The P and sediment retention were negligible compared with agricultural land and the watershed average for both plantation forestry and forests. A paired correlation analysis for a sample of 10,000 pixels from all land use/land cover classes showed a high positive correlation between carbon storage and vertebrate diversity (Fig. 3.S1). A high negative correlation between groundwater recharge and both vertebrate diversity and carbon storage can be observed.

In the Big Sunflower watershed, corn and wheat production had a significantly lower mean carbon storage, vertebrate diversity and P retention than did forests (Fig. 3.4f, g). By contrast, the sediment retention and groundwater recharge were higher for corn and wheat production than for forests. A paired correlation analysis for a sample of 10,000 pixels from all land use/land cover classes showed a high positive correlation between carbon storage and vertebrate diversity (Fig. 3.S2). A lower negative correlation between groundwater recharge and both vertebrate diversity and carbon storage can be observed.

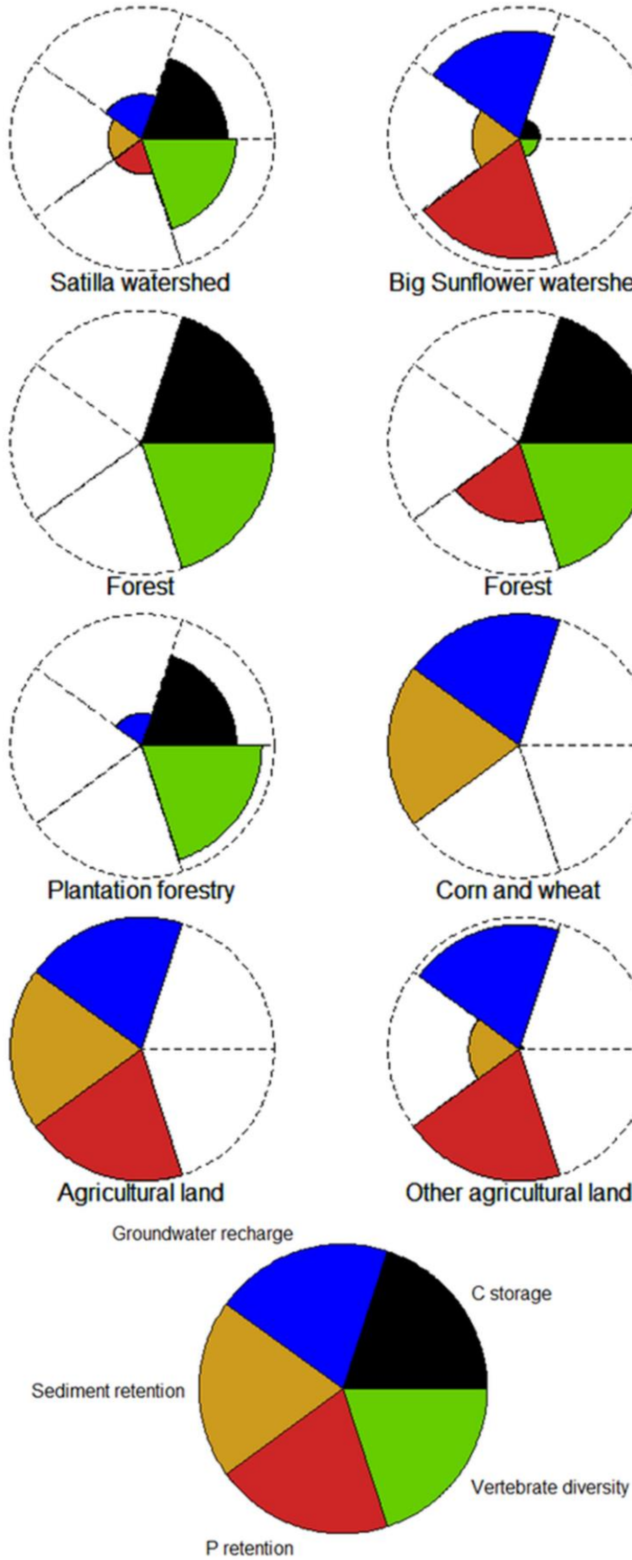

Fig. 3.4: ESS supply (arithmetic mean) for the entire watershed (a, e), natural or semi-natural forest as counterfactual (b, f) for plantation forestry (c) and corn and wheat production (g). The highest arithmetic mean value for each ESS category is used maximum to scale the radar charts for the Satilla (a-d) and Big Sunflower watersheds (e-h) separately.

3.4.3 The influence of topography and soil properties, landscape composition, configuration and naturalness on ecosystem service supply

The explained variation in ESS supply for both watersheds did not differ if the combined latitudinal and longitudinal information was included, c.f. Fig. 3.5c, d, or excluded, c.f. Fig. 3.5a, b.

The RDA without geographic information for the Satilla watershed (Fig. 3.5a) as a plantation forestry system showed that the landscape naturalness, i.e., the land use gradient and the hemeroby index (human impact), only explained 10 %, and if combined with other indicator groups, a further 20 % of the variation in the ESS supply was explained. Landscape composition only explained 4 %, and if combined with other indicator groups, 18 % of variation in the ESS supply was explained. The selected indicators are the largest patch index for agricultural land and shrub- and grassland. The landscape configuration explained less than 1 %, and if combined with other indicator groups, 10 % of the variation was explained. The selected indicators are the effective mesh size of shrub- and grassland and water bodies as well as the landscape shape index of forest, plantation forestry and agricultural land. The topography and soil factors explained 4 %, and if combined with other indicator groups, 14%.

The RDA without geographic information (Fig. 3.5b) for the Big Sunflower watershed as an agricultural system showed that the landscape naturalness, i.e., the land use gradient, explained 23 %, and if combined with other groups, a further 49 % of the variation in the ESS supply was explained. The landscape composition explained 2 %, and if combined with other groups, explained a further 48 %. The selected indicators are the largest patch index for agricultural land, urban land and water, the edge density for forest and for other land use/land cover categories as well as the Shannon’s diversity of land use/land cover. The landscape configuration and other indicator groups explained 52 % of the variation. The selected indicators are the effective mesh size for forest and the landscape shape index for agricultural land. Topography and soil factors were of minor importance.

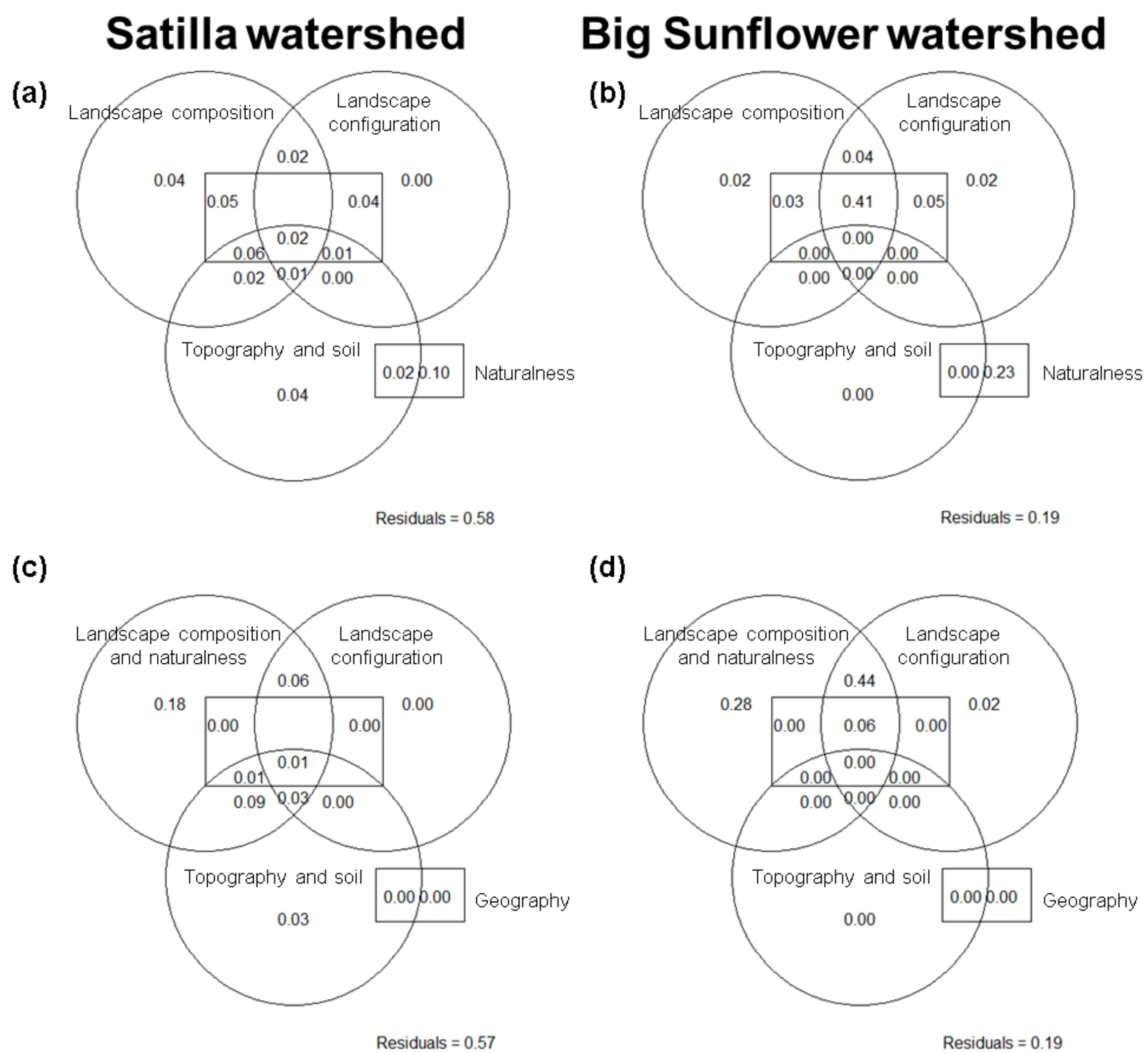

Fig. 3.5: Variation partitioning for ESS supply in the Satilla (a, c) and Big Sunflower (b, d) watersheds without geographic location (a-b) and with geographic location (c-d). p<0.01 and values <0 are not shown; the indicated values display the variance captured by the indicator groups on landscape composition, naturalness, landscape configuration, topography and soil parameters selected from Table 3.1; the indicators were selected based on permutation p-values; the variance is captured as adjusted R^2 for single (non-overlapping areas) and combined categories (overlapping areas).

3.4.4 Plot and landscape characteristics to distinguish sites of sufficient and insufficient ESS supply

In total, 0.2 % of the area of the Big Sunflower watershed had sufficient ESS supply, and 0.9 % had insufficient ESS supply. The results for the overall area are shown in the previous section. Corn and wheat production, i.e., 4 % of the area of the Big Sunflower watershed, accounted for 1 % of the sites of sufficient ESS supply and 5 % of the sites of insufficient ESS supply. The major landscape scale factors that promoted sufficient ESS supply were a higher effective mesh size of forests, a higher landscape shape index for agricultural land and a higher edge density of shrub, grassland and water bodies (Table 3.4). By contrast, a higher land use intensity, a higher edge density of forests and higher landscape shape indices for urban land promoted insufficient ESS supply at the landscape scale. A higher share of corn and wheat production slightly favored insufficient ESS supply. At the plot scale, the higher slope may be associated with insufficient ESS supply.

Table 3.4: Factors characterizing sufficient and insufficient ESS supply in the Big Sunflower watershed (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to sufficient ESS supply; a negative value for the standardized β indicates that an explanatory variable is contributing to insufficient ESS supply.

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	0.2938	0.4829	0.608	0.54288	
Topography and soil parameters					
Elevation [m]	-6.6488	0.2702	-24.605	< 0.0001	***
Slope [%]	-12.0045	0.3313	-36.235	< 0.0001	***
Silt content [%]	2.5642	0.2503	10.244	< 0.0001	***
Saturated hydraulic conductivity [$\mu m\ s^{-1}$]	2.5663	0.5208	4.928	< 0.0001	***
Soil erodibility	-0.6872	0.1632	-4.212	< 0.0001	***
Depth to water table [mm]	3.4742	0.2796	12.426	< 0.0001	***
Available water holding capacity [$cm\ cm^{-1}$]	-0.8073	0.2871	-2.812	0.00493	**
Naturalness					
Land use intensity	-6.0203	0.4517	-13.329	< 0.0001	***
Hemeroby	-0.975	0.2592	-3.761	0.00017	***
Landscape composition					
Corn/Wheat production, 300 m buffer [%]	-0.8912	0.3559	-2.504	0.01228	*
Edge density (forest)	-1.7479	0.3269	-5.347	< 0.0001	***
Edge density (shrub- and grassland)	10.4326	0.8355	12.487	< 0.0001	***
Edge density (open water)	5.0185	0.6683	7.509	< 0.0001	***
Largest patch index (urban)	1.3618	0.3367	4.045	< 0.0001	***
Landscape configuration					
Effective mesh size (forest)	10.904	1.2152	8.973	< 0.0001	***
Landscape shape index (agricultural land)	11.4842	0.6129	18.737	< 0.0001	***
Landscape shape index (open water)	3.9223	0.3983	9.848	< 0.0001	***
Landscape shape index (urban)	-2.6192	0.2023	-12.947	< 0.0001	***

In total, 0.1 % of the area of the Satilla watershed showed sufficient ESS supply, and 0.6 % showed insufficient ESS supply. The plantation forestry, i.e., 28 % of the area of the Satilla watershed in 2006, accounted for 19 % of the sites of sufficient ESS supply and for 1.6 % of the sites of insufficient ESS supply. The major landscape scale factors that were favorable for a sufficient ESS supply were a higher edge density as well as a high largest patch index of forest and plantation forestry (Table 3.S1). At the plot scale, a higher available water holding capacity and a higher depth to water table were beneficial. In contrast, a higher landscape shape index of forests, a greater land use intensity, a higher share of wetlands and a higher largest patch index of agricultural land promoted insufficient ESS supply at the landscape scale.

The likelihood ratio tests showed a significant difference when compared to a null model for the Big Sunflower watershed (χ^2=20440, df=18, p<2.2e-16) and the Satilla (χ^2=35944, df=22, p<2.2e-16) (the set of explanatory variables shown in Table 3.4 and Table 3.S1 had a significant explanatory value when compared against the model without explanatory variables).

3.5 Discussion

3.5.1 Ecosystem service supply synergies and tradeoffs

The higher carbon storage and biodiversity in the Satilla watershed may be related to the higher resemblance of plantation forestry to natural or semi-natural forest, see [34]. Higher P and sediment retention in the Big Sunflower watershed are attributable to the higher P application and bare soil in agricultural production systems, resulting in a higher demand for related regulating services, see Fig. 3.3a, b. The higher rate of groundwater decline in the Big Sunflower watershed [111] hints at a stronger mismatch of groundwater abstraction and recharge compared with the Satilla watershed.

Generally, tradeoffs for agricultural production systems have been shown in other studies, e.g., [116, 117, 14, 15], but not for plantation forestry or for the intensive feedstock production systems evaluated in this study. Overall, the plantation forestry deviates less from the counterfactual natural or semi-natural forest than from agricultural production systems such as corn and wheat; thus, we conclude that the plantation forestry system is preferable for the modeled ESS bundle and biodiversity.

3.5.2 Advantages and constraints of the ecosystem service modeling scheme

We achieved the models' purpose of providing a broad picture of water quality [118] and of assessing annual groundwater recharge rates [87]. For such application, we achieved a reasonably good validity (Fig. 3.3); our results are comparable with those of Qiu and Turner [14] and Terrado et al. [119]. In addition, simple tools such as InVEST are more likely to be an option for practitioners for regional scale assessments [118], e.g., for bioenergy, agricultural or forestry production systems, compared to significantly more complex and resource-intensive models such as the Soil-Water Assessment Tool (SWAT) [120]. For example, InVEST may be used in data scarce situations without monthly or daily precipitation data, the latter required by SWAT for nutrient and sediment modeling. Therefore, this study design is suitable for regions with high or low data availability, which will facilitate comparable analyses for bioenergy production systems around the world to compare ESS supply and biodiversity, e.g., within this study. However, tools such as InVEST or SWB do not support simulation studies requiring numerous runs to find Pareto optimal solutions of ESS supply, e.g., [121], or biodiversity, e.g., [122].

Considering only the higher biomass yield of pine plantations and the lower yields from corn stover and cereal straw [39], the smaller tradeoffs of pine plantations may be seen as preferable than agricultural biomass at first glance. Bennett and Balvanera [16] similarly argued that the tradeoffs of provisioning ESS should be minimized to ensure a balanced ESS supply. Focusing on the yield, i.e., for bioenergy, and the environmental side may disregard the social side of the "food, energy and environment trilemma" [21]. This trilemma may be solved if (i) there are no competing uses of wheat straw or corn stover [39, 21] and (ii) if both corn stover and wheat straw are produced regardless; the identified ESS tradeoffs with forests do not exclusively need to be attributed to a potentially used residual biomass.

3.5.3 Benefits of avoided and potential further competition for biomass

Due to the missing competing uses for the agricultural residues in the Mississippi Delta, their potential use for bioenergy instead of burning may create synergies. For example, (i) negative impacts on soil structure, local microbiology, the water holding capacity of soil and soil fertility from burning could be avoided [123, 124], and (ii) we could reduce GHG emissions or at least generate energy as an additional use. However, the actual sustainable residue removal rates and GHG emissions depend on the local environmental conditions, e.g., soil organic carbon or water availability and conversion and use aspects, e.g., the harvesting technique, transportation and the type of energy carrier. Therefore, further studies may integrate results from this study into life-cycle assessments to compare these iLUC free bioenergy feedstocks with a fossil fuel energy carrier. For the plantation forestry system, future research may investigate (i) the global production pattern changes due to the decline of the pulp and paper industry in the southeastern US to identify potential indirect impacts, such as iLUC, and (ii) the ESS supply and tradeoffs of current and future pulp and paper exporting regions of the world that partly substitute production capacities in the southeastern US and that may be of interest; e.g., Eucalyptus plantations in Brazil or some parts of sub-Saharan Africa may reveal other economic, social or environmental issues.

3.5.4 Factors explaining the variance of ecosystem service supply

We confirmed the hypothesis of Frank et al. [125] that the degree of naturalness of the landscape affects ESS supply. Wrbka et al. [103] previously demonstrated this relation for the human appropriation of net primary productivity, e.g., biomass use. The impact of landscape composition and configuration on ESS supply was only slightly less important, as proposed by Syrbe and Walz [102] and Frank et al. [125], e.g., the largest patch index, the effective mesh size or the landscape shape index in both watersheds. For example, in the Satilla watershed, the substantial influence of the largest patch index of agricultural land and of the effective mesh size of shrub and grassland habitats have shown that not only the existence but also the location and connection of shrub and grassland patches are important for ESS supply. Further influential factors related to the climate are not considered in this study because both case studies are in the same humid subtropical climate zone according to Koppen-Geiger [126]; however, this may be reasonable for case studies with climatic gradients, e.g., [95], or when comparing case studies in different climate zones, e.g., a humid subtropical climate compared with a semi-arid tropical climate.

3.5.5 Thresholds and local demand for ecosystem services

To link ESS supply to the local demand, we related the actual supply to locally set environmental thresholds when available for sediment and P concentrations in surface waters, as in Terrado et al. [119]. Considering local preferences may more likely allow us to determine whether ESS provide a human benefit. However, this consideration is not regularly performed in ESS modeling, see, e.g., [3, 15]. Such socially accepted environmental thresholds for groundwater recharge or biodiversity.

When comparing the sites above and below thresholds, the plantation forestry contributed to sufficient ESS supply. In the Big Sunflower watershed, ESS supply may be enhanced by a higher complexity of agricultural patches. It simultaneously requires combining agricultural land with, e.g., shrub and grassland or forest patches rather than urban land, as shown by the negative impact of increasing land use intensity. This is supported by the beneficial effect of an increase in size and number of shrub and grassland patches. It is indirectly shown by the beneficial effect of a higher edge density and a higher effective mesh size of forest patches. In the Satilla watershed, it may be beneficial to increase the size and connectivity of forest patches. By contrast, it does not seem beneficial to enhance the complexity of forest patches. This may be explained by the fact that more complex forest patches have a larger share of non-forest land use, e.g., pine plantations, agricultural or urban land. These results are in line with the results of the tradeoff analysis showing the higher supply of carbon storage and biodiversity toward plantation forestry. However, a higher dominance of agricultural land decreases ESS supply, whereas plantation forestry still increases ESS supply. Future research should assess the relevance of landscape composition and configuration and naturalness of the landscape in other solid biomass production systems in other parts of the world.

In practice, the rules of certification schemes or the rules set by local authorities in landscape planning should include rules on landscape composition and configuration. It may be reasonable to consider an additional assessment of ESS supply at the landscape scale in improved certification schemes, e.g., for bioenergy. An assessment at the plot scale of the individual feedstock producer seems incomplete. Even if the assessment would go beyond the management practices, the local environmental factors, topography and soil parameters explained only a small share of the variation in ESS supply. For example, preserving or creating a landscape mosaic may better balance the supply of ESS beyond the bioenergy feedstock in the production region. One concrete option could be a higher number of connected forest patches as buffer strips alongside rivers in the Big Sunflower watershed or other natural vegetation, such as grassland in agricultural watersheds [30]. For the Satilla river, we may see forested buffer strips in Fig. 3.1, which are required in Georgia’s Mountain and River Corridor Protection Act [127]. Another strategy to reduce the intensity of agricultural production could be 2nd generation bioenergy feedstocks, such as perennial bioenergy grasses or short rotation coppice species. Such bioenergy-providing species may provide both biomass and higher ESS (sediment and nutrient retention) and biodiversity, c.f. [30, 4].

3.5.6 Potential future use of indicators on landscape structure

We suggest including indicators of landscape structure to ensure a harmonized level of sustainability in certification schemes, which is particularly relevant if the biomass is largely traded, e.g., wood pellets in the form of pine plantations in the southeast US; they should also be subjected to legal frameworks, which are currently absent or variable, for landscape planning. For a broader application in sustainability assessments beyond bioenergy, the impact of landscape structure on ESS supply should be tested in other agricultural or forest production systems.

3.6 Acknowledgments

The authors would like to thank Charlotte Main for partial literature review and data preparation. The authors thank Ralf Seppelt, Daniel Doktor, and the reviewers for helpful comments on earlier drafts of this manuscript.

3.7 References

[1] MEA Board. Ecosystems and human well-being : current state and trends : findings of the condition and trends working group. Hassan R, Scholes R, Ash N, editors. Washington, DC: Island Press; 2005.

[2] Gasparatos A, Stromberg P, Takeuchi K. Biofuels, ecosystem services and human wellbeing: Putting biofuels in the ecosystem services narrative. Agriculture, Ecosystems & Environment 2011;142(3–4):111-28.

[3] Meehan TD, Gratton C, Diehl E, Hunt ND, Mooney DF, Ventura SJ, et al. Ecosystem-Service Tradeoffs Associated with Switching from Annual to Perennial Energy Crops in Riparian Zones of the US Midwest. PLoS ONE 2013;8(11):e80093.

[4] Werling BP, Dickson TL, Isaacs R, Gaines H, Gratton C, Gross KL, et al. Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. Proceedings of the National Academy of Sciences 2014.

[5] Chum H, Faaij A, Moreira J, Berndes G, Dhamija P, Dong H, et al. Bioenergy. In: Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, et al., editors. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press; 2011.

[6] Lamers P, Junginger M, Hamelinck C, Faaij A. Developments in international solid biofuel trade—An analysis of volumes, policies, and market factors. Renewable and Sustainable Energy Reviews 2012;16(5):3176-99.

[7] Goh CS, Junginger M, Cocchi M, Marchal D, Thrän D, Hennig C, et al. Wood pellet market and trade: a global perspective. Biofuels, Bioproducts and Biorefining 2013;7(1):24-42.

[8] Diaz-Chavez R, Berndes G, Neary D, Elia Neto A, Fall M. Water quality assessment of bioenergy production. Biofuels, Bioproducts and Biorefining 2011;5(4):445-63.

[9] Meyfroidt P, Rudel TK, Lambin EF. Forest transitions, trade, and the global displacement of land use. Proceedings of the National Academy of Sciences 2010;107(49):20917-22.

[10] Brockerhoff EG, Jactel H, Parrotta JA, Ferraz SFB. Role of eucalypt and other planted forests in biodiversity conservation and the provision of biodiversity-related ecosystem services. Forest Ecology and Management 2013;301(0):43-50.

[11] Bringezu S, O'Brien M, Schütz H. Beyond biofuels: Assessing global land use for domestic consumption of biomass: A conceptual and empirical contribution to sustainable management of global resources. Land Use Policy 2012;29(1):224-32.

[12] Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65:151-69.

[13] Lamers P, Thiffault E, Paré D, Junginger M. Feedstock specific environmental risk levels related to biomass extraction for energy from boreal and temperate forests. Biomass and Bioenergy 2013;55(0):212-26.

[14] Qiu J, Turner MG. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proceedings of the National Academy of Sciences 2013;110(29):12149-54.

[15] Raudsepp-Hearne C, Peterson GD, Bennett EM. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences 2010;107(11):5242-7.

[16] Bennett EM, Balvanera P. The future of production systems in a globalized world. Frontiers in Ecology and the Environment 2007;5(4):191-8.

[17] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global Consequences of Land Use. Science 2005;309(5734):570-4.

[18] Kremen C, Miles A. Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs. Ecology and Society 2012;17(4).

[19] Junginger M, van Dam J, Zarrilli S, Ali Mohamed F, Marchal D, Faaij A. Opportunities and barriers for international bioenergy trade. Energy Policy 2011;39(4):2028-42.
[20] Muth Jr DJ, Bryden KM, Nelson RG. Sustainable agricultural residue removal for bioenergy: A spatially comprehensive US national assessment. Applied Energy 2013;102(0):403-17.
[21] Tilman D, Socolow R, Foley JA, Hill J, Larson E, Lynd L, et al. Beneficial Biofuels—The Food, Energy, and Environment Trilemma. Science 2009;325(5938):270-1.
[22] Fonseca MB, Burrell A, Gay H, Henseler M, Kavallari A, M'Barek R, et al. Impacts of the EU biofuel target on agricultural markets and land use: a comparative modelling assessment. In: Burrell A, editor. JRC Scientifc and Technical Reports. Luxembourg: JRC-Institute for Prospective Technological Studies; 2010.
[23] Goldman RL, Thompson BH, Daily GC. Institutional incentives for managing the landscape: Inducing cooperation for the production of ecosystem services. Ecological Economics 2007;64(2):333-43.
[24] Tscharntke T, Klein AM, Kruess A, Steffan-Dewenter I, Thies C. Landscape perspectives on agricultural intensification and biodiversity – ecosystem service management. Ecology Letters 2005;8(8):857-74.
[25] Zhang W, Ricketts TH, Kremen C, Carney K, Swinton SM. Ecosystem services and dis-services to agriculture. Ecological Economics 2007;64(2):253-60.
[26] Lewandowski I, Faaij APC. Steps towards the development of a certification system for sustainable bio-energy trade. Biomass and Bioenergy 2006;30(2):83-104.
[27] van Dam J, Junginger M, Faaij APC. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renewable and Sustainable Energy Reviews 2010;14(9):2445-72.
[28] Tscharntke T, Milder JC, Schroth G, Clough Y, DeClerck F, Waldron A, et al. Conserving biodiversity through certification of tropical agroforestry crops at local and landscape scales. Conservation Letters 2015;8(1):14-23.
[29] Ghazoul J, Garcia C, Kushalappa CG. Landscape labelling: A concept for next-generation payment for ecosystem service schemes. Forest Ecology and Management 2009;258(9):1889-95.
[30] Manning P, Taylor G, E. Hanley M. Bioenergy, Food Production and Biodiversity – An Unlikely Alliance? GCB Bioenergy 2015;7(4):570–6.
[31] Mitchell ME, Bennett E, Gonzalez A. Linking Landscape Connectivity and Ecosystem Service Provision: Current Knowledge and Research Gaps. Ecosystems 2013;16(5):894-908.
[32] Frank S, Fürst C, Witt A, Koschke L, Makeschin F. Making use of the ecosystem services concept in regional planning—trade-offs from reducing water erosion. Landscape Ecology 2014:1-15.
[33] Bourke D, Stanley D, O'Rourke E, Thompson R, Carnus T, Dauber J, et al. Response of farmland biodiversity to the introduction of bioenergy crops: effects of local factors and surrounding landscape context. GCB Bioenergy 2014;6(3):275-89.
[34] Brockerhoff E, Jactel H, Parrotta J, Quine C, Sayer J. Plantation forests and biodiversity: oxymoron or opportunity? Biodivers Conserv 2008;17(5):925-51.
[35] USGS and USDOI. Landfire 1.2.0 - Environmental Site Potential layer. Wahington, DC2010 [cited 2014 January 15]; Available from: http://landfire.cr.usgs.gov/viewer/.
[36] Susaeta A, Lal P, Alavalapati J, Carter DR. Modelling the impacts of bioenergy markets on the forest industry in the southern United States. International Journal of Sustainable Energy 2013;32(6):544-61.
[37] Snipes CE, Nichols SP, Poston DH, Walker TW, Evans LP, Robinson HR. Current Agricultural practices of the Mississippi Delta. Prairie: Mississippi Agricultural and Forestry Experiment Station; 2005.
[38] USDA NASS. Corn and winter wheat production in bushels for Mississippi and the Upper Delta and the Lower Delta 2013. Washington, DC2014 [cited 2014 May 20]; Available from: http://quickstats.nass.usda.gov.

[39] Perlack RD, Stokes B, Downing M, Eaton LM, Graham RL, Langholtz MH, et al. U.S. Billion-Ton Update: Biomass Supply for a Bioenergy and Bioproducts Industry. Oak Ridge: Oak Ridge National Laboratory; 2011.

[40] Daly C, Halbleib M, Smith JI, Gibson WP, Doggett MK, Taylor GH, et al. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology 2008;28(15):2031-64.

[41] CICES. Common International Classification of Ecosystem Services. 2013 [cited 2013 January 15]; Available from: http://cices.eu/.

[42] Efroymson R, Dale V, Kline K, McBride A, Bielicki J, Smith R, et al. Environmental Indicators of Biofuel Sustainability: What About Context? Environmental Management 2013;51(2):291-306.

[43] McBride AC, Dale VH, Baskaran LM, Downing ME, Eaton LM, Efroymson RA, et al. Indicators to support environmental sustainability of bioenergy systems. Ecological Indicators 2011;11(5):1277-89.

[44] Scarlat N, Dallemand J-F. Recent developments of biofuels/bioenergy sustainability certification: A global overview. Energy Policy 2011;39(3):1630-46.

[45] Alexander RB, Smith RA, Schwarz GE, Boyer EW, Nolan JV, Brakebill JW. Differences in Phosphorus and Nitrogen Delivery to The Gulf of Mexico from the Mississippi River Basin. Environmental Science & Technology 2008;42(3):822-30.

[46] Dale VH, Polasky S. Measures of the effects of agricultural practices on ecosystem services. Ecological Economics 2007;64(2):286-96.

[47] Cardinale BJ, Duffy JE, Gonzalez A, Hooper DU, Perrings C, Venail P, et al. Biodiversity loss and its impact on humanity. Nature 2012;486(7401):59-67.

[48] Maes J, Paracchini ML, Zulian G, Dunbar MB, Alkemade R. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biological Conservation 2012;155(0):1-12.

[49] Graham RL, Nelson R, Sheehan J, Perlack RD, Wright LL. Current and Potential U.S. Corn Stover Supplies. Agron J 2007;99(1):1-11.

[50] Kadam KL, McMillan JD. Availability of corn stover as a sustainable feedstock for bioethanol production. Bioresource Technology 2003;88(1):17-25.

[51] McCarty JL. Remote Sensing-Based Estimates of Annual and Seasonal Emissions from Crop Residue Burning in the Contiguous United States. Journal of the Air & Waste Management Association 2011;61(1):22-34.

[52] Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford: Oxford University Press; 2011.

[53] Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron D, et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 2009;7(1):4-11.

[54] Bridgham S, Megonigal JP, Keller J, Bliss N, Trettin C. The carbon balance of North American wetlands. Wetlands 2006;26(4):889-916.

[55] Ruesch A, Gibbs H. New IPCC Tier-1 Global Biomass Carbon Map for the Year 2000. Oak Ridge: Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory; 2008 [cited 2014 January 14]; Available from: http://cdiac.ornl.gov/epubs/ndp/global_carbon/carbon_documentation.html.

[56] Sainju UM, Whitehead WF, Singh BP. Carbon accumulation in cotton, sorghum, and underlying soil as influenced by tillage, cover crops, and nitrogen fertilization. Plant Soil 2005;273(1-2):219-34.

[57] Smith JE, Heath LS, Skog KE, Birdsey RA. Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. General Technical Report NE. Delaware: Northern Research Station, Forest Service; 2006.

[58] Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Paris; 2006.
[59] Birdsey RA. Carbon Storage and Accumulation in United States Forest Ecosystems. General Technical Report WO. Radnor: Northeastern Forest Experiment Station, Forest Service; 1992.
[60] Franzluebbers AJ. Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil and Tillage Research 2005;83(1):120-47.
[61] USDA NASS. Cropland Data Layer (CDL). Fairfax2010 [cited 2013 November 20]; Available from: www.nass.usda.gov/research/Cropland/SARS1a.htm.
[62] USGS GAP. National Land Cover, Version 2. 2011 [cited 2013 November 25]; Available from: http://gapanalysis.usgs.gov/gaplandcover/data/.
[63] Lehner B, Verdin K, Jarvis A. New Global Hydrography Derived From Spaceborne Elevation Data. Eos, Transactions American Geophysical Union 2008;89(10):93-4.
[64] FAO Geonetwork. Global map of monthly reference evapotranspiration - 10 arc minutes (GeoLayer). Rome, Italy2004 [cited 2014 January 20]; Available from: http://www.fao.org/geonetwork/srv/en/main.home.
[65] USDA NRCS. Web Soil Survey - Soil Survey Geographic (SSURGO) Database. 2014 [cited 2014 January 20]; Available from: http://websoilsurvey.nrcs.usda.gov/.
[66] Allen RG, Pereira LS, Raes D, Smith M. Crop evapotranspiration - Guidelines for computing crop water requirements. FAO Irrigation and drainage paper. Rome; 2001.
[67] Scurlock JMO, Asner GP, Gower ST. Worldwide Historical Estimates of Leaf Area Index, 1932-2000. Oak Ridge Oak Ridge National Laboratory; 2001.
[68] Burns LA, Allen BW, Barber MC, Bird SL, Cheplick JM, Fendley MJ, et al. Pesticide and Industrial Chemical Risk Analysis and Hazard Assessment, Version 3.0. Athens: Environmental Research Laboratory, Office of Research and Development; 1992.
[69] Canadell J, Jackson RB, Ehleringer JB, Mooney HA, Sala OE, Schulze ED. Maximum rooting depth of vegetation types at the global scale. Oecologia 1996;108(4):583-95.
[70] Mishra HS, Rathore TR, Pant RC. Root growth, water potential, and yield of irrigated rice. Irrig Sci 1997;17(2):69-75.
[71] Schenk HJ, Jackson RB. The Global Biogeography of Roots. Ecological Monographs 2002;72(3):311-28.
[72] USDA NRCS. Engineering Field Handbook. Washington, DC: U.S. Department of Agriculture; 2012.
[73] Campbell Group. Juvenile Pine Plantation Fertilization recommendations. Waycross: Georgia Biomass; 2013.
[74] Harmel D, Potter S, Casebolt P, Reckhow K, Green C, Haney R. Compilaton of measured nutrient load data for agricultural land uses in the United States. JAWRA Journal of the American Water Resources Association 2006;42(5):1163-78.
[75] Reckhow KH, Beaulac M, Simpson J. Modeling Phosphorous Loading and Lake Response under Uncertainty: A Manual and Compilation of Export Coefficients. Washington, DC; 1980.
[76] Tallis HT, Ricketts T, Guerry AD, Wood SA, Sharp R, Nelson E, et al. InVEST 2.5.6 User’s Guide. Stanford; 2013.
[77] Wischmeier WH, Smith DD. Predicting rainfall erosion losses: A guide to conservation planning. Washington, DC; 1978.
[78] Roose E. Land husbandry - Components and strategy. FAO Soils Bulletin. Rome; 1996.
[79] Perica S, Martin D, Pavlovic S, Roy I, St. Laurent M, Trypaluk C, et al. Precipitation-Frequency Atlas of the United States. NOAA Atlas 14. Silver Spring: National Weather Service; 2013.
[80] Dissmeyer GE, Foster GR. A guide for predicting sheet and rill erosion on forest land. Technical publication. Atlanta: Forest Service, Southern Region; 1984.
[81] Fernandez C, Wu JQ, McCool DK, Stöckle CO. Estimating water erosion and sediment yield with GIS, RUSLE, and SEDD. Journal of Soil and Water Conservation 2003;58(3):128-36.

[82] Martin A, Gunter J, Regens J. Estimating erosion in a riverine watershed bayou liberty-tchefuncta river in louisiana. Environ Sci & Pollut Res 2003;10(4):245-50.
[83] Morgan R. Soil erosion and conservation. 3rd ed. Oxford: Wiley-Blackwell; 2005.
[84] USDA NRCS. C Factor used for erosion prediction in Michigan. Michigan State University; 2002 [cited 2014 January 20]; Available from: http://www.iwr.msu.edu/rusle/doc/cfactors.pdf.
[85] USDA NRCS and ARS. Revised Universal Soil Loss Equation (RUSLE) EPA Pesticide Project. National Resources Conservation Service and Agricultural Research Service 2000.
[86] Suárez LA. PRZM-3, A Model for Predicting Pesticide and Nitrogen Fate in the Crop Root and Unsaturated Soil Zones: Users Manual for Release 3.12.2. Washington, DC: Ecosystems Research Division, National Exposure Research Laboratory; 2005.
[87] Dripps WR, Bradbury KR. A simple daily soil–water balance model for estimating the spatial and temporal distribution of groundwater recharge in temperate humid areas. Hydrogeol J 2007;15(3):433-44.
[88] Westenbroek SM, Kelson VA, Dripps WR, Hunt RJ, Bradbury KR. SWB - A modified Thornthwaite-Mater Soil-Water-Balance code for estimating groundwater recharge. Techniques and Methods 6-A31. Reston; 2010.
[89] Menne MJ, Durre I, Vose RS, Gleason BE, Houston TG. An Overview of the Global Historical Climatology Network-Daily Database. Journal of Atmospheric and Oceanic Technology 2012;29(7):897-910.
[90] Wolock DM. Estimated mean annual natural ground-water recharge in the conterminous United States. Reston2003 [cited 2014 March 25]; Available from: http://water.usgs.gov/lookup/getspatial?rech48grd.
[91] Priest S. Evaluation of ground-water contribution to streamflow in coastal Georgia and adjacent parts of Florida and South Carolina. U.S. Geological Survey Scientific Investigations Report 2004. Reston; 2004.
[92] Welch H, Green C, Coupe R. The fate and transport of nitrate in shallow groundwater in northwestern Mississippi, USA. Hydrogeol J 2011;19(6):1239-52.
[93] Kramer E, Conroy MJ, Elliott MJ, Anderson EA, Bumback WR, Epstein J. A GAP Analysis of Georgia. Athens: Institute of Ecology and Georgia Cooperative Fish & Wildlife Research Unit, University of Georgia; 2003.
[94] Vilella FJ, Minnis RB, Elmore RD, Bowman JL, Evans DL, McCombs J. A GAP Analysis of Mississippi. Starkville: Mississippi Cooperative Fish and Wildlife Research Unit, Mississippi State University; 2003.
[95] Brandt P, Abson DJ, DellaSala DA, Feller R, von Wehrden H. Multifunctionality and biodiversity: Ecosystem services in temperate rainforests of the Pacific Northwest, USA. Biological Conservation 2014;169(0):362-71.
[96] Dauber J, Hirsch M, Simmering D, Waldhardt R, Otte A, Wolters V. Landscape structure as an indicator of biodiversity: matrix effects on species richness. Agriculture, Ecosystems & Environment 2003;98(1–3):321-9.
[97] Jones KB, Neale A, Nash M, Van Remortel R, Wickham J, Riitters K, et al. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 2001;16(4):301-12.
[98] Mouchet MA, Lamarque P, Martín-López B, Crouzat E, Gos P, Byczek C, et al. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change 2014;28(0):298-308.
[99] R Development Core Team. R: A language and environment for statistical computing. 2009 [cited 2013 November 05]; Available from: http://www.r-project.org/.
[100] Yeh S, Berndes G, Mishra GS, Wani SP, Elia Neto A, Suh S, et al. Evaluation of water use for bioenergy at different scales. Biofuels, Bioproducts and Biorefining 2011;5(4):361-74.
[101] Immerzeel DJ, Verweij PA, van der Hilst F, Faaij APC. Biodiversity impacts of bioenergy crop production: a state-of-the-art review. GCB Bioenergy 2014;6(3):183-209.

[102] Syrbe R-U, Walz U. Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. Ecological Indicators 2012;21(0):80-8.
[103] Wrbka T, Erb K-H, Schulz NB, Peterseil J, Hahn C, Haberl H. Linking pattern and process in cultural landscapes. An empirical study based on spatially explicit indicators. Land Use Policy 2004;21(3):289-306.
[104] McGarigal K, Cushman SA, Ene E. FRAGSTATS v4: Spatial Pattern Analysis Program for Categorical and Continuous Maps. Amherst: University of Massachusetts; 2012 [cited 2014 April 01]; Available from: http://www.umass.edu/landeco/research/fragstats/fragstats.html.
[105] Heikkinen RK, Luoto M, Virkkala R, Rainio K. Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in an agricultural–forest mosaic. Journal of Applied Ecology 2004;41(5):824-35.
[106] Borcard D, Gillet F, Legendre P. Numerical ecology with R. 1st ed. New York Springer; 2011.
[107] Blanchet FG, Legendre P, Borcard D. Forward selection of explanatory variables. Ecology 2008;89(9):2623-32.
[108] Villa F, Voigt B, Erickson JD. New perspectives in ecosystem services science as instruments to understand environmental securities. Philosophical Transactions of the Royal Society B: Biological Sciences 2014;369(1639).
[109] West PC, Gibbs HK, Monfreda C, Wagner J, Barford CC, Carpenter SR, et al. Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. Proceedings of the National Academy of Sciences 2010;107(46):19645-8.
[110] Clark BR, Hart RM, Gurdak JJ. Groundwater availability of the Mississippi embayment. U.S. Geological Survey professional paper. Reston: U.S. Geological Survey; 2011.
[111] Konikow LF. Groundwater depletion in the United States (1900−2008). U.S. Geological Survey Scientific Investigations Report 2013. Reston; 2013.
[112] Payne DF, Abu R, Malek, Clarke JS. Simulation of Ground-Water Flow in Coastal Georgia and Adjacent Parts of South Carolina and Florida -Predevelopment, 1980, and 2000. Scientific Investigations Report Reston; 2005.
[113] Gos P, Lavorel S. Stakeholders' expectations on ecosystem services affect the assessment of ecosystem services hotspots and their congruence with biodiversity. International Journal of Biodiversity Science, Ecosystem Services & Management 2012;8(1-2):93-106.
[114] Martín-López B, Montes C, Benayas J. The non-economic motives behind the willingness to pay for biodiversity conservation. Biological Conservation 2007;139(1–2):67-82.
[115] Bennett EM, Peterson GD, Gordon LJ. Understanding relationships among multiple ecosystem services. Ecology Letters 2009;12(12):1394-404.
[116] Bateman IJ, Harwood AR, Mace GM, Watson RT, Abson DJ, Andrews B, et al. Bringing Ecosystem Services into Economic Decision-Making: Land Use in the United Kingdom. Science 2013;341(6141):45-50.
[117] Goldstein JH, Caldarone G, Duarte TK, Ennaanay D, Hannahs N, Mendoza G, et al. Integrating ecosystem-service tradeoffs into land-use decisions. Proceedings of the National Academy of Sciences 2012;109(19):7565-70.
[118] Vigerstol KL, Aukema JE. A comparison of tools for modeling freshwater ecosystem services. Journal of environmental management 2011;92(10):2403-9.
[119] Terrado M, Acuña V, Ennaanay D, Tallis H, Sabater S. Impact of climate extremes on hydrological ecosystem services in a heavily humanized Mediterranean basin. Ecological Indicators 2014;37, Part A(0):199-209.
[120] Arnold JG, Fohrer N. SWAT2000: current capabilities and research opportunities in applied watershed modelling. Hydrological Processes 2005;19(3):563-72.
[121] Lautenbach S, Volk M, Strauch M, Whittaker G, Seppelt R. Optimization-based trade-off analysis of biodiesel crop production for managing an agricultural catchment. Environmental Modelling & Software 2013;48(0):98-112.

[122] Convertino M, Valverde LJ, Jr. Portfolio Decision Analysis Framework for Value-Focused Ecosystem Management. PLoS ONE 2013;8(6):e65056.
[123] Johnson JF, Novak J, Varvel G, Stott D, Osborne S, Karlen D, et al. Crop Residue Mass Needed to Maintain Soil Organic Carbon Levels: Can It Be Determined? BioEnergy Research 2014;7(2):481-90.
[124] McCarty JL, Justice CO, Korontzi S. Agricultural burning in the Southeastern United States detected by MODIS. Remote Sensing of Environment 2007;108(2):151-62.
[125] Frank S, Fürst C, Koschke L, Makeschin F. A contribution towards a transfer of the ecosystem service concept to landscape planning using landscape metrics. Ecological Indicators 2012;21(0):30-8.
[126] Peel MC, Finlayson BL, McMahon TA. Updated world map of the Köppen-Geiger climate classification. Hydrol Earth Syst Sci 2007;11(5):1633-44.
[127] O.C.G.A. 12-2-8 Georgia Mountain and River Corridor Protection Act. 2010.
[128] Kim K-H, Pauleit S. Landscape character, biodiversity and land use planning: The case of Kwangju City Region, South Korea. Land Use Policy 2007;24(1):264-74.
[129] Baessler C, Klotz S. Effects of changes in agricultural land-use on landscape structure and arable weed vegetation over the last 50 years. Agriculture, Ecosystems & Environment 2006;115(1–4):43-50.
[130] USGS GAP. Slope. 2011 [cited 2013 November 25]; Available from: http://gapanalysis.usgs.gov/species/data/download/#avoidance.
[131] USGS GAP. Aspect. 2011 [cited 2013 November 25]; Available from: http://gapanalysis.usgs.gov/species/data/download/#avoidance.
[132] USGS GAP. Human impact avoidance. 2011 [cited 2013 November 25]; Available from: http://gapanalysis.usgs.gov/species/data/download/#avoidance.
[133] EPD. Satilla River Basin Dissolved Oxygen TMDLs. Atlanta: Georgia Department of Natural Resources; 2001.
[134] MDEQ. Total Maximum Daily Load - Total Nitrogen and Total Phosphorus For Selected Large Rivers in the Delta - Yazoo River Basin. Jackson; 2008.
[135] MDEQ. TMDL for Organic Enrichment, Nutrients and Sediment for the Big Sunflower River - Yazoo River Basin - Sunflower, Coahoma, Washington, Humphreys, and Sharkey Counties, Mississippi. Jackson: Mississippi State University; 2003.
[136] Bridgeman TG, Jones JM, Shield I, Williams PT. Torrefaction of reed canary grass, wheat straw and willow to enhance solid fuel qualities and combustion properties. Fuel 2008;87(6):844-56.
[137] Mullen CA, Boateng AA, Goldberg NM, Lima IM, Laird DA, Hicks KB. Bio-oil and bio-char production from corn cobs and stover by fast pyrolysis. Biomass and Bioenergy 2010;34(1):67-74.

3.8 Supporting Information

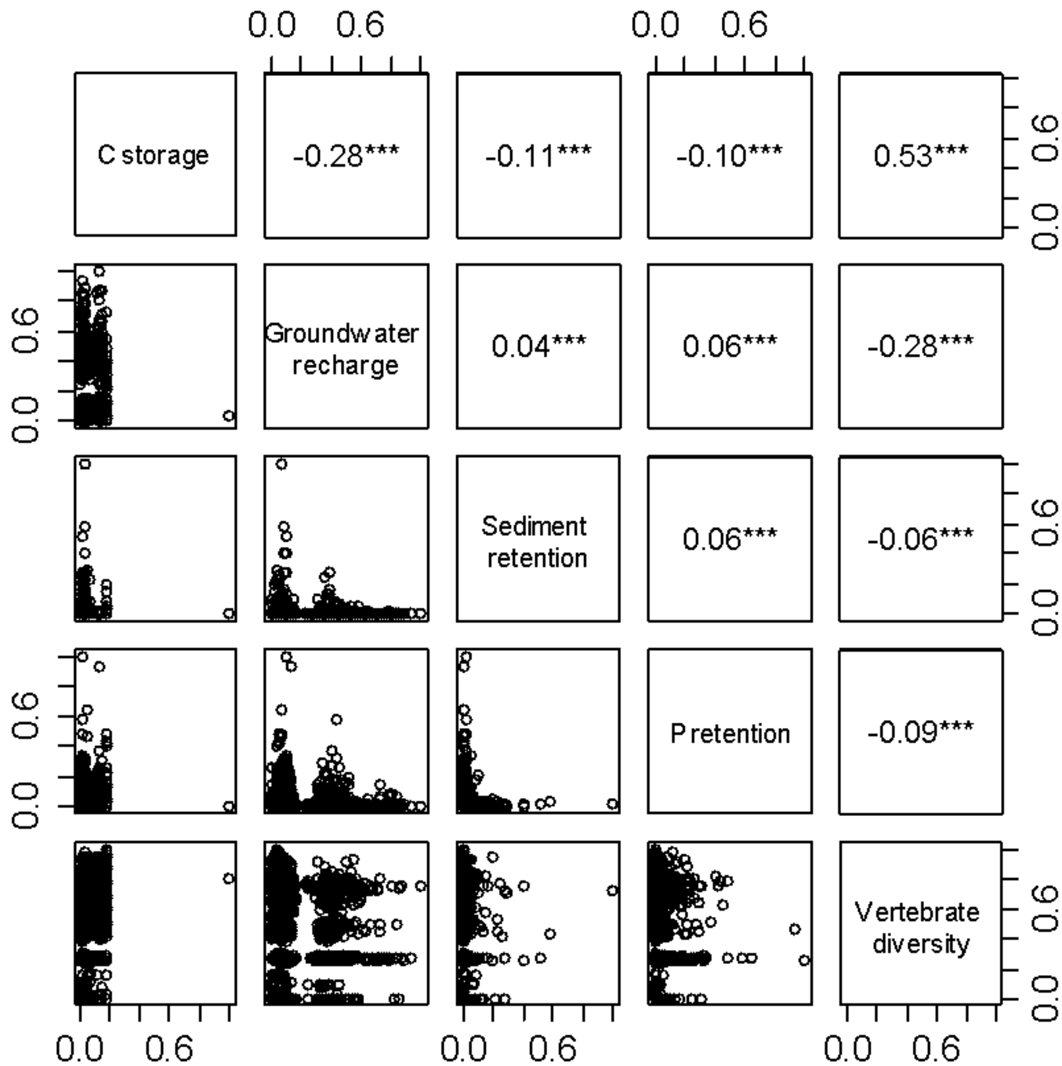

Fig. 3.S1: Paired correlation analysis of ESS supply in the Satilla watershed (p<0.001 (***); p<0.01 (**), p<0.05 (*)).

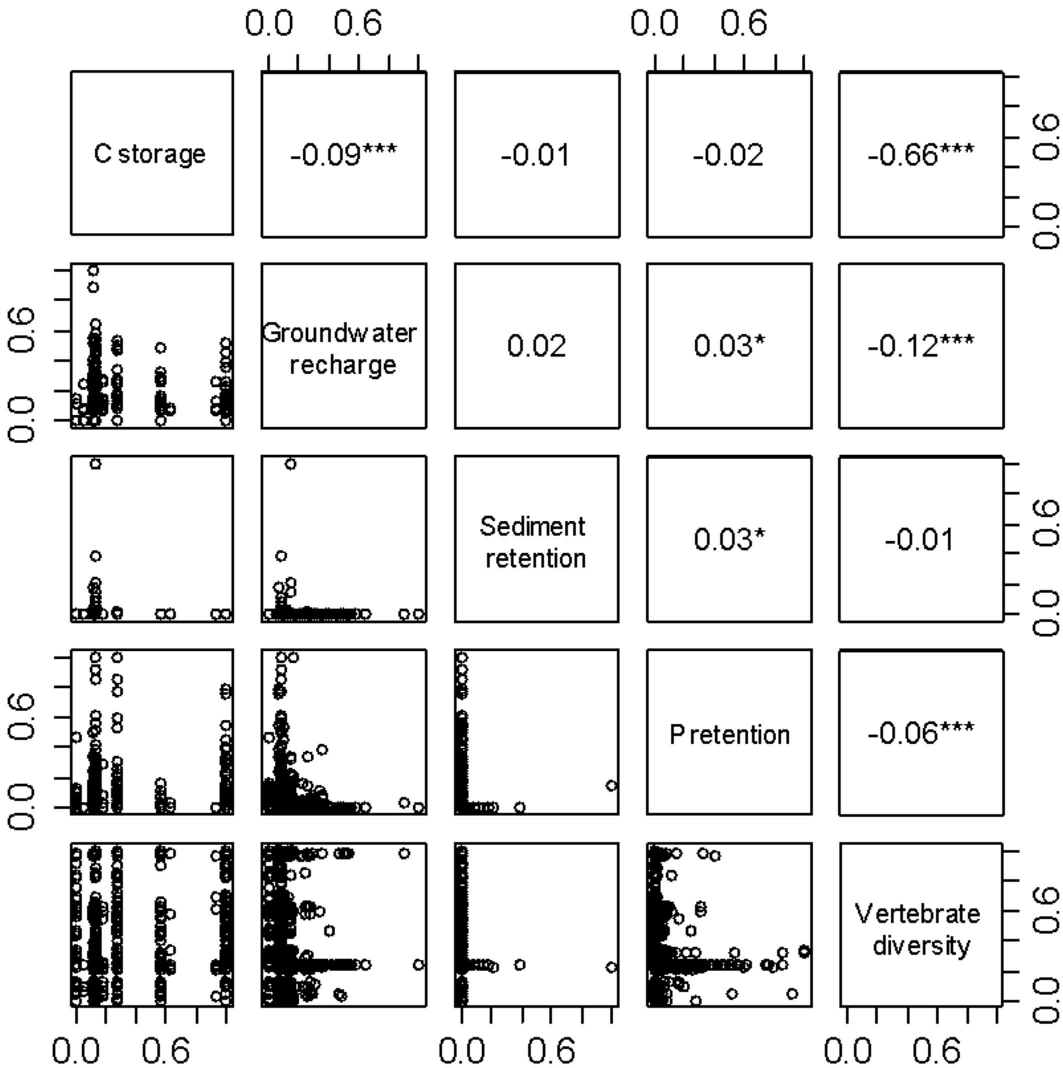

Fig. 3.S2: Paired correlation analysis of ESS supply in the Big Sunflower watershed (p<0.001 (***); p<0.01 (**), p<0.05 (*)).

Table 3.S1: Factors characterizing sufficient and insufficient ESS supply in the Satilla watershed (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to sufficient ESS supply; a negative value for the standardized β indicates that an explanatory variable is contributing to insufficient ESS supply.

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-1.49444	0.25723	-5.81	< 0.0001	***
Topography and soil parameters					
Slope [%]	0.76804	0.11812	6.502	< 0.0001	***
Aspect [°]	0.57418	0.06733	8.528	< 0.0001	***
Silt content [%]	1.84472	0.16523	11.165	< 0.0001	***
Soil erodibility	-3.09937	0.121	-25.615	< 0.0001	***
Depth to water table [mm]	3.39417	0.07826	43.37	< 0.0001	***
Available water holding capacity [cm cm^{-1}]	4.41303	0.22092	19.975	< 0.0001	***
Distance to stream [m]	0.88542	0.17021	5.202	< 0.0001	***
Natualness					
Land use intensity	-5.7782	0.07737	-74.679	< 0.0001	***
Hemeroby	-1.19755	0.14577	-8.215	< 0.0001	***
Landscape composition					
Wetlands, 300 m buffer [%]	-4.92968	0.22509	-21.901	< 0.0001	***
Edge density (plantation forestry)	4.32167	0.45223	9.556	< 0.0001	***
Largest patch index (plantation forestry)	2.92441	0.21905	13.35	< 0.0001	***
Largest patch index (forest)	6.11463	0.27058	22.598	< 0.0001	***
Largest patch index (agricultural land)	-3.25041	0.18299	-17.762	< 0.0001	***
Landscape configuration					
Connectance index (forest)	0.46992	0.07394	6.356	< 0.0001	***
Connectance index (agricultural land)	-0.37559	0.06626	-5.668	< 0.0001	***
Connectance index (barren)	0.78323	0.23461	3.338	0.00084	***
Landscape shape index (forest)	-4.26863	0.45722	-9.336	< 0.0001	***
Landscape shape index (shrub- and grassland)	-0.5583	0.17222	-3.242	0.00119	**
Landscape shape index (agricultural land)	0.6267	0.19153	3.272	0.00107	**
Landscape shape index (open water)	1.31774	0.11516	11.443	< 0.0001	***
Landscape shape index (urban)	1.47012	0.10881	13.511	< 0.0001	***

4 Assessing regional-scale impacts of short rotation coppices on ecosystem services by modeling land use decisions

This chapter is submitted to PLOS ONE:

Schulze J, Frank K, Priess JA, Meyer MA. Assessing regional-scale impacts of short rotation coppices on ecosystem services by modeling land use decisions. (*submitted*)

4.1 Abstract

Second generation feedstocks, for example, short rotation coppices (SRCs), are being discussed as chance to meet the world's continuously growing energy demand. However, identifying the environmentally best feedstock option requires a comprehensive impact assessment considering multiple ecosystem services (ESS) and biodiversity as trade-offs are inherent. In this study, we place SRCs in the landscape of the Mulde watershed in Central Germany by modeling farmers' land use decisions under different economic and policy-driven scenarios using an agent-based model (ABM). This allows for assessing regional-scale impacts on multiple ESS despite missing large-scale implementation. Meeting the regional demand for solid biomass of combined heat and power (CHP) plants, the required area and locations for SRCs hardly affect ESS. Similarly in the policy-driven scenario, placing SRCs on low or high quality soils to provide ecological focus areas (EFA), as promoted in the EU, hardly affects ESS. Only a substantial increase in the SRC production area, beyond regional demand of CHP plants, will have a relevant effect, namely a negative impact on food production as well as a positive on biodiversity and regulating ESS. Even if beneficial impacts on ESS occur, our results indicate that SRCs hardly stimulated the number of balanced ESS bundles. In that respect, future research should assess whether other potentially beneficial land use options such as cover crops or management practice significantly enhance the number of balanced ESS bundles. Coupling ABMs with biophysical ESS models can contribute to a comprehensive impact assessment of currently hardly implemented bioenergy feedstocks supported by spatial deployment that emerges under potential future scenarios.

4.2 Introduction

Several environmental and social concerns about bioenergy production arose in the past years [1, 2] while, at the same time, the world's energy demand is continuously growing [3]. Solving this discrepancy is among the societal challenges of the next decades. One option to tackle this are second generation (2G) feedstocks: "perennial, ligno-cellulosic feedstocks that are non-food crops" [4] that are believed to be politically encouraged in the near future [5]. However, crucial for a successful bioenergy usage are science-based safeguards for the adoption of the best feedstocks [6]. This requires a comprehensive impact assessment of 2G feedstocks considering multiple ecosystem services (ESS) as trade-offs, especially between provisioning and regulating ESS, are inherent [7-9]. In temperate climate, prominently discussed 2G feedstocks are short rotation coppices (SRCs) [10].

SRCs are fast-growing trees, in the EU mostly the species poplar and willow, which are partly commercially grown as perennial energy crops on agricultural land [11]. Plantations are harvested every few years and afterwards stump sprouting takes place. After several of these rotations, the land is re-cultivated. SRCs may fulfill multiple bioeconomic purposes: they serve as source of material, heat and power supply. At the same time, several environmental advantages are being discussed. They are believed to increase biodiversity [5, 12, 13] and to positively affect soil and water quality [14, 15]. Furthermore, the reformed Common Agricultural Policy (CAP) (2014-2020) couples subsidy payments to the obligation that at least 5% of a farmer's arable land are managed as ecological focus areas (EFAs). SRCs qualify as EFAs due to their beneficial impact on the environment [16]. Despite the known environmental benefits, currently only approximately 6500 ha SRCs are established in Germany [17]. However, Kraxner et al. [18] project worldwide an increase of SRC plantations to 190-250 million ha by 2050.

Several studies modeling the potential supply of perennial energy crops and accompanied impacts exist (e.g., [19], Meehan et al. [19], Aust et al. [20], Tölle et al. [21]). At this, the spatial configuration of energy crops is believed to affect the assessment [22]. Holland et al. [5] and Milner et al. [4] emphasize the research gap to upscale feedstock production to commercial scale. As data on the spatial allocation of currently hardly implemented land use options is missing, models of potential supply often base upscaling processes on heuristics that place SRCs in the landscape. Meehan et al. [19], for example, replace annual by perennial energy crops. Tölle et al. [21] locate SRCs on land with suitable cultivation conditions, e.g., sufficient available soil water capacity. Missing existing larger commercial SRC plantations, scenarios reflecting farmers' decisions within the existing and potential future political framework are needed to determine more realistic spatial allocations of SRCs. Therefore, we believe that it is valuable to explicitly incorporate human decision-making for the spatial allocation of SRCs in such models. Several studies have shown that the inclusion of human decision-making in models is crucial [23, 24]. In the case of SRCs, this is especially important as individual restraints of farmers have been identified as important factor for the slow uptake [25, 26]. Equation-based simulation models could be used to incorporate human decision-making. However, these do not allow incorporating decisions of multiple heterogeneous, interacting agents. In contrast, this can be achieved by using agent-based models (ABMs). ABMs are computational models simulating macro-level phenomena emerging from micro-level decision-making of heterogeneous, interacting agents [27].

In this study, we use a spatially-explicit ABM to simulate the decision-making of interacting farmers facing the choice between SRCs and conventional annual agricultural activity (crops or fallow land). This decision is strongly influenced by socio-economic framework conditions that can be varied in the model to represent different economic scenarios. In addition, embedded in the recent discussion on "Greening" in Germany [28], we assess two policy scenarios where a share of the landscape is cultivated with SRCs to fulfill the CAP requirements for EFAs. Currently, to receive payments, for example, 5% of agricultural land need to be set aside; alternatively SRCs can be cultivated with a weighting factor of 0.3 [29]. We apply the model to the Mulde watershed in Central Germany to assess the impact of SRC expansion at regional scale on crop yields, carbon storage, nutrient and sediment retention, and biodiversity. We cover these local/regional ESS as those are hardly covered in common environmental assessments of bioenergy feedstock production, e.g., LCAs [30, 31]. Thereby, we assess where SRCs establish, given the assumptions we have made in the ABM, and quantify occurring ESS synergies and trade-offs under different economic and policy-driven scenarios at the regional scale.

4.3 Materials and Methods

We modeled SRC development and ESS for the reference year 2006, for which major land use/land cover (LU/LC) data is available.

4.3.1 Study site

The investigated share of the Mulde watershed is mostly located in the German Federal State Saxony (see Fig. 4.1) and amounts to about 5,791 km². The Mulde is a tributary of the Elbe river and formed by its headwaters Zwickauer Mulde and Freiberger Mulde, which have their source in the Ore Mountains. The altitude ranges from 70 to 1214 m. Climatically, the precipitation in 2006 in the Mulde watershed with its humid continental climate (834.6 mm, SD: 180.6 mm) was 9 % lower than the normal climate conditions for the period from 1991 to 2011 [32]; the minimum and maximum average temperatures in 2006 in the Mulde watershed (T_{min}: 4.5 °C, SD: 0.8 °C, T_{max}: 13.1 °C, SD: 1.4 °C) deviated less than 1 °C from the normal climate conditions for the period from 1991 to 2011 [32]. The amplitude of precipitation ranged between 500 mm and 1290 mm in 2006. The loess soils in the lowlands are dominated by crop production, whereas the Ore Mountains are dominated by forestry [33]. Winter wheat (24%), winter rapeseed (18%), and winter barley (12%) dominated the cropland in 2006. Currently, SRC only amounts to 0.03 % of the agricultural land in Saxony [34]. There are only few SRC sites in the Mulde watershed, which are often trial sites. Contrastingly, there are about 36 combined heat and power (CHP) plants in Saxony [35] that may use SRC products. 15 of these CHP plants are located in the Mulde watershed.

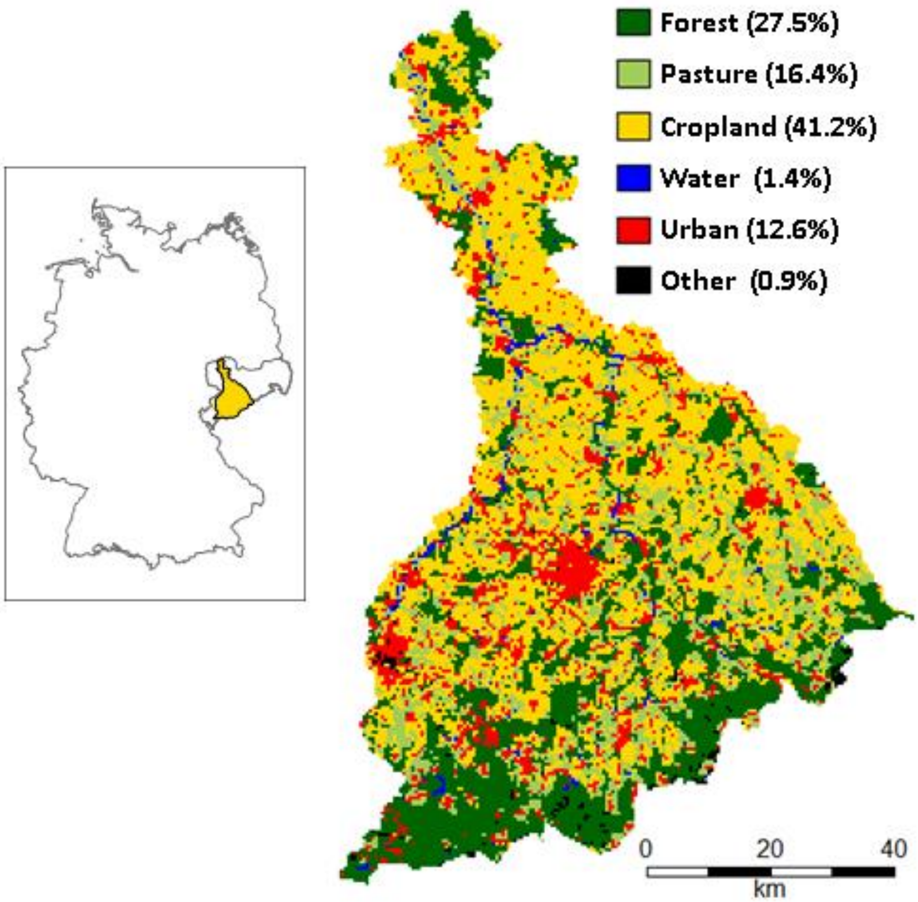

Fig. 4.1: Land use/land cover in the Mulde watershed and its location in Germany [36, 37].

4.3.2 Agent-based model

The model used within this study is an extended version of the ABM described in Weise [38]. Here, we present a short description of the model, whereas the full model description using the Overview, Design Concepts and Detail (ODD) protocol [39, 40] can be found in the supporting information S1 Text. The main aim of the model is to assess changes in agricultural landscapes, in particular SRC expansion, by modeling individual land use decisions.

The spatially explicit model consists of an artificial landscape that is subdivided into pixels, each being farmed by an individual land user (agent). The underlying landscape consists of a soil quality layer and sites of customers for SRC products, i.e., CHP plants. The agent cultivates land to generate income through the production of agricultural goods. The agent can choose between three different land use options: SRCs, annual crops, or fallow land. Among these, the agent chooses the land use option that will yield the maximum net profit. The net profit for a land use option is given by the difference between revenue and costs. The revenue is influenced by market price and site-specific yield, while the costs are incorporated via production and transportation costs. At this, the yield is determined by physical site-conditions, i.e., soil quality, while transportation costs of SRC products depend on the distance to CHP plants. Although traditional and self-interested profit maximization, i.e., the rational of the homo economicus, is widely accepted as decision criterion in economics, also non-commercial factors are believed to influence agricultural decisions [41]. However, Brown et al. [42] showed with a survey in the UK that economic factors are of main importance. Non-economic factors are less important and only slightly influence decisions to cultivate bioenergy crops such as the willingness to reduce GHG emissions. Therefore, we see this simplification as appropriate for our model design. For SRC practice in Germany, farmers' are provided with advisory material (e.g., the manual of Skodawessely et al. [43] or the profitability calculator provided by the research project AGROWOOD [44]). To reflect the situation in practice, we adapt the therein suggested profitability calculation as decision criterion.

The market prices for wood chips from SRCs and for annual agricultural products are given by the relation of exogenously given demands and the endogenously resulting supply that is determined by the agents' decisions. Hereby, we follow a standard assumption in microeconomic theory [45, 46].

The results based on the artificial landscape in the ABM needed to be transferred to the real landscape (Mulde watershed) to assess impacts of SRC expansion. Therefore, the initial artificial landscape for the ABM needed to fulfill statistical characteristics present in the Mulde watershed.

Therefore, the distribution of soil quality for the ABM was generated using a random generator that returns uniformly distributed, spatially correlated numbers with fixed arithmetic mean and correlation length. For this purpose, the standard method of Cholesky decomposition was used (see Appendix A in Thober et al. [47] for details). Using the Cholesky factor in the generation of the random field guarantees that the covariance among all cells is considered. The soil quality distribution of the artificial landscape was adapted to the soil quality distribution in the Mulde watershed. (1) We set the arithmetic mean from the artificial landscape to the one in the Mulde watershed. (2) We divided the continuous soil qualities into 6 classes with the same proportion as given in the empirical spatial dataset soil quality [48]. (3) We set the correlation length by adapting spatial autocorrelation (using Moran's I up to a neighborhood of 49*49 cells) of the artificial landscape to the real landscape.

The number of CHP plants in the initial landscape was set according to the areal density in the Mulde watershed given by the data described in the previous section.

With the procedure described above, we generated 100 initial landscapes to receive an appropriate level of accuracy and to avoid effects of stochastic outliers. The ABM ran on these initial landscapes and generated land use patterns for the investigated land use options, i.e., SRCs, annual crops, and fallow land. The location of a specific land use option could be described in terms of site-specific characteristics of the initial landscape. Subsequently, we used this information to project the specific land use options into a real landscape. In our case, we quantified probabilities of SRCs presence depending on soil quality and distance to CHP plants.

Based on these probabilities and the soil quality and CHP plants data, we randomly allocated the SRCs in the Mulde watershed. Therefore, we randomly selected sites for SRCs based on the total intended area of SRCs in the Mulde watershed (from now on called target amount) and on the probability of SRCs occurrence depending on soil quality and distance to CHP plants. Therefore, each pixel is assigned its probability of being selected as SRC (based on a probability derived from the ABM and data on soil quality and distance). For each pixel, a random number between 0 and 1 is drawn and all pixels where this number is less than the probability are selected as SRC. This procedure is repeated until the number of selected pixels reaches at least the targeted amount. If the number of SRC pixels exceeds the target amount, SRC pixels are deselected according to their probabilities until target amount is reached.

The majority of parameters in the ABM was based on literature (for details see full model description in the supporting information S1 Text). The demand for annual agricultural crops was calibrated using the LU/LC data for the Mulde case study. This demand was set by aligning the initial shares of land use options under the baseline scenario with those in the Mulde watershed.

4.3.3 Scenarios for SRC development

For the standard scenario, we assumed that the CHP plants currently present in the Mulde watershed were the only customers of wood chips and that these entirely met their biomass demand by wood chips from SRCs. We calculated the share of SRCs needed to meet the demand of the current CHP plants in the region derived from the reference values in FNR [49] and set the standard wood demand to the value needed to achieve this share of SRCs.

Increasing global demand for wood combined with limited forest resources will most likely increase prices for wood in the future [50]. Therefore, we assessed the increase of wood chips demand by comparing the standard scenario with three further scenarios (medium, high and very high demand). In these additional scenarios, we did not spatially explicit allocate further CHP plants in the landscape because the regional energy and heat demand would unlikely further increase. We rather expect that external drivers will raise the demand for solid biomass obtainable from SRCs. Considering global bioenergy trade, Matzenberger et al. [50] expect an 80-times increase of solid biomass trade between world regions in the period 2010 to 2030 for a synthesis of 22 scenario models.

In addition to the economic scenarios simulated with the ABM, we included two policy scenarios. Embedded in the recent discussion on “Greening” in Germany [28], we assumed that 16.67% of all arable land would be used to cultivate SRCs. To receive payments, 5% of arable land needs to be designated as EFAs, which can be simply set aside land. Alternatively, SRCs can be cultivated with a weighting factor of 0.3 [29]. We assumed that SRCs would be located on land with low soil quality for economic reasons. To assess the impact of soil quality, we also compared this scenario to a second policy scenario where the best 16.67% of the entire arable land with respect to soil quality would be converted to SRCs to analyze potential ESS impacts.

4.3.4 Ecosystem services and biodiversity

4.3.4.1 Provisioning ecosystem services

For all scenarios, we calculated the crop and SRC yields. For crop production, we spatially downscaled the average yield per ha at the district level [51, 52]. We considered the impact of soil and climatic heterogeneity on yields by calculating the arithmetic mean of the agricultural yield potential for each district [53]. We assigned each pixel the yield available at district level and raised or lowered this value depending on the actual agricultural yield potential of the pixel relative to the district arithmetic mean.

We selected poplar as SRC species due to the dry climate in the agriculturally dominated lowlands. To model a spatially explicit KUP yield, we used the regression model developed in Saxony by Ali [54]:

$$Yield = a_4(a_1 * C + a_2 * P * SQI + a_3 * \frac{T}{AWC})^{a_5} \quad (1)$$

with

$$a_4 = -1.13 * 10^{-9} * N^2 + 2.54 * 10^{-5} * N + 0.028 \quad (2)$$

and

$$a_5 = 3.41 * 10^{-9} * N^2 - 5.01 * 10^{-5} * N + 2.614 \quad (3)$$

where *C* is the rotation cycle, *P* sum of the precipitation for the months May and June, *SQI* the soil quality index, *T* average temperature for the months April until July, *AWC* the available water holding capacity, *N* the planting density and a_1 up to a_5 are species specific parameters. Based on the existing practice in Saxony, we assumed the use of the most common poplar clone Max. with the parameters and datasets given in Table 4.1.

Table 4.1: Parameter and datasets used to calculate yield of the poplar clone Max.

	Item	Value	References
N	Planting density [*n* ha^{-1}]	9446	TU Dresden/AgroForNet [34]
C	Rotation cycle [a^{-1}]	5.5	TU Dresden/AgroForNet [34]
a_1		1.569	Ali [54]
a_2		0.0004	Ali [54]
a_3		-23.198	Ali [54]
P	Precipitation (sum May-June) [mm]		Jäckel et al. [32]
T	Average temperature April-July [°C]		Jäckel et al. [32]
SQI	Soil quality index		LfULG [48]
AWC	Available water holding capacity [mm]		LfULG [48]

4.3.4.2 Regulating ecosystem services and biodiversity

We used InVEST (Integrated Valuation of Environmental Services and Tradeoffs) to calculate different regulating ESS. InVEST uses ecological production functions to simulate the provision of ESS under different scenarios. First, we calculated the amount of carbon stored according to the IPCC guidelines [55], supported by InVEST [56, 57], with the data indicated in Table 4.2 for 2006. Second, we modeled the phosphorous (P) export and retention with InVEST. Based on the runoff, the P inputs were routed through the respective watershed. The retention largely depends on the topography and vegetative cover. Third, we modeled the amount of retained sediment with InVEST based on the universal soil loss equation [58, 59]. The baseline scenario for P and sediment retention was validated in Meyer et al. [60].

Table 4.2: Data items for carbon storage (No. 2, 16), P retention and export (No. 1 -7, 10-12), sediment retention and export (No. 1-3, 8-9, 13-15 and for biodiversity (No. 2, 17-21); we refined the default parameter of InVEST with the indicated sources (No. 10 - 15); methodological sources are equally included.

	Input datasets	References
1	DEM (3 arc-seconds) [m]	Lehner et al. [77]
2	LU/LC	European Environment Agency (EEA) [36], Wochele et al. [37]
3	Potential Natural Vegetation	LfULG [78]
4	Reference Evapotranspiration (10 arc-min) [mm a^{-1}]	FAO Geonetwork [79]
5	Precipitation [mm a^{-1}]	Jäckel et al. [32]
6	Depth to any soil restrictive layer [mm]	Panagos et al. [80], LfULG [48]
7	Available water holding capacity [cm cm^{-1}]	Panagos et al. [80], LfULG [48]
8	Erosivity (R) [MJ mm ha^{-1} h^{-1} a^{-1}]	Bräunig [81]
9	Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	LfULG [48], Bischoff [82]
10	Rooting depth [mm]	
11	P export [kg ha^{-1} a^{-1}]	Reckhow et al. [83]
12	P retention efficiencies [%]	
13	cover-management factor (C)	
14	Support practice factor (P)	SMUL [84]
15	Vegetation sediment retention efficiency [%]	
16	Carbon pools [t ha^{-1}]	Fraver et al. [85], Bohn et al. [86], Müller-Using and Bartsch [87], Wördehoff et al. [88], Polley and Henning [89], Strogies and Gniffke [90], BGR [91]
17	Population density [*n* km^{-2}]	Priess et al. [92]
18	Street and railway map	BKG [93]
19	N deposition	Builtjes et al. [94]
20	Critical N loads	Builtjes et al. [94]
21	Terrestrial ecoregions	Olson et al. [95]

4.3.4.3 SRC impacts on regulating ESS bundles

We used cluster analysis to identify ESS bundles occurring for different concentrations of SRC in the Mulde watershed and selected the methods based on Mouchet et al. [62]. For the cluster analysis, we used a random sample of 1,000 pixels to reduce the effect of spatial autocorrelation. We identified ESS bundles by K means and displayed them as starplots in R [63]. ESS bundles are described as "sets of services that appear together repeatedly" by Raudsepp-Hearne et al. [8]. To explain differences between ESS bundles, we applied a binomial logistic regression model. In the regression model, we tested indicators of landscape composition and naturalness, soil, topography, and climate, see Table 4.S5. We calculated landscape composition and naturalness indicators in a moving window approach for a buffer radius of 5 km, which was ten times the LU/LC pixel size. We removed variables with variance inflation factors >10 to reduce multicollinearity. Next, we removed non-significant explanatory variables in a backward stepwise manner based on the Akaike information criterion. Next, the significance of the final model was tested against a null model using a likelihood ratio test. To assess the spatial autocorrelation of the final model, we added geographic coordinates and tested for significant difference towards the final model without geographic coordinates with the likelihood ratio test.

4.4 Results

4.4.1 SRC distribution and associated ecosystem services impacts under economic and policy-driven scenarios

Spatial distribution of SRCs depended on the economic and policy-driven scenarios (Fig. 4.2). The share of SRCs differed substantially between the four economic scenarios (2%, 4%, 14%, and 24% of the total area for standard, medium, high, and very high demand respectively). However, SRC distributions showed similar characteristics under the four scenarios: sites with high soil quality indices showed hardly any deployment. SRCs are only economically viable on inferior soils, where they can compete with annual crops. Only in scenario 1 (standard demand), SRCs were located near existing CHP plants because in the other scenarios no local consumption was assumed (see section "Scenarios for SRC development"). Distribution of SRCs under the first policy-driven scenario, where 16.67% of agricultural land with the lowest soil quality indices was converted to SRC cultivation (see scenario 5), was similar to the economic scenario with high demand. In contrast, under the second policy-driven scenario (see scenario 6), SRCs were located within the central Mulde watershed, where soil quality indices were high.

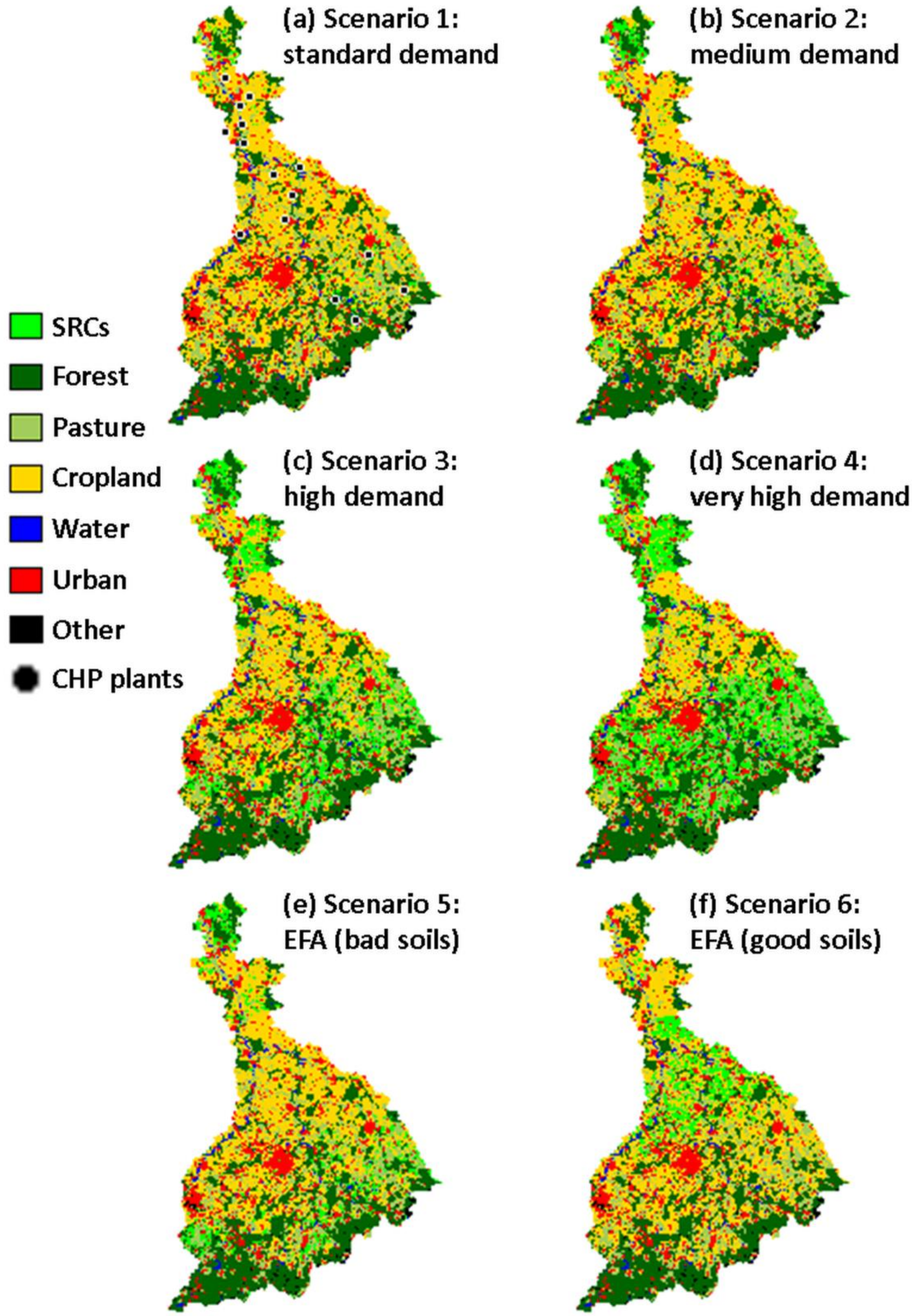

Fig. 4.2: Deployment of SRCs in the Mulde watershed for four economic (1-4) and two policy-driven (5-6) scenarios. The economic scenarios are based on the ABM. The policy scenarios reflect the potential deployment of SRCs to fulfill the requirements for EFAs (ecological focus areas). The dots indicate existing CHP plants [35].

In general, economic and policy-driven scenarios affected provisioning and regulating services as well as biodiversity (Fig. 4.3). Scenario 1 (standard demand), i.e., demand from the currently existing CHP plants solely met by SRCs, did not have a substantial effect on the investigated ESS. Only a large increase in demand (scenarios 3 and 4) for woody products from SRCs revealed substantial trade-offs between the provision of annual agricultural products and SRC yields as well as regulating ESS and biodiversity. For example in scenario 3, compared to the baseline scenario SRC deployment on 33% of cropland synergistically raised biodiversity (+22%) and carbon storage (+5%) and reduced phosphorus (-7%) and sediment export (-19%) from a regional perspective.

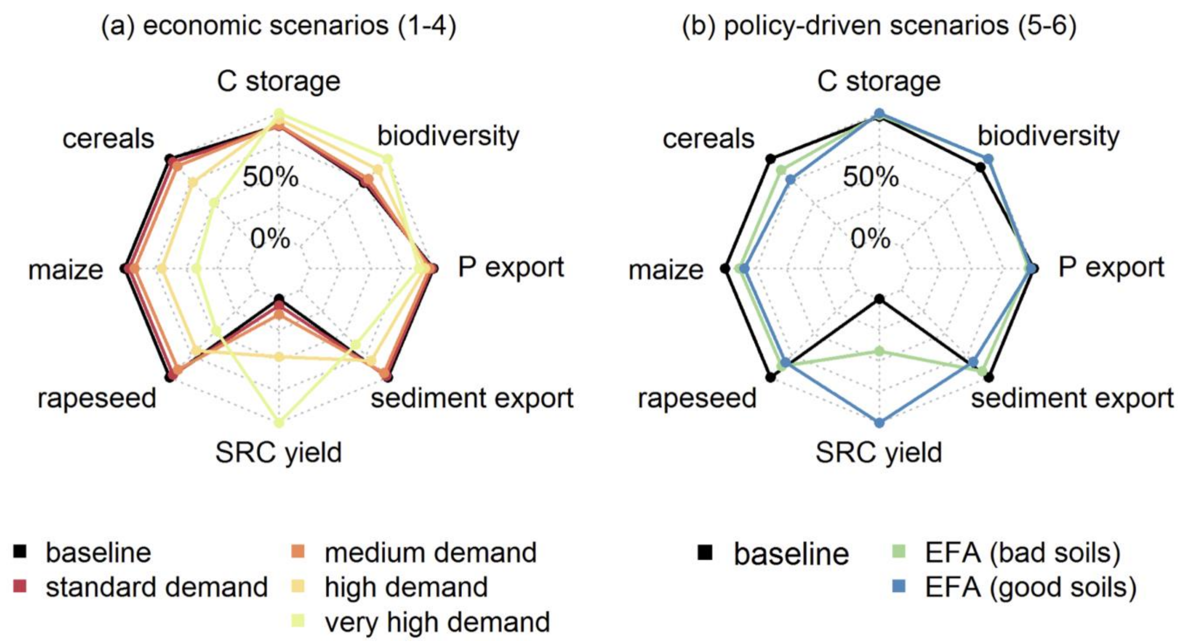

Fig. 4.3: Trade-offs between provisioning and regulating ESS in (a) the economic and (b) the policy-driven scenarios, each set compared to the baseline scenario (black line). For each ESS, the scenario values are normalized with respect to the maximum value obtained, in other words the maximum value of all scenarios is set to 100% and differences of the remaining scenarios are given in percent of the maximum value. For most of the ESS higher values implies a better performance, only for P and sediment export a lower value is better.

Interestingly, the two policy-driven scenarios led to different trade-off effects. SRCs placed on good soils positively affected SRC yield (17% of cropland), carbon storage (+3%) and sediment export (-18%) at high costs of annual crops (ranging from -16% to -23% depending on the crop) (see scenario 6). In contrast, SRCs placed on bad soils, led to slightly higher reduction in phosphorus export (+4%) while annual crops were less negatively affected (ranging from -12% to -13% depending on the crop) and sediment export (-7%) and carbon storage (+1%) less positively affected (see scenario 5). Effects on biodiversity hardly differed between the two policy-driven scenarios (ranging from 10% to 11%).

4.4.2 SRC impacts on regulating ESS bundles

Most of the ESS bundles prevailed independent from the share of SRC in the landscape that varied between the scenarios (see Fig. 4.4). Only ESS bundle 2 strongly differed between the baseline scenario (a), scenario 2 (medium demand) (b) vs. the more homogenous scenarios 4 (very high demand) and 5 (EFA (bad soils)) (c, d). However, the abundance of ESS bundles changed. For example, scenario 4 had a much higher abundance of the high biodiversity bundle 2 than scenario 5. Vice versa, bundles 1 and 4 hardly differed between scenario 4 and 5. These bundles were less abundant in scenario 4, i.e., especially P and sediment retention became less important in the economic scenario 4 compared with the policy scenario 5.

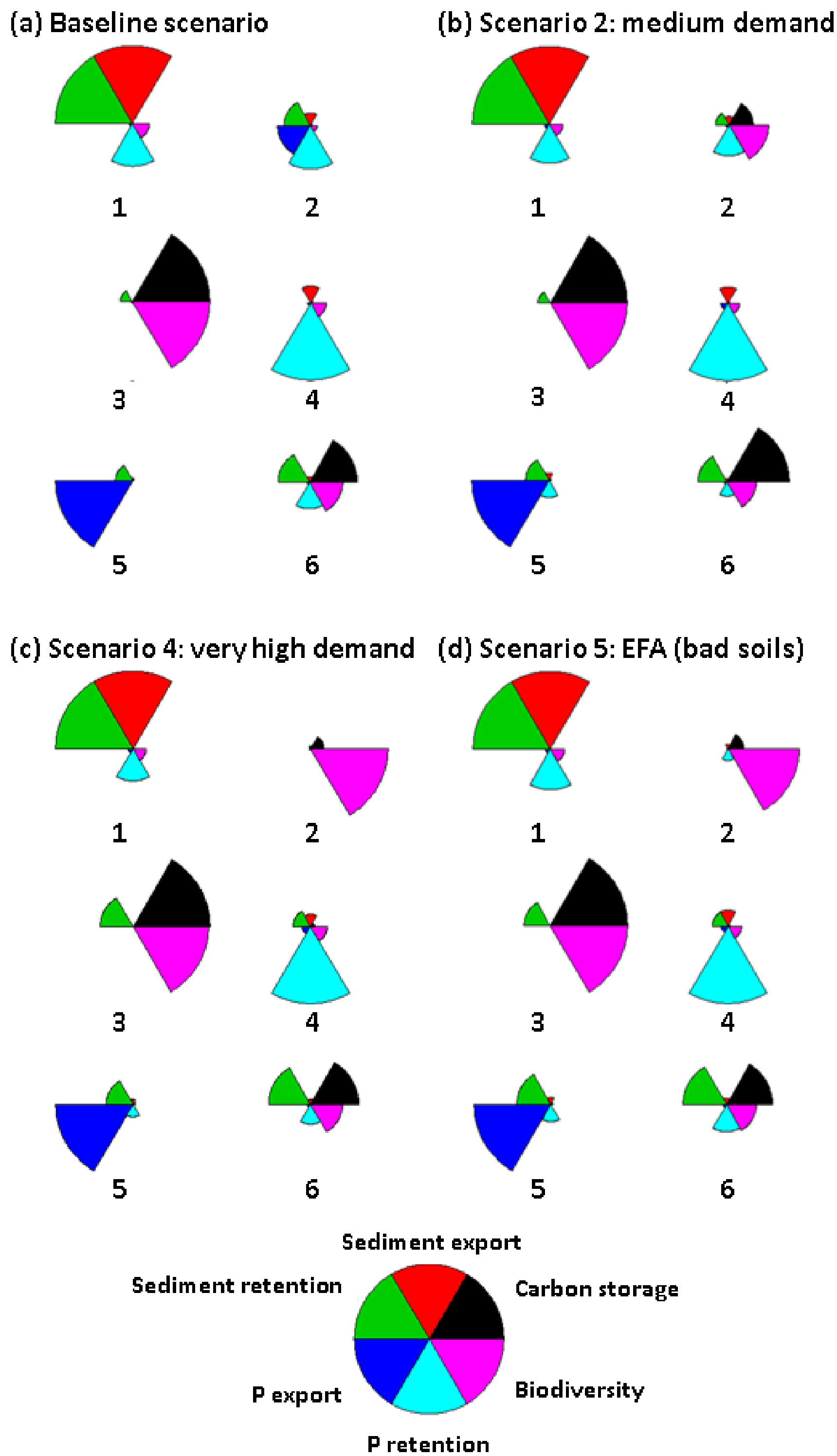

Fig. 4.4: ESS bundles for the baseline scenario (a), two economic scenarios (b) and (c), and a policy scenario (d). The abundance of the ESS bundles based on K means is in (a) (n: 1770, 1317, 1391, 5765, 529, 1978), (b) (n: 1810, 1669, 1228, 5611, 1197, 1235), (c) (n: 1247, 3396, 1478, 3982, 1073, 1574), and (d) (n: 1748, 992, 1515, 5613, 1180, 1702). The highest arithmetic mean value for each ESS category is used as maximum to scale the radar charts.

Analyzing the differences between the bundles in scenario 4, a higher share of SRC in the landscape mostly increased the abundance of bundles 2 and 3, see Fig. 4.5.

Cluster	1	2	3	4	5	6
1	-	- 3.17	- 1.55	- 0.72	1.29	-
2	1.86e-13 ***	-	- 1.17	-	4.29	5.12
3	0.032525 *	0.05931 .	-	-	2.73	- 0.88
4	0.01112 *	< 2e-16 ***	< 2e-16 ***	-	0.68	-
5	0.01670 *	< 2e-16 ***	5.99e-08 ***	0.004387 **	-	-
6	-	0.000871 ***	0.0268 *	-	-	-

Fig. 4.5: % SRC characterizing ESS bundles in scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to the cluster with the lower ordinal number; a negative value for the standardized β indicates that an explanatory variable is contributing to the cluster with the higher ordinal number. The entire regression results are listed in supporting information S2 File.

In that respect, we might especially enhance either (i) biodiversity or (ii) carbon storage, biodiversity, and sediment retention. However, the share of SRC in the landscape hardly enhanced the occurrence of the more balanced bundle 6. For example, the factors that contribute to the balanced bundle 6 instead of bundle 5 with high P export were a higher slope and a higher available water holding capacity (Table 4.3). A higher soil quality index and higher precipitation enhanced the occurrence of bundle 5. The high explanatory power of the biophysical factors and the rather low explanatory power of SRC showed that a balanced bundle was hardly obtainable through modified landscape composition, e.g., with a high share of SRCs in the landscape.

Table 4.3: Factors characterizing ESS cluster 5 and 6 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 5; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 6. The likelihood ratio test showed a significant difference when the final model was compared to a null model ($\chi 2= 1398.4$, df = 14, $p<2.2e-16$). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference ($\chi 2= 83.3$, df = 3, $p<2.2e-16$).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	13.5768	0.9016	15.059	< 2e-16	***
Topography and soil parameters					
Elevation [m]	-8.0942	0.9503	-8.518	< 2e-16	***
Slope [%]	-11.8836	0.6643	-17.89	< 2e-16	***
Aspect [°]	0.5111	0.2936	1.741	0.0817	.
Curvature [*score*]	-6.1341	0.7359	-8.336	< 2e-16	***
Effective rooting depth [mm]	-8.4734	0.8305	-10.203	< 2e-16	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	-1.7053	0.9607	-1.775	0.0759	.
Erosivity (R) [MJ mm ha^{-1} h^{-1} a^{-1}]	-8.4734	0.8305	-10.203	< 2e-16	***
Available water holding capacity [cm cm^{-1}]	-22.1665	2.2753	-9.742	< 2e-16	***
Soil quality index	9.9122	1.0365	9.563	< 2e-16	***
Climate					
Precipitation [mm]	3.5771	0.5564	6.429	1.28e-10	***
Reference Evapotranspiration [mm a^{-1}]	-3.6937	0.5329	-6.932	4.16e-12	***
Landscape composition					
Pasture, 5 km buffer [*%*]	2.3507	0.4175	5.63	1.8e-08	***
Water, 5 km buffer [*%*]	-2.7981	0.461	-6.069	1.29e-09	***
Urban, 5 km buffer [*%*]	2.0017	0.4844	4.132	0.0000359	***

4.5 Discussion

4.5.1 Ecosystem services and biodiversity under economic and policy-driven scenarios

Major aim of this study was to analyze the impact of SRCs on ESS and biodiversity to identify synergies and trade-offs under different economic and policy-driven scenarios. SRCs are discussed as sustainable 2G feedstock for energy production. Our approach of placing SRCs in the landscape by modeling farmers' decisions allows to close the gap of existing research synthesizing mostly plot/field scale studies for ESS [4] and conceptually discussing, but not testing the beneficial impact of SRC on ESS at the regional scale [64]. Only few studies such as Fürst et al. [65] have assessed their impact on multiple ESS and biodiversity at the regional scale, but without modeling farmers' decision making to assess commercial SRC deployment. Simulating farmers' decisions allowed us to develop spatially explicit SRC distributions, given the assumptions made in the ABM, at a commercial scale for an environmental and ESS impact assessment as requested by several authors for 2G feedstocks, e.g., Holland et al. [5]. Beyond existing SRC impact assessments with a focus on carbon storage, e.g., Milner et al. [4], we spatially explicitly model multiple ESS.

In the investigated economic scenarios, farmers preferably cultivated SRC plantations in the southern and northern part of the Mulde watershed with dominating low quality soils. SRCs seem to be competitive with annual crops on these low quality soils. In that respect, we can confirm Hellmann and Verburg [66] who assume "[…] that it is unattractive to cultivate biofuel crops on locations with relatively very high yields of cereals and root crops due to economic competition" (p. 2414). In scenario 1 (standard demand), we tested the impact of switching current input of CHP plants in the Mulde watershed to wood chips coming from SRCs. In this scenario, SRCs establish in the proximity of existing CHP plants. This is in line with Kocoloski et al. [67] and Vanloocke et al. [68] who state that 2G feedstocks are likely to be clustered around biorefineries. Concerning the impact assessment, we showed that the investigated provisioning and regulating ESS and biodiversity would only be slightly affected in scenario 1. In particular, food production will not be affected much. Only substantially promoting SRCs would have a significant impact in decreasing order relative to the baseline scenario as follows: biodiversity, sediment export, P export, and carbon sequestration. This is in accordance to Fürst et al. [65] who showed for a case study in Central Saxony, Germany, that a substantial increase of SRC production by up to 30% (depending on the region) would beneficially affect the provision of ESS and biodiversity. Meehan et al. [19] also showed a decrease in the P export to surface water and an increase in carbon sequestration by switching annual to perennial crops. For the ESS assessment, we chose simple tools such as InVEST as they allow to model multiple regulating ESS with rather high reliability relative to the effort, e.g., Meyer et al. [69]. However, simple tools are hardly disadvantageous as we focus on the relative comparison of different scenarios which would equally contain possible imprecisions.

Besides the economic scenarios, we assessed instruments in two policy scenarios for their impact on SRC deployment and ESS. We compared the more likely policy scenario 5 ("EFA (bad soils)") with the rather hypothetical scenario 6 ("EFA (good soils)"). Both scenarios show different impacts on the regarded ESS. In scenario 6, SRC yield, carbon storage, and sediment export are positively affected at high costs of annual agricultural production. In contrast, scenario 5 is slightly more beneficial for P export while crop production is not tremendously decreased. Both scenarios have the same share of SRCs in the landscape, but with differing spatial distribution answering the research need raised by Holland et al. [5]. Holland et al. [5] raised the question whether the distribution of feedstocks might affect the provision of ESS. This underlines the importance for aiming at realistic distributions of energy crops, for example by explicitly modeling farmers' decisions. In addition, it shows potential to reflect the current EU policy to couple subsidy payments to EFAs. Our study reveals that the rules on the EFAs' properties are insufficiently specified for an effective environmental improvement.

To be able to identify the impact of SRCs in their proximity, we clustered ESS bundles and analyzed how the share of SRCs in the landscape affects the occurrence of the respective bundles. Thereby, we assessed occurring trade-offs between multiple ESS and biodiversity, which are inherent in the 2G feedstock expansion [7-9]. In that respect, our approach helps to balance competing services for deployment decisions as required by Holland et al. [5]. We showed that SRCs especially enhanced either (i) biodiversity or (ii) carbon storage, biodiversity, and sediment retention, and broadened the findings for single ESS from the synthesis by Holland et al. [5]. However, balanced ESS bundles are hardly obtainable even with a high share of SRCs in the landscape. Their occurrence can be better explained by biophysical factors. The beneficial impact of SRCs on multiple ESS and biodiversity as discussed in several studies, e.g., Manning et al. [64] or Holland et al. [5], is rather low for the individual SRC plot. In that respect, it would be interesting to compare SRCs with other options that may enhance a landscape's multifunctionality. Other options exist with respect to land use, e.g., catch or cover crops to reduce P or sediment export, or management practices, e.g., such as low input and no-tillage agriculture, for the cropland converted to SRCs. This comparison is especially relevant regarding the low deployment of SRCs under current market conditions. Under these current market conditions, SRCs would need to be equally subsidized as the other mentioned options. Beyond, a comparison on the economic efficiency of subsidies for the different options would be of interest.

4.5.2 Transferability of methods and results

From a socio-economic perspective, the assumptions underlying the ABM further determine the transferability of results In the ABM, we assume profit maximization as decision rule. Although this rational and self-interested decision-maker is widely accepted in economics, also non-commercial factors are believed to influence agricultural decisions [41]. In Germany, farmers' interested in SRC practice are provided with advisory material, e.g., the manual of Skodawessely et al. [43], or a profitability calculators provided by different projects, e.g., AGROWOOD [44]. Therefore, we implemented the therein suggested profit maximization determining decision criterion for the ABM. This ABM was developed to model regional deployment of SRCs when this production practice is expected to reach its mature commercial phase. Beyond modeling SRC deployment in the commercial phase, one may equally model the current process of SRC deployment, e.g., to explain the slow deployment of SRCs. This would require additional factors such as practical challenges associated with SRCs and other new land use options, which could be influential [25, 70, 26].

For our ABM, we used a field size of 45 ha in the artificial landscape, resembling the mean field size in eastern Germany [71], without the option to split the area for different agricultural practices. Considering smaller field sizes could change the results; for example, SRCs might become profitable on sites with good soil qualities, where annual production is not profitable due to other reasons. The ABM could be adapted to address pre-mature commercial phases or smaller field sizes.

Under these assumptions, the ABM can be applied to different regions and policy settings described in the following. Within the assumed profit maximization, we focused on the major site-specific influence factors of cropping decisions, i.e., soil quality and distance to CHP plants. From the environmental perspective, this is an appropriate assumption for our study area as all possible arable land is located in a plain area with moderate variation in slope and water holding capacity. In study regions, which fulfill these characteristics, the ABM and its results for the artificial landscape may be applied to assess the economic viability and deployment of SRCs by knowing major implementation parameters, e.g., soil quality or CHP plants' distribution. However, without reapplying the ABM, the results should already be representative for similar German soil climate clusters ("Boden-Klima-Räume"), which comprise large areas of eastern Germany (Saxony and Thuringia (Saxon-Thuringian Hills and Upper Lusatia)) [72]. In addition, this region is one of Germany's major crop production areas. With respect to land use intensity, agricultural companies and field structures in this region remained mostly stable after the re-privatization of the Agricultural Production Cooperatives [73], which allow for comparable highly mechanized land management. Further environmental transfer of the results beyond Germany might be equally feasible with different approaches that control for environmental heterogeneity as developed in Meyer et al. [60].From a policy perspective, increasing demand in the ABM can resemble instruments like the promotional policies under the Renewable Energies Act in Germany and other national laws implementing the EU Renewable Energy Directive

(EU RED) that affect the prices for woody biomass. Furthermore, these market-related changes may also be affected by occurring novel conversion technologies [74]. Beyond the EU RED, other environmental policies such as forest protection policies and reforestation initiatives may also affect biomass demand and regional shifts [75]. In that respect, our approach of coupling an ABM with ESS and environmental assessments may be equally applied to assess impact of changing supply and demand patterns in the forestry sector on regional ESS.

Overall, transferability of results to regions with different biophysical and socio-economic framework conditions is generally given. However, the presented methodology needs to be adapted as suggested above, e.g., by removing the effect of environmental heterogeneity, or by adapting the ABM model for landscape characteristics of potential study regions. Furthermore, the ABM itself can be applied to address different deployment phases by adapting rules of the decision process.

Coupling an ABM with environmental assessment tools such as InVEST and Globio shows several advantages compared to simpler suitability assessments (e.g., Meehan et al. [19] or Tölle et al. [21]). The latter often rely on predefined thresholds, but miss farmers' decision making in the site selection process. Tölle et al. [21] state that "the extent and magnitude of LULCC [land use land cover change] due to renewable energies are not yet quantified for Germany". With our approach, we do not predefine the share of SRCs, but this is an emergent property of the ABM. This enables to apply the same methodology to different settings, such as other decision processes or different economic or policy scenarios, and to investigate how this changes the allocation of SRCs without predefined suitability rules. Suitability rules are most likely hardly transferable even at the national scale due to heterogeneous management and environmental conditions.

4.6 Conclusions

In this study, we quantified how SRCs would affect multiple ESS and biodiversity under different economic and policy-driven scenarios in the Mulde watershed in Central Germany. Only a substantial increase in SRC production areas will considerably reduce food production as well as the provision of regulating services. However, SRCs hardly provide balanced ESS bundles, i.e., do not locally enhance the multifunctionality. Overall, our study showed that coupling agent-based modeling with environmental and ESS assessment models can contribute to a comprehensive impact assessment of currently hardly implemented bioenergy feedstocks by aiming at more realistic spatial deployment that would emerge in potential future scenarios.

4.7 Acknowledgments

The authors would like to thank Erik Gawel for conceptual advice on model design and Christian Hoyer for data preparation.

4.8 References

[1] Gasparatos A, Stromberg P, Takeuchi K. Biofuels, ecosystem services and human wellbeing: Putting biofuels in the ecosystem services narrative. Agriculture Ecosystems & Environment 2011;142(3-4):111-28.

[2] Rathmann R, Szklo A, Schaeffer R. Land use competition for production of food and liquid biofuels: An analysis of the arguments in the current debate. Renewable Energy 2010;35(1):14-22.

[3] IEA. Energy Technology Perspectives 2010: Scenarios and Strategies to 2050. Paris: International Energy Agency (IEA); 2010.

[4] Milner S, Holland RA, Lovett A, Sunnenberg G, Hastings A, Smith P, et al. Potential impacts on ecosystem services of land use transitions to second generation bioenergy crops in GB. GCB Bioenergy 2015.
[5] Holland RA, Eigenbrod F, Muggeridge A, Brown G, Clarke D, Taylor G. A synthesis of the ecosystem services impact of second generation bioenergy crop production. Renewable and Sustainable Energy Reviews 2015;46:30-40.
[6] Tilman D, Socolow R, Foley JA, Hill J, Larson E, Lynd L, et al. Beneficial Biofuels-The Food, Energy, and Environment Trilemma. Science 2009;325(5938):270-1.
[7] Power AG. Ecosystem services and agriculture: tradeoffs and synergies. Philosophical Transactions of the Royal Society B-Biological Sciences 2010;365(1554):2959-71.
[8] Raudsepp-Hearne C, Peterson GD, Bennett EM. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences of the United States of America 2010;107(11):5242-7.
[9] Werling BP, Dickson TL, Isaacs R, Gaines H, Gratton C, Gross KL, et al. Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. Proceedings of the National Academy of Sciences of the United States of America 2014;111(4):1652-7.
[10] Hastings A, Tallis MJ, Casella E, Matthews RW, Henshall PA, Milner S, et al. The technical potential of Great Britain to produce ligno-cellulosic biomass for bioenergy in current and future climates. GCB Bioenergy 2014;6(2):108-22.
[11] Njakou Djomo S, Ac A, Zenone T, De Groote T, Bergante S, Facciotto G, et al. Energy performances of intensive and extensive short rotation cropping systems for woody biomass production in the EU. Renewable and Sustainable Energy Reviews 2015;41(0):845-54.
[12] Rowe RL, Hanley ME, Goulson D, Clarke DJ, Doncaster CP, Taylor G. Potential benefits of commercial willow Short Rotation Coppice (SRC) for farm-scale plant and invertebrate communities in the agri-environment. Biomass & Bioenergy 2011;35(1):325-36.
[13] Sage R, Cunningham M, Boatman N. Birds in willow short-rotation coppice compared to other arable crops in central England and a review of bird census data from energy crops in the UK. Ibis 2006;148:184-97.
[14] Makeschin F. Effects of Energy Forestry on Soils. Biomass & Bioenergy 1994;6(1-2):63-79.
[15] Schmidt-Walter P, Lamersdorf NP. Biomass Production with Willow and Poplar Short Rotation Coppices on Sensitive Areas-the Impact on Nitrate Leaching and Groundwater Recharge in a Drinking Water Catchment near Hanover, Germany. BioEnergy Research 2012;5(3):546-62.
[16] Pe'er G, Dicks LV, Visconti P, Arlettaz R, Baldi A, Benton TG, et al. AGRICULTURE POLICY EU agricultural reform fails on biodiversity. Science 2014;344(6188):1090-2.
[17] Drossart I, Mühlenhoff J. Holzenergie Bedeutung, Potenziale, Herausforderungen. Renews Spezial: Agentur für Erneuerbare Energien; 2013.
[18] Kraxner F, Nordström E-M, Havlík P, Gusti M, Mosnier A, Frank S, et al. Global bioenergy scenarios – Future forest development, land-use implications, and trade-offs. Biomass and Bioenergy 2013;57(0):86-96.
[19] Meehan TD, Gratton C, Diehl E, Hunt ND, Mooney DF, Ventura SJ, et al. Ecosystem-Service Tradeoffs Associated with Switching from Annual to Perennial Energy Crops in Riparian Zones of the US Midwest. PLoS ONE 2013;8(11):e80093.
[20] Aust C, Schweier J, Brodbeck F, Sauter UH, Becker G, Schnitzler JP. Land availability and potential biomass production with poplar and willow short rotation coppices in Germany. GCB Bioenergy 2014;6(5):521-33.
[21] Tölle MH, Gutjahr O, Busch G, Thiele JC. Increasing bioenergy production on arable land: Does the regional and local climate respond? Germany as a case study. Journal of Geophysical Research: Atmospheres 2014;119(6):2711-24.

[22] Asbjornsen H, Hernandez-Santana V, Liebman M, Bayala J, Chen J, Helmers M, et al. Targeting perennial vegetation in agricultural landscapes for enhancing ecosystem services. Renewable Agriculture and Food Systems 2014;29(2):101-25.

[23] Le QB, Seidl R, Scholz RW. Feedback loops and types of adaptation in the modelling of land-use decisions in an agent-based simulation. Environmental Modelling & Software 2012;27-28:83-96.

[24] Parker DC, Hessl A, Davis SC. Complexity, land-use modeling, and the human dimension: Fundamental challenges for mapping unknown outcome spaces. Geoforum 2008;39(2):789-804.

[25] Glithero NJ, Wilson P, Ramsden SJ. Prospects for arable farm uptake of Short Rotation Coppice willow and miscanthus in England. Applied Energy 2013;107(0):209-18.

[26] Sherrington C, Bartley J, Moran D. Farm-level constraints on the domestic supply of perennial energy crops in the UK. Energy Policy 2008;36(7):2504-12.

[27] Holland JH. Complex adaptive systems. Daedalus - Journal of the American Academy of Arts and Science 1992;121:17-30.

[28] Lupp G, Steinhäußer R, Starick A, Gies M, Bastian O, Albrechtb J. Forcing Germany's renewable energy targets by increased energy crop production: A challenge for regulation to secure sustainable landuse practices. Land Use Policy 2014;36:296-306.

[29] Regulation (EU) No 1307/2013. Regulation (EU) No 1307/2013 of the European Parliament and of the Council of 17 December 2013 establishing rules for direct payments to farmers under support schemes within the framework of the common agricultural policy and repealing Council Regulation (EC) No 637/2008 and Council Regulation (EC) No 73/2009 Off J EU 2013. L 347: 608-70.

[30] Koellner T, Geyer R. Global land use impact assessment on biodiversity and ecosystem services in LCA. The International Journal of Life Cycle Assessment 2013;18(6):1185-7.

[31] Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65(0):151-69.

[32] Jäckel G, Zink M, Marx A, Aufbereitung von gemessenen und simulierten Klimadaten, 2012, Mitteldeutsches Klimabüro, Leipzig.

[33] Altermann M, Ruske R. Die Region Leipzig-Halle-Bitterfeld: Geologic Lund Borden. In: Feldmann RH, K.; Auge, H.; Flachowsky, J.; Klotz, S.; Krönert, R., editor. Regeneration und nachhaltige Landnutzung - Konzepte für belastete Regionen: Springer, Berlin, Heidelberg; 1997.

[34] TU Dresden/AgroForNet. Kurzumtriebsplantagen in Sachsen. 2013 [cited 2015 April 30]; Available from: http://www.energieholz-portal.de/257-0-KUP-in-Sachsen.html.

[35] Das S, Eichhorn M, Hopffgarten MV, Lang E, Priess J, Thrän D. Spatial Analysis of the Potential of District Heating from Existing Bioenergy Installations in Germany. 20th European Biomass Conference and Exhibition; 2012; Milan: ETA-Florence Renewable Energies.

[36] European Environment Agency (EEA), Corine Land Cover 2006 raster data, 2013, European Environment Agency.

[37] Wochele S, Priess J, Thrän D, O'Keeffe S. Crop allocation model "CRAM" - an approach for dealing with biomass supply from arable land as part of a life cycle inventory. 22nd European Biomass Conference and Exhibition; 2014; Hamburg: ETA-Florence Renewable Energies.

[38] Weise H. Land use change in the context of bioenergy production: impact assessment using agent-based modelling. University of Osnabrück; 2014.

[39] Grimm V, Berger U, Bastiansen F, Eliassen S, Ginot V, Giske J, et al. A standard protocol for describing individual-based and agent-based models. Ecological Modelling 2006;198(1–2):115-26.

[40] Grimm V, Berger U, DeAngelis DL, Polhill JG, Giske J, Railsback SF. The ODD protocol: A review and first update. Ecological Modelling 2010;221(23):2760-8.

[41] Renting H, Rossing WAH, Groot JCJ, Van der Ploeg JD, Laurent C, Perraud D, et al. Exploring multifunctional agriculture. A review of conceptual approaches and prospects for an integrative transitional framework. Journal of Environmental Management 2009;90:S112-S23.

[42] Brown C, Bakam I, Smith P, Matthews R. An agent-based modelling approach to evaluate factors influencing bioenergy crop adoption in North East Scotland. GCB Bioenergy 2015.
[43] Skodawessely, Pretzsch, Bemmann. Beratungshandbuch zu KUP: Eigenverlag der TU Dresden; 2010.
[44] AGROWOOD. 2015 [cited 2015 July 13]; Available from: http://www.agrowood.de/ergebnisse.php.
[45] Engelkamp P, Sell F. Einführung in die Volkswirtschaftslehre: Springer, Berlin, Heidelberg; 2007.
[46] Mankiw NG, Taylor MP. Economics: Thomson Learning Services, Toronto; 2006.
[47] Thober S, Mai J, Zink M, Samaniego L. Stochastic temporal disaggregation of monthly precipitation for regional gridded data sets. Water Resources Research 2014;50(11):8714-35.
[48] LfULG, Auswertekarten Bodenschutz 1:50.000, 2012, Saxon State Office for Environment Agriculture and Geology, Freiberg.
[49] FNR. Basisdaten Bioenergie Deutschland. Gülzow: Fachagentur Nachwachsende Rohstoffe e. V. ; 2013.
[50] Matzenberger J, Kranzl L, Tromborg E, Junginger M, Daioglou V, Sheng Goh C, et al. Future perspectives of international bioenergy trade. Renewable and Sustainable Energy Reviews 2015;43(0):926-41.
[51] StaLa Sachsen. Bodennutzung und Ernte im Freistaat Sachsen - Feldfrüchte, Obst, Wein und Gemüse 2006. Statistische Berichte - Land- und Forstwirtschaft, Fischerei. Kamenz: Statistisches Landesamt des Freistaates Sachsen; 2007.
[52] TLS. Ernte- und Betriebsberichterstattung - Feldfrüchte und Grünland in Thüringen - 2006. Statistische Bericht. Erfurt: Thüringer Landesamt für Statistik; 2007.
[53] BGR, Ackerbauliches Ertragspotential der Böden in Deutschland 1 : 1 000 000, 2014, Bundesanstalt für Geowissenschaften und Rohstoffe, Hannover.
[54] Ali W. Modelling of Biomass Production Potential of Poplar in Short Rotation Plantations on Agricultural Lands of Saxony, Germany. Dresden University of Technology; 2009.
[55] Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K. Guidelines for National Greenhouse Gas Inventories. National Greenhouse Gas Inventories Programme. Institute for Global Environmental Strategies; 2006.
[56] Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford: Oxford University Press; 2011.
[57] Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron D, et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 2009;7(1):4-11.
[58] Tallis HT, Ricketts T, Guerry AD, Wood SA, Sharp R, Nelson E, et al. InVEST 2.5.6 User’s Guide. Stanford; 2013.
[59] Wischmeier WH, Smith DD. Predicting rainfall erosion losses: A guide to conservation planning. Washington, DC; 1978.
[60] Meyer MA, Seppelt R, Witing F, Priess JA. How can we compare environmental and ecosystem service assessments for biomass production systems in different parts of the world? Environmental Research Letters under review.
[61] Alkemade R, van Oorschot M, Miles L, Nellemann C, Bakkenes M, ten Brink B. GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss. Ecosystems 2009;12(3):374-90.
[62] Mouchet MA, Lamarque P, Martin-Lopez B, Crouzat E, Gos P, Byczek C, et al. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change-Human and Policy Dimensions 2014;28:298-308.
[63] R Development Core Team. R: A language and environment for statistical computing. 2009 [cited 2013 November 05]; Available from: http://www.r-project.org/.

[64] Manning P, Taylor G, Hanley EM. Bioenergy, Food Production and Biodiversity – An Unlikely Alliance? GCB Bioenergy 2015;7(4).
[65] Fürst C, Frank S, Witt A, Koschke L, Makeschin F. Assessment of the effects of forest land use strategies on the provision of ecosystem services at regional scale. Journal of Environmental Management 2013;127:S96-S116.
[66] Hellmann F, Verburg PH. Spatially explicit modelling of biofuel crops in Europe. Biomass and Bioenergy 2011;35(6):2411-24.
[67] Kocoloski M, Griffin WM, Matthews HS. Impacts of facility size and location decisions on ethanol production cost. Energy Policy 2011;39(1):47-56.
[68] Vanloocke A, Bernacchi CJ, Twine TE. The impacts of Miscanthus x giganteus production on the Midwest US hydrologic cycle. Global Change Biology Bioenergy 2010;2(4):180-91.
[69] Meyer MA, Chand T, Priess JA. Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLoS ONE 2015;10(3):e0116336.
[70] Sattler C, Nagel UJ. Factors affecting farmers' acceptance of conservation measures - A case study from north-eastern Germany. Land Use Policy 2010;27:70-7.
[71] Fischer C, Flohre A, Clement LW, Batáry P, Weisser WW, Tscharntke T, et al. Mixed effects of landscape structure and farming practice on bird diversity. Agriculture, Ecosystems & Environment 2011;141(1–2):119-25.
[72] Roßberg D, Michel V, Graf R, Neukampf R. Definition von Boden-Klima-Raumen fur die Bundesrepublik Deutschland. Nachrichtenblatt des deutschen Pflanzenschutzdienstes 2007;59(7):155-61.
[73] Blumöhr T, Brandl M, Breitenfeld J, Dahl S, Führer J, Gabka D, et al. Agrarstrukturen in Deutschland - Einheit in Vielfalt - Regionale Ergebnisse der Landwirtschaftszählung. In: Federal Statistical Office and the statistical offices of the Länder, editor. Stuttgart: Statistisches Landesamt Baden-Württemberg; 2011.
[74] Edel M, Thraen D. The Economic Viability of Wood Energy Conversion Technologies in Germany. International Journal of Forest Engineering 2012;23(2):102-13.
[75] Meyfroidt P, Lambin EF, Erb K-H, Hertel TW. Globalization of land use: distant drivers of land change and geographic displacement of land use. Current Opinion in Environmental Sustainability 2013;5(5):438-44.
[76] Müller B, Bohn F, Dreßler G, Groeneveld J, Klassert C, Martin R, et al. Describing human decisions in agent-based models – ODD + D, an extension of the ODD protocol. Environmental Modelling & Software 2013;48:37-48.
[77] Lehner B, Verdin K, Jarvis A. New Global Hydrography Derived From Spaceborne Elevation Data. Eos, Transactions American Geophysical Union 2008;89(10):93-4.
[78] LfULG, Potentielle natürliche Vegetation (pnV) in Sachsen, 2011, Saxon State Office for Environment Agriculture and Geology, Freiberg.
[79] FAO Geonetwork. Global map of monthly reference evapotranspiration - 10 arc minutes (GeoLayer). Rome, Italy2014 [cited 2014 January 14]; Available from: http://www.fao.org/geonetwork/srv/en/main.home.
[80] Panagos P, Van Liedekerke M, Montarella L, The European Soil Database distribution version 2.0, 2006, European Commission/European Soil Bureau Network, Ispra.
[81] Bräunig A, Dokumentation zur Berechnung und Ableitung R-Faktor Sachsen, 2013, LfULG, Freiberg.
[82] Bischoff R, K-Faktor (MMK 100), 2014, TLUG, Jena.
[83] Reckhow KH, Beaulac M, Simpson J. Modeling Phosphorous Loading and Lake Response under Uncertainty: A Manual and Compilation of Export Coefficients Washington, DC; 1980.
[84] SMUL. Erosionsminderung. 2013 [cited 2014 August 10]; Available from: http://www.umwelt.sachsen.de/umwelt/4566.asp.

[85] Fraver S, Wagner RG, Day M. Dynamics of coarse woody debris following gap harvesting in the Acadian forest of central Maine, U.S.A. Canadian Journal of Forest Research 2002;32(12):2094-105.
[86] Bohn U, Hettwer C, Gollub G. Anwendung und Auswertung der Karte der natürlichen Vegetation Europas : Beiträge und Ergebnisse des internationalen Workshops auf der Insel Vilm. Bonn: Deutschland / Bundesamt für Naturschutz; 2005.
[87] Müller-Using S, Bartsch N. Decay dynamic of coarse and fine woody debris of a beech (Fagus sylvatica L.) forest in Central Germany. European Journal of Forest Research 2009;128(3):287-96.
[88] Wördehoff R, Spellmann H, Evers J, Nagel J. Kohlenstoffstudie Forst und Holz Niedersachsen. Göttingen: Universitätsverlag Göttingen; 2011.
[89] Polley H, Henning P. Die Bundeswaldinventur in der Geodateninfrastruktur des Thünen-Instituts (BWI 2002). Eberswalde2012 [cited 2014 August 15]; Available from: https://gdi.ti.bund.de/geoserver/bwi_2009dt/ows?SERVICE=WFS&REQUEST=GetCapabilities.
[90] Strogies M, Gniffke P. Berichterstattung unter der Klimarahmenkonvention der Vereinten Nationen und dem Kyoto-Protokoll 2012 - Nationaler Inventarbericht zum Deutschen Treibhausgasinventar 1990 – 2010. Dessau: Umweltbundesamt; 2012.
[91] BGR, Nutzungsdifferenzierte Bodenübersichtskarte der Bundesrepublik Deutschland 1:1.000.000 (BÜK1000N) (Serie), 2013, Bundesamt für Geowissenschaften und Rohstoffe, Hannover.
[92] Priess JA, Heinze M, Egli L, Pomm S, Lang E, Masurowski F, et al. Assessing some of the social and environmental consequences of changes in population and settlement pattern expected until 2050 for a Central European region. Environmental Modelling & Software under review.
[93] BKG, Digitales Basis-Landschaftsmodell - Basis-DLM, 2014, Bundesamt für Kartographie und Geodäsie, Leipzig.
[94] Builtjes P, Hendriks E, Koenen M, Schaap M, Banzhaf S, Kerschbaumer A, et al., Abschlussbericht zum UFOPLAN-Vorhaben FKZ 3707 64 200: Erfassung, Prognose und Bewertung von Stoffeinträgen und ihren Wirkungen in Deutschland (Modelling of Air Pollutants and Ecosystem Impact – MAPESI). 2011, Federal Environmental Agency, Dessau-Rosslau.
[95] Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, et al. Terrestrial Ecoregions of the World: A New Map of Life on Earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. BioScience 2001;51(11):933-8.

4.9 Supporting Information

4.9.1 S1 Text: Full model description of ABM

This document contains the full description of the agent-based model (ABM). The ABM is an extended version of the model described in Weise [1]. Here, the model description follows the ODD+D protocol [2]. As an extension of the widely used ODD protocol [3, 4], the ODD+D protocol puts particular focus on the presentation of human decision-making.

Purpose

The model has been developed to understand the determinants and impacts of the expansion of short rotation coppices (SRCs). The aim of the presented study was to transfer derived insights to the Mulde watershed in Central Germany and to analyse impacts of SRCs on multiple ecosystem services and biodiversity.

Entities, state variables and scales

The model contains following entities: individual land users as uniform type of agents, the landscape, grids cells as spatial units and economic markets of different agricultural products as institutions. Table 4.S1 gives an overview on model entities and associated parameters and state variables.

Table 4.S1: Entities, parameters and state variables of the model.

Entity	Parameters	State variables
Agent	Location	Profit
	discount rate	
Landscape	Size	shares of land use types
	number of CHP plants	
	mean soil quality	
	spatial correlation of soil quality	
grid cell	Location	land use
	soil quality	
	distances to CHP plants	
market for annual agricultural crops	Demand	price
		supply
market for SRC	Demand	price
		supply

Exogenous drivers of land use decisions, which are not influenced by processes during a model run, are: soil qualities, demands, number and location of combined heat and power (CHP) plants. Space is explicitly considered in the model by an abstract landscape that consists of a regular grid. One grid cell represents approximately 45 ha and the model landscape comprises 50x50 grid cells. Each cell is occupied by one agent who stays in that cell for the whole simulation and decides in each time step on the land use type in that cell. One time step represents one year and simulations were run for 50 years.

Process overview and scheduling

Here, the processes of the model are briefly specified to allow a general overview of the model and its dynamics. For a detailed description of the processes see section submodels.

Process 1: Initialization of landscape

Before the start of the simulation the landscape, i.e., distribution of soil quality and spatial allocation of CHP plants, is randomly generated.

Process 2: Decision-making of agents

In each time step all agents decide sequentially in a random order between three different land use types: no cultivation, annual agricultural crops for food or feed production, and SRCs.

Process 3: Calculation of regional supply and market prices

After each agent decision, the market prices for the different commodities (woody products from SRCs and other agricultural crops from annual cultivation) is updated based on the exogenously given demands and current regional supply.

Design concepts

Theoretical and empirical background

A main concept is the implementation of markets for the different agricultural commodities (woody products from SRCs and other agricultural crops from annual cultivation). Markets prices in the model are determined by supply and demand, a standard assumption in economic theory [5, 6]. At this, market price is determined by an externally given demand and a supply that is solely generated by the land use decisions of the agents (i.e., endogenous markets). The demands for products from SRCs as well as for other agricultural crops are fixed (i.e., price insensitive).

The agents are rational profit maximizers, i.e., agents have clear preferences over all possible land use options and aim at maximizing their income. At this, it is assumed that agents have full information (that are available in the model) and the needed cognitive ability to process all possible options. This decision model was chosen to be close to established theory. Furthermore, we believe that profit maximization is an appropriate assumption for industrial agricultural decisions. To enable the comparison between land use types with different lifespans, the equivalent annual annuity approach (see for example Brigham and Houston [7]) from investment theory was chosen. This approach is appropriate as it is often recommended to land users interested in SRC practice in Germany (for example Schweinle and Franke [8]) and has been used in several studies on the financial analysis of SRCs [9].

Individual decision-making

The agents, namely individual farmers, decide between three different land use types: no cultivation, annual crops for food or feed production and SRCs. Agents follow a rational profit maximization approach using an equivalent annual annuity approach (see above). Agents adapt to changing market prices. Neither social norms nor cultural values are incorporated in the model. Spatial aspects are incorporated as the distance to CHP plants influences the decision via resulting transportation costs. Temporal aspects play a role as discounting of future profits is incorporated in the equivalent annual value approach. Discount rates are seen as subjective discount rates which can vary depending on personal risk aversion [10].

Learning

Learning is not incorporated in the model.

Individual sensing

The land user knows its current land use, the location factors of its land, i.e., soil quality and distance to CHP plants, and the current market price. The agents do not perceive information from other agents directly. However, they do sense the current total supply which comprises all agents currently chosen land use type. The sensing process is not modelled explicitly, it is not erroneous and no costs of sensing are incorporated.

Individual prediction

Agents are not able to predict changes in market prices. However, they are able to predict how their own decision will impact the current market price.

Interaction

Agents interact indirectly via the endogenous market. The land use decision of an agent influences the market prices, which then influences the land use decisions of other agents.

Collectives

There are no collectives incorporated in the model.

Heterogeneity

Agents are heterogeneous with regard to the location factors of their land. Here, the soil qualities as well as the distances to CHP plants differ between cells. Soil quality influences productivity of annual crops as well as of SRCs. Distances to CHP plants determine transportation costs SRC products.

Stochasticity

Agents make their decisions in a random order.

Observation

With the goal of this study to transfer results on the SRC distribution to the Mulde watershed in Central Germany, conditional probabilities of SRC occurrence based on location factors are observed as main output variable.

Implementation details

The model was implemented in C++ using Embarcadero® C++Builder® 2010.

Initialization

The model is initialized with 2500 agents and cells. At the beginning, all cells are not under cultivation. A fixed number of CHP plants is randomly placed within the landscape. The distribution of soil qualities is randomly generated with a fixed mean soil quality and a fixed spatial correlation (see section submodels for details). The initial landscape, i.e., CHP plants and soil qualities, varies between the simulations.

Input

No input data is used.

Submodels

Initialization of landscape

At the beginning of each simulation, the underlying landscape is randomly generated. Soil qualities are assigned to cells and CHP plants are spatially allocated within the landscape. Based on this underlying landscape, locations for SRCs in dependence on soil quality and distance to CHP plants, are simulated with the ABM. These results need to be transferred to the real landscape to assess impacts of SRC expansion in a specific case study, here the Mulde watershed. Therefore, the underlying landscape for the ABM, i.e., distribution of soils qualities and number of CHP plants, need to fulfill statistical characteristics present in the Mulde watershed.

Distribution of soil qualities

The distribution of soil qualities for the ABM is generated using a random generator that returns uniformly distributed, spatially correlated numbers with fixed arithmetic mean and correlation length. For this purpose, the standard method of Cholesky decomposition is used (see Appendix A in Thober et al. [11] for details). Using the Cholesky factor in the generation of the random field guarantees that the covariance among all cells is considered. The soil quality distribution of the generated landscape was adapted to the soil quality distribution in the Mulde watershed. (1) We set the arithmetic mean from the generated landscape to the one in the Mulde watershed. (2) We divide the continuous soil qualities into 6 classes by a mean preserving procedure with the same areal shares as given in the data, as data on soil quality are available in 6 classes [12]. (3) We set the correlation length by adapting spatial correlation (using Moran’s I up to a neighborhood of 49*49 cells) of the generated landscapes to the real landscape.

Spatial allocation of infrastructures

A fixed number of CHP plants is randomly located within the landscape, with no further constraints. The number of CHP plants is set according to the areal density in the Mulde watershed given by the data described in Das et al. [13].

Decision-making of agents

The agent chooses between three land use types: SRCs, annual agricultural crops (ANN) or no cultivation (NoC). From these, the agent chooses the option that maximizes its profits. The profit calculation differs between the three land use types. No cultivation yields neither costs nor revenue and its profit for agent *i* is therefore:

$$P_{NoC}^{i} = 0 \; \forall i \tag{1}$$

For annual agricultural crops the following profit function applies:

$$P_{ANN}^{i}(t) = p_{ANN}(t) * prod_{ANN}^{i} - pca \tag{2}$$

where $p_{ANN}(t)$ is the current market price (calculated by equation 10), $prod_{ANN}^{i}$ the productivity of annuals crops in the cell of agent i and pca the production costs of annuals. The productivity of annual crops is determined by the location factor soil quality by assuming a linear relationship:

$$prod_{ANN}^{i} = sq^{i} \in [0,1] \tag{3}$$

where sq^{i} is the soil quality of the cell of agent i. As pointed out by Zhang et al. [14] soil properties strongly impact the agricultural output. Similarly, we assume that productivity and soil quality are positively correlated. Thereby, we follow the concept of using soil values ("Bodenwertzahl") to classify German soils [15]. The soil value is a measure for differences in net yield under proper cultivation that are solely determined by differences in soil [15]. With equation 3, a soil value sq^{i} of 0.5 represents a reduction in net yield by 50% of the maximal yield [16, 14].

As SRCs represent long-term investment decisions, concepts of intertemporal choice should be taken into account in the profit calculation. The underlying idea is that people value profit differently at different points in time. For this study, the equivalent annual annuity approach (for example described in Brigham and Houston [7]) from investment theory was chosen. This approach is appropriate as it is often recommended to land users interested in SRC practice in Germany (for example Schweinle and Franke [8]) and has been used in several studies on the financial analysis of SRCs [9].

In a first step, the profit of agent *i* in year *t* $P_{SRC}^{i}(t)$ over the whole life time of the SRC is calculated by:

$$P_{SRC}^{i}(t) = \begin{cases} p_{SRC}(t) * prod_{SRC}^{i} * rot - costs(t), if\ t \bmod rot = 0 \\ - costs(t), else \end{cases} \tag{4}$$

where $p_{SRC}(t)$ is the current market price in year t for SRC products produced in one year on optimal soil conditions calculated by equation 10, $prod^{i}$ the productivity of SRCs in the cell of agent i, *rot* the number of years after which SRCs are harvested and $costs(t)$ are all incurring costs in year t. The productivity of SRCs is given by:

$$prod^{i} = \begin{cases} prod_{min} + 0.2, if\ sq^{i} \geq 0.5 \\ prod_{min}, if\ sq^{i} < 0.5 \end{cases} \tag{5}$$

where $prod_{min}$ is the productivity on cells with a soil quality sq^i below 0.5. Hence, the productivity of SRCs is assumed to decrease on poor soils (as was found by Ali [17]). At this, the dependence on soil quality is less pronounced than for annual crops (see equation 3) because studies showed that biomass yield from SRCs is dependent on further factors such as age of plantation or precipitation [17]. Nevertheless, in our model the consideration of biophysical location factors is restricted to the soil quality due to simplicity reasons.

Finally, all occurring costs are calculated by:

$$costs(t) = \begin{cases} ic, if\ t = 0 \\ hc + rot * tc, if\ t\ mod\ rot = 0 \\ hc + rot * tc + rc, if\ t = LT \\ 0,\ else \end{cases} \tag{6}$$

where rot is the number years after which SRCs are harvested, ic are the investment costs, hc the harvest costs, tc the transportation costs of wood produced per year and rc the recovery costs. In the initial year the investment costs ic are due, at the end of each rotation cycle harvest costs hc as well as transportation costs to the CHP plant tc occur and finally at the end of the lifetime additional recovery costs of the land rc have to be paid. The transportation costs are linearly dependent on the distance to CHP plant:

$$tc(d) = (tc_{min} + tc_{slope} * d) * prod_{SRC} * yield \tag{7}$$

where d is the distance to the CHP plant calculated as Euclidean distance [18], tc_{min} are the transport costs for minimal distance, tc_{slope} the slope with which transport costs are increasing with increasing distance, $prod^i$ the productivity of SRCs in the cell of agent i and $yield$ is the yield of SRC products produced in one year on optimal soil conditions.

From the sequence of profits $P^i_{SRC}(t)$, the net present value is calculated as the sum of the discounted profits:

$$NPV^i = \sum_{t=0}^{LT} (1 + r)^{-t} * P^i_{SRC}\,(t) \tag{8}$$

where LT is the lifetime of the plantation, r the discount rate and $P^i_{SRC}(t)$ the profit in year t calculated by equation 4.

Subsequently, the equivalent annual value EAV is calculated from the net present value NPV to enable the comparison of land use options with unequal lifespans:

$$EAV^i = \frac{1}{1-(1+r)^{-LT}} * NPV^i \tag{9}$$

where r is the discount rate, LT the lifetime of a SRC plantation and NPV^i the net present value calculated by equation 8.

In a final step, the agent compares the equivalent annual value EAV^i with the possible profit from annual agricultural production $P^i_{ANN}(t)$ and chooses the option with the higher profit. If both, the equivalent annual value EAV^i of SRC and the profit of annual agricultural crops $P^i_{ANN}(t)$ would yield negative incomes, the agent decides to not cultivate its land in the current year.

Calculation of regional supply and market prices

After each decision step, the regional supplies $S_j(t)$ and the market prices $p_j(t)$ for the different products j, i.e., ANN and SRC, are updated by calculating:

$$p_j(t) = \frac{D_j}{S_j(t)} \text{ with } S_j(t) = \sum_{i=1}^{N} h^i_j(t) \quad (10)$$

where D_j is the demand for product $j \in \{ANN, SRC\}$, N the number of agents and $h^i_j(t)$ the harvest amount of product j in cell i given by:

$$h^i_{ANN}(t) = \begin{cases} prod^i_{ANN}, \text{ if land use is ANN} \\ 0, \text{ if land use is not ANN} \end{cases} \quad (11)$$

$$h^i_{SRC}(t) = \begin{cases} prod^i_{SRC}, \text{ if land use is SRC} \\ 0, \text{ if land use is not SRC} \end{cases} \quad (12)$$

Parameter set

Table 4.S2 shows the names of all parameters used in the model, their values and, if available, the references for their parameterization.

Table 4.S2: Parameters of the model.

Parameter	Symbol	Value	Unit	Reference
Technical parameters				
Number of agents	N	2500	-	-
Number of time steps	T	50	Years	-
Agent				
Discount rate	r	6%	-	average value of discount rates used in SRC studies included in a review by

				Kasmioui and Ceulemans [9]
Field size	fs	~45	Ha	according to mean field sizes in East Germany [19]
Landscape				
Mean soil quality		0.62	-	chosen based on mean soil quality in case study [12]
Correlation length	ρ	0.05	-	chosen by adapting spatial correlation of the generated landscapes to the real landscape [12]
Number of CHP plants	if	2	-	chosen based on total expansion of landscape and number per area based on Das et al. [13]
Annual crops				
Demand for annual crops	D_{ANN}	11000	money units per year and ha	chosen based on initial shares of agricultural and fallow land currently present in case study [20, 21]
Production costs per ha	pca	2.4	money units per ha	Landwirtschaftskammer Niedersachsen [22]
SRCs				
Demand for SRC products	D_{SRC}	140	money units per year and ha	chosen based on current demand in case study (solely given by CHP plants present) [13]
Investment costs per ha	ic	4.7	money units	Schweinle and Franke [8], Wagner et al. [23]
Recovery costs per ha	rc	3.6	money units per ha	Schweinle and Franke [8], Wagner et al. [23]
Harvest costs per ha	hc	1.6	money unit per ha	Schweinle and Franke [8], Wagner et al. [23]
Rotation cycle	rot	4	Years	Aylott et al. [24], Hillier et al. [25]

Lifetime	LT	20	Years	maximal number of years to not count as forest [26]
Minimal productivity	$prod_{min}$	0.8	-	-
Minimal transportation costs per dry ton	tc_{min}	0.02	money units per dry ton	linear regression based on values from Kröber et al. [27], Strohm et al. [28], Wagner et al. [23]
Slope transportation costs	tc_{slope}	0.001	money units per dry ton and distance	linear regression based on values from Kröber et al. [27], Strohm et al. [28], Wagner et al. [23]
Yield per ha	$yield$	12	dry tons per ha	Aust et al. [29]

References

[1] Weise H. Land use change in the context of bioenergy production: impact assessment using agent-based modelling. University of Osnabrück; 2014.
[2] Müller B, Bohn F, Dreßler G, Groeneveld J, Klassert C, Martin R, et al. Describing human decisions in agent-based models – ODD + D, an extension of the ODD protocol. Environmental Modelling & Software 2013;48:37-48.
[3] Grimm V, Berger U, Bastiansen F, Eliassen S, Ginot V, Giske J, et al. A standard protocol for describing individual-based and agent-based models. Ecological Modelling 2006;198(1–2):115-26.
[4] Grimm V, Berger U, DeAngelis DL, Polhill JG, Giske J, Railsback SF. The ODD protocol: A review and first update. Ecological Modelling 2010;221(23):2760-8.
[5] Engelkamp P, Sell F. Einführung in die Volkswirtschaftslehre: Springer, Berlin, Heidelberg; 2007.
[6] Mankiw NG, Taylor MP. Economics: Thomson Learning Services, Toronto; 2006.
[7] Brigham E, Houston J. Fundamentals of Financial Management: Cengage Learning; 2006.
[8] Schweinle J, Franke E. Beratunsghandbuch zu Kurzumtriebsplantagen. In: Skodawessely PB, editor: Eigenverlag der TU Dresden; 2010.
[9] Kasmioui OE, Ceulemans R. Financial analysis of the cultivation of poplar and willow for bioenergy. Biomass and Bioenergy 2012;43(0):52-64.
[10] Barberis N, Thaler R. Handbook of the Economics of Finance. In: Constantinides GM, Harris M, Stulz R, editors: Elsevier Science; 2003.
[11] Thober S, Mai J, Zink M, Samaniego L. Stochastic temporal disaggregation of monthly precipitation for regional gridded data sets. Water Resources Research 2014;50(11):8714-35.
[12] LfULG, Auswertekarten Bodenschutz 1:50.000, 2012, Saxon State Office for Environment Agriculture and Geology, Freiberg.
[13] Das S, Eichhorn M, Hopffgarten MV, Lang E, Priess J, Thrän D. Spatial Analysis of the Potential of District Heating from Existing Bioenergy Installations in Germany. 20th European Biomass Conference and Exhibition; 2012; Milan: ETA-Florence Renewable Energies.
[14] Zhang W, Ricketts TH, Kremen C, Carney K, Swinton SM. Ecosystem services and dis-services to agriculture. Ecological Economics 2007;64(2):253-60.

[15] GD NRW. Wertzahlen der Bodenschätzung. 2014 [cited 2015 April 15]; Available from: www.gd.nrw.de.

[16] Petzold R, Butler-Manning D, Feldwisch N, Glaser T, Schmidt PA, Denner M, et al. Linking biomass production in short rotation coppice with soil protection and nature conservation. Iforest-Biogeosciences and Forestry 2014;7:353-62.

[17] Ali W. Modelling of Biomass Production Potential of Poplar in Short Rotation Plantations on Agricultural Lands of Saxony, Germany. Dresden University of Technology; 2009.

[18] Deza MM, Deza E. Encyclopedia of Distances: Springer, Berlin, Heidelberg; 2013.

[19] Fischer C, Flohre A, Clement LW, Batáry P, Weisser WW, Tscharntke T, et al. Mixed effects of landscape structure and farming practice on bird diversity. Agriculture, Ecosystems & Environment 2011;141(1–2):119-25.

[20] European Environment Agency (EEA), Corine Land Cover 2006 raster data, 2013, European Environment Agency.

[21] Wochele S, Priess J, Thrän D, O'Keeffe S. Crop allocation model "CRAM" - an approach for dealing with biomass supply from arable land as part of a life cycle inventory. 22nd European Biomass Conference and Exhibition; 2014; Hamburg: ETA-Florence Renewable Energies.

[22] Landwirtschaftskammer Niedersachsen. Maispreisrechner. 2014 [cited 2015 April 16]; Available from: http://www.lwk-niedersachsen.de/index.cfm/portal/betriebumwelt/nav/360/article/17878.html.

[23] Wagner P, Schweinle J, Setzer F, Kröber M, Dawid M. DLG-Standard zur Kalkulation einer Kurzumtriebsplantage. DLG-Merkblatt 372; 2012.

[24] Aylott MJ, Casella E, Tubby I, Street NR, Smith P, Taylor G. Yield and spatial supply of bioenergy poplar and willow short-rotation coppice in the UK. New Phytologist 2008;178:358–70.

[25] Hillier J, Whittaker C, Dailey G, Aylott M, Casella E, Richter G, et al. Greenhouse gas emissions from four bioenergy crops in England and Wales: Integrating spatial estimates of yield and soil carbon balance in life cycle analyses. GCB Bioenergy 2009;1(4):267-81.

[26] BWldG Gesetz zur Erhaltung des Waldes und der Förderung der Forstwirtschaft. Bundesministerium für Justiz und Verbraucher, 2010.

[27] Kröber M, Becker R, Reike J, Wolf H. Beratunsghandbuch zu Kurzumtriebsplantagen. In: Skodawessely PB, editor: Eigenverlag der TU Dresden; 2010.

[28] Strohm K, Schweinle J, Liesebach M, Osterburg B, Rödl A, Baum S, et al. Kurzumtriebsplantagen aus ökologischer und ökonomischer Sicht. Arbeitsberichte aus der VTI-Agrarökonomie, No. 06/2012; 2012.

[29] Aust C, Schweier J, Brodbeck F, Sauter UH, Becker G, Schnitzler JP. Land availability and potential biomass production with poplar and willow short rotation coppices in Germany. GCB Bioenergy 2014;6(5):521-33.

4.9.2 S2 File: Tables supporting Fig. 4.8

Table 4.S3: Provisioning ESS values for the economic (scenarios 1-4) and the policy-driven scenarios (scenarios 5-6) are indicated compared to the baseline scenario (italics).

Provisioning ecosystem services	SRC yield	Cereals		Maize silage		Rapeseed	
	[t a^{-1}]	[t a^{-1}]	Δ baseline scenario	[t a^{-1}]	Δ baseline scenario	[t a^{-1}]	Δ baseline scenario
Baseline scenario (2006)	*0*	*621,068*		*787,037*		*127,765*	
Scenario 1: standard demand	50,085	598,641	-4%	757,771	-4%	123,170	-4%
Scenario 2: medium demand	117,178	567,409	-9%	720,617	-8%	115,864	-9%
Scenario 3: high demand	433,996	455,015	-27%	547,316	-30%	88,829	-30%
Scenario 4: very high demand	934,079	307,553	-50%	329,840	-58%	59,357	-54%
Scenario 5: EFA (bad soils)	170,108	543,606	-12%	695,843	-12%	111,159	-13%
Scenario 6: EFA (good soils)	403,301	476,539	-23%	663,341	-16%	105,279	-18%

Table 4.S4: Regulating ESS values for the economic (scenarios 1-4) and the policy-driven scenarios (scenarios 5-6) are indicated compared to the baseline scenario (italics).

Regulating ecosystem services	C storage		Sediment export		P export		Biodiversity	
	[t a^{-1}]	Δ baseline scenario	[t a^{-1}]	Δ baseline scenario	[kg a^{-1}]	Δ baseline scenario	MSA (mean)	Δ baseline scenario
Baseline scenario (2006)	*37,318,089*		*66,871*		*341,343*		*0.126*	
Scenario 1: standard demand	37,158,360	0%	65,533	-2%	336,371	-1%	0.129	2%
Scenario 2: medium demand	37,517,730	1%	63,500	-5%	330,895	-3%	0.134	7%
Scenario 3: high demand	39,346,230	5%	53,891	-19%	318,658	-7%	0.153	22%
Scenario 4: very high demand	41,280,573	11%	41,843	-37%	301,156	-12%	0.174	38%
Scenario 5: EFA (bad soils)	37,826,181	1%	61,969	-7%	326,501	-4%	0.139	11%
Scenario 6: EFA (good soils)	38,305,760	3%	54,989	-18%	333,165	-2%	0.139	10%

4.9.3 S3 File: Tables listing the set of potential explanatory variables for Table 4.3 and Fig. 4.5 and the entire regression results for Fig. 4.5

Table 4.S5: Potential variables to explain ESS cluster differences

Independent variable	Unit	Methodological reference (data source)
Landscape composition		Jones et al. [1]
Share of LU/LC classes in the neighborhood:		European Environment Agency (EEA) [2], Wochele et al. [3]
Forest	[%]	
Pasture	[%]	
SRC	[%]	
Cropland	[%]	
Urban	[%]	
Water	[%]	
Distance to stream	[m]	Kissel et al. [4]
Naturalness		
Urbanity	[*score*]	European Environment Agency (EEA) [2], Wochele et al. [3], Meyer et al. [5]
Topography		Wrbka et al. [6]
Elevation	[m]	Lehner et al. [7]
Slope	[%]	Lehner et al. [7]
Curvature	[*score*]	Lehner et al. [7])
Aspect	[°]	Lehner et al. [7]
Soil parameters		Qiu and Turner [8]
Effective rooting depth	[mm]	Panagos et al. [9], LfULG [10]
Available water holding capacity	[cm cm^{-1}]	Panagos et al. [9], LfULG [10]
Soil quality index ("Ackerzahl")	[*score*]	LfULG [10]
Erodibility (K)	[t ha h ha^{-1} MJ^{-1} mm^{-1}]	LfULG [10], Bischoff [11]
Climate		
Precipitation	[mm a^{-1}]	Jäckel et al. [12]
Reference Evapotranspiration (10 arc-min)	[mm a^{-1}]	FAO Geonetwork [13]
Erosivity (R)	[MJ mm ha^{-1} h^{-1} a^{-1}]	Bräunig [14]

Table 4.S6: Factors characterizing ESS cluster 1 and 2 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 1; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 2. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only an insignificant and small difference (χ^2= 0.98931, df = 3, p= 0.8038).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-4.16	1.325	-3.14	0.00169	**
Topography and soil parameters					
Elevation [m]	-5.4933	1.3285	-4.135	0.0000355	***
Slope [%]	-2.8955	1.4718	-1.967	0.04914	*
Aspect [°]	1.2244	0.3847	3.183	0.00146	**
Curvature [*score*]	-9.3121	1.6165	-5.761	8.37e-09	***
Effective rooting depth [mm]	-6.8905	1.535	-4.489	0.00000716	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	3.3915	0.8047	4.214	0.000025	***
Available water holding capacity [cm cm^{-1}]	8.009	1.7576	4.557	0.00000519	***
Soil quality index	15.0027	1.3344	11.243	< 2e-16	***
Climate					
Precipitation [mm]	2.1837	0.8962	2.437	0.01482	*
Landscape composition					
Pasture, 5 km buffer [*%*]	-1.5396	0.693	-2.222	0.02631	*
SRC, 5 km buffer [*%*]	-3.173	0.4312	-7.358	1.86e-13	***
Water, 5 km buffer [*%*]	-1.3005	0.5975	-2.176	0.02953	*
Distance to stream [m]	0.7343	0.418	1.757	0.07896	.

Table 4.S7: Factors characterizing ESS cluster 1 and 3 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 1; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 3. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only an insignificant and small difference (χ^2= 1.6238, df = 3, p= 0.654).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	12.3201	2.1133	5.83	5.55e-09	***
Topography and soil parameters					
Elevation [m]	-14.3714	2.1349	-6.732	1.68e-11	***
Slope [%]	-12.0602	1.5815	-7.626	2.42e-14	***
Aspect [°]	0.9365	0.4958	1.889	0.058903	.
Curvature [*score*]	-9.8478	1.8065	-5.451	0.00000005	***
Effective rooting depth [mm]	-8.2678	2.1071	-3.924	0.0000871	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	4.8404	1.3888	3.485	0.000491	***
Available water holding capacity [cm cm^{-1}]	-11.5222	4.018	-2.868	0.004135	**
Soil quality index	15.7233	1.8145	8.665	< 2e-16	***
Climate					
Precipitation [mm]	6.2493	1.1237	5.561	2.68e-08	***
Reference Evapotranspiration [mm a^{-1}]	-8.7854	1.2352	-7.113	1.14e-12	***
Landscape composition					
Forest, 5 km buffer [%]	-8.3005	1.396	-5.946	2.75e-09	***
SRC, 5 km buffer [%]	-1.55	0.725	-2.138	0.032525	*
Urban, 5 km buffer [%]	-3.1857	1.055	-3.02	0.00253	**
Water, 5 km buffer [%]	-9.8478	1.8065	-5.451	0.00000005	***

Table 4.S8: Factors characterizing ESS cluster 1 and 4 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 1; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 4. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only an insignificant and small difference (χ^2= 1.6238, df = 3, p= 0.654).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-2.5874	0.9145	-2.829	0.00467	**
Topography and soil parameters					
Slope [%]	-4.7428	0.8784	-5.399	6.69e-08	***
Aspect [°]	1.3759	0.2312	5.951	2.67e-09	***
Curvature [*score*]	-11.1975	1.0019	-11.176	< 2e-16	***
Effective rooting depth [mm]	5.5924	0.778	7.188	6.58e-13	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	1.1685	0.6112	1.912	0.05591	.
Available water holding capacity [cm cm^{-1}]	13.2404	1.5589	8.493	< 2e-16	***
Climate					
Precipitation [mm]	-1.9768	0.4761	-4.152	0.000033	***
Landscape composition					
Forest, 5 km buffer [%]	-1.7254	0.645	-2.675	0.00748	**
Pasture, 5 km buffer [%]	-1.9516	0.4701	-4.152	0.000033	***
SRC, 5 km buffer [%]	-0.7229	0.2847	-2.539	0.01112	*
Urban, 5 km buffer [%]	-1.6837	0.4214	-3.995	0.0000647	***
Water, 5 km buffer [%]	-1.8504	0.3095	-5.979	2.25e-09	***
Distance to stream [m]	0.9989	0.2494	4.005	0.0000619	***

Table 4.S9: Factors characterizing ESS cluster 1 and 5 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 1; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 5. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference (χ^2= 13.691, df = 3, p=0.003357).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-18.6672	1.6865	-11.069	< 2e-16	***
Topography and soil parameters					
Aspect [°]	1.2086	0.4037	2.994	0.00276	**
Curvature [*score*]	4.4168	1.7502	2.524	0.01162	*
Effective rooting depth [mm]	11.6498	1.1754	9.912	< 2e-16	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	-2.5422	0.9076	-2.801	0.00509	**
Available water holding capacity [cm cm^{-1}]	30.1855	3.0182	10.001	< 2e-16	***
Climate					
Precipitation [mm]	-3.0773	0.71	-4.334	0.0000146	***
Reference Evapotranspiration [mm a^{-1}]	3.0096	0.8986	3.349	0.00081	***
Landscape composition					
Forest, 5 km buffer [*%*]	1.6219	1.0783	1.504	0.13254	
SRC, 5 km buffer [*%*]	1.2877	0.538	2.393	0.0167	*
Urban, 5 km buffer [*%*]	-2.9236	0.6477	-4.514	0.00000637	***
Distance to stream [m]	0.9935	0.3631	2.736	0.00622	**

Table 4.S10: Factors characterizing ESS cluster 1 and 6 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 1; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 6. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a insignificant and small difference (χ^2= 7.804, df = 3, p=0.05024).

Explanatory variable	**Stand. β**	**SE**	**z value**	**Pr(>|z|)**	
(Intercept)	5.0229	1.3484	3.725	0.000195	***
Topography and soil parameters					
Slope [%]	-12.8686	1.2161	-10.582	< 2e-16	***
Aspect [°]	1.1075	0.3625	3.056	0.002246	**
Curvature [*score*]	-10.8784	1.4333	-7.59	3.2e-14	***
Effective rooting depth [mm]	-3.2587	1.5762	-2.068	0.038686	*
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	4.2407	0.9493	4.467	0.00000792	***
Available water holding capacity [cm cm^{-1}]	-9.5672	3.0626	-3.124	0.001785	**
Soil quality index	11.007	1.2649	8.702	< 2e-16	***
Climate					
Precipitation [mm]	-2.4918	0.567	-4.394	0.0000111	***
Reference Evapotranspiration [mm a^{-1}]	-3.6111	0.6969	-5.182	0.00000022	***
Landscape composition					
Forest, 5 km buffer [*%*]	-5.7837	0.8862	-6.527	6.72e-11	***
Urban, 5 km buffer [*%*]	-3.5488	0.6864	-5.17	0.000000234	***

Table 4.S11: Factors characterizing ESS cluster 2 and 3 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 2; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 3. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference (χ^2= 193.6, df = 3, p<2.2e-16).

Explanatory variable	**Stand. β**	**SE**	**z value**	**Pr(>|z|)**	
(Intercept)	13.5787	0.9501	14.293	< 2e-16	***
Topography and soil parameters					
Elevation [m]	-4.2517	0.6777	-6.273	3.53e-10	***
Slope [%]	-8.0223	0.4982	-16.104	< 2e-16	***
Curvature [*score*]	-1.3272	0.5149	-2.577	0.00996	**
Effective rooting depth [mm]	-1.9267	0.6103	-3.157	0.00159	**
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	-2.1902	0.4599	-4.763	0.00000191	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	-4.8972	0.7237	-6.767	1.32e-11	***
Available water holding capacity [cm cm^{-1}]	-3.8205	1.1678	-3.271	0.00107	**
Soil quality index	4.1765	0.6871	6.078	1.22e-09	***
Climate					
Precipitation [mm]	3.9363	0.4165	9.451	< 2e-16	***
Reference Evapotranspiration [mm a^{-1}]	-4.0255	0.4519	-8.908	< 2e-16	***
Landscape composition					
Forest, 5 km buffer [%]	-9.0033	0.6267	-14.365	< 2e-16	***
SRC, 5 km buffer [%]	-1.1683	0.6195	-1.886	0.05931	.
Cropland, 5 km buffer [%]	-5.8619	0.6239	-9.396	< 2e-16	***
Urban, 5 km buffer [%]	-4.2552	0.7278	-5.847	5.01e-09	***

Table 4.S12: Factors characterizing ESS cluster 2 and 4 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 2; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 4. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference (χ^2= 21.818, df = 3, p=7.118e-05).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	2.4899	0.3547	7.02	2.22e-12	***
Topography and soil parameters					
Elevation [m]	2.7317	0.4134	6.608	3.91e-11	***
Slope [%]	-2.8792	0.3474	-8.287	< 2e-16	***
Curvature [*score*]	2.0256	0.3997	5.067	0.000000404	***
Effective rooting depth [mm]	2.7337	0.3928	6.96	3.4e-12	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	-1.1764	0.4451	-2.643	0.00822	**
Soil quality index	-2.3995	0.3739	-6.418	1.38e-10	***
Climate					
Precipitation [mm]	-0.6136	0.2669	-2.299	0.02152	*
Landscape composition					
Forest, 5 km buffer [%]	-3.9043	0.3359	-11.623	< 2e-16	***
Pasture, 5 km buffer [%]	-3.8755	0.2541	-15.251	< 2e-16	***
Cropland, 5 km buffer [%]	-5.3337	0.249	-21.424	< 2e-16	***
Water, 5 km buffer [%]	-0.4836	0.227	-2.131	0.03312	*
Urban, 5 km buffer [%]	-5.2251	0.3204	-16.31	< 2e-16	***
Distance to stream [m]	0.7177	0.164	4.375	0.0000121	***

Table 4.S13: Factors characterizing ESS cluster 2 and 5 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 2; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 5. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference (χ^2= 77.314, df = 3, p<2.2e-16).

Explanatory variable	**Stand. β**	**SE**	**z value**	**Pr(>|z|)**	
(Intercept)	-11.833	0.6648	-17.8	< 2e-16	***
Topography and soil parameters					
Elevation [m]	6.8486	0.6644	10.308	< 2e-16	***
Slope [%]	7.286	0.5695	12.793	< 2e-16	***
Aspect [°]	-0.4876	0.2189	-2.227	0.0259	*
Curvature [*score*]	7.0636	0.6369	11.09	< 2e-16	***
Effective rooting depth [mm]	3.751	0.3849	9.745	< 2e-16	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	0.9223	0.4415	2.089	0.0367	*
Available water holding capacity [cm cm^{-1}]	7.2733	1.1526	6.31	2.78e-10	***
Climate					
Precipitation [mm]	-3.8094	0.4601	-8.279	< 2e-16	***
Reference Evapotranspiration [mm a^{-1}]	2.4038	0.3913	6.143	8.08e-10	***
Landscape composition					
Forest, 5 km buffer [*%*]	2.6093	0.4436	5.881	4.07e-09	***
SRC, 5 km buffer [*%*]	4.2857	0.275	15.582	< 2e-16	***
Urban, 5 km buffer [*%*]	-2.8127	0.3803	-7.396	1.41e-13	***
Water, 5 km buffer [*%*]	2.799	0.3916	7.148	8.78e-13	***
Distance to stream [m]	0.4219	0.2437	1.731	0.0834	.

Table 4.S14: Factors characterizing ESS cluster 2 and 6 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 2; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 6. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed only a small difference (χ^2= 5.8011, df = 3, p=0.1217).

Explanatory variable	**Stand. β**	**SE**	**z value**	**Pr(>|z|)**	
(Intercept)	2.8028	0.6123	4.578	0.0000047	***
Topography and soil parameters					
Elevation [m]	-1.1768	0.6015	-1.956	0.05042	.
Slope [%]	-9.7763	0.4681	-20.887	< 2e-16	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	-2.1495	0.4172	-5.152	0.000000257	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	-1.586	0.6864	-2.311	0.020848	*
Available water holding capacity [cm cm^{-1}]	-6.1998	0.9994	-6.204	5.51e-10	***
Soil quality index	2.5041	0.4196	5.968	2.41e-09	***
Climate					
Precipitation [mm]	0.8913	0.3854	2.313	0.020729	*
Reference Evapotranspiration [mm a^{-1}]	-2.1111	0.3973	-5.314	0.000000107	***
Landscape composition					
Forest, 5 km buffer [%]	0.8249	0.5702	1.447	0.147976	
Pasture, 5 km buffer [%]	1.4148	0.3818	3.705	0.000211	***
SRC, 5 km buffer [%]	5.1188	0.3262	15.691	< 2e-16	***
Urban, 5 km buffer [%]	-1.6235	0.4996	-3.249	0.001156	**
Distance to stream [m]	1.0736	0.2377	4.516	0.00000631	***

Table 4.S15: Factors characterizing ESS cluster 3 and 4 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 3; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 4. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed a difference (χ^2= 263.88, df = 3, p<2.2e-16).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-10.657	0.782	-13.628	< 2e-16	***
Topography and soil parameters					
Elevation [m]	7.1763	0.651	11.024	< 2e-16	***
Slope [%]	6.6011	0.4615	14.302	< 2e-16	***
Aspect [°]	0.397	0.2064	1.923	0.054429	.
Curvature [*score*]	2.5544	0.4855	5.262	0.000000143	***
Effective rooting depth [mm]	5.3606	0.6235	8.597	< 2e-16	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	2.4447	0.4629	5.282	0.000000128	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	2.7122	0.7113	3.813	0.000137	***
Available water holding capacity [cm cm^{-1}]	6.4823	1.2161	5.33	9.81e-08	***
Soil quality index	-7.4393	0.7068	-10.525	< 2e-16	***
Climate					
Precipitation [mm]	-4.1861	0.4152	-10.083	< 2e-16	***
Reference Evapotranspiration [mm a^{-1}]	4.958	0.4443	11.16	< 2e-16	***
Landscape composition					
Forest, 5 km buffer [%]	3.6181	0.4707	7.687	1.5e-14	***
Pasture, 5 km buffer [%]	-4.8487	0.411	-11.797	< 2e-16	***
Cropland, 5 km buffer [%]	-1.1955	0.3567	-3.352	0.000804	***
Urban, 5 km buffer [%]	-1.8508	0.6099	-3.035	0.002407	**
Water, 5 km buffer [%]	0.7118	0.2836	2.51	0.012081	*
Distance to stream [m]	1.4863	0.2471	6.015	1.8e-09	***

Table 4.S16: Factors characterizing ESS cluster 3 and 5 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 3; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 5. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed a difference (χ^2= 144.66, df = 3, p<2.2e-16).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-18.5685	1.2397	-14.978	< 2e-16	***
Topography and soil parameters					
Elevation [m]	11.4527	1.2047	9.507	< 2e-16	***
Slope [%]	10.5595	0.8205	12.87	< 2e-16	***
Aspect [°]	-0.5439	0.3568	-1.525	0.12736	
Curvature [*score*]	8.3907	0.8978	9.346	< 2e-16	***
Effective rooting depth [mm]	2.8857	0.64	4.509	0.00000652	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	3.2043	0.7	4.578	0.0000047	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	3.1952	1.2583	2.539	0.01111	*
Available water holding capacity [cm cm^{-1}]	4.9752	1.6323	3.048	0.0023	**
Climate					
Precipitation [mm]	-6.6601	0.7157	-9.306	< 2e-16	***
Reference Evapotranspiration [mm a^{-1}]	5.4704	0.6914	7.912	2.54e-15	***
Landscape composition					
Forest, 5 km buffer [%]	5.4258	0.9595	5.655	1.56e-08	***
Pasture, 5 km buffer [%]	-1.7785	0.6044	-2.943	0.00325	**
SRC, 5 km buffer [%]	2.7294	0.5037	5.419	5.99e-08	***
Urban, 5 km buffer [%]	-4.5193	0.8372	-5.398	6.75e-08	***
Water, 5 km buffer [%]	3.494	0.4831	7.232	4.76e-13	***
Distance to stream [m]	0.8502	0.3922	2.168	0.03018	*

Table 4.S17: Factors characterizing ESS cluster 3 and 6 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 3; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 6. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed a difference (χ^2= 125.55, df = 3, p<2.2e-16).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	-1.8993	0.754	-2.519	0.0118	*
Topography and soil parameters					
Elevation [m]	3.4718	0.6967	4.983	0.000000626	***
Slope [%]	-0.7261	0.4055	-1.791	0.0734	.
Curvature [*score*]	1.0382	0.4139	2.508	0.0121	*
Effective rooting depth [mm]	1.8645	0.4076	4.574	0.00000479	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	2.9894	0.7264	4.115	0.0000387	***
Available water holding capacity [cm cm^{-1}]	-2.5648	0.6445	-3.979	0.0000691	***
Climate					
Precipitation [mm]	-2.556	0.4207	-6.075	1.24e-09	***
Reference Evapotranspiration [mm a^{-1}]	1.9031	0.4313	4.412	0.0000102	***
Landscape composition					
Pasture, 5 km buffer [*%*]	-3.839	0.353	-10.875	< 2e-16	***
SRC, 5 km buffer [*%*]	-0.8789	0.3969	-2.215	0.0268	*
Cropland, 5 km buffer [*%*]	-2.4368	0.5384	-4.526	0.000006	***
Urban, 5 km buffer [*%*]	-6.4621	0.6624	-9.755	< 2e-16	***
Water, 5 km buffer [*%*]	0.705	0.3103	2.272	0.0231	*
Distance to stream [m]	1.1016	0.27	4.081	0.0000449	***

Table 4.S18: Factors characterizing ESS cluster 4 and 5 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 4; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 5. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed a difference (χ^2= 106.42, df = 3, p<2.2e-16).

Explanatory variable	**Stand. β**	**SE**	**z value**	**Pr(>|z|)**	
(Intercept)	-7.1283	0.6139	-11.611	< 2e-16	***
Topography and soil parameters					
Elevation [m]	3.3888	0.7016	4.83	0.00000137	***
Slope [%]	3.6113	0.5236	6.898	5.29e-12	***
Aspect [°]	-0.7757	0.2084	-3.722	0.000198	***
Curvature [*score*]	7.1101	0.6311	11.266	< 2e-16	***
Effective rooting depth [mm]	1.7167	0.3216	5.338	9.42e-08	***
Erosivity (R) [MJ mm ha-1 h-1 a-1]	1.144	0.6893	1.66	0.096987	.
Available water holding capacity [cm cm^{-1}]	9.6669	0.9167	10.545	< 2e-16	***
Climate					
Precipitation [mm]	-1.9202	0.3993	-4.809	0.00000152	***
Reference Evapotranspiration [mm a^{-1}]	1.2854	0.3731	3.445	0.000572	***
Landscape composition					
Pasture, 5 km buffer [*%*]	0.5522	0.2955	1.869	0.061644	.
SRC, 5 km buffer [*%*]	0.6792	0.2384	2.849	0.004387	**
Urban, 5 km buffer [*%*]	-2.5269	0.2943	-8.587	< 2e-16	***
Water, 5 km buffer [*%*]	2.1468	0.3519	6.101	1.05e-09	***

Table 4.S19: Factors characterizing ESS cluster 4 and 6 for scenario 4 (backward logistic regression). A positive value for the standardized β indicates that an explanatory variable is contributing to cluster 4; a negative value for the standardized β indicates that an explanatory variable is contributing to cluster 6. The likelihood ratio test showed a significant difference when the final model was compared to a null model (χ^2= 1398.4, df = 14, p<2.2e-16). Comparing the final model with a model including x- and y-coordinates, the likelihood ratio test showed a difference (χ^2= 97.124, df = 3, p<2.2e-16).

Explanatory variable	Stand. β	SE	z value	Pr(>|z|)	
(Intercept)	6.9057	0.5422	12.736	< 2e-16	***
Topography and soil parameters					
Elevation [m]	-4.4778	0.4575	-9.788	< 2e-16	***
Slope [%]	-7.526	0.3655	-20.593	< 2e-16	***
Curvature [*score*]	-0.7616	0.3964	-1.921	0.054699	.
Effective rooting depth [mm]	-3.7839	0.5278	-7.17	7.52e-13	***
Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	-1.0316	0.3741	-2.757	0.005827	**
Available water holding capacity [cm cm^{-1}]	-9.3981	1.0753	-8.74	< 2e-16	***
Soil quality index	6.1995	0.5561	11.149	< 2e-16	***
Climate					
Precipitation [mm]	1.588	0.3397	4.674	0.00000295	***
Reference Evapotranspiration [mm a^{-1}]	-3.208	0.3495	-9.179	< 2e-16	***
Landscape composition					
Forest, 5 km buffer [*%*]	-1.6118	0.3679	-4.381	0.0000118	***
Pasture, 5 km buffer [*%*]	2.6629	0.3135	8.494	< 2e-16	***
Urban, 5 km buffer [*%*]	-1.3244	0.3744	-3.537	0.000404	***

References

[1] Jones KB, Neale A, Nash M, Van Remortel R, Wickham J, Riitters K, et al. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 2001;16(4):301-12.

[2] European Environment Agency (EEA), Corine Land Cover 2006 raster data, 2013, European Environment Agency.

[3] Wochele S, Priess J, Thrän D, O'Keeffe S. Crop allocation model "CRAM" - an approach for dealing with biomass supply from arable land as part of a life cycle inventory. 22nd European Biomass Conference and Exhibition; 2014; Hamburg: ETA-Florence Renewable Energies.

[4] Kissel M, Fuentes A, Deane-Drumnond C. The evolution of the human niche: assessing and describing the development of complex decision-making in the Pleistocene through an open-access, comparative database. American Journal of Physical Anthropology 2015;156:188-.

[5] Meyer MA, Chand T, Priess JA. Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLoS ONE 2015;10(3):e0116336.

[6] Wrbka T, Erb KH, Schulz NB, Peterseil J, Hahn C, Haberl H. Linking pattern and process in cultural landscapes. An empirical study based on spatially explicit indicators. Land Use Policy 2004;21(3):289-306.

[7] Lehner B, Verdin K, Jarvis A. New Global Hydrography Derived From Spaceborne Elevation Data. Eos, Transactions American Geophysical Union 2008;89(10):93-4.

[8] Qiu J, Turner MG. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proceedings of the National Academy of Sciences 2013;110(29):12149-54.

[9] Panagos P, Van Liedekerke M, Montarella L, The European Soil Database distribution version 2.0, 2006, European Commission/European Soil Bureau Network, Ispra.

[10] LfULG, Auswertekarten Bodenschutz 1:50.000, 2012, Saxon State Office for Environment Agriculture and Geology, Freiberg.
[11] Bischoff R, K-Faktor (MMK 100), 2014, TLUG, Jena.
[12] Jäckel G, Zink M, Marx A, Aufbereitung von gemessenen und simulierten Klimadaten, 2012, Mitteldeutsches Klimabüro, Leipzig.
[13] FAO Geonetwork. Global map of monthly reference evapotranspiration - 10 arc minutes (GeoLayer). Rome, Italy2014 [cited 2014 January 14]; Available from: http://www.fao.org/geonetwork/srv/en/main.home.
[14] Bräunig A, Dokumentation zur Berechnung und Ableitung R-Faktor Sachsen, 2013, LfULG, Freiberg.

5 How can we compare environmental and ecosystem service assessments for biomass production systems in different parts of the world?

This chapter is under review at Environmental Research Letters:

Meyer MA, Seppelt R, Witing F, Priess JA. How can we compare environmental and ecosystem service assessments for biomass production systems in different parts of the world? (*under review*)

5.1 Abstract

Increasing global trade of forestry or agricultural products raises the question, how local/regional environmental impacts (EIs) of different locations can be compared. To enable comparisons of the effects of land use activities at various production locations, it is necessary to control for environmental heterogeneity, i.e., the variation of biotic and abiotic conditions. In this study, we use three approaches to control for environmental heterogeneity applying (i) environmental stratification, (ii) potential natural vegetation (PNV), and (iii) regionally set environmental thresholds to compare EIs of solid biomass production, e.g., for bioenergy. We utilize production regions with managed forests and plantation forestry for subtropical (Satilla watershed, southeastern US), tropical (Rufiji basin, Tanzania), and temperate climate (Mulde watershed, central Germany). All approaches allow for comparing the EIs of different land use/land cover (LU/LC) classes between and within production regions and also the EIs with each other. The different approaches rank EIs for LU/LC within a production region according to their naturalness (forest, plantation forestry, cropland). We identified the PNV approach as conceptually the most reliable, but with major deficiencies concerning feasibility and relevance. The PNV approach explicitly includes most of the factors that drive environmental heterogeneity in contrast to the stratification and threshold approach. We also show that stratification based comparison facilitates a concise global application. Regional environmental thresholds only implicitly include environmental heterogeneity varying by individual case and cover only few EIs. Further studies are needed to validate the methodologies for other land use systems and environmental conditions.

5.2 Introduction

Increasing pressures on land resources as well as increasing global trade of (solid) biomass and other agricultural commodities are calling for comparative approaches to assess the EIs related to the production of commodities, e.g., [1], to identify ways of sustainable production [2]. Thus, methods are needed, which enable us to compare EIs and ecosystem services (ESS) in different production systems across the globe. This study is seen as a contribution to develop a set of methods, which finally enables us to minimize detrimental EIs, via identifying the least harmful combinations of production systems and environmental conditions, keeping in mind that social and economic impacts, costs or benefit, may play similar or even dominating roles.

Policies such as the EU Renewable Energy Directive (EU RED) are designed to save greenhouse gas (GHG) emissions, diversify, secure and create energy supply, and create jobs [3-6]. Biomass for bioenergy typically originates from various sources, e.g., agriculture or forestry, and production systems, e.g., managed forest or plantation forestry, from different parts of the world and is traded in large quantities. For example, Lamers et al. [7] expect an increase in EU solid biomass imports from 2010 until 2020 by about 300 percent to 236 PJ, even under the currently very likely implementation of sustainability requirements of the EU RED. Increasing global trade of raw or processed biomass may change regional land use systems, an effect which should be avoided by standardized sustainability requirements in certification schemes complying, e.g., with the EU RED [8]. However, these requirements are currently only weakly specified. Biomass producers need to prove that they fulfill the sustainability criteria, usually via complying with certification schemes. For that purpose, global EIs such as GHG emissions are assessed with harmonized and standardized life-cycle assessments [9, 10]. Such global EI assessment approaches of biomass production often hardly represent major local/regional socio-economic and environmental processes. For example, Dale et al. [11] state that soil and water quality impacts of biomass production are not reasonably covered by standardized and non-place-based indicators as used in life-cycle assessments.

Land use systems for bioenergy production are an example that needs a methodology to assess the environmental sustainability at the local/regional scale. At the local/regional scale, environmental assessments lack harmonization and consistency in methodology, e.g., ESS assessments [12]. Earlier studies on the quality of indicators for the environmental assessment, e.g., [4, 13, 14], identified that certification schemes often give preference to cause-related environmental indicators, e.g., on best management practices in agriculture or forestry. To assess the human impact on the environment through biomass for bioenergy or other products in different production regions, it will be necessary to apply a methodology that controls for environmental heterogeneity. Especially problematic is the fact that harmonized approaches need to be able to deal with different levels of data availability, i.e., a situation with high data availability, e.g., in the US, or data scarce conditions as found for many developing countries or countries restricting access to data.

Environmental heterogeneity, the spatial variation of biotic (vegetation and land cover) and abiotic conditions (climate, topography, and soil) [15], is often considered in the context of biodiversity modelling, e.g., [16]. Kienast et al. [17] argue that environmental heterogeneity similarly affects the level of positive, e.g., ESS, and negative EIs, but requires further spatially explicit quantification. Neglecting environmental heterogeneity makes comparing the sustainability of different production regions impossible. In existing ESS research, no methodology is applied consistently to compare and to assess the sufficiency of ESS supply [18]. To date, several conceptual frameworks exist, which propose different ways to quantify an ecosystem’s capacity to provide ESS, the actual ESS supply as well as ESS demand [19, 20]. Both studies, synthesizing various frameworks and approaches to tackle the question of sustainable ESS supply, could not identify a generally applicable methodology, suitable for a large set of ESS.

There are different options for sustainability assessments to account for the variation in EIs due to environmental heterogeneity. Potential options are threshold or target values or baseline conditions [21, 22]. The former are supposed to be set by policy makers or in a larger societal discourse. Stakeholders may set, e.g., critical pollutant loads, considering the regional ecosystem and its desired state [21]. Baseline conditions may be obtained on different pathways. EIs may be normalized in relation to the minimal and maximal impact in the respective environmental stratum. Metzger et al. [23] roughly quantify ESS and stratify them at European scale. A second approach is to relate the EIs of the current LU/LC to PNV representing conditions prior to land use activities [24]. West et al. [25] compare carbon storage between current LU/LC and PNV at the global scale. However, both options are not used to compare the sustainability of EIs at the regional scale, which is a major assessment scale with respect to ESS [26]. Instead, several studies, e.g., [27], define arbitrary thresholds of hot and cold spots of ESS, e.g., the 20%- and 80%-quantiles of the ESS values in a study region. This approach produces at least two groups, positive and negative locations with respect to EIs, and, e.g., allows to identify factors that characterize both groups for improved land management [28]. Such arbitrary selection of high or low thresholds for EIs may not necessarily make sense in another region, depending on environmental conditions and land use activities. For instance, the top 20% of an area regarding biodiversity may still be less diverse in an intensively managed region than the attainable improvement after conservation efforts. Vice versa, an extensively managed region may still have a higher biodiversity in the lowest 20% of the area than the upper 20% of an agro-industrial region.

In this paper, we model EIs of biomass production in two land use systems with plantation forestry in the southeast US (Satilla watershed) and in Tanzania (Rufiji basin) and one with secondary forest used for biomass production in central Germany (Mulde watershed) as exemplary production regions. (i) We assume that the EIs are higher in regions with more intensive biomass production. For example, pine plantations in the southeast US may have lower EIs than managed forests in central Germany. (ii) We expect that different approaches to control for environmental heterogeneity may yield different results and conclusions regarding EIs. We aim at testing and comparing methods to account for environmental heterogeneity, with the objective to make the EI of land use activities comparable in different world regions. To achieve this objective, we environmentally stratify EIs, use PNV as a reference case, and apply environmental thresholds. We test if and how the results of these approaches overlap by using the hot and cold spot approach commonly used in ESS assessments. In addition, we analyze the LU/LC composition of the hot and cold spots identified by the different approaches. (iii) We analyze the suitability of the applied methodologies for further use with respect to reliability, feasibility and relevance, as major categories for indicator-based environmental assessments [29, 13].

5.3 Material and Methods

5.3.1 Study sites

We selected three production regions, Fig. 5.1, of which two represent major global solid biomass supply regions (southeast US and Tanzania) and the remaining a major consumer and producer region ((central) Germany) [7]. The selected production regions cover a wide range of socio-economic and environmental conditions:

- Southeast USA (Satilla watershed): Developed country with a from the EU differing legislation on the sustainability of bioenergy, but with a standard and voluntary schemes applied to ensure broader sustainability of bioenergy [14]
- Tanzania (Rufiji basin): Developing country with an existing, but weakly enforced legislation and without largely applied standards and schemes on the sustainability of bioenergy [30].
- Central Germany (Mulde watershed): Developed country with enforced legislation concerning the sustainability of bioenergy, binding for biofuels and bioliquids, but supposed to be applied to solid bioenergy by national or supra-national solutions (legislation, schemes, standards) in the EU member states [31].

In addition to the regulatory differences, the production regions used in this paper represent different climates: humid continental climate (Mulde watershed: 5,791 km²), humid subtropical climate (Satilla watershed: 8,760 km²) and tropical wet and dry savanna climate (Rufiji basin: 176,301 km²).

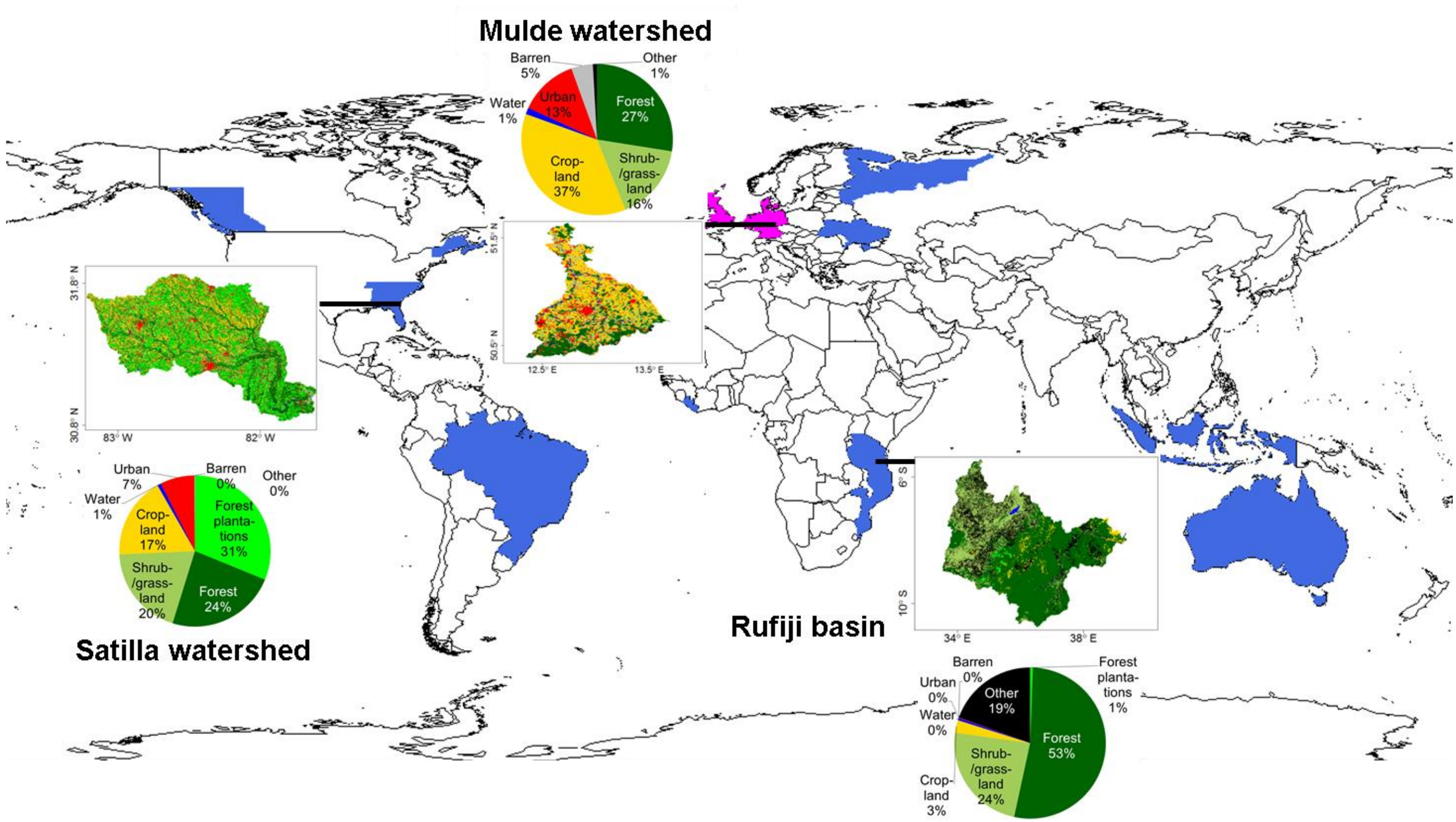

Fig. 5.1: Case studies representing temperate, subtropical and tropical regions for current solid biomass supply and demand based on [7]; light blue: solid biomass supply, magenta: solid biomass demand; the legend with the LU/LC classes indicates the respective share for the Satilla and Mulde watersheds and the Rufiji basin.

5.3.2 Environmental impact assessment of carbon storage, erosion, nutrient load, and biodiversity

We modeled the amount of carbon stored based on the IPCC guidelines [32], partly with InVEST (Integrated Valuation of Environmental Services and Tradeoffs) [33, 34], with the data indicated in Table 5.1 for 2006 (Satilla and Mulde watershed) and 2005 (Rufiji basin) respectively. With the InVEST model, we estimated P export and retention. We validated the modeled P export against the corresponding P concentration measurements of the stations Little Satilla near Offerman, Satilla River at Atkinson (USGS 02227500/USGS 2228000 (Satilla watershed)), Msembe (1KA59 (Rufiji basin)), Errln, Niederschlema, and Bad Düben (Mulde watershed*).* We also modeled the amount of retained sediment with InVEST based on the universal soil loss equation [35, 36], i.e., sheet erosion, and the LU/LC specific sediment removal rates. We validated the sediment export against the suspended sediment concentration for the stations Little Satilla near Offerman, Satilla River at Atkinson (USGS 02227500/USGS 2228000 (Satilla watershed)), Msembe (1KA59 (Rufiji basin)), Errln, Wechselburg, and Bad Düben (Mulde watershed).

Furthermore, we modeled impacts on biodiversity with the Globio model as described in [37] as an approach equally applicable in data scarce environments such as Tanzania.

Table 5.1: Data items; for InVEST (Carbon storage: No. 2, 16; P retention and export: No. 1 -7, 10-12; Sediment retention and export: No. 1-3, 8-9, 13-15) and for Globio (No. 2, 17-21)) with sources for used data and formulas; the InVEST default assumptions of modelling parameters are refined based on the indicated sources (No. 10 - 15).

	Input datasets	Southeast US	Tanzania	Central Germany
1	DEM (3 arc-seconds) [m]	[38]	[38]	[38]
2	LU/LC	[39]	[40-42]	[43, 44]
3	Potential Natural Vegetation	[45]	[46]	[47]
4	Reference Evapotranspiration (10 arc-min) [mm a^{-1}]	[48]	[48]	[48]
5	Precipitation [mm a^{-1}]	[49]	[50]	[51]
6	Depth to any soil restrictive layer [mm]	[52]	[53]	[54, 55]
7	Available water holding capacity [cm cm^{-1}]	[52]	[53]	[54, 55]
8	Erosivity (R) [MJ mm ha^{-1} h^{-1} a^{-1}]	[56, 57]	[58, 59]	[60]
9	Erodibility (K) [t ha h ha^{-1} MJ^{-1} mm^{-1}]	[52]	[53]	[61, 54]
10	Rooting depth [mm]	[62-66]	[67]	
11	P export [kg ha^{-1} a^{-1}]	[68, 69]	[67]	[69]
12	P retention efficiencies [*%*]		[67]	
13	cover-management factor (C)	[70-75, 36]	[67]	
14	Support practice factor (P)	[71, 73, 76, 75]	[67]	[77]
15	Vegetation sediment retention efficiency [*%*]		[67]	
16	Carbon pools [Mg ha^{-1}]	[32, 78, 27, 79, 80]	[67, 81]	[82-88]
17	Population density [*n* km^{-2}]	[89]	[90]	[91]
18	Street and railway map	[92, 93]	[94]	[95]
19	N deposition	[96]	-	[97]
20	Critical N loads	[98, 99]	-	[97]
21	Terrestrial ecoregions	[100]	[100]	[100]

5.3.3 Assessing environmental impacts under environmental heterogeneity

We compared three approaches to control for environmental heterogeneity in biomass production systems in different parts of the world:

- environmental stratification according to [101],
- PNV [45-47] as a reference case, and
- environmental thresholds or target values that may be regionally set, depending on the EI [102-104].

For environmental stratification, we used a spatial dataset with 128 global environmental strata [101] or environmentally homogenous groups, which are spatial subgroups with regions of comparable environmental properties, especially climate. We normalized the EI within each environmental stratum with equation (1):

$$f_{strat}(x_i, X_j) = \frac{x_i - min(X_j)}{max(X_j) - min(X_j)} \tag{1}$$

with x_i as EI variable value in the environmental stratum j and X_j as all EI values in the environmental stratum j, see [23] for a European production region. Therefore, we need to model the EIs only for the current LU/LC dataset.

Alternatively, we modeled EIs for current LU/LC and for PNV as a reference case, e.g., [25, 105]. We calculated a score for each EI using equation (2):

$$f_{PNV}(x_k, y_k) = \frac{x_k - y_k}{y_k} \tag{2}$$

with x_k as EI variable for current LU/LC and y_k as the EI value for PNV.

Regional environmental thresholds defined by stakeholders or authorities only exist for the negative EIs, P and sediment export, in the Satilla and Mulde watersheds, but not for other EIs. In the US, typically P and sediment loading thresholds are set as the *Total Maximum Daily Loadings* [106] for most of the surface water pollutants and are translated to land-based thresholds, e.g., sediment yields/soil erosion rates and P export rates, see Table 5.2. Similarly, the P and suspended solids concentrations in the Mulde watershed are converted as presented by Ludwig et al. [107]. Since sediment loadings do not have thresholds in the Satilla watershed, we used the average value of unimpaired watersheds in the Piedmont and southeastern Plains ecoregions. We calculated a score for each EI using equation (3):

$$f_{thres}(x_m, z) = \frac{x_m - z}{z} \tag{3}$$

with x_i as EI variable and z as the threshold values for the EI value.

For better visualization of the data, we transform the input data for equation (1) and the output of equations (2) and (3) with equation (4):

$$f_{transform}(x) = \begin{cases} -ln(-x+1), x < 0 \\ ln(x+1), else \end{cases} \quad (4)$$

Table 5.2: Environmental thresholds for P and Sediment export set by environmental protection agencies (partly with public consultation); for the Satilla watershed, threshold values for water quality with "high ecological status" (class I) according to the European Water Framework Directive were taken.

Thresholds	Value	Unit	Data source
P export (Satilla watershed)	2.31	[kg ha^{-1} a^{-1}]	[102]
P export (Mulde watershed)	0.03	[kg ha^{-1} a^{-1}]	[104]
Sediment yield (Satilla watershed)	543.63	[kg ha^{-1} a^{-1}]	[103]
Sediment yield (Mulde watershed)	64.73	[kg ha^{-1} a^{-1}]	[104]

5.3.4 Congruence of sustainability assessment approaches

Based on the techniques compiled by Mouchet et al. [28], we tested the cross-predictive capacity with diagnostic test statistics [108]. We applied the diagnostic test statistic with R [109], which provides a score between 0 and 1 for the agreement of correctly identified pixels of sufficient, i.e., sensitivity, and insufficient EIs, i.e., specificity, i.e., above and below a critical value in the stratification and threshold approaches and the EIs modeled for PNV. The PNV and the threshold approach categorize EIs into two groups. However, the stratification approach does not categorize the EIs into two groups (positive and negative). It only allows to compare EIs between production regions or LU/LC classes. One option to have two groups of EIs, positive and negative, would be to apply the concept of hot- and coldspots from ESS research. Hot- and coldspots, i.e., locations of high or low supply of ESS, are often determined by arbitrarily defined 10%-quantiles as thresholds to classify the ESS into groups. We used the concept to assess the congruence of the PNV and the stratification approach for all 10%-quantiles (10%-90) and the arithmetic mean of the EIs. Comparing the stratification approach with arbitrarily set quantiles and PNV or thresholds helps to (i) characterize the different production regions with respect to the intensity of land use activities. To characterize the identified hotspots of the stratification approach, 70-90%-quantiles as typical cases in ESS research [110, 27], and the PNV approach, we compared (1) the spatial extent and (2) the dominating LU/LC classes. (ii) We aimed at identifying cases in which PNV, stratification and thresholds may be used interchangeably, i.e., to identify the degree of overlapping of hotspot pixels in all approaches.

5.3.5 Reliability, feasibility and relevance of the approaches

There are several studies on blueprints for further harmonizing the concepts in ESS and environmental assessment and management with the objective to enhance the comparability of regional production regions [26, 111]. The criteria to select suitable environmental indicators equally apply for methodologies to develop baseline conditions under environmental heterogeneity, i.e., reliability and conceptual soundness, feasibility, and relevance for the end user [13, 29]. From existing literature, we collected sub-criteria for the three categories listed above. For reliability, we used the criteria: worldwide consistent methodology and datasets [112], regional stakeholder and expert involvement [22], environmental heterogeneity within a study region [113], and the range of environmental factors considered [16]. For feasibility, we used the criteria: global data coverage

[113], easily applicable for various EIs [114], and the development effort of the methodology if no data is available [115]. For relevance, we used the criteria: clear classification in positive and negative locations with respect to EIs [115], and the relative comparison of globally distributed production regions [113], e.g., to compare the EIs of plantation forestry between two production regions.

5.4 Results

5.4.1 Environmental impact assessment of biomass production systems across production regions

After applying the stratification, PNV, and threshold approaches, we could compare each EI for different LU/LC (cropland (yellow), plantation forestry (light green) and natural or semi-natural forest (dark green)) between and within different production regions, see Fig. 5.2, Fig. 5.S1, Fig. 5.S2, and Table 5.S1.

In the stratification and PNV approach, Fig. 5.2, forestry in the Mulde watershed showed the highest value for carbon storage. Plantation forestry had the highest values for sediment retention in the Rufiji basin and for P retention in the Satilla watershed. Biodiversity was highest for plantation forestry in the Rufiji basin. The top rated production region with respect to positive EIs was similar between both approaches. However, the order of the EIs differed between the two approaches. In the stratification approach, Fig. 5.S1, forestry in the Mulde watershed had lower sediment export, whereas plantation forestry in the Satilla watershed had lower P export. In the PNV approach, plantation forestry in the Satilla watershed had higher sediment and P export. In the threshold case, plantation forestry in the Satilla watershed had higher sediment export, whereas forestry in the Mulde watershed had higher P export. In total, the stratification and PNV approach agreed if we were interested in the production region with the highest EI value for biomass production. However, the ranking of the different biomass production options with respect to EIs was less consistent between the PNV and stratification approach.

The ranking of the individual LU/LC classes and EIs between the production regions only agreed for the stratification and PNV approach for three EIs, i.e., sediment and P retention for cropland and P retention for plantation forestry, see Fig. 5.2 and Table 5.S1. For negative EIs, Fig. 5.S1 and Table 5.S1, the ranking only agreed for the stratification and PNV approach for P export. The ranking agreed for the PNV and threshold approaches for five out of ten cases.

Comparing current LU/LC with PNV, the decline in carbon storage and biodiversity was lowest in the Rufiji basin. For P and sediment retention, the increase was highest in the Satilla watershed and the Rufiji basin respectively. For sediment export, we had the highest increase in the Satilla watershed. For P export, we had a higher increase in the Mulde watershed. Comparing the EIs with each other for the current LU/LC and PNV, P retention deviated most from PNV as a baseline for the entire study areas, forestry and plantation forestry. Sediment retention with an increase towards PNV and carbon storage with a decrease towards PNV deviated less. Only for cropland in the Rufiji basin, we had a different pattern. Negative EIs were less homogeneous in that respect. The ranking within the production regions agreed mostly between the stratification and PNV approach. It followed the logic of higher positive EIs and lower negative EIs for forest than for plantation forestry than for cropland. Sediment retention in the Satilla watershed and P retention in the Rufiji basin showed deviating orders of LU/LC classes between the two approaches.

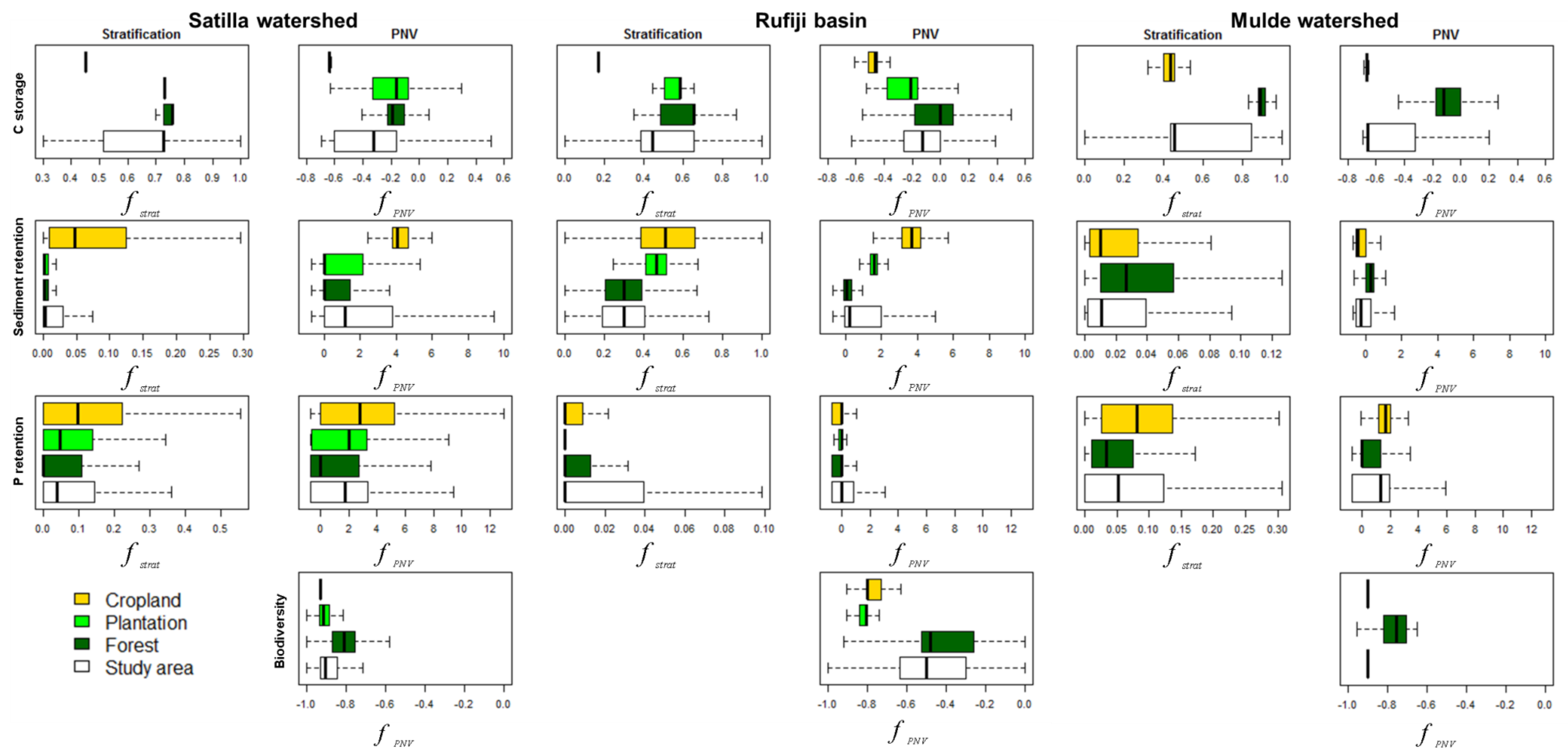

Fig. 5.2: Beneficial EIs standardized with the stratification and PNV approach to control for environmental heterogeneity. After applying the stratification and PNV approach, we compare each EI for different LU/LC (cropland (yellow), plantation forestry (light green) and natural or semi-natural forest (dark green)) between and within different production regions. For the stratification approach, each EI is standardized with f_{strat} for each environmental stratum i, i.e., homogeneous bioclimatic groups; for the PNV approach, the relative difference between PNV and the current EIs is displayed with f_{PNV}; values >0 indicate higher EIs supply in the current LU/LC than for PNV. For better visualization, the input data for f_{strat} as well as the scores f_{PNV} (PNV for carbon storage, sediment and P retention) have been transformed with $f_{transform}$.

5.4.2 Congruence of sustainability assessment approaches

Comparing the PNV and stratification approach for positive EI hotspots, see Table 5.3, the Satilla and Mulde watershed were most congruent at similar quantiles. In the Rufiji basin, the PNV and stratification approaches were most congruent at different quantiles. Therefore, we did not have the same intensity of EIs by setting similar quantiles. The sensitivity, i.e., first values in the bracket, was generally lower than the specificity. It means that if stratification with quantiles was used instead of PNV, the risk that locations of strong positive EIs were left out was higher. It was less likely that hotspots that were not classified as such through PNV would be classified as hotspots through stratification.

Comparing the PNV and the stratification approach for negative EIs, the congruence between the Satilla and Mulde watersheds was less comparable than for positive EIs. Stratification rather overestimated the size and number of hotspots of sediment export, whereas it rather underestimated P export hotspots. The NA-values and the low specificity comparing the stratification with the threshold case and the PNV with the threshold case respectively reflected the low number of P and sediment export rates above the locally set thresholds. Therefore, thresholds only selected hotspots of extremely negative EIs, whereas PNV and the stratification approach with quantiles differentiated more in their sustainability assessment.

Table 5.3: Degree of agreement between different approaches to identify hotspots of EIs; the percentages indicate the quantiles for the stratification approach which agree best with the PNV or threshold approach. We indicate the best fitting value for all EIs (Total) and the EIs per production region (C storage, sediment retention and export, P retention and export); sensitivity (agreement/overlap between two approaches of pixel classified as hotspots) and specificity scores (agreement/overlap between two approaches for pixels not classified as hotspots) are indicated in brackets; score values of 0 indicate no agreement and 1 indicates complete agreement between two approaches; For example, the best agreement between the stratification and PNV approaches occurred for the 90%-quantile for the Satilla watershed; for the entire results, see Tables 5.S2-S12.

Positive EIs	Total	C storage	Sediment retention	P retention
Stratification/PNV				
Satilla	50% (1.4/2.8)	90% (0.2/1.0)	60% (0.5/0.9)	40% (0.8/0.9)
Rufiji	70% (1.6/2.5)	70-90% (0.8/0.8)	mean (0.5/0.7)	70% (0.4/0.8)
Mulde	50% (1.7/2.6)	90% (0.7/1.0)	60% (0.7/0.8)	40% (0.9/0.9)
Negative EIs			**Sediment export**	**P export**
Stratification/PNV				
Satilla			mean (1.0/0.2)	50% (0.7/1.0)
Mulde			60% (1.0/0.4)	40% (0.6/0.7)
Stratification/ threshold				
Satilla			mean (1.0/0.0)	NA
Mulde			NA	mean (0.0/0.4)
PNV/threshold				
Satilla			(0.5/1.0)	NA
Mulde			(0.9/0.0)	(0.8/0.0)

5.4.3 LU/LC differences between hotspots (PNV vs. stratification approach)

In the Satilla watershed, 1.1 % of the area was classified as EI hotspots in the PNV approach and 0.8 % with the 80%-quantile of the stratification approach, center of the common range of 70-90% as indicated in section 5.3.4. Plantation forestry accounted for 1 % (PNV approach) and 39 % (80%-quantile in the stratification approach) of total hotspots, Fig. 5.3. This deviation showed that the EI values were high for plantation forestry compared with other LU/LC in the production region, but were mostly lower than for PNV. The remaining hotspots were nearly exclusively natural or semi-natural forests. In the Rufiji basin, 13.8 % of the area could be classified as hotspots in the PNV approach and 0.7 % in the stratification approach (80%-quantile). The current LU/LC in the Rufiji basin was closer to the potential natural state and the stratification approach (80%-quantile) did not account for that aspect. Natural or semi-natural forest accounted for 62.4 % (PNV approach) and 73.7 % (80%-quantile in the stratification approach) of total hotspots. The remaining hotspots were shrub- and grassland, 25 % (PNV approach) and 26.3 % (80%-quantile in the stratification approach), and 12.4 % mosaic vegetation and cropland (PNV approach). In the Mulde watershed, 3.1 % of the area could be classified as hotspots in the PNV and 0.5 % in the stratification approach (80%-quantile). In both cases all hotspots were forests. In the Rufiji basin, the composition of hotspots with respect to LU/LC shows a similar pattern except the fact that mosaic vegetation, denoted as “Other” in Fig. 5.3, was a major hotspot in the PNV approach. Comparing the 90%- and 70%-quantiles with the 80%-quantile showed that LU/LC classes with higher land use intensity were included with a lower quantile value. The general LU/LC composition of the hotspots was mostly similar. The pattern strongly deviated only for the Satilla watershed for carbon storage in the 90%-quantile with a nearly exclusive dominance of forest, and for the Mulde watershed for carbon storage in the 70%-quantile with a mixture of urban land, cropland, and shrub- and grassland.

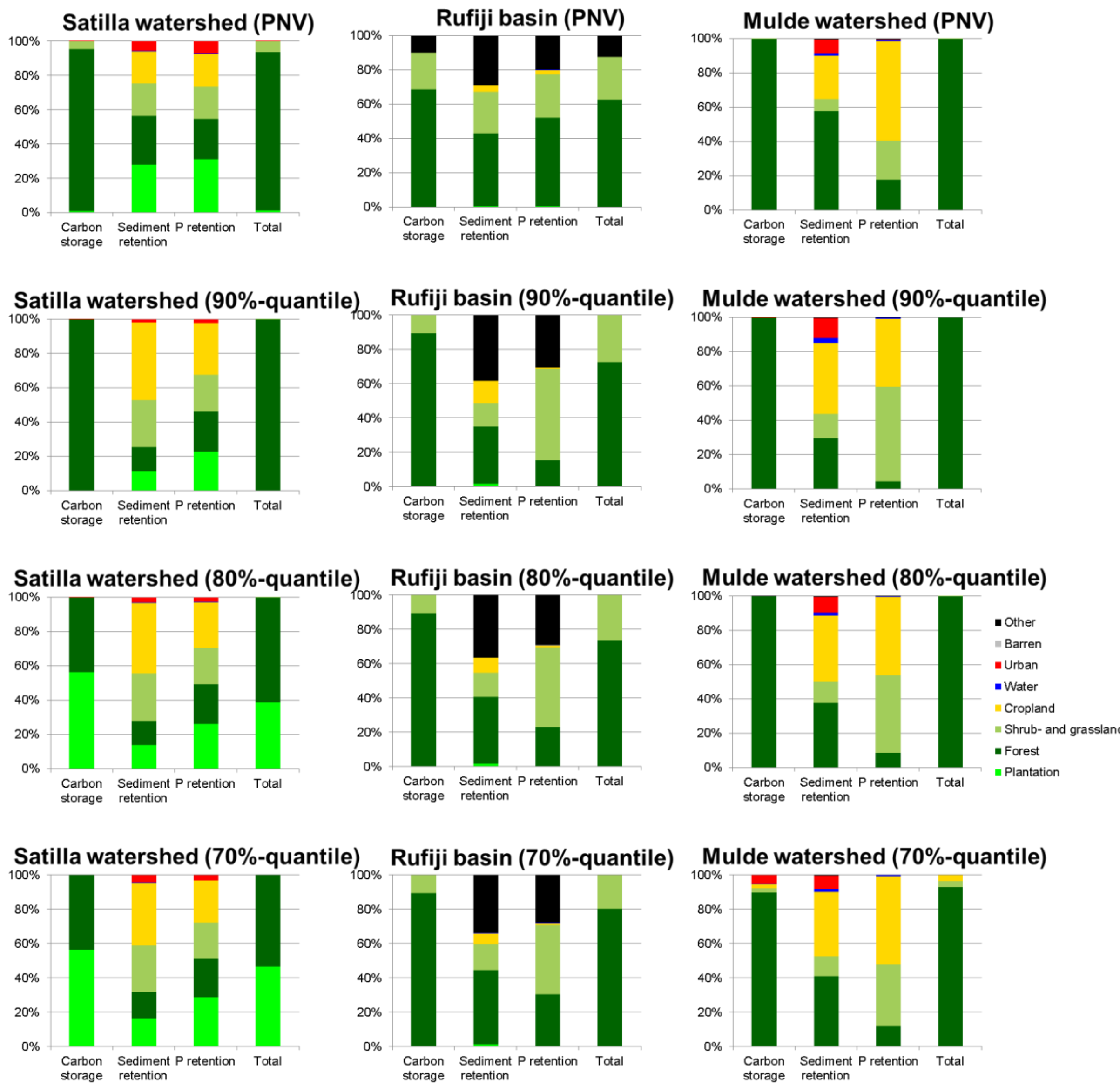

Fig. 5.3: LU/LC composition of individual positive EIs and the agreement of different comparative approaches; the LU/LC composition is displayed for hotspots of EIs. The hotspots of EIs are all pixels with EI values for the current LU/LC > PNV and the top 10 to 30 % percent of pixels with EI values for the stratification approach.

5.4.4 Reliability, feasibility and relevance of the approaches

The reliability of the comparative approaches presented in this study, largely depended on a *consistent methodology* applied and the data required to analyze EIs in production systems distributed across the globe. The environmental stratification approach which is based on a single global dataset promised more comparability than using regional PNV datasets, see Table 5.4. The PNV approach may be less comparable as various ecological concepts are partly used and partly not. For example, natural disturbance of the landscape through fire is considered in the Satilla watershed, but not in the other two production regions. The involvement of *regional stakeholders and experts* may create inconsistencies at larger scales due to the potentially different regional strategies to define thresholds.

The available environmental thresholds, see section 5.3.3, are typically defined for an entire area and do not consider *environmental heterogeneity within a region*, in contrast to the stratification and PNV approach. The *range of environmental factors* to assess environmental heterogeneity varies between the approaches. The PNV approach implicitly includes more abiotic factors (topography and soil) as well as biotic factors (vegetation and land cover) that contribute to environmental heterogeneity. The environmental threshold approach may implicitly consider such factors as in the ‘critical load’ concept, the level of nutrient input below which no harmful alteration of the regional ecosystems occurs [99], in the Satilla watershed or the water framework directive [116] in the Mulde watershed for the nutrient and erosion related EIs.

Environmental stratification and PNV can be applied to all of the EIs in this study (*global data coverage*). Environmental thresholds could clearly determine if positive EIs are sufficient and negative ones exceeding the defined limits and calling for action, whereas the other two approaches do not give any or no direct indication for action (stratification and PNV) (*easily applicable*). Due to the broader range of input data or the need to set up expert panels, both the PNV and the threshold approach require more (local/regional) knowledge and effort than the stratification approach, especially when trying to compare a (large) number of production sites located in different regions (*development effort*).

In contrast to the thresholds and PNV approaches, the stratification approach does not allow for binary solutions, i.e., to distinguish between sufficient and insufficient EIs (*positive and negative locations*). Results based on stratification need to be compared with a reference case, e.g., another LU/LC class, production region, etc. Therefore, they may not provide clear answers regarding their level of impact compared to other production sites. All approaches allow for a *relative comparison* of EIs of different LU/LC classes and between different land use systems and production systems in relation to each other or by relating the EIs of current LU/LC with PNV or threshold values.

Table 5.4: Characterizing the environmental stratification, PNV, and threshold approaches concerning their reliability, feasibility and relevance; the sub-criteria have been collected from [15, 22, 112-115]. Whether the sub-criteria are fulfilled is indicated as following: green: fulfillment, yellow: partial fulfillment, red: no fulfillment.

		Stratification	PNV	Thresholds
Reliability	Worldwide **consistent methodology** and datasets	green	yellow	red
	Regional stakeholders and experts involved	red	green	green
	Environmental heterogeneity within a region	yellow	green	red
	Range of environmental factors considered	red	green	yellow
Feasibility	**Global data coverage**	green	yellow	yellow
	Easily applicable for various environmental impacts	green	yellow	red
	Development effort if no data available	green	red	red
Relevance	Classification in **positive and negative locations**	red	yellow	green
	Relative comparison of globally distributed case studies	green	green	green

5.5 Discussion

5.5.1 Different approaches – congruent impact/sustainability assessments?

The stratification and PNV approaches show an increase in beneficial and a decrease in harmful EIs related to land use intensity, i.e., cropland > plantation forestry > forest, an effect consistent with results of Brockerhoff et al. [117]. Both approaches allow to compare the EI of biomass production between and within production regions and with each other. For plantation forestry (Satilla watershed and Rufiji basin) and forestry (Mulde watershed), forestry in the Mulde watershed had the positive EI values as similarly identified in the PNV and stratification approach. However, the relative order partly deviates. In that respect, it would produce the most positive EIs if we used biomass from forests in central Germany. Practical consequences would be to (i) increase the biomass sourcing from beneficial biomass production locations or to (ii) analyze which factors contribute to higher positive EIs, e.g., forest management or governance instruments such as certification schemes.

We compared the congruence of different approaches with the hotspot approach from ESS research [110, 27]. We could show that hotspots of the stratification approach and the PNV approach do not necessarily agree in their outcome. After quantifying the similarity between the different approaches with diagnostic test statistics, we found that defining hotspots based on quantiles may create deviating rankings in different land use systems depending (i) on the land use intensity or the similarity to the natural state in the region and (ii) on the set of EIs assessed. The comparison showed that missing baseline conditions (a natural or desired state of the environment) in the stratification approach with quantiles is less reliable in discovering strong or weak human modifications of the environment. Baseline conditions should provide or model EIs as independent as possible from current land use activities. For example, the share of plantation forestry in the Satilla watershed at the EI hotspots is significantly larger for the stratification approach (80%-quantile) (39%) than for the PNV approach (1%) for a comparable area (1.1 vs. 0.8 % of the watershed). Stratification may overrate the beneficial EI of plantation forestry. In the Rufiji basin, the areas classified as hotspots with the PNV approach account for 13.8% of the region, whereas the stratification approach only classified 0.7 % of the area as hotspots. In total, stratification more likely provides comparable results to PNV if the quantiles to determine hotspots are based on the individual EIs and are not set for the entire set of indicators. Therefore, in studies using a hotspot approach, it should be questioned, whether it is reasonable to aim for maximizing the set of EIs or ESS, see, e.g., [27]. It may be more reasonable to assess whether in an ecosystem or watershed EIs or ESS are balanced or not, see Foley et al. [24].

Threshold values to identify locations with high negative EIs may not always be useful. For instance, it is not if most of the area is below environmental thresholds as in our example for sediment and P concentrations. It may indicate (i) weak sustainability requirements or (ii) low EI/sustainable land use activities. If they indicate sustainable land use activities, thresholds may only identify very strong hotspots of negative EIs. Consequently, the EIs of biomass production would be negligible. In addition, thresholds may not be defined or available as in one of our examples, the Rufiji basin.

To consider stakeholders' preferences with respect to EIs or ESS, it is reasonable to give different EIs spatially explicit weights depending on their over- or undersupply. Locations of over- or undersupply may be regions or LU/LC classes that strongly deviate from the baseline derived in the

PNV approach. However, (participatory) weighting calls for transparent procedures, e.g., to enable comparisons between weighted and unweighted states. Obviously, differences in stakeholders' interests and regional preferences or local/regional regulations may lower the comparability of threshold-based approaches.

5.5.2 Reliability, feasibility and relevance of the approaches

The generally more reliable PNV approach can be improved by (i) assessing a consistent standard set of environmental factors, and (ii) providing more transparency to reveal remaining inconsistencies, e.g., with a protocol listing additional environmental factors and describing the modeling approach. The major advantage of stratification based on a global dataset used in this study is that we overcome the potential heterogeneity of the expert-based, regional PNV approach. The major advantage of PNV over stratification is that it is less dependent on the minima and maxima within the study. For example, if a study region is intensively used, sites with positive EIs may be very far from the natural state, but may not be revealed as such. This may be misleading in environmental or ESS assessments if sites with high and low management intensity are compared. From our production regions, we can conclude that information about land use intensity may help to enhance the reliability of the results, i.e., production regions with a more comparable land use intensity more likely agree between the PNV and stratification approach, see the hotspot example in section 5.4.2. The stratification approach should be amplified by complementary environmental factors and validated in additional studies to ensure a comprehensive and comparable removal of environmental heterogeneity. The threshold approach provides only a limited indication on the extent of the environmental impacts. It classifies land use in locations of positive and negative impacts based stakeholders' or governmental preferences or regulations. In that respect it provides clear answers for decision makers. Nevertheless, a general concept to adapt thresholds to environmental heterogeneity is not available, e.g., varying sediment loading rates depending on soil type and topography, or setting stricter threshold values for vulnerable ecosystems with respect to nutrient input, e.g., peatlands.

Environmental thresholds for positive EIs or ESS are not readily available except applications based on the concept of critical loads for environmental pollutants [114]. Major reasons may be that thresholds (i) require considerable effort to thoroughly consider local environmental conditions and (ii) may require different methodologies for their development for individual EIs or ESS, e.g., the concept of ESS capacities [118]. Such need for individual methodologies for each EI is less suitable for an increasing number of studies that assess sets of EIs or ESS and their hot- and coldpots as well as their interactions, see Mouchet et al. [28].

Recent papers discuss options to assess stakeholders' preferences regarding the quantity and type of acceptable EIs and the levels of positive EIs or ESS, with, e.g., thresholds [18]. Governments increasingly assess or prescribe to assess ESS sufficiency in general [18] or the acceptable level of EIs, e.g., in the context of biomass for bioenergy [13]. For example, certification schemes, e.g., for bioenergy or agriculture and forestry in general, partly require local environmental thresholds to ensure low levels of negative EIs [13]. Although both the stratification and PNV approach may overcome some reliability and feasibility deficiencies of thresholds, currently they do not seem to be in the focus of governments or authorities. One reason for the limited use of the PNV approach may be the fact that in many regions a natural state is hardly attainable after long histories of land use activities, even if they were stopped [119]. Nevertheless, we consider it advantageous that the PNV approach reveals EI-levels in relation to a reference system, i.e. the (potential) natural state.

5.6 Conclusions

We could show that the different approaches, stratification, PNV, and thresholds, in this study allow to compare EIs of biomass production between different world regions despite large environmental heterogeneity. However, the different approaches to control for environmental heterogeneity rank the biomass production systems in deviating orders. With respect to reliability, the PNV approach performs best, whereas this approach is less feasible and only partly relevant. Both the environmental stratification and threshold approaches revealed several conceptual deficiencies for the intended use. Our results indicate that the stratification approach is more feasible for application, while the threshold approach could be considered as more relevant, due to the closer links to stakeholders and authorities. To address the deficiencies of the PNV and stratification approach identified in this paper, additional comparative EI assessments are needed aiming at broader sets of land use activities and environmental conditions. Covering major combinations of environmental conditions and socio-economic factors, e.g., [120], it should be possible to determine the preferred and acceptable level of EIs. Such a categorization may help to determine critical intensities of land use relative to PNV. Furthermore, additional studies may enable us to identify conditions under which comparable results for the PNV and stratification approach can be obtained, or to which extent it is sufficient to apply the stratification approach with less environmental parameters compared with the more complex PNV approach. Therefore, further studies should investigate, which trade-offs arise between using less detailed, but homogenous global datasets and more detailed, but heterogeneous local/regional datasets.

5.7 Acknowledgments

This work was partly funded by the EU FP7 project SECTOR (Production of Solid Sustainable Energy Carriers by Means of Torrefaction), grant agreement number 282826, and the Helmholtz Programme “Terrestrial Environmental Research”. The authors would like to thank Christian Hoyer, Christian Schweitzer, and Charlotte Main for data preparation. The authors would also like to thank Matthew McCartney (IWMI) for providing data on the hydrology of the Rufiji basin. The authors would like to thank Christian Hoyer and Carola Pätzold for helpful comments on earlier versions of the manuscript.

5.8 References

[1] Lambin EF, Meyfroidt P. Global land use change, economic globalization, and the looming land scarcity. Proceedings of the National Academy of Sciences 2011;108(9):3465-72.

[2] Haberl H, Erb K-H, Krausmann F, Running S, Searchinger TD, Smith WK. Bioenergy: how much can we expect for 2050? Environmental Research Letters 2013;8(3):031004.

[3] Demirbas A. Political, economic and environmental impacts of biofuels: A review. Applied Energy 2009;86, Supplement 1(0):S108-S17.

[4] Lewandowski I, Faaij APC. Steps towards the development of a certification system for sustainable bio-energy trade. Biomass and Bioenergy 2006;30(2):83-104.

[5] Ragauskas AJ, Williams CK, Davison BH, Britovsek G, Cairney J, Eckert CA, et al. The path forward for biofuels and biomaterials. Science 2006;311(5760):484-9.

[6] WG III. IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Edenhofer O, Pichs-Madruga R, Sokona Y, Seyboth K, Matschoss P, Kadner S, et al., editors. Cambridge and New York: Cambridge University Press; 2012.

[7] Lamers P, Hoefnagels R, Junginger M, Hamelinck C, Faaij A. Global solid biomass trade for energy by 2020: an assessment of potential import streams and supply costs to North-West Europe under different sustainability constraints. GCB Bioenergy 2015;7(4):618-34.
[8] Directive 2009/28/EC. On the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.2009.04.23. Off J EU 2009. L 140: 16-62.
[9] Cherubini F. GHG balances of bioenergy systems – Overview of key steps in the production chain and methodological concerns. Renewable Energy 2010;35(7):1565-73.
[10] Wang M, Han J, Dunn JB, Cai H, Elgowainy A. Well-to-wheels energy use and greenhouse gas emissions of ethanol from corn, sugarcane and cellulosic biomass for US use. Environmental Research Letters 2012;7(4):045905.
[11] Dale VH, Lowrance R, Mulholland P, Robertson GP. Bioenergy sustainability at the regional scale. Ecology and Society 2010;15(4):23.
[12] Seppelt R, Dormann CF, Eppink FV, Lautenbach S, Schmidt S. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. Journal of Applied Ecology 2011;48(3):630-6.
[13] Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65:151-69.
[14] Van Dam J, Junginger M, Faaij APC. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renewable and Sustainable Energy Reviews 2010;14(9):2445-72.
[15] Stein A, Kreft H. Terminology and quantification of environmental heterogeneity in species-richness research. Biological Reviews 2015;90(3):815–36.
[16] Stein A, Gerstner K, Kreft H. Environmental heterogeneity as a universal driver of species richness across taxa, biomes and spatial scales. Ecology Letters 2014;17(7):866-80.
[17] Kienast F, Bolliger J, Potschin M, de Groot R, Verburg P, Heller I, et al. Assessing Landscape Functions with Broad-Scale Environmental Data: Insights Gained from a Prototype Development for Europe. Environmental Management 2009;44(6):1099-120.
[18] Wong CP, Jiang B, Kinzig AP, Lee KN, Ouyang Z. Linking ecosystem characteristics to final ecosystem services for public policy. Ecology Letters 2015;18(1):108-18.
[19] Geijzendorffer IR, Martín-López B, Roche PK. Improving the identification of mismatches in ecosystem services assessments. Ecological Indicators 2015;52(0):320-31.
[20] Villamagna AM, Angermeier PL, Bennett EM. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity 2013;15(0):114-21.
[21] Moldan B, Janoušková S, Hák T. How to understand and measure environmental sustainability: Indicators and targets. Ecological Indicators 2012;17(0):4-13.
[22] Willemen L, Veldkamp A, Verburg PH, Hein L, Leemans R. A multi-scale modelling approach for analysing landscape service dynamics. Journal of Environmental Management 2012;100(0):86-95.
[23] Metzger MJ, Rounsevell MDA, Acosta-Michlik L, Leemans R, Schröter D. The vulnerability of ecosystem services to land use change. Agriculture, Ecosystems & Environment 2006;114(1):69-85.
[24] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global Consequences of Land Use. Science 2005;309(5734):570-4.
[25] West PC, Gibbs HK, Monfreda C, Wagner J, Barford CC, Carpenter SR, et al. Trading carbon for food: Global comparison of carbon stocks vs. crop yields on agricultural land. Proceedings of the National Academy of Sciences 2010;107(46):19645-8.
[26] Crossman ND, Burkhard B, Nedkov S, Willemen L, Petz K, Palomo I, et al. A blueprint for mapping and modelling ecosystem services. Ecosystem Services 2013;4(0):4-14.

[27] Qiu J, Turner MG. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proceedings of the National Academy of Sciences 2013;110(29):12149-54.
[28] Mouchet MA, Lamarque P, Martín-López B, Crouzat E, Gos P, Byczek C, et al. An interdisciplinary methodological guide for quantifying associations between ecosystem services. Global Environmental Change 2014;28(0):298-308.
[29] Kurtz JC, Jackson LE, Fisher WS. Strategies for evaluating indicators based on guidelines from the Environmental Protection Agency's Office of Research and Development. Ecological Indicators 2001;1(1):49-60.
[30] Maltsoglou I, Khwaja Y. Bioenergy and food security - The BEFS analysis for Tanzania. Environment and Natural Resources. Working Paper. Rome: FAO; 2010.
[31] Com 2010/11 Report from the Commission to the Council and the European Parliament on sustainability requirements for the use of solid and gaseous biomass sources in electricity, heating and cooling. Publications Office 2010.
[32] Eggleston HS, Buendia L, Miwa K, Ngara T, Tanabe K. Guidelines for National Greenhouse Gas Inventories. National Greenhouse Gas Inventories Programme. Institute for Global Environmental Strategies; 2006.
[33] Kareiva P, Tallis H, Ricketts TH, Daily GC, Polasky S. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford: Oxford University Press; 2011.
[34] Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron D, et al. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 2009;7(1):4-11.
[35] Tallis HT, Ricketts T, Guerry AD, Wood SA, Sharp R, Nelson E, et al. InVEST 2.5.6 User's Guide. Stanford; 2013.
[36] Wischmeier WH, Smith DD. Predicting rainfall erosion losses: A guide to conservation planning. Washington, DC; 1978.
[37] Alkemade R, van Oorschot M, Miles L, Nellemann C, Bakkenes M, ten Brink B. GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss. Ecosystems 2009;12(3):374-90.
[38] Lehner B, Verdin K, Jarvis A. New Global Hydrography Derived From Spaceborne Elevation Data. Eos, Transactions American Geophysical Union 2008;89(10):93-4.
[39] Gap Analysis Program (GAP), National Land Cover, Version 2, 2011, U.S. Geological Survey, Reston.
[40] Defourny P, Vancutsem C, Bicheron P, Brockmann C, Nino F, Schouten L, et al. GLOBCOVER: a 300 m global land cover product for 2005 using Envisat MERIS time series. ISPRS Commission VII Mid-term Symposium "Remote Sensing From Pixels to Processes; 2006; Enschede.
[41] Green Resources, Reforestation of degraded grasslands in Uchindile & Mapanda, Tanzania, 2009, VCS, Washington, DC
[42] Hunting Technical Services, Tanzania Land Cover and Land Use, 1996, Tanzania Ministry of Natural Resources and Tourism, Hemel Hemstead.
[43] European Environment Agency (EEA), Corine Land Cover 2006 raster data, 2013, European Environment Agency.
[44] Wochele S, Priess, J., Thrän, D., O'Keeffe, S. Crop allocation model "CRAM" - an approach for dealing with biomass supply from arable land as part of a life cycle inventory. 22nd European Biomass Conference and Exhibition EU BC&E; 2014: ETA-Florence Renewable Energies.
[45] Landfire, Landfire 1.2.0 - Environmental Site Potential layer, 2010, (Geological UDotI, Survey), Wahington, DC.
[46] van Breugel P, Kindt R, Lillesø JP, Bingham M, Demissew S, Dudley C, et al., Potential Natural Vegetation Map of Eastern Africa: An interactive vegetation map for Ethiopia, Kenya, Malawi, Rwanda, Tanzania, Uganda and Zambia. Version 1.1., 2012, Forest and Landscape (Denmark)/ICRAF, Copenhagen/Nairobi.

[47] LfULG, Potentielle natürliche Vegetation (pnV) in Sachsen, 2011, Saxon State Office for Environment/Agriculture/Geology, Freiberg.
[48] FAO Geonetwork, Global map of monthly reference evapotranspiration - 10 arc minutes (GeoLayer), 2014, FAO, Rome, Italy.
[49] Daly C, Halbleib M, Smith JI, Gibson WP, Doggett MK, Taylor GH, et al. Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology 2008;28(15):2031-64.
[50] Huffman GJ, Stocker EF, Bolvin DT, Nelkin EJ, Adler RF, TRMM Version 7A 3B43 Data Sets, 2013, NASA/GSFC, Greenbelt.
[51] Jäckel G, Zink M, Marx A, Aufbereitung von gemessenen und simulierten Klimadaten, 2012, Mitteldeutsches Klimabüro, Leipzig.
[52] Soil Survey Staff, Web Soil Survey - Soil Survey Geographic (SSURGO) Database, 2014, Natural Resources Conservation Service (USDA).
[53] FAO/IIASA/ISRIC/ISSCAS/JRC, Harmonized World Soil Database (version 1.2), 2012, FAO/IIASA, Rome/Laxenburg.
[54] LfULG, Auswertekarten Bodenschutz 1:50.000, 2012, Saxon State Office for Environment/Agriculture/Geology, Freiberg.
[55] Panagos P, Van Liedekerke M, Montarella L, The European Soil Database distribution version 2.0, 2006, European Commission/European Soil Bureau Network, Ispra.
[56] Perica S, Martin D, Pavlovic S, Roy I, St. Laurent M, Trypaluk C, et al. Precipitation-Frequency Atlas of the United States. NOAA Atlas 14. Silver Spring: National Weather Service; 2013.
[57] Roose E. Land husbandry - Components and strategy. FAO Soils Bulletin. Rome; 1996.
[58] Hijmans RJ, Cameron SE, Parra JL, Jones PG, Jarvis A. Very high resolution interpolated climate surfaces for global land areas. International Journal of Climatology 2005;25(15):1965-78.
[59] Renard KG, Freimund JR. Using monthly precipitation data to estimate the R-factor in the revised USLE. Journal of Hydrology 1994;157(1–4):287-306.
[60] Bräunig A, Dokumentation zur Berechnung und Ableitung R-Faktor Sachsen, 2013, LfULG, Freiberg.
[61] Bischoff R, K-Faktor (MMK 100), 2014, TLUG, Jena.
[62] Burns LA, Allen BW, Barber MC, Bird SL, Cheplick JM, Fendley MJ, et al. Pesticide and Industrial Chemical Risk Analysis and Hazard Assessment, Version 3.0. Athens: Environmental Research Laboratory, Office of Research and Development; 1992.
[63] Canadell J, Jackson RB, Ehleringer JB, Mooney HA, Sala OE, Schulze ED. Maximum rooting depth of vegetation types at the global scale. Oecologia 1996;108(4):583-95.
[64] Mishra HS, Rathore TR, Pant RC. Root growth, water potential, and yield of irrigated rice. Irrigation Science 1997;17(2):69-75.
[65] Schenk HJ, Jackson RB. The Global Biogeography of Roots. Ecological Monographs 2002;72(3):311-28.
[66] USDA-NRCS. Engineering Field Handbook. Washington, DC: USDA Soil Conservation Service; 2012.
[67] Leh MDK, Matlock MD, Cummings EC, Nalley LL. Quantifying and mapping multiple ecosystem services change in West Africa. Agriculture, Ecosystems & Environment 2013;165(0):6-18.
[68] Campbell Group. Juvenile Pine Plantation Fertilization recommendations. Waycross: Georgia Biomass; 2013.
[69] Reckhow KH, Beaulac M, Simpson J. Modeling Phosphorous Loading and Lake Response under Uncertainty: A Manual and Compilation of Export Coefficients Washington, DC; 1980.
[70] Dissmeyer GE, Foster GR. A guide for predicting sheet and rill erosion on forest land. Technical publication: Forest Service, Southern Region; 1984.

[71] Fernandez C, Wu JQ, McCool DK, Stöckle CO. Estimating water erosion and sediment yield with GIS, RUSLE, and SEDD. Journal of Soil and Water Conservation 2003;58(3):128-36.
[72] Institute of Water Research. C Factor used for erosion prediction in Michigan. Michigan State University; 2002 [cited 2013 January 20]; Available from: http://www.iwr.msu.edu/rusle/doc/cfactors.pdf.
[73] Martin A, Gunter J, Regens J. Estimating erosion in a riverine watershed Bayou Liberty-Tchefuncta river in Louisiana. Environmental Science and Pollution Research 2003;10(4):245-50.
[74] Morgan R. Soil erosion and conservation. 3rd ed. Oxford: Wiley-Blackwell; 2005.
[75] USDA NRCS and ARS. Revised Universal Soil Loss Equation (RUSLE) - EPA Pesticide Project. Washington, DC: National Resources Conservation Service and Agricultural Research Service 2000.
[76] Suárez LA. PRZM-3, A Model for Predicting Pesticide and Nitrogen Fate in the Crop Root and Unsaturated Soil Zones: Users Manual for Release 3.12.2. Washington, DC: Ecosystems Research Division, National Exposure Research Laboratory; 2005.
[77] SMUL. Erosionsminderung. 2013 [cited 2014 August 25]; Available from: http://www.umwelt.sachsen.de/umwelt/4566.asp.
[78] Franzluebbers AJ. Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil and Tillage Research 2005;83(1):120-47.
[79] Ruesch A, Gibbs H. New IPCC Tier-1 Global Biomass Carbon Map for the Year 2000. Oak Ridge: Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory; 2008 [cited 2014 January 14]; Available from: http://cdiac.ornl.gov/epubs/ndp/global_carbon/carbon_documentation.html.
[80] Smith JE, Heath LS, Skog KE, Birdsey RA. Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States. General Technical Report NE. Delaware: Northern Research Station, Forest Service; 2006.
[81] Willcock S, Phillips OL, Platts PJ, Balmford A, Burgess ND, Lovett JC, et al. Towards Regional, Error-Bounded Landscape Carbon Storage Estimates for Data-Deficient Areas of the World. PLoS ONE 2012;7(9):e44795.
[82] BGR, Nutzungsdifferenzierte Bodenübersichtskarte der Bundesrepublik Deutschland 1:1.000.000 (BÜK1000N) (Serie), 2013, Bundesamt für Geowissenschaften und Rohstoffe, Hannover.
[83] Bohn U, Hettwer C, Gollub G. Anwendung und Auswertung der Karte der natürlichen Vegetation Europas : Beiträge und Ergebnisse des internationalen Workshops auf der Insel Vilm. Bonn: Deutschland / Bundesamt für Naturschutz; 2005.
[84] Fraver S, Wagner RG, Day M. Dynamics of coarse woody debris following gap harvesting in the Acadian forest of central Maine, U.S.A. Canadian Journal of Forest Research 2002;32(12):2094-105.
[85] Müller-Using S, Bartsch N. Decay dynamic of coarse and fine woody debris of a beech (Fagus sylvatica L.) forest in Central Germany. European Journal of Forest Research 2009;128(3):287-96.
[86] Polley H, Henning P. Die Bundeswaldinventur in der Geodateninfrastruktur des Thünen-Instituts (BWI 2002). Eberswalde2012 [cited 2014 August 18]; Available from: https://gdi.ti.bund.de/geoserver/bwi_2009dt/ows?SERVICE=WFS&REQUEST=GetCapabilities.
[87] Strogies M, Gniffke P. Berichterstattung unter der Klimarahmenkonvention der Vereinten Nationen und dem Kyoto-Protokoll 2012 - Nationaler Inventarbericht zum Deutschen Treibhausgasinventar 1990 – 2010. Dessau: Umweltbundesamt; 2012.
[88] Wördehoff R, Spellmann H, Evers J, Nagel J. Kohlenstoffstudie Forst und Holz Niedersachsen. Göttingen: Universitätsverlag Göttingen; 2011.
[89] Center for International Earth Science Information Network - CIESIN - Columbia University, Centro Internacional de Agricultura Tropical - CIAT. Gridded Population of the World, Version 3

(GPWv3): Population Density Grid, Future Estimates. NASA Socioeconomic Data and Applications Center (SEDAC); 2005.
[90] Tatem AJ, Noor AM, von Hagen C, Di Gregorio A, Hay SI. High Resolution Population Maps for Low Income Nations: Combining Land Cover and Census in East Africa. PLoS ONE 2007;2(12):e1298.
[91] Priess JA, Heinze M, Egli L, Pomm S, Lang E, Masurowski F, et al. Assessing some of the social and environmental consequences of changes in population and settlement pattern expected until 2050 for a Central European region. Environmental Modelling & Software 2015;under review.
[92] TIGER, Rails National Geodatabase, 2013, U.S. Census Bureau, Washington, DC.
[93] TIGER, Roads National Geodatabase, 2013, U.S. Census Bureau, Washington, DC.
[94] FAO Geonetwork, Roads of Tanzania - AFRICOVER, 2014, FAO, Rome.
[95] BKG, Digitales Basis-Landschaftsmodell - Basis-DLM, 2014, Bundesamt für Kartographie und Geodäsie, Leipzig.
[96] Illinois State Water Survey, National Atmospheric Deposition Program (NRSP-3) - CLAD NTN Maps, 2013, NADP Program Office, Champaign.
[97] Builtjes P, Hendriks E, Koenen M, Schaap M, Banzhaf S, Kerschbaumer A, et al., Abschlussbericht zum UFOPLAN-Vorhaben FKZ 3707 64 200: Erfassung, Prognose und Bewertung von Stoffeinträgen und ihren Wirkungen in Deutschland (Modelling of Air Pollutants and Ecosystem Impact – MAPESI). 2011, Federal Environmental Agency, Dessau-Rosslau.
[98] Blett TF, Lynch JA, Pardo LH, Huber C, Haeuber R, Pouyat R. FOCUS: A pilot study for national-scale critical loads development in the United States. Environmental Science & Policy 2014;38(0):225-36.
[99] Pardo LH, Fenn ME, Goodale CL, Geiser LH, Driscoll CT, Allen EB, et al. Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. Ecological Applications 2011;21(8):3049-82.
[100] Olson DM, Dinerstein E, Wikramanayake ED, Burgess ND, Powell GVN, Underwood EC, et al. Terrestrial Ecoregions of the World: A New Map of Life on Earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. BioScience 2001;51(11):933-8.
[101] Metzger MJ, Bunce RGH, Jongman RHG, Sayre R, Trabucco A, Zomer R. A high-resolution bioclimate map of the world: a unifying framework for global biodiversity research and monitoring. Global Ecology and Biogeography 2013;22(5):630-8.
[102] EPD. Satilla River Basin Dissolved Oxygen TMDLs. Atlanta: Georgia Department of Natural Resources; 2001.
[103] EPD. Total Maximum Daily Load Evaluation for Headstall Creek in the Savannah River Basin for Sediment. Atlanta: Georgia Department of Natural Resources; 2005.
[104] SMUL, PLa, POh. Bericht über die Entwicklung der Wasserbeschaffenheit in ausgewählten deutsch - tschechischen Grenzwasserläufen - Auswertung der abgestimmten physikalischen, chemischen und biologischen Kenngrößen von 1993 bis 2006. Sächsische Landesamt für Umwelt und Geologie, Povodí Labe, Povodí Ohře; 2007.
[105] Hainz-Renetzeder C, Schneidergruber A, Kuttner M, Wrbka T. Assessing the potential supply of landscape services to support ecological restoration of degraded landscapes: A case study in the Austrian-Hungarian trans-boundary region of Lake Neusiedl. Ecological Modelling 2015;295(0):196-206.
[106] Munoz-Carpena R, Vellidis G, Shirmohammadi A, Wallender W. Evaluation of modeling tools for TMDL development and implementation. Transactions of the ASABE 2006;49(4):961-5.
[107] Ludwig W, Dumont E, Meybeck M, Heussner S. River discharges of water and nutrients to the Mediterranean and Black Sea: Major drivers for ecosystem changes during past and future decades? Progress in Oceanography 2009;80(3–4):199-217.
[108] Altman DG, Bland JM. Diagnostic tests 1: sensitivity and specificity. BMJ 1994;308:1552.
[109] R Development Core Team. R: A language and environment for statistical computing. 2009 [cited 2013 November 05]; Available from: http://www.r-project.org/.

[110] Eigenbrod F, Armsworth PR, Anderson BJ, Heinemeyer A, Gillings S, Roy DB, et al. The impact of proxy-based methods on mapping the distribution of ecosystem services. Journal of Applied Ecology 2010;47(2):377-85.
[111] Seppelt R, Fath B, Burkhard B, Fisher JL, Grêt-Regamey A, Lautenbach S, et al. Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. Ecological Indicators 2012;21(0):145-54.
[112] Chiarucci A, Araújo MB, Decocq G, Beierkuhnlein C, Fernández-Palacios JM. The concept of potential natural vegetation: an epitaph? Journal of Vegetation Science 2010;21(6):1172-8.
[113] Metzger MJ, Brus DJ, Bunce RGH, Carey PD, Gonçalves J, Honrado JP, et al. Environmental stratifications as the basis for national, European and global ecological monitoring. Ecological Indicators 2013;33(0):26-35.
[114] Groffman P, Baron J, Blett T, Gold A, Goodman I, Gunderson L, et al. Ecological Thresholds: The Key to Successful Environmental Management or an Important Concept with No Practical Application? Ecosystems 2006;9(1):1-13.
[115] Bishop K, Beven K, Destouni G, Abrahamsson K, Andersson L, Johnson RK, et al. Nature as the “Natural” Goal for Water Management: A Conversation. AMBIO 2009;38(4):209-14.
[116] Birk S, Bonne W, Borja A, Brucet S, Courrat A, Poikane S, et al. Three hundred ways to assess Europe's surface waters: An almost complete overview of biological methods to implement the Water Framework Directive. Ecological Indicators 2012;18(0):31-41.
[117] Brockerhoff E, Jactel H, Parrotta J, Quine C, Sayer J. Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity and Conservation 2008;17(5):925-51.
[118] Schröter M, Barton DN, Remme RP, Hein L. Accounting for capacity and flow of ecosystem services: A conceptual model and a case study for Telemark, Norway. Ecological Indicators 2014;36(0):539-51.
[119] Somodi I, Molnár Z, Ewald J. Towards a more transparent use of the potential natural vegetation concept – an answer to Chiarucci et al. Journal of Vegetation Science 2012;23(3):590-5.
[120] Václavík T, Lautenbach S, Kuemmerle T, Seppelt R. Mapping global land system archetypes. Global Environmental Change 2013;23(6):1637-47.

5.9 Supporting Information

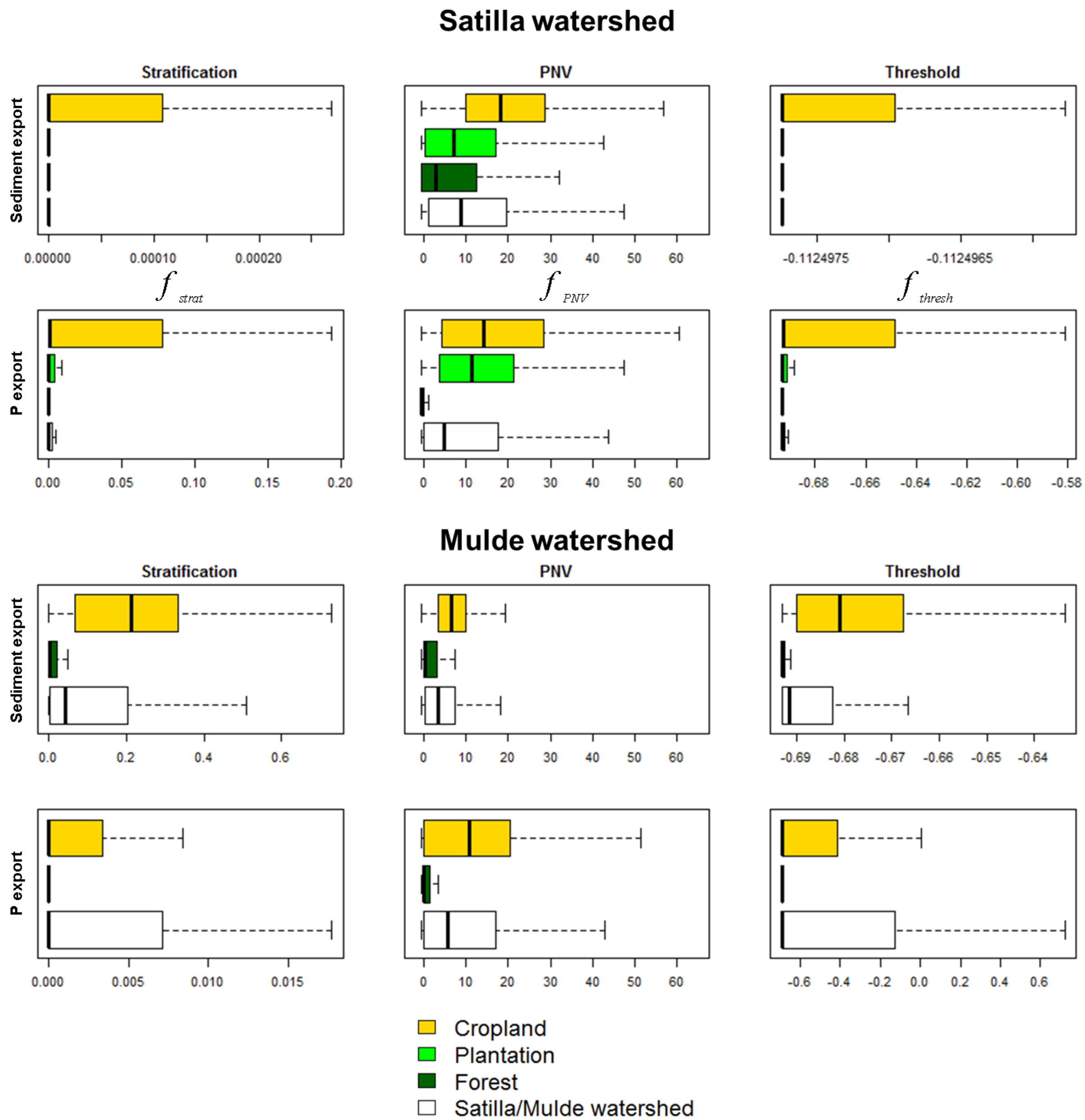

Fig. 5.S1: Negative EIs standardized with the stratification, PNV, and threshold approach to control for environmental heterogeneity. After applying the stratification, PNV and threshold approach, we can compare each EI for different LU/LC (cropland (yellow), plantation forestry (light green) and natural or semi-natural forest (dark green)) between and within different case studies. For the stratification approach, each EI is standardized with f_{strat} for each environmental stratum i, i.e., homogeneous bioclimatic groups; for the PNV approach, the relative difference between PNV and the current Environmental impact is displayed with f_{PNV}; values >0 indicate higher EIs in the current LU/LC than for PNV. For the PNV and threshold approach, the relative difference between PNV/threshold and the current Environmental impact is displayed with f_{PNV} and f_{thresh} respectively; values >0 indicate higher a higher EI creation in the current LU/LC than for PNV/than the threshold. For better visualization, the input data for f_{strat} as well as the scores f_{PNV} and f_{thresh} have been transformed with $f_{transform}$.

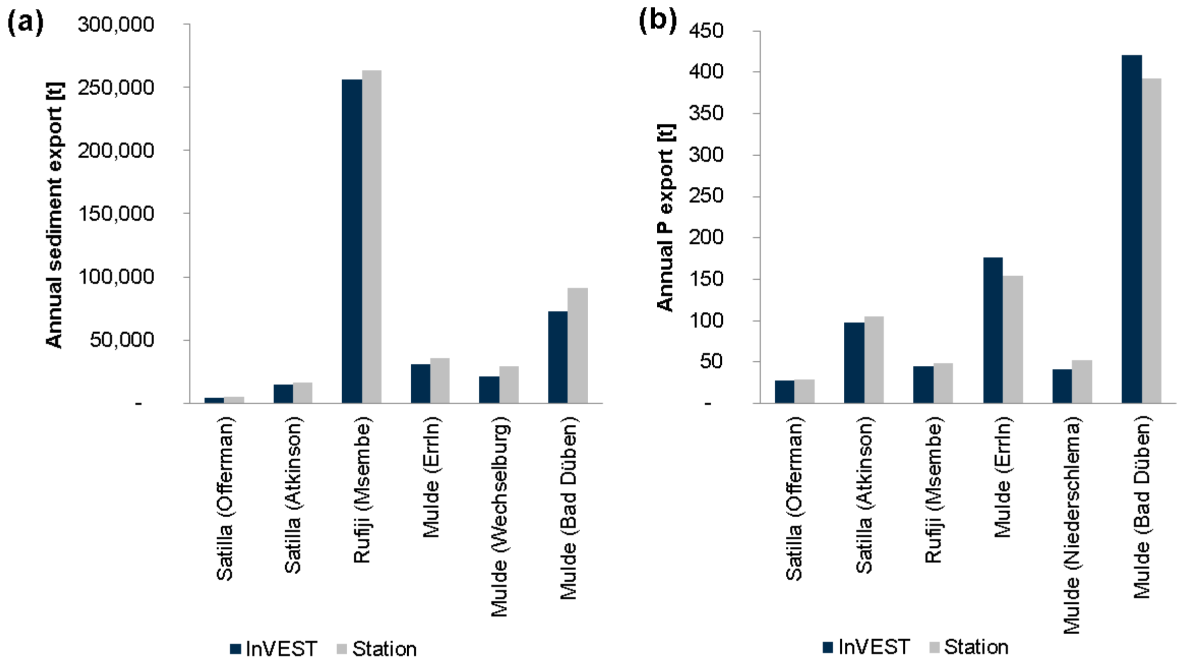

Fig. 5.S2: Comparison of measured and modeled annual sediment (a) and P export (b) for Fig. 5.2 and Fig. 5.S1; the station names for the case studies are listed in brackets. The agreement between measured and modeled annual P and sediment export ranges from 72 to 114 percent.

Table 5.S1: Arithmetic mean of positive and negative EIs for the stratification, PNV, and threshold approaches of the data displayed in Fig. 5.2 and Fig. 5.S1.

		Carbon storage		Sediment		P retention		Biodiversity		Sediment export		P export	
		mean	SD	mean	SD	mean	SD	mean	SD	mean	SD	mean	SD
Satilla watershed	***Stratification***												
	Study area	0.63	0.13	0.03	0.06	0.09	0.12			0.00	0.01	0.04	0.11
	Forest	0.74	0.02	0.02	0.05	0.07	0.12			0.00	0.00	0.00	0.00
	Plantation	**0.69**	0.08	**0.02**	0.04	**0.09**	0.11			**0.00**	0.00	**0.03**	0.08
	Cropland	0.45	0.00	0.08	0.09	0.14	0.14			0.00	0.01	0.10	0.19
	PNV												
	Study area	-0.47	0.36	1.85	2.11	4.31	9.63	-0.88	0.08	12.40	12.97	9.60	11.74
	Forest	-0.18	0.22	0.96	1.76	1.88	6.04	-0.81	0.09	7.66	10.91	0.16	3.28
	Plantation	**-0.30**	0.30	**1.10**	1.76	**4.76**	10.09	**-0.91**	0.04	**10.59**	11.78	**13.06**	10.81
	Cropland	-0.89	0.02	4.02	1.48	6.98	11.86	-0.92	0.04	20.46	13.36	15.97	12.88
	Threshold												
	Study area									-0.11	0.00	-0.65	0.15
	Forest									-0.11	0.00	-0.69	0.00
	Plantation									**-0.11**	0.00	**-0.67**	0.08
	Cropland									-0.11	0.00	-0.56	0.32
Rufiji basin	***Stratification***												
	Study area	0.49	0.18	0.30	0.16	0.04	0.09						
	Forest	0.60	0.14	0.30	0.14	0.02	0.05						
	Plantation	**0.57**	0.06	**0.46**	0.08	**0.03**	0.09						
	Cropland	0.15	0.05	0.52	0.18	0.02	0.05						
	PNV												
	Study area	-0.08	0.27	1.14	3.46	3.82	10.56	-0.46	0.23				
	Forest	-0.02	0.28	0.69	4.07	3.95	10.83	-0.40	0.23				
	Plantation	**-0.23**	0.19	**1.58**	0.69	**3.68**	10.53	**-0.80**	0.12				
	Cropland	-0.46	0.10	4.31	5.59	3.05	9.74	-0.74	0.13				
Mulde watershed	***Stratification***												
	Study area	0.56	0.21	0.04	0.06	0.08	0.09			0.12	0.15	0.07	0.18
	Forest	**0.89**	0.04	**0.05**	0.06	**0.05**	0.06			**0.02**	0.05	**0.00**	0.02
	Cropland	0.44	0.06	0.04	0.06	0.09	0.08			0.22	0.17	0.02	0.06
	PNV												
	Study area	-0.51	0.27	-0.03	0.79	0.95	1.80	-0.87	0.08	4.54	4.63	9.95	11.63
	Forest	**-0.10**	0.17	**0.35**	0.52	**0.37**	1.45	**-0.77**	0.09	**2.18**	3.71	**3.44**	7.21
	Cropland	-0.66	0.02	-0.12	0.69	1.40	1.62	-0.90	0.00	6.92	4.53	12.54	11.72
	Threshold												
	Study area									-0.68	0.02	0.27	1.90
	Forest									**-0.69**	0.00	**-0.55**	0.60
	Cropland									-0.67	0.03	-0.06	1.29

Table 5.S2: Agreeing options between the stratification (10%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	270,360	53,269	2	6,166,658	1.00	0.01	0.04	1.00
		Sediment retention	5,010,177	81,589	831,083	567,440	0.86	0.13	0.90	0.09
		P retention	4,497,642	-	1,992,647	-	0.69	NaN	1.00	-
Rufiji	Stratification/PNV	C storage	688,086	97,797	1,216	912,818	1.00	0.10	0.43	0.99
		Sediment retention	1,104,436	84,861	425,489	85,131	0.72	0.50	0.93	0.17
		P retention	985,354	-	714,563	-	0.58	NaN	1.00	-
Mulde	Stratification/PNV	C storage	609,293	906,832	-	7,592,710	1.00	0.11	0.07	1.00
		Sediment retention	3,477,892	588,579	4,720,059	322,305	0.42	0.65	0.92	0.11
		P retention	5,339,562	-	3,769,273	-	0.59	NaN	1.00	-
Satilla	Stratification/PNV	Sediment export	5,377,418	-	1,112,871	-	0.83	NaN	1.00	-
		P export	5,131,148	-	1,359,141	-	0.79	NaN	1.00	-
	Stratification/threshold	Sediment export	6,490,284	-	5	-	1.00	NaN	1.00	-
		P export	6,397,447	-	92,842	-	0.99	NaN	1.00	-
Mulde	Stratification/PNV	Sediment export	8,639,093	-	469,742	-	0.95	NaN	1.00	-
		P export	8,009,905	-	1,098,930	-	0.88	NaN	1.00	-
	Stratification/threshold	Sediment export	9,108,772	-	63	-	1.00	NaN	1.00	-
		P export	6,881,740	-	2,227,095	-	0.76	NaN	1.00	-

Table 5.S3: Agreeing options between the stratification (20%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	270,207	1,212,102	155	5,007,825	1.00	0.19	0.05	1.00
		Sediment retention	4,556,138	276,579	636,093	1,021,479	0.88	0.21	0.82	0.30
		P retention	4,497,642	-	1,992,647	-	0.69	NaN	1.00	-
Rufiji	Stratification/PNV	C storage	674,850	214,334	14,452	796,281	0.98	0.21	0.46	0.94
		Sediment retention	1,003,352	153,769	356,581	186,215	0.74	0.45	0.84	0.30
		P retention	985,354	-	714,563	-	0.58	NaN	1.00	-
Mulde	Stratification/PNV	C storage	609,293	1,819,809	-	6,679,733	1.00	0.21	0.08	1.00
		Sediment retention	3,400,147	1,421,717	3,886,921	400,050	0.47	0.78	0.89	0.27
		P retention	5,339,562	-	3,769,273	-	0.59	NaN	1.00	-
Satilla	Stratification/PNV	Sediment export	5,377,418	-	1,112,871	-	0.83	NaN	1.00	-
		P export	5,131,148	-	1,359,141	-	0.79	NaN	1.00	-
	Stratification/threshold	Sediment export	6,490,284	-	5	-	1.00	NaN	1.00	-
		P export	6,397,447	-	92,842	-	0.99	NaN	1.00	-
Mulde	Stratification/PNV	Sediment export	6,912,367	95,040	374,702	1,726,726	0.95	0.05	0.80	0.20
		P export	8,009,905	-	1,098,930	-	0.88	NaN	1.00	-
	Stratification/threshold	Sediment export	7,287,006	-	63	1,821,766	1.00	-	0.80	-
		P export	6,881,740	-	2,227,095	-	0.76	NaN	1.00	-

Table 5.S4: Agreeing options between the stratification (30%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	270,207	1,212,102	155	5,007,825	1.00	0.19	0.05	1.00
		Sediment retention	4,086,831	456,301	456,371	1,490,786	0.90	0.23	0.73	0.50
		P retention	4,497,642	-	1,992,647	-	0.69	NaN	1.00	-
Rufiji	Stratification/PNV	C storage	618,021	373,125	71,281	637,490	0.90	0.37	0.49	0.84
		Sediment retention	895,844	216,252	294,098	293,723	0.75	0.42	0.75	0.42
		P retention	985,354	-	714,563	-	0.58	NaN	1.00	-
Mulde	Stratification/PNV	C storage	609,293	2,656,909	-	5,842,633	1.00	0.31	0.09	1.00
		Sediment retention	3,294,289	2,226,743	3,081,895	505,908	0.52	0.81	0.87	0.42
		P retention	5,339,562	-	3,769,273	-	0.59	NaN	1.00	-
Satilla	Stratification/PNV	Sediment export	3,727,062	285,463	827,408	1,650,356	0.82	0.15	0.69	0.26
		P export	5,131,148	-	1,359,141	-	0.79	NaN	1.00	-
	Stratification/threshold	Sediment export	4,554,465	-	5	1,935,819	1.00	-	0.70	-
		P export	6,397,447	-	92,842	-	0.99	NaN	1.00	-
Mulde	Stratification/PNV	Sediment export	6,120,743	214,301	255,441	2,518,350	0.96	0.08	0.71	0.46
		P export	8,009,905	-	1,098,930	-	0.88	NaN	1.00	-
	Stratification/threshold	Sediment export	6,376,121	-	63	2,732,651	1.00	-	0.70	-
		P export	6,881,740	-	2,227,095	-	0.76	NaN	1.00	-

Table 5.S5: Agreeing options between the stratification (40%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	268,510	2,215,631	1,852	4,004,296	0.99	0.36	0.06	1.00
		Sediment retention	3,605,439	623,938	288,734	1,972,178	0.93	0.24	0.65	0.68
		P retention	3,728,675	1,827,149	165,498	768,967	0.96	0.70	0.83	0.92
Rufiji	Stratification/PNV	C storage	553,655	519,129	135,647	491,486	0.80	0.51	0.53	0.79
		Sediment retention	782,887	273,287	237,063	406,680	0.77	0.40	0.66	0.54
		P retention	985,354	-	714,563	-	0.58	NaN	1.00	-
Mulde	Stratification/PNV	C storage	609,293	3,522,932	-	4,976,610	1.00	0.41	0.11	1.00
		Sediment retention	3,140,597	2,983,934	2,324,704	659,600	0.57	0.82	0.83	0.56
		P retention	4,962,656	3,266,555	502,718	376,906	0.91	0.90	0.93	0.87
Satilla	Stratification/PNV	Sediment export	3,167,185	385,832	727,039	2,210,233	0.81	0.15	0.59	0.35
		P export	5,131,148	-	1,359,141	-	0.79	NaN	1.00	-
	Stratification/threshold	Sediment export	3,894,219	-	5	2,596,065	1.00	-	0.60	-
		P export	6,397,447	-	92,842	-	0.99	NaN	1.00	-
Mulde	Stratification/PNV	Sediment export	5,320,868	325,309	144,433	3,318,225	0.97	0.09	0.62	0.69
		P export	5,170,809	804,438	294,492	2,839,096	0.95	0.22	0.65	0.73
	Stratification/threshold	Sediment export	5,465,238	-	63	3,643,534	1.00	-	0.60	-
		P export	3,238,206	-	2,227,095	3,643,534	0.59	-	0.47	-

Table 5.S6: Agreeing options between the stratification (median) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	255,720	3,228,388	14,642	2,991,539	0.95	0.52	0.08	1.00
		Sediment retention	3,100,245	767,772	144,900	2,477,372	0.96	0.24	0.56	0.84
		P retention	3,242,460	1,989,962	2,685	1,255,182	1.00	0.61	0.72	1.00
Rufiji	Stratification/PNV	C storage	723,080	499,674	280,133	197,030	0.72	0.72	0.79	0.64
		Sediment retention	666,811	327,202	183,148	522,756	0.78	0.38	0.56	0.64
		P retention	533,023	397,626	316,937	452,331	0.63	0.47	0.54	0.56
Mulde	Stratification/PNV	C storage	609,293	4,513,352	-	3,986,190	1.00	0.53	0.13	1.00
		Sediment retention	2,911,792	3,666,012	1,642,626	888,405	0.64	0.80	0.77	0.69
		P retention	4,360,046	3,574,901	194,372	979,516	0.96	0.78	0.82	0.95
Satilla	Stratification/PNV	Sediment export	2,622,905	490,631	622,240	2,754,513	0.81	0.15	0.49	0.44
		P export	3,237,302	1,351,297	7,844	1,893,846	1.00	0.42	0.63	0.99
	Stratification/threshold	Sediment export	3,245,140	-	5	3,245,144	1.00	-	0.50	-
		P export	3,152,304	-	92,842	3,245,143	0.97	-	0.49	-
Mulde	Stratification/PNV	Sediment export	4,512,454	427,778	41,964	4,126,639	0.99	0.09	0.52	0.91
		P export	4,368,449	912,961	185,969	3,641,456	0.96	0.20	0.55	0.83
	Stratification/threshold	Sediment export	4,554,355	-	63	4,554,417	1.00	-	0.50	-
		P export	2,327,323	-	2,227,095	4,554,417	0.51	-	0.34	-

Table 5.S7: Agreeing options between the stratification (60%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	255,720	3,228,388	14,642	2,991,539	0.95	0.52	0.08	1.00
		Sediment retention	2,543,874	860,430	52,242	3,033,743	0.98	0.22	0.46	0.94
		P retention	2,595,995	1,992,517	130	1,901,647	1.00	0.51	0.58	1.00
Rufiji	Stratification/PNV	C storage	506,254	831,224	183,048	179,391	0.73	0.82	0.74	0.82
		Sediment retention	547,454	377,837	132,513	642,113	0.81	0.37	0.46	0.74
		P retention	477,165	511,755	202,808	508,189	0.70	0.50	0.48	0.72
Mulde	Stratification/PNV	C storage	609,293	5,370,267	-	3,129,275	1.00	0.63	0.16	1.00
		Sediment retention	2,581,392	4,246,496	1,062,142	1,218,805	0.71	0.78	0.68	0.80
		P retention	3,555,997	3,681,736	87,537	1,783,565	0.98	0.67	0.67	0.98
Satilla	Stratification/PNV	Sediment export	2,088,435	605,190	507,681	3,288,983	0.80	0.16	0.39	0.54
		P export	2,591,079	1,354,102	5,039	2,540,069	1.00	0.35	0.50	1.00
	Stratification/threshold	Sediment export	2,596,111	-	5	3,894,173	1.00	-	0.40	-
		P export	2,503,276	-	92,842	3,894,171	0.96	-	0.39	-
Mulde	Stratification/PNV	Sediment export	3,637,570	463,778	5,964	5,001,523	1.00	0.08	0.42	0.99
		P export	3,520,086	975,482	123,448	4,489,819	0.97	0.18	0.44	0.89
	Stratification/threshold	Sediment export	3,643,471	-	63	5,465,301	1.00	-	0.40	-
		P export	1,416,439	-	2,227,095	5,465,301	0.39	-	0.21	-

Table 5.S8: Agreeing options between the stratification (70%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	255,264	3,932,250	15,098	2,287,677	0.94	0.63	0.10	1.00
		Sediment retention	1,935,790	901,375	11,297	3,641,827	0.99	0.20	0.35	0.99
		P retention	1,947,079	1,992,639	8	2,550,563	1.00	0.44	0.43	1.00
Rufiji	Stratification/PNV	C storage	451,033	910,446	238,269	100,169	0.65	0.90	0.82	0.79
		Sediment retention	423,342	423,717	86,633	766,225	0.83	0.36	0.36	0.83
		P retention	389,707	594,295	120,268	595,647	0.76	0.50	0.40	0.83
Mulde	Stratification/PNV	C storage	609,293	6,361,460	-	2,138,082	1.00	0.75	0.22	1.00
		Sediment retention	2,136,183	4,712,170	596,468	1,664,014	0.78	0.74	0.56	0.89
		P retention	2,700,669	3,737,291	31,982	2,638,893	0.99	0.59	0.51	0.99
Satilla	Stratification/PNV	Sediment export	1,569,502	735,286	377,585	3,807,916	0.81	0.16	0.29	0.66
		P export	1,944,185	1,356,239	2,902	3,186,963	1.00	0.30	0.38	1.00
	Stratification/threshold	Sediment export	1,947,082	-	5	4,543,202	1.00	-	0.30	-
		P export	1,854,245	-	92,842	4,543,202	0.95	-	0.29	-
Mulde	Stratification/PNV	Sediment export	2,731,419	468,510	1,232	5,907,674	1.00	0.07	0.32	1.00
		P export	2,658,338	1,024,617	74,313	5,351,567	0.97	0.16	0.33	0.93
	Stratification/threshold	Sediment export	2,732,588	-	63	6,376,184	1.00	-	0.30	-
		P export	505,556	-	2,227,095	6,376,184	0.19	-	0.07	-

Table 5.S9: Agreeing options between the stratification (80%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	255,264	3,932,250	15,098	2,287,677	0.94	0.63	0.10	1.00
		Sediment retention	1,295,810	910,424	2,248	4,281,807	1.00	0.18	0.23	1.00
		P retention	1,298,056	1,992,645	2	3,199,586	1.00	0.38	0.29	1.00
Rufiji	Stratification/PNV	C storage	451,033	910,446	238,269	100,169	0.65	0.90	0.82	0.79
		Sediment retention	292,428	462,794	47,556	897,139	0.86	0.34	0.25	0.91
		P retention	284,942	659,521	55,042	700,412	0.84	0.48	0.29	0.92
Mulde	Stratification/PNV	C storage	608,219	7,285,954	1,074	1,213,588	1.00	0.86	0.33	1.00
		Sediment retention	1,575,703	5,062,574	246,064	2,224,494	0.86	0.69	0.41	0.95
		P retention	1,810,236	3,757,742	11,531	3,529,326	0.99	0.52	0.34	1.00
Satilla	Stratification/PNV	Sediment export	1,081,472	896,285	216,586	4,295,946	0.83	0.17	0.20	0.81
		P export	1,297,215	1,358,298	843	3,833,933	1.00	0.26	0.25	1.00
	Stratification/threshold	Sediment export	1,298,053	-	5	5,192,231	1.00	-	0.20	-
		P export	1,205,216	-	92,842	5,192,231	0.93	-	0.19	-
Mulde	Stratification/PNV	Sediment export	1,821,485	469,460	282	6,817,608	1.00	0.06	0.21	1.00
		P export	1,793,630	1,070,793	28,137	6,216,275	0.98	0.15	0.22	0.97
	Stratification/threshold	Sediment export	1,821,704	-	63	7,287,068	1.00	-	0.20	-
		P export	10	405,338	1,821,757	6,881,730	0.00	0.06	0.00	0.18

Table 5.S10: Agreeing options between the stratification (90%-quantile) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	254,524	5,366,635	15,838	853,292	0.94	0.86	0.23	1.00
		Sediment retention	648,305	911,948	724	4,929,312	1.00	0.16	0.12	1.00
		P retention	649,029	1,992,647	-	3,848,613	1.00	0.34	0.14	1.00
Rufiji	Stratification/PNV	C storage	451,033	910,446	238,269	100,169	0.65	0.90	0.82	0.79
		Sediment retention	152,961	493,319	17,031	1,036,606	0.90	0.32	0.13	0.97
		P retention	158,208	702,779	11,784	827,146	0.93	0.46	0.16	0.98
Mulde	Stratification/PNV	C storage	606,071	8,194,645	3,222	304,897	0.99	0.96	0.67	1.00
		Sediment retention	872,486	5,270,240	38,398	2,927,711	0.96	0.64	0.23	0.99
		P retention	909,280	3,767,669	1,604	4,430,282	1.00	0.46	0.17	1.00
Satilla	Stratification/PNV	Sediment export	609,932	1,073,774	39,097	4,767,486	0.94	0.18	0.11	0.96
		P export	649,029	1,359,141	-	4,482,119	1.00	0.23	0.13	1.00
	Stratification/threshold	Sediment export	649,024	-	5	5,841,260	1.00	-	0.10	-
		P export	556,187	-	92,842	5,841,260	0.86	-	0.09	-
Mulde	Stratification/PNV	Sediment export	910,829	469,687	55	7,728,264	1.00	0.06	0.11	1.00
		P export	910,466	1,098,512	418	7,099,439	1.00	0.13	0.11	1.00
	Stratification/threshold	Sediment export	910,821	-	63	8,197,951	1.00	-	0.10	-
		P export	4	1,316,215	910,880	6,881,736	0.00	0.16	0.00	0.59

Table 5.S11: Agreeing options between the stratification (arithmetic mean) and PNV approach as well as threshold approach for positive and negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Environmental impact	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	Stratification/PNV	C storage	256,080	3,088,707	14,282	3,131,220	0.95	0.50	0.08	1.00
		Sediment retention	1,557,133	908,693	4,020,484	3,979	0.28	1.00	1.00	0.18
		P retention	383,965	-	6,106,319	5	0.06	-	1.00	-
Rufiji	Stratification/PNV	C storage	512,439	786,365	176,863	224,250	0.74	0.78	0.70	0.82
		Sediment retention	645,324	336,643	173,707	544,243	0.79	0.38	0.54	0.66
		P retention	336,083	631,302	83,261	649,271	0.80	0.49	0.34	0.88
Mulde	Stratification/PNV	C storage	609,293	6,455,818	-	2,043,724	1.00	0.76	0.23	1.00
		Sediment retention	1,950,846	4,850,159	458,479	1,849,351	0.81	0.72	0.51	0.91
		P retention	3,489,899	3,689,224	80,049	1,849,663	0.98	0.67	0.65	0.98
Satilla	Stratification/PNV	Sediment export	377,097	1,105,998	5,000,321	6,873	0.07	0.99	0.98	0.18
		P export	928,978	1,359,118	23	4,202,170	1.00	0.24	0.18	1.00
	Stratification/threshold	Sediment export	5,377,414	1	1,112,870	4	0.83	0.20	1.00	0.00
		P export	836,159	-	92,842	5,561,288	0.90	-	0.13	-
Mulde	Stratification/PNV	Sediment export	3,326,842	466,401	3,341	5,312,251	1.00	0.08	0.39	0.99
		P export	1,433,737	1,091,037	7,893	6,576,168	0.99	0.14	0.18	0.99
	Stratification/threshold	Sediment export	3,330,120	-	63	5,778,652	1.00	-	0.37	-
		P export	8	785,473	1,441,622	6,881,732	0.00	0.10	0.00	0.35

Table 5.S12: Agreeing options between the PNV and the threshold approach for negative EIs; true positive (TP), true negative (TN), false positive (FP), positive predictive value (PPV), negative predictive value (NPV).

Case study	Methods	Ecosystem disservice	TP	TN	FP	FN	PPV	NPV	Sensitivity	Specificity
Satilla	PNV/threshold	Sediment export	5,377,414	1	4	1,112,870	1.00	0.00	0.83	0.20
		P export	5,038,306	-	92,842	1,359,141	0.98	-	0.79	-
Mulde	PNV/threshold	Sediment export	8,639,030	-	63	469,742	1.00	-	0.95	-
		P export	5,832,715	49,905	2,177,190	1,049,025	0.73	0.05	0.85	0.02

6 Synthesis and discussion

6.1 Suitability of certification schemes for a sustainability assessment

The current globally applicable and EU-centric governance mechanisms, sustainability certification schemes, showed several deficiencies as shown in detail in chapter 2. Four major deficiencies (i-iv) and their implications will be discussed within and beyond the scope of bioenergy in this section.

(i) Certification schemes as market driven tools predominately aimed at feasible indicators and were partly aligned with legislative requirements. In that respect, if clearly defined requirements in the EU RED exist, certification schemes will implement very precise C&Is. For example, conversion of “land with high biodiversity value” or” high carbon stocks” (Art. 17 [1]) is not allowed. However, fuzzy requirements led to diverging indicator sets in certification schemes. Fuzzy requirements were often included as vague formulations in the EU RED, e.g., to consider impacts on soil and water or on “basic ecosystem services” [1]. Most certification schemes used more feasible than reliable indicators for the less specified requirements for environmental impacts. For example, they often measured drivers instead of the actual environmental response of biomass production, thus reducing the reliability (c.f. chapter 2). An example would be accounting for the area of clearcuts in pine plantations instead of assessing water quality impacts due to potentially increased erosion rates.

This dissertation confirms the rational of existing research, e.g., Scarlat and Dallemand [2], to prescribe minimum sustainability requirements. For environmental impact categories without minimum requirements such as soil quality, certification schemes strongly differ in their quality and comprehensiveness (c.f. chapter 2). Therefore, a comparable assessment for different environmental impact categories without minimum standards is hardly given. In that respect, certification schemes could be improved upon with a legally defined minimum set of indicators for each environmental impact category, and the obligation to develop additional indicators complementing the minimum set. This approach has the advantage to allow at least for a minimum comparability of certification schemes. It also ensures that certification schemes go beyond the minimum requirements in the underlying legislation, which was the case for biodiversity indicators in certification schemes (c.f. chapter 2). However, independent from the fact that certification schemes for the EU RED could be improved and more harmonized, it is crucial that they not only reflect the European notion for sustainable biomass production. Van Dam et al. [3] identified that the EU is exporting their environmental sustainability criteria to third countries insufficiently considering local needs such as food security in the social sustainability dimension, especially from developing countries. This fact is also problematic in other global governance initiatives for sustainability such as the Clean Development Mechanism [4]. Therefore, the EU should more precisely define their criteria for certification schemes. Future certification schemes should also consider the preferences of national or regional authorities and affected local stakeholders in the biomass producing countries. It may be therefore reasonable to question to which extent is sufficient to rely on voluntary international certification schemes without strengthening the national governance of producing countries as a crucial question for certification schemes in general [5].

(ii) Certification schemes predominately applied indicators on management practices at the plot/field scale without aggregating them to the landscape scale. For example, certification schemes assessed the use of irrigation practices for the individual farmer but disregarded the impact of large-scale water abstraction of multiple farmers at the watershed level. These findings hint to a general problem for voluntary certification schemes beyond bioenergy. Voluntary certification schemes of other products like coffee, for instance, are equally elaborate for identifying environmentally harmful management practices at the plot scale [6]. However, they are rather weak in assessing the large scale impact of land-use change arising due to increased production of cash crops for the world market [5]. As this issue exists not only in the context of bioenergy but also for other agricultural or forestry production areas, it might reasonable to put it on the policy agenda for environmental treaties and (voluntary) certification schemes for sustainable forestry and agricultural production for the world market.

(iii) Certification schemes hardly linked ESS use such as biomass for bioenergy with underlying ecosystem structures and processes. Environmental feedbacks and interaction were barely analyzed [7]. Indicators in certification schemes typically considered immediate impacts of land use on ESS. For example, they assume a direct link that irrigation practices enhance or reduce water use. But certification schemes often do not consider the impacts of changed irrigation practices on ground- and surface water resources and underlying processes of the regional water cycle. Intermediate buffering or amplifying environmental processes or human activities are not taken into account but shown as relevant by Niemeijer and de Groot [8] in general and in chapter 2 for bioenergy certification schemes. The developed analytical framework for biomass for bioenergy could also be used to analyze assessment approaches for biomass and agricultural production systems such as certification schemes for forestry or agriculture or indicator based assessments by governmental agencies. Biermann et al. [5] revealed that such complex interactions are equally missing in regulatory frameworks in the context of water governance and ESS assessments at the landscape scale. The developed analytical framework could be used as a starting point to assess the quality and comprehensiveness of environmental monitoring schemes for water management. For example, Vlachopoulou et al. [9] identified links between the Water Framework Directive and the ESS cascade applied in this dissertation (c.f. chapter 2). Revising the indicators in the analytical framework developed in chapter 2 on water quality and quantity and underlying ecosystem structures and processes would allow for the evaluation of alternative water management options within the EU. This revision of indicators in the analytical framework might also be equally applicable to the critical load concept as applied in the US. Applying the analytical framework may reveal limitations in the schemes and guide improvement options. Poppy et al. [10] reveal that payment schemes for ESS often focus on a single ESS instead of ESS bundles and their interaction. Therefore, the analytical framework (c.f. chapter 2) could be used to compare different payment schemes for ESS assessing their effectiveness in improving the supply of multiple ESS and underlying environmental interactions. However, the analysis of certification schemes in this dissertation and the developed analytical framework focused on assessing the human impact on the environment reflecting the dominant notion in the EU RED as underlying legislation [11]. Future research should provide an integrated and balanced analysis of socio-economic and environmental impacts to obtain a more complete impact assessment of biomass production systems as equally identified by Awudu and Zhang [12]. This dissertation also agrees with Fischer et al. [13] that several processes at the social side of socio-ecological systems need more research effort as in the context of governance of natural resources like biomass production.

(iv) Certification schemes also did not propose or require a methodology to compare the environmental or ESS impacts of a biomass production system with a baseline such as an ecosystem's capacity. Indicators often miss target values, thresholds, or "wishful" environmental conditions (e.g., PNV) to assess the absolute or relative compliance of the biomass production system with sustainability requirements. To overcome these issues, this dissertation proposed a set of capacity indicators such as the maximum nutrient removal potential or a minimum viable species population size (c.f. chapter 2). Meijaard et al. [14] equally propose this for ESS in forest certification schemes. However, the reliability, feasibility, and relevance of different approaches for multiple ESS have been discussed in chapter 5. For removing most of the environmental heterogeneity, the PNV approach should be taken. For a worldwide consistent application with minimal effort, the stratification approach should be taken as one globally applicable dataset exists and no second modeling round for environmental impacts is needed. Therefore, a more likely implementation would be the stratification approach due to the strong preference towards feasibility in certification schemes. However, the currently only named concept in certification schemes is the threshold approach, which performed rather badly with respect to reliability and feasibility. Nevertheless, it is still seen as most relevant by stakeholders and government agencies as shown in chapter 5. Therefore, the implementation barrier for the PNV and stratification approach is likely higher than for a threshold approach.

6.2 Ecosystem service assessments in different biomass production systems at the regional scale

This dissertation filled the gap on assessing the environmental sustainability of biomass production at the regional scale and demonstrated the relevance of landscape structure for bundles of ESS instead of studies for single ESS (c.f. chapter 3). The assessment of multiple ESS based on simulated SRC plantations allowed to close the research gap of missing impact assessments of commercial-scale SRC plantations (c.f. chapter 4). This section discusses (i) implications of the results for regional scale environmental assessments for bioenergy and (ii) social and environmental implications of regional scale SRCs' deployment as an example for novel biomass production options.

(i) The regional analyses of ESS impacts and their drivers (c.f. chapters 3 to 5) showed that it is insufficient to apply a governance mechanism like certification schemes that focuses on single feedstock producers (e.g., farmers, foresters), and to prescribe management practices at the plot/field scale as discussed in chapter 6.1. In both a plantation forestry and an agriculturally dominated watershed in the Southeastern US, indicators on landscape structure and naturalness explained more than 30 percent of the variation in supply for multiple ESS (c.f. chapter 3), thus confirming the conceptual ideas of Birkhofer et al. [15]. For SRCs, this dissertation partly disagrees with Holland et al. [16] based on a synthesis of studies for the plot/field scale or with Manning et al. [17] discussing it conceptually. An increasing share of SRCs in the landscape, in different scenarios between 2 and 24% of the study area, did not raise the number of balanced ESS bundles as defined by Foley et al. [18]. However, as shown for the study regions in the Southeastern US (c.f. chapter 3) various landscape factors like landscape naturalness and structure as well as soil, topography, and climate parameters may all equally influence ESS supply and the occurrence of ESS bundles. For that reason, certification schemes could integrate indicators on landscape structure and naturalness in their assessments to test for comparable sustainability standards in globally distributed biomass production regions. Studies within (c.f. chapter 4) beyond the scope of bioenergy (Qiu and Turner [19]) have shown that landscape structure may explain the occurrence of hot and cold spots of ESS bundles. Syrbe and Walz [20] conceptually discuss the impact of landscape structure on ESS.

To overcome these limitations of assessments at the plot scale, certification schemes should assess the impact of land use and management in the entire landscape to avoid ESS trade-offs, especially with the tools used in this dissertation: InVEST [21], Globio [22], and SWB [23]. An integration of regional-scale assessments into certification schemes seems more reliable in contrast to exclusively relying on assessments by regional or national authorities in the respective feedstock production regions, and they may help to reduce unwanted ESS trade-offs. Without assessments at the regional scale, it is impossible to determine whether ESS trade-offs are reduced through environmentally sound management practices. The latter approach is unlikely to be fruitful in practice in countries with variable or absent legal frameworks and functioning governance bodies for landscape planning. Prior to such analysis, one should conduct further studies in major (solid) biomass and crop production regions to underpin these findings for the Southeastern US. However, it was possible to show the implications for the Southeastern US, the expected major exporting region of modern solid biomass (i.e., wood pellets) to the EU by 2020 [24], and a potential future exporting region, Tanzania, [25]. Nevertheless, further empirical validation under different environmental and management conditions would help to generalize the results.

With respect to ESS research beyond bioenergy, this dissertation equally helped to close the spatial incongruence between local/plot scale experimental research on ESS and the need for knowledge on ESS bundles and their interactions at the regional scale, e.g., for policy making [15]. It also showed in chapter 3, how the focus of existing studies on ESS synergies and trade-offs either on agricultural (e.g., Power [26]) or forestry production systems (e.g., Cademus et al. [27]) could be overcome. However, these results from individual case studies need to be synthesized and to be tested for suitability in less human modified landscapes and for other globally distributed land-use systems.

(ii) Considering the demand for solid biomass of current CHP plants in the case study in the Mulde watershed in Central Germany (about 50,000 t per year), SRCs hardly positively affected regulating ESS. The corresponding SRC production on 2% of the study area reduced sediment and P export by 2%, and biodiversity, i.e., an increase in habitat quality by 2% (c.f. chapter 4). Only a significant increase largely beyond regional solid biomass demand for CHP plants enhanced the supply of regulating ESS (c.f. chapter 4). For use in policy and practice, the results showed that SRCs hardly enhanced ESS and biodiversity in deployment scenarios for the current market and political environment, i.e., fulfilling the requirements for EFAs in the CAP entirely with SRCs. If SRCs should be used to enhance regulating ESS supply and biodiversity, it would (a) be at the price of a considerable loss of food and feed production, (b) assume super-regional demand for biomass from SRCs, and (c) require considerable subsidy payments to stimulate commercial deployment or the implementation of legislative instruments such as the EFAs in the CAP. For (a), it is necessary to compare the indirect impacts of the displacement of substituting production options. Indirect land-use change and associated positive or negative sustainability impacts may equally occur. For (b), it would require super-regional competitiveness, which is hardly given even for the local market as shown in the scenario with standard demand for solid biomass for SRCs in chapter 4. For (c), it may be more reasonable to compare SRCs with other options like the use of catch or cover crops. Such comparison may help to identify the relative efficiency of SRCs.

6.3 Comparison of worldwide biomass production systems

This dissertation showed that regional environmental impacts and ESS can be modeled in a similar manner in biomass production regions in different parts of the world. This section discusses (i) major technical challenges and (ii) potential options for the future application of sustainability assessments beyond the scope of bioenergy.

(i) The analysis of certification schemes identified the need for an approach to make environmental impacts comparable for biomass production options in different world regions; Certification schemes missed such an approach (e.g., thresholds or target values) (c.f. chapter 2). The comparison of environmental impacts of land use options requires controlling for environmental heterogeneity (c.f. chapter 5). The stratification and PNV approaches show decreasing beneficial and lower harmful environmental impacts relative to land use intensity, i.e., cropland > plantation forestry > forest, as stated in Brockerhoff et al. [28]. For solid biomass production options, plantation forestry (Southeastern US and Tanzania) and forestry (Central Germany), the PNV and stratification approach identified forestry in Central Germany with the highest amount of positive environmental impacts (carbon storage, sediment retention, and phosphorous retention). In contrast to comparing biomass production systems with mostly homogenous datasets as chapter 3, any approach to model environmental or ESS impacts at the global scale needs to consider the heterogeneity in the locally available datasets (c.f. chapter 5). For example, the biomass production region in Tanzania partially required using global datasets at low spatial resolution, whereas ESS modeling in the Southeastern US mostly relied on local datasets with high spatial resolution. Equally, global heterogeneous availability of data delimits choice of environmental impact and ESS modeling tools. For example, changes in groundwater recharge as one major concern regarding bioenergy production [29] is hardly possible due to missing options for validation as it was case in Tanzania.

(ii) Beyond the focus of biomass production for bioenergy, this dissertation clarified the deficiencies of the different approaches to control for environmental heterogeneity for an absolute or relative environmental sustainability assessment. Comparable to the EU Water Framework Directive that aims in an abstract way at achieving the natural state of water bodies [30], the more reliable PNV or the more feasible stratification approach could be seen as starting point to create such needed reference state for different environmental impacts or ESS in different land-use systems worldwide. Applying these approaches, one may go beyond the approach in chapter 3 relating the sets of ESS or environmental impacts for different LU/LC classes within a study. Beyond the stratification approach, modelling ESS supply based on PNV helps to assess the distance from the natural state of current ESS supply to implement an assessment as conceptually discussed in Foley et al. [18]. Applying this approach to different production regions for one commodity, e.g., biomass or crop production, will allow to assess the current human alteration of the ecosystem [31]. It may provide a basis for stakeholder consultation based on the information of how far a current land-use system is away from the (potential) natural reference state. The stakeholder discussions would receive neutral reference conditions to decide whether the management intensity or the LU/LC composition should be modified. The preferences will differ between stakeholder groups, but also between developed and developing countries as demonstrated for water related environmental impacts or ESS [30]. Depending on the societal discourse, the acceptable level of environmental impacts or the required level of ESS may largely vary, but may range between the outcome for the PNV approach and an intensively managed agricultural or forestry systems, disregarding highly urbanized and industrialized regions. Several normative questions arise in this context and with respect to future research for implementation: Which state of the land-use system do local and remote stakeholders, e.g., in biomass importing regions, prefer? How should different stakeholders' preferences be considered?

6.4 Conclusions and future research needs

The focus of this dissertation was to assess regional environmental and ESS impacts of biomass production for bioenergy promoted by the EU RED. The EU RED contributed to rising global biomass trade for bioenergy and the need for comparing regional environmental and ESS impacts worldwide as shown in chapter 5. However, this issue may need to be tackled in general for agricultural and forest products. (i) Ahead of bioenergy policies, forest protection policies have already started to displace the demand for and production of forestry and agricultural products to other countries in the world with weaker governance structures [32]. (ii) The biomass resources assessed in this dissertation may not only serve an energetic but also a material use, as shown in chapter 3 for the increasing availability of forest biomass for bioenergy due to a decline in the pulp and paper industry in the Southeastern US. (iii) For other bioenergy feedstocks such as corn cobs, the food vs. fuel debate showed strong interlinkages of the markets for food and bioenergy [33], which calls for environmental assessments independent from the final use. Therefore, the findings in this dissertation for biomass for bioenergy hint to three major challenges that future research may analyze in the context of environmental and ESS assessments for agricultural and forestry production systems. First (i), to identify the regional environmental impacts of global trade of agricultural and forestry products, it is necessary to create the link between tradeflows and regional environmental and ESS assessments. Secondly (ii), it is necessary to improve the methods for modeling ESS supply and demand consistently to determine which environmental impacts of global trade flows are critical for regional stakeholders. Finally (iii), it is necessary to consider how the first two steps could be transferred from individual model crops or products and case studies to further products and world regions; the first two research needs are hardly implementable for all agricultural and forestry products and all world regions in a quite realistic and feasible manner:

i. Linking global trade flows of forestry and agricultural products with regional environmental and ESS assessments

Beyond the bioenergy focus, increasing global trade of raw or processed agricultural or forestry products may induce further land-use change. Several existing spatially explicit global environmental models link global economic models on trade flows, with global, standardized environmental impact indicators, see Davis et al. [34] for an exemplary study on CO_2 emissions. However, human induced socioeconomic changes in land-use systems such as enhanced biomass production have regionally varying impacts and should not be modeled at a global scale [35]. Common approaches miss to link global trade flows, one of the major driving forces of land-use change, with local/regional process based models [32] that include an understanding of socio-economic and environmental processes. For example, Dale et al. [36] state that soil and water quality impacts of land-use change are not reasonably covered by standardized and non-place-based indicators as used in LCAs. Spatially explicit assessments can consider the impact of landscape heterogeneity on environmental impacts or ESS [37].

Global trade flows of agricultural and forestry products, which are a major driver of land-use change, can be conceptualized in various ways. Henders and Ostwald [38] identified a variety of methods such as material flow analysis or multi-regional-input-output models that may be used as a starting point to model distant drivers of land-use change, i.e., teleconnections. On the one hand, generic trade flow models, neither crop nor biomass production system specific, often miss information on land management or socio-economic factors such as population growth or changes in consumption patterns. Considering these factors may improve the accuracy of trade-flow patterns, which could change regional land use systems and ESS, see Poppy et al. [10]. On the other hand, coupling the mentioned ESS models with models conceptualizing teleconnections comprehensively and detailed for all relevant natural resources may create too much complexity and hinder further application. Therefore, it might be reasonable to focus on model crops or products and several case study regions first. In that respect, the approaches of regional scale environmental and ESS assessments (c.f. chapters 3 and 4) and the worldwide comparison of regional studies (c.f. chapter 5) could contribute to future investigations on how changes in agricultural or forestry products trade flows affect the balance of regional ESS supply and demand. Regional production for the world market beyond the regional scale may change regional supply and demand patterns of ESS.

ii. Modeling ESS demand and supply consistently and reliably

Chapters 3, 4, and 5 have exemplarily demonstrated regional environmental and ESS assessment approaches for various agricultural and forestry production systems. The spatial modeling of ESS or environmental impacts may be done with InVEST [21] or other assessment tools such as ESTIMAP [39] or SWB [23]. However, the named tools are developed for static modeling of environmental impacts or ESS based on biophysical input data and predominately emphasize on the supply perspective. ESS modeling approaches often have a low temporal resolution ESS supply [15], e.g., annual values, although ESS demand and supply for many ESS may seasonally fluctuate, e.g., water supply and demand in comparison between the dry and the rainy season.

Wolff et al. [40] reviewed few existing studies on assessing ESS demand. They request to improve the assessment of ESS demand and to adapt the assessment techniques depending on the individual ESS, which includes improvements with respect to spatial and temporal resolution. With respect to this dissertation, conceptually refined ESS demand modeling would allow for the improvement of results in chapter 3. It would also allow to assess ESS instead of environmental impacts, done in chapter 5. In this context, it is necessary to consider that ESS will be only realized if ESS supply and demand meet at the relevant spatial and temporal scales.

iii. Transferability of results from case studies

One option is to transfer regional case study results as gained in chapters 3, 4, and 5. Therefore, it may be reasonable to first apply ESS models to model specific natural resources or land-system archetypes, representative combinations of land use intensity, environmental conditions and socioeconomic factors [41], to identify drivers of ESS supply and demand. Such analysis could delimit the number of decisive socio-economic and environmental factors. However, the large scale of the land system archetypes requires spatial and qualitative refinements, especially due to partly misleading aggregations for applications at the regional scale. Therefore, further research on the transferability of regional case study results is needed and different options such as methods from meta-analyzes need to be evaluated.

6.5 References

[1] Directive 2009/28/EC. On the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC.2009.04.23. Off J EU 2009. L 140: 16-62.

[2] Scarlat N, Dallemand J-F. Recent developments of biofuels/bioenergy sustainability certification: A global overview. Energy Policy 2011;39(3):1630-46.

[3] Van Dam J, Junginger M, Faaij APC. From the global efforts on certification of bioenergy towards an integrated approach based on sustainable land use planning. Renewable and Sustainable Energy Reviews 2010;14(9):2445-72.

[4] Newell P, Jenner N, Baker L. Governing Clean Development: A Framework for Analysis. Development Policy Review 2009;27(6):717-39.

[5] Biermann F, Abbott K, Andresen S, Bäckstrand K, Bernstein S, Betsill MM, et al. Transforming governance and institutions for global sustainability: key insights from the Earth System Governance Project. Current Opinion in Environmental Sustainability 2012;4(1):51-60.

[6] Tscharntke T, Milder JC, Schroth G, Clough Y, DeClerck F, Waldron A, et al. Conserving Biodiversity Through Certification of Tropical Agroforestry Crops at Local and Landscape Scales. Conservation Letters 2015;8(1):14-23.

[7] Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65:151-69.

[8] Niemeijer D, de Groot R. Framing environmental indicators: moving from causal chains to causal networks. Environment, Development and Sustainability 2008;10(1):89-106.

[9] Vlachopoulou M, Coughlin D, Forrow D, Kirk S, Logan P, Voulvoulis N. The potential of using the Ecosystem Approach in the implementation of the EU Water Framework Directive. Science of The Total Environment 2014;470–471:684-94.

[10] Poppy GM, Chiotha S, Eigenbrod F, Harvey CA, Honzák M, Hudson MD, et al. Food security in a perfect storm: using the ecosystem services framework to increase understanding; 2014.

[11] German L, Schoneveld G. A review of social sustainability considerations among EU-approved voluntary schemes for biofuels, with implications for rural livelihoods. Energy Policy 2012;51:765-78.

[12] Awudu I, Zhang J. Uncertainties and sustainability concepts in biofuel supply chain management: A review. Renewable and Sustainable Energy Reviews 2012;16(2):1359-68.

[13] Fischer J, Gardner TA, Bennett EM, Balvanera P, Biggs R, Carpenter S, et al. Advancing sustainability through mainstreaming a social–ecological systems perspective. Current Opinion in Environmental Sustainability 2015;14:144-9.

[14] Meijaard E, Wunder S, Guariguata MR, Sheil D. What scope for certifying forest ecosystem services? Ecosystem Services 2014;7(0):160-6.

[15] Birkhofer K, Diehl E, Andersson J, Ekroos J, Früh-Müller A, Machnikowski F, et al. Ecosystem services – current challenges and opportunities for ecological research. Frontiers in Ecology and Evolution 2015;2.

[16] Holland RA, Eigenbrod F, Muggeridge A, Brown G, Clarke D, Taylor G. A synthesis of the ecosystem services impact of second generation bioenergy crop production. Renewable and Sustainable Energy Reviews 2015;46:30-40.

[17] Manning P, Taylor G, E. Hanley M. Bioenergy, Food Production and Biodiversity – An Unlikely Alliance? GCB Bioenergy 2015;7(4):570-6.

[18] Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, et al. Global Consequences of Land Use. Science 2005;309(5734):570-4.

[19] Qiu J, Turner MG. Spatial interactions among ecosystem services in an urbanizing agricultural watershed. Proceedings of the National Academy of Sciences 2013;110(29):12149-54.

[20] Syrbe R-U, Walz U. Spatial indicators for the assessment of ecosystem services: Providing, benefiting and connecting areas and landscape metrics. Ecological Indicators 2012;21:80-8.

[21] Vigerstol KL, Aukema JE. A comparison of tools for modeling freshwater ecosystem services. Journal of Environmental Management 2011;92(10):2403-9.

[22] Alkemade R, van Oorschot M, Miles L, Nellemann C, Bakkenes M, ten Brink B. GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss. Ecosystems 2009;12(3):374-90.
[23] Westenbroek SM, Kelson VA, Dripps WR, Hunt RJ, Bradbury KR. SWB - A modified Thornthwaite-Mater Soil-Water-Balance code for estimating groundwater recharge. Techniques and Methods 6-A31. Reston; 2010.
[24] Goh CS, Junginger M, Cocchi M, Marchal D, Thrän D, Hennig C, et al. Wood pellet market and trade: a global perspective. Biofuels, Bioproducts and Biorefining 2013;7(1):24-42.
[25] Lamers P, Hoefnagels R, Junginger M, Hamelinck C, Faaij A. Global solid biomass trade for energy by 2020: an assessment of potential import streams and supply costs to North-West Europe under different sustainability constraints. GCB Bioenergy 2015;7(4):618-34.
[26] Power AG. Ecosystem services and agriculture: tradeoffs and synergies; 2010.
[27] Cademus R, Escobedo F, McLaughlin D, Abd-Elrahman A. Analyzing Trade-Offs, Synergies, and Drivers among Timber Production, Carbon Sequestration, and Water Yield in Pinus elliotii Forests in Southeastern USA. Forests 2014;5(6):1409-31.
[28] Brockerhoff E, Jactel H, Parrotta J, Quine C, Sayer J. Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity and Conservation 2008;17(5):925-51.
[29] Yeh S, Berndes G, Mishra GS, Wani SP, Elia Neto A, Suh S, et al. Evaluation of water use for bioenergy at different scales. Biofuels, Bioproducts and Biorefining 2011;5(4):361-74.
[30] Bishop K, Beven K, Destouni G, Abrahamsson K, Andersson L, Johnson RK, et al. Nature as the "Natural" Goal for Water Management: A Conversation. AMBIO: A Journal of the Human Environment 2009;38(4):209-14.
[31] Meyer MA, Seppelt R, Witing F, Priess JA. How can we compare environmental and ecosystem service assessments for biomass production systems in different parts of the world? Environmental Research Letters under review.
[32] Meyfroidt P, Lambin EF, Erb K-H, Hertel TW. Globalization of land use: distant drivers of land change and geographic displacement of land use. Current Opinion in Environmental Sustainability 2013;5(5):438-44.
[33] Valentine J, Clifton-Brown J, Hastings A, Robson P, Allison G, Smith P. Food vs. fuel: the use of land for lignocellulosic 'next generation' energy crops that minimize competition with primary food production. GCB Bioenergy 2012;4(1):1-19.
[34] Davis SJ, Peters GP, Caldeira K. The supply chain of CO2 emissions. Proceedings of the National Academy of Sciences 2011;108(45):18554-9.
[35] van Asselen S, Verburg PH. A Land System representation for global assessments and land-use modeling. Global Change Biology 2012;18(10):3125-48.
[36] Dale VH, Lowrance R, Mulholland P, Robertson GP. Bioenergy sustainability at the regional scale. Ecology and Society 2010;15(4):23.
[37] Wong CP, Jiang B, Kinzig AP, Lee KN, Ouyang Z. Linking ecosystem characteristics to final ecosystem services for public policy. Ecology Letters 2015;18(1):108-18.
[38] Henders S, Ostwald M. Accounting methods for international land-related leakage and distant deforestation drivers. Ecological Economics 2014;99(0):21-8.
[39] Zulian G, Polce C, Maes J. ESTIMAP: a GIS-based model to map ecosystem services in the European union. Annali di Botanica 2014;4:1-7.
[40] Wolff S, Schulp CJE, Verburg PH. Mapping ecosystem services demand: A review of current research and future perspectives. Ecological Indicators 2015;55(0):159-71.
[41] Václavík T, Lautenbach S, Kuemmerle T, Seppelt R. Mapping global land system archetypes. Global Environmental Change 2013;23(6):1637-47.

Curriculum vitae

Personal details

Name	**Markus Andreas Meyer**
Date of birth	**17th of April 1987**
Place of birth	**Salzgitter**
Nationality	**German**

Education

Since 01/2013	**External PhD student (Geography), Martin Luther University Halle-Wittenberg, Faculty of Natural Sciences III** *Spatially explicit environmental assessment of global biomass production for bioenergy at the regional scale*
02/2012 – 07/2012	**Master's thesis, Karlsruhe Institute of Technology, Institute for Technology Assessment and Systems Analysis** *Economic analysis of energy and matter generation from microalgae*
10/2010 – 09/2012	**Master of Science, University of Hohenheim and Swedish University of Agricultural Science (double degree)** *Environmental Science*
09/2006 – 02/2010	**Bachelor of Science, Frankfurt School of Finance & Management** *Business Administration*
09/2008 – 12/2008	**Semester abroad, Université du Québec à Montréal** *Business Administration*
08/1999 – 06/2006	**Abitur, Gymnasium Salzgitter-Bad**

Professional experience

Since 04/2015	**University of Munich (LMU),** Munich (lecturer) *Department of Geography*
08/2012 – 07/2015	**Helmholtz Centre for Environmental Research,** Leipzig (research assistant) *Department* Computational *Landscape Ecology*
08/2006 – 10/2010	**Deutsche Postbank AG,** Bonn (employee) *Risk management, Controlling, vocational training*

Eching, 06.08.2015

List of publications

Peer-reviewed publications

Meyer MA, Priess JA. Indicators of bioenergy-related certification schemes – An analysis of the quality and comprehensiveness for assessing local/regional environmental impacts. Biomass and Bioenergy 2014;65:151-69.

Meyer MA, Weiss A. Life cycle costs for the optimized production of hydrogen and biogas from microalgae. Energy 2014;78:84-93.

Meyer MA, Chand T, Priess JA Comparing bioenergy production sites in the Southeastern US regarding ecosystem service supply and demand. PLOS ONE 2015;10(3):e0116336.

Conference proceedings

Meyer MA, Priess JA (2013) Local/regional environmental impacts of (solid) biofuels - evaluation of certification scheme indicators. 21st European Biomass Conference: Setting the course for a biobased economy. Proceedings of the International Conference held in Copenhagen, Denmark, 3-7 June 2013. Florence, Italy: EU BC&E Proceedings.

Meyer MA, Priess JA (2014) Local/regional environmental impacts of solid biofuels – The case od the southeastern US. International Bioenergy Conference 2014. Proceedings of the International Conference held in Manchester, UK, 11-13 March 2014. Manchester, UK: BBSRC.

Meyer MA, Priess JA (2014) Ökosystemleistungen und „Trade-offs" in forstlichen und landwirtschaftlichen Biomasseproduktionssystemen für Bioenergie im Südosten der USA. In: Ulrike Tappeiner, Uta Schirpke, Roberta Bottarin, Stefan Zerbe, editors. "IALE-D Jahrestagung 2014 - Ökologie, Resilienz und Management unserer Landschaft". Proceedings of the International Conference held in Bolzano, Italy, 15-17 October 2014. Bolzano, Italy: IALE - Region Deutschland.

Acknowledgments

Here, I would like to acknowledge the people supporting the completion of this dissertation. This dissertation work was made possible by the funding through the EU FP7 project SECTOR (Production of Solid Sustainable Energy Carriers by Means of Torrefaction), grant agreement number 282826, the Helmholtz Research Program "Terrestrial Environments", the Helmholtz Interdisciplinary Graduate School for Environmental Research (HIGRADE), and the Helmholtz Research School for Ecosystem Services under Changing Land-use and Climate (ESCALATE).

This dissertation was conducted at the Department Computational Landscape Ecology, UFZ, Leipzig. In particular, I would like to thank Prof. Dr. Ralf Seppelt for the supervision and very useful discussions towards finalizing this dissertation. I am greatly obliged to Dr. Jörg Priess for his scientific support and his guidance as my immediate supervisor. I thank him especially for introducing me to the scientific fields in this thesis such as bioenergy and ecosystem services. In that respect, I would also like to thank the other people, who actively contributed to the success of this thesis: Jule Schulze, Tanzila Chand, and Felix Witing.

Especially, I would like to thank my immediate colleagues for the inspiring and very pleasant working atmosphere throughout these three years: Karla Locher Krause, Christian Hoyer, Susanne Mühlner, Carola Pätzold, Christian Schweitzer, and all other colleagues from the Department CLE, UFZ. Many thanks also to my mentor Nina Schwarz for her reassurance during the ups and downs of this period. Special thanks to Sonya Salimy for language editing for several chapters of this dissertation.

Furthermore, I would like to express my thanks to all colleagues and collaborators from the project SECTOR. Especially, I would like the numerous people for the time and data provided in the study regions in the Southeastern US, Tanzania, and Central Germany.

Finally, I would like to thank my family and friends, especially Henning, for supporting me during this period.

Selbstständigkeitserklärung

Ich erkläre an Eides statt, dass ich diese Arbeit selbstständig und ohne fremde Hilfe verfasst, keine anderen als die von mir angegebenen Quellen und Hilfsmittel benutzt und die den benutzten Werken wörtlich oder inhaltlich entnommenen Stellen als solche kenntlich gemacht habe.

Datum

Markus Meyer