Assessing ecosystem services for informing decision making on sustainable land management under climate change

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

der

Naturwissenschaftlichen Fakultät III
Agrar- und Ernährungswissenschaften,
Geowissenschaften und Informatik

der Martin-Luther-Universität Halle-Wittenberg

vorgelegt von
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geb. am 13.03.1981 in Dresden

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Tag der Verteidigung: Halle (Saale), den 18.6.2018

"Learning is experience. Everything else is just information."

Albert Einstein

"Only when the last tree has been cut down, the last fish been caught, and the last stream poisoned, will we realize we cannot eat money."

Cree Indian Prophecy

Foreword

Experiencing human-nature interaction is critical for providing answers to today`s environmental challenges and it is this experience that laid the foundation for this work. My studies in Landscape Ecology and Nature Conservation allowed me to combine the disciplines of ecology, environmental economics and ethics, providing me with knowledge and skills for assessing human-nature interactions and for exploring strategies for more sustainable development. My work for the Royal Swedish Academy of Sciences gave me first-hand experience in assessing the impacts of global climate change on ecosystems. The study programme in Global Change Ecology equipped me with the tools to connect knowledge across local to global scales. In my work for the International Union for Conservation of Nature (IUCN), I applied the knowledge and tools for informing the design of climate policies on the potential of using carbon payments for reducing forest loss. The gained skills and experiences were essential for contributing to The Economics of Ecosystems and Biodiversity (TEEB): providing economic rationale for sustainable resource management and nature conservation, while being fully aware of the limitations of economic perspectives.

Combined, the knowledge, skills and experiences I gained throughout this journey form the foundation for this dissertation. I hope the work and results presented in this dissertation can inform the way we do science for supporting a wiser stewardship of our planet. I am deeply grateful to my mentors, friends and family for this joint journey.

Acknowledgments

I am grateful to Ralf Seppelt for his excellent supervision, guidance and advice on this dissertation and in particular for his patience. Furthermore, I would like to thank especially Heidi Wittmer for constructive comments throughout the development and completion of this dissertation. Special thanks also to my colleagues at the GLUES Project and at the Helmholtz Centre for Environmental Research – UFZ for inspiring discussions and a pleasant work environment.

Regarding Chapter 2, I would like to thank all participants of the workshop for reviewing the database and for sharing their insights and advice, including Alexandra Dehnhardt, Jesko Hirschfeld, Karsten Grunewald, Achim Schäfer, Daniel Sutter, Burkhard Schweppe-Kraft, Irene Ring, Björn Bünger and Astrid Matthey. Funding for Chapter 2 was provided by the German Federal Environment Agency Umweltbundesamt (UBA) as part of the call on ”Methodenkonvention 3.0 – Weiterentwicklung und Erweiterung der Methodenkonvention zur Schätzung von Umweltkosten”, which is part of the research plan (UFOPLAN) of the German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMUB) (funding ID: 3715 14 102 0).

I am also grateful to all participants and stakeholders involved in Chapter 3 for sharing their experience and insights from their work within the projects of the Sustainable Land Management Program. The German Federal Ministry of Education and Research (BMBF) provided the funding for this research (BMBF grant 01LL0901A: GLUES).

Furthermore, I would like to thank Tomáš Václavík, Gillian A. Cerbu, Elizabeth A. Law and Ralf Seppelt for their comments and contributions to earlier drafts of Chapter 4 and Ameur M. Manceur, Katharina Gerstner and Gi-Mick Wu for guidance on statistical analysis. I am also grateful to Stewart Maginnis, Jeff Sayer and Earl Saxon for insights and discussions on the challenges involved in designing REDD+ policies. Chapter 4 was also supported by the BMBF grant 01LL0901A: GLUES.

Finally, I am deeply grateful to my parents for their support throughout my education and to Julia for her encouragement.

Summary

This dissertation identifies and addresses key research questions related to the assessment of ecosystem services for informing decision making on sustainable land management under climate change. Despite the increasing focus in environmental science on assessing the benefits nature provides to society, so-called ecosystem services, there is little evidence on how scientific information on ecosystem services actually informs decision making. This raises questions whether science on ecosystem services actually achieves the goal of informing decision making. This dissertation addresses this issue at national and international level and provides insights on:

1. studies of monetary valuation of ecosystem service available for Germany and their relevance for informing decisions on national policies on the potential costs of ecosystem service loss (Chapter 2);
2. conceptual considerations for designing problem-oriented ecosystem service assessments that can inform decision making on real-world problems (Chapter 3);
3. using a problem-oriented approach for assessing factors determining the carbon performance of projects reducing emissions from deforestation and forest degradation, in order to inform the design of a climate policy that explicitly targets ecosystem services for ensuring sustainable land management and climate change mitigation (Chapter 4).

Based on a literature review, Chapter 2 provides the first systematic assessment of monetary values for regulating and cultural ecosystem services for Germany. A database was created that can serve as a decision support tool, providing an easy access to decision relevant information on the monetary value of ecosystem services. In total, 109 monetary valuation studies of ecosystem services were identified for Germany with the majority focusing on forests and wetlands. Few studies relate to grasslands although this ecosystem experiences the greatest loss. Only 6 out of 109 studies (5.5 %) comply with selection criteria relevant for informing national policies targeted by the methodological convention (Methodenkonvention) for assessing costs of environmental damage by the German Federal Environment Agency Umweltbundesamt (UBA). Overall, monetary values for regulating and cultural ecosystem services are scattered and scarce compared to information on provisioning services, which is accounted for in detail in national statistics. For achieving greater relevance of ecosystem service valuation studies for decision making, the design of ecosystem service assessments and the choice of indicators could benefit from targeting a clearly defined problem that is of concern for decision makers or of relevance for decision making processes.

Motivated by these insights, Chapter 3 develops a framework and outlines options for designing problem-oriented ecosystem service assessments. Based on the review of existing frameworks for ecosystem service assessments, a lack of explicit guidance on tailoring ecosystem service assessments to information needs of decision makers was identified. For closing this gap, Chapter 3 proposes a problem-oriented approach for

assessing ecosystem services, which is informed by the review of existing frameworks and the experience of four case studies in Brazil, China, Madagascar, and Vietnam. Indicators are identified that can help focusing assessments on decision relevant questions.

This approach of ensuring policy relevance by focusing assessments on a clearly defined policy question is applied in Chapter 4. Reducing carbon emissions from deforestation and forest degradation (REDD+) is a policy under the United Nations Framework Convention on Climate Change (UNFCCC) and a critical component of policies under the Paris Climate Agreement. REDD+ is based on the principles of payments for ecosystem services, providing positive incentives for avoiding carbon emissions from deforestation. While REDD+ offers multiple benefits for climate change mitigation, biodiversity conservation and development, trade-offs between these multiple objectives are to be expected. Hence a better understanding of factors determining carbon performance and potential trade-offs can inform the design of REDD+ projects. The assessment in Chapter 4 identified 66 REDD+ projects that are validated by carbon standards and combined estimate to conserve 9.1 million hectares of forests - an area the size of Portugal - with expected net emission reductions of 1.6 $GtCO_2e$. Multiple linear regression analysis reveals that emission reductions are positively associated with historical deforestation rates, governance effectiveness and a project design for avoiding planned deforestation. However, projects delivering multiple ecosystem services within their project area achieve lower emission reductions, indicating trade-offs in ecosystem services. Private stakeholders favour projects with high carbon performance and carbon rights are private for 75.8% of total net emission reductions, while local communities hold carbon rights to only 10.4% of emission reductions. The analysis informs the design of the REDD+ policy on the need for safeguards for addressing trade-offs in ecosystem services and ensuring equitable access to carbon rights in particular for forest communities. As most emission reductions are generated through avoiding planned deforestation, there is the risk of simply shifting drivers of deforestation to other regions. Hence there is the need to better monitor the potential displacement of deforestation and related carbon emission across regions and continents.

The findings of this dissertation highlight that informing decision making on sustainable land management under climate change requires multidisciplinary and integrative approaches that include, but are not limited to, the assessment of ecosystem services. Thereby, biophysical and socio-economic indicators on ecosystem services should be assessed and reported in units that allow for comparison of values between studies and across scales. Furthermore, future research needs to include impacts on biodiversity and ecosystem services across regions and continents, so-called teleconnections. Ultimately, alternative production and consumption patterns need to be developed for reducing negative impacts on biodiversity. Thereby, not only maximizing carbon sequestration for mitigating climate change should be considered as criterion for sustainability, but also criteria related to the multiple values biodiversity and ecosystem services have for society.

Zusammenfassung

Diese Dissertation identifiziert und adressiert Kernfragen der Forschung bezüglich der Erfassung von Ökosystemleistungen, um Entscheidungen für eine nachhaltige Landnutzung unter Einfluss des Klimawandels zu informieren. Trotz des zunehmenden Fokus der Umweltwissenschaften auf die Erfassung des Nutzens der Natur für die Gesellschaft, den sogenannten Ökosystemleistungen, gibt es wenige Hinweise darauf, wie wissenschaftliche Informationen zu Ökosystemleistungen tatsächlich auch Entscheidungen informieren. Dies wirft Fragen auf, ob die Forschung zu Ökosystemleistungen tatsächlich das Ziel erreicht, Entscheidungen zu informieren. Diese Dissertation adressiert diese Frage auf nationaler und internationaler Ebene und gibt Einblicke zu:

1. Studien zur monetären Bewertung von Ökosystemleistungen verfügbar für Deutschland und ihre Relevanz, Entscheidungen zu nationalen Politiken über mögliche Kosten durch den Verlust von Ökosystemleistungen zu informieren (Kapitel 2);
2. konzeptionellen Überlegungen für die Entwicklung von problemorientierten Ökosystemleistungsanalysen, welche Entscheidungen mit Bezug zu realen Problemen informieren können (Kapitel 3);
3. der Verwendung eines problemorientierten Ansatzes für die Analyse von Faktoren, welche die Kohlenstoffleistung von Projekten zur Reduzierung von Emissionen durch Entwaldung und Walddegradation bestimmen, um die Entwicklung eines Politikinstruments zu informieren, welches explizit auf Ökosystemleistungen für eine nachhaltige Landnutzung und Vermeidung des Klimawandels abzielt (Kaptiel 4).

Basierend auf einer Literaturrecherche liefert Kapitel 2 die erste systematische Erfassung von monetären Werten für regulierende und kulturelle Ökosystemleistungen für Deutschland. Es wurde eine Datenbank geschaffen, welche eine Entscheidungshilfe darstellen kann, indem sie einen einfachen Zugang zu entscheidungsrelevanten Informationen über monetäre Werte von Ökosystemleistungen bietet. Insgesamt wurden 109 Studien mit monetären Werten für Ökosystemleistungen für Deutschland identifiziert, wovon sich die Mehrzahl auf Wälder und Feuchtgebiete bezieht. Wenige Studien beziehen sich auf Grünland, obwohl dieses Ökosystem den größten Rückgang erfährt. Nur 6 von 109 Studien (5,5%) erfüllen Auswahlkriterien zur Eignung als Entscheidungshilfe für nationale Politiken, auf welche die Methodenkonvention des Umweltbundesamts (UBA) zur Bestimmung von Kosten durch Umweltschäden abzielt. Zusammenfassend lässt sich feststellen, dass monetäre Werte für regulierende und kulturelle Ökosystemleistungen verstreut und rar sind, wenn man diese mit bereitstellenden Ökosystemleistungen vergleicht, welche detailliert in nationalen Statistiken erfasst werden. Um eine größere Relevanz von Bewertungen von Ökosystemleistungen für Entscheidungen zu erreichen, könnte das Design und die Wahl von Indikatoren von einem klar definierten Bezug auf ein Problem mit Wichtigkeit für Entscheidungsträger oder Entscheidungsprozesse profitieren.

Von diesen Einsichten motiviert, entwickelt Kapitel 3 einen Ansatz und zeigt Möglichkeiten für die Entwicklung einer problemorientierten Erfassung von Ökosystemleistungen auf. Basierend auf der Literaturrecherche existierender Ansätze zur Erfassung von Ökosystemleistungen wurde festgestellt, dass explizite Vorgaben für eine Analyse mit klarem Bezug auf den Informationsbedarf von Entscheidungsträgern fehlen. Um diese Lücke zu schließen, stellt Kapitel 3 einen problemorientierten Ansatz für die Erfassung von Ökosystemleistungen vor, welcher auf der Analyse existierender Ansätze sowie Erfahrungen aus vier Fallstudien aus Brasilien, China, Madagaskar und Vietnam beruht. Es werden Indikatoren identifiziert, welche helfen können Analysen stärker auf entscheidungsrelevante Fragen zu fokussieren.

Dieser Ansatz, auf klar definierte Politikfragen zu fokussieren, wird in Kapitel 4 angewandt. Die Reduzierung von Kohlenstoffemissionen von Entwaldung und Walddegradation (REDD+) ist ein Politikinstrument der Klimarahmenkonvention der Vereinten Nationen (UNFCCC) und wichtiger Bestandteil des Klimaabkommens von Paris. REDD+ basiert auf dem Prinzip Ökosystemleistungen durch Zahlungen zu honorieren und so einen positiven Anreiz für die Vermeidung von Kohlenstoffemissionen zu erzeugen. Während REDD+ vielfältigen Nutzen für Vermeidung des Klimawandels, Erhalt von Biodiversität und Entwicklung bieten kann, sind auch Zielkonflikte zu erwarten. Daher kann ein besseres Verständnis von Faktoren, welche Einfluss auf die Kohlenstoffleistung sowie mögliche Zielkonflikte haben, die Entwicklung von REDD+ Projekten informieren. Die Analyse in Kapitel 4 identifizierte 66 REDD+ Projekte, welche durch Kohlenstoffstandards bestätigt sind und zusammen 9.1 Millionen Hektar Wald schützen – ein Gebiet der Größe Portugals – und die Vermeidung von Emissionen in Höhe von 1.6 $GtCO_2e$ erwarten. Die multiple lineare Regressionsanalyse zeigt, dass die Vermeidung von Emissionen positiv mit der historischen Entwaldungsrate, der Wirksamkeit staatlicher Steuerung und einem Projektdesign für die Vermeidung geplanter Entwaldung korreliert. Projekte, deren Gebiete vielfältige Ökosystemleistungen erbringen, zeigen jedoch niedrigere Emissionsreduktionen, welches auf Zielkonflikte zwischen der Erreichung verschiedener Ökosystemleistungen hindeutet. Private Akteure bevorzugen Projekte mit hoher Kohlenstoffleistung und Kohlenstoffrechte sind für 75,8% der totalen Netto-Emissionsvermeidung in privater Hand, wohingegen lokale Gemeinden nur 10,4% der Kohlenstoffrechte besitzen. Diese Analyse zeigt somit den Bedarf von Schutzmaßnahmen zur Reduzierung von Zielkonflikten zwischen Ökosystemleistungen sowie für Maßnahmen zur Sicherstellung eines gerechten Zugangs zu Kohlenstoffrechten insbesondere für Gemeinden auf. Da die meisten Emissionsreduktionen durch die Vermeidung geplanter Entwaldung erreicht werden besteht das Risiko, dass es zu einer Verschiebung der Ursachen von Entwaldung in andere Regionen kommt. Daher besteht der Bedarf, die potenzielle Verlagerung von Entwaldung sowie die damit verbundenen Emissionen über Regionen und Kontinente hinweg besser zu erfassen.

Die Erkenntnisse dieser Dissertation heben hervor, dass es für die Bereitstellung von Informationen als Entscheidungshilfen für eine nachhaltige Landnutzung unter Berücksichtigung des Klimawandels multidisziplinäre und integrative Ansätze bedarf, welche Ökosystemleistungen einbeziehen, aber nicht nur auf diese begrenzt sind. Dabei sollten biophysikalische und sozio-ökonomische Indikatoren in Einheiten erfasst und berichtet werden, die einen Vergleich von Werten zwischen Studien und über Skalen hinweg erlauben. Des Weiteren sollte zukünftige Forschung Auswirkungen auf Biodiversität und Ökosystemleistungen über Regionen und Kontinente hinweg erfassen, sogenannte Telekonnektionen. Letztendlich bedarf es der Entwicklung alternativer Produktionsverfahren und Konsummustern, um die negativen Auswirkungen auf Biodiversität zu reduzieren. Dabei sollte nicht nur die Maximierung der Bindung von Kohlenstoff für die Vermeidung des Klimawandels als Kriterium für Nachhaltigkeit verwendet werden, sondern auch Kriterien, welche die vielfältigen Werte von Biodiversität und Ökosystemleistungen für die Gesellschaft einbeziehen.

List of articles and reports published as result of this dissertation

The following articles and reports were published before submission of this dissertation and originate from work and results of this dissertation.

Chapter 2 is available as project report (Sachstandsbericht) in German together with the corresponding database (both files are enclosed on CD at the back of this dissertation):

Förster, J., S. Schmidt, B. Bartkowski, N. Lienhoop, C. Albert, H. Wittmer (2017) Sachstandsbericht AP 2: Schätzung der Umweltkosten infolge Schädigung oder Zerstörung von Ökosystemen und Biodiversitätsverlust. Methodenkonvention 3.0 – Weiterentwicklung und Erweiterung der Methodenkonvention zur Schätzung von Umweltkosten., Helmholtz-Zentrum für Umweltforschung GmbH – UFZ, Leipzig im Auftrag des Umweltbundesamtes (UBA).

Förster, J., S. Schmidt, B. Bartkowski, N. Lienhoop, C. Albert, H. Wittmer (2017) Datenbank monetärer Werte von Ökosystemleistungen und Biodiversität in Deutschland. Beitrag zur Methodenkonvention 3.0 des Umweltbundesamtes (UBA). Version 1.0, Helmholtz-Zentrum für Umweltforschung GmbH – UFZ, Leipzig im Auftrag des Umweltbundesamtes (UBA).

Chapter 3 of this dissertation is a contribution to a Special Feature and is published under license by the Resilience Alliance in the journal Ecology and Society:

Förster, J., J. Barkmann, R. Fricke, S. Hotes, M. Kleyer, S. Kobbe, D. Kübler, C. Rumbaur, M. Siegmund-Schultze, R. Seppelt, J. Settele, J. H. Spangenberg, V. Tekken, T. Václavík, and H. Wittmer (2015) Assessing ecosystem services for informing land-use decisions: a problem-oriented approach. *Ecology and Society* 20 (3): 31. DOI: 10.5751/ES-07804-200331 URL: https://www.ecologyandsociety.org/vol20/iss3/art31/

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List of abbreviations

AIC	Akaike Information Criterion
APD	Avoiding Planned Deforestation
BMBF	German Federal Ministry of Education and Research
BMUB	German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety
CBD	Convention on Biological Diversity
CCB	Climate, Community and Biodiversity Standard
CE	Choice Experiment
CICES	Common International Classification of Ecosystem Services
COP	Conference of the Parties
CV	Contingent Valuation
DST	Decision Support Tool
EIA	Environmental Impact Assessment
ES	Ecosystem Services
ES_{total}	Total number of Ecosystem Service categories in project area
ESVD	Ecosystem Service Valuation Database
EU	European Union
GLUES	Global Assessment of Land Use Dynamics, Greenhouse Gas Emissions and Ecosystem Services
INNOVATE	Interplay among multiple uses of water reservoirs via innovative coupling of substance cycles in aquatic and terrestrial ecosystems
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IUCN	International Union for Conservation of Nature
LEGATO	Land-use intensity and Ecological Engineering – Assessment Tools for risks and Opportunities in irrigated rice based production systems
MAES	Mapping and Assessment of Ecosystems and their Services
MAgPIE	Model of Agricultural Production and its Impact on the Environment

MC	Methodological Convention – Methodenkonvention of the German Federal Environment Agency Umweltbundesamt (UBA)
MCDA	Multi-Criteria Decision Analysis
MIKE HYDRO	Multipurpose, map-based decision support tool for integrated water resources analysis, planning and management of river basins
MONERIS	Modelling Nutrient Emissions in River Systems
NDCs	Nationally Determined Contributions
NER	Net Emission Reductions
NER_{std}	Standardized Net Emission Reductions (per hectare and year)
NER_{total}	Total Net Emission Reductions (across all analysed REDD+ projects)
NPP	Net Primary Productivity
PECS	Programme on Ecosystem Change and Society
PES	Payments for Ecosystem Services
PV	Plan Vivo
R	The R Project for Statistical Computing
REDD+	Reducing Emissions from Deforestation and Degradation in developing countries and promoting sustainable forest management for conserving and enhancing forest carbon stocks
SDGs	Sustainable Development Goals
SES	Social-Ecological System
SLM	Sustainable Land Management
SuLaMa	Participatory research to support sustainable land management on the Mahafaly Plateau in south-western Madagascar
SuMaRiO	Sustainable Management of River Oases along the Tarim River
SWIM	Soil and Water Integrated Model
TEEB	The Economics of Ecosystems and Biodiversity
TEEB DE	Naturkapital Deutschland - The Economics of Ecosystems and Biodiversity
UBA	German Federal Environment Agency Umweltbundesamt

UFOPLAN	Research plan (Umweltforschungsplan) of the German Federal Ministry for the Environment, Nature Conservation, Building and Nuclear Safety (BMUB)
UFZ	Helmholtz Centre for Environmental Research
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
VCS	Verified Carbon Standard
VCS VM	Verified Carbon Standard Methodology
WTP	Willingness to Pay

1 Introduction

1.1 Motivation

Over the past decades, there has been increasing focus in environmental science on assessing the benefits that nature provides to society using the concept of ecosystem services (Vihervaara et al. 2010; Seppelt et al. 2011; Abson et al. 2014). Often, the aim of research on ecosystem services is to inform decision making on sustainable development in order to enhance human wellbeing and to conserve biodiversity (Daily et al. 2009). For example, the National Strategy for sustainable development of the German government has the goal to secure the provision of ecosystem services and conserve biodiversity in Germany and internationally (Die Bundesregierung 2016). However, there is little evidence on how information on ecosystem services actually informs decision making (Laurans et al. 2013) and whether the generated knowledge is relevant for decision makers (Honey-Rosés & Pendleton 2013; Martinez-Harms et al. 2015). This raises questions on whether science on ecosystem services actually achieves the goal of informing decision making. This dissertation addresses this knowledge gap on the relevance of science on ecosystem services for decision making by:

1. reviewing studies on monetary valuation of ecosystem services available for Germany, in order to inform the methodological convention (Methodenkonvention) of the German Federal Environment Agency Umweltbundesamt (UBA) on options for using this information in decision making on national policies (Chapter 2);
2. advancing conceptual considerations for ecosystem service assessments in order to enhance their relevance for decision making (Chapter 3);
3. assessing factors determining the performance of a policy that explicitly targets ecosystem services, using the example of the climate policy for reducing emissions from deforestation and degradation in developing countries (REDD+) (Chapter 4).

1.2 Ecosystem service concepts and research questions

Ecosystem services describe the benefits that nature provides for the wellbeing of people and society (Daily 1997; Millennium Ecosystem Assessment 2005; TEEB 2010a). Biodiversity, comprised of species, genes and ecosystems, is the foundation for the provision of ecosystem services (Cardinale et al. 2012). Ecosystem services include essential elements that support human life: the provision of food, water and materials; the regulation of water flows and quality; maintenance of soil productivity, air quality and carbon sequestration for climate regulation. The contribution of nature to aesthetics, spiritual inspiration, recreation as well as science and traditional knowledge are part of the cultural dimension of ecosystem services. In essence, biodiversity provides benefits in the form of ecosystem services, which ensure the wellbeing of people and society at large.

Ecosystem services are not free gifts from nature (Spangenberg, Görg, et al. 2014). First, the potential of ecosystems to provide certain benefits needs to be recognized. Second, resources need to be invested, such as knowledge, labour, time, materials or money, in order to access and harness these benefits. For example, the use of plants for food production requires knowledge and resources for their cultivation. The venom of snakes can be deadly, but with knowledge and materials it can be turned into medicine for the treatment of hypertension and cancer (Vyas et al. 2013). However, there are also many ecosystem processes that provide benefits without any input from humans, for example the uptake of carbon dioxide by oceans and forests for climate regulation.

Traditions, belief systems, political systems, markets and regulations influence which ecosystem services are used and how. Hence, human agency determines what aspects of biodiversity and ecosystems are regarded to be ecosystem services and how these are used (Spangenberg, Görg, et al. 2014). This process of human-nature interaction is shaping ecosystems and landscapes at local, national and global scale. Therefore, knowledge on ecosystem services is regarded to be useful for informing decisions on ecosystem management (Daily et al. 2009).

Ecosystems provide bundles of multiple ecosystem services with the characteristics of the ecosystem service bundles differing between landscapes, management practices and social-ecological systems (Raudsepp-Hearne et al. 2010). For example, provisioning ecosystem services such as food production dominate in agricultural landscapes, while regulating ecosystem services such as water regulation and flood control dominate in more natural landscapes (Foley et al. 2005). Information on ecosystem services can inform decision makers on trade-offs involved in land-use decisions, e.g. who gains and who loses as a result of land-use changes (Howe et al. 2014). Hence, knowledge on ecosystem services is in particular relevant for informing decisions on sustainable ecosystem management aiming at maintaining human wellbeing while conserving biodiversity (Goldman et al. 2008).

Scientific research on ecosystem services has become popular over the past decades with a substantive increase in published literature (Seppelt et al. 2011). The Millennium Ecosystem Assessment identified that a range of indicators and data exists on the biophysical aspects of ecosystem services, however, information on the economic dimension of ecosystem services is scarce (Millennium Ecosystem Assessment 2005). This has been regarded as a shortcoming in particular when it comes to informing decision making, which is often based on expected economic impacts. Therefore, the international initiative on 'The Economics of Ecosystems and Biodiversity' (TEEB) was initiated by the German government together with the European Commission and the United Nations Environment Programme (UNEP) with the strong support of multiple donor countries. The primary focus of the TEEB initiative is to inform policies and decision makers on the multiple values that ecosystems and biodiversity provide to society. The TEEB reports compiled existing data on the monetary and non-monetary value of ecosystem services (de Groot et al. 2012), synthesized economic valuation methods (TEEB 2010b) and provided guidance to decision makers on the economic importance of ecosystem services from local and regional (TEEB 2012) to global scale (TEEB 2011).

The ecosystem service concept is also of increasing relevance for national and international policies. In particular the Convention on Biological Diversity (CBD) together with the United Nations Framework Convention on Climate Change (UNFCCC) have been instrumental in mainstreaming a focus on ecosystem services based on the recognition that biodiversity and ecosystems provide benefits vital to society and human wellbeing (Vihervaara et al. 2010). Hence, the ecosystem service concept is also instrumental for working towards achieving the Sustainable Development Goals (SDGs) (Griggs et al. 2013).

At European level, policies and research efforts explicitly target ecosystem services. For example, Action 5 of the European Biodiversity Strategy states: "*Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020."* (European Commission 2011). For achieving this, the MAES-working group has been established, which supports ecosystem service mapping and research in multiple European countries.

In Germany, the concept of landscapes providing functions with relevance to society has a long tradition within the disciplines of landscape planning, landscape ecology and nature conservation (Bobeck & Schmithüsen 1949, Succow 1988, Bastian & Schreiber 1994). Although the term 'ecosystem services' is not explicitly mentioned, the conservation of biodiversity and ecosystems for their vital functions for the wellbeing of society is recognized and targeted by law through the Bundesnaturschutzgesetz (BNatSchG 2009). The assessment of landscape functions is applied in particular in the context of landscape planning (Albert et al. 2012), which is also true for other European countries such as The Netherlands (de Groot 1992). While Albert et al. (2012) diagnose considerable overlap in the concepts of landscape functions and ecosystem services, they also stress that the ecosystem service concept has advantages when assessing and evaluating benefits that ecosystems provide to human wellbeing. The concept of ecosystem services is found to be in particular useful for analysing the costs and benefits of land-use changes. Hence there is ample room for synergies between both concepts, in particular by combining the long established and widely applied methods of landscape planning with the ecosystem service concept for evaluating the benefits ecosystems provide to society (Albert et al. 2012).

The initiative Naturkapital Deutschland – TEEB DE has the goal to highlight the economic relevance of biodiversity and ecosystem services for Germany and informs decision making on how biodiversity conservation benefits the wellbeing of citizens in cities and rural landscapes, and contributes to climate change mitigation and sustainable development (Naturkapital Deutschland – TEEB DE 2012, Doyle et al. 2014). Given the increasing relevance of the ecosystem services concept it is also included in the National Strategy for sustainable development of the German government for guiding decision making at national and international level (Die Bundesregierung 2016).

Often it is assumed that more knowledge on ecosystem services will lead to more awareness of the significance ecosystems have for human wellbeing, which will inform decision making and consequently trigger change towards more sustainable land management (Daily et al. 2009). However, this assumption is contested, for example, by the fact that only 8 out of 340 ecosystem service valuation studies in peer-reviewed journals actually mention how their results played a role in decision making (Laurans et al. 2013). There is a gap between the expectation of the science community concerning the relevance of their work for decision making and the reality of decision makers using information on ecosystem services in the management of natural resources and policy making (Daily et al. 2009).

Furthermore, it has been diagnosed that integrating ecosystem services into planning and decision making would benefit from standardization of assessment methods and ecosystem service information (Galler et al. 2016). Policy processes such as regulatory impact assessments (Gesetzesfolgenabschätzung) could be guided by information on ecosystem services. Already today the German Federal Environment Agency Umweltbundesamt (UBA) sets the monetary value for the cost of carbon dioxide emissions at 80€/tCO_2 (Methodenkonvention, Umweltbundesamt 2012). This carbon price is applied in public land-use decisions and informs on the value ecosystems have for carbon sequestration and for climate change mitigation. However, the importance of other ecosystem services is usually not considered. This is due to the fact that in particular regulating and cultural ecosystem services are hard to quantify and value, leaving great uncertainties about trade-offs involved in land-use decision. Therefore, the German Federal Environment Agency Umweltbundesamt (UBA) is seeking pragmatic approaches towards quantifying the monetary value of ecosystem services in order to better account for costs related to the degradation and loss of ecosystems. This information is of relevance, for example, for assessing environmental costs in regulatory impact assessments (Gesetzesfolgenabschätzung). Hence there is the research question of:

> **Research question 1:** *What information on monetary values of ecosystem services is available for Germany and how can it be used for informing decision making on the design of policies?*

The ecosystem service concept is not only regarded to be a bridging concept between natural and social science but also to connect science, policy and practice (Braat & de Groot 2012). Hence, informing decision making through information on ecosystem services is at the core of the concept (Braat & de Groot 2012) (Daily et al. 2009).

Following the Millennium Ecosystem Assessment and the TEEB initiative, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was created, which has the goal to address the gaps between science and decision making in a more strategic and formalized approach. It is the mission of IPBES to assess "…*the state of biodiversity and of the ecosystem services it provides to society, in response to requests from decision makers"* (IPBES 2017). While in the past it was often regarded to be a one-directional flow of information, with environmental science providing knowledge to decision makers, IPBES has established a process that allows for the creation of information on biodiversity and ecosystem services in response to a clearly defined demand for information in decision making.

It is also suggested that the ecosystem service concept changes the discourse: While biodiversity conservation is often regarded to pose a trade-off to economic development, the ecosystem service concept helps to focus on win-win options for conservation and development (de Groot et al. 2010). This is supported by scientific evidence that biodiversity underpins most ecosystem services (Cardinale et al. 2012). Furthermore, the concept is also found to promote stakeholder engagement in environmental planning as it makes consequences of decisions on land-use change more explicit for stakeholder groups (Galler et al. 2016).

However, challenges remain concerning the use and acceptance of the ecosystem service concept in science and practice (Schröter et al. 2014). The use of the ecosystem service concept is criticized for being vague with inconsistencies in the underlying frameworks, leaving room for interpretation (Nahlik et al. 2012). Therefore it is suggested that more decision-oriented approaches to ecosystem service assessments are needed for overcoming these obstacles (Nahlik et al. 2012). This requires an analysis of how the ecosystem service concept is currently used and how it can be improved in order to better address information needs for decision making. Hence there is the research question of:

> **Research question 2:** *What gaps exist in current ecosystem service frameworks and how can the design of ecosystem service assessments be improved in order to increase their relevance for decision making?*

Despite the above mentioned shortcomings, the ecosystem service concept is already applied in the formulation of environmental policies. Under the UNFCCC it is recognized that ecosystems play a critical role for carbon sequestration and therefore for climate change mitigation. Since deforestation and forest degradation are estimated to contribute 12% to anthropogenic greenhouse gas emissions (Houghton et al. 2012), a policy has been developed that explicitly targets the conservation of forests for carbon sequestration: reducing emissions from deforestation and degradation in developing countries (REDD+) (UNFCCC 2008). The policy of REDD+ is based on the principle of payments for ecosystem services (PES) with carbon emissions that are avoided through the implementation of REDD+ activities being traded as carbon credits on voluntary carbon markets. Attaching a monetary value to the ecosystem service of carbon sequestration serves as an incentive for forest users to switch from deforestation to forest conservation. It also gives economic weight to the development of national and international policies that aim at reducing forest loss.

REDD+ is an example for a policy that is based on the ecosystem service concept and explicitly targets the management of ecosystems services for climate change mitigation. However, questions have been raised whether a focus on ecosystem services is actually helpful for sustainable land management. There is the concern that policies that focus on managing natural habitats with a utilitarian perspective that is prioritizing an ecosystem service over others can undermine biodiversity conservation (Ridder 2008) and social equity (Corbera 2012). Therefore, safeguards have been developed for ensuring that REDD+ projects not only focus on maximizing carbon sequestration but include biodiversity conservation and equitable stakeholder participation in their objectives (UNFCCC 2011).

While this is reasonable from a perspective of ensuring biodiversity conservation and equity, there is empirical evidence that overloading schemes of payments for ecosystem services (PES) with too many side-objectives can undermine their goal of achieving their primary objective (Wunder et al. 2008). In the case of REDD+ the primary objective is to reduce carbon emissions from deforestation and degradation. Hence there is the question of:

> **Research question 3:** *How does the inclusion of multiple ecosystem services in the design of REDD+ projects impact their performance of reducing carbon emission from deforestation and forest degradation?*

The research questions identified above address key challenges of using information on ecosystem services in decision making on sustainable land management and climate change mitigation. This dissertation addresses these research questions by analyzing the current knowledge on monetary values of ecosystem services in Germany, assessing gaps in the conceptual frameworks underlying ecosystem service assessments, and by focusing on the specific example of the REDD+ policy for understanding how ecosystem service information can support decision making on sustainable land management under climate change.

1.3 Overview of dissertation structure

For assessing what information on monetary values of ecosystem services is available for Germany and how this can be used for informing the design of policies (research question 1), Chapter 2 provides a review of monetary valuation studies for Germany. The review is a contribution to advancing the methodological convention (Methodenkonvention) of the German Federal Environment Agency Umweltbundesamt (UBA), which aims at informing regulatory impact assessments (Gesetzesfolgenabschätzung) on the costs of environmental impacts. Opportunities and challenges related to assessing and using monetary values for ecosystem services are explored in order to better inform decision making on land use in Germany (Albert et al. 2012).

One challenge is that the characteristics of ecosystems, their services and benefits are highly site-specific and dependent on the social-ecological context. Hence it is difficult to synthetize ecosystem service information across sites, transfer it to other regions, or up-scale from local to national level (Spash & Vatn 2006). This site-dependence requires ecosystem services assessments to take into account locally specific landscape characteristics including the decision-making contexts. However, this is a time-intensive and expensive process with both resources often being scarce in situations of decision making. Therefore, practitioners and decision makers are seeking pragmatic approaches that allow the development and use of standardized information on ecosystem services in decision making.

In order to better understand how the design of ecosystem service assessments can be improved for increasing their relevance for decision making (research question 2), Chapter 3 reviews existing frameworks of ecosystem service assessments, identifies gaps and proposes an alternative framework with a more problem-oriented approach.

The developed framework helps to ensure that ecosystem service assessments better target information needs relevant for decision makers (Chapter 3, Fig. 3.2) and was co-designed and tested in four international case studies in Brazil, China, Madagascar and Vietnam. The four case studies are part of the Sustainable Land Management Program funded by the German Federal Ministry of Education and Research (BMUB), which also financed this dissertation as part of the synthesis project GLUES - Global Assessment of Land Use Dynamics, Greenhouse Gas Emissions and Ecosystem Services. It was recognized by the BMUB that there is the need for scientific synthesis across the research projects of Programme for creating knowledge that can advance the science of ecosystem services and inform policies. Therefore, Chapter 3 is based on the collaboration with partners of the Sustainable Land Management Program and synthesizes knowledge of multiple research projects. Such synthesis of scientific information on ecosystem services is critical for decision support. Hence, the `Programme on Ecosystem Change and Society (PECS): Knowledge for Sustainable Stewardship of Social-ecological Systems` was created, an interdisciplinary research programme that is part of the international science platform Future Earth. The GLUES project has been endorsed by PECS and Chapter 3 of this dissertation is a contribution to a Special Feature of the PECS-Programme in the Journal of Ecology and Society (Förster et al. 2015).

For assessing how information on ecosystem services can be relevant for the design of policies (research question 3), Chapter 4 assesses the performance of projects under the climate policy of reducing emissions from deforestation and degradation in developing countries (REDD+). While the main objective of REDD+ is to reduce deforestation for maintaining the ecosystem service of carbon sequestration, trade-offs with multiple side-objectives for securing other ecosystem services and biodiversity conservation are to be expected (Wunder et al. 2008; Phelps et al. 2012; Bustamante et al. 2014). In order to inform the design of REDD+ policies on potential trade-offs, 66 REDD+ projects were analyzed for factors that influence the performance of reducing carbon emissions. Using meta-analysis, factors of biophysical and socio-economic context as well as factors concerning the design of REDD+ projects were assessed. Besides others, this included the number of ecosystem services present in the project area. The analysis in Chapter 4 is a variable-oriented meta-analysis using mixed meta-analytical methods (Magliocca et al. 2015). This approach is considered to be suitable for analyzing regional and global environmental change with an explicit focus on informing decision making (Rudel 2008).

2 Incorporating environmental costs of ecosystem service loss in political decision making: a synthesis of monetary values for Germany

Summary

Germany faces on-going degradation and loss of biodiversity and ecosystems. As a consequence, goods and services provided by nature for human well-being, so-called ecosystem services, are lost. While the negative ecological impacts of ecosystem conversion are known, the economic costs are neglected in decision making. To fill this gap, the German Federal Environment Agency Umweltbundesamt (UBA) aims at developing standard estimates for environmental costs of ecosystem service loss. For informing this process, a literature review was conducted and a database of monetary values for regulating and cultural ecosystem services in Germany has been developed. In total, 109 monetary valuation studies of ecosystem services were identified with the majority focusing on forests and wetlands. After applying a set of selection criteria, only 6 out of 109 valuation studies (5.5%) were identified to be relevant for informing decisions on national policies. Overall, monetary information on regulating and cultural ecosystem services is scattered and scarce compared to information on provisioning services, which is accounted for in detail in national statistics. This imbalance in information likely contributes to the distortion in land-use policies, giving preference to maximizing provisioning services in agricultural production and forestry, while neglecting the societal relevance of regulating and cultural services. Therefore, decision makers have to account for the trade-off in relying on only few cost estimates that are scientifically robust, while being pragmatic enough to include also vague estimates in cases where data is lacking. Overall, it was found that few scientific studies use indicators for ecosystem services that are relevant for informing policies and there is the need for scientific studies to better target the specific information needs in decision making. As monetary estimates provide only a partial representation of ecosystem benefits, it is recommended that decision making should also use complementary and qualitative information that accounts for the multiple values biodiversity and ecosystems provide to human wellbeing.

2.1 Introduction

Germany faces on-going degradation of ecosystems with negative consequences for biodiversity and the services ecosystems provide to individuals and society. Key drivers are urbanisation, land sealing for infrastructure and settlements, and conversion of grassland to cropland (Tietz et al. 2012; Niedertscheider et al. 2014). A number of national policies aim at reducing the loss of biodiversity and ecosystems, including the federal law on nature conservation (Bundesnaturschutzgesetz*)* (BNatSchG 2009), the national strategy on biological diversity (BMUB 2007) and the National Strategy for sustainable development, which includes Germany's commitment to contribute to the achievement of Sustainable Development Goals (SDGs) (Die Bundesregierung 2016).

Despite these policies and strategies, the degradation and loss of biodiversity and ecosystems continues, causing social costs to society. For example, the conversion of wetlands to agricultural land aims at increasing benefits from crop production but causes costs due to declining water quality (Dehnhardt 2002), the emission of soil carbon (Grossmann & Dietrich 2012) and damages from flood events as less water is being absorbed in the landscape (Hartje & Grossmann 2013). Current changes in laws and regulations, for example for reducing the impacts of development of urban areas, fall short in reducing land sealing and ecosystem loss (Sachverständigenrat für Umweltfragen SRU 2017).

Making costs and benefits related to ecosystem services more explicit is believed to inform decision making on more sustainable land-use options both in economic and ecological terms (Bateman et al. 2013). Ecosystems and their configuration across the landscape provide multiple ecosystem services, so-called ecosystem service bundles, for a diverse range of beneficiaries within society (Raudsepp-Hearne et al. 2010). Current land-use decisions often focus only on a few selected ecosystem services, prioritizing provisioning services with market value (e.g. agricultural crop production), while ignoring regulating services (e.g. water provision) and cultural services (e.g. landscape aesthetics) that are not valued in markets (Bateman et al. 2013). Hence, land-use decisions often aim at increasing private benefits from market goods, for example from crop and timber production, neglecting public benefits from ecosystem services such as water regulation, carbon sequestration and landscape aesthetics.

Costs related to the loss of regulating and cultural ecosystem services are mainly borne by the public, e.g. in the form of increased costs for the provision of drinking water or by damages to health (TEEB 2012). The costs of ecosystem service loss can occur in the form of damage costs, abatement costs or costs for replacing ecosystems with alternative man-made structures and services. Compensating or reversing the degradation and loss of biodiversity and ecosystem services through habitat restoration or replacement with man-made infrastructure and services can be expensive or is simply impossible.

Estimates for economic costs and benefits of land-use options can inform decision making on the multiple benefits biodiversity and ecosystems provide to human wellbeing as well as on the economic consequences of ecosystem loss (Sukhdev & Kumar 2008; TEEB

2010a). For example, it has been shown that the economic benefits of conserving biodiversity and ecosystems outweigh the costs of conservation when benefits of ecosystem services are accounted for (Wüstemann et al. 2014). However, most of economic valuation studies focus only on a few ecosystem services including agricultural crops, carbon sequestration, water quality, recreation (e.g. number of visitors) or willingness to pay for conservation (Bateman et al. 2013; Wüstemann et al. 2014) as there are gaps in biophysical and socio-economic data for other ecosystem services (Luck et al. 2009). The benefit-cost ratio of conserving ecosystem services would increase even further, if more ecosystem services were to be included in the accounting.

As decision making is increasingly based on economic considerations, including cost-benefit analysis, there is concern that decision making will continue to ignore the costs of losing biodiversity and ecosystems as long as the costs and benefits related to ecosystem services are not included in monetary terms (Sukhdev & Kumar 2008). Furthermore, while costs of protecting nature have to be considered in ex-ante policy impact assessments, the inclusion of benefits from nature conservation in form of ecosystem services is often optional. Therefore, benefits from ecosystem services are only considered, if reliable monetary estimates are available. Hence there is an increasing focus on including monetary values of ecosystem services in the assessment of land-use decisions in order to better account for the costs and benefits related to impacts on ecosystems and their ecosystem services (Fisher et al. 2008; Naturkapital Deutschland – TEEB DE 2012).

In Germany, the Federal Environment Agency Umweltbundesamt (UBA) is aiming at establishing standardized estimates of monetary values for the benefits of ecosystem services in order to account in decision making for ecosystem service benefits and costs related to their loss. Already today, the UBA methodological convention (UBA Methodenkonvention) provides standardized cost estimates for a range of environmental impacts. For example, the cost of carbon emission is currently estimated at 80€/tCO_2 based on damage costs resulting from climate change (Umweltbundesamt 2012). This standardized cost estimate informs decision making in public procurement or regulatory impact assessments (Gesetzesfolgenabschätzungen). Currently, the UBA methodological convention is updated with the aim of determining cost estimates for ecosystem service loss caused by land conversions in order to inform policy processes at national level including regulatory impact assessments.

It is the aim of this study to review the state of evidence of economic benefits of ecosystem services and costs related to ecosystem service loss for land-use changes in Germany and to derive recommendations for formulating cost estimates for the use of policy impact assessments. Challenges involved in economic valuation and in generalization of values are highlighted as well as the implications these challenges have for using economic values of ecosystem services in decision making.

2.2 Materials and methods

First, information needs were identified for updating the methodological convention of the Umweltbundesamt (UBA). Second, a literature review was conducted for developing a database with economic values for regulating and cultural ecosystem services. Third, a consultation process was conducted involving a) experts in ecosystem service assessment and valuation for ensuring quality and completeness of the literature review and b) experts from the Umweltbundesamt (UBA) for agreeing on criteria for selecting valuation studies relevant for informing national policies. Based on the outcome of this review process, challenges and opportunities for using economic values of ecosystem services in decision making are highlighted and recommendations for their application are derived.

2.2.1 Defining information needs

In the process of updating the UBA methodological convention, a lack of information on cost estimates of ecosystem service loss have been identified for the following major land conversion processes in Germany:

I. Conversion of extensively or intensively used grassland into arable land (including loss of fringes of water bodies and small forest formations and coppice);
II. Conversion of grassland, arable land, forests and accompanying vegetation to sealed surfaces including settlements and roads;
III. Drainage of wetlands;

Given Germany's large imports of agricultural commodities from tropical forest regions and related conversion of tropical forests with impacts on ecosystem services (Kissinger et al. 2012, Schmitz et al. 2015, Liu et al. 2016), tropical forest conversion was also included:

IV. Conversion of tropical rainforest into grassland or arable land.

2.2.2 Literature review

For identifying studies with monetary valuation of ecosystem services related to the ecosystems and conversion processes I.-IV., bibliographic databases (e.g. Web of Science) and databases with monetary values for ecosystem services (e.g. ESVD) were searched (Supplementary material S2.2). Both peer-reviewed and grey literature were considered. Although grey literature is often not peer-reviewed as academic publications, it can be a useful complementary resource (Rothstein & Hopewell 2009). Provisioning services such as agricultural production and timber production are not considered in this review, as these are ecosystem services that are already captured in land-use statistics at local and national level (Statistisches Bundesamt 2017). Instead, the focus of this review is primarily on regulating and cultural services that are usually not captured in land-use statistics.

The review includes mainly primary valuation studies for ensuring complete recording of information on biophysical and socio-economic context, study design, valuation methods and underlying assumptions (Supplementary material S2.3 contains the recorded data).Various valuation methods are at hand to assess the economic costs of ecosystem service loss associated with land-use changes and to derive cost estimates. A fundamental element of the ecosystem service paradigm is the recognition that changes in ecosystems influence the provision of ecosystem services, and that these changes in services have influence on human welfare. In economic terms, an increase in the flow of ecosystem services is regarded as benefits and a decrease in flows is regarded as costs. These benefits and costs reflect the preferences of individual stakeholders affected by the change. Both market and non-market valuation methods can be used to estimate the change of economic value associated with the changes in ecosystem services flow. Market valuation means economic values are derived from market prices. Examples include the forgone economic value of agricultural products or timber, which is sold on a market (market analysis) due to expansion of settlements, the costs of offset activities to compensate for a new road (restoration costs) or water treatment due to soil runoff when grassland is converted to arable land (damage cost). Many ecosystem services are not traded in markets and therefore have no market price. In this case, it is necessary to assess the economic value of a decreased flow of ecosystem services through direct or indirect non-market valuation methods. Direct methods (also called stated preference methods) refer to contingent valuation (CV) and choice experiments (CE), where the affected general public is asked directly in a survey for their willingness to pay (WTP) to obtain a land-use change (to value the benefits of an increased ecosystem services flow) or their WTP to avoid a land-use change (to value the costs of a decreased ecosystem services flow). WTP can also be obtained indirectly by assuming that economic value is reflected in the costs incurred to travel to specific sites, such as recreational visits to wetland areas (travel cost method), or additional property prices paid to live in specific environment, e.g. in the vicinity of a forest (hedonic pricing method). In the latter two approaches, economic value is ‘revealed’ through observable behaviour (Garrod & Willis 1999; Hansjürgens & Lienhoop 2015).

Given that the outcome of monetary valuation studies is highly dependent on the context of the study area and the choice of valuation method, the following characteristics were recorded in the database including:

- full reference of study;
- ecosystem service classified according to TEEB (2010) and CICES (Haines-Young & Potschin 2012);
- spatial and temporal dimension (area of study site, location, year of valuation etc.);
- information on biophysical and socio-economic context;
- valuation method, sample size and underlying assumptions;
- discount rate;
- monetary value (minimum, mean, median, maximum).

2.2.3 Reporting of monetary values in database

The database includes the original monetary values as provided by studies together with inflation-adjusted values in Euro (with 2014 as base year). Monetary values were adjusted to 2014 values using the consumer price index for the year of valuation relative to the year 2014 based on the Deutsche Bundesbank (2016) (Equation 1, Supplementary material S2.8). In a second step, estimates in the currency Deutsche Mark (DM) were converted to Euro (€) using the general currency conversion factor of 1 Euro = 1.95583 DM.

Equation 1:

$$Value\ in\ €_{2014} = Value_{reported} * \frac{VPI_{2014}}{VPI_{Year\ of\ valuation}}$$

Values from other countries and in other currencies were inflation-adjusted using the respective consumer price index (Equation 1; Supplementary material S2.8). For allowing comparability of values across countries, values were adjusted for Purchasing Power Parity (PPP, World Bank 2016) and converted to Euro (Equation 2).

Equation 2:

$$Value\ in\ €_{2014} = Value_{2014} * \frac{PPP_{Germany}}{PPP_{Country\ of\ valuation\ study}}$$

In a third step, groups of monetary values with similar metrics were formed including:

i. €/ha/a
ii. €/ha
iii. €/Person/a
iv. other;

The classification used for ecosystem services follows the Common International Classification of Ecosystem Services (CICES) of the European Environmental Agency. Valuation methods were grouped in accordance with the database of de Groot et al. (2012) in order to ensure compatibility with existing databases. For the conversion of values of larger study areas into values per hectare, linear scale effects were assumed. Values per household were divided by the average number of household members in Germany (1.99 members) based on the German Federal Statistical Office (Statistisches Bundesamt DESTATIS 2015). For studies from other countries, household values were divided by the respective average number of household members (Supplementary material S2.8). Similarly, inflation-adjustment and currency conversions are based on assumptions of data homogeneity.

2.2.4 Identification of monetary values relevant for the methodological convention of the Umweltbundesamt (UBA)

During a two-day workshop the results of the literature review were assessed by external experts on ecosystem service assessments and monetary valuation for ensuring quality and completeness of the literature review. This ensures that the most relevant valuation studies for biodiversity and ecosystem services are included in the database.

The monetary values obtained from the literature review must allow a certain degree of generalization in order to be representative for the conversion processes I.-IV. and to allow for informing policy impact assessments and decision making at national level. In consultation with experts from the Umweltbundesamt (UBA), criteria were identified for evaluating the suitability of valuation studies for deriving cost estimates for ecosystem service loss and for informing the methodological convention of the Umweltbundesamt (UBA). These selection criteria for monetary valuation studies include:

a) Thematic focus of study is at least on one of the relevant conversion processes and ecosystems (I-IV);

b) Explicit description of biophysical and socio-economic context;

c) Transparency of study design, methods and underlying assumptions;

d) Monetary values refer to a distinct, clearly identifiable ecosystem service or ecosystem service bundle;

e) Monetary values are derived using common valuation methods (cost-based or benefit-based approaches);

f) Monetary values are reported in Euro per hectare (ii. €/ha) or allow for currency conversion and unit-adjustment;

g) Representativeness of monetary values: the reasoning for minimum – maximum ranges of values should be linked to ranges in biophysical or socio-economic factors (e.g. carbon content of ecosystem per hectare).

2.3 Results

Based on the review of literature and existing databases, 257 studies were identified with a thematic focus on ecosystem service valuation in Germany and a focus on ecosystems and land cover types related to conversion processes I.-III. (grasslands, arable lands, wetlands, forests and sealed surfaces) (Fig. 2.1). Of the 257 studies 109 turned out to be distinct valuation studies with a total of 638 monetary values for ecosystem services (Fig. 2.2). The largest number of monetary values is available for wetlands (n = 169) and forests (n = 170). 21 out of 109 studies comply with the selection criteria a.) to e.) with study design and information on biophysical and socio-economic context being sufficiently transparent. Only six studies comply with all selection criteria a) to g) providing 101 monetary values. These studies were used for informing the methodological convention of the German Federal Environment Agency Umweltbundesamt (UBA) on possible costs involved in the loss or degradation of ecosystem services (Table 2.1).

Monetary values for ecosystem services from tropical forests were derived from already existing databases including de Groot et al. (2012) and the literature review by Ojea et al. (2016) (Fig. 2.3). From the 23 studies with 171 monetary values, 114 values comply with the criteria on transparency (criteria a. to e.). Five aggregated monetary values based on the meta-analysis of Ojea et al. (2016) comply with criteria a. to g and have been selected for informing the UBA methodological convention.

In total, the database contains 809 monetary values from 132 valuation studies for ecosystems in Germany and tropical forests. Almost half of the monetary values (46%, n = 375) provide estimates for stocks or marginal changes in ecosystem services within the same ecosystem type (Fig. 2.4). About one third of monetary values (36%, n = 288) address ecosystem conversion processes (I.-IV.). Wetland conversion is the process for which most monetary values of ecosystem services are available (20%, n = 161) and includes estimates for wetland restoration.

The monetary values originate from studies with a great diversity of valuation methods. In total, about 11 major groups of valuation methods have been identified (Fig. 2.5). Using replacement costs as means for valuing ecosystem services is the most common approach in Germany, followed by choice experiments. For valuing ecosystem services of tropical forests, willingness to pay and market price methods dominate. However, the majority of monetary values originate from valuation studies that apply a mix of valuation methods.

According to the Common International Classification of Ecosystem Services (CICES), the reviewed valuation studies address 20 ecosystem service classes (Fig 2.6 a.-g. and 2.7). Some of the studies also value bundles of ecosystem services, e.g. the joint valuation of recreation, aesthetics and habitat provision for biodiversity.

For ecosystems in Germany, the class “Biodiversity (habitat, species) (2.3.1.)” is valued most frequently (n = 237). It includes the appreciation of people for ecosystems to provide habitat for species and related diversity within ecosystems and across landscapes. Ecosystem services with a high number of monetary values also include “Physical experience (recreation) (3.1.1.)" in particular for forests (n = 50) and "Water quality (N and P retention) (2.3.4.)" in particular for wetlands (n = 72). Agricultural production and timber production are not considered, as the focus of this assessment is on regulation and cultural ecosystem services.

For tropical forests, frequently valued ecosystem services include "Biodiversity (habitat, species) (2.3.1.)", "Food provision (1.1.1.), bundles of multiple ecosystem services, "Material provision (1.2.1.)" and "Physical experience (recreation) (3.1.1.)" (Fig. 2.7).

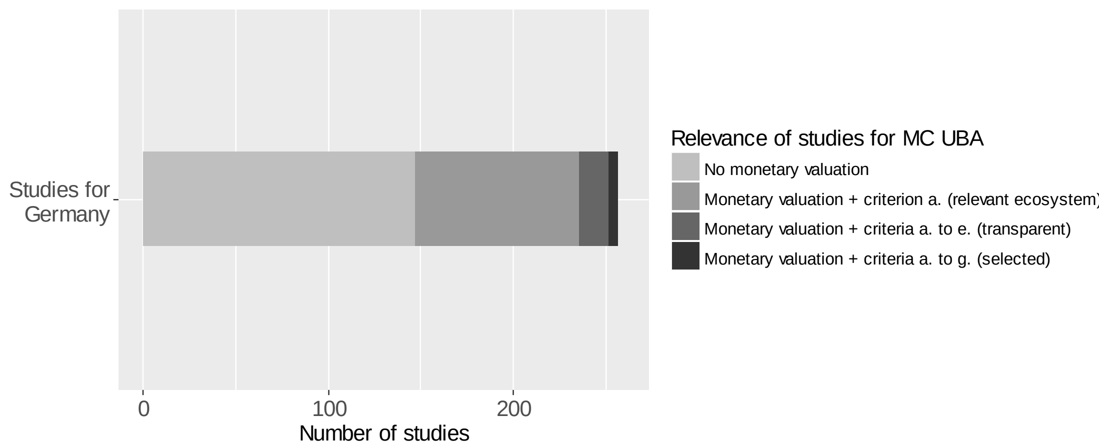


Figure 2.1: Number of monetary valuation studies for ecosystem services in Germany.
In total, 257 studies with a focus on ecosystem service valuation in Germany were reviewed. 148 studies do not estimate monetary values or only cite values from other valuation studies. 109 studies are primary valuation studies, estimating monetary values for ecosystem services related to conversion processes I-III (including grasslands, arable lands, wetlands, forests and sealed surfaces). Of the 109 studies 21 studies are sufficiently transparent (complying with selection criteria *a.* to *e.*). Of the 21 studies only six studies comply with all selection criteria (a. to g.) by being sufficiently transparent, reporting monetary values in a common unit (e.g. € per ha) and minimum-maximum ranges can be explained by biophysical or socio-economic context. Only these studies were selected for informing the methodological convention (MC) of the Umweltbundesamt (UBA).

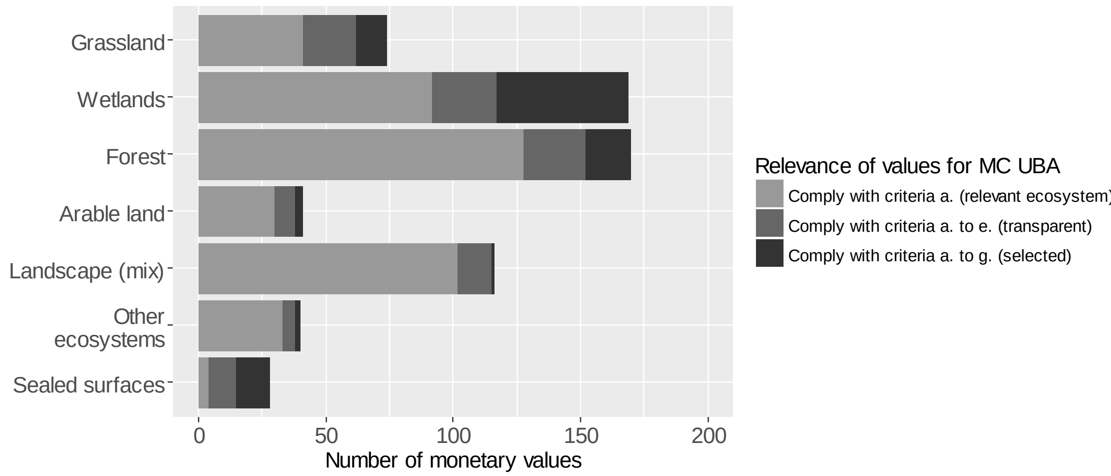


Figure 2.2: Number of monetary values for ecosystem services of common land-cover types in Germany. The database contains 633 monetary values for ecosystem services from 109 primary valuation studies that focus on at least one of the ecosystems involved in the conversion processes (I. - IV.). The majority of monetary values have been identified for forests and wetlands. 15 studies with 204 monetary values are sufficiently transparent and comply with selection criteria a. to e. Six studies with 101 monetary values comply with all selection criteria (a. - g.) and have highest relevance for informing the methodological convention (MC) of the Umweltbundesamt (UBA) on possible costs involved in the loss or degradation of ecosystem services.

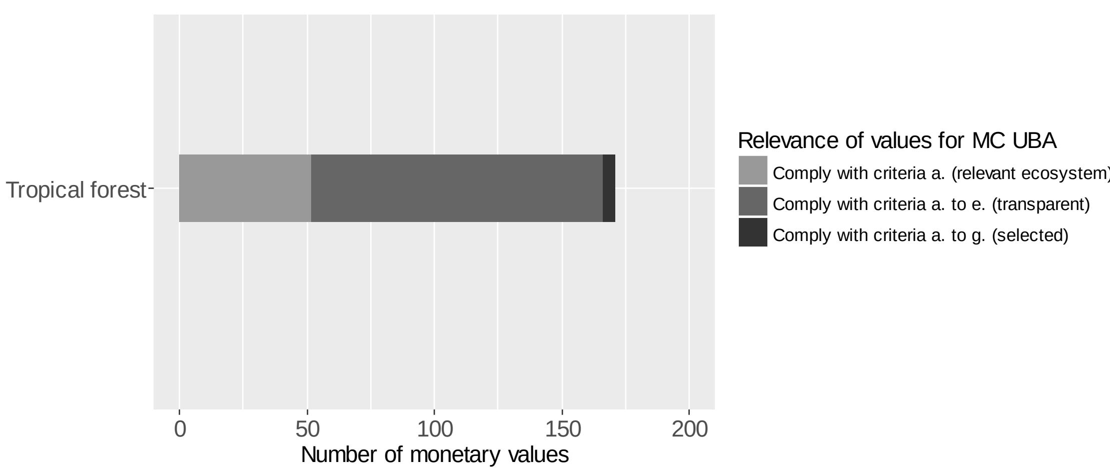


Figure 2.3: Number of monetary values for ecosystem services of tropical forests. Tropical forests are addressed in conversion process IV. The database contains 171 monetary values for ecosystem services of tropical forests from a total of 23 monetary valuation studies. Of the 171 monetary values 114 comply with criteria a. to e. with regards to the transparency of study design and methods. Five aggregated monetary values from the review by Ojea et al. (2016) comply with criteria a. - g. and have highest relevance for informing the methodological convention (MC) of the Umweltbundesamt (UBA) on possible costs involved in the loss or degradation of ecosystem services.

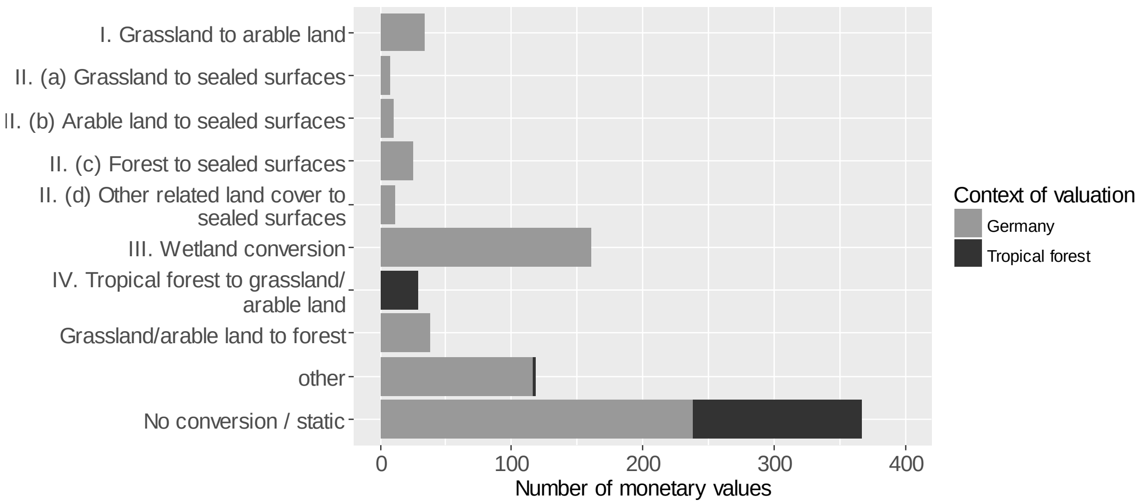


Figure 2.4: Number of monetary values for ecosystem services impacted by ecosystem conversion processes. Almost half (46%, n = 375) of the monetary values for ecosystem services originate from valuation studies that estimate stocks or marginal changes within the same ecosystem type (no conversion). 36% of monetary values (n = 288) address one of the four relevant conversion process (I.-IV.). Wetland conversion (III.) is the process with most monetary values (n = 161).

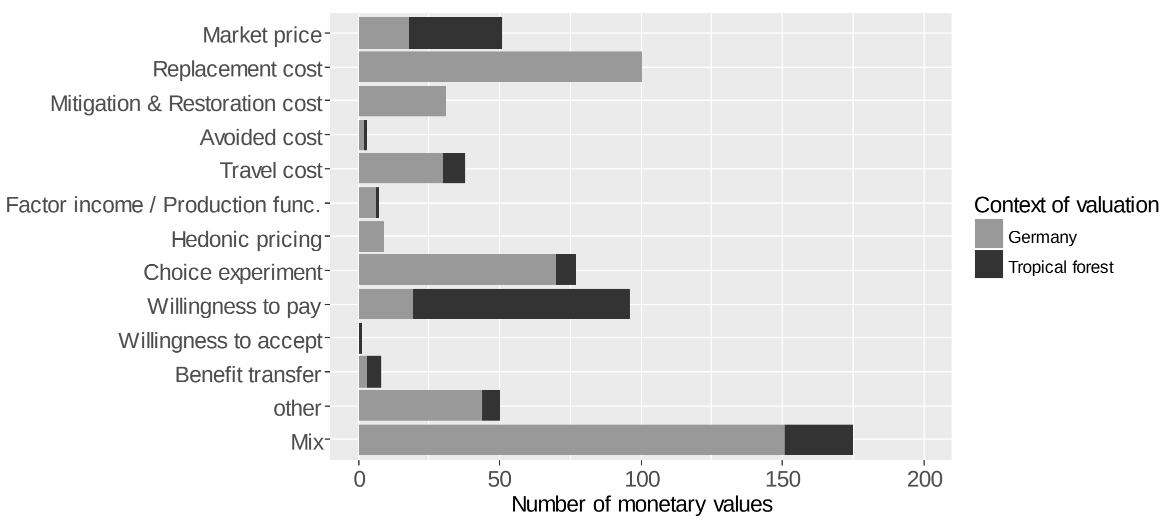


Figure 2.5: Monetary valuation methods. The majority of monetary values originate from valuation studies that apply a mix of valuation methods. Using replacement costs as means for valuing ecosystem services is a common approach in Germany, followed by choice experiments. In tropical regions, willingness to pay and market price methods dominate.

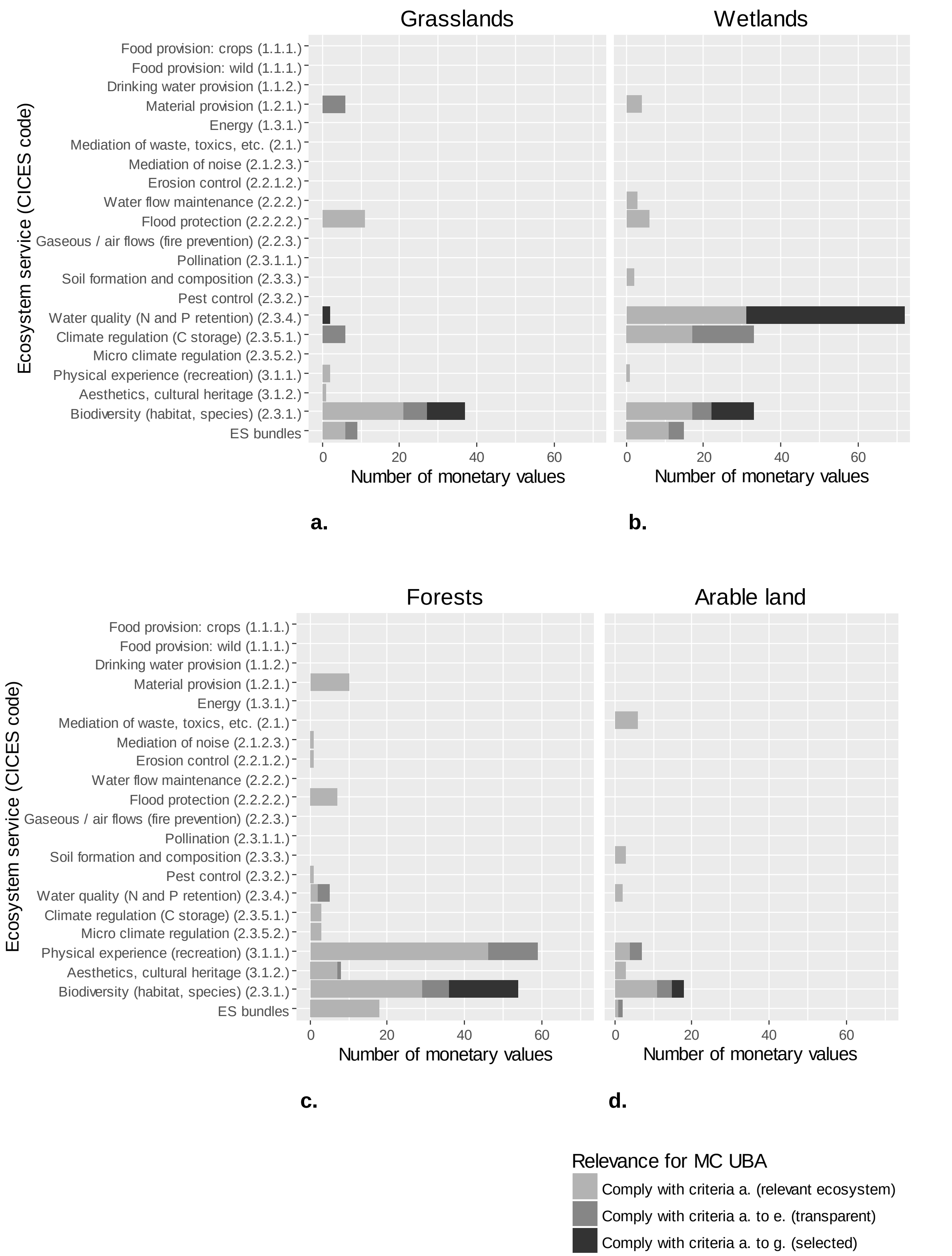
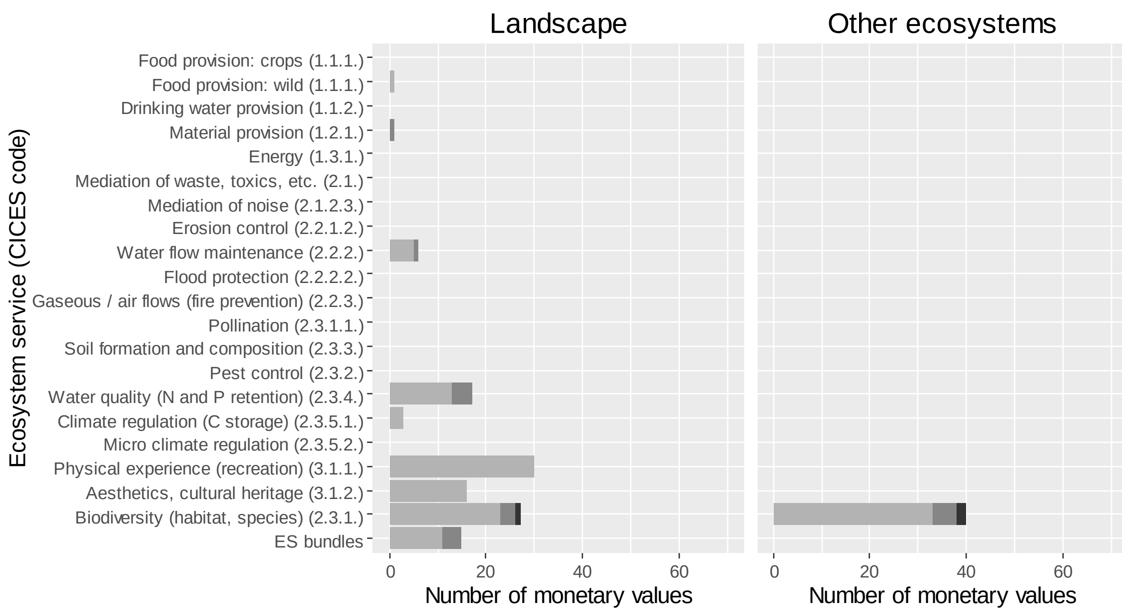
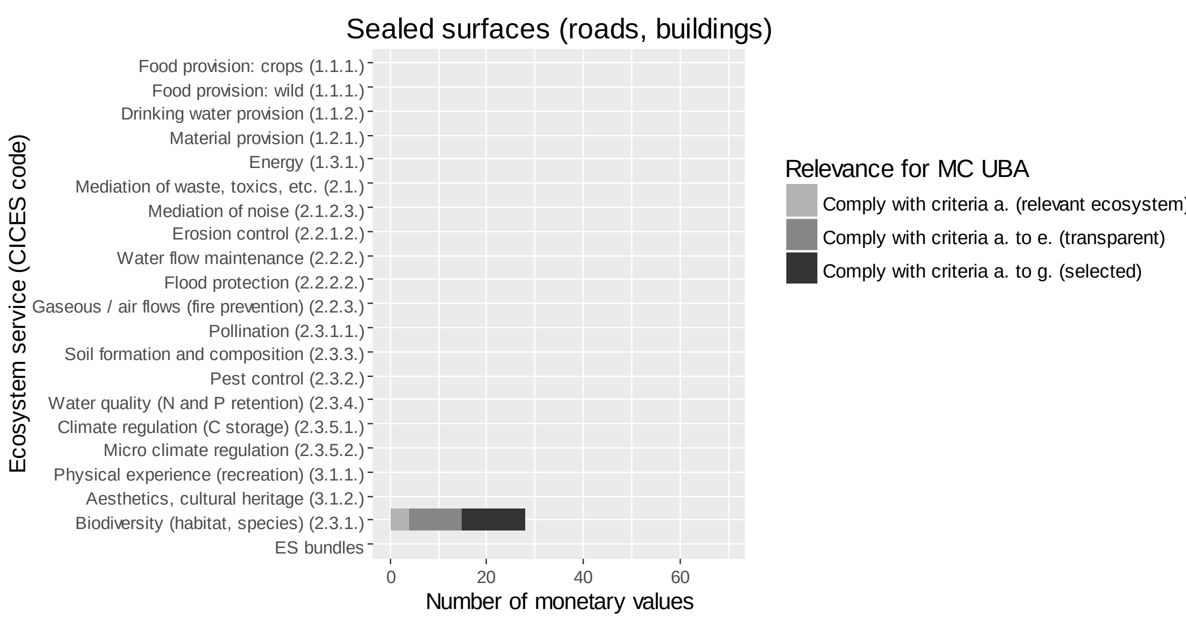


Figure 6 (a. - g.): (continues next page)



e. **f.**



g.

Figure 2.6 (a. – g.): Number of monetary values for ecosystem services in Germany (classified according to CICES). The graph includes all ecosystem services classes addressed by the 109 reviewed valuation studies for Germany. The class “Biodiversity (habitat, species) (2.3.1.)” is valued most frequently across all ecosystem types (a. to g.) and includes the appreciation of people for ecosystems to provide habitat for species and diversity of ecosystems across landscapes. Ecosystem services with a high number of monetary values also include “Physical experience (recreation) (3.1.1.)” in particular for forests and “Water quality (N and P retention) (2.3.4.)” in particular for wetlands. **Note:** Agricultural production and timber production are not considered in this review as these ecosystem services are already captured in land-use statistics at local and national level (Statistisches Bundesamt 2017).

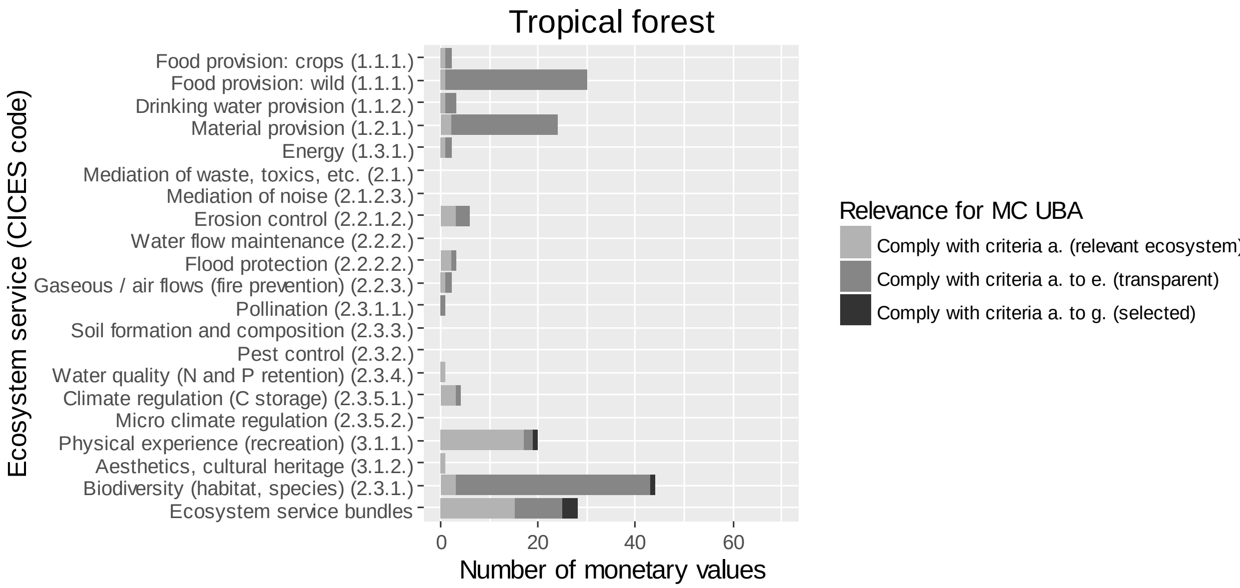


Figure 2.7: Number of monetary values for ecosystem services in tropical forests (classified according to CICES). The class “Biodiversity (habitat, species) (2.3.1.)” is valued most frequently and includes the appreciation of people for ecosystems to provide habitat for species and diversity of ecosystems across landscapes. This is followed by the distinct ecosystem service classes “Food provision: wild (1.1.1.)”, “Material provision (1.2.1.) and “Physical experience (recreation) (3.1.1.)”. ”Ecosystem service bundles” with multiple ecosystem service classes are also frequently valued. Colour coding indicates the relevance of values for informing the methodological convention (MC) of the Umweltbundesamt (UBA) on possible costs involved in the loss or degradation of ecosystem services.

2.3.1 Selected ecosystem service valuation studies

Based on the selection criteria (a. to g.), six ecosystem service valuation studies for Germany and one meta-analysis of valuation studies for tropical forests were selected for informing the methodological convention (MC) of the Umweltbundesamt (UBA) on potential costs in terms of ecosystem service loss (Table 2.1).

Table 2.1: Studies with monetary values of ecosystem services complying with criteria for informing national policies (criteria are defined in section 2.2.4.).

Reference	Publication type	Focus of valuation	Ecosystem service (CICES code)	Related conversion process	Minimum ranges of monetary values (inflation-adjusted with 2014 as base year)	Comment
Born et al. 2012	Project report	Benefit of wetlands for nutrient retention. Replacement costs for alternative approaches for removing nitrate N.	Water quality (retention of nitrate N & phosphate P) (2.3.4.)	III. Wetland conversion. Benefit of wetland for removing 1 kg N: Benefit of one hectare wetland for N retention:	6.16€/kg N 663.27 – 809.07 €/ha	
Born et al. 2012	Project report	Benefit of wetlands for nutrient retention. Replacement costs for alternative approaches for removing phosphate P.	Water quality (retention of nitrate N & phosphate P) (2.3.4.)	III. Wetland conversion. Benefit of wetland for removing 1 kg P: Benefit of one hectare wetland for P retention:	61.60€/kg P 159.14€/ha	
Grossmann 2012	Peer-reviewed publication	Benefit of each additional hectare inundated/ restored wetland for N and P retention.	Water quality (retention of nitrate N & phosphate P) (2.3.4.)	III. Wetland conversion. Restoring riparian wetland area for achieving reduction in N and P load by: Scenario 1 (S1): 5% Scenario 2 (S2): 15% Scenario 3 (S3): 25% Scenario 4 (S4): 35%	Valuation of benefit each additional hectare of inundated/restored wetland has for N and P retention: S1: 1,636.39 - 1,834.13 €/ha S2: 1,2664.67 - 1,3059.08 €/ha S3: 21,172.63 - 25,027.94 €/ha S4: 43,188.58 - 56,556.55 €/ha	
Horbat et al. (unpubl.)	Project report	Valuation of benefit of wetland restoration for N retention.	Water quality (retention of nitrate N) (2.3.4.)	III. Wetland conversion. Scenario 1 (S1): Restoring riparian wetland area from 4748 ha to 6426 ha. Scenario 2 (S2): Restoring riparian wetland area from 4748 ha to 8494 ha.	S1: 649.51 €/ha/year S2: 233.22 €/ha/year	Peer-reviewed publication of data is recommended.

Table 2.1 (continued)

Horbat et al. (unpubl.)	Project report	Valuation of benefit of wetland restoration for P retention.	Water quality (retention of phosphate P) (2.3.4.)	III. Wetland conversion. Scenario 1: Restoring riparian wetland area from 4748 ha to 6426 ha. Scenario 2: Restoring riparian wetland area from 4748 ha to 8494 ha.	S1: 615.72 €/ha/year S2: 229.11 €/ha/year	Peer-reviewed publication of data is recommended.
Ott et al. (2006)	Project report	Cost of habitat restoration	Biodiversity (habitat, species) (2.3.1.)	II.) Restoration of sealed surfaces.	9,273.91 - 9,4265.21 €/ha (net present value)	Requires update of underlying assumptions.
Reutter & Matzdorf (2013)	Book chapter	Monetary valuation of nitrate (N) retention and leakage to freshwater as result of changes in intensity of grassland use.	Water quality (retention of nitrate N) (2.3.4.)	I.) Grassland conversion. Scenario 1: low intense use of grassland to high intense use of grassland (increase of N emissions: 20 kg N/ha/year) Scenario 2: low intense use of grassland to arable land (increase of N emissions: 70 kg N/ha/year)	S1: 10.92 €/ha/year S2: 65.63 €/ha/year	Underlying assumptions of monetary values for N and P retention could be updated using 6 € per kg N and 60 € per kg P.
Schweppe-Kraft (unpdate based on Schweppe-Kraft 1998)	Report from 1998 updated in 2016 (unpublished update)	Cost of habitat restoration: grasslands	Biodiversity (habitat, species) (2.3.1.)	I.) Grassland conversion. Restoration of grasslands of different habitat quality.	31,811.17 - 91,457.11 €/ha (net present value)	Based on habitat-valuation-point system. Monetary value per habitat-point is based on Schweppe-Kraft (1998). Requires updating.
Schweppe-Kraft (unpdate based on Schweppe-Kraft 1998)	Report from 1998 updated in 2016 (unpublished update)	Cost of habitat restoration: forests	Biodiversity (habitat, species) (2.3.1.)	II.) Forest restoration. Restoration of forests of different habitat quality.	43,740.35 - 91,457.11 €/ha (net present value)	Requires update (see above).
Schweppe-Kraft (unpdate based on Schweppe-Kraft 1998)	Report from 1998 updated in 2016 (unpublished update)	Cost of habitat restoration: wetlands	Biodiversity (habitat, species) (2.3.1.)	III. Wetland conversion. Restoration of wetlands of different habitat quality.	67,598.73 - 95,433.50 €/ha (net present value)	Requires update (see above).

Table 2.1 (continued)

Ojea et al. (2016)	Peer-reviewed publication	Benefit from tropical forests	Physical experience (recreation) (3.1.1.)	IV. Tropical forest (no conversion).	682.91 €/ha/a	Based on meta-analysis of multiple valuation studies
Ojea et al. (2016)	Peer-reviewed publication	Benefit from tropical forests	Biodiversity (habitat, species) (2.3.1.)	IV. Tropical forest (no conversion).	3960.74 €/ha/a	Based on meta-analysis of multiple valuation studies
Ojea et al. (2016)	Peer-reviewed publication	Benefit from tropical forests	Ecosystem service bundle: air quality and water regulation (excluding carbon)	IV. Tropical forest (no conversion).	5287.27 €/ha/a	Based on meta-analysis of multiple valuation studies
Ojea et al. (2016)	Peer-reviewed publication	Benefit from tropical forests	Ecosystem service bundle: "food and fibre"	IV. Tropical forest (no conversion).	4267.11 €/ha/a	Based on meta-analysis of multiple valuation studies

As example of how a standard cost estimate for an ecosystem service can be used to inform on economic costs involved in land-cover conversions, the standard cost estimate of 80€/tCO_2, which is currently used by the Umweltbundesamt (UBA) for estimating damage costs of carbon emissions, is applied to the carbon balance of land-cover change reported by the German Government under the Kyoto Protocol (Supplementary material S2.4; Umweltbundesamt 2014).

2.4 Discussion

In Germany, impact assessments of new policy proposals increasingly rely on monetary cost-benefit analysis, which often do not consider costs of ecosystem service loss. This study provides a first systematic and comprehensive review of monetary valuation studies of ecosystem services for common ecosystems and land-cover conversion processes in Germany. In addition, this review includes information on the potential costs of ecosystem service loss caused by tropical deforestation, which is relevant for accounting for the costs of ecosystem service loss due to imports of agriculture and forest commodities from tropical forest regions (Kissinger et al. 2012, Schmitz et al. 2015, Liu et al. 2016). As such, this literature review and the developed database serve as a reference for informing on potential costs and benefits involved in land-cover change in terms of ecosystem services loss in Germany and tropical forest regions.

2.4.1 Key findings

Gaps in knowledge on the economic dimension of the benefits biodiversity and ecosystems contribute to human wellbeing in Germany were identified. Considering that provisioning services, including agricultural production and forestry, are accounted for in detail in local and national statistics (Statistisches Bundesamt 2017), the identified 109 studies with monetary values for regulating and cultural ecosystem services in Germany since the 1980s are strikingly small in number. This confirms concerns that current decision making processes are distorted, giving preference to maximizing provisioning services in agricultural production and forestry, while neglecting the relevance of regulating and cultural services for society (Bateman et al. 2013).

Furthermore, only 6 out of 109 ecosystem service valuation studies (5.5 %) were found to comply with all selection criteria (a. to g.) for informing policy impact assessments targeted by the methodological convention (Methodenkonvention) of the German Federal Environment Agency Umweltbundesamt (UBA) (Fig. 2.6 and Table 2.1). This highlights the need for monetary valuation studies to be more policy relevant by: i) being more transparent and robust with regards to study design and valuation methods, by ii) assessing and reporting information on ecosystem services in common and comparable units (e.g. providing information on biophysical and socio-economic indicators in values per hectare and/or per capita) and, if possible, by iii) explaining minimum-maximum ranges in monetary values by measurable changes in biophysical or socio-economic indicators (Fig. 2.6). This would enhance the interpretation of the reported monetary values on ecosystem services in light of the original valuation studies and allow for judging their suitability, credibility, and reliability for informing decisions on policy design.

The majority of valuation studies focus only on a few ecosystems and ecosystem services (Fig. 2.6), revealing blind spots in the literature on ecosystem service valuation. Forests and wetlands have received greatest attention in ecosystem service valuation in Germany (Fig. 2.2) with a focus on habitat provision for biodiversity, recreation, and nutrient (N and P) retention for freshwater quality (Fig. 2.6). Other regulating and cultural ecosystem services are less frequently assessed, including pollination, soil formation, erosion control and pest control. Potential explanations for the focus of valuation studies on forests and wetlands include that there is a long history of research on these ecosystems in Germany. Biodiversity, recreation and water quality are also topics of public interest and therefore such research is more likely to be supported by donors and decision makers, while other ecosystem services are less visible and recognized.

One of the gaps includes the lack of literature on monetary valuation of ecosystem services of grasslands. While grasslands are heavily affected by land-cover conversion in Germany (Tietz et al. 2012, Niedertscheider et al. 2014), only 14 studies were found to address the monetary value of ecosystem services of grasslands (Fig. 2.2). Reutter & Matzdorf (2013) estimate that the intensification of grassland use increases nitrate (N) emissions by 20 kg per hectare and year, while the conversion of grasslands to arable land increases nitrate emissions by 70 kg per hectare and year, causing monetary costs of about

10.92 - 65.63 $€_{2014}$ per ha per year (Table 2.1). These are costs society has been bearing as a result of grassland loss throughout the past and these costs continue to occur today.

The use of fertilizers on agricultural land and the lack of natural ecosystems buffering nitrate from reaching freshwater systems is a major cause for the continuous increase in nitrate concentrations and it is expected that the resulting increase in efforts for purifying drinking water from nitrate will increase the costs for water users by 32 to 45 % (Umweltbundesamt 2017; Oelmann et al. 2017). Currently, the European Commission has taken legal steps against the German government due to continuously high nitrate concentrations in water bodies in Germany, which exceed the thresholds of the European Union Nitrate Directive (European Commission 2016). Due to the lack of effective measures and policies for reducing nitrate concentrations, the German government is facing the payment of significant fines.

2.4.2 Challenges and limitations of using the identified monetary values

Monetary values for nutrient retention allow for generic conclusions on the benefits ecosystems provide in terms of capturing nutrients (nitrate N and phosphate P) (e.g. Grossmann 2012, Born et al. 2012, Horbat et al. (unpubl.), Table 2.1). Currently, standard estimates for replacement costs for nutrient retention are given at 6.16 € per kg N and 60.60 $€_{2014}$ per kg P (e.g. Born et al. 2012, Table 2.1). However, it is important to note that these estimates are only a partial reflection of the true costs that incur to society when nutrients enter freshwater systems. These cost estimates are based on the replacement cost method by determining the monetary value of nutrient retention provided by ecosystems based on assumptions for costs that would incur, if nutrient loads in the water were to be reduced using technical measures. However, this is only a partial representation of costs, as the replacement costs do not include damage costs caused by excess of nutrients in freshwater systems causing species loss, impacts on human health and decline in aesthetic and recreational values due to deterioration of water quality.

Given these shortcomings of the replacement cost method, it is recommended to use the damage cost method for estimating the costs of ecosystem service loss. This recommendation is in line with already existing guidance by the UBA methodological convention on estimating the costs of carbon emissions based on damages caused by climate change impacts. Using the damage-cost approach, the costs of carbon emissions are currently estimated to be at 80 €/tCO_2 (Umweltbundesamt 2013). This estimate for the social cost of carbon emissions has become an established reference in Germany. For example, it is used for determining the cost of wetland degradation in terms of carbon emissions and for estimating the benefits of restoring wetlands for mitigating carbon emissions (Schäfer 2009). Applying this cost estimate to the biophysical carbon values used in the national reporting under the Kyoto Protocol (Umweltbundesamt 2014) allows for a rough estimation of the costs caused by carbon emissions from land-cover change in Germany (Supplementary material S2.4). Using similar standardized biophysical indicators for other ecosystems and ecosystem services, for example for nutrient retention (N and P) by wetlands, could allow for an equally rough estimation of the benefits ecosystems provide for nutrient retention at national scale.

For some of the selected studies an update of the monetary estimates is recommended, using more recent information on the monetary benefits of biodiversity and ecosystem services. For example, Schweppe-Kraft (1998) provides restoration costs for a diversity of habitats based on the habitat-valuation-point system (Biotopwertpunkte). This habitat-valuation-point system is used for assessing the ecological quality of habitats and is well-established in land-use planning in Germany. It is applied, for example, in environmental impact assessments (EIA) for informing decision making on options for conserving, mitigating, restoring and offsetting environmental impacts. Although the monetary values presented by Schweppe-Kraft (Table 2.1) take into account a recent update of the habitat-valuation-point system, the underlying economic model used for determining the monetary value of a single habitat-valuation point is based on an outdated value of the willingness to pay for biodiversity conservation from the 1990s (Schweppe-Kraft 1998). As the socio-economic context of the 1990s, when the original study was conducted, is very different from today's context (e.g. due to changes in income, unemployment rates, demography, etc.), the use of monetary values of past valuation studies within today's reality involves large uncertainties (Dittrich et al. 2017). Therefore, an update of the monetary values of Schweppe-Kraft (Table 2.1) is recommended, using more recent estimates for the monetary benefits of biodiversity and ecosystem services. The same applies for studies following a similar methodological approach, such as Ott et al. (2006).

For ecosystem services of tropical forests, the study by Ojea et al. (2016) provides monetary values based on a review of existing valuation studies and a meta-analysis using linear regression analysis. While such an analysis can provide an important contribution to establishing more general estimates for the monetary value of ecosystem services, one has to be aware that the context of the original valuation studies is lost in the process of the meta-analysis. The original valuation studies have been designed for addressing a particular research or policy question within a specific biophysical and socio-economic context and at a specific spatial scale (e.g. local or national). This information is not contained in the aggregated values. Therefore, it is critical to consult the original valuation studies when interpreting and using aggregated values for informing decision making.

As shown with the examples above, one has to be aware that monetary estimates for ecosystem services are only "snapshots" of a few selected costs or benefits and economic values of ecosystem services account only for a subset of benefits biodiversity and ecosystem services provide to human wellbeing (Spangenberg & Settele 2016). In addition, monetary valuation of non-market goods – a characteristic that applies to most regulating and cultural ecosystem services – involves methodological and conceptual challenges, as the loss of multiple values, in particular cultural and intrinsic values, is not being represented in monetary estimates (Spangenberg & Settele 2016).

Each ecosystem provides bundles of multiple ecosystem services with a great diversity of values and benefits (Raudsepp-Hearne et al. 2010). However, ecosystem service assessments often focus on a few selected ecosystem services, neglecting the benefits of multiple ecosystem services. Valuation methods can also address only certain aspects of benefits, with multiple values not being accountable in monetary terms. Hence, monetary values of single ecosystem services should be interpreted as minimum values. The total cost of ecosystem loss is likely to be larger.

Therefore, relying exclusively on monetary values is a narrow approach to decision making, which can lead to outcomes in favour of or against biodiversity conservation (Adams 2014). Economic valuation is rather an illustration of potential economic costs involved in decision options that should be complemented also by other methods and indicators that allow for integration of multiple values of biodiversity and ecosystem services in decision making (Spangenberg & Settele 2016). Furthermore, decision makers do not want to rely only on economic information and demand also other types of information (Ruckelshaus et al. 2015). Taking into account the multiple values of biodiversity and ecosystems, including their intrinsic and cultural values, is essential for an inclusive decision making process (Jacobs et al. 2016).

2.4.3 Recommendations on the use of monetary values for ecosystem services

The monetary values recorded in the database and the selected values presented in Table 2.1 can provide a first indication on the benefits regulating and cultural ecosystem services provide to human wellbeing in Germany. However, it is important to be aware of the methodological challenges of monetary valuation of ecosystem services in order to judge the credibility and suitability of monetary values for informing decision making. Monetary valuation is done on a case-by-case basis (Helm & Hepburn 2012) and the judgment of the credibility and suitability of monetary values for informing decision making should follow a case-by-case approach. For using monetary values of ecosystem services in decision making, it should be demonstrated that the monetary value fits the context and purpose of the particular situation of a decision (e.g. see Johnston et al. (2015) for guidance on benefit transfer). Due to the diversity in the biophysical and socio-economic contexts of study sites and the diversity in valuation methods (Fig. 2.5), such a benefit transfer requires a thorough review of the primary valuation studies with regards to their suitability for informing a particular decision context. Furthermore, as ecosystem service valuation studies use a great diversity of methods (Fig. 2.5), it is not advised to aggregate monetary values across different valuation studies. Instead, ranges of minimum and maximum values should be used for ecosystem services in order to reflect the diversity in valuation methods and socio-ecological contexts.

Given the outlined limitations in the monetary values available for ecosystem services in Germany, decision making has to account for the trade-off in relying on few cost estimates for ecosystem services that are scientifically robust, while being pragmatic enough to include also vague estimates from studies that may not comply with the defined selection criteria (a. – g.). This review and the generated database can serve as tool for identifying ecosystem service valuation studies that are relevant for informing decision-making processes. However, the database should not be used as a one-stop-shop for an arbitrary use of monetary values. The database can guide the identification of monetary valuation studies of ecosystem services relevant for informing decision making, but it does not replace a careful assessment of the original valuation studies for informing a particular decision context.

Finally, it has been shown for spatial planning that decision makers prefer a mix of multiple indicators that allow weighing decision options for different criteria within a specific decision context (Albert et al. 2014). Multi-dimensional frameworks as, for example, multi-criteria decision analysis (MCDA), allow for the inclusion of quantitative and qualitative information on multiple values of biodiversity and ecosystems. This can open up discourses in decision-making processes and help reflect the views and values of multiple stakeholders (TEEB 2012; Saarikoski, Mustajoki, et al. 2013; Förster et al. 2015; Lienhoop et al. 2015; Spangenberg & Settele 2016). Therefore, the use of monetary values of ecosystem services should be accompanied also by information on other indicators in order to allow for inclusive decision-making processes that take into account the multiple values of biodiversity and ecosystem services.

2.4.4 Conclusions

This review highlights significant gaps in knowledge on the monetary value of ecosystem services in Germany and a lack of studies relevant for informing decision making. Therefore, it is recommended that future ecosystem service valuation studies should better target the specific information needs of decision makers in order to provide information on ecosystem service indicators that are relevant for informing decision making at local and national level. While using monetary values on ecosystem services can open up the debate on the relevance of biodiversity and ecosystem services for society and inform the design of policies including cost-benefit analysis, decision making should not only rely on single monetary values for ecosystem services. This would bear the risk of underestimating the benefits of ecosystem services and costs involved in ecosystem loss. Given that biodiversity and ecosystem services provide multiple values (Díaz et al. 2015), it should be recognized that monetary valuation is only one approach of many for assessing the importance of nature for human well-being (Spangenberg & Settele 2016).

3 Assessing ecosystem services for informing land-use decisions: a problem-oriented approach

Summary

Assessments of ecosystem services that aim at informing decisions on land management are increasing in number around the globe. Despite selected success stories, evidence for ecosystem service information being used in decision making is weak, partly because ecosystem service assessments are found to fall short in targeting information needs by decision makers. To improve their applicability in practice, we compared existing concepts of ecosystem service assessments with focus on informing land-use decisions and identified opportunities for enhancing the relevance of ecosystem service assessments for decision making. In a process of co-design, building on experience of four projects in Brazil, China, Madagascar, and Vietnam, we developed a step-wise approach for better targeting ecosystem service assessments toward information needs in land-use decisions. Our problem-oriented approach aims at (1) structuring ecosystem service information according to land-use problems identified by stakeholders, (2) targeting context-specific ecosystem service information needs by decision makers, and (3) assessing relevant management options. We demonstrate how our approach contributes to making ecosystem service assessments more policy relevant and enhances the application of ecosystem service assessments as a tool for decision support.

3.1 Introduction

Assessments of ecosystem services are increasing in number (Seppelt et al. 2011; Abson et al. 2014), but it is questioned whether they actually generate knowledge that is relevant for decision makers (Honey-Rosés and Pendleton 2013, Laurans et al. 2013, Martinez-Harms et al. 2015). The majority of ecosystem service assessments tend to generate knowledge on ecological functions and economic values (Abson et al. 2014) with little consideration of the information demand by decision makers for addressing a particular land-use problem (Honey-Rosés & Pendleton 2013). For example, only 8 out of 340 cases of ecosystem service valuation published in scientific literature actually report how information on the value of ecosystem services is used in local decision making (Laurans et al. 2013). Ecosystem service assessments have not yet proven to effectively change land management and policies in public and private sectors (Ruckelshaus et al. 2013, Abson et al. 2014).

Nonetheless, ecosystem service assessments can be an attractive tool for supporting decisions on land use, as they can highlight benefits and trade-offs between different land-use options, ideally by integrating biophysical and socio-economic methods (Daily et al. 2009, Fisher et al. 2009, TEEB 2010, Ruckelshaus et al. 2013). Therefore, ecosystem service assessments are increasingly used in decision-oriented processes, including Environmental Impact Assessments (EIA) (e.g. Pischke and Cashmore 2006) and land-use planning for biodiversity conservation (Goldman et al. 2008) and catchment management (e.g. Ruckelshaus et al. 2013). The ecosystem service concept is also popular in national and international policy processes, including national ecosystem assessments, the Aichi Biodiversity Targets of the United Nations Convention on Biological Diversity (CBD), the Work Plan of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), and the Biodiversity Strategy of the European Union.

The term ‘ecosystem services’ describes benefits that ecosystems - comprised of species, genes, biotic and abiotic structures and processes – provide to human well-being (Millennium Ecosystem Assessment 2005, Fisher et al. 2009). Harnessing and managing ecosystem services often requires knowledge on the potential of ecosystems to provide ecosystem services and takes the investment of skills, labour, materials, and energy (Spangenberg et al. 2014a). The cultural and political context influences, which ecosystem services are appropriated and how. Land use is then the result of this complex human-ecosystem interaction, which is described as social-ecological system (SES) (Ostrom 2007). Components or processes of ecosystems only become ecosystem services, if someone actively or passively benefits from them (Jax et al. 2013). Hence, the definition of ecosystem services involves subjective judgments of what is perceived as benefit, making ecosystem services a normative concept (Jax et al. 2013; Schröter et al. 2014). Using a broad interpretation, in which ecosystem service benefits are based on multiple values, the ecosystem service concept can be valuable for decision support: it allows assessing human dependence on ecosystems through inter- and transdisciplinary research, integrating perspectives and values of different stakeholder groups, and guiding decisions on resource use (Reyers et al. 2010; Jax et al. 2013; Abson & Hanspach 2014; Schröter et al. 2014). A narrow interpretation, in which ecosystem service benefits are only based on monetary values, evokes criticism of the ecosystem service concept for being anthropocentric, fostering a utilitarian and economic perspective, with the risk of promoting commodification and exploitation of nature (Turnhout et al. 2013; Schröter et al. 2014). Due to this normative character, there is no standard interpretation and application of the ecosystem service concept, but it is clear that it requires transparency about its context, purpose, and definitions (Jax et al. 2013).

Since 1997 the number of scientific publications addressing ecosystem services has increased 27-fold, particularly in the natural-science literature (Abson et al. 2014). Biophysical characteristics of ecosystem services (e.g. Egoh et al. 2009), their cultural and social significance (e.g. Chan et al. 2012), and economic value (e. g. Christie et al. 2012) are assessed and integrated into models (e.g. Nelson et al. 2009) and maps (e.g. Crossman et al. 2013) that describe interdependencies and trade-offs between land-use options. However, interdisciplinary ecosystem service assessments remain the exception with only 8.5% of ecosystem service studies being truly interdisciplinary (Abson et al. 2014).

Integrating a social-ecological system (SES) perspective into ecosystem service assessments, with land use being viewed as a system of interlinked natural and socio-political processes, offers a way of making such assessments more relevant to decision making (Spangenberg et al. 2014a). A SES perspective within ecosystem service assessments allows (i) the analysis of how human demand constitutes potential services (Spangenberg et al. 2014b), (ii) the identification of dependencies of ecosystem service users on ecosystems, and (iii) an understanding of trade-offs among management options (Cowling et al. 2008; Seppelt et al. 2011; Carpenter et al. 2012).

Guidance exists on integrating a SES perspective into ecosystem service assessments (e.g. Reyers et al. 2013), accounting for cultural and social values (Chan et al. 2012a, Chan et al. 2012b), using ecosystem service information in landscape planning and management (de Groot et al. 2010), and mainstreaming ecosystem services into policies and practice (Cowling et al. 2008; Daily et al. 2009). However, the attempt to account for all social-ecological factors can make ecosystem service assessments a complex and resource-intense endeavour (Cowling et al. 2008, Chan et al. 2012a). Experience from practice shows that complex assessments are not necessarily more helpful for decision support (Ruckelshaus et al. 2015). Decision makers do not necessarily need an exhaustive understanding of the social-ecological system, but they need sufficient arguments to make a choice between land-use options. Therefore, designing problem-oriented ecosystem service assessments, which focus on the information demand by decision makers, can help making ecosystem service assessments more decision relevant (Honey-Rosés & Pendleton 2013).

To address this challenge, we compared existing frameworks for assessing ecosystem services in social-ecological systems. We identified prevailing gaps in these approaches and, based on the experience from four case studies in Brazil, China, Madagascar and Vietnam, we co-designed and tested a problem-oriented ecosystem service assessment approach that prioritizes information demand by decision makers. We discuss how our approach contributes toward making ecosystem service assessments a more relevant tool for decision making. The case studies are part of the Sustainable Land Management (SLM) Program, funded by the German Federal Ministry for Education and Research (BMBF), with the objective of fostering transformations toward more sustainable land stewardship (Eppink et al. 2012). It is part of the Program on Ecosystem Change and Society (PECS) (Carpenter et al. 2012).

3.2 Building on field experience

Building on the experience of four place-based projects (Fig. 3.1), we collaboratively identified aspects which are critical for a problem-oriented ecosystem service assessments, using workshops and expert consultations. The four case studies use ecosystem service assessments to guide decisions on land-use problems related to agriculture, water use, and ecosystem conservation at local to regional scales.

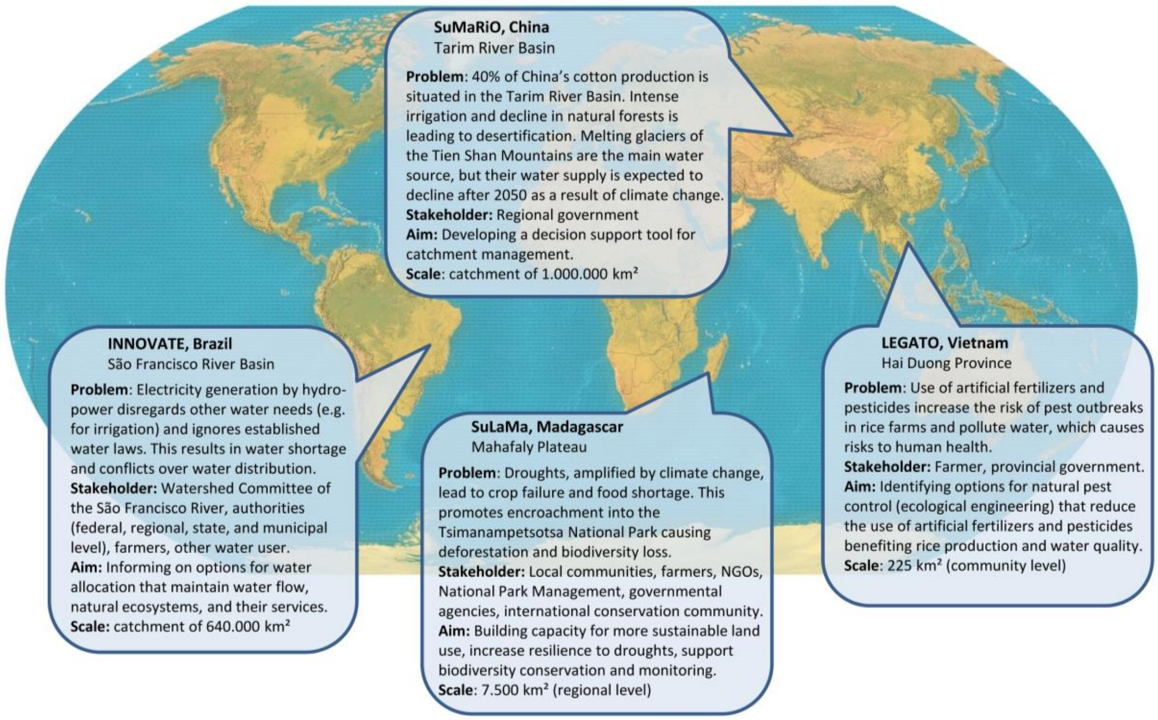


Figure 3.1: Case studies of the Sustainable Land Management Program for which the problem-oriented approach was developed and exemplified. Videos summarizing each case study can be accessed at the Program's website. (URL: http://modul-a.nachhaltiges-landmanagement.de/de/mediathek-modul-a/videobeitraege/).

In Madagascar, the SuLaMa project identifies options for enhancing the resilience of local communities to shortages in food and water supply caused by climate variability, and for mitigating encroachment into a protected area (Fig. 3.3). The LEGATO project in Vietnam analyses rice farming practices that enhance natural pest control, increase yields, and reduce the use of pesticides causing water pollution (Settele et al. 2013) (Fig. 3.4). In the São Francisco River watershed in Brazil, the INNOVATE project analyses ecosystem services to support the Watershed Committee in addressing conflicts over water use for irrigation agriculture, electricity generation from hydropower, and domestic water use, while maintaining sufficient water flow for river ecosystems (Siegmund-Schultze et al. 2015) (Fig. 3.5). In the Tarim River Basin in China, the SuMaRiO project informs the regional government on benefits and trade-offs involved in water use for cotton irrigation and the conservation of riparian forests, considering threats related to desertification and climate change (Rumbaur et al. 2015) (Fig. 3.6).

We compare our approach with eight existing frameworks (Fig. 3.2) that focus on assessing ecosystem services within social-ecological systems (SES) with the aim of providing decision support (Cowling et al. 2008; Carpenter et al. 2009; Daily et al. 2009; Ostrom 2009; Chan et al. 2012a, TEEB 2012; Reyers et al. 2013; Martinez-Harms et al. 2015)

Only three out of eight frameworks provide explicit guidance for focusing ecosystem service assessments on decision relevant problems. The TEEB approach (TEEB 2012) and Chan et al. (2012a) require to (1) agree on the problem, in order to (2) prioritize ecosystem services according to their relevance to the problem and stakeholders, and to (3) identify information needs by decision makers. However, the TEEB approach remains vague in how to assess ecosystem services from a SES perspective and Chan et al. (2012a) target mainly cultural values Martinez-Harms et al. (2015) emphasize the importance of a stakeholder-driven problem identification and specification of objectives at the beginning of the assessment process, but they note that only 8% of case studies actually use stakeholder consultations in this process. Nevertheless, they provide little guidance on how to target problems and objectives relevant to decision makers. The other five approaches acknowledge the need to account for concerns of stakeholders, but the gaps under 'Scoping phase A' (Steps 1-3 on the left side of Fig. 3.2) depict the lack of explicit guidance on tailoring ecosystem service assessments to decision needs.

All approaches assume that developing an understanding of the social-ecological context and analysing the flow of ecosystem services, their benefits, and trade-offs (Assessment phase B, Fig. 3.2) will generate information relevant to decision making (Implementation phase C, Fig. 3.2). This can be achieved, for example, through assessing the governance and resource system (Ostrom 2009), undertaking social and biophysical assessments (Cowling et al. 2008), analysing the link between governance context and ecosystem service (Carpenter et al. 2009), and establishing social-ecological production functions (Reyers et al. 2013). However, trade-off analysis alone is not leading to changes in decision making (Daily et al. 2009). Focusing on the importance of ecosystem service information for decision making only after it has been generated involves the risk of missing decision relevant information. Furthermore, judging the relevance of information by scientific criteria can lead to advice that is lacking a policy perspective. It is recognized that, besides improving the science, a better integration of ecosystem service information in the development of policies and institutions is needed (Daily et al. 2009).

We propose closing these gaps by better tailoring ecosystem service assessments to problems at the very beginning of the assessment process and targeting specific information needs of decision makers. Building on the experience of the four case studies (Fig. 3.1), we developed a problem-oriented ecosystem service assessment approach to provide practical guidance for the assessment and synthesis of ecosystem service information with a focus on informing land-use decisions (Fig. 3.2). Our approach comprises a scoping phase (A), assessment phase (B), and implementation phase (C), and follows 5 steps: (Step 1) specify and agree with stakeholders on the problems to be addressed, (Step 2) identify ecosystem service beneficiaries and ES most relevant to decision making, (Step 3) define information needs of decision makers, (Step 4) assess ecosystem service flow within the SES context and impact of changes on ecosystem service benefits and trade-offs, and finally (Step 5) synthetize and integrate the generated information into processes of decision support. The approach is not intended to replace the existing frameworks, but to provide complementary guidance for designing and implementing ecosystem service assessments that are more relevant for decision making.

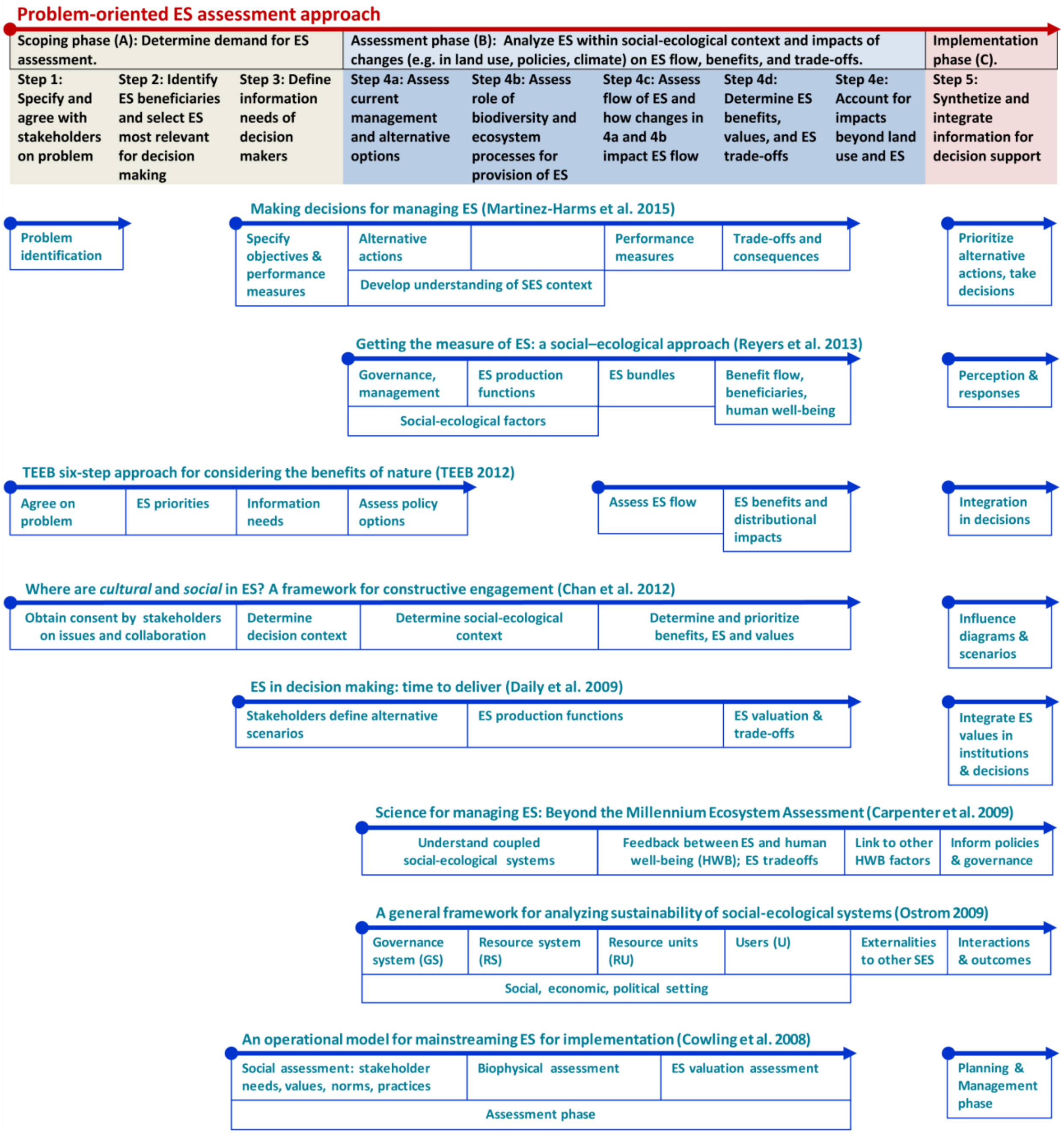


Figure 3.2: The problem-oriented approach for assessing ecosystem services (ES) (at top) of the Sustainable Land Management (SLM) Program compared with other approaches for assessing ecosystem services using a social-ecological systems (SES) perspective. Steps can be applied sequentially (arrows), interchangeably, and repetitively within iterative assessment procedures.

3.3 Application

In the following the problem-oriented approach of the SLM Program is exemplified along the four case studies (Fig. 3.3 to 3.6). The approach is not a static, prescriptive blueprint for a linear assessment process. Each ecosystem service assessment is a unique undertaking, adapted to a specific decision within a social-ecological system and point in time, producing context specific outcomes. Hence, designing and implementing ecosystem service assessments, aiming at more sustainable land management options, requires trans-disciplinary expertise that accommodates different types of knowledge and allows responding to context specific information needs (Görg et al. 2014). Ideally, ecosystem service assessments are embedded in a science-practice partnership that enables co-generation of knowledge, which is both user-inspired and user-relevant (Ntshotsho et al. 2015).

The presented approach is flexible in that the sequence of steps can be altered and the thematic and methodological focus can be adapted to stakeholder needs. Applied in an iterative process, information generated in one step can inform previous and consecutive steps in feedback loops. The normative character of the ecosystem service concept helps to take into account different cultural and socio-economic contexts and decision-making processes (Schröter et al. 2014) and to integrate multiple types of knowledge (e.g. combining traditional and scientific information). Integrative tools, which combine methods of natural and social science and synthesize qualitative and quantitative information (e.g. multi-criteria analysis, tools for spatial analysis, and social-ecological models), are increasingly applied for ecosystem service assessments (e. g. Bagstad et al. 2013).

In the following sections each step is explained in more detail and an overview of questions, methods, and indicators is provided (see Tables 3.1 to 3.3).

3.3.1 Scoping phase (A)

Step 1: Specify and agree with stakeholders on problem.

Land use related problems, drivers, and impacts are identified in step 1 through consultations of experts and stakeholders, review of literature, and available data (Table 3.1). As stakeholders are not a homogenous group (e.g. politicians and farmers are both decision makers), consensus on often multilayered problems cannot be taken for granted. For example, in the case of competition for scarce resources, ecosystem service information can empower one party over others, leading to inequalities and potential conflicts. Thus, analysing the distribution of benefits and disbenefits and the impacts on power relations is an important starting point for determining the focus and scales of the assessment.

For example, stakeholder interviews and constellation analysis (e.g. Bruns et al. 2011) helped INNOVATE in Brazil and SuMaRiO in China to identify large-scale water allocation issues at a catchment scale (area of 640.000 km^2 and 1 million km^2 respectively) (step 1 in Fig. 3.5 and 3.6). In these catchments, water use involves trade-offs between irrigation, hydropower production, and maintaining minimum ecological flow for sustaining natural ecosystems that provide habitat for biodiversity and mitigate desertification (e.g. Siew et al. 2014). In contrast, the projects SuLaMa in Madagascar and LEGATO in Vietnam target farmers who make decisions on crop and livestock production ranging from a few hectares up to regional scales within mosaic landscapes (areas of 7500 km^2 and 225 km^2). SuLaMa and LEGATO aim at enhancing resilience of agricultural production against droughts and pest outbreaks in order to increase food security and household income, while ensuring biodiversity conservation (step 1 in Fig. 3.3 and 3.4).

To ensure a focus on ‘real-life’ problems, LEGATO followed an approach of co-design and co-production. Using stakeholder dialogues, relevant partners including local decision makers, farmers, researchers, and research institutions were consulted to identify research needs and elucidate synergies in capacities, knowledge, and skills. This process also ensured political acceptance and support of the project by all partners, taking into account institutional settings, involving different levels of local and regional governance, and respecting power structures.

Step 2: Identify ecosystem service beneficiaries and select ecosystem service most relevant for decision making.

Step 2 covers prioritization of ecosystem service according to their relevance to the identified problem, affected stakeholders, and the decision to be informed (Chan et al. 2012a, TEEB 2012) (Table 3.1). Special attention should be given to diverging interests and the distribution of benefits and costs. To do so, it is critical to integrate a range of knowledge sources of multiple stakeholder groups, including farmers, indigenous peoples, decision makers in public administration and private businesses, but also researchers and experts with particular knowledge of the system. The focus on prioritized ecosystem services has the advantage of targeting ecosystem service assessments toward specific land-use problems, taking into account available capacities and resources. However, as many ecosystem services are co-produced in bundles with benefits and costs to different stakeholders, the analysis must not be limited to single ecosystem services, monetary benefits, or selected stakeholders, which would ignore ecological context and distributional effects.

Table 3.1: Examples of questions, actions, and indicators for determining the demand for an ecosystem services (ES) assessment - Scoping phase (A).

Scoping phase (A): Determine the demand for the ES assessment		
Questions	**Actions**	**Indicators (qualitative & quantitative)**
Step 1: Specify and agree with stakeholders on problem		
Who are stakeholder groups and which problems are they concerned about? Are these problems caused by or linked to land use? Which socio-economic or ecological drivers influence the problem? What are the spatiotemporal scales of the problem and who is affected? Are problems related to policies?	Consulting stakeholders, decision makers, and experts using participatory approaches, e.g. interviews, group consultations, surveys, and multi-criteria-analysis (e.g. Saarikoski et al. 2013) Exploring available data and statistics on environmental and socio-economic variables. Literature analysis.	Issues addressed in meetings and interviews with stakeholder groups, decision makers and experts; Status and trends of environmental variables (e.g. water quality, habitat size, yield, climate etc.); Status and trends in socio-economic variables (e.g. income, health, access to resources etc.); Size of affected area and population.
Step 2: Identify ES beneficiaries and select ES most relevant for decision making		
Which stakeholder groups or experts should be involved in ES identification and prioritization? Which ecosystems and ES are related to the problem? Which ES benefits are of particular importance to stakeholders? Are they part of a co-produced ES bundle? Who suffers from disbenefits / trade-offs? Which distributional challenges emerge?	Consulting stakeholders, decision makers and experts on preferences for certain ES bundles and related trade-offs (e.g. Martín-López et al. 2014). Allowing flexibility for accommodating different knowledge types, values, and convictions. Adapting terminology and classification to stakeholder needs, while ensuring compatibility with common ES classification systems (e.g. Fisher et al. 2009, Haines-Young and Potschin 2012).	Types of benefits derived from ES (e.g. consumption, income, etc.), types of disbenefits; Stakeholder groups and number of people benefiting from ES (beneficiaries and ES demand) or suffering disbenefits; Location and area of ecosystems that provide direct and indirect benefits to stakeholder groups (ES supply); Location and area of region that is benefiting from ES provision (ES demand); Importance of ES benefits for wellbeing of stakeholders and related disbenefits.
Step 3: Define information needs of decision makers		
Who is taking decisions on land use? Are stakeholders and decision makers aware of ES benefits and the positive and negative impacts of land-use decisions? Are there decision-making processes or policies for which ES information could be relevant? Would it improve decisions? If so, is there a window of opportunity for using ES information in current or upcoming land-use decisions? On what criteria are land-use decisions based so far (economic benefits, traditional rules, etc.) by which group of decision makers? Does a link to ES exist (irrespective of the terminology used)? When in the decision process is what type of information needed by whom and for which purpose? Which level of detail is required? What are knowledge gaps related to the identified problems and ES? Are they relevant for the decision to be taken?	Using participatory methods (collaborative planning, workshops, consultations) for addressing complex land-use conflicts, involving relevant stakeholders, decision makers, and experts in identifying possible resolutions (e.g. Saarikoski et al. 2013) Analysing potential knowledge gaps, conflicting interests of stakeholder groups, and beneficiaries of ES information (e.g. empowerment of certain groups). Providing lessons learned in comparable decision contexts. For example Garrick et al. (2009) compare how ES information influenced decisions on water management in two basins in the U.S.A. and Australia. Exploring historical data on information used in decision making. For example Wilkinson et al. (2013) compare historical changes in the use of ES information for urban planning in Melbourne and Stockholm.	Stakeholder groups involved in decision making and their respective interests; Stakeholder groups not involved and reasons for exclusion; Awareness of decision makers of identified problem and ES; Decisions or decision processes mentioned by decision makers; Social-ecological variables mentioned by decision makers to be of relevance; Timing of decision processes; Problems, decisions, and variables identified by the research team but not mentioned by decision makers, or only by subgroups.

Figure 3.3: Problem-oriented approach for assessing ecosystem services (ES): case of the SuLaMa project on subsistence farming in Madagascar.

Scoping phase (A): Determine the demand for the ES assessment.			**Assessment phase (B): Analyze ES within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ES flow, benefits, and trade-offs.**					**Implementation phase (C).**
Step 1: Specify and agree with stakeholders on problem and relevant research question	**Step 2: Identify ES beneficiaries and select ES most relevant for decision making**	**Step 3: Define information needs of decision makers**	**Step 4a: Assess current management and alternative options**	**Step 4b: Assess role of biodiversity and ecosystem processes for provision of ES**	**Step 4c: Assess flow of ES and how changes in 4a and 4b impact ES flow**	**Step 4d: Determine ES benefits, values, and ES trade-offs**	**Step 4e: Account for impacts beyond land use and ES**	**Step 5: Synthetize and integrate information for decision support**
Communities suffer from food insecurity and few or no alternative income sources. **Managers of the Tsimanampetsotsa National Park** identified deforestation and forest degradation mainly caused by farmers as causes for biodiversity loss. **Conflicts** between farmers and biodiversity conservation in National Park. **Drivers causing the problem include:** poverty, population growth, crop failure due to weather events including droughts, soil degradation, pests damaging crops, poor infrastructure (storage, market access).	**Farmers benefit from:** **Provision** of food from cultivated crops, livestock, wild alimentary plants, and wildlife, medicinal plants, fodder for livestock, fuelwood and timber from forests. **Regulation** of soil fertility, pest control, water availability. **Cultural values** including livestock as indicator for wealth and status, use of wild plants for ritual purposes. **National Park Management benefits from** maintaining habitat for biodiversity conservation.	Information is needed on **options for diversifying income strategies of farmers** and for building resilience to environmental shocks (droughts) and income loss. **How can the already existing coping strategies be enhanced?** Information is needed on use of drought resistant crops, enhancing soil fertility and pest control, and livestock management, sustainable use of wild plants and wildlife. **National Park Management** requires better monitoring of biodiversity and habitat quality.	Current subsistence **farming practices already include diversified strategies** for crop and livestock production in order to reduce risks related to droughts and pests. Option 1: Explore local crop varieties that are more resistant **to droughts and pests** (e.g. bird resistant millet). Options 2: Test **alternative sources of fodder for livestock** during droughts in order to avoid encroachment in protected area. Option 3: Promote **home gardens** for planting vegetables for domestic consumption and sale on markets. Option 4: Assess role of **wild plants and wildlife for income** diversification.	Option 1: Analyze **resistance** of different crop varieties to **droughts and pests** and test options for enhancing **soil fertility**. Option 2: Analyze **productivity** of Samata, a plant potentially serving as alternative fodder for livestock. Option 3: Analyze **productivity of home gardens.** Option 4: Assess **wild plants and animals used by households.** Monitoring of National Park: analyze species abundance, deforestation and forest degradation rates as **indicators for habitat quality**.	Option 1: **Local corn** is fast growing but little drought resistant. **Local manioc** is resistant against pests, but achieves lower prize on market as it is less tasty. Resistance of **millet** to birds needs further testing. Option 2: Suitability of **Samata as fodder** is tested (biomass and nutrient content). **Water** need for Samata could conflict with water need for food crops. Option 3: **Home gardens** have positive influence on food production for personal consumption and for sale on market. Option 4: **Wild plants and wildlife provide** food in periods of droughts. Medicinal plants are important for health care. **Trade-offs** to be assessed: Competition among farmers for water, land and labor. Changes in spatial patterns of land use and impacts on encroachment into national park.	Option 1: **Farmers** benefit form crop diversification. It can improve food security, income, and increase resilience to droughts. However, best use of local varieties is still unclear. Option 2: **Fodder cultivation (Samata)** has potential to support livestock production. The potential scale of this practice is unclear yet. Option 3: **Home gardens** enhance food security and income for farmers and in particular women. Option 4: **Wild plants and animals** contribute to food security and health. The extent to which these options can reduce encroachment and deforestation in the national park is still being assessed.	**Cultural impacts:** change in livestock production can lead to **social changes,** as the number of livestock determines the social status. Changes in livestock distribution can impact **behavior** of cattle thieves. **Tourism sector** can increase in region with potential impacts on income generation and cultural values.	**Recommendations** to local organizations and decision makers include: **Diversification of crop and fodder production** (option 1 and 2) can be a viable option for enhancing resilience of local communities, since diversification is already part of traditional farming practices. However, the **effect on increasing income is unclear yet. Water remains to be a limiting factor.** Farmers and in particular women benefit from home gardens as a source of food and income. **Home gardens** (option 3) **should be promoted.** The effect of options 1-4 on reducing encroachment into the national park is still unclear.

For example in Vietnam, the involvement of different farmer groups and generations was needed to realize that traditional rice farming practices maintain species compositions that provide natural pest control, while artificial pesticides together with fertilizers cause water pollution and health issues. Thus, better understanding of farming practices that enhance natural pest control and reduce use of pesticides was identified to be the focus of the LEGATO project (step 2, Fig. 3.5). However, institutional issues can also play a role in prioritizing ecosystem services. Due to the relevance of rice farming for local and national economy, LEGATO sought contact to provincial governors, heads of administration, and national senators. Consequently, both direct and indirect beneficiaries of rice production were included among stakeholders. This helped reveal ecosystem services related to rice production, identify disciplinary overlaps, and fill gaps in the choice of decision makers to be involved.

There is the risk of overlooking ecosystem services or stakeholder groups that have not been prioritized in the first place, but are found to be important later in the assessment process. For example, in the INNOVATE project in Brazil, the relatively new and not yet generally recognized Watershed Committee was identified as an important stakeholder group after a series of expert consultations (step 1 and 3, Fig. 3.5). Furthermore, unexpected events can impact project priorities. During the course of the INNOVATE project a particularly strong drought triggered societal concerns over water quantity. Hence, ecosystem services related to water quantity increased in importance.

This decision-focused approach differs from the recommendation by Reyers et al. (2013), who suggest to assess the entire bundle of ecosystem services in order to address the full range of consequences and trade-offs involved in decision making. While assessing the entire bundle of ecosystem services is certainly important for a complete trade-off analysis, it is often constrained by the lack of resources and information. It is also not necessarily required in every decision context. For the case of the LEGATO project in Vietnam, for example, tourism and industrial development are likely to increase in importance for household income, but up to now they play a secondary role within the assessment, as the main focus is on enhancing pest control in rice farming systems (step 2, Fig. 3.4).

Whether the entire bundle of ecosystem services or only a subset of prioritized ecosystem services should be assessed is determined by the problem to be addressed (step 1), the different stakeholders and the decisions to be informed (step 3), and available methods and resources, including capacities, budget, and time. However, synergies and trade-offs involved in decisions and differences in preferences and impacts between stakeholder groups should be considered.

Figure 3.4: Problem-oriented approach for assessing ecosystem services (ES): case of the LEGATO project on rice farming in Vietnam (Settele et al. 2013).

Scoping phase (A): Determine the demand for the ES assessment.			**Assessment phase (B): Analyze ES within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ES flow, benefits, and trade-offs.**					**Implementation phase (C).**
Step 1: Specify and agree with stakeholders on problem and relevant research question	**Step 2: Identify ES beneficiaries and select ES most relevant for decision making**	**Step 3: Define information needs of decision makers**	**Step 4a: Assess current management and alternative options**	**Step 4b: Assess role of biodiversity and ecosystem processes for provision of ES**	**Step 4c: Assess flow of ES and how changes in 4a and 4b impact ES flow**	**Step 4d: Determine ES benefits, values, and ES trade-offs**	**Step 4e: Account for impacts beyond land use and ES**	**Step 5: Synthetize and integrate information for decision support**
Rice farmers suffer from water pollution caused by the use of pesticides and frequent incidences of rice pests. Loss of land due to conversion for industry, settlement and infrastructure. **Poverty:** Farmers have few or no alternative income sources. Production costs of rice farming are increasing due to use of artificial fertilizers and pesticides. **Development plans of** provincial government focus on promoting tourism and industry with little attention to sustainable farming practices. **Drivers:** poverty, water scarcity and pollution, pest infestations of rice farms, population growth, expansion of urban and industrial areas.	**Rice farmers benefit from:** Provisioning ES: rice yield and biomass; Regulating ES: soil fertility, pest control and pollination; Cultural ES: heritage, identity, aesthetics, education and tourism. **Tourists benefit from** heritage of historical buildings and landscape, religious purpose of temples, pilgrimage; leisure from recreational ES. **Government** has leading role in modernizing rice production and managing transition from an agricultural society to urban, industrial society.	**Farmers require knowledge on alternative farming practices** that can: - reduce the use of artificial pesticides and fertilizers; - reduce water pollution; - promote natural pest control; **-** enhance soil nutrients; - reduce post-harvest losses (e.g. dryers, safe storage facilities). **-** increase farm-gate selling price (access to new markets). **Can natural pest control replace artificial pesticides and fertilizers while maintaining yields and income?** Identify **alternative income** sources, e.g. from tourism.	**Rice farming:** Assess role of already existing **traditional farming practices** for natural pest control, enhancing crop yields, and promoting soil nutrient cycles. Test and apply methods of **ecological engineering** that promote natural pest control. **Explore options for diversifying income:** Producing and marketing organic rice products; Engaging in tourism-based income generation; Developing tourism infrastructure to attract international tourists to cultural heritage and landscape amenities. Consider role of societal drivers in particular the potential impacts of **urbanization and industrialization** on future development of rice farming.	**Assessment of natural pest control:** Analyze species diversity of insects in rice fields for assessing the influence of different farming practices on the abundance of natural predators that control pests (e.g. damsel- and dragonflies). **Pollinators:** Assess abundance of pollinators in order to assess influence of biodiversity on productivity of crops other than rice (e.g. fruits). **Soil nutrients:** Conduct field experiments on decomposition and soil nutrient cycling. **Tourism:** Assess link between landscape structure and aesthetics value and recreation.	**Assess how farming practices: enhance yield** of rice, vegetables, fruits, nuts, spices, meat, pork, and chicken; **influence regulating ES such as** natural pest control, pollination, and functional diversity; **impact water quality** with relevance for domestic water use and health. **Applying ecological engineering** can potentially enhance natural pest control, enhance yields of farms, reduce need for artificial pesticides and reduce water pollution. **Influence of land use on cultural services:** Traditional land-use practices are linked to cultural identity, social structures and rituals connected to nature. Historical-cultural identity is supported by old temples and other historical sites.	**Beneficiaries of ES:** Rice farmers, millers, traders, consumers, employees in tourism, city dweller seeking recreation, government (taxes). **Potential benefits of changing farming practices to ecological engineering:** more stable rice yields, higher market value of organic rice, cost savings and health benefits from reduced use of pesticides, cleaner water, enhanced reputation of farmers from producing clean and healthy products, providing perspective to young farmers, stronger cultural identity and aesthetics value due to more diverse landscape, benefiting tourism (jobs, income). **Trade-off: farming competes with** industry offering jobs and income.	**Training** in ecological engineering can provide farmers with **capacity, awareness and skills** that are also of relevance in other sectors and enhance **diversificatio n of income strategies.** **Reduction of dependency** on agro-chemicals allows more self-determined farming with potential to reduce costs. Socio-economic impacts of **industrial and urban development** are significant but are not assessed.	**Revise recommendations by regional government on rice farming.** **Revitalize traditional farming practices** that require less artificial fertilizers and pesticides, as this benefits pest control, water quality and income. **Enhance knowledge** on role of biodiversity for pest control. **Provide training** on ecological engineering in particular for young generation of farmers and decision makers. **Build capacity** for assessing trade-offs between different land-use options. **Guarantee long-term lease.** Enlarge fields. **Promote marketing of organic** farm products. **Communicate results** to intergovernmental organizations, e.g. Asian Development Bank (ADB).

The perception of ecosystem services and related terminology can differ between stakeholder groups, localities, and cultural contexts. The ecosystem service concept can serve as an analytical tool for translating context specific terms into an agreed ecosystem service classification system (e.g. Haines-Young and Potschin 2012). For example, in stakeholder consultations of the LEGATO project, it was not the goal to educate stakeholders about the ecosystem service concept, but to collect their knowledge on the benefits they receive from ecosystems expressed in their own terms. The ecosystem service concept was then used to unify the various terms and enable synthesis and further analysis. Translation back into stakeholder-specific terms should be considered when disseminating results during the assessment process (e.g. in step 5).

Step 3: Define information needs of decision makers.

Knowledge gaps in decision-making processes have to be addressed in order to ensure that an ecosystem service assessment generates relevant information (TEEB 2012) (Table 3.1). Identifying options for integrating ecosystem services related knowledge in ongoing decision-making processes supports the uptake of assessment results in decision processes (Ruckelshaus et al. 2015).

For example, the regional Watershed Committee of the São Francisco River in Brazil is in the process of developing a new water management plan for the next ten years. In a series of stakeholder workshops, members of the committee identified gaps in understanding the impacts of decisions on water-related ecosystem services. Sharing knowledge among all stakeholders helped building trust. As a consequence the Watershed Committee asked the INNOVATE project to contribute to filling the knowledge gaps. Thus, INNOVATE used hydrological models to inform about the amount of water available for irrigation, supply of drinking water, electricity generation, and critical ecological processes under different scenarios of decision making and climate change.

In the Tarim Basin in China, there is the need to generate a common understanding of impacts and trade-offs involved in decisions on land and water use across the region, in order to inform the development of the five-year-plan at a national and provincial level. The SuMaRiO project involves multiple institutions at regional level, each with competing interests and responsibilities in managing water distribution, agricultural production, forests, and biodiversity conservation (step 3, Fig. 3.6). Adequate and sensitive management of tensions is critical for developing a concerted strategy for the entire Tarim Basin. Hydrological models operating at a basin-scale were chosen to better understand the effects of different options for water distribution and land use (step 4). Based on this, a decision support tool was developed, allowing institutions to test different decision scenarios (step 5).
The assessment process also contributes to enhancing transparency and communication among different stakeholder groups.

Figure 3.5: Problem-oriented approach for assessing ecosystem services (ES): case of the INNOVATE project in the River São Francisco Watershed in Brazil.

Scoping phase (A): Determine the demand for the ES assessment.			**Assessment phase (B): Analyze ES within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ES flow, benefits, and trade-offs.**					**Implementation phase (C).**
Step 1: Specify and agree with stakeholders on problem and relevant research question	**Step 2: Identify ES beneficiaries and select ES most relevant for decision making**	**Step 3: Define information needs of decision makers**	**Step 4a: Assess current management and alternative options**	**Step 4b: Assess role of biodiversity and ecosystem processes for provision of ES**	**Step 4c: Assess flow of ES and how changes in 4a and 4b impact ES flow**	**Step 4d: Determine ES benefits, values, and ES trade-offs**	**Step 4e: Account for impacts beyond land use and ES**	**Step 5: Synthetize and integrate information for decision support**
Water shortage during prolonged periods of drought causes **conflicts over water management between multiple stakeholders** across multiple scales. **Water uses include: Water withdrawal** by farmers for irrigation, by households and communal water suppliers for domestic use (drinking water), by local users, and **transferred** to distant user regions. **Minimum water flow required** for ecological processes, fishery, aquaculture and navigation. **Water storage** for electricity generation from hydropower. Constructions of dams caused displacement of farmers to less fertile areas. **Lack of coordination between stakeholder groups** represented in the Watershed Committee).	**Maintenance of water flow benefits:** **-** domestic water use by households, - irrigation by farmers within and outside catchment, - hydroelectric power generation, - navigation, - ecological processes. **Maintenance of water quality** benefits domestic use, irrigation, livestock health, aquaculture, fisheries, and leisure. **Maintenance of soil fertility** benefits agricultural production. Negative effects from nutrient export to river. **Erosion control** benefits navigation and hydropower potential besides other.	**Watershed Committee** requires information for renewing its **water management plan**. The plan is developed for the next ten years and involves strategic decisions on water allocation. Information on options and effects of: - **distributing water** among different users, while **ensuring sufficient water flow and quality** for maintaining health of river ecosystem; - **reducing soil erosion** and nutrient export; - **identifying alternative income sources** related to tourism; - **coping with conflicts** in prioritization and decision making.	There is lack of **enforcing existing regulations** over water use or they show little effectiveness (e.g. water price not signaling its scarcity). Option 1: Enhanced **coordination** between stakeholder groups. Option2: Improve water **monitoring** for informing effectiveness of actions taken. Option 3: **Enhance existing water management** including sanitation. Option 4: **Protect and restore riparian vegetation** for controlling erosion and nutrient input. Option 5: **Enhance soil nutrients and adsorption capacity**. Option 6: **Diversify agricultural production** and aquaculture. Option 7: **Use income from water permits to** support and monitor current management and restoration projects.	Applying field measurements and hydrological and nutrient emission modelling for assessing: **Impacts of current and future land and water use** on water flow and quality; Options for **enhancing soil nutrients and reducing sedimentation**; Effect of land use on **water nutrients** (e.g. phosphorus); **Impact of reservoir management on macorphytes;** **Impacts of different land use intensities on biodiversity** (amphibians, and terrestrial plant species). **Considering the role of natural ecosystems** (e.g. riparian forests) for water provision and quality.	Hydrologic and hydro-economic models are used for assessing **water availability** under different water and land use scenarios. The Watershed Committee aims at **conserving riparian and spring ecosystems as they are found to be of importance for water supply and quality.** This option is not only of ecological relevance but also highly political, since it has been promoted as alternative to the contested transfer of water out of the catchment. **Trade-off: Actions for soil improvement** may promote further conversion of natural vegetation (Caatinga reserves) to cropland with negative impacts on biodiversity and water availability.	Who are winner and loser of water and land management strongly depends on **negotiations and power relations among users.** Several water claims are mutually exclusive. **Farmers benefit** from irrigation and water for livestock production; **Fishermen** benefit from maintenance of water flow and quality for fish production. **Villages and towns** benefit from supply for domestic use. **Local and indigenous communities** benefit from intrinsic and spiritual value of natural landscape. **Companies** benefit from water for hydroelectric power generation, industries, mining, water diversion projects, transport (waterways), tourism. **Water shortage and erosion** has multiple **negative impacts and costs** to different users.	Water shortage and land degradation drives **migration to towns** for alternative income options. Construction of dams causes **expulsion of people** from the region. **Distant water users** benefiting from water transfer, while also involving equity concerns. **Change in traditional water use** also has **cultural effects** on local and indigenous people.	The project is developing water and land use models which test scenarios of different land and water management options for decision support. **Further recommendations to the Watershed Committee include:** Support **democratic, inclusive and transparent decision-making processes** allowing for effective participation of all stakeholder groups. **Harmonization of and easy access to existing water monitoring** programs, data and statistics. **Capacity building related to importance of riparian ecosystems:** assess and monitor ongoing restoration projects with careful consideration of ecosystem services.

In Vietnam, rice farmers and authorities expressed their interest in low-cost measures for stabilizing or enhancing rice yields, reducing pre- and post-harvest losses (in particular through pest control), reducing water pollution from pesticide use, enhancing soil nutrients, and improving income and livelihood. The LEGATO project compared traditional and conventional farming systems for biological pest control, rice yields, nutrient cycling in soils, and impacts on water quality (step 4, Fig. 3.4). The analysis of the ecological processes related to biological pest control required species sampling over several growing seasons. This focus mainly determined the design, spatial scale, and timing of the assessment. Interactions with other practices that affect the farming system (e.g. tourism or forestry) were also investigated.

Careful consideration of the actual information needs by decision makers is important to ensure that ecosystem service assessments apply indicators and methods, which provide the type and detail of information required for a specific decision. At the same time, the expectations of stakeholders and decision makers about what an ecosystem service assessment can deliver need to be kept realistic, in order to ensure that assessment results are used appropriately and that misinterpretations and disappointments are avoided.

3.3.2 Assessment phase (B)

Step 4: Analyse ecosystem services within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ecosystem service flow, benefits, and trade-offs.

The previous steps provide the focus for the social-ecological analysis in step 4, which is divided into five sub-steps compatible with other SES approaches (Fig. 3.2): the assessment of current and alternative management options (4a), ecological factors relevant for producing ecosystem services (4b), the flow of ecosystem services (4c), ecosystem service benefits and trade-offs (4d), and impacts beyond land use and ecosystem services (4e) (Table 3.2).

Step 4a: Assess current management and alternative options.

Identifying policies and management options requires an understanding of the current land-use policies and practices within their socio-economic and cultural context (Cowling et al. 2008; Ostrom 2009, Chan et al. 2012a). Within ecological limits, landscapes offer a range of potential land-use options and configurations. Which of the land-use options are implemented and which of the ecosystem service benefits are appropriated and by whom partly depends on the ability of the different stakeholder groups and beneficiaries to influence land-use decisions (Spangenberg, von Haaren, et al. 2014). Social, cultural, and economic processes shape ecosystem service generation, with power relations, property and access rights, investment of time, labour, and resources determining the ecosystem service potential realized across a landscape.

Figure 3.6: Problem-oriented approach for assessing ecosystem services (ES): case of the SuMaRiO project in the Tarim River Basin in China (Rumbaur et al. 2015).

Scoping phase (A): Determine the demand for the ES assessment.			**Assessment phase (B): Analyze ES within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ES flow, benefits, and trade-offs.**					**Implementation phase (C).**
Step 1: Specify and agree with stakeholders on problem and relevant research question	**Step 2: Identify ES beneficiaries and select ES most relevant for decision making**	**Step 3: Define information needs of decision makers**	**Step 4a: Assess current management and alternative options**	**Step 4b: Assess role of biodiversity and ecosystem processes for provision of ES**	**Step 4c: Assess flow of ES and how changes in 4a and 4b impact ES flow**	**Step 4d: Determine ES benefits, values and ES trade-offs**	**Step 4e: Account for impacts beyond land use and ES**	**Step 5: Synthetize and integrate information for decision support**
Irrigation of cotton causes land **degradation** and increases the risk of **desertification**. **Melting glaciers** of Tien Shan Mountains are main water source. Water flow from glaciers is expected to decline after 2050 due to **climate change.** **Overuse of irrigation** is having multiple negative impacts on ecosystems including: **salinization** of soils and river water, **changes in ground water** level, soil degradation leading to **productivity loss of agricultural land,** degradation of natural forests. **Loss of natural forest** causes **biodiversity loss**, reduction in storm protection, and increase in **wind erosion** and sand drift.	**Most important ES of the Tarim River Basin include:** **Water** for agriculture, industry, households, and natural ecosystems; **Fiber:** 40% of Chinas cotton is produced in the region, heavily relying on water for irrigation. **Food:** water sustains irrigation agriculture and livestock. **Natural forests provide storm protection and erosion control,** mitigating sand drift, sandstorms, and desertification. **Habitat** for 60% of global population of Euphrat poplar (*Populus euphratica*). **Trade-off in water use for irrigation** vs. water needed for maintaining natural vegetation.	Information on ES is requested by **institutional stakeholders in Xinjiang Region:** **Water Resources Bureau** and **Tarim River Basin Management Bureau:** How does climate change impact glacier melt, water flow, and future water availability? **Agriculture Department:** How can salinization of soils be reduced? **Forestry Administration:** How do changes in groundwater impact natural forests? **Environmental Protection Department**: What are options for reducing salinization and desertification?	**Comparing impacts of different scenarios of climate change and land use on hydrology and socio-economy:** **Climate change:** Assessing impact of a) extreme, b) moderate, and c) low rise in global temperature on hydrology of the Tarim Basin. **Land use and socio-economic scenarios**: a) maximizing cotton production versus b) complete protection of natural vegetation and c) mosaic landscape of cotton production with natural vegetation. **Testing alternatives to cotton production:** Indian hemp (*Apocynum sp.*) is a salt tolerant plant that can be used for fiber production. It can be used for restoring saline soils. Economically viable production is being tested.	**Assessment of: Climate change** using climate models for predicting changes in precipitation and temperature; **Hydrology** using model MIKE HYDRO for computing water discharge, water use and allocation. **Agricultural changes** using agro-economic models for comparing productivity of cotton on "healthy" soils and on saline soils with the productivity of the alternative fiber plant *Apocynum sp*. **Forest changes** analyzing state of forests on shallow and deep ground-water levels, forest distribution, and use of fuelwood. **Desertification** analyzing area affected.	**Assessing impact of changes in climate and land use on ES flow:** **Water provision** from glacial melt compared with water demand for irrigation agriculture, maintenance of natural ecosystems. **Fiber production:** Compare yields of cotton with *Apocynum sp.* **Fuelwood:** Assess harvest of fuel wood in riparian forest. **Storm and erosion control by forests:** Assess avoidance of sand drift, soil erosion, sandstorm and avoided damage to infrastructure (e.g. roads). **Habitat:** Asses species composition of forests, comparing different stages of forest use and degradation. **Recreation:** Assess aesthetic appreciation by visitors.	**Water provision** benefits agriculture, industry, domestic use (e.g. drinking water). **Fiber from cotton provides** income to small private and large state farms and large-scale investors. **Fuel wood** is used by households for heating and cooking. **Storm protection, erosion control by forests:** estimating avoided damage to roads. **Provision of habitat** species abundance. **Aesthetics:** benefits to tourism. **Trade-offs:** *Apocynum sp.* finds lower acceptance by farmers than cotton as production is more expensive providing less income. Farmers prefer alternative water-intense crops e.g. Chinese dates.	External effects are not addressed in this study but include: Water use and environmental impacts by **other economic sectors** such as oil and gas mining; Impact of desertification on **other regions and infrastructure** **Health effects** by dust due to wind erosion; Potential **land-use conflicts** effecting minorities; Effects of land use and land degradation on **migration**.	**Findings include:** By 2060 glaciers are likely to have disappeared due to climate change, which is likely to lead to a **decrease in runoff** and water shortage in the Tarim River (Sorg et al. 2012). Based on models a **Decision Support Tool (DST)** is developed allowing institutional stakeholders to **evaluate possible consequences** of scenarios of land and water use. **Impacts of different scenarios on ES provision are rated** (good, medium poor); The **DST has mainly educational purposes;** **Simple solutions do not exist.** Decision involves trade-offs in agriculture and nature conservation. The DST can inform decision making on likely impacts on ES provision.

Table 3.2: Examples of questions, actions, and indicators for assessment phase (B).

Assessment phase (B): Analyse ecosystem services (ES) within social-ecological context and impacts of changes (e.g. in land use, policies, climate) on ES flow, benefits, and trade-offs.

Questions	Actions	Indicators (qualitative & quantitative)
Step 4a: Assess current management and alternative options		
What are historical and current land-use practices and which policies and institutions influence change? How are future changes expected to influence land use and ES provision? What formal and informal policies, norms, and rules influence land-use decisions? Which drivers influence land-use practices and policies (e.g. cultural or economic drivers)? What are potential alternative land-use options and policies? Which freedom of choice do local farmers have?	Analysing how policies and institutions influence land-use practices in order to identify options for improving resource use and governance (e.g. Rathwell and Peterson 2012). Providing evidence from success stories in other regions in order to identify alternative options. For example Goldman et al. (2008) found that using ES information had a positive influence on the success of conservation projects. Developing social-ecological models and scenarios of future changes together with stakeholders and decision makers for understanding drivers for ES provision and likely trade-offs (e.g. Reed et al. 2013).	Types of land-use practices and change over time; Laws, regulations and financial mechanisms such as subsidies, taxes, or fines; Institutions governing land use; Developments in market price of crops and market access; Formal regulations e.g. related to pesticides and nutrients use; Traditional and informal rules e.g. on cropping cycles, types of crops used; Cultural rules and norms e.g. rites related to land use; Level of decision making (by individual farmer or by central government).
Step 4b: Analyze role of biodiversity and ecological processes for provision of ES		
Which elements of biodiversity and ecosystem processes are important for ES provision over an extended period of time? How do land use and other relevant driver impact biodiversity and ecosystems (e.g. changes in population, policies, markets, and climate)? What are likely impacts of alternative land-use options and policies on biodiversity and ecosystem processes?	Choosing methods that resonate with decision makers and adapting them to particular information needs to ensure credibility of ES data for decision making. For example mapping and modelling of ES can be targeted to specific stakeholder needs (e.g. Petter et al. 2013, Crossman et al. 2013). Using in-situ field measurements for monitoring biodiversity and ecosystem processes, e.g. species presence or hydrological monitoring. Analysing historical trends in land use and conditions of ecosystems using remote sensing.	Mapping forest area and assessing species composition e.g. for estimating potential for carbon storage and biodiversity conservation; Model influence of drivers on biodiversity and ecological processes relevant for ES provision; Presence or absence of species important for pest control; Sediment content in river water, e.g. as indicator for role of vegetation for water quality and erosion.
Step 4c: Assess flow of ES and how changes in 4a and 4b impact ES flow.		
How do biodiversity and ecosystem processes contribute to the provision of ES? How do changes in land use and other drivers influence ES flow (e.g. changes in population, policies, markets, and climate)? How would alternative land-use options and/or policies impact ecosystems and ES flow?	Assessing impacts of changes in management on ES flow, using integrative methods and tools, including socio-economic and ecological models (e.g. Bagstad et al. 2013). Modelling impacts of land-use change on ES flow such as erosion, sediment load, nutrient concentration in water or water availability (e.g. Villa et al. 2014). Assessing impact of changes in crop growth on yield or changes in species composition on spread of pests. Assessing impact of changes in forest use on carbon stocks, availability of wood for fuel and construction, bush meat, medicinal plants, etc.	Water flow in river under different land use, land cover, or climate scenarios; Comparing crop yield for different stages of soil degradation; Abundance of pests in relation to species composition; Water quality (e.g. nutrient or sediment content) for different scenarios of land use and cover; Erosion control by vegetation for different land-use scenarios; Carbon sequestration by forest under different forest management options.

Table 3.2: (continued)

Step 4d: Determine ES benefits, values, and ES trade-offs resulting from changes in 4a-4c.		
Who are ES beneficiaries? Who are recipients of disbenefits? What are the ES benefits? What are the ES disbenefits? Which social and cultural values are affected positively and negatively by the service/disservice? Which socio-economic values are affected amongst the different stakeholder groups? What human inputs (e.g. knowledge, skills, resources, costs, etc.) are required for accessing ES? Which indicators and methods for assessing the benefits/disbenefits of ES are relevant and meaningful to different stakeholders and decision makers?	Assessing benefits and disbenefits of ES bundles for different stakeholder groups and land-use types (Raudsepp-Hearne et al. 2010; Goldstein et al. 2012; Martín-López et al. 2014). Using multi-criteria-analysis and cost-benefit-analysis to account for both qualitative and quantitative ES information in assessing the impacts of land-use changes on human well-being (e.g. Sijtsma et al. 2013). Assessing impacts on social and cultural values such as status, sense of place, social relations (e.g. Chan et al. 2012a, Chan et al. 2012b). Assessing monetary and non-monetary values of ES (e.g. Christie et al. 2012, Viglizzo et al. 2012). Mapping cultural ES (e.g. Plieninger et al. 2013).	Impact of changes in crop yield on income and status of farmers and decision makers; Impact of changes in pests on yield, income, and subsequent changes in land management; Impacts of changes in water availability on water user, e.g. changes in water price, changes in crop yield; Health benefits, e.g. due to improvement in water quality; Health damage cost; Impact of changes in forest cover on erosion, hunting success, carbon stocks; Changes in water treatment costs; Saved costs of sediment removal from reservoirs for hydropower production.
Step 4e: Impacts beyond land use and ES		
Which other sectors or institutions beyond land use are affected by changes in ES flow and benefits/disbenefits? Which cultural and social impacts occur due to changes in ES (e.g. impacts on traditions, norms, rituals)?	Analysing impacts on education, social norms, traditional practices, rituals, social structures. Identifying links to other sectors and infrastructure related to energy, transport, communication, etc. Assessing changes in distribution of wealth and income, political stability and social security, self-determination vs. transfer dependency.	Educational benefits and capacity building due to assessment process; Access to new knowledge and technology; Behavioural changes of land user e.g. crowding out effects (Rode et al. 2015); Changes in access to infrastructure, markets, and communication; Income distribution patterns; Changes in the hierarchies of social structures.

In the Tarim River Basin in China, land-use decisions are centralized but involve multiple government institutions (Land and Resources Bureau and departments of Agriculture, Forestry and Environmental Protection) that make decisions at regional level following guidelines by the central government. Complex trade-offs exist in land and water use for cotton production, hydropower generation, forestry, and conservation of natural habitats (e.g. Feike et al. 2015). To better understand the impacts of different land-use options, scenarios were developed including climate change with high and low water availability, and land use with different intensities of cotton production and nature conservation. In field experiments, alternatives to irrigation-intense cotton production were tested using the salt-tolerant plant Apocynum sp. This plant is suitable for fibre production and can be used for the restoration of degraded agricultural soils. Throughout the assessment process interviews and discussions with stakeholders informed the development and testing of the different options.

In the case of the São Francisco watershed in Brazil, analyses of past and current water governance found that comprehensive water policies already exist for addressing water distribution issues, especially at the federal level. However, the implementation and enforcement of these policies is weak and the water monitoring is inadequate to measure the effectiveness of policies. INNOVATE addressed these immediate information needs of the Watershed Committee by developing guidance on implementation of existing policies and improving water monitoring (step 5).

LEGATO`s ecosystem service assessment compared traditional and conventional rice farming systems for factors that impact income and livelihoods of farmers, including institutional settings and world views that may guide different land management decisions, biological pest control, rice yields, and nutrient cycling in soils (step 4a, Fig. 3.4).

In the case of the SuLaMa project in Madagascar, decisions of farmers and smallholders are largely based on traditional knowledge (step 4a, Fig. 3.3). Crops are primarily cultivated for subsistence, with surpluses being traded as a source of income. Besides crops, livestock plays an important role for people’s livelihood. It provides a fallback resource in periods of crop failures and also determines social status. Current land use leads to ecosystem degradation and encroachment in the Tsimanampetsotsa National Park. This situation is aggravated by cattle thieves driving farmers to graze their livestock in forested areas. Thus, the SuLaMa project analysed the drivers of degradation, their impacts on biodiversity and ecosystem service provision, and explored options of more sustainable land use. Besides others, this includes fodder production for livestock as means for reducing grazing pressure and the use of home gardens as means of diversifying sources of income.

Step 4b: Assess role of biodiversity and ecosystem processes for provision of ecosystem services.

In this step, ecological processes and biodiversity indicators relevant for the provision of the prioritized ecosystem services are identified and analyzed. This includes biophysical measurements, modelling of ecological processes, and biodiversity assessments as well as characterization of relevant drivers. Again, multiple sources of knowledge should be taken into account including scientific, traditional, and indigenous knowledge. Biophysical assessment methods are numerous, and factors influencing the choice of methods include: the type of biophysical indicators required for addressing the information needs, available expertise and resources, available data, and extent to which primary data have to be measured in the field.

In the Tarim Basin in China, the SuMaRiO project used the hydrological model MIKE HYDRO for estimating water discharge and allocation for irrigation. Cotton yields on intact soils were compared with yields on degraded soils, and productivity of the more salt-tolerant crop Apocynum sp. were tested in the field to inform model simulations of alternative crop production. Methods of forest monitoring were used to assess how forest biodiversity and its role for erosion control are impacted by changes in groundwater levels.

In INNOVATE, the hydrological model SWIM and the nutrient emission model MONERIS were calibrated and adjusted for the São Francisco River. The MAgPIE model was used to estimate future land use under climate change. Hydro-economic analysis was performed for a sub-region of the catchment. A species distribution model of the semiarid Caatinga vegetation was set up with Maxent. While these models mainly use secondary data, primary data on biodiversity and alternative land-use options were collected in the field.

LEGATO in Vietnam analysed the role of biodiversity for pest control, conducting inventories of species (e.g. of parasitoids or damsel- and dragonflies) that control pests. Impacts of fertilizers and pesticides on ecological processes were investigated via field inventories of pollinators, native and alien plant species, soil organisms, and nutrient cycles. This was accompanied by surveys among farmers to assess productivity of rice fields for the different farming systems. The analysis of the ecological processes was the main factor determining the design, spatial scale, and timing of the assessment.

Step 4c: Assess flow of ecosystem service and how changes in 4a and 4b impact ecosystem service flow.

In this step, the interplay between social (4a) and ecological factors (4b) and their role for the production and flow of ecosystem services is assessed. A causal relationship between ecological factors and the provision of ecosystem services is often anticipated, but it is rarely proven or quantified (Carpenter et al. 2009, Reyers et al. 2013). Proxy indicators are often used in cases where direct measurements of ecosystem services are missing or for simplifying the analysis (e.g. changes in forest cover as proxy for carbon sequestration). Additional validation is required in case proxies are used to transfer results across different sites.

Given the complexity involved in social-ecological systems, computer-based models are often the first choice for analysing climate change impacts, drivers of land-use change, their impacts on ecosystem service flow, and alternative land-use scenarios. This is in particular true for large-scale assessments as undertaken by INNOVATE and SuMaRiO (Fig. 3.5 and 3.6) (e.g. Krysanova et al. 2015). Validating models based on empirical data and discussing their plausibility with scientists and stakeholders is critical to ensure that model outputs provide relevant information for decision making. In the Tarim Basin in China, hydrological modelling combined with stakeholder consultations helped inform decision makers about potential impacts of land-use decisions on water availability. Through this process the relevance of forest conservation for protecting infrastructure and agricultural land from desertification was communicated to respective stakeholders.

Field surveys and experiments allow ground trothing the assumptions on ecosystem service flows. In Madagascar, the SuLaMa project used household surveys to analyse the relevance of ecosystem services for household income, including yields of different crop varieties, productivity of home gardens, fodder production using Samata (Euphorbia stenoclada), and use of wild plants. Inventories of insect species in rice fields in Vietnam elucidated the benefits which local communities obtain from traditional farming practices that support natural pest control (LEGATO, Fig. 3.4).

Step 4d: Determine ecosystem service benefits, values, and ecosystem service trade-offs.

Valuation of biodiversity and ecosystem services depends on the perception of stakeholders that benefit from ecosystem services or suffer disbenefits (Görg et al. 2014). There are multiple values which stakeholders can attach to biodiversity and ecosystem services, including social, cultural, and economic (monetary and non-monetary) values (TEEB 2012, Chan et al. 2012a). Demonstrating these values with analytical methods in quantitative and qualitative terms can be a challenge; in particular, when it comes to spiritual and cultural values, public goods, and future generations. The types of values to be assessed and the choice of methods and indicators should be tailored to each specific decision.

Although increasing in popularity, monetary valuation of ecosystem services is not necessarily required or useful in every decision context. Alternative and complementary methods for addressing social and cultural values can be more relevant to decision makers (Limburg et al. 2002, Daily et al. 2009, Abson and Termansen 2011, TEEB 2012, Chan et al. 2012b, Ruckelshaus et al. 2013, Sijtsma et al. 2013). Multi-criteria analysis is an option for integrating qualitative and quantitative information on values in decision making (e.g. Fontana et al. 2013). There is also an increasing number of tools for data integration (Bagstad et al. 2013).

In particular, traditional land-use practices cater multiple values. Rice farming in Vietnam is not only a source of food and income, but it is deeply interlinked with local culture and traditions, which developed around rice farming over generations. Hence, in the LEGATO project, alternative rice-farming practices were not only evaluated for their benefits in terms of income and environmental impacts, but also for their impacts on local culture and identity. Rice farming systems based on traditional knowledge are expected to account for ecological processes, using locally adapted crop varieties, which require less input of artificial fertilizer and pesticides. Such systems are expected to enhance natural pest control, thus requiring less chemical inputs, which in turn reduces related costs and benefits water quality. Traditional farming is also promoting a sense of place by strengthening local traditions and social bonds (Tekken & Settele 2014). This has potential benefits for tourism, which brings new income sources to the region (but can also exert stress on traditions and social bonds). Accessing markets for organic products can potentially provide a long-term perspective also for younger rice farmers.

Similarly, in Madagascar, land-use practices are strongly linked to local culture through traditional knowledge and religious beliefs. Besides analysing crop yield, food availability and cash income, the SuLaMa project also accounted for cultural values involved in each of the analysed land-use practices. Wild plants do not only serve as food or medicine but also fulfil important roles in traditions and rites. The number of livestock determines the social status of households, providing an incentive to increase livestock numbers, which can enhance grazing pressure.

In the case of watershed management addressed by INNOVATE in Brazil and SuMaRiO in China, ecosystem service valuation targets more long-term investment decisions across regional scales. Stakeholders were asking for quantitative information on water flow, crop yield, costs of water provision, costs of ecosystem degradation, and impacts on income. Ecosystem service valuation was used to identify the winners and losers of different watershed management strategies. In the Tarim Basin in China, SuMaRiO project assessed the ecological and economic potential of Apocynum sp. as an alternative to cotton production (Thevs et al. 2012). The value of natural forests for reducing wind erosion and desertification was analysed by estimating avoided costs from reduced loss in agricultural land and reduced infrastructure maintenance, e.g. cleaning sand from roads.

Step 4e: Account for impacts beyond land use and ecosystem services.

Decision making within the assessed social-ecological system can have external effects on other social-ecological systems (Ostrom 2009). Shifts in land use can impact stakeholder sectors and land-use systems within and outside the study region. Valuation of ecosystem services can have impact on cultural values or behaviour. For example, introducing monetary values as an argument for conservation of biodiversity can replace cultural and intrinsic motivations for conservation (crowding-out effects) (Rode et al. 2015).

In the assessment of watersheds in Brazil (INNOVATE) and China (SuMaRiO), it is recognized that changes in land and water use greatly impact migration of people in and out of the region, although it is not the central focus of the assessment. The INNOVATE project acknowledged plans for artificial water transfer to regions outside the watershed and the severe impacts this can have on the future development of the entire catchment. Due to the lack of transparency regarding the details of these plans, this factor is subject to speculation. In the Tarim Basin in China, mining of oil and gas is an important water user, but this sector was beyond the scope of the SuMaRiO project due to limited resources and political reasons. Although cattle theft is a major problem in Madagascar, it was not the focus of the SuLaMa project to assess behavioural changes of cattle thieves in response to changes in cattle production. In Vietnam, industrial development impacts income opportunities, causing migration of young people to cities and a decline in farming population. This issue is documented by the LEGATO project but not assessed in detail since these drivers are beyond the project’s influence.

Although such external effects cannot always be analysed in detail, it is critical to recognize their existence. They substantiate the discussion of uncertainties of the findings and help embedding the findings of ecosystem service assessments into the larger decision context.

3.3.3 Implementation phase (C)

Step 5: Synthetize and integrate information for decision support.

Step 5 focuses on the use of ecosystem service information for decision support based on the synthesis of information generated in the previous steps (Table 3.3). The outcomes of ecosystem service assessments depend on the information needs defined in scoping phase

A and need to be adapted to the particular ecological, socio-economic, and cultural context. Assessment results can help change stakeholder perspectives and trigger changes in the management of biodiversity and ecosystem services (Ruckelshaus et al. 2015). Whether this change is for better or worse depends on how the information is used and by whom. Avoiding that ecosystem service information leads to adverse impacts, e.g. the commodification and exploitation of nature (Turnhout et al. 2013; Schröter et al. 2014), requires broad stakeholder participation and transparency in defining and using ecosystem service information (Chan et al. 2012a, Jax et al. 2013).

Table 3.3: Examples of questions, actions, and indicators for Implementation phase (C).

Implementation phase (C)		
Questions	**Actions**	**Indicators (qualitative & quantitative)**
Step 5: Synthetize and integrate information for decision support		
How to communicate the generated ecosystem service (ES) information, so it is adopted by stakeholders? Are there windows of opportunities for bringing assessment results to the attention of key decision makers, institutions, or including it in public debates? How can the generated ES information trigger changes in policies and practices? How to ensure that these changes improve the sustainability of land use? Are there important knowledge gaps that require an iteration of assessment steps?	Promoting science-practice partnerships from the start to enable co-design of user-inspired and user-relevant knowledge (Milner-Gulland et al. 2010, Ntshotsho et al. 2015). Promoting use of assessment results through user-adapted decision support tools such as participatory models, maps, guidelines, user-targeted publications, and websites (e.g. Liekens et al. 2013). Consulting stakeholders, decision makers, and experts on the use of ES information. Establishing monitoring system for tracking positive and negative changes. Repeating assessment steps if necessary.	Awareness of stakeholder groups on availability of ES information, e.g. through the use of assessment results or published reports. Monitoring of qualitative and quantitative changes in ES using indicators e.g. for water quality, sediment load, crop yield, carbon stock etc. (e.g. Feld et al. 2009). The type of ES information and tools used by stakeholders in decision processes.

Integrating ecosystem service information into decision making and changing land management to more sustainable practices require adaptive management (Cowling et al. 2008), involving an iterative and participatory process of prioritizing management actions, monitoring their performance, and adjusting management practices in accordance with the defined objectives (Martinez-Harms et al. 2015). The outcome can be as unique as the assessment process itself, depending on the specific social-ecological context. Hence, guidance on integrating ecosystem service information into decision making can only remain general. However, science-practice partnerships, involving close collaboration of practitioners and scientists from outset of the assessment, can help generate user-inspired and user-relevant knowledge that promotes effective management on the ground (Ntshotsho et al. 2015).

In the INNOVATE project, guidelines for the watershed management of the São Francisco River in Brazil were discussed with stakeholders, in order to improve water monitoring and inform existing policies and restoration efforts. Collaboration with local and regional research organizations ensures capacity building for future assessments in the region. Supporting ongoing restoration and conservation projects with data on biodiversity and land use may pave the way for a more careful consideration of natural resources in decision making. Recommendations are provided in writing, presented in live events, and discussed and refined during stakeholder consultations. These efforts can also support the development of more transparent and democratic decision-making processes for water management.

The decision support tool developed by the SuMaRiO project in China supports institutions at national and provincial level in testing different scenarios of land and water use (Siew et al. 2014). The tool has mainly educational purpose and allows the involved institutions to better understand possible impacts of land-use decisions on ecosystem services. Although it is a simplification of the watershed, the tool supports institutions in developing an improved understanding of the complexity of the system and general trends across the watershed.

Enhancing the use of home gardens has been identified by the SuLaMa project in Madagascar as a viable option that improves income of local households and increases resilience to environmental disturbances (e.g. pests and droughts). Local acceptance of this strategy is expected to be high, as it builds on existing land-use practices and benefits women in particular. With regards to potential alternative strategies for crop and fodder production, more investigation is needed in order to get a better understanding of possible adverse impacts (e.g. an increase in livestock production could cause conflicts over scarce water resources). Modern farming practices were previously introduced by development organizations but subsequently abandoned for the lack of local acceptance, indicating complex social-ecological challenges involved in establishing alternative land-use practices.

Educating and training farmers and government officials in ecological engineering is identified by the LEGATO project as an important component of supporting rice farmers in Vietnam. “Farmer field schools” and “entertainment education” including soap opera episodes on radio and TV (Escalada et al. 1999, Heong et al. 2008, Heong et al. 2014) proved to be effective tools for education about the practices of ecological engineering. Furthermore, based on the ecosystem service assessment, policy advice was developed for regional and national government departments to better integrate knowledge on biodiversity and ecosystem services in rice farming policies. Provincial administrations insisted on the participation of representatives of the agricultural administration in farmer trainings in order to build capacity for repeating them on a province-wide scale. In addition, the project was frequently consulted for advice on provincial development plans. Despite this success, the generated information can become irrelevant to decision makers; for example, if other issues on the political agenda become more relevant, or in case of mismatch of competencies between project partners.

3.4 Discussion and conclusions

Initiatives like the SLM Program and PECS aim at applying ecosystem service assessments to inform decisions on specific land-use problems. However, simply generating ecosystem service information does not guarantee its relevance for decision making (Laurans et al. 2013). Often science-driven ecosystem service assessments focus only on biophysical functions (Honey-Rosés & Pendleton 2013), ignoring diversity in ecosystem service benefits and information needs by decision makers. Social and political processes in the provision and distribution of ecosystem services and resulting social, distributional, and economic impacts are often not analysed. The presented problem-oriented approach was developed to better target ecosystem service assessments to specific information needs by decision makers. The approach builds on the analysis of empirical experience of four place-based ecosystem service assessments (Fig. 3.1) and existing ecosystem service frameworks (Fig. 3.2).

The presented approach stresses the need to: a) identify land-use problems (step 1) and related information needs by decision makers (step 3) from the outset of the assessment process, and (b) focus on decision-relevant ecosystem service information throughout the assessment process (step 2 and step 4).

Step 1 and step 3 are useful for focusing ecosystem service assessments on land-use problems from a stakeholder point of view within a particular local or regional decision context. This promotes both engagement of relevant stakeholders and the building of trust between stakeholder groups. Trust among stakeholders is important for sharing knowledge but also for acknowledging relevant knowledge gaps. This includes, for example, local knowledge on diversifying crop production as a means of building resilience to droughts and pests in Madagascar (SuLaMa, Fig. 3.3), and knowledge on the relevance of local practices for enhancing resistance of rice farming to pests in Vietnam (LEGATO, Fig. 3.4).

Targeting the assessments on priorities relevant for decision making (step 2 and step 4) helps to integrate ecosystem service information into ongoing policy processes (step 5). For example, the SuMaRiO project (Fig. 3.6) informs the development of the five-year-plan for the Tarim Basin in China about ecosystem service trade-offs involved in cotton production. Having a clear focus on decision-relevant land-use problems from the outset of the assessment enhances the probability that the generated ecosystem service information will be integrated in the decision process.

The presented approach also facilitates the establishment of partnerships with decision-relevant institutions, the development of a common understanding of the issues at stake, and the building of trust between stakeholders involved in the assessment. For example, it enabled the INNOVATE project (Fig. 3.5) to establish a close working relationship with the Watershed Committee of the São Francisco River in Brazil, allowing effective communication of information needs of decision makers to the scientists conducting the ecosystem service assessment. This also allows the transfer of assessment findings back to relevant stakeholders and decision makers, highlighting where regional and national policies and development priorities override interests of local land user.

The clarity of problems and information needs is also important to agree on assessment goals and the type of decision support that an ecosystem service assessment can realistically deliver within a given context and with available resources. The process of co-design with stakeholders allows identifying opportunities for the ecosystem service assessment to provide a meaningful contribution to a specific decision-making process. This is important to clarify limitations and avoid overly ambitious expectations. Ecosystem service assessments can trigger changes in decision making; in particular, if they are linked to ongoing decision-making processes. The development of decision support tools and guidelines can be useful in promoting this process. Nevertheless, the impact of technical decision support tools should not be overestimated, as decision processes are often complex negotiations dependent on multiple factors that are beyond the scope of an ecosystem service assessment.

Ecosystem service assessments are unlikely to deliver ultimate solutions to the identified problems. When ecosystem service assessments become part of a political process, they can contribute to solutions but also trigger new conflicts. For example, the INNOVATE project identified that the ecosystem service assessment can help making decisions on water management more transparent and thereby facilitate stakeholder involvement in water management. However, more transparency in decision making is not always wanted by all stakeholders or decision makers.

Nonetheless, achieving a shared understanding of the role of ecosystem services within the social-ecological context can already be beneficial for the decision-making process. Designing ecosystem service assessments is a learning process where the design is refined and re-adjusted in the course of the assessment process and in response to newly acquired knowledge. To paraphrase Albert Einstein, assessments should be as simple as possible, but no simpler. We recognize that step-wise approaches are a simplification of the process required to fully understand the complexities involved in social-ecological systems (Rogers et al. 2013). However, our approach is meant to provide pragmatic guidance for making ecosystem service assessments more policy-relevant by focusing the design of assessments on particular land-use problems, stakeholder priorities, and information needs in order to explore options for more sustainable land management.

4 Global assessment of factors determining expected carbon performance of REDD+ projects

Summary

Reducing carbon emissions from deforestation and forest degradation (REDD+) is a critical component of policies for mitigating climate change. Here I investigate the potential of REDD+ projects for reducing carbon emissions and assess factors determining their performance using meta-analysis. I identified 66 REDD+ projects validated by carbon standards that estimate to conserve 9.1 million hectares of forests expecting net emission reductions of 1.6 $GtCO_2e$. Carbon performance is positively associated with historical deforestation rates, governance effectiveness and project design for avoiding planned deforestation. Projects seeking multiple ecosystem services trade off this multifunctionality with lower emission reductions. Private stakeholders favour projects with high carbon performance and carbon rights are private for 75.8% of total net emission reductions. Local communities are expected to gain land tenure security in 65% of projects, but hold carbon rights to only 10.4% of emission reductions. This emphasizes the need for safeguards ensuring equitable outcomes of REDD+.

4.1 Introduction

Reducing emissions from deforestation and degradation in developing countries and promoting sustainable forest management for conserving and enhancing forest carbon stocks (REDD+) is critical for achieving the Paris Agreement of holding global warming below 2°C compared to pre-industrial levels (UNFCCC 2008; UNFCCC 2015; Houghton et al. 2015; Grassi et al. 2017). Over the past decade, more than 453 REDD+ initiatives have been launched (Simonet et al. 2016) and understanding factors promoting or constraining their carbon performance can guide the future design and implementation of REDD+ policies and projects.

Originally, REDD+ is based on the principles of payments for ecosystem services (PES), a market-based policy instrument, whereby agreed-upon actors are compensated for environmental services they supply or manage (Wunder et al. 2008; Leimona et al. 2015). In REDD+, payments for forest carbon are an incentive for mitigating deforestation drivers (Weatherley-Singh & Gupta 2015), whereby avoided carbon emissions are traded as verified carbon units in carbon markets (Sandker et al. 2010). Strategies for achieving emission reductions and benefit distribution are specified within the design process of individual projects. This project-based approach has evolved into a results-based aid effort dominated by nonmarket development funding for national and sub-national REDD+ initiatives (Angelsen 2017).

Existing REDD+ initiatives are typically located in tropical countries with high forest carbon stocks, high baseline emissions from deforestation, high scores in government effectiveness and a high number of threatened species (Cerbu et al. 2011). While REDD+ offers multiple benefits for climate change mitigation, biodiversity conservation and development (Chhatre & Agrawal 2009; Olsson & Ouattara 2013; Bustamante et al. 2014; Gilroy et al. 2014; Labrière et al. 2015), trade-offs are to be expected (Bustamante et al. 2014; Phelps et al. 2012; Bottazzi et al. 2014). For example, there is concern that the categorization of forests into tradable carbon units promotes activities for maximizing carbon sequestration while undermining biodiversity conservation and excluding local communities from accessing forests for livelihood needs (Bottazzi et al. 2014; Corbera 2012; Pokorny et al. 2013). With large areas of tropical forests being privately owned (Richards & VanWey 2015), there is the risk that a market-driven approach of REDD+ leads to further privatization of forests, exacerbating the marginalization of forest-dependent communities (Atela et al. 2015). Therefore, REDD+ policies demand compliance with safeguards for the *“protection and conservation of natural forests and their ecosystem services”* and for ensuring *“full and effective participation of relevant stakeholders, in particular indigenous peoples and local communities”* (UNFCCC 2011). A better understanding of factors determining carbon performance and related trade-offs can help achieving efficient and fair REDD+ projects.

4.2 Materials and methods

I used information provided in design documents of REDD+ projects together with information on their national context to first, quantify the potential of REDD+ projects for mitigating carbon emissions and second, identify factors determining their expected carbon performance.

My sample includes all REDD+ projects (n = 66) that have gone through a third-party validation process by major international carbon standards including the Verified Carbon Standard (VCS), the Climate, Community and Biodiversity (CCB) Standard, and Plan Vivo (PV) (Fig. 4.1, Supplementary Table 4.1). The projects expect emission reductions to be additional and permanent, and report total net emission reductions for the project area over a specific carbon accounting period in metric tonnes of carbon dioxide equivalents (NER_{total} in tCO_2e) (Fig. 4.2, Supplementary Table 4.2).

For quantifying net emission reductions (NER in tCO_2e) expected to be generated by REDD+ projects and for identifying factors explaining the variance in the expected carbon performance of REDD+ projects (NER_{std} in tCO_2e ha^{-1} yr^{-1}), I conducted a meta-analysis using multiple linear regression analysis (Rudel 2008).

4.2.1 Data source

Data on REDD+ projects was derived from project design documents, which were downloaded from online databases of the respective carbon standards (Supplementary Table 4.1 and 4.2). Supplementary List S4.1 provides an overview of all analysed project design documents.

Only REDD+ projects in a mature stage were included, meaning that projects 1) are operational, 2) generate or are about to generate carbon credits for trade in carbon markets, and 3) underwent validation by at least one major international carbon standard, including the Verified Carbon Standard (VCS), the Standard of the Climate, Community and Biodiversity (CCB) Alliance and Plan Vivo (Fig. 4.1, Supplementary Table 4.2). These selection criteria were applied in order to ensure that expected emission reductions are reported according to current standards in monitoring, reporting and verification, and account for uncertainties.

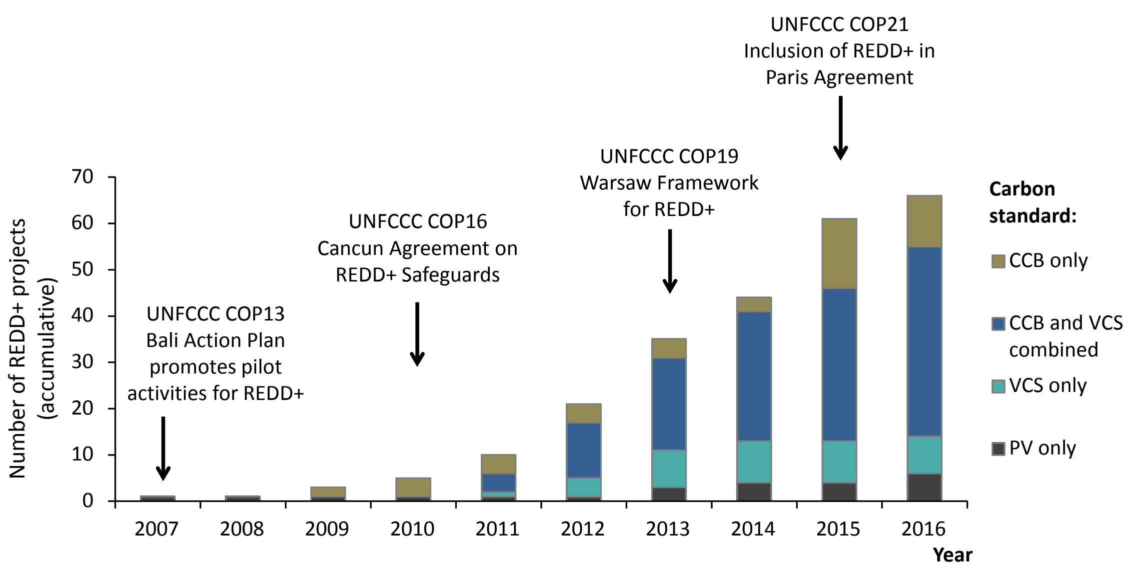


Figure 4.1: Number of REDD+ projects validated by major carbon standards and accumulated over time. In total, 66 REDD+ projects were identified that have been validated by carbon certification standards, including the Verified Carbon Standard (VCS), the Standard of the Climate, Community and Biodiversity (CCB) Alliance, and Plan Vivo (PV) (as of September 10, 2016). Arrows indicate REDD+ policies under the United Nations Framework Convention on Climate Change (UNFCCC) between 2007 and 2016.

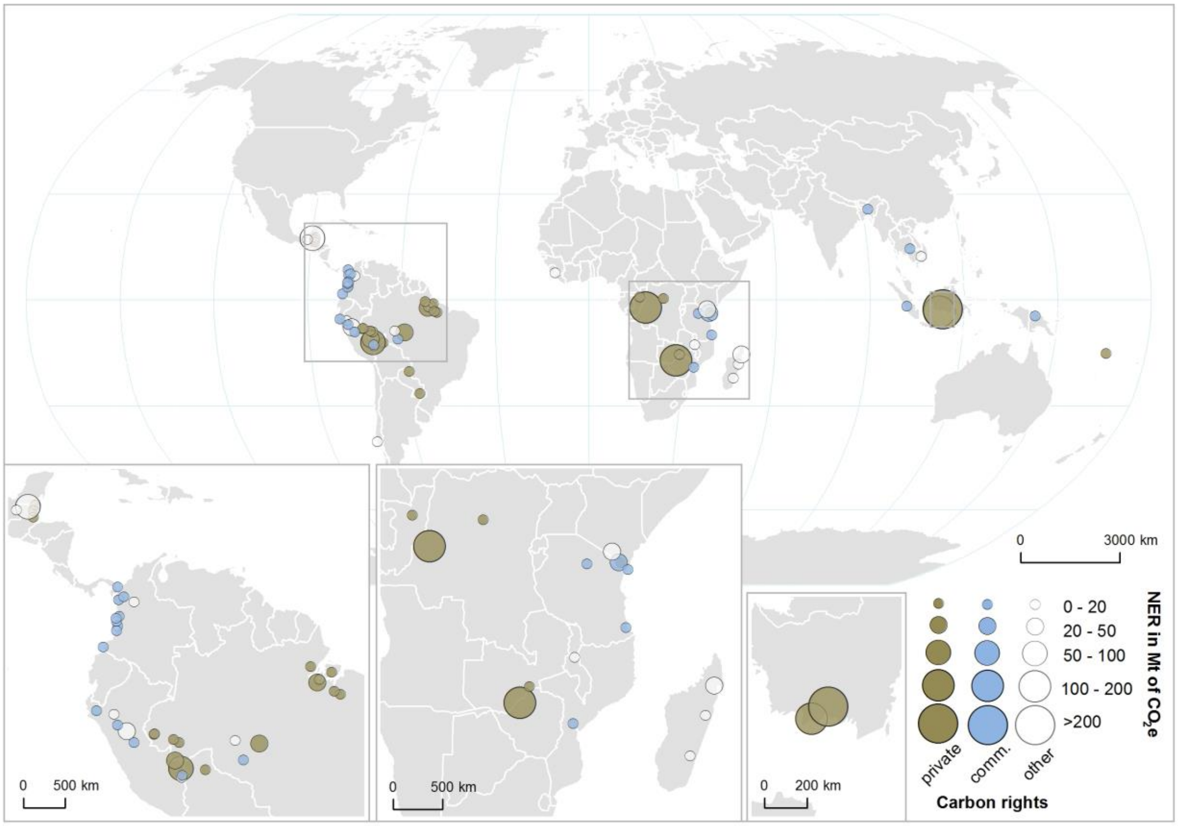


Figure 4.2: Location of REDD+ projects validated by major carbon certification standards (n = 66, as of September 10, 2016). Size of circles indicates the expected net emission reductions (NER) in $MtCO_2e$ as reported by REDD+ projects. Colours illustrate the holder of carbon rights: private, communities, other (including NGOs, governments, multiple/trust). Particularly in large REDD+ projects (NER > 100 $MtCO_2e$) carbon rights are owned by private stakeholders.

4.2.2 Net emission reductions (NER) reported by REDD+ projects

Projects report NER in tCO_2e expected within the project area (in hectares) during a defined carbon accounting period (in years) (Supplementary Table 4.2). As measure for carbon performance of REDD+ projects, NER were standardized by converting reported emission reductions into NER per hectare per year (NER_{std} in $tCO_2e\ ha^{-1}\ yr^{-1}$). Projects estimate NER *ex ante* comparing business-as-usual scenarios for changes in forest carbon stocks without REDD+ interventions with scenarios with REDD+ interventions. Emission reductions can be over- or underestimated due to a) uncertainties in measuring and monitoring changes in carbon stocks and b) uncertainties in scenarios for the expected impacts of REDD+ activities. Actual emission reductions as result of project implementation are measured and monitored throughout the carbon accounting period.

VCS and CCB projects (n = 60) estimate expected net emission reductions following carbon accounting guidelines established by the Verified Carbon Standard (VCS) (VCS methodologies VM0004, VM0007, VMD0006, VM0009, VM0010, VM0011, VMD0015). Projects certified by Plan Vivo (n= 6) follow different procedures of estimating and reporting emission reductions, which can lead to inconsistencies in carbon accounting. Projects of all three carbon standards were included, since carbon credits certified by these standards are assumed to be additional based on best knowledge currently available.

Projects report net emission reductions after deducting emissions caused by project implementation and leakage of emissions due to the displacement of deforestation drivers. For example, for projects avoiding planned deforestation guidelines by VCS recommend up to 40% deduction from gross emission reductions in order to account for potential leakage (VCS VM0004). This value is adjusted on a case-by-case basis. Hence, net emission reductions reported by REDD+ projects are considered to be conservative estimates. However, leakage is mainly accounted for at local to national scale but not at international scale. This can be a potential source of error since displacement of deforestation is occurring also at international scale related to trade of commodities (Meyfroidt & Lambin 2009; DeFries et al. 2010; Leblois et al. 2017; DeFries et al. 2013).

In order to account for potential risks of not achieving the expected emission reductions, projects estimate a risk buffer in percent of NER. The carbon accounting rules by VCS require a minimum risk buffer of 10% of NER, which are non-tradable carbon credits that need to be kept in reserve. If carbon monitoring reveals that expected net emission reductions are not achieved, carbon credits of the risk reserve are cancelled. For the projects included in my analysis, 60 out of 66 projects report a risk buffer (Supplementary Table 4.2). The average risk buffer of the analysed REDD+ projects is 16%. In the regression analysis NER_{std} values were used that include the risk buffer, as these values represent the emission reductions expected to be achieved.

There is also the possibility that the analysed REDD+ projects are underestimating expected net emission reductions, since 19 projects report NER only for the first 10 years and expect additional emission reductions throughout the project lifetime of up to 60 years.

4.2.3 Testing hypotheses on variables explaining variance in expected carbon performance of REDD+ projects

For testing hypotheses on variables explaining the variance in expected carbon performance of REDD+ projects, information on A) biophysical context, B) socio-economic context and C) project design was extracted from project design documents and from global data sets (Fig. 4.3). Supplementary Table 4.3 contains a detailed description of the variables included in the analysis.

Qualitative information provided by project design documents was coded as binary categorical variables (Yes/No), e.g. the type of forest management strategies applied in REDD+ activities. For assessing the environmental co-benefits provided by forests in the project areas, the project design documents were analysed for the benefits that stakeholders derive from ecosystems within the project area (e.g. the use of land and forest). The reported benefits were classified in accordance with the ecosystem service categories of The Economics of Ecosystems and Biodiversity (TEEB 2010b) and recorded the total number of ecosystem service categories for the project area (ES_{total}) (Supplementary Table 4.3, variable 27). Given that REDD+ projects report ecosystem benefits mainly in qualitative terms, the variable ES_{total} only indicates the diversity of environmental co-benefits derived from forest landscapes. It neither allows for conclusions on the quality and quantity of benefits provided by forest landscapes nor for conclusions on environmental impacts resulting from the use of forest landscapes. There are likely to be differences between projects in the emphasis of reporting ecosystem benefits. However, since all carbon standards require projects to report the use of land and forests within the project area, the differences in reporting environmental co-benefits are likely to be minor.

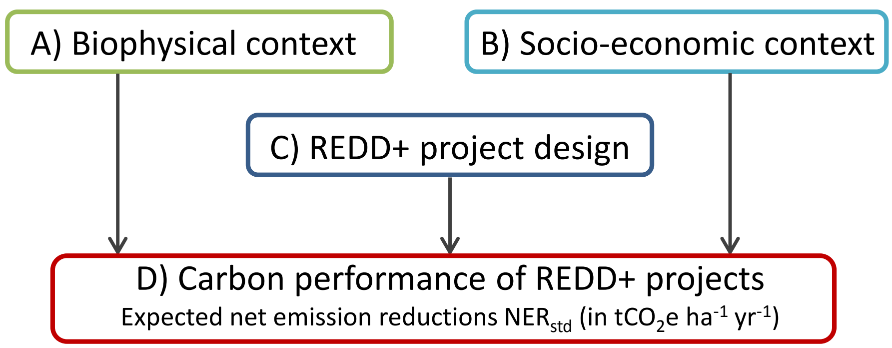


Figure 4.3: Conceptual design for assessing factors associated with the carbon performance of REDD+ projects. Multiple linear regression analysis was used for testing hypotheses for variables of A) biophysical context, B) socio-economic context, and C) project design explaining the variance in D) the expected carbon performance of REDD+ projects (NER_{std} in tCO_2e ha^{-1} yr^{-1}). Supplementary Table 4.3 provides a detailed description of the variables.

4.2.4 Selection of independent variables

For the statistical analysis of factors explaining the variance in carbon performance of REDD+ projects in NER_{std} in tCO_2e ha^{-1} yr^{-1}, a total of 27 explanatory variables were assessed, using multiple linear regression analysis (Supplementary Figure 4.1). The variables were selected based on hypotheses for their relationship with expected carbon performance of REDD+ projects (Supplementary Table 4.3). All statistical analyses were done using the R software (The R Foundation for Statistical Computing, Version 3.3.1, 2016).

Only non-collinear variables of A) biophysical context, B) socio-economic context, and C) project design were included in the regression analysis (Pearson’s $r > -0.5$ and $r < 0.5$; Pearson's x^2-test $p > 0.05$; Welch’s two-sample t-test $p > 0.05$). For assessing collinearity of continuous variables, the Pearson’s correlation coefficient (r) was determined using R function cor() of package *stats* (version 3.3.1) (Supplementary Figure 4.2). Collinearity of categorical variables was assessed using the Pearson's x^2-test (R function chisq.test () of package *stats* version 3.3.1) (Supplementary Table 4.4). Collinearity of pairs of categorical and continuous variables was assessed using the Welch’s two-sample t-test (R function t.test () of package *stats* version 3.3.1) and boxplots (R function boxplot () of package *graphics* version 3.3.1).

4.2.5 Assessing factors explaining expected carbon performance

As indicator for the expected carbon performance of REDD+ projects, I standardized the expected net emission reductions for hectare and year (NER_{std} in $tCO_2e\ ha^{-1}\ yr^{-1}$), allowing for cross-project comparison. Using multiple linear regression analysis, I assessed the dependence of expected carbon performance of REDD+ projects on a total of 27 variables for the A) biophysical and B) socio-economic context of the project area and C) for the project design (Fig. 4.3, Supplementary Figure 4.1, Supplementary Table 4.3).

4.2.6 Multiple linear regression analysis

The multiple linear regression model that would best explain the variance in expected carbon performance of REDD+ projects was selected based on the Akaike information criterion (AIC). Due to the large number of explanatory variables, the regression analysis followed a tiered approach. First, I used regression analysis for testing hypotheses with variable combinations that belong to only one of the three groups: A) biophysical context, B) socio-economic context and C) project design (Supplementary Figure 4.1 and Supplementary Table 4.5 to 4.7). Second, variables of the three groups were combined for testing hypotheses explaining carbon performance of REDD+ projects (Supplementary Table 4.8). Finally, the regression model with the variable combination yielding the lowest value for AIC was identified (Fig. 4.5, Supplementary Table 4.9). For visualizing the output of the regression model, I used the R function visreg () of package *visreg* (version 2.3).). As there was no data available for the Net Primary Productivity (NPP) of the project area for the project with ID 66, it was excluded from the multiple linear regression analysis. I declare that the data supporting the findings of this study are available within the paper and its supplementary information files.

4.3 Results and discussion

4.3.1 Expected contribution to climate change mitigation

The 66 REDD+ projects expect to conserve 9.1 million hectares of forests, an area almost the size of Portugal, with total net emission reductions of 1.6 $GtCO_2e$ (16% risk of non-permanence) (Fig. 4.4, Supplementary Table 4.2).

For the year 2015, REDD+ projects reported total net emission reductions of 0.05 $GtCO_2e$, corresponding to about 1.29% of global annual CO_2 emissions from deforestation (4.03 ± 1.83 $GtCO_2e\ yr^{-1}$) (Smith et al. 2014). If the Nationally Determined Contributions (NDCs) of Parties to the Paris Agreement for reducing carbon emissions were to be achieved, they would still fall short of the global emission reductions required for staying below the 2°C and 1.5°C targets (UNEP 2016). This emission gap is quantified to be 12 to 15 $GtCO_2e$ for reaching an emission pathway by 2030 that would allow meeting the 2°C and 1.5°C target with a 66% to 50% probability. Until 2030, the 66 REDD+ projects expect emission reductions of 0.95 $GtCO_2e$, corresponding to 7.9% and 6.3% of emission reductions required for closing the emission gap of the respective target. However, the reporting of NDCs remains vague on the extent to which REDD+ activities are included (Grassi et al. 2017). Therefore, more transparent reporting of emission reductions originating from REDD+ activities is needed in the NDCs in order to avoid double counting of REDD+ contributions to the targets of the Paris Agreement.

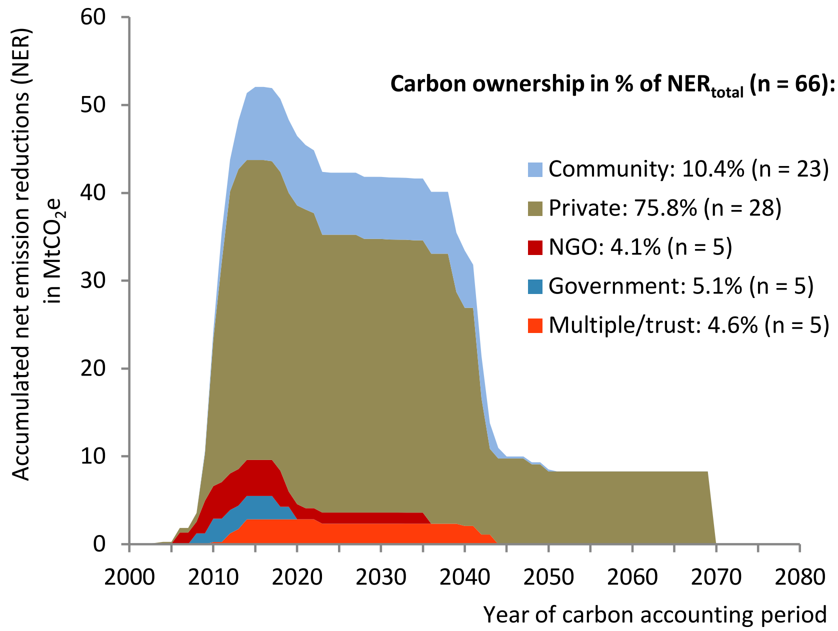


Figure 4.4: Expected net emission reductions (NER) of REDD+ projects (n = 66) accumulated over the carbon accounting period and according to ownership of carbon rights (colours) and percentage of total expected net emission reductions (NER_{total} = 1.6 $GtCO_2e$). 19 REDD+ projects report NER only for the first 10 years of the carbon accounting period, explaining the peak in 2015 - 2016, when all projects report emission reductions. Private actors hold the carbon rights for the majority (75.8%) of total net emission reductions.

4.3.2 National context and project design determine carbon performance

From the 27 variables four variables were identified that parsimoniously explain 78% of the variance in expected carbon performance of REDD+ projects (NER_{std} in tCO_2e ha^{-1} yr^{-1}). The expected carbon performance is positively associated with (1) national deforestation rate (1990-2005), (2) national government effectiveness index, and (3) avoiding planned deforestation (APD) as forest management strategy. However, carbon performance is negatively related to (4) the number of environmental co-benefits reported for the project area (ES_{total}) (Fig. 4.5, Supplementary Table 4.9).

I show that projects in countries with high historical deforestation rates (1990–2005) and a high score in government effectiveness expect to achieve high net emission reductions per hectare per year (Fig. 4.5). Government effectiveness is known to explain the location of REDD+ initiatives (Cerbu et al. 2011) with my findings emphasizing the importance of governance also for the carbon performance of REDD+ initiatives. Strengthening forest governance at national level is likely to benefit REDD+ (Minang & van Noordwijk 2013; Vatn & Vedeld 2013), in particular if integrated into existing land-use policies (Ezzine-de-Blas et al. 2016).

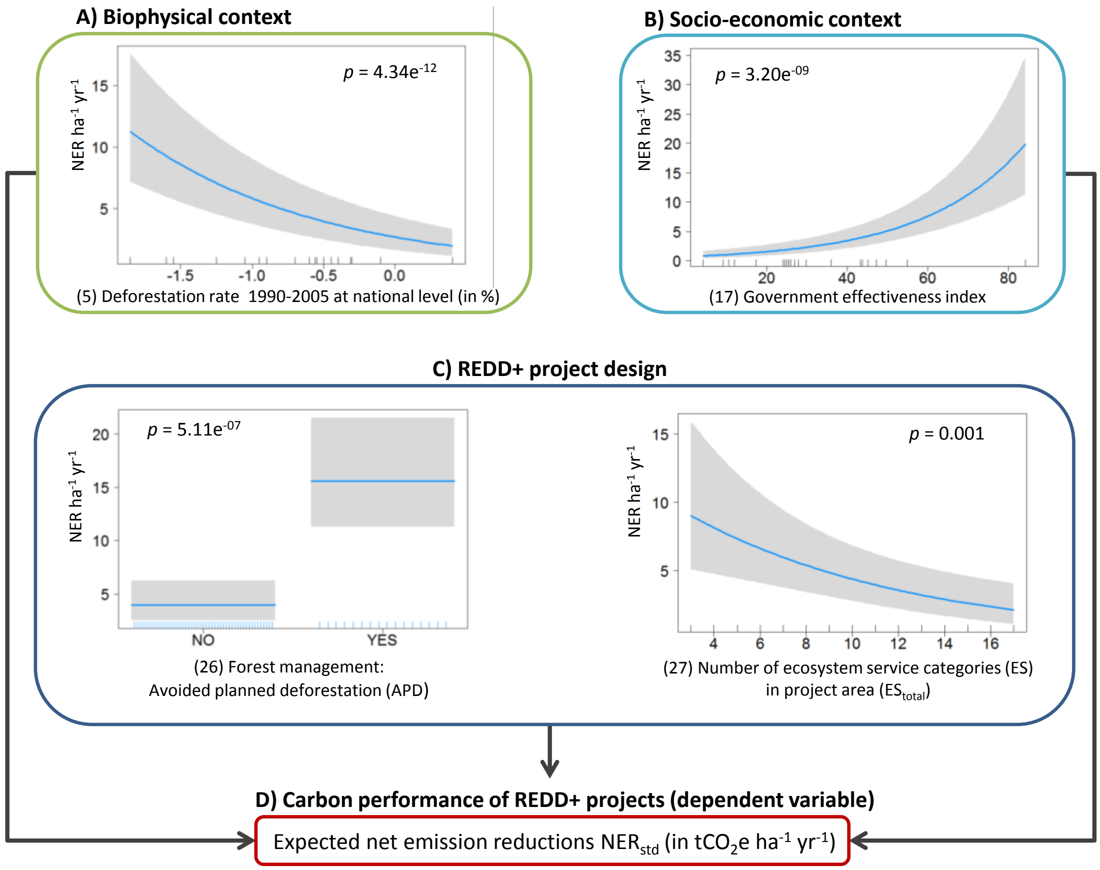


Figure 4.5: Result of the multiple linear regression model with four variables explaining 78% of the variance in expected carbon performance of REDD+ projects (NER_{std} in tCO_2e ha^{-1} yr^{-1}). Grey shaded areas denote 95% confidence interval. Tick marks at x-axis represent project data.

4.3.3 Targeting legal drivers of deforestation is highly carbon effective

Projects designed for avoiding planned deforestation (APD) represent only 12.8% of the total forest area of the analysed REDD+ projects (n = 66) but expect to generate 52.2% of total net emission reductions. This includes projects turning legal logging concessions or concessions for palm oil plantations into conservation concession. Projects avoiding planned deforestation expect to achieve significantly higher net emission reductions per hectare per year (median NER_{std} = 17.2 tCO_2e ha^{-1} yr^{-1}; mean NER_{std} = 22.5 tCO_2e ha^{-1} yr^{-1}) than non-APD projects (median NER_{std} = 4.7 tCO_2e ha^{-1} yr^{-1}; mean NER_{std} = 5.6 tCO_2e ha^{-1} yr^{-1}). From the 27 variables included in the regression analysis, avoiding planned deforestation has the largest statistical effect on the expected carbon performance of REDD+ projects (Fig. 4.5, Supplementary Table 4.9).

For example in Borneo, Indonesia, two projects (Supplementary Table 4.2, project ID 13 and 27) avoid the conversion of highly carbon-rich peat land forests to palm oil plantations. Together they account for 38.4% of total expected net emission reductions with a carbon performance of NER_{std} = 92.5 tCO_2e ha^{-1} yr^{-1} and NER_{std} = 55.3 tCO_2e ha^{-1} yr^{-1} respectively. When excluding these two projects from the regression analysis, avoiding planned deforestation still remains a significant variable ($p = 1.4e^{-04}$), confirming the importance of avoiding planned deforestation in explaining carbon performance.

The large share of emission reductions originating from avoiding planned deforestation highlights the role of reducing legal deforestation in current REDD+ activities. It also supports concerns that REDD+ activities are less effective in addressing illegal and indirect drivers of deforestation (Weatherley-Singh & Gupta 2015; Pasgaard et al. 2016; Mertz et al. 2012). Avoiding planned deforestation usually involves a change in the legal status of forest, changing concessions for the production of commodities (e.g. forest and agricultural products) into conservation concessions (e.g. Supplementary Table 4.2, project ID 13 and 27). For ensuring the permanence of this change, effective governance across sectors is critical (Vatn & Vedeld 2013; Ravikumar et al. 2015; Sunderlin et al. 2015), in particular for addressing deforestation drivers related to the demand and supply of commodities (Weatherley-Singh & Gupta 2015).

4.3.4 High risk of displacement of deforestation due to commodity trade

Private actors favour the highly carbon effective strategy of avoiding planned deforestation, with 14 out of 16 projects avoiding planned deforestation being located on private forest land. This indicates that REDD+ policies can provide an alternative business model to the production of commodities linked to deforestation (e.g. production of timber and palm oil). However, the strategy of avoiding planned deforestation involves the risk of simply displacing drivers of deforestation by shifting the sourcing of commodities driving deforestation between countries and regions (Meyfroidt & Lambin 2009; DeFries et al. 2010; Leblois et al. 2017; DeFries et al. 2013). The effect of displacing deforestation and related carbon emissions from one place to another is known as leakage. Currently, REDD+ projects account for leakage mainly at local and national scales. However, in order to account for the risk of leakage due to international trade of commodities, carbon accounting for REDD+ has to address leakage also at a global scale. Furthermore, REDD+ projects have to be accompanied by policies at an international level for avoiding the displacement of deforestation drivers across countries and regions (Nepstad et al. 2013; Broekhoven & Wit 2014; le Polain de Waroux et al. 2016).

4.3.5 Trade-offs in carbon performance and environmental co-benefits

For assessing the role of environmental co-benefits in the performance of REDD+ projects, the environmental benefits reported for the project area were analysed and classified these according to ecosystem service categories (ES_{total}). The analysis reveals that REDD+ projects reporting a high number of ecosystem service categories, i.e. with a high diversity of environmental co-benefits, expect a lower performance in reducing carbon emissions (Fig. 4.5, Supplementary Table 4.9). This trade-off is pronounced in particular for differences in land ownership. Projects with communities as land owners report a higher number of ecosystem service categories for the project area (median 12 ES_{total}) but achieve lower emission reductions (median NER_{std} = 4.8 $tCO_2e\ ha^{-1}\ yr^{-1}$; mean NER_{std} = 5.8 $tCO_2e\ ha^{-1}\ yr^{-1}$) than projects with private land owners (median 9 ES_{total}; median NER_{std} = 5.2 $tCO_2e\ ha^{-1}\ yr^{-1}$; mean NER_{std} = 12.4 $tCO_2e\ ha^{-1}\ yr^{-1}$).

Differences in motivation for engaging in REDD+ activities and in criteria for site selection are possible reasons for these trends. While private investors are likely to base site selection for REDD+ projects on criteria of maximizing income from tradable carbon credits (Corbera 2012), communities are likely to engage in REDD+ projects for securing access to multiple benefits. However, empirical evidence has shown that overloading the design of payments for ecosystem services with multiple objectives for environmental and social-economic outcomes can undermine their efficiency of delivering on their main objective (Wunder et al. 2008). Not all desired co-benefits are beneficial for maximizing carbon storage in forests and trade-offs in delivering carbon performance and environmental and social co-benefits through REDD+ are to be expected (Bustamante et al. 2014; Phelps et al. 2012; Budiharta et al. 2014).

Nonetheless, the design of REDD+ strategies can benefit from targeting multiple ecosystem services. Projects with a focus on ecosystem services are found to better integrate land-use sectors, such as forestry and agriculture, in the design of conservation strategies (Goldman et al. 2008). The focus on multiple ecosystem services can help to reconcile competing interests of forest stakeholders and contribute to mitigating drivers of deforestation and degradation. Furthermore, not all projects are implemented with the goal of maximizing carbon performance. For example, projects with pre-existing protected areas for biodiversity conservation (n = 17) cite the prospect of generating additional funding through carbon finance as motivation for engaging in REDD+ activities. In this case, biodiversity conservation is the main objective, with REDD+ being a means of financing conservation.

4.3.6 Privatization of carbon rights

Ownership of land tenure and carbon rights are identified to be underlying factors explaining patterns in expected emission reductions (Fig 4.6). While projects with private land tenure generate 42.4% of total expected net emission reductions, the carbon rights of 75.8% of total expected net emission reductions are privately owned (Fig. 4.6a, Supplementary Table

4.10). Private actors prefer to engage in REDD+ projects with a high carbon performance, as projects on private land or with private carbon ownership have a significantly higher carbon performance than other projects (Fig. 4.6b and 4.6c). Furthermore, private actors are successful in acquiring carbon rights of REDD+ projects situated in forests owned by governments or communities: while land tenure is private in 21 out of 66 REDD+ projects, private actors hold carbon rights of 28 REDD+ projects. In comparison, projects on community-owned land (n = 20) generate 19.5% of total expected net emission reductions, while communities own carbon rights to only 10.4% of total net emission reductions. Similarly, projects on government land (n = 13) generate 19.0% of total expected net emission reductions, while only 5.1% of carbon rights are in government ownership (Fig. 4.6a, Supplementary Table 4.10).

A plausible explanation for the dominance of private ownership of carbon rights is that private actors gain access to carbon rights in return for financial investment in designing and implementing REDD+ projects. This indicates that REDD+ policies are successful in promoting private sector engagement. However, the findings also show that REDD+ projects lead to a privatization of forest carbon with governments and communities potentially losing out on benefits (Bottazzi et al. 2014; Nhantumbo & Camargo 2013; Howe et al. 2014). Therefore, safeguards could be important for ensuring equitable access and benefit sharing.

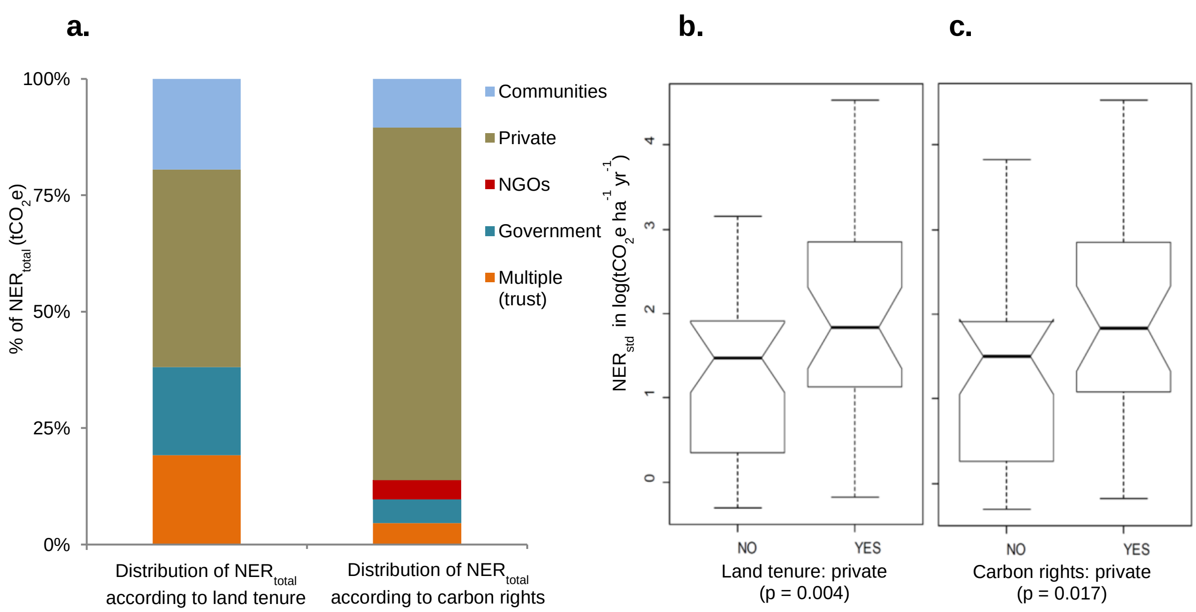


Figure 4.6: Distribution of expected net emission reductions (NER) according to land tenure and carbon rights. **a.** Percentage of total net emission reductions (NER_{total}, n= 66) distributed according to land tenure (left bar) and carbon rights (right bar). Projects with private land tenure contribute 42.2% to NER_{total} (left bar). 75.8% of carbon rights are in private ownership (right bar). The large share of carbon rights in private ownership can be explained by a privatisation of carbon rights originating from community-owned land, government land and land under mixed tenure regimes.
b. Projects with private land tenure and **c.** projects with private owners of carbon rights expect to achieve significantly ($p < 0.05$) higher net emission reductions per hectare and year (NER_{std} in $\log(tCO_2e\ ha^{-1}\ yr^{-1})$) than other projects.

4.3.7 REDD+ safeguards can benefit carbon performance

Benefit sharing in form of direct monetary payments is uncommon in REDD+ projects with only 16 out of 66 REDD+ projects (24%) providing monetary payments to local communities. For REDD+ projects with private carbon rights, only 7 out of 28 projects (25%) share benefits through monetary payments. Hence there is the risk that REDD+ projects promote elite capture and increase inequalities (Bottazzi et al. 2014). In Brazil, for example, private landholders with large forest areas are likely to benefit most from payments for ecosystem services such as REDD+ (Richards & VanWey 2015).

The majority of REDD+ projects claim to provide benefits to local communities through securing tenure rights: 43 out of 66 REDD+ projects (65%), including 14 projects with private carbon rights, report to secure or strengthen land tenure rights of local communities. Addressing development priorities and livelihood needs of local communities, including land tenure security, can indeed contribute to mitigate local drivers of deforestation (Bottazzi et al. 2014; Atela et al. 2015). Furthermore, community-based forest management is often associated with low deforestation rates (Chhatre & Agrawal 2009; Bottazzi et al. 2014; Porter-Bolland et al. 2012). It has been argued that securing land tenure of communities secures multiple forest services, including carbon sequestration, with benefits for communities and private investors (Ding et al. 2016). However, strengthening property rights alone is not sufficient for reducing deforestation and local institutions with strong enforcement capacity and environmental norms are needed (Bottazzi et al. 2014). Otherwise there is the risk that REDD+ projects can promote the enclosure of common forest land with the potential of triggering conflicts over forest resources (Scheba & Rakotonarivo 2016).

Mitigating potential conflicts and trade-offs should be addressed from the outset of project design instead of expecting win-win situations to occur by default (Howe et al. 2014). Including communities in the design process is found to positively influence the performance of payments for ecosystem services and can reduce trade-offs (Wunder et al. 2008; Chhatre et al. 2012). Therefore, striking a balance between efficiency in reducing carbon emissions and securing the needs of local communities is critical for the success of REDD+ (Leimona et al. 2015; Bottazzi et al. 2014; Ituarte-Lima et al. 2014). Implementing REDD+ safeguards for securing tenure rights of communities, ensuring equitable access to carbon rights and establishing benefit sharing could promote efficient and fair REDD+ projects. As the results show that effective governance is explaining high carbon performance of REDD+ projects, strengthening governance is likely to benefit both carbon performance and REDD+ safeguards for reducing trade-offs. Improving governance is also critical for ensuring permanence of emission reductions. This is in particular true for over half of total net emission reductions originating from avoiding planned deforestation, as these REDD+ projects rely on the assumption that the change in legal status from concessions for logging or palm oil plantations to conservation concessions is actually enforced.

4.4 Conclusions

Currently, there is no systematic accounting of the contribution of REDD+ towards achieving the climate change mitigation targets of the Paris Agreement (Grassi et al. 2017). This analysis provides a baseline for expected net emission reductions reported by REDD+ projects validated by major carbon standards. Over the next decades, and throughout the lifetime of REDD+ projects, monitoring of the actual carbon performance is required for confirming emission reductions. My finding that half of emission reductions of REDD+ projects are expected from avoiding legally planned deforestation in forest areas used for commodity production highlights that there is a significant risk of displacing deforestation drivers (leakage) due to trade of commodities. REDD+ monitoring will have to account for this potential leakage not only at national but also at regional and global scales. For ensuring that REDD+ projects generate truly additional emission reductions, REDD+ policies will have to address also deforestation drivers linked to the demand and supply of forest and agricultural commodities.

Finally, as net emission reductions reported in the design documents of REDD+ projects and validated by carbon standards are currently the only available data on the carbon performance of REDD+ projects, this assessment can inform the Nationally Determined Contributions of the Parties to the Paris Agreement on emission reductions originating from REDD+ projects. The collected data can be revised and updated as confirmed estimates for net emission reductions become available throughout the lifetime of REDD+ projects.

5 Synthesis and discussion

Despite the increasing focus in environmental science on assessing the benefits nature provides to society - commonly referred to as ecosystem services (Vihervaara et al. 2010; Seppelt et al. 2011; Abson et al. 2014), there is little evidence on how scientific information on ecosystem services informs decision making (Laurans et al. 2013) and whether the generated knowledge is actually relevant for decision makers (Honey-Rosés & Pendleton 2013; Martinez-Harms et al. 2015).

With this dissertation, I identify and address key research questions related to the assessment of ecosystem services for informing decision making on sustainable land management under climate change. The research findings derived from this dissertation will hopefully contribute to making ecosystem service science more relevant for decisions on sustainable land management and related climate policies, by:

1. providing a better understanding of the scientific knowledge on the monetary value of ecosystem services available for decision support in Germany (Chapter 2).

2. advancing the conceptual design of ecosystem service assessments through proposing a problem-oriented approach that allows targeting decision relevant ecosystem service information (Chapter 3).

3. informing the design of the REDD+ policy on trade-offs between carbon sequestration and other ecosystem services, the need for safeguards for ensuring access rights of forest communities to carbon rights, and the need for better monitoring and accounting of displacement of deforestation caused by international commodity trade (Chapter 4).

4. demonstrating the use of methods for the synthesis, integration and meta-analysis of qualitative and quantitative information on ecosystem services for informing science-policy processes (Chapter 2: the use of a database as decision support tool; Chapter 3: the synthesis of conceptual approaches and guidance for ecosystem service assessments; Chapter 4: informing policy design using multiple linear regression analysis).

In the following, the research findings of this dissertation are discussed in more detail.

5.1 Review of monetary valuation studies of ecosystem services for informing decision making in Germany

Germany faces on-going degradation and loss of biodiversity and related ecosystems (Niedertscheider et al. 2014). As a consequence, goods and services provided by biodiversity to human well-being, the so-called ecosystem services, are lost. While the ecological impacts of ecosystem conversion are known, they are often ignored and neglected in decision making. Therefore, integrating information on the costs of ecosystem service loss into decisions on land-use planning and policy design is regarded to be critical for strengthening economic arguments for avoiding biodiversity loss and degradation (TEEB 2010, Naturkapital Deutschland – TEEB DE 2012).

For addressing the question of what information on monetary values of ecosystem services is available for Germany and how it can be used for informing decision making on policies (research question 1), a literature review of monetary valuation studies of ecosystem services was conducted. Together with experts from the German Federal Environment Agency Umweltbundesamt (UBA), a set of criteria were identified for synthesizing information of monetary value studies of ecosystem services according to their relevance for informing decision making on national policies. Information on monetary values of ecosystem services highly depends on study design including the biophysical and social-ecological context of the study site and the choice of valuation methods and indicators. Therefore, it was critical to not only record monetary values of ecosystem services but also qualitative information related to study context, design, indicators and methods. This allows for interpreting the recorded monetary values on ecosystem services in light of the original valuation studies and judging their suitability, credibility, and reliability for informing a particular decision. This transparency on data quality is important for identifying opportunities and limitations of using ecosystem service information in decision making.

As result of this approach, a database was created that allows accessing information on monetary values of ecosystem services together with qualitative information on study design, context, indicators and methods (the database is provided on CD as part of this dissertation). This database is the first systematic compilation of monetary values for ecosystem services in Germany and can serve as a resource and tool for supporting the integration of information on ecosystem services in decision making. In total, 109 monetary valuation studies of ecosystem services were identified for ecosystems in Germany with the majority focusing on forests and wetlands. Few studies relate to grasslands although this ecosystem experiences the greatest loss (Tietz et al. 2012). Monetary values for regulating and cultural ecosystem services are scattered and scarce compared to information on provisioning services, which is accounted for in detail in national statistics. This imbalance in information likely contributes to the distortion in land-use policies, giving preference to maximizing provisioning services in agricultural production and forestry, while neglecting the societal relevance and preferences for maintaining regulating and cultural services. Due to imports of commodities from tropical forest regions, deforestation of tropical forests and related loss of ecosystem services is also of relevance for decision making in Germany (Kissinger et al. 2012, Schmitz et al. 2015, Liu et al. 2016). Therefore, monetary valuation studies of ecosystem services for tropical forests were included in the database based on already existing databases and literature reviews.

Overall, only 6 out of 109 studies (5.5 %) comply with selection criteria relevant for informing national policies targeted by the methodological convention of the German Federal Environment Agency Umweltbundesamt (UBA) (Fig. 2.6 and Table 2.1). These criteria aim at ensuring comparability of monetary values across single studies in order to base decision making on a range of monetary values that reflect diversity in biophysical properties and social-ecological contexts. These selection criteria include that monetary valuation studies are sufficiently transparent and robust with regards to study design and valuation methods, that monetary values are reported in common and comparable units (e.g. €/ha), and that the minimum-maximum ranges in monetary values are explained by measurable changes in biophysical or socio-economic indicators (Fig. 2.6).

Chapter 2 shows that few scientific studies use indicators for regulating and cultural ecosystem services that are relevant for informing policies at national level. Often study design and indicators relate to particular local interests that are not compatible with information needs at national level. While it is certainly meaningful to target a particular decision context for producing decision-relevant outcomes (Förster et al. 2015), the design of ecosystem service assessments should also be compatible with indicators across multiple scales in order to allow their use in meta-analyses for informing decision making (e.g. Gerstner et al. 2017).

Due to the small number of monetary valuation studies identified to be of relevance for decision making at national level, decision makers have to account for the trade-off in relying on few cost estimates that are scientifically robust, while being pragmatic enough to include also vague estimates in cases where data is lacking. This highlights the need for ecosystem service valuation studies to include indicators that are relevant for particular decision contexts but also of relevance across multiple scales.

One way of enhancing the compatibility of studies is to better align biophysical and socio-economic indicators on ecosystems services so that they better complement each other when used in monetary valuation (e.g. ensure that biophysical indicators can be converted to units per hectare or that socio-economic indictors allow for conversion to units per capita). This could allow associating, for example, spatial changes in biophysical and socio-economic contexts to changes in the economic significance of ecosystem services.

For achieving greater relevance of ecosystem service valuation studies for decision making across local and national scales, potential users of ecosystem service information should be involved in the identification and development of ecosystem services indicators. Thereby, the study design should target a clearly defined problem that is of concern for decision makers and that is of relevance for decision making processes (e.g. environmental impact assessments). However, it is important to be aware that monetary estimates provide only a partial representation of ecosystem benefits and that decision makers do not want to rely only on economic information (Ruckelshaus et al. 2015). Therefore, decision making should account for the multiple values biodiversity and ecosystems provide to human wellbeing and not only rely on monetary values. These multiple values includes information, for example, on the value of an area for biodiversity conservation (e.g. for threatened species), on the relevance of biodiversity and ecosystems for the identity of people, for spiritual and cultural values, and information on the existence value of nature. In summary, Chapter 2 provides the first systematic review of monetary values of regulating and cultural ecosystem services for Germany and informs the German Federal Environment Agency Umweltbundesamt (UBA) on the suitability of monetary values of ecosystem services for informing policies at national level. The review highlights knowledge gaps in particular related to monetary values for regulating and cultural ecosystem services. The small number of monetary valuation studies with relevance for decision making on national policies emphasizes the need for scientific studies to better address indicators with relevance across multiple scales. This does not only include information on monetary values of ecosystem services, but also information on the multiple other values biodiversity and ecosystem services provide to society. Options for ensuring that ecosystem service assessments are more decision relevant are investigated in Chapter 3 of this dissertation.

5.2 Target information needs by decision makers using a problem-oriented approach for ecosystem service assessments

The majority of ecosystem service assessments tend to generate knowledge on ecological functions and economic values (Abson et al. 2014) with little consideration of the information demand by decision makers for addressing a particular land-use problem (Honey-Rosés and Pendleton 2013). For example, only 8 out of 340 cases of ecosystem service valuation published in scientific literature actually report how the information on values of ecosystem services is used in local decision making (Laurans et al. 2013). In Germany, only 6 out of 109 (5.5 %) studies with monetary valuation of ecosystem services comply with selection criteria relevant for informing national policies (Förster et al. 2017). Therefore, ecosystem service assessments have not yet proven to effectively change land management and policies in public and private sectors (Abson et al. 2014, Ruckelshaus et al. 2015).

For addressing the question of what gaps exist in current ecosystem service frameworks and how the design of ecosystem service assessments can be improved in order to increase their relevance for decision making (research question 2), Chapter 3 compares existing frameworks targeted at assessing ecosystem services in social-ecological systems (Fig 3.2). This review identified that most conceptual frameworks for assessing ecosystem services lack explicit guidance on tailoring ecosystem service assessments to the information needs of decision makers (depicted by the gaps in Steps 1-3 of “Scoping phase A” on the left side of Fig. 3.2). Only three out of eight frameworks include a focus on decision relevant problems (TEEB 2012, Chan et al. 2012a, Martinez-Harms et al. 2015) and emphasize the need for a stakeholder-driven process of problem identification. Furthermore, Martinez-Harms et al. (2015) note that only 8% of case studies use stakeholder consultations during the process of problem identification.

For closing the identified gaps, Chapter 3 proposes a problem-oriented approach for assessing ecosystem services (Fig. 3.2), which is derived from the experience of four case studies in Brazil, China, Madagascar, and Vietnam (Fig. 3.1). Like this dissertation, these case studies are part of the Sustainable Land Management (SLM) Program, funded by the German Federal Ministry for Education and Research (BMBF), with the objective of fostering transformations toward more sustainable land stewardship (Eppink et al. 2012). The conceptual considerations build on insights gained during my involvement in the TEEB initiative, which included the assessment of case studies with a focus on how to ensure that information on ecosystem services is relevant for decision making (Russi et al. 2013; TEEB case studies 2017). Furthermore, the process of developing the framework involved scientists with long-standing experience in collaborating with stakeholders in the particular study sites, which ensured the inclusion of context-specific and decision relevant information (Fig. 3.3 to 3.6).

The proposed approach in Chapter 3 comprises a scoping phase (A), assessment phase (B), and implementation phase (C), and follows 5 steps: (Step 1) specify and agree with stakeholders on the problems to be addressed, (Step 2) identify ecosystem service beneficiaries and ecosystem services most relevant to decision making, (Step 3) define information needs of decision makers, (Step 4) assess ecosystem service flow within the social-ecological context and the impact changes have on ecosystem service benefits and trade-offs, and finally (Step 5) synthetize and integrate the generated information into processes of decision support. The approach is not intended to replace the existing frameworks, but to provide complementary guidance for enhancing the relevance of ecosystem service assessments for decision making. Thereby it is critical to ensure that the information generated on ecosystem services is also viewed in light of other information with relevance for decision making concerning human-wellbeing.

Despite this proposed step-wise approach one has to be aware that processes of designing assessments with relevance for decision making are often time intensive involving multiple stakeholders in a dynamic process of co-design. This can be a messy process with unforeseen iterations due to changing stakeholder priorities or the adaptation of the research design to changes in environmental or socio-economic conditions. These dynamics are not reflected in the rather linear design of the proposed framework (Fig. 3.2). In reality, ecosystem service assessments might prioritize certain steps over others, the order of the assessment components can change, and multiple assessment processes run in parallel and with iterations (Berghöfer et al. 2016). Although ecosystem service assessments might divert from a step-wise structure, the questions and indicators identified for each component of the proposed approach (Table 3.1 to 3.3) can help in focusing ecosystem service assessments on decision relevant questions.

Although this approach of focusing on a clearly defined policy question has been developed mainly for place-based ecosystem service assessments, it was helpful for informing the meta-analysis conducted in Chapter 4 by ensuring its relevance for decision making and policy design. Chapter 4 also shows that information on ecosystem services is only one of many indicators that need to be considered when conducting decision-relevant assessments.

5.3 Informing the design of climate policies by assessing factors influencing the carbon performance of projects reducing carbon emissions from deforestation and forest degradation

Reducing emissions from deforestation and degradation in developing countries and promoting sustainable forest management for conserving and enhancing forest carbon stocks (REDD+) is a policy under the United Nations Framework Convention on Climate Change (UNFCCC). Reducing emissions from forest loss is regarded to be a critical contribution to achieving the Paris Agreement of holding global warming below 2°C compared to pre-industrial levels (UNFCCC 2008; UNFCCC 2015; Houghton et al. 2015; Grassi et al. 2017). The REDD+ policy is based on the principles of payments for ecosystem services (PES), a market-based policy instrument, whereby agreed-upon actors are compensated for environmental services they supply or manage (Wunder et al. 2008; Leimona et al. 2015). In REDD+, payments for forest carbon are an incentive for mitigating deforestation drivers (Weatherley-Singh & Gupta 2015), whereby avoided carbon emissions are traded as verified carbon units in carbon markets (Sandker et al. 2010).

While REDD+ offers multiple benefits for climate change mitigation, biodiversity conservation and development (Chhatre & Agrawal 2009; Olsson & Ouattara 2013; Bustamante et al. 2014; Gilroy et al. 2014; Labrière et al. 2015), trade-offs are to be expected (Bustamante et al. 2014; Phelps et al. 2012; Bottazzi et al. 2014). For example, there is concern that the categorization of forests into tradable carbon units promotes activities for maximizing carbon sequestration while undermining biodiversity conservation and excluding local communities from accessing forests for livelihood needs (Bottazzi et al. 2014; Corbera 2012; Pokorny et al. 2013). Therefore, REDD+ policies demand compliance with safeguards for the *“protection and conservation of natural forests and their ecosystem services”* and for ensuring *“full and effective participation of relevant stakeholders, in particular indigenous peoples and local communities”*(UNFCCC 2011). Hence a better understanding of factors determining carbon performance and related trade-offs can help achieving efficient and fair REDD+ projects.

For addressing the question of how the inclusion of multiple ecosystem services in the design of REDD+ projects impact their performance of reducing carbon emission from deforestation and forest degradation (research question 3), I conducted a meta-analysis of REDD+ projects. First, I quantified the net emission reductions (NER in tCO_2e) expected to be generated by REDD+ projects and second, I identified factors explaining the variance in the expected carbon performance of REDD+ projects (NER_{std} in $tCO_2e\ ha^{-1}\ yr^{-1}$) using multiple linear regression analysis (Rudel 2008). Among other factors, the meta-analysis included the total number of ecosystem service categories reported for the project area, serving as an indicator for the diversity of environmental co-benefits derived from the forest landscapes in REDD+ projects.

In order to ensure that the analysis addresses questions relevant for decision making, the selection of variables for the meta-analysis is informed by extensive discussions with experts working on the design of REDD+ policies. I followed multiple meetings on the development of REDD+ at Conferences of the Parties (COPs) to the UNFCCC and CBD and consulted experts from the World Bank, IUCN, and non-governmental organizations.

This includes also information from my previous work on assessing the potential for REDD+ policies on the ground in Western Ghana (Förster 2009; Sandker et al. 2010). These consultations together with my experience gained in the field ensured that the design of this meta-analysis is addressing key questions relevant for the design of the REDD+ policy. Besides defining a clear policy question, it was also ensured that relevant biophysical and socio-economic indicators were included as outlined in the assessment approach developed in Chapter 3. Such integrated assessments with a clear policy question are also in demand by interdisciplinary scientific journals that aim at publishing research with relevance for addressing real world problems, such as the journal Nature Climate Change.

The assessment identified 66 REDD+ projects validated by carbon standards (Fig. 4.1 and 4.2) that are estimating to conserve 9.1 million hectares of forests, which is equivalent to the area of Portugal, and are expecting net emission reductions of 1.6 $GtCO_2e$ (Fig. 4.4). The multiple regression analysis revealed that the carbon performance of these projects is positively associated with historical deforestation rates, governance effectiveness and project design for avoiding planned deforestation (Fig. 4.5). However, projects with multiple ecosystem services within their project area are related with lower emission reductions. This is likely due to multiple land uses within the project area that are generating multiple benefits in form of ecosystem services, but at the same time undermining the carbon storage in the forest.

Furthermore, the assessment identified that private stakeholders seem to benefit most from REDD+ policies. Private stakeholders favour projects with high carbon performance and carbon rights are private for 75.8% of total net emission reductions across all 66 projects (Fig. 4.4). Local communities are expected to gain land tenure security in 65% of projects, but hold carbon rights to only 10.4% of emission reductions. This emphasizes the need for safeguards that can ensure equitable access to benefits resulting from REDD+ projects.

One reason for the large share of private ownership in carbon rights is that private actors favour the highly carbon effective strategy of avoiding planned deforestation, with concessions for logging or palm oil plantations being converted into conservation concessions. This indicates that REDD+ policies can provide an alternative business model to the production of commodities linked to deforestation (e.g. production of timber and palm oil). However, the strategy of avoiding planned deforestation involves the risk of simply displacing drivers of deforestation by shifting the sourcing of commodities driving deforestation between countries and regions (Meyfroidt & Lambin 2009; DeFries et al. 2010; Leblois et al. 2017; DeFries et al. 2013). The effect of displacing deforestation and related carbon emissions from one place to another is known as leakage. Currently, the carbon monitoring of REDD+ projects accounts for leakage mainly at local and national scales. However, in order to account for the risk of replacing deforestation across countries due to international trade of commodities, carbon accounting for REDD+ has to address leakage also at a global scale. Furthermore, REDD+ projects have to be accompanied by policies at an international level for avoiding the displacement of deforestation drivers across countries and regions (Nepstad et al. 2013; Broekhoven & Wit 2014; le Polain de Waroux et al. 2016).
With these findings, the meta-analysis in Chapter 4 informs the design of REDD+ policies on the need for safeguards that address trade-offs between carbon sequestration and

ecosystem services from the multifunctional use of forests as well as the need for ensuring equitable access to carbon rights in particular for forest communities. Furthermore, it highlights that a large part of emission reductions of current REDD+ projects originate from reducing planned deforestation, which involves the risk of simply shifting deforestation to other countries and regions through international trade of commodities. This leakage effect has to be taken into account in carbon monitoring and when accounting the contribution of the REDD+ policy toward achieving the goals of the Paris Agreement for climate change mitigation.

5.4 Ensuring relevance of research questions for decision making through engagement in science-policy processes

For ensuring relevance of the research questions addressed in this dissertation for decision making, my engagement in interactive science-policy processes was critical. The process of identification of the research questions but also the research itself was part of science-policy processes and involved the consultation of key knowledge holders and stakeholders at relevant levels.

In Chapter 2 the relevance of the research for informing science and policy was ensured through the consultation of experts on ecosystem service valuation and of representatives from the German Federal Environment Agency Umweltbundesamt (UBA). This was critical for defining the criteria for identifying and selecting the monetary valuation studies that are of relevance for decision making. Chapter 3 is building on the lessons learned in stakeholder consultations conducted by the contributors from each of the projects. The synthesis of this experience in Chapter 3 allowed developing a more problem-oriented approach, which can help to ensure that assessments of ecosystem services better target information with relevance for decision making. The research question addressed in Chapter 4 was developed based on my consultations and exchange with stakeholders, including local land users in Ghana (Förster 2009; Sandker et al. 2010) and insights gained in international science-policy processes that focused on designing the REDD+ policies, including meetings under the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biological Diversity (CBD). The study design was also informed by discussions with scientists and representatives from the International Union for Conservation of Nature (IUCN), the World Bank, the initiative The Economics of Ecosystems and Biodiversity (TEEB) and from international non-governmental organizations. These consultations and the engagement in science-policy processes helped to target the research of this dissertation towards solving real-world problems concerning the design of policies for sustainable land management under climate change.

Decision-oriented research on ecosystem services for informing sustainable land management requires the engagement of scientists in science-policy processes. The outcomes of such science-policy driven research goes beyond scientific publications and can include the co-development of decision support tools for targeting very specific information needs of particular stakeholder groups. Ensuring a fruitful exchange of scientists with relevant stakeholders from policy and practice can promote the prioritization of decision-relevant research questions and the development of decision-oriented research designs. However, this often involves considerable amount of time, resources and creativity for developing methods and tools that are not commonly used in traditional science (e.g. involving indigenous knowledge on land-use practices). Therefore, the processes and outcomes of decision-relevant research should be appreciated by donors for research funding and by employers at research organizations, for example, by not only measuring scientific excellence in form of scientific publications but also by the relevance of research for informing decision making and policies.

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is an example for an initiative that is promoting the inclusion of perspectives from decision makers and indigenous people in scientific assessments. Such initiatives can help that applied environmental science becomes more meaningful and accessible for decision making on real-world problems, in particular related to challenges concerning sustainable land management and climate change.

5.5 Methods for synthesis, integration and meta-analysis of information on ecosystem services for decision support

This dissertation demonstrates how research methods for the synthesis, integration and meta-analysis of qualitative and quantitative information on ecosystem services can be applied towards supporting decision making on sustainable land management and related policies.

Chapter 2 demonstrates the use of a database for collecting, structuring and synthesizing ecosystem service information in order to inform decision making on national policies. Thereby, the database does not only include information on monetary values of ecosystem services and valuation methods. It also includes information on a set of criteria that were defined for judging the relevance of the data for informing decision making on national policies. These criteria were defined in collaboration with relevant experts from science as well as users of the database from the German Federal Environment Agency Umweltbundesamt (UBA). As mentioned above, this allows interpreting the recorded monetary values on ecosystem services in light of the original valuation studies and judging their suitability, credibility, and reliability for informing a particular decision. This transparency on data quality is important for identifying opportunities and limitations of using ecosystem service information in decision making. Hence, the created database serves as a repository for ecosystem service valuation studies that allows easy access to ecosystem service data with relevance for both science and decision making. Therefore, the database can serve as a tool for supporting decision making on environmental policies in Germany.

Chapter 3 provides a synthesis of conceptual approaches for ecosystem service assessments, which helped identifying the lack of explicit guidance on tailoring ecosystem service assessments to the information needs of decision makers. Using a literature review in combination with the synthesis of empirical experiences from stakeholder consultations in four case studies helped to develop a more problem-oriented approach to the assessment of ecosystem services. The developed guidance can help to better target the research design of ecosystem service assessments to decision relevant questions and to choose appropriate qualitative and quantitative methods for answering the questions.

Chapter 4 uses quantitative assessment methods for the meta-analysis of projects reducing emissions from deforestation with the goal of informing the design of the REDD+ climate policy. The use of multiple-linear regression analysis helped to identify factors determining carbon performance of REDD+ projects. The results show that besides information on ecosystem services also other information on study context and project design are relevant for informing policies on sustainable land management under climate change. This stresses the need for integrating also other biophysical and socio-economic information in the assessment of ecosystem services when aiming at informing decision making.

Hence, the research conducted for this dissertation shows that for informing real-world decision making, integrative assessment methods are needed that allow the synthesis of qualitative and quantitative information on a diverse range of indicators, which include, but are not limited to, ecosystem services. Therefore, ecosystem service assessments should be regarded as one of multiple components in multidisciplinary and integrative assessments for informing decision making on sustainable land management under climate change.

5.6 Conclusions

This dissertation, first, identifies major gaps in scientific knowledge on the monetary value of ecosystem services in Germany and limitations that need to be addressed in order to enhance the relevance of ecosystem service assessments for decision making on national policies. Second, it developed a more problem-oriented approach for ecosystem service assessments in order to enhance their policy relevance. Third, it informs the design of the REDD+ climate policy by identifying factors that have an influence on the amount of emission reductions expected by projects reducing deforestation, including trade-offs involved in the integration of multiple ecosystem services in project design.

Overall, the dissertation demonstrates that for informing real-world decision making, integrative assessment methods are required that allow the synthesis of qualitative and quantitative information on a diverse range of decision-relevant indicators. This includes, but should not be limited to, indicators on ecosystem services. Hence, ecosystem service assessments should be regarded as one of multiple components of multidisciplinary and integrative assessments required for informing decision making on sustainable land management under climate change.

5.7 Future research needs

Chapter 2 highlights the need for future monetary valuation studies of ecosystem services to assess and report biophysical and socio-economic indicators in units that allow for a better comparison of values between studies and the transfer of information across scales. This includes, for example, expressing values for biophysical and socio-economic indicators per hectare and per capita units. Furthermore, monetary values are only one way of expressing the significance of biodiversity and ecosystem services for society. There are multiple other values and forms of expressing societal relevance, for example, values related to human wellbeing but also information on the ecological significance of species and ecosystems. Chapter 3 provides examples for a range of indicators that can address multiple values. However, it remains vague on proposals for specific approaches and methods for assessing the multiple values of ecosystem services. Hence, future research should address how multiple values of biodiversity and ecosystem services can be assessed and integrated into decision-making processes. Finally, Chapter 4 highlights the need for future research on the impacts that resource use has on biodiversity and ecosystem services in distant places through teleconnections across regions and continents. The findings of Chapter 4 show that avoiding planned deforestation, e.g. by converting logging concessions or palm oil concessions into conservation areas, is a popular strategy for reducing deforestation. However, there is the risk that this strategy is simply shifting the production of commodities to other regions and continents, causing deforestation elsewhere. Therefore, there is not only the need for research on more sustainable land-use practices and sustainable production of commodities, but also the need for research into alternative consumption patterns that can reduce the demand for commodities that are currently driving deforestation (e.g. research on alternatives to using palm oil as biofuel). Thereby, not only maximizing carbon sequestration for mitigating climate change should be considered as criterion for sustainability, but also criteria related to the multiple values of biodiversity and ecosystem services need to be considered. In conclusion, multidisciplinary and integrative approaches are required for assessing the sustainability of land management under climate change, which includes, but is not limited to, the assessment of impacts on biodiversity and ecosystem services.

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Supplementary materials Chapter 2

S2.1 to S2.8 are part of the Excel file on the CD enclosed at the back of this dissertation.

S2.1 Introduction to database of monetary values of ecosystem services and biodiversity in Germany

S2.2 Overview of databases and publications reviewed for identifying studies with monetary valuation of ecosystem services.

S2.3 Database Master containing reviewed studies and monetary values of ecosystem services and biodiversity

S2.4 Carbon balance of land use in Germany converted to monetary values

S2.5 Definitions of ecosystem services applied in database (CICES and TEEB)

S2.6 Value ranges of monetary values for land cover conversions I-IV (selected in consultation with UBA)

S2.7 Value ranges of monetary values for land cover conversions I-IV (selected at expert workshop)

S2.8 Conversion indices applied for standardization in Euro-2014 values

Supplementary materials Chapter 4

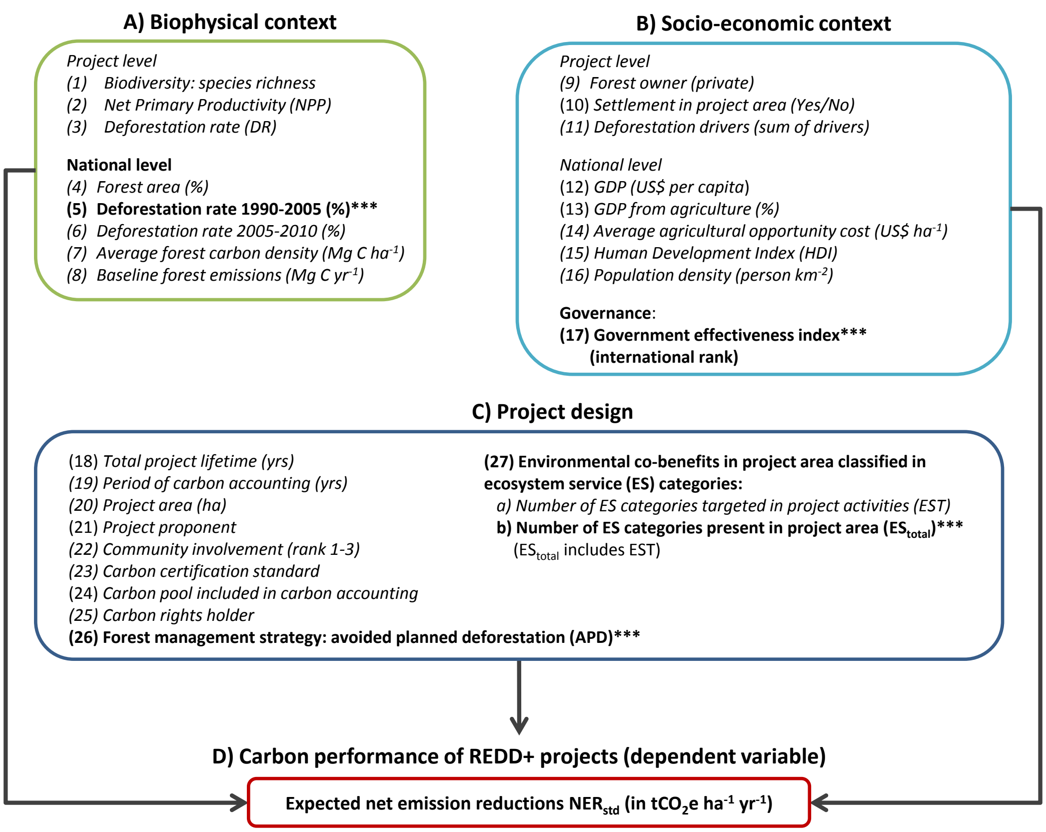
S4.1: REDD+ projects included in analysis

Supplementary Table 4.1 (Excel file on CD): REDD+ projects included in analysis.

Supplementary Table 4.2 (Excel file on CD): Project information.

Supplementary List S4.1 (PDF file on CD): Analysed project design documents of REDD+ projects.

S4.2: Conceptual design and variables included in analysis



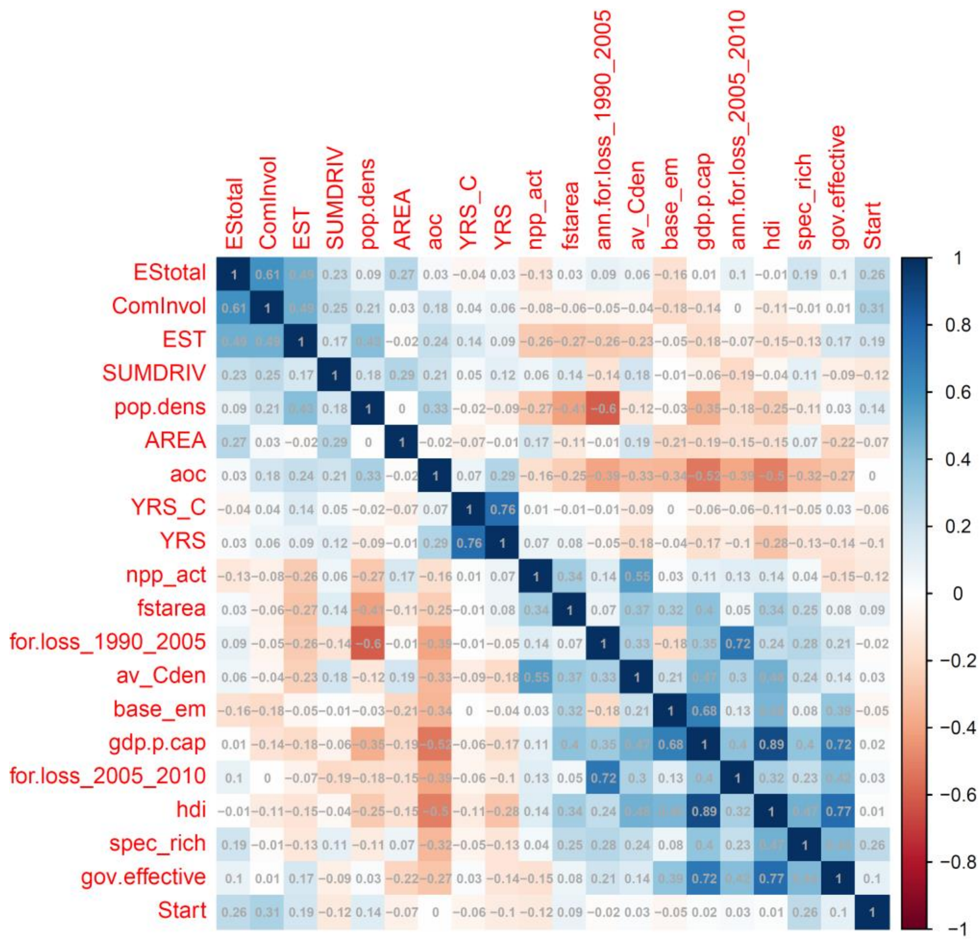
Supplementary Figure 4.1: Variables included in the analysis of factors explaining variance in expected carbon performance of REDD+ projects in terms of net emission reductions per hectare and year (NER_{std} in $tCO_2e\ ha^{-1}\ yr^{-1}$). The multiple linear regression model explaining the variance in NER_{std} with lowest value for the Akaike information criterion (AIC) includes four statistically significant variables (p < 0.05; marked with ***). Excluded variables are shown in *italics*.

S4.3: Independent variables (predictors)

Supplementary Table 4.3 (Excel file on CD): **Variables included in assessing the variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) reported by REDD+ projects**: A) biophysical variables (Supplementary Table 3.1), B) socio-economic variables (Supplementary Table 3.2), and C) variables of project design (Supplementary Table 3.3). Statistical analysis followed a stepwise procedure (Step 1-3).

S4.4: Assessing collinearity of independent variables (predictors)

a) Continuous variables: Pearson's correlation coefficient r



Supplementary Figure 4.2: Pearson's correlation coefficient of independent continuous variables for A) biophysical context, B) socio-economic context and C) project design. Only variables with correlation coefficients r > -0.5 or r < 0.5 were included in regression analysis.

b) Categorical variables: Pearson's x^2 -test

Supplementary Table 4.4 (Excel file on CD): Result of Pearson's x^2-test. Only significant values (p < 0.05) are shown.

S4.5: Multiple linear regression analysis for testing hypotheses and model comparison

Supplementary Table 4.5 (Excel file on CD): Multiple linear regression with biophysical variables A) explaining variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) reported by REDD+ projects.

Supplementary Table 4.6 (Excel file on CD): **Multiple linear regression with socio-economic variables B)** explaining variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) reported by REDD+ projects.

Supplementary Table 4.7 (Excel file on CD): **Multiple linear regression with project design variables C)** explaining variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) reported by REDD+ projects.

Supplementary Table 4.8 (Excel file on CD): **Multiple linear regression with combination of variables from biophysical (Model A 1), socio-economic (Model B 1), and project design variables (Model C 1)** explaining variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) reported by REDD+ projects.

S4.6: Multiple linear regression model with lowest value for AIC

Supplementary Table 4.9: **Multiple linear regression with variables of biophysical A), socio-economic B), and project design C)** that explain variance in expected carbon performance (NER_{std} in tCO_2e ha^{-1} yr^{-1}) with lowest value for the Akaike information criterion (AIC).

Model: NER_{std} = ƒ (ann.for.loss_1990_2005, gov.effective, APD, ES_{total}) + e	**Estimate**	**Pr(>|t|)**
(Intercept)	0.382	0.272
(5) National annual deforestation rate 1990-2005 (ann.for.loss_1990_2005)	-0.769	$4.34e^{-12}$ ***
(17) Government effectiveness index (gov.effective)	0.04	$3.20e^{-09}$ ***
(26) Avoided planned deforestation (APD: YES)	1.365	$5.11e^{-07}$ ***
(27) Number of ecosystem service categories (ES) in project area (ES_{total})	-0.103	0.001 ***
Akaike infromation criterion (AIC)		448.8
Cox & Snell pseudo R^2		0.78
Nagelkerke / Cragg & Uhler's pseudo R^2		0.78

S4.7: Net emission reductions of REDD+ projects according to land tenure, carbon rights and avoiding planned deforestation

Supplementary Table 4.10: Net emission reductions of REDD+ projects in terms of percentage of total expected net emission reductions (NER_{total}) and in terms of carbon performance (NER_{std} in $tCO_2e\ ha^{-1}\ yr^{-1}$) according to land tenure (S-Table 4.10.1), holder of carbon rights (S-Table 4.10.2) and for REDD+ projects with avoiding planned deforestation (APD) (S-Table 4.10.3).

S-Table 4.10.1: Net emission reductions of REDD+ projects according to ownership of land in project area

Land tenure	**Government**	**Communities**	**Private**	**NGOs**	**Multiple (trust)**	Total
Number of REDD+ projects	13	20	21	0	12	66
% of NER_{total} (tCO_2e) (n = 66)	19.0	19.5	42.4	0.0	19.1	100
Mean NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	5.7	5.8	12.4	0.0	15.6	9.7
Median NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	2.6	4.8	5.2	0.0	5.4	4.8

S-Table 4.10.2: Net emission reductions of REDD+ projects according to holders of carbon rights

Holder of carbon rights	**Government**	**Communities**	**Private**	**NGOs**	**Multiple (trust)**	Total
Number of REDD+ projects	5	23	28	5	5	66
% of NER_{total} (tCO_2e)	5.1	10.4	75.8	4.1	4.6	100
Mean NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	3.1	5.7	14.9	11.0	4.1	9.7
Median NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	2.6	4.8	7.5	3.4	4.4	4.8

S-Table 4.10.3: Net emission reductions of REDD+ projects with avoided panned deforestation (APD) according to carbon rights

Holder of carbon rights	**Government**	**Communities**	**Private**	**NGOs**	**Multiple (trust)**	**Total**
Number of REDD+ projects with APD	0	1	14	1	0	16
% of NER_{total} (tCO_2e)	0.0	0.3	51.9	0.04	0.0	52.2
Mean NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	0.0	2.0	22.3	45.7	0.0	22.5
Median NER_{std} ($tCO_2e\ ha^{-1}\ yr^{-1}$)	0.0	2.0	17.2	45.7	0.0	17.2

Content of CD enclosed at back of this dissertation

Supplementary materials Chapter 2: PDF-file of project report Förster et al. (2017) and an Excel file containing S2.1 – S2.8 (database for Chapter 2).

Supplementary materials Chapter 4: PDF file with Supplementary List S4.1 and an Excel file containing Supplementary Tables 4.1 – 4.8 (database for Chapter 4).

Curriculum vitae

Name	**Johannes Förster**
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Place of birth	**Dresden**
Nationality	**German**

Education

Since 04/2013	**External Ph. D. student (Geography), Martin Luther University Halle-Wittenberg, Faculty of Natural Sciences III** *Title of Dissertation: Assessing ecosystem services for informing decision making on sustainable land management under climate change*
10/2006 – 08/2009	**Global Change Ecology (M.Sc.) University Bayreuth, Germany** *M. Sc. thesis: The Potential for Reducing Emissions from Deforestation and Degradation (REDD) in Western Ghana*
01/2004 – 04/2006	**Physical Geography (M.Sc.) Umea University, Sweden** *M. Sc. thesis: 400-year reconstruction of lake water carbon*
01/2004 – 04/2006	**Biology (B.Sc.) Umea University, Sweden** *B. Sc. thesis: CO_2-saturation in lake water*
10/2001 – 01/2004	**Vordiplom in Landscape Ecology and Nature Conservation, University Greifswald, Germany**
1992 – 2000	**Abitur, Friedrich-Schleiermacher-Gymnasium Niesky**

Professional experience

Since 10/2009	**Helmholtz Centre for Environmental Research - UFZ, Leipzig** *Scientific staff in Dept. Environmental Politics / Dept. Conservation Biology / Dept. Economics; Ph. D. candidate in Dept. Computational Landscape Ecology*
10/2007 – 03/2008	**International Union for Conservation of Nature (IUCN)** *Junior Prof. Associate, Forest Conservation Progr.,Switzerland*
02/2006 – 04/2006 **05/2005 – 09/2005** **06/2004 – 09/2004**	**Royal Swedish Academy of Sciences, Abisko, Sweden** *Abisko Scientific Research Station* *Scientific staff and research assistant*

Leipzig, 29 August 2017

List of publications

Peer-reviewed journal articles

Cumming, T. L., R. T. Shackleton, **J. Förster**, J. Dini, A. Khan, M. Gumula, I. Kubiszewski (2017) Achieving the National Development Agenda and the Sustainable Development Goals (SDGs) through Investment in Ecological Infrastructure: A Case Study of South Africa. *Ecosystem Services*. DOI:10.1016/j.ecoser.2017.05.005

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Förster, J. and H. Wittmer (2012) Towards a needs-oriented appraisal of ecosystem services: the TEEB Stepwise Approach. Poster Session 2: Ecosystem Services. Planet under Pressure Conference, London, March 26-29, 2012.

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Selbstständigkeitserklärung / *Declaration under Oath*

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.
I declare under penalty of perjury that this thesis is my own work entirely and has been written without any help from other people. I used only the sources mentioned and included all the citations correctly both in word or content.

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Datum	Name