REMOTE SENSING OF ECOSYSTEM SERVICES

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Abstract

Ecosystem services are the benefits that humans obtain from ecosystems for human well-being. They are essential for human well-being, but severely threatened by human activities. Therefore, spatially explicit mapping and regular monitoring approaches of ecosystems services are crucial to evaluate their state. Remote sensing offers versatile capabilities in doing so. This thesis demonstrated and exemplified the use of remote sensing for spatially explicit (i) ecosystem service mapping and (ii) monitoring and for (iii) identifying the drivers of change and their effects on ecosystem services.

The foundation of the thesis built an observation-based approach for spatially explicit ecosystem service mapping. To accomplish this, the ecosystem service cascade was further developed from a theoretical ecosystem service mapping approach to a framework based on observations. This approach was applied on an urban-rural gradient in Switzerland to retrieve spatial heterogeneity in two ES, namely CO_2 regulation and food supply. The spatial scaling of this approach to Switzerland was conducted with satellite data and allowed monitoring of four ecosystem services (i.e. CO_2 regulation, soil erosion prevention, air quality regulation, recreational hiking) for more than a decade. Besides investigating spatial and temporal ecosystem service trends, ecosystem service interactions such as trade-offs and synergies were also identified. Furthermore, these ecosystem service time series enabled detecting the impacts of climatologies (i.e. precipitation, temperature and rel. sunshine duration) and non-climatological effects on ecosystem service change.

This thesis provided a framework to map ecosystem services spatially explicit based on observations, which enabled detecting spatial ecosystem service heterogeneity within single and among different ecosystems. High spatial variability in ecosystem services was identified in more natural ecosystems such as forests and grasslands, in contrast to highly managed systems like agricultural areas with low spatial ecosystem service heterogeneity.

Ecosystem service monitoring revealed decreasing trends in CO_2 regulation and air quality in Switzerland between 2004 and 2014, while soil erosion prevention increased and recreational hiking remained stable. These trends were characterized by regional differences indicating amongst others altitudinal dependencies. The

ecosystem service interactions identified synergies for all ecosystem service combinations at national scale except for CO_2 regulation and air quality regulation, which were a trade-off. ES interactions displayed a scale dependency, since interactions changed from national to regional scale, and differed across regions indicating altitudinal dependencies.

Based on the ecosystem service time series, this thesis investigated climatological and non-climatological effects on ecosystem service trends. The findings showed that climatological effects had the strongest influence on ecosystem service change except for recreational hiking. Climatological impacts affected mainly ecosystem service trends in the Jura and the Swiss Midlands, while the Central and Southern Alps indicated spatially heterogeneous patterns of climatological and non-climatological effects. Temperature was for all ecosystem services investigated the most important climatology.

This thesis critically discussed the advantages and drawbacks of these approaches for ecosystem service mapping and monitoring and provided findings for decision-making and the evaluation of policy targets. Future research directions proposed included the extension of the ecosystem services mapped and monitored with remote sensing data, the operationalization of remote sensing based ecosystem service monitoring with essential ecosystem service variables and to improve communication between ecosystem service research and policy and practice.

In conclusion, by using remote sensing for ecosystem service assessments, this thesis contributed to spatially explicit and continuous ecosystem service mapping and monitoring, and to identifying the impacts of drivers of change on ecosystem services in an original way. Thus, this thesis fostered spatially explicit ecosystem service assessments and contributed to a paradigm shift from pure ecosystem service mapping towards regular ecosystem service monitoring.

Zusammenfassung

Ökosystemdienstleistungen umschreiben den Nutzen bzw. die Leistungen, die Ökosysteme bereitstellen und dem Wohlergehen der Menschen dienlich sind. Sie sind ein entscheidender Bestandteil des menschlichen Wohlergehens, jedoch stark durch menschliche Aktivitäten bedroht. Die Zustandsbewertung von Ökosystemdienstleistungen ist notwendig und durch räumlich explizite Kartierungs- und regelmäßige Überwachungsansätze möglich. Die Fernerkundung bietet hierfür vielseitige Einsatzmöglichkeiten und stand im Zentrum dieser Doktorarbeit. Im Speziellen wurden sowohl der Nutzen der Fernerkundung für die räumlich explizite (i) Kartierung und (ii) Überwachung von Ökosystemdienstleistungen, als auch (iii) die Bestimmung von Veränderungsursachen und deren Auswirkungen auf Ökosystemdienstleistungen untersucht.

Die Basis dieser Doktorarbeit stellte ein auf Erdbeobachtungsdaten basierendes Konzept zur räumlich expliziten Kartierung von Ökosystemdienstleistungen dar. Hierfür wurde die Ökosystemdienstleistungskaskade von einem theoretischen Ansatz zur Kartierung von Ökosystemdienstleistungen zu einem erdbeobachtungsbasierenden Konzept weiterentwickelt. Dieser Ansatz wurde auf einen Stadt-Land Gradient in der Schweiz angewandt. Damit konnte das Potenzial des neuen Ansatzes am Beispiel der räumliche Heterogenität zweier Ökosystemdienstleistungen, der CO_2 Regulierung und der Nahrungsbereitstellung demonstriert werden. Die Skalierung dieses Überwachungsansatzes auf die ganze Schweiz wurde mit Satellitendaten realisiert. Dieser ermöglichte exemplarisch die schweizweite Überwachung von vier Ökosystemdienstleistungen, namentlich der CO_2 Regulierung, der Vermeidung von Bodenerosion, der Regulierung der Luftqualität, sowie dem Erholungspotenzial von Wanderungen über mehr als ein Jahrzehnt. Neben der räumlichen und zeitlichen Bestimmung der Ökosystemdienstleistungstrends wurden auch deren Interaktionen (Trade-offs und Synergien) untersucht. Zudem wurden die Auswirkungen klimatologischer und nicht-klimatologischer Einflüsse auf die Veränderungen von Ökosystemdienstleistungen mit Hilfe dieser Zeitreihen erfassen.

Der in dieser Doktorarbeit vorgeschlagene Ansatz ermöglichte die Untersuchung und Quantifizierung der räumlichen Heterogenität von Ökosystemdienstleistungen sowohl innerhalb eines als auch zwischen verschiedenen Ökosystemen. Eine hohe räumliche

Variabilität der Ökosystemdienstleistungen wurde in naturbelasseneren Ökosystemen wie etwa Wäldern und Grasländern erfasst. Stark genutzte Ökosysteme wie landwirtschaftliche Flächen wiesen eine substanziell geringere räumliche Heterogenität auf.

Im Zeitraum von 2004 bis 2014 zeigte die Überwachung von Ökosystemdienstleistungen abnehmende Trends für die CO_2 Regulierung und die Regulierung der Luftqualität in der Schweiz auf, während die Vermeidung von Bodenerosion zunahm und das Erholungspotential von Wanderungen unverändert blieb. Diese Trends deuteten auf regionale Unterschiede in Abhängigkeit der Höhenlage hin. Eine detaillierte Untersuchung von Interaktionen zwischen Ökosystemdienstleistungen zeigte Synergien für alle untersuchten Ökosystemdienstleistungskombinationen auf nationaler Ebene auf, lediglich die Kombination von CO_2 Regulierung und Regulierung der Luftqualität offenbarte einen Trade-off. Zudem wiesen die Untersuchungen auf eine Skalenabhängigkeit von Interaktionen hin, da sich diese einerseits von nationaler zu regionaler Ebene veränderten, andererseits auch eine Höhenabhängigkeit zeigten.

Basierend auf genannten Ökosystemdienstleistungszeitreihen untersuchte diese Doktorarbeit zudem klimatologische und nicht klimatologische Effekte auf Ökosystemdienstleistungstrends. Abgeleitete Ergebnisse zeigten deutlich, dass klimatologische Effekte den größten Einfluss auf die Veränderungen von Ökosystemdienstleistungen hatten, lediglich die Veränderungen im Erholungspotenial von Wanderungen stellten eine Ausnahme dar. Klimatologische Effekte beeinflussten hauptsächlich Ökosystemdienstleistungstrends im Jura und Schweizer Mittelland, während die Zentral- und die südlichen Alpen von räumlich heterogenen Mustern klimatologischer und nicht-klimatologischer Effekte gekennzeichnet waren. Temperatur war für alle Ökosystemdienstleistungen der wichtigste klimatologische Faktor.

Eine kritische und umfangreiche Diskussion der Vor- und Nachteile verwendeter Ansätze zur Kartierung und Überwachung von Ökosystemdienstleistungen komplettierte diese Arbeit. Zudem lieferte diese Doktorarbeit Ergebnisse für Entscheidungsprozesse und die Evaluation politischer Zielvorgaben. Als zukünftige Forschungsschwerpunkte wurden die Erweiterung der fernerkundungsbasierten Ökosystemdienstleistungsansätze, die Operationalisierung fernerkundungsgestützter

Überwachungsansätze mit essentiellen Ökosystemdienstleistungsvariablen und die Verbesserung der Kommunikation zwischen Wissenschaft und Politik und Praxis definiert.

Diese Doktorarbeit leistete durch die Nutzung von Fernerkundung einen Beitrag zur räumlich expliziten und kontinuierlichen Überwachung von Ökosystem-dienstleistungen und zur Bestimmung der Ursachen ihrer Veränderungen. Somit förderte sie die räumlich explizite Bewertung von Ökosystemdienstleistungen und trug zu einem Paradigmenwechsel von der reinen Kartierung zur regelmäßigen Überwachung von Ökosystemdienstleistungen bei.

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List of Abbreviations

APEX	Airborne Prism Pxperiment
ASTRA	Bundesamt für Strassen und Schweizer Wanderwege
AQR	Air quality regulation
β	Remaining impact of soil erosion
C	Concentration of PM_{10}
CO_2R	Carbon dioxide regulation
CEST	Central European summertime
CUE	Carbon use efficiency
C_V	Vegetation cover
DEM	Digital elevation model
DMSP-OLS	Defense Meteorological Satellite Program Operational Linescan System
EBVs	Essential biodiversity variables
eESVs	Essential ecosystem service variables
EC	Eddy-covariance
E_{flow}	Ecosystem service flow of soil erosion prevention
E_{pot}	Ecosystem service potential supply of soil erosion prevention
ES	Ecosystem service
ε	Residual component
ESDAC	European Soil Data Centre
FE	Fixed effects
FLEX	European Space Agency's 8th Earth explorer mission Fluorescence Explorer
FWHM	Full-width-at-half-maximum
GEO BON	Group on Earth Observation Biodiversity Observation Network
GPP	Gross primary production
$GPP_{cropland}$	Gross primary production of cropland
GPP_{forest}	Gross primary production of forest
$GPP_{grassland}$	Gross primary production of grassland
GPP_{ann}	Annual gross primary production
GPP_{inst}	Instantaneous gross primary production
GPP_{day}	Daily gross primary production
GPP_{ECO}	Gross primary production measured by eddy-covariance tower
GPP_{mod}	Modeled gross primary production
GPP_{scaled}	Scaled gross primary production
H	Harvest
HCRF	Hemispherical conical reflectance factor
HI	Harvest index
IPBES	Inter-governmental Platform on Biodiversity and Ecosystem Services
IRGA	Intrared gas analyzer
JRC	Joint Research Center

K	Soil erodibility
LAI	Leaf area index
LiDAR	Light detection and ranging
LS	Topographic factors length and steepness
LWF	Long-term forest research site
MAES	Working group on Mapping and Assessment on Ecosystems and their Services
MC	Moisture content
MODIS	Moderate-resolution imaging spectrometer
NABEL	National Air Pollution Monitoring Network
NDVI	Normalized difference vegetation index
NEE	Net ecosystem exchange
NPP	Net primary production
NBP	Net biome production
NSL	Nighttime stable lights
P	Precipitation
PAR	Photosynthetic active radiation
PM_{10}	Particulate matter with a diameter less than 10µm
PPFD	Photosynthetic photon flux density
PUD	Photo-user-days
R	Rainfall erosivity
R_A	Autotrophic respiration
R_H	Heterotrophic respiration
$R_{H\text{-}EC}$	Heterotrophic respiration estimated by eddy-covariance measurements
$R_{H\text{-}Raich}$	Heterotrophic respiration estimated according to Raich et al. (2002)
R_{ECO}	Ecosystem respiration
RH	Recreational hiking
RH_{pot}	Ecosystem service potential supply of recreational hiking
RH_{flow}	Ecosystem service flow of recreational hiking
RI	Standardized ratio index
RQs	Research questions
RS	Remote sensing
R_S	Soil respiration
RSR	Root to shoot ratio
SAGA	System for Automatic Geoscientific Analyses
SAR	Synthetic aperture radar
SEEA	System of Environmental-Economic Accounting
SEEA EEA	System for Environmental Economic Accounts Experimental Ecosystem Accounting
SEP	Soil erosion prevention
SD	Standard deviation
SIF	Sun-induced chlorophyll fluorescence
SPA-crop	Soil-Plant-Atmosphere Crop model
SSI	Spectral sampling interval

SVM	Support vector machine
T	Time step
T_a	Air temperature
TEEB	The Economics of Ecosystems and Biodiversity
V_d	Deposition velocity
WSL	Swiss Federal Institute for Forest, Snow and Landscape Research
Υ	Structural Impact
$Yield_{maize}$	Yield of maize
$Yield_{sugar\ beet}$	Yield of sugar beet

Chapter 1

Introduction

1.1 Ecosystem services – highlighting the benefits of nature to foster sustainable development

Ecosystems provide a variety of benefits to humans, from provisioning of freshwater, food and clean air, to erosion control, flood protection and recreation (Daily, 1997; de Groot et al., 2002). These benefits that people obtain from ecosystems for human well-being can be described as ecosystem services (ES) (Millennium Ecosystem Assessment, 2005), and are essential for human well-being. This dependency of society on ES is represented in three ES categories defined by the Millennium Ecosystem Assessment (2005): *provisioning services*, which are products received from nature such as freshwater and food supply; *regulating services*, obtained from the regulation of ecosystem processes like climate regulation, soil erosion prevention and air quality regulation; and *cultural services* that refer to non-material benefits people obtain from ecosystems, e.g. recreation and aesthetic beauty of a landscape.

However, ES are often taken for granted, considered to be infinite and free of charge. The reality, however, is quite different: ES are severely threatened by human activities. Planetary boundaries have been crossed leading to irreversible changes in the Earth system (Steffen et al., 2015). Land use change, habitat degradation, pollution, over-exploitation and climate change affect ecosystems and their services worldwide (Folke et al., 2016; Millennium Ecosystem Assessment, 2005). The ecosystem service concept aims at (re)connecting people with nature and at raising awareness about human dependency on ES by focusing not only on tangible, but also on nonmaterial, intangible ES (Fisher et al., 2009; Schröter et al., 2014b). In this way, the concept bridges ecological and social system and can be used as a starting point to define, determine, monitor and value services from nature. Making ES explicit not only helps to emphasize the importance and necessity of protecting ecosystems, it can also inform management decisions about areas of low and high ES provision and provide additional information about the social-ecological systems that we live in (Science for the Environment Policy, 2015). While the ES concept is often criticized for providing an anthropocentric, instead of a stronger ecocentric view (Schröter et al., 2014b), this concept still provides valuable tools to foster policies that optimize the sustainable use of ecosystems.

1.1.1 Background

The idea of humans benefiting from nature, both directly and indirectly, is certainly not new, and can be traced back several thousand years, much longer than the scientific analysis of ES. Plato already recognized the importance of nature's services for human well-being, when realizing that extensive deforestation caused soil erosion in 400 BC (Feen, 1996; Runnels, 1995). A later example is the Easter Island society that realized too late how strongly they depended on nature and its ES. Extensive deforestation on their island caused soil erosion, a lack of raw materials, in particular for constructing ships and consequently, a strong population decline (Mieth and Bork, 2003). The modern ES concept emerged in the late 1970s, when Westman (1977) emphasized the importance of recognizing and valuing ES benefits. At the same time, protecting indispensable ES evolved as additional argument for fighting species extinction – in addition to their intrinsic value (Ehrlich and Ehrlich, 1981, 1972). In the 1990s the ES concept gained momentum (Figure 1.1) and became widely known in the scientific literature due to Daily's book on "Nature's Services: Societal Dependence on Natural Ecosystems" (1997) and Costanza et al.'s study on "The value of the world's ecosystem services and natural capital" (1997). This increasing scientific interest in both environmental science and economics brought ES into political agendas. A major global United Nation's study, the Millennium Ecosystem Assessment (2005), investigated the state of ecosystems and the human impacts on them. It revealed the most rapid and extensive ecosystem changes in human history over the past 50 years, resulting in degradation of many ecosystem services worldwide. In combination with The Economics of Ecosystems and Biodiversity (TEEB; Kumar, 2010), these two reports contributed significantly to mainstreaming and developing the ES concept as well as to further fostering scientific interest in ES (Figure 1.1).

The ES concept has developed into a large and rapidly expanding research field that aims to measure, map, assess, and value aspects of societal dependence on nature. Additionally, the growing application of the ES approach in planning and regulatory contexts has caused fundamental shifts in environmental governance. Policy makers have included ES in political agendas, through platforms such as the Aichi Biodiversity Targets of the UN's Convention on Biological Diversity (Convention on Biological Diversity, 2010), the Biodiversity Strategy 2020 of the European Union

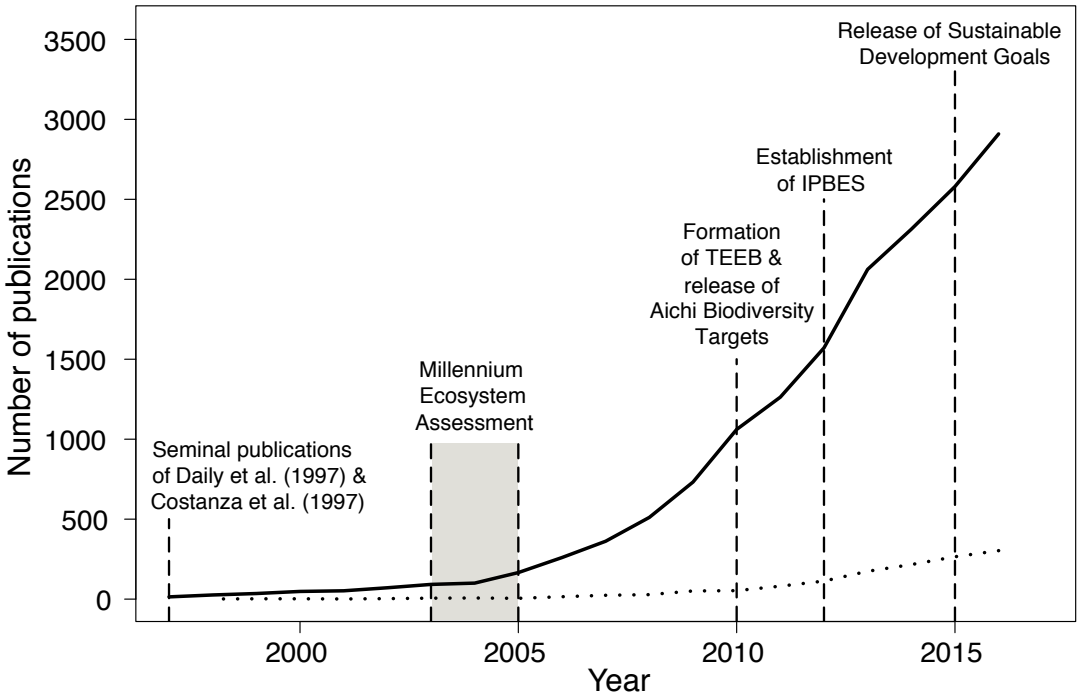

Figure 1.1 Time line of ES publications from 1997 until 2016 based on Scopus (on 31 May 2017): published articles with "ecosystem service*" (solid line) and with "ecosystem service*" & map* (dotted line) in the article title, keywords or abstract. Vertical dashed lines indicate important dates of publications and policy decisions that fostered ES research.

(European Commission, 2011) and the Sustainable Development Goals (United Nations, 2015). Through this political shift, ES have been intrinsically tied together with biodiversity. Despite their relationship being still a source of disagreement (Mace et al., 2012), it is generally recognized that sustainability can only be achieved by considering both, biodiversity and ES. Consequently, policy makers have required non-economic and economic assessments that investigate how biodiversity and ES loss can be translated into loss of human well-being and welfare loss (e.g. TEEB, System of Environmental-Economic Accounting (SEEA)), respectively. Additionally, the Inter-governmental Platform on Biodiversity and Ecosystem Services (IPBES) was established by 90 governments to assess the state of ecosystems, biodiversity and their services and to promote the ES concept in science, police and practice (Díaz et al., 2015; Larigauderie and Mooney, 2010).

This rapid development of the ES concept has attracted extensive discussions, applications and research (Kull et al., 2015; McDonough et al., 2017). These range from questioning the usefulness of the concept (Kull et al., 2015; Schröter et al.,

2014b) and striving for developments in terminologies and definitions (de Groot et al., 2002; Fisher et al., 2009; Hein et al., 2016) to ES mapping and assessment methods (Jacobs et al., 2015; Maes et al., 2016, 2014; Malinga et al., 2015). Moreover, the adequate quantification of less tangible benefits – cultural ES that require enhanced inclusion of social science approaches – remains a subject of debate (Chan et al., 2012). The ES concept aims to foster interdisciplinary work to develop a comprehensive approach for research and implementation in practice (Díaz et al., 2015). This concept is increasingly established as anthropocentric argument for protection and sustainable management of ecosystems (Wood and DeClerck, 2015).

1.1.2 Ecosystem service mapping – spatial assessment of nature

Implementing the ES concept in practice requires spatial assessments since ES vary in space and time (Boyd, 2008). There is a distinction between ES potential supply, which refers purely to the biophysical supply of a service by an ecosystem, irrespective of its human use, and ES flow defined as the actual service used by humans (Hein et al., 2016). Both can have different spatial contexts. The spatial heterogeneity of ES potential supply evolves on the one hand due to spatial variability in underlying environmental processes and conditions such as water availability, soil properties, topography, vegetation (Schröter et al., 2015). On the other hand it is affected by human influences represented in land use and land management (Maes et al., 2012). ES flow depends on ES potential supply and the locations of the beneficiaries. ES potential supply can be provided in the same area as ES flow, for example in the case of food supply (Figure 1.2a). The spatial relationship between ES potential supply and flow may be directional. For instance, potential supply of flood regulation is performed by a river or wetland, but ES flow is located downslope, where humans use the service (Figure 1.2b). Alternatively, ES flow may take place at more distant areas compared to potential supply, described as omnidirectional, for example in the case of carbon sequestration (Figure 1.2c) (Fisher et al., 2009; Syrbe and Walz, 2012).

Spatially explicit ES assessments are required to support improved management decisions, to measure and verify the progress of national and international environmental policies (e.g. Aichi Biodiversity Targets, Sustainable Development

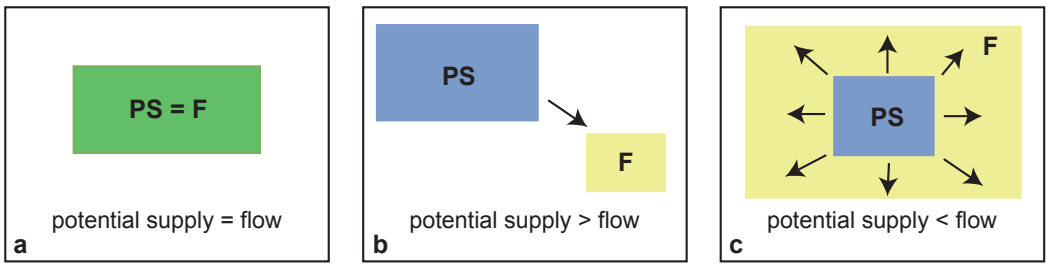

Figure 1.2 Spatial relationships between ES potential supply (PS) and ES flow (F) (adapted from Fisher et al. (2009) and Raudsepp-Hearne and Peterson (2016)). (a) PS and F take place in the same area (e.g. food supply), (b) PS is directional with F that is generated e.g. down stream or downslope (e.g. soil erosion prevention), and (c) PS is omnidirectional, where F occurs in the surrounding (e.g. carbon sequestration).

Goals), and to examine the efficiency of conservation measures and environmental payment schemes (e.g. payments for ecosystem services, agri-environmental schemes, REDD+) (de Groot et al., 2010; Feld et al., 2009; Maes et al., 2016, 2013; O'Connor et al., 2015). This demand has resulted in the development of a separate field within ES research dedicated to ES mapping, which has substantially increased in scientific publications over the last decade (Figure 1.1, dotted line) (Andrew et al., 2015; Maes et al., 2014; Schägner et al., 2013; Seppelt et al., 2011). This scientific interest in ES mapping has been partly fostered by initiatives targeting ES modeling and mapping like the Natural Capital Project (www.naturalcapitalproject.org) and the working group on Mapping and Assessment on Ecosystems and their Services (MAES) (Maes et al., 2013) .

Over this period, various ES mapping approaches have evolved covering a wide range of data requirements, spatial scales, accuracies and applicability (Malinga et al., 2015; Martínez-Harms and Balvanera, 2012). They can be grouped into four categories (Eigenbrod et al., 2010; Martínez-Harms and Balvanera, 2012; Schröter et al., 2015): look-up tables, spatial interpolation, regression models, and causal relationships (Table 1.1). The look-up table method allocates a constant ES value to each land cover class, for which values were derived from previous studies at other places and spatial scales, and expert knowledge (Burkhard et al., 2009; Jacobs et al., 2015; Kandziora et al., 2013). Spatial interpolation maps the distribution of ES based on spatial autocorrelation of in-situ measured data points (Sumarga and Hein, 2014).

Regression models predict the relationship between explanatory variables such as environmental data and a response variable like ES field data (Lavorel et al., 2011; Schröter et al., 2014a). Causal relationships model ES based on existing knowledge about the influence of environmental variables on ES distribution and abundance (Nelson and Daily, 2010; Remme et al., 2014). The different methods have specific

Table 1.1 ES mapping approaches based on model categories and their specific characteristics (based on Martínez-Harms and Balvanera, 2012; Schröter et al., 2015).

Model categories	Data requirements	Spatial scalability	Accuracy	Application purpose	Literature
Look-up table	Land cover map combined with literature values or expert knowledge	No	low	fast, superficial	Burkhard et al., 2009; Jacobs et al., 2015; Kandziora et al., 2013
Spatial interpo-lation	Point data e.g. from national statistics, census data, field data	Yes			Sumarga and Hein, 2014
Regression model	In situ measurements in combination with spatial environmental data e.g. climate, topography, soil properties, remotely sensed data (e.g. NDVI, vegetation traits)	Yes, de-pending on the scale of the input data			Lavorel et al., 2011; Schröter et al., 2014a
Causal relation-ship	Spatial environmental data e.g. climate, topo-graphy, soil pro-perties, remotely sensed data (e.g. NDVI, vegetation traits); in situ data, literature values	Yes, de-pending on the scale of the input data	high	extensive	Nelson and Daily, 2010; Remme et al., 2014

data requirements, spatial scalability, accuracy (i.e. how well a spatial model estimates the real spatial ES distribution) and application purposes, which determine as well their frequency of use in scientific literature (Table 1.1). Beside the ES model type, input data for the different ES models highly determine the spatial explicitness of the retrieved ES maps. Utilization of land cover maps as a basis for ES mapping is often the case for all four ES model categories. However, this usually neglects the spatial heterogeneity of ES within land cover classes (Eigenbrod et al., 2010). The first mapping approaches using RS derived ES properties or proxies have provided incentives to improve the mapping of ES heterogeneity (Andrew et al., 2014; Remme et al., 2014; Schröter et al., 2014a; Strauch and Volk, 2013).

Although all the aforementioned approaches result in the retrieval of ES maps, so far not one approach has been established as a standard, which complicates the operationalization of ES in practice. The lack of spatial explicitness in ES mapping and consequently, the neglect of ES heterogeneity in space remain as key challenges (Bennett et al., 2015).

1.1.3 Monitoring ecosystem services

Spatially explicit ES mapping is not only important for identifying ES heterogeneity, but is also the basis for ES monitoring. Current environmental policies such as the Aichi Biodiversity Targets (Convention on Biological Diversity, 2010), the European Union Biodiversity Strategy (European Commission, 2011) and the Sustainable Development Goals (United Nations, 2015), as well as research (Cord et al., 2017, 2015; Karp et al., 2015; O'Connor et al., 2015; Tallis et al., 2012) require and call for regular and systematic monitoring of ES to improve management decisions and to verify progress towards environmental policies, payment schemes and efficiency of conservation measures. Multiple studies have indicated degradation and loss of ES (Millennium Ecosystem Assessment, 2005; Runting et al., 2017) and a crossing of stable planetary boundaries with unknown sudden, rapid and surprising consequences for ecosystems, their services and human well-being (Steffen et al., 2015). Nevertheless, there still exist neither national nor global, systematic monitoring systems that regularly measure ES change through space and time (Cord et al., 2017; Karp et al., 2015; O'Connor et al., 2015). Therefore, we are not able to foresee areas

with irreversible ES change, but are surprised by such shifts due to this lack of early warning systems.

1.1.3.1 Trends

Technically, the capacity is already available to start detecting changes in ecosystems worldwide. But data gaps remain, in particular with regard to global data sets that impede our understanding of socio-ecological problems and consequently, ES monitoring (Tallis et al., 2012). Methodologically, changes in ES should not only be based on qualitative modifications, i.e. changes in the extent of land cover classes, but also on quantitative changes related to shifts in ecosystem conditions that affect ecosystem processes, functioning and finally ES (European Commission et al., 2013). Therefore, the Group on Earth Observations Biodiversity Observation Network (GEO BON) recently started the search for essential ecosystem service variables as addition to essential biodiversity variables (Pereira et al., 2013) that should standardize effective observation of ES status and trends, including their supply, use, value and contributions to well-being over time.

Attempts have been made to address these gaps through case studies of ES monitoring that have been conducted from regional to global scales. Karp et al. (2015) used a combination of national statistics and modeling approaches with data collected worldwide for each country to investigate worldwide changes in eight ES between 1996 and 2005. Temporal changes revealed increases in several provisioning services such as crop and game meat production, while timber and hydropower supply were decreasing. In contrast, Schirpke et al. (2013) investigated long-term trends of six ES at regional scale in the Alps based on historical land use changes between 1954 and 2011. Results showed increasing ES trends until around the year 2000, after which ES trends decreased or remained stable. Guerra et al. (2016) monitored fluctuating changes in soil erosion prevention at landscape scale over 60 years. The ES trends were based on models driven by land cover maps for each decade. However, basing ES monitoring on land cover maps and census data provides only individual snapshots in time, restricted to five to ten year periods, when new data are released (Karp et al., 2015). Hence, ES trends are the result of interpolating such temporal snapshots. For this reason, it is widely recognized that the considerable potential of

RS for monitoring ES will fill this gap in the future (Cord et al., 2017, 2015; Pettorelli et al., 2016; Skidmore et al., 2015; Tallis et al., 2012).

1.1.3.2 Interactions

ES mapping and monitoring enable the identification of ES interactions (Maes et al., 2012). Due to the many processes and functions in ecosystems, not only individual, but also multiple ES can be provided at the same time (Bolliger et al., 2011; Schindler et al., 2014). These ES interact with each other and are influenced by human resource management decisions leading to different ES relationships (Foley et al., 2005). We distinguish between ES trade-offs that occur, when an increase in one ES leads to a decrease in another, and synergies that represent an increase in both ES (Bennett et al., 2009; Rodriguez et al., 2006). Knowing how ES interact and influence each other is crucial for sustainable management decisions to support not only the provision of one but of multiple ES (Liu et al., 2015; Willemen et al., 2012). A trade-off, for example, represents the conversion of forest into agricultural land to foster food supply, but causing declines in several services such as water flow and quality regulation, infectious disease mediation, and carbon sequestration. In contrast, a synergy is the development of an agroforestry system that combines agriculture with forestry, which supports several ES (Fagerholm et al., 2016) and enables multifunctional landscapes (Foley et al., 2005).

To date, most research has mainly focused on interactions between provisioning and non-provisioning ES (Howe et al., 2014). Consequently, nearly three times more ES trade-offs were identified compared to synergies (Lee and Lautenbach, 2016). The most frequently determined trade-off is between “biomass production” (e.g. timber and fodder) and “climate regulation”, while synergies are dominated by interactions between “habitat and gene pool protection services” and regulating services like “soil formation regulating services” (Lee and Lautenbach, 2016). However, ES interactions can depend on the spatial scale of investigation (Hein et al., 2006; Lee and Lautenbach, 2016; Raudsepp-Hearne and Peterson, 2016; Willemen et al., 2012). This means, while a pair of ES caused a trade-off at local scale, it can result in a synergy at national scale. However, so far most studies investigated ES interactions only at one scale, preferably at regional and plot scale, while continental and global scale were

least studied (Lee and Lautenbach, 2016). ES mapping and monitoring approaches should be scalable to fit the spatial scale of decision-making, which can often occur at different scales to habitat or geographical boundaries (Daily et al., 2009).

1.1.4 Understanding drivers of ecosystem service change

Ecosystems and the services they provide, have been degraded and lost in the past (Millennium Ecosystem Assessment, 2005; Mooney et al., 2009). To ensure sustainable development of our planet in the future, it is not only important to monitor ES, their changes and interactions, but as well to identify the drivers causing these changes, how they interact and at which spatial and temporal scales they operate (Rounsevell et al., 2010). A driver of change is defined as an ecological or human-induced factor that affects ecosystem processes, functioning and consequently ES directly or indirectly (Millennium Ecosystem Assessment, 2005). Drivers of change can be direct having an explicit effect on ecosystems, such as land use and land cover change, climate change, pollution and invasive species. Indirect drivers of change are more diffuse such as demographic, economic, sociopolitical, cultural as well as scientific and technological drivers. They can act at different spatial and temporal scales, which adds complexity in monitoring their impacts on ES. In this thesis, we mainly focus on direct drivers of change (from here on referred to as "drivers of change") and how they influence ES.

The two most studied direct drivers of ES change are land use change (Lawler et al., 2014) and climate change with a particular focus on temperature and precipitation (Runting et al., 2017; Scholes, 2016). Assessing impacts of individual attributes of drivers (e.g. precipitation, temperature, landscape fragmentation, and land use change) on ES results in detailed information, but might not provide a comprehensive investigation. Multiple attributes of drivers of change affect ES often simultaneously and make a comprehensive assessment necessary. For example, climate change may decrease food supply in certain areas due to decreasing precipitation and increasing temperatures, even though increased CO_2 concentration in the atmosphere may fertilize photosynthetic rates and thus net primary production (Rosenzweig et al., 2014). In general, the investigated impacts of climate and land use change on ES have mainly been negative, but varied across services, drivers, and assessment methods

(Runting et al., 2017; Scholes, 2016). In particular, food supply is projected be strongly negatively affected by climate and land use change in the future (Challinor et al., 2014; Rosenzweig et al., 2014). However, these determined causal relationships between ES and drivers of change were based on future projections of ES change by land use and climate scenarios (Lawler et al., 2014; Martinez-Harms et al., 2017; Nelson et al., 2010; Rosenzweig et al., 2014), while only few studies have determined the effects of drivers of change on actually measured ES trends (Egarter Vigl et al., 2016; Guerra et al., 2016; Nelson et al., 2013; Schirpke et al., 2013). Special attention has been given to climate change impacts on provisioning services (particularly food supply, raw material and freshwater) and carbon sequestration as regulating service, whereas cultural services received the least attention (Runting et al., 2017). All in all, the understanding of actual impacts of multiple direct and indirect drivers of change on ES is still limited, in particular for regulating and cultural services.

1.2 Remote sensing for ecosystem service assessment

Remote sensing (RS) is the "science of acquiring information about an object without being in direct physical contact with the object" (Jensen, 2007). RS measures the surface properties of an object through its interference (i.e. scattering, absorption and emission) with electromagnetic radiation. Therefore, RS plays a key role in studying environmental processes spatially (Pettorelli et al., 2014; Skidmore et al., 2015). RS is considered as an indispensable tool with unique capabilities for ES assessments, in particular for services that are characterized by ecosystem surface properties. For instance, provisioning services such as food supply and timber, and regulating services like carbon sequestration and soil erosion prevention can be assessed by RS (Ayanu et al., 2012; Cord et al., 2017; de Araujo Barbosa et al., 2015). The main advantages of RS for ES assessments are (i) the retrieval of spatially continuous and explicit information about ecosystem properties and functioning; (ii) the regular and long-term temporal extent of RS data, which enables ES monitoring and (iii) the spatial scalability of the retrieved ES information to match policy relevant scales (Cord et al., 2017; de Araujo Barbosa et al., 2015; O'Connor et al., 2015; Pettorelli et al., 2014; Rose et al., 2015).

RS's direct contribution to ES assessment lies within spatially explicit estimations of ecosystem properties, whereas the indirect contribution is using these very properties as input variables to mechanistic ES models. The ES cascade, as a theoretical framework for ES assessments, integrates these aspects (Haines-Young and Potschin, 2010) (Figure 1.3).

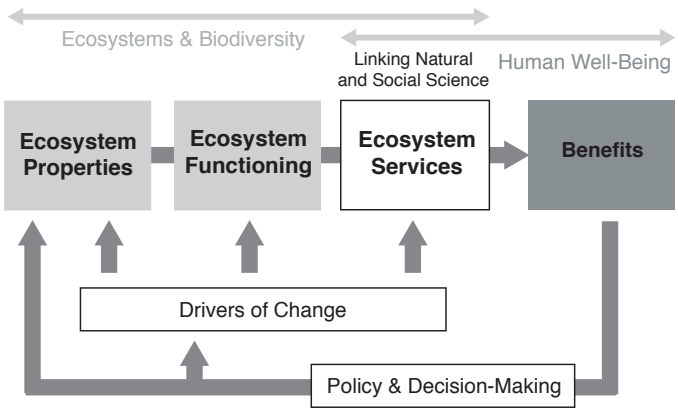

Figure 1.3 Ecosystem service cascade displaying the relationship between ecosystem properites, functioing services and benefits for human well-being (according to Haines-Young and Potschin (2010) and Burkhard et al. (2014)).

It is based on ecosystem properties to estimate ecosystem functioning, ES and ultimately the benefits to human well-being, their (monetary) values and how these different components are affected by direct and indirect drivers of change. Ecosystem properties describe the conditions of an ecosystem, while ecosystem functioning refers to the flow of energy and materials of biotic and abiotic components within and between ecosystems that allows the provision of ES (Mitchell et al., 2015). Mapping and monitoring of all parts of this cascade is needed to ensure sustainable use of ES. So far, RS has already directly and indirectly contributed to individual parts of this cascade by assessing (i) ecosystem properties and functioning, (ii) ES, and (iii) drivers of change and their impacts on ES (Figure 1.4). These three key contributions of RS to the ES cascade will be discussed in the following sections.

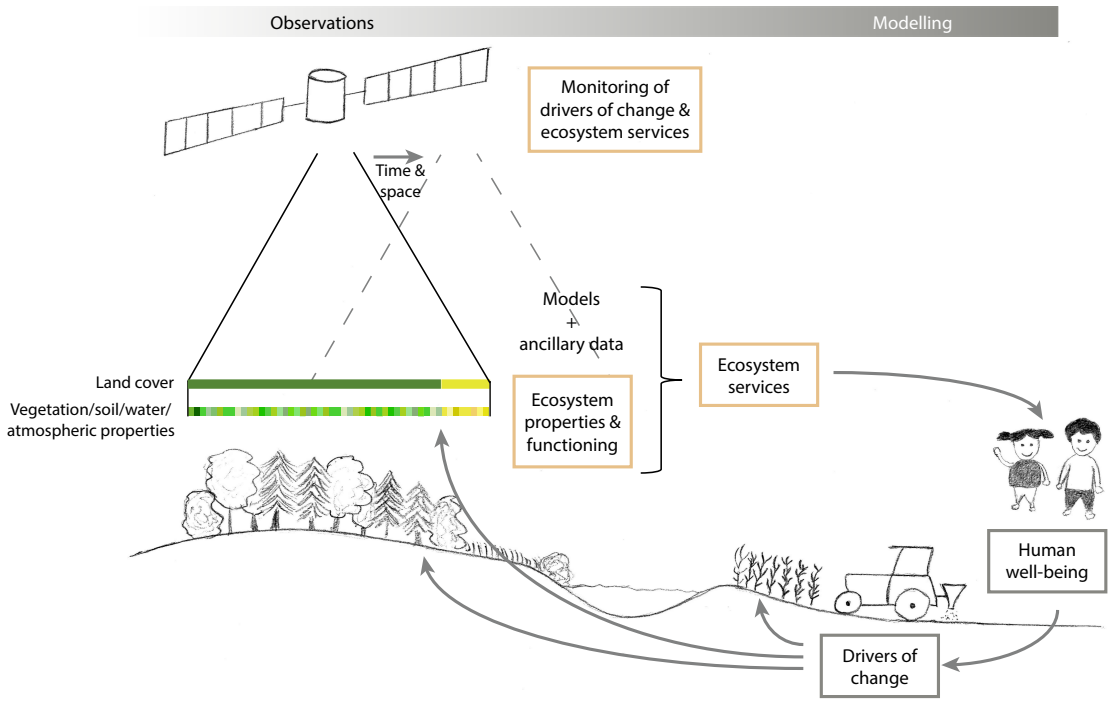

Figure 1.4 Contributions of remote sensing to ecosystem service assessments based on the ecosystem service cascade (adapted from Haines-Young and Potschin, 2010). The orange squares represent the three key contributions referring to (i) mapping of ecosystem properties and functioning, (ii) mapping of ecosystem services and (iii) monitoring of drivers of change and ecosystem services. Each of these three are described in more detail in sections 1.2.1-1.2.3.

1.2.1 Mapping of ecosystem properties and functioning

The first and most direct contribution of RS to the ES cascade is through mapping of ecosystem properties and functioning. Estimations of ecosystem properties and functioning have been extensively assessed in RS and cover a wide range from soil properties (e.g. carbon content, texture and roughness) (Forkuor et al., 2017; Mulder et al., 2011), vegetation traits (e.g. leaf area index, chlorophyll and carotenoid content) (Asner et al., 2017; Gitelson et al., 2006; Homolová et al., 2013) and atmospheric composition (e.g. aerosol and nitrogen dioxide concentrations) (Lamsal et al., 2008; Popp et al., 2012; van Donkelaar et al., 2010) to water quality (e.g. dissolved organic matter, chlorophyll and phytoplankton concentrations, and sediment loads) (Dörnhöfer and Oppelt, 2016; Kiefer et al., 2015; Odermatt et al., 2012). Various methods can be used to estimate these different ecosystem properties such as spectral indices (Gitelson et al., 2006), which are combinations of multiple spectral bands, regression (Homolová et al., 2014) and radiative transfer models (Laurent et al., 2014; Si et al., 2012). Further, ecosystem functioning, in particular of vegetation such as gross primary production (GPP) and net primary production (NPP) were assessed from local to global scale with the help of Earth observation data (Damm et al., 2015; Gitelson et al., 2012; Guanter et al., 2014; Zhao et al., 2005). This extensive RS research on ecosystem prosperities and functioning represents a solid ground for spatially explicit ES assessments. Some ES, in particular provisioning and regulating services, are often more directly linked to RS derived ecosystem properties than cultural ES, which would require a combination of RS and social science data. For example, sun-induced chlorophyll fluorescence (SIF) is related to photosynthetic activity of vegetation and can be used to estimate GPP, an important ecosystem function for many ES (Garbulsky et al., 2013), while leaf area index (LAI) is needed as input data for mechanistic models to estimate air quality regulation (Manes et al., 2016). In contrast, cultural services strongly rely on social science data, e.g. derived from surveys and semi-structured interview and are often only indirectly related to RS data. For instance, satellite-based nighttime stable lights, which are light emissions of human infrastructure, can provide information about the natural- and remoteness of landscapes for estimating recreation as ES (Levin et al., 2015).

1.2.2 Mapping of ecosystem services

In general, ES cannot be appropriately assessed with RS data alone, but require a coupling of Earth observations, ancillary data sets (e.g. *in situ*, literature and census data) and mechanistic models to estimate ES (Cord et al., 2017; de Araujo Barbosa et al., 2015). While the ES cascade suggests to assess ES based on properties and functioning, most ES mapping approaches (56 %) use only RS derived land cover data as basis for their assessment (de Araujo Barbosa et al., 2015). The use of land cover data only comes along with the drawbacks of discrete, non-continuous ES maps that lack spatial heterogeneity in ES (see section 1.1.2).

In contrast, ES studies based on RS derived ecosystem properties result in more spatially explicit and continuous ES maps that capture both quality and quantity of ES (Cord et al., 2017). The most studied ES with RS data are services directly linked to RS measurements such as food supply (Reeves et al., 2005; Xin et al., 2013), carbon sequestration (Kross et al., 2013; Olivas et al., 2010; Wißkirchen et al., 2013) and carbon stocks (Asner et al., 2010; de Araujo Barbosa et al., 2015; Wicaksono et al., 2016). Within the RS based ES studies, it can be distinguished between proxy-based estimations and mechanistic models. Proxy-based approaches estimate directly ecosystem properties and functioning with RS as suggested in the ES cascade (see section 1.2.1), but utilize these derived RS products directly to refer to an ES. Examples are vegetation indices such as the normalized difference vegetation index (NDVI) that are used as estimate for biomass and food supply (Krishnaswamy et al., 2009; Taugourdeau et al., 2014) and as indicator for net CO_2 uptake (Kross et al., 2013). In contrast, mechanistic models follow the ES cascade and use RS estimations of ecosystem properties as input data set combined with ancillary data to assess e.g. air quality regulation with RS-based estimations of air pollutants (Mozumder et al., 2013) and LAI (Manes et al., 2016). However, the coupling of RS based ecosystem properties and functioning with mechanistic ES models has only been rarely implemented and represents a key challenge for ES and RS research.

1.2.3 Monitoring of drivers of change and ecosystem services

The remaining linkage between RS and the ES cascade is the monitoring of drivers of change and of their impacts on ecosystem properties, functioning, and finally ES

(Figure 1.3). With this link, RS extents its contributions to ES mapping by a temporal component that allows detection and monitoring of drivers of change, and how they affect ES over time. RS is a tool that allows repeatable and long-term ES assessments with continuous spatial coverage. Due to an increasing number of satellite missions with open access data in recent years, there is a large variety of RS data sets available with different spectral, temporal and spatial extents (see overview in Vihervaara et al. (2017)) for analyzing past and monitoring future changes in ecosystem properties, functioning and services. For example, studies were able to quantify deforestation rates as a driver of change using time-series of Landsat images and, thus, estimate modifications in provisioning ES such as raw materials (Volante et al., 2012; Wang et al., 2006).

However, as already mentioned in section 1.1.3, ES monitoring is so far still mainly based on changes in land cover due to discrete land cover based ES mapping methods. These approaches neglect the advantages of RS as a tool to assess and map both drivers of change as well as quality and quantity of ES. Therefore, recent ES mapping approaches based on RS-derived ecosystem properties are valuable, in particular for ES monitoring. While several ecological applications of RS have successfully monitored changes in ecological and environmental processes e.g. in phenology (de Jong et al., 2013; Garonna et al., 2016), systematic RS-based ES monitoring is still lacking. However, environmental and RS scientists have identified this gap and call for interdisciplinary approaches to fill this void (Cord et al., 2017, 2015; Karp et al., 2015).

1.2.4 Gaps in remote sensing of ecosystem services

Mapping of ES has developed rapidly in the last decade, but some key challenges still remain. Several recent studies (Bennett et al., 2015; Cord et al., 2017; Karp et al., 2015; Pettorelli et al., 2014a; Tallis et al., 2012) as well as international initiatives such as the Group on Earth Observation Biodiversity Observation network (GEO BON) have emphasized research goals for ES mapping and monitoring. Three selected challenges will be tackled in this thesis.

Firstly, many ES mapping approaches lack spatially explicitness and continuity, which are necessary for detailed ES assessment (Eigenbrod et al., 2010; Hein et al.,

2015; Remme et al., 2014). Policy targets, ecosystem accounting and the evaluation of conservation measures and ES payment schemes require detailed spatial information about ES (Cord et al., 2017; Tallis et al., 2012). However, the majority of the current ES mapping approaches rely on land cover maps in combination with in situ and literature data as well as expert knowledge that result in discrete ES maps (de Araujo Barbosa et al., 2015). Hence, ES research called for the use of RS to assess ES in a spatially explicit manner to fill this gap (Ayanu et al., 2012; Cabello et al., 2012; Cord et al., 2017).

Secondly, ES research needs to fulfill a paradigm shift from pure ES mapping towards monitoring of ES, their trends and interactions. In particular, international and national policy regulation such as the Aichi Biodiversity targets, the Sustainable Development Goals and the EU Biodiversity Strategy 2020 include targets to conserve ES (Geijzendorffer et al., 2017). These require regular and standardized evaluations that measure ES trends and the progress towards these targets, which emphasizes the need for an ES monitoring system (Cord et al., 2017; Karp et al., 2015). First studies monitored ES with land cover based approaches and ancillary data such as census data (Egarter Vigl et al., 2016; Guerra et al., 2016; Karp et al., 2015; Schirpke et al., 2013). However, the better temporal coverage and the need to assess also quantitative changes within ecosystems, call for the use RS data to monitor ES (Cord et al., 2017, 2015).

Thirdly, monitoring ES needs to be combined with identifying the causes of ES change to ensure sustainable development of the Earth in the future. The influence of both climatic and non-climate drivers should be investigated simultaneously to understand their complex interactions and their relative and cumulative impacts on ES change. Determining drivers of change and their impacts on ES can improve decision-making (Martinez-Harms et al., 2017; Runting et al., 2017), e.g. by identifying the most important areas of conservation actions and by reevaluating past conservation measures and agri-environmental payment schemes. This can mitigate current and avoid future ES change and ensure a sustainable development of ecosystems.

1.3 Thesis' aims

Mapping ecosystem services is a key topic for environmental policies and management decisions in landscape planning and conservation. Since ecosystem service mapping and monitoring approaches are often lacking spatial explicitness, scalability and longer temporal extents, there is a need for filling this void. Remote sensing offers versatile capabilities in doing so.

The aim of this thesis is to advance the use of remote sensing as tool for spatially explicit ecosystem service assessments covering different spatial and temporal scales. This thesis focuses on Switzerland, where data availability was adequate to map several ecosystem services covering different spheres. The particular research goals are to:

- map the biophysical part of the ecosystem service cascade at regional scale to assess ecosystem service heterogeneity.
- monitor ecosystem services to determine trends of and interactions (i.e. trade-offs and synergies) between multiple ecosystem services at regional and national scale between 2004 and 2014.
- examine the influence of climatological and non-climatological drivers on the ecosystem service trends determined.

1.3.1 Hypotheses and research questions

The general hypotheses are:

I. that ecosystem service heterogeneity is higher within natural ecosystems such as forest and grasslands compared to highly managed systems like agricultural fields;

II. that regulating ecosystem services decreased in Switzerland in the last decade due to changes in land use and land management, while cultural ecosystem services remained stable;

III. that ecosystem service interactions (i.e. trade-offs and synergies) remained stable over time; and

IV. that ecosystem service trends were more influenced by climatological than by non-climatological effects due to stronger effects of climate change in higher altitudinal areas.

Three research questions (RQ) were formulated as follows:

RQ1: How can remote sensing be used as a tool to map ecosystem services and what is the spatial variability of ecosystem services?

RQ2: What are the spatial and temporal ecosystem service trends and how do the ecosystem services interact (i.e. trade-off and synergy) in Switzerland over time?

RQ3: What are the effects of climatological and non-climatological drivers of change on ecosystem service trends in Switzerland?

1.3.2 Structure of the thesis

Chapter 1 provides definitions and the state-of the art of the thesis, and presents its objectives and research questions.

Chapter 2 describes an approach for spatially explicit mapping of two ES at landscape scale following an observation-based ecosystem service cascade and investigates the spatial heterogeneity in ES supply. Hypothesis I is analysed in this Chapter using airborne imaging spectroscopy.

Chapter 3 tackles larger spatial and temporal scales to implement the monitoring of three regulating and one cultural ES in Switzerland between 2004 and 2014. Mechanistic ES model were combined with satellite remote sensing data to allow the detection of trends and interactions in ES potential supply and flow. Hypothesis II and III were evaluated in this Chapter at national and regional scale.

Chapter 4 determines the influence of climatological and non-climatological drivers on previously detected ES trends. This Chapter tests Hypothesis IV in Switzerland.

Finally, **Chapter 5** summarizes and discusses the main findings of the thesis and provides concluding remarks and an outlook on possible future research directions.

Chapter 2

From instantaneous to continuous: Using imaging spectroscopy and *in situ* data to map two productivity-related ecosystem services

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and has been modified to list all cited references in the Bibliography chapter.

Abstract

Spatially well-informed decisions are essential to sustain and regulate processes and ecosystem services (ES), and to maintain the capacity of ecosystems to supply services. However, spatially explicit ES information is often lacking in decision-making, or exists only as ES maps based on categorical land cover data. Remote sensing (RS) opens new pathways to map ES, in particular biophysical ES supply. We developed an observation-based concept for spatially explicit and continuous ES mapping at landscape scale following the biophysical part of the ES cascade. We used Earth observations in combination with *in situ* data to map ecosystem properties, functions, and biophysical ES supply. We applied this concept in a case study to map two ES: carbon dioxide regulation and food supply. Based on Earth observations and *in situ* data, we determined the ecosystem property Sun-Induced chlorophyll Fluorescence (SIF) to indicate ecosystem state and applied scaling models to estimate gross primary production (GPP) as indicator for ecosystem functioning and consequently carbon dioxide regulation and food supply as ES.

Resulting ES maps showed heterogeneous patterns in ES supply within and among ecosystems, which were particularly evident within forests and grasslands. All investigated land cover classes were sources of CO_2, with averages ranging from 66 to 748 g C m^2 yr^1, after considering the harvest of total above ground biomass of crops and the storage organ, except for forest being a sink of CO_2 with an average of 105 g C m^2 yr^1. Estimated annual GPP was related to food supply with a maize grain yield average of 9.5 t ha^1 yr^1 and a sugar beet root yield of 110 t ha^1 yr^1. Validation with *in situ* measurements from flux towers and literature values revealed a good performance of our approach for food supply (relative RMSE of less than 23 %), but also some over- and underestimations for carbon dioxide regulation. Our approach demonstrated how RS can contribute to spatially explicit and continuous ES cascade mapping and suggest that this information could be useful for environmental assessments and decision-making in spatial planning and conservation.

Authors' contributions (alphabetical order): DB, AD, MES designed the study and developed the methodology. DB, AD, EPL, AR collected the data. DB, AD performed the analysis. All authors wrote the manuscript.

2.1 Introduction

Investigating the impact of human-induced changes on ecosystem processes, properties, and services is essential to gain an understanding of their trade-offs and impacts on human well-being (Tallis et al., 2012). The concept of ecosystem services (ES) is useful to monitor ecosystem changes and their respective implications for human well-being (Karp et al., 2015). Research on ES has strongly increased in the last two decades including the assessment and mapping of ES at different spatial scales (Malinga et al., 2015).

A general concept to assess ES is the ecosystem service cascade (Haines-Young and Potschin, 2010). It considers biophysical ecosystem processes as a basis to quantify ecosystem functioning and ES, and links these to benefits for humans and (monetary) valuation. Implementing the ES cascade by mapping its constituents in a spatially explicit way is crucial to advance environmental assessment and monitoring capabilities, which facilitates decision-making in spatial planning (Maes et al., 2012).

However so far, many ES mapping approaches have focused only on the estimation of ES supply and demand (Martínez-Harms and Balvanera, 2012; Wolff et al., 2015). Additionally, most mapping approaches often rely on land cover maps and ES estimates specific to land cover classes obtained from expert knowledge (Burkhard et al., 2014), literature research (Schirpke et al., 2013), *in situ* measurements (Lavorel et al., 2011), and biophysical models (Nelson et al., 2009; Schulp et al., 2014). Resulting ES maps are often discrete and correlate with underlying land cover. Furthermore, they neglect much of the spatial variability of ES supply within and among land cover classes (Eigenbrod et al., 2010), which can substantially vary due to micro-climatic conditions, small-scale nutrient availability, or species composition. This information, however, is in particular relevant at local and regional scales for policy decisions related to environmental assessment and spatial planning (Hauck et al., 2013; Maes et al., 2012).

For the purpose of natural capital accounting, this was tackled by developing the framework System of Environmental-Economic Accounting (SEEA), which targets the spatial explicit assessment of ecosystem extent, condition, capacity, and service supply and use (Hein et al., 2015). This approach has fostered recent studies on more advanced spatially explicit ES mapping (Remme et al., 2014; Schröter et al., 2014).

Nevertheless, remote sensing (RS) is still rarely used in ES mapping, although it opens new pathways to map ES spatially explicit, in particular the biophysical supply of ES, and enables rapid assessments of ecosystem processes and variables across scales (Ustin et al., 2004). The main use of RS data in ES mapping is still land cover classification (de Araujo Barbosa et al., 2015). Some studies obtain ES from RS data, but apply relatively phenomenological relationships between ES and RS proxies (e.g., vegetation indices) that do not follow the ES cascade and limit their applicability across ecosystems (Krishnaswamy et al., 2009; Taugourdeau et al., 2014). Only a few recent ES mapping studies applied advanced RS products (e.g. gross primary production (GPP), leaf area index (LAI)) to generate spatially explicit ES maps (Homolová et al., 2014; Remme et al., 2014; Schröter et al., 2014). Generally, due to a lack of advanced RS products in ES research, detailed spatially continuous ES maps for environmental assessments and decision-making has been limited (Chan et al., 2006).

The objective of this study is to provide a concept based on a combination of multitemporal RS and *in situ* data together with mechanistic knowledge to map ES at landscape scale in a spatially continuous and explicit way. Our approach follows the modified biophysical part of the ES cascade and builds upon observation data to estimate ecosystem processes and functions required to deriving ES supply. In a case study, we apply this concept to two productivity-related ES. We use sun-induced chlorophyll fluorescence (SIF), a proxy for photosynthetic activity, as ecosystem property to estimate the ecosystem function GPP in a heterogeneous landscape in Switzerland. Mechanistic models are then used to derive the biophysical supply of two ES, namely “carbon dioxide regulation” and “food supply”. Carbon dioxide regulation is defined as net biome production (NBP), with GPP minus autotrophic and hetero- trophic respiration. Food supply is determined as yield of crops per year. We analyze the heterogeneity in ES supply to demonstrate the added value of RS applications for ES mapping.

2.2 Observation-based ecosystem service cascade

The ecosystem service cascade is a conceptual framework to assess ES (Haines-Young and Potschin, 2010). Research on quantifying and mapping each of the cascade components is needed, for example, to determine how climate change influences ecosystems, their services and human well-being (Maes et al., 2012). Therefore, through the input of *in situ* and RS data, we suggest an observation-based ES mapping cascade, adapted from the biophysical part of the ES cascade (Figure 2.1). We use *in situ* and RS data as input to the cascade. These observational data are the basis for estimating ecosystem properties, functioning, ES and ultimately the benefits to human well-being. Ecosystem properties indicate the condition or state of ecosystems. Ecosystem functioning involves the flow of energy and materials of biotic and abiotic components within and between ecosystems that allows the provision of ES (Mitchell et al., 2015). ES are split in biophysical supply and actual flow. Biophysical supply is the full potential of ecological functions in ecosystems to provide a given ES, without consideration of whether humans recognize, use or value that function (Villamagna et al., 2013). In this framework, which we follow in this paper, the actual use of biophysical ES supply by humans is defined as actual flow of ES. Note that, in the SEEA framework, these terms are labeled ecosystem capacity to generate service and flow of ecosystem services (Hein et al., 2016), respectively. In the SEEA, supply by the ecosystem equals use by people (whereas demand from people may be higher) (European Commission et al., 2013).

The adapted, observation-based cascade allows mapping of all its biophysical components, but requires well-established links between the individual components involved. These links are based on mechanistic relationships and biophysical models, which enable transferring mapped ecosystem properties into functions and finally into biophysical ES supply. Spatially explicit mapping of individual parts of the cascade allows detailed analysis of how human activities and climate change influence ecosystem properties, functioning, services, and human benefits. Derived maps can be used to inform decision-makers for applications e.g. in spatial planning and conservation. In the following approach, we test the biophysical part of the proposed frame- work by estimating two productivity-related ES from optical Earth observation data for a heterogeneous landscape in Switzerland.

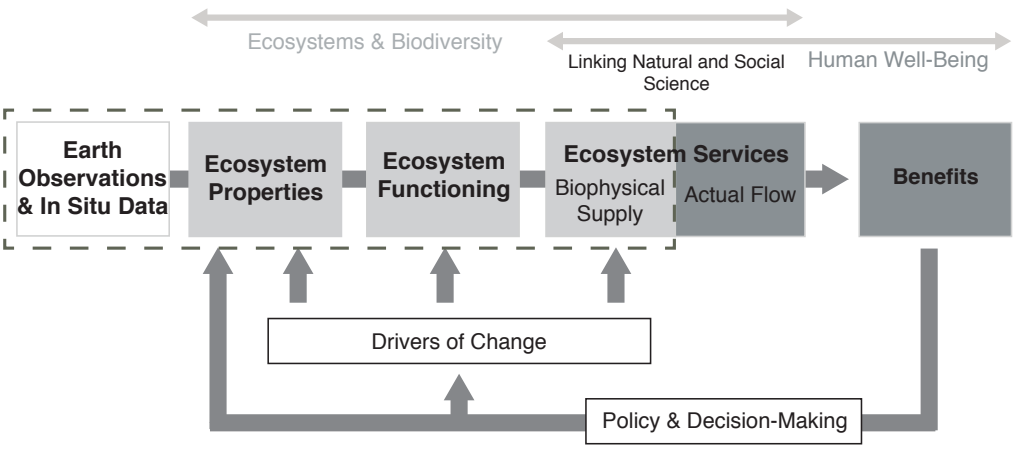

Figure 2.1 Observation-based ecosystem service (ES) cascade for spatially explicit ES mapping (adapted from Burkhard et al. (2014); Haines-Young and Potschin (2010)). This study focuses only on the bio- physical part of this cascade (black dashed rectangle).

2.3 Study area and data

2.3.1 Study area

The study area comprises the Laegern mountain (866 m a.s.l.) and its surroundings (circa 50 km^2), located in the Swiss Plateau, approximately 15 km northwest of Zurich, Switzerland. The area is characterized by heterogeneous land cover/use including grasslands, croplands (principally maize and sugar beet), forests and urban areas (Figure 2.2). The Laegern forest is a mixed deciduous forest with a complex structure and is dominated by beech (Fagus silvatica L.), ash (Fraxinus excelsior L.), sycamore maple (Acer pseudoplatanus L.) and Norway spruce (Picea abies (L. Karst.)) (Eugster et al., 2007; Schneider et al., 2014). The forest hosts a long-term forest ecosystem research site (LWF) of the Swiss Federal Institute for Forest, Snow and Landscape Research (WSL) and is equipped with scientific instrumentation as part of several measurement networks, including FLUXNET (Baldocchi et al., 2001), AERONET (Holben et al., 1998), SPECNET (Gamon et al., 2006), and the Swiss National Air Pollution Monitoring Network (NABEL).

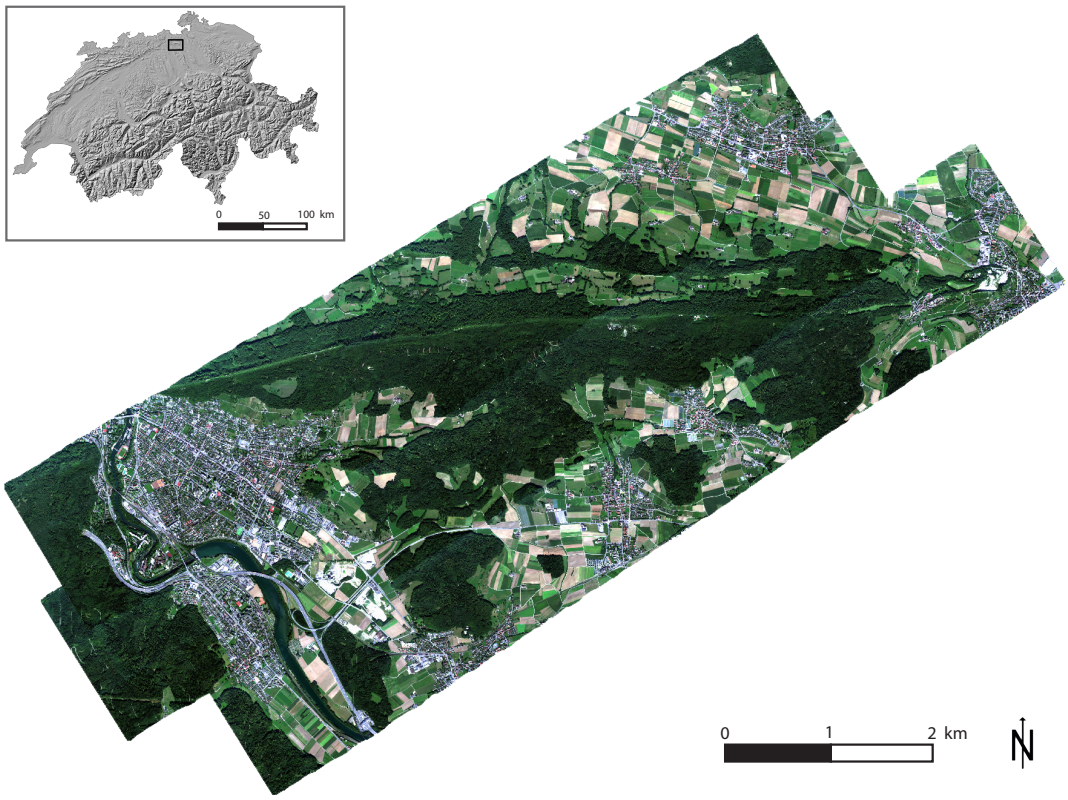

Figure 2.2 Image mosaic of Airborne Prism Experiment (APEX) data of the Laegern area and its location in Switzerland (top left) (DEM courtesy of Swisstopo).

2.3.2 Remote sensing data

2.3.2.1 Airborne imaging spectrometer data

Airborne data were acquired with the Airborne Prism EXperiment (APEX), a dispersive pushbroom imaging spectrometer ranging from 372 to 2540 nm in 532 contiguous spectral bands. The spectral sampling interval (SSI) and the full-width-at-half-maximum (FWHM) are 0.45-7.5nm and 0.86-15.0 nm, depending on the wavelength (Schaepman et al., 2015).

Three APEX flight lines were acquired on 12 July at 8:30 am and 3 September 2013, at 12:00 pm Central European Summer Time (CEST), respectively. The flight height was about 6′400 m a.s.l., resulting in a ground pixel size of 2 m x 2 m. All flight lines were pre-processed and radiometrically calibrated (Hueni et al., 2013), followed by georectification (PARGE, Schläpfer and Richter (2002)) and atmospheric compensation (ATCOR4, Richter and Schläpfer (2002)). The two resulting data

products per acquisition date were georectifly calibrated radiance data and top-of-canopy hemispherical conical reflectance factor (HCRF) data.

2.3.3 Calibration and validation data

2.3.3.1. Eddy-covariance measurements

Eddy-covariance (EC) fluxes (Baldocchi, 2003) were measured continuously in 2013 at the Lägeren mixed forest site (47°28′42″ N, 8°21′52″ E, 682 m a.s.l.) and at the intensively managed Chamau grassland site (47°12′36.8″ N, 8°24′37.6″ E, 393 m a.s.l.; note: Laegern is the name of the mountain range and Lägeren the name used for the Swiss FluxNet site). The EC instrumentation consisted of an open-path infrared gas analyzer (IRGA) (model LI-7500 (Lägeren and Chamau), LI- COR Inc., Lincoln, NE, USA) and a three-dimensional sonic anemometer-thermometer (model HS (Lägeren) and R3-50 (Chamau), Gill Instruments, Lymington, UK). EC measurements were made at a frequency of 20 Hz and processed to half-hourly averages using the EddyPro software (v6.1.0, LI-COR Inc., USA). Flux quality post-processing was done following Vickers and Mahrt (1997). Standardized gap filling and partitioning of NEP into GPP (GPP measured by eddy-covariance tower = GPP_{EC}) and ecosystem respiration (R_{ECO}) was performed using the method from Barr et al. (2004). Measurements of photosynthetic photon flux density (PPFD) quantified in $\mu mol\ m^2\ s^1$ were converted to radiance units ($mW\ m^2\ sr^1\ nm^1$), representing the photosynthetic active radiation (PAR) to be consistent with Monteith's light use efficiency concept and the SIF radiance units.

2.3.3 Modeled maize and sugar beet gross primary production and net ecosystem exchange data

Since EC flux data were not available across the surrounding maize and sugar beet cropland areas, the Soil-Plant-Atmosphere Crop (SPA-Crop, see Sus et al. (2010)) model was used to generate daily and annual estimates of GPP and net ecosystem exchange (NEE) for the 2013 crop season. SPA-Crop simulates the cropland ecosystem carbon cycle at fine temporal (half-hourly) and vertical scales (ten canopy and twenty soil layers). This simulation includes leaf-level processes of

photosynthesis using the Farquhar model (Farquhar and von Caemmerer, 1982) and transpiration using the Penman-Monteith equation (see Jones (2014)).

SPA-Crop was initialised with approximate sowing and harvest dates: 23 April and 22 October for maize, 20 March and 28 September for sugar beet. Half-hourly meteorological observations from the Lägeren site were used to drive SPA-Crop for 2013 growing season and included temperature, atmospheric CO_2, wind speed, short-wave radiation, relative humidity, PAR, precipitation and atmospheric pressure. To provide a realistic simulation of carbon allocation, SPA-Crop also includes different carbon partitioning schemes for maize and sugar beet, which were based on empirical observations of crop growth cycles detailed in Penning de Vries et al. (1989). Specifically, this carbon partitioning describes the fraction of assimilated carbon that is allocated amongst the roots, leaves, stem and storage (i.e. grain) organs as a function of development stage. The development stage is calculated as the accumulation of daily development rates − a function of temperature and photoperiod (Streck et al., 2003).

2.4 Methods

2.4.1 Ecosystem property: retrieving sun-induced chlorophyll fluorescence from APEX data

Sun-induced chlorophyll fluorescence (SIF) is a faint energy signal emitted by plants in the spectral range from 650 to 850 nm. SIF is a by-product of photosynthesis and is considered the most direct measure of photosynthetic activity, providing a mechanistic approach to estimate ecosystem productivity (Damm et al., 2010; Frankenberg et al., 2011; Guanter et al., 2014). SIF was retrieved at 760 nm by exploiting the broad O_2-A oxygen absorption band according to Damm et al. (2015) for both APEX acquisition dates (12 July and 3 September 2013).

We masked two parallel lines along track per flight line due to wires installed on the sensor (Schaepman et al., 2015). These wires are needed for geometric monitoring during in-flight calibration. For some applications the missing information below those wires can be interpolated, however for the sensitive retrieval of sun-induced

chlorophyll fluorescence, it was more accurate to not interpolate those areas but to mask them instead.

2.4.2 Ecosystem functioning: modeling and mapping of gross primary production

GPP calculations comprised of three processing steps, (i) the estimation of instantaneous GPP maps (GPP_{inst}) from APEX SIF values, (ii) the retrieval of daily GPP maps (GPP_{day}) based on GPP_{inst} and modeled GPP (GPP_{mod}) or measured GPP_{EC} diurnals, and (iii) the estimation of annual GPP maps (GPP_{ann}) based on GPP_{day} maps and annual GPP_{mod} or GPP_{EC} data.

2.4.2.1 Instantaneous gross primary production maps based on sun-induced chlorophyll fluorescence

Estimating GPP_{inst} from SIF requires ecosystem-specific functions (Damm et al., 2015; Guanter et al., 2012; Verrelst et al., 2016). Therefore, we used a supervised support vector machine (SVM) classification with a radial basis function kernel and 10fold cross validation (Mountrakis et al., 2011) to derive seven land cover classes from the APEX images. Forest, grassland and croplands (i.e. maize and sugar beet) were classified as target land cover classes for the GPP estimation (overall accuracy = 83.6 %, kappa value = 0.8; for further details on the SVM classification, see Appendix 2A.1).

We estimated GPP_{inst} by deriving and applying three ecosystem- specific hyperbolic GPP-SIF functions according to Damm et al. (2015) for similar Swiss ecosystems:

$$GPP_{cropland} = \frac{34.60 \cdot SIF}{0.08+SIF} \qquad R^2 = 0.88 \qquad rRMSE = 3.4\% \qquad (2.1)$$

$$GPP_{grassland} = \frac{36.51 \cdot SIF}{0.37+SIF} \qquad R^2 = 0.68 \qquad rRMSE = 22.3\% \qquad (2.2)$$

$$GPP_{forest} = \frac{37.37 \cdot SIF}{0.78+SIF} \qquad R^2 = 0.52 \qquad rRMSE = 14.7\% \qquad (2.3)$$

The SVM classification was used to relate Eqs. (2.1–2.3) to corresponding image areas. The $GPP_{cropland}$ function (Eq. (2.1)) was applied to maize and sugar beet, since

SIF-GPP relationships are rather robust across crop types (Guanter et al., 2014, 2012).

Resulting maps of GPP_{inst} represented both a snapshot in time for 12 July and 3 September 2013, whereas decision-makers in land management and conservation are interested in larger periods of years to decades. We consequently applied a temporal scaling to account for this mismatch.

2.4.2.2 Daily and annual aggregation of instantaneous gross primary production maps

GPP_{day} was estimated per pixel for the days of overflight using land cover specific half-hourly modeled GPP data (GPP_{mod}) for maize and sugar beet, whereas the measured GPP data (GPP_{EC}) was used for forest and grassland.

A two-hour running average was applied to smooth the diurnal GPP_{EC} and GPP_{mod} curves. Resulting curves were then linearly scaled to the pixel specific GPP_{inst} values considering the moment of the APEX flight (Figure 2.3a). GPP_{day} for 12 July and 3 September 2013 was finally obtained as integral of the scaled GPP_{EC} or GPP_{mod} curves on a per pixel basis.

The annual sum of GPP 2013 (GPP_{ann}) was calculated per pixel using both GPP_{day} estimates for 12 July and 3 September 2013, as well as annual GPP_{EC} or GPP_{mod} data (Figure 2.3b). A harmonic function with mean and three frequencies (year, half-year, 4 months) was fitted to the annual GPP_{EC} or GPP_{mod} time series of each selected land cover class using the R packages “zoo” (Zeileis and Gorthendieck, 2005) and “bfast” (Verbesselt et al., 2010a, 2010b). This fitted harmonic curve was scaled for each pixel by minimizing their least squares to the two known GPP_{day} values. GPP_{ann} per pixel was calculated as integral of scaled GPP_{EC} or GPP_{mod} curves.

GPP_{ann} estimates were validated by comparing average values and standard deviations (SDs) per land cover class with GPP_{EC} or GPP_{mod} data as well as values from publications that investigated similar land cover classes under comparable climatic conditions.

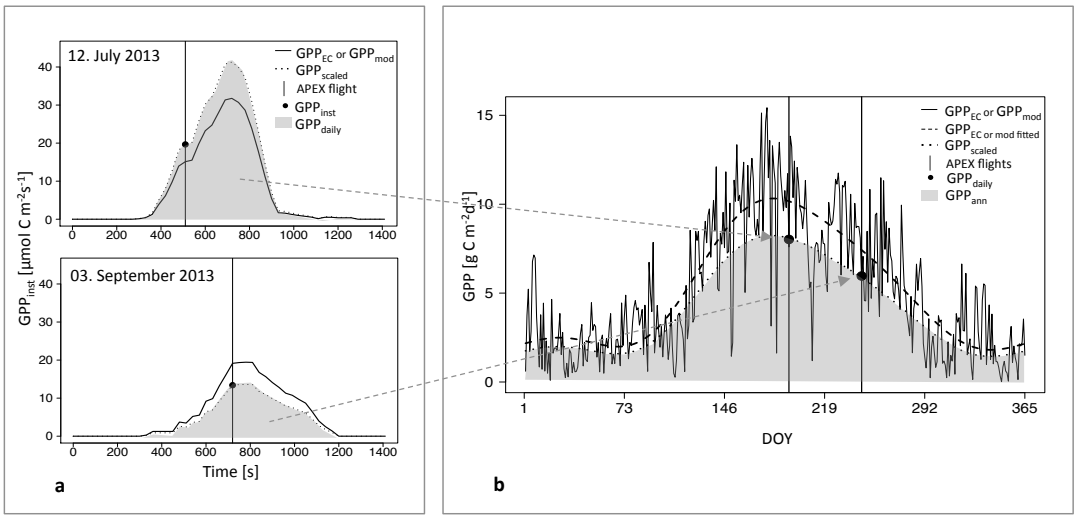

Figure 2.3 Temporal scaling of GPP measurements. (a) Scaling of diurnal EC flux tower or modeled GPP data (black line: GPP_{EC} or GPP_{mod}, black dotted line: GPP_{scaled}) based on an instantaneous GPP value (black dot). Daily GPP is estimated as integral of the diurnal curve of 12 July 2013 (a top) and 3 September 2013 (a bottom). (b) Scaling of the fitted annual GPP flux tower or modeled curve (black line: GPP_{EC} or GPP_{mod}, black broken line: $GPP_{EC\ or\ mod\ fitted}$, black dotted line: GPP_{scaled}) by minimizing the least squares of the two daily GPP estimates from 12 July and 3 September 2013 (black dots). The integral of the scaled annual curve represents the annual GPP value.

2.4.3 Ecosystem services: mapping the biophysical supply of ecosystem services

Various ES can be mechanistically associated to GPP. GPP is the actual uptake of atmospheric carbon dioxide (CO_2) due to the process of plant photosynthesis, which facilitates plant growth. GPP is thus related to biomass production and is a principle driver of CO_2 exchange between the atmosphere and biosphere. Both ES, food production and CO_2 regulation, are important for human well-being and their supply will be a key issue in the context of global change.

2.4.3.1 Carbon dioxide regulation

We defined CO_2 regulation as net biome production (NBP). Positive NBP values indicate a sink of CO_2, while negative values express a source of CO_2. NBP was calculated as the difference between GPP_{ann} and the sum of annual autotrophic respiration (R_A), heterotrophic respiration (R_H) and harvest (H):

$$NBP = GPP - R_A - R_H - H = CUE \cdot GPP_{ann} - R_H - H, \quad \text{with} \quad (2.4)$$

$$R_A = (1 - CUE) \cdot GPP_{ann} \quad (2.5)$$

R_A can be approximated as a function of the ecosystem specific carbon use efficiency (CUE) and GPP (Gifford, 2003). CUE values were obtained from literature for the three selected land cover classes (Appendix 2A.2 Table 2A2.1). All selected CUEs represented average values since CUE depends on land management, species composition and stand age (DeLucia et al., 2007; Gilmanov et al., 2007; Kutsch et al., 2010). R_H is the heterotrophic respiration, which was estimated by two different approaches to assess uncertainties in R_H estimations. First, we computed R_H for the study area according to Raich et al. (2002) ($R_{H\text{-}Raich}$) as soil respiration based on 1 km spatial mean monthly air temperature (T_a) and mean monthly precipitation (P) data of 2013 (MeteoSwiss, 2013):

$$R_{H-Raich} = 1.250 \cdot e^{(0.05452 \cdot T_a)} \cdot \left[\frac{P}{4.259+P}\right] \quad (2.6)$$

Second, R_H was partitioned from the EC flux tower measurements following Barr et al. (2004) ($R_{H\text{-}EC}$). H is the harvest of biomass exported from ecosystems. Note that in the ecosystems that we examined relatively little carbon is removed due to wood harvest or fires, and we therefore did not consider these types of carbon removal from the ecosystems. The harvest of grassland was 349 g C m^2 yr^1 and was the average of annually measured grassland harvest at the Chamau test site in 2006 and 2007 (Zeeman et al., 2010). Crop harvest was a result of the SPA-Crop model and was computed as sum of carbon in foliage, stem and storage organ (i.e. grain for maize and root for sugar beet). Harvest of maize was 806.73 g C m^2 and 845.46 g C m^2 for sugar beet.

Note that our NBP estimation did not consider any carbon inputs into ecosystems such as organic fertilization due to a lack of spatially explicit data. The validation of resulting annual NBP values was based on comparison of average NBP per land cover class to either EC flux tower or modeled NBP data.

2.4.3.2 Food supply

We defined food supply as crop yield, i.e. annual crop production per hectare per year. Adapted from Xin et al. (2013), we estimated maize and sugar beet yield (in t ha^{1} yr^{1}) as a function of annual GPP and crop specific parameters:

$$Yield_{maize} = \frac{GPP_{ann} \cdot HI}{RSR \cdot MC \cdot 1000}, \quad \text{and} \tag{2.7}$$

$$Yield_{sugar\ beet} = \frac{GPP_{ann} \cdot HI}{RSR}, \tag{2.8}$$

where GPP_{ann} is the estimated annual GPP of each crop, RSR is the root to shoot ratio, HI is the harvest index in percent (indicating harvestable crops per above ground crop biomass in the case of maize and per total biomass in the case of sugar beet, which has a relatively high root-to-shoot ratio), and MC is the moisture content in percent, which here is only relevant for maize. All variables were crop specific and mostly obtained from *in situ* measurements (Appendix 2A.2 Table 2A.2.2). We differentiated only two crop types, maize and sugar beet, as those were the only crops growing in the study area on July 12 and September 3 2013. For sugar beet, we estimated beet yield, whereas the grain yield was estimated for maize. For validation, we compared resulting yield estimates with in situ data and reported literature values.

2.5 Results

2.5.1 Sun-induced chlorophyll fluorescence

We retrieved SIF for APEX acquisitions on 12 July, 8:30 am CEST, and 3 September 2013, 12:00 pm CEST (Figure 2.6a). For all land cover classes, SIF average values and their distribution were higher in September than in July, except for grassland, which remained nearly equal (Figure 2.4a). This difference is mainly due to different observation times and related variations of irradiance intensities rather than indicating physiological differences. Forest, grassland and sugar beet had similar SIF averages values in July, while maize was lower. In September, the SIF average of maize was the highest, followed by sugar beet, forest and grassland. This indicates highest photosynthetic activity in maize compared to the other land cover classes.

2.5.2 Gross primary production

Annual GPP was heterogeneously distributed across the study area and showed substantial differences among ecosystems but also within them (Figure 2.6b). GPP_{ann} was particularly high in agricultural areas and grasslands north of the Laegern forest (i.e., 1650 g C m^2 yr^1 for selected agricultural areas, 3305 g C m^2 yr^1 for selected grasslands) and in the southern part of the study area. Lower GPP_{ann} values were located in the Laegern forest, in particular on the north-facing slopes (i.e., 1490 g C m^2 y^1) due to reduced irradiance, and in the southern and eastern part of the forest dominated by coniferous trees (i.e., 1213 g C m^2 yr^1). The mean value of GPP_{ann} for maize was 1560 g C m^2 yr^1, sugar beet was 1744 g C m^2 yr^1, grassland was 2997 g C m^2 yr^1, and forest was 1625 g C m^2 yr^1 (Figure 2.4b). The spatial variability of GPP_{ann} within ecosystems was large in grasslands and forests, while it appeared rather low in croplands (Figure 2.6b). The GPP_{ann} for croplands showed smaller variations of GPP (Figure 2.4b). In contrast, forests and grasslands showed a high intraclass variation of GPP_{ann}, with the largest value range for grassland.

Validation of GPP_{ann} estimations with EC flux tower and modeled GPP data revealed relative errors of 7.5 % for sugar beet, 13.2 % for forest, 13.4 % for grassland and 18.3 % for maize (Figure 2.4b).

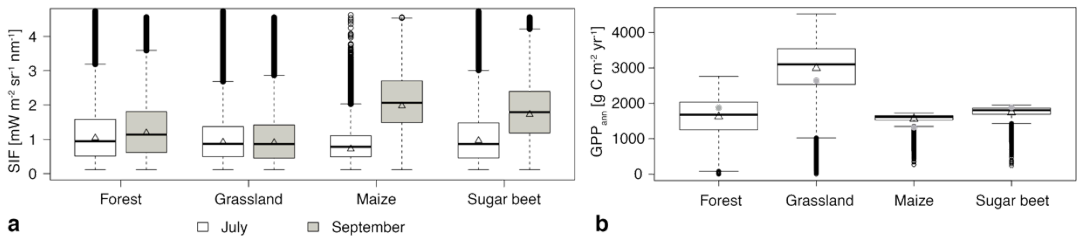

Figure 2.4 Boxplots of sun-induced chlorophyll fluorescence (SIF) from 12 July and 3 September 2013 (a) and annual GPP (b) for selected land cover classes (median = black line inside the box, mean = triangle, grey dot = EC flux tower measurement or modeled GPP data for validation). It must be noted that differences in SIF between July and September are partly caused by the different observation times (i.e. 8:30 CEST on 12 July and 12:00 CEST on 3 September).

2.5.3 Ecosystem service supply

2.5.3.1 Carbon dioxide regulation

Carbon dioxide regulation estimated from NBP based on $R_{H\text{-}EC}$ was more similar to EC flux tower measurements or modeled NBP data when compared to NBP estimated with $R_{H\text{-}Raich}$ (Appendix 2A.3 Table 2A.3.1). Highest absolute deviations to the reference data were found for grassland and maize, with 177 g C m^2 yr^1 and 143 g C m^2 yr^1, respectively. Therefore, we used $R_{H\text{-}EC}$ to map carbon dioxide regulation. The resulting map showed a small-scale heterogeneity in ES supply in the study area (Figure 2.6c). The Laegern forest was characterized by higher carbon dioxide regulation supply in the southwest, intermediate supply in the center and lower supply on the northern slopes and the southeast of the Laegern mountain. The carbon dioxide regulation supply in grasslands was also characterized by a heterogeneous spatial distribution (Figure 2.6c). In contrast, maize and sugar beet fields in the northern and southern part of the study area showed more homogeneous carbon dioxide regulation. Highest average carbon dioxide regulation was obtained for forest with 105 g C m^2 yr^1, followed by grassland with 66 g C m^2 yr^1, maize with -652 g C m^2 $yr^1$and sugar beet with 748 g C m^2 yr^1 (Appendix 2A.3 Table 2A.3.1). All land cover classes were sources of CO_2 except for forest, which was a CO_2 sink. The

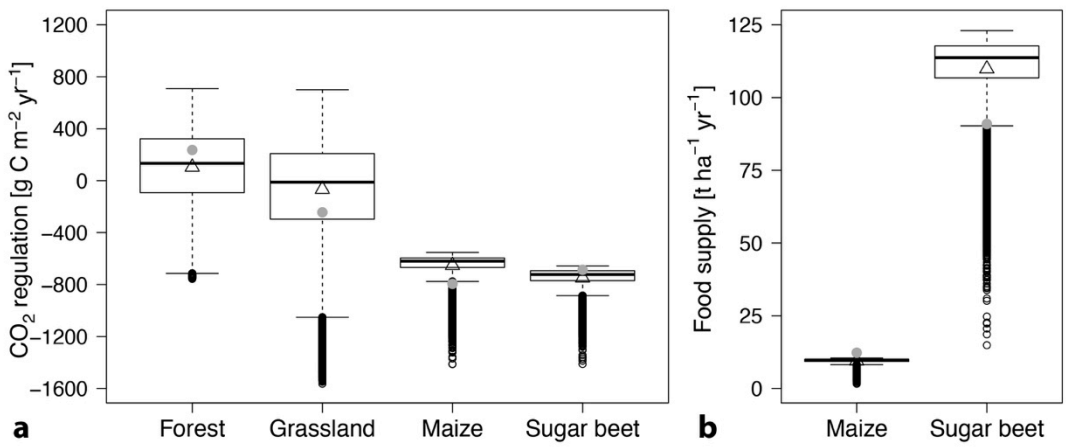

Figure 2.5 Boxplots of carbon dioxide regulation (NBP based on $R_{H\text{-}EC}$) (a) and food supply (b) for the selected land cover classes (median = black line inside the box, mean = triangle, grey dot = validation data (i.e. (a) eddy-covariance flux tower measurements or modeled data, (b) *in situ* data)).

variation of carbon dioxide regulation was largest in grasslands and forests, followed by maize and sugar beet (Figure 2.5a).

2.5.3.2 Food supply

Contrasting to the small-scale supply of carbon dioxide regulation, patterns of the ES food supply were rather homogeneous among and within crop fields (Figure 2.6d). Sugar beet showed a higher food supply, with an average beet yield of 110 t ha^{1} yr^{1}, compared to a maize grain yield of 9.5 t ha^{1} yr^{1} indicating a bipolar distribution (Figure 2.5b, Appendix 2A.3 Table 2A.3.1). Note that we compare the two crops in one Figure, but their economic values are quite different with sugar beet the lower value crop (on a per kg basis). Comparison with *in situ* yield measurements of 2015 from agricultural fields close to our study area resulted in relative errors of 20 % for sugar beet and 23 % for maize.

2.6 Discussion

2.6.1 Observation-based ecosystem service cascade mapping

We suggested an observation informed ES cascade using *in situ* and Earth observation data to assess and map ecosystem properties, functioning and biophysical ES supply. Our attempt to demonstrate the applicability of the proposed ES mapping cascade and to validate retrieved results required a heterogeneous set of *in situ*, Earth observation and literature data as well as several assumptions. The results of this approach and its uncertainties will be set in context with other studies and discussed hereafter.

2.6.1.1 Sun-induced chlorophyll fluorescence

SIF was used as an informative proxy of an ecosystem property related to actual rates of plant photosynthesis, providing the base to mechanistically constrain estimates of GPP at ecosystem level. Recent advances in this research field indicate sensitivity of SIF for plant photosynthesis (Rossini et al., 2015; Yang et al., 2015) and empirically

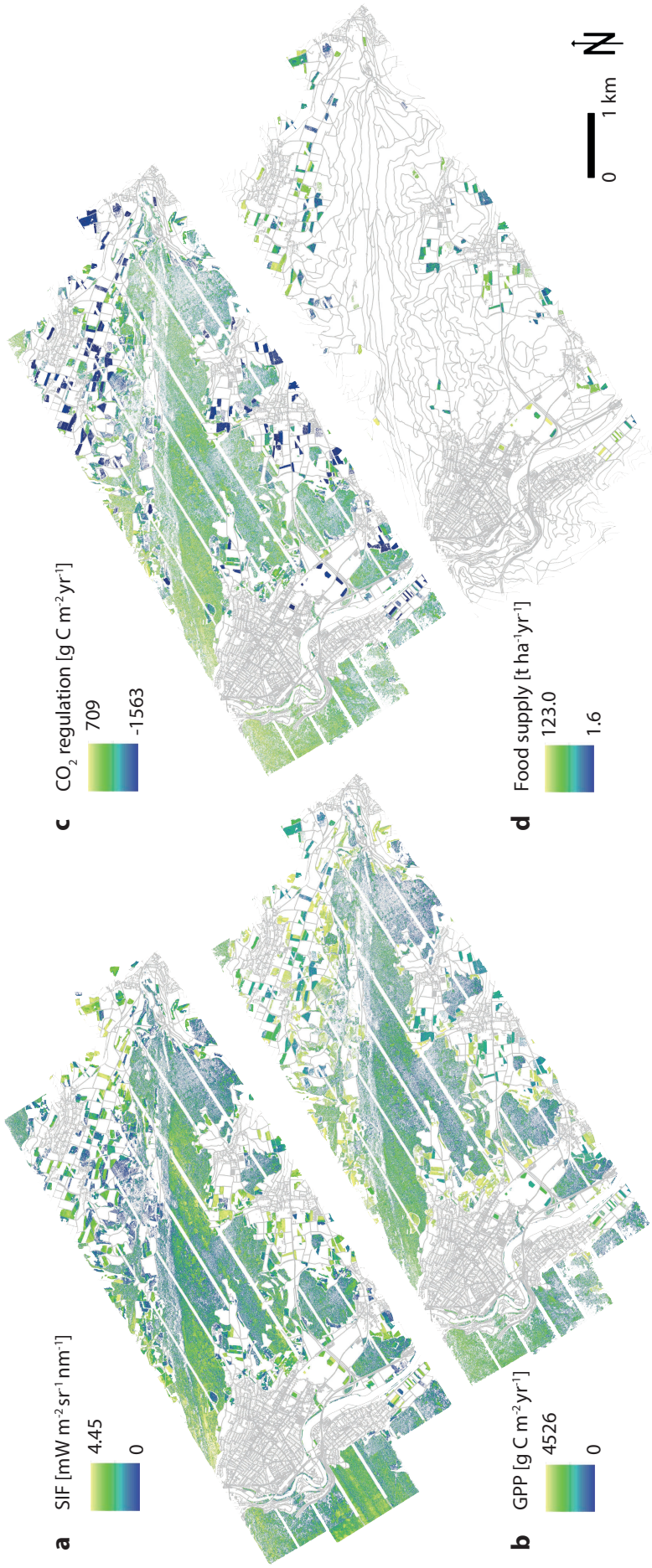

Figure 2.6 Mapped ES cascade at Laegern area: (a) Sun-induced chlorophyll fluorescence (SIF, from 3 September 2013) as ecosystem property; (b) annual gross primary production (GPP) as ecosystem function; (c) carbon dioxide regulation (based on $R_{H\text{-}EC}$) and (d) food supply. The white areas are not vegetated or data retrieval was not accurately possible. These areas were excluded from all further analysis. The grey lines indicate roads and urban areas.

demonstrate relationships between SIF and GPP (Damm et al., 2015; Parazoo et al., 2013). Estimated SIF on 12 July and 3 September 2013 had higher values at the south-facing Laegern mountain ridge when compared to those on the north-facing slopes. This difference in SIF is due to higher irradiance. Lower SIF values on 12 July compared to the 3 September are a consequence of an earlier acquisition time of 8:30 CEST compared to 12:00 CEST. Our approach related actual GPP estimates based on SIF to EC based GPP diurnals and allowed a reliable retrieval of daily and annual GPP estimates. Further, it was demonstrated that spatial SIF pattern of forest remain relatively stable during the day and the season (Yang et al., 2015). In consequence, the differing acquisition times do not complicate the interpretation of seasonal GPP differences as obtained in our results.

The retrieved SIF value ranges between 0 and 5 mW m^2 nm^1 sr^1 are comparable to other research carried out in similar environments (Damm et al., 2015). It must be acknowledged that the spectral resolution of used APEX data is sub-optimal for SIF retrievals, likely biasing absolute SIF values. However, since the intention of this study was to demonstrate the feasibility of the mapping approach, these issues do not compromise conclusions that can be drawn from our results. In the long run, dedicated fluorescence missions such as the European Space Agency's 8th Earth explorer mission Fluorescence Explorer (FLEX) or existing atmospheric missions such as OCO-2 (Frankenberg et al., 2014) or Sentinel-5p (Guanter et al., 2015) will provide more accurate SIF values on a regular basis.

2.6.1.2 Gross primary production

We used a mechanistic link between SIF and GPP to calculate instantaneous maps of GPP for the study area. This link between *in situ* measured SIF and GPP was first documented in Damm et al. (2010) with further empirical (Frankenberg et al., 2011; Lee et al., 2013; Parazoo et al., 2014) and mechanistic analyses (Damm et al., 2015; Lee et al., 2015; Verrelst et al., 2016), thus confirming the existence and robustness of this species-specific link across temporal and spatial scales. However, instantaneous GPP estimates for maize and sugar beet could have been improved by using crop specific SIF-GPP equations instead of an average crop equation (cf. Damm et al.

(2015) for a detailed discussion on the impact of vegetation type on SIF-GPP relationships).

For GPP_{inst} and GPP_{day} we found satisfying agreements with reference GPP measurements (data not shown). Estimates of GPP_{ann} were based on combining SIF and temporal scaling using EC flux tower and modeled GPP time series. The complexity and diversity of the data involved resulted in relatively low deviations between predicted and *in situ* measured GPP.

Differences in GPP_{ann} for crops were distinct, with the lowest relative errors for sugar beet (7.4 %) and highest for maize (18 %). This can be partly explained by inaccuracies in the parameterization and thus the estimation of annual GPP time series of maize using the SPA-Crop model. Additionally, agricultural management practices in the study area (e.g., the application of fertilizer and pesticides, organic farming) were not considered, but can substantially influence agricultural productivity. Kutsch et al. (2010) analysed the carbon balance of differently managed crop rotation sites under differing climatic conditions across Europe based on EC measurements. Reported estimates of GPP_{ann} for multiple croplands ranged from 807 g C m^2 yr^1 to 1624 g C m^2 yr^1. Our estimations fit within this range, but are closer to the higher, indicating fertile and productive agricultural fields in our study area.

We found a 13.2 % underestimation of GPP_{ann} for forest when compared to GPP_{EC} and literature values (Etzold et al., 2011). This is within the measurement accuracy and the distribution of GPP_{ann} values (Figure 2.4b). Differences of GPP_{ann} and GPP_{EC} could be partly explained by difficulties in comparing gas exchange measurements with optical re- mote sensing based estimates of functional vegetation information over complex structured vegetation canopies (Damm et al., 2015). Measured SIF signals from APEX correspond only to top-of-canopy, while GPP_{ann} obtained from an EC flux tower integrates over the full vertical extend of the canopy. The larger spatial heterogeneity of forest GPP_{ann} on the southern slope of Laegern compared to the northern is justified by differing irradiance pattern on both, the north- and south-facing slopes.

GPP_{ann} derived for grassland showed an uncertainty of 13.4 % and the highest average values (i.e., 2995 g C m^2 yr^1), indicating high productivity. Although the high productivity found for the study area corresponds well to the reference grassland in

Chamau, an intensively managed grassland (Merbold et al., 2014; Zeeman et al., 2010), probably not all grasslands in our study area are as intensively managed. For example, estimates of GPP_{ann} of other Swiss EC flux tower grassland sites (e.g. Oensingen) and years are less productive, with an average of 1893 g C m^2 yr^1 (Wißkirchen et al., 2013). This indicates an overestimation of GPP_{ann} for some grasslands in the study area, since we assume partly extensive grassland management. Potential reasons could be a mismatch between the annual GPP_{EC} curve used to scale daily to annual GPP values, which is likely not representative for actually applied management such as grass cutting or fertilization in our study area.

In the near future, some of the observed uncertainties are likely to be reduced by replacing the scaling approach based on the GPP time series of one site with novel approaches combining photosynthesis models with time series of new Earth observation data such as SIF (Lee et al., 2015; Parazoo et al., 2014; van der Tol et al., 2014, 2009). Such approaches will allow simulating GPP spatially and temporally continuous across ecosystems and minimizing uncertainties when scaling instantaneous GPP measurements to daily and annual values.

2.6.1.3 Ecosystem services

The observation-based ES cascade finally allowed providing maps of ES. Estimates of carbon dioxide regulation deviated stronger from EC flux tower measurements compared to GPP_{ann}. Forest carbon dioxide regulation, for example, was half of the reference value and only a quarter compared to the average NBP measured at Lägeren between 2005 and 2009 (Etzold et al., 2011). NBP of maize and sugar beet displayed differences compared to the modeled reference value but are in the range of carbon dioxide regulation values as reported for European croplands (Ceschia et al., 2010; Kutsch et al., 2010). Nevertheless, our NBP estimates of croplands were probably too small, representing a too big source of CO_2 as harvest included all carbon of the above ground biomass (i.e. sum of carbon in foliage, stem and storage organ). In general, the amount of remaining crop residues on a field depends highly on the agricultural practice. Stem and foliage of maize sometimes remain on the field and are tilled in the ground as plant litter manure. However, it is as well common to harvest the total above ground biomass of maize and to use it as feed. In this case only some

small amounts of crop residues (e.g. maize stubble) remain on the field. In literature, carbon removal due to harvest ranges for maize from 479 to 794 g C m^2 for grain and 327 g C m^2 for stem and foliage, and for sugar beet from 630 to 850 g C m^2 for the sum of above ground biomass and storage organ (Ceschia et al., 2010). Our harvest data used for maize and sugar beet fit to the literature values, but are at the upper end of the range.

For grassland, estimated carbon dioxide regulation was higher than the EC flux tower data and values from literature (Zeeman et al., 2010). As discussed for GPP_{ann}, this value highly depends on the management of the system, which was intensive at our EC flux tower site Chamau, but likely not representative for all fields in our study area. Consequently, carbon dioxide regulation was overestimated at landscape scale.

The diversity of the calibration data used partly explains the divergence of RS derived and *in situ* carbon dioxide regulation. Further, we used a fixed CUE value per land cover class, although it was shown that CUE varies depending on land management, species composition and stand age (DeLucia et al., 2007; Gilmanov et al., 2007; Kutsch et al., 2010). A main source of uncertainty, however, is the difficulty to accurately estimate spatial-temporal distribution of heterotrophic respiration, a carbon flux contributing to NBP and determining carbon dioxide regulation. We tested two approaches, one suggested by Raich et al. (2002) that employs spatial mean monthly air temperature and precipitation data to estimate soil respiration and the other used *in situ* measured heterotrophic respiration. The difference between both approaches (cf. Appendix 2A.3 Table 2A.3.1) reveals that coarse approximations of relevant environmental factors (i.e., 1 km aggregated data) could not capture the small-scale spatial variation of heterotrophic respiration of different land cover classes in a heterogeneous landscape like our study area. *In situ* measured values revealed the differences in heterotrophic respiration among the selected land cover classes, but lacked information on their spatial distribution. Consequently, improved estimates of heterotrophic respiration are necessary and require spatial information of sufficient granularity to account for small scale and short-term changes of this important carbon flux.

We found a good agreement of estimated food supply with *in situ* values. However, the use of *in situ* data and literature values to parameterize mechanistic equations limits the applicability of this approach to other crops, the operational implementation

for larger areas, and might affect the absolute accuracy and suppress spatial heterogeneity of ES supply. Nevertheless, obtained yields are in agreement with *in situ* (i.e. field measurements) and published Swiss agricultural statistics (Swiss Federal Office for the Environment, 2013). Hence, for both ES models used, increased reliability can be achieved when moving from discrete to continuous parameterizations of mechanistic ES calculations.

2.6.2. Remote sensing to assess spatial heterogeneity in ecosystem service supply

Derived maps of biophysical components of the suggested observation-based ES cascade revealed spatial heterogeneity of all components both within and among ecosystems. This spatial explicitness and continuity are key advantages of this approach and allow for an advanced understanding of how climate change influences ecosystem properties, function, ES and consequently human well-being. Such detailed results and differences in ecosystems can provide an added value for ES mapping that has not been exploited so far. Key for the suggested approach is the use of RS, suggesting a straightforward adaptation to other ES as well, for example erosion control, microclimate regulation, and freshwater availability. It is important to consider the need to use appropriate RS and *in situ* data to strictly follow the cascading approach. This requirement also limits the range of applications, in particular for cultural ES (e.g., spiritual, aesthetic, educational, and recreational). Nevertheless, using RS in ES mapping can provide an added value for stakeholders and decision-makers and the resulting maps could facilitate decision-making e.g. in environmental planning and conservation.

2.7 Conclusion

We demonstrated the capability of RS in general and imaging spectroscopy in particular to assess ecosystem properties, functioning and services following an observation-based ES cascade. Moving the assessment of ES from empirical relationships and land cover maps towards mechanistic approaches in combination with RS and *in situ* data is important to continuously assess ES. Resulting maps of

ecosystem properties, functions, and ES provide complementary information of ecosystems and how they are impacted by global environmental change. We based the proposed ES assessment approach on SIF and GPP exemplifying carbon dioxide regulation and food supply as ES. We conclude on the importance of accurately linking ecosystem properties, functions and ES in the observation informed ES cascade. This requires more in depth mechanistic understanding of these links and more ac- curate data, as in our case on heterotrophic respiration. Furthermore, alternative strategies and approaches of mapping ES using multi-temporal and satellite imaging spectroscopy data should be developed in future research to widen the range of ES that can be connected to earth observation.

Acknowledgements

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Supporting Information

Appendix 2A.1: Classification of land cover classes

The estimation of instantaneous GPP from APEX derived SIF is ecosystem-specific. Therefore, we used a supervised support vector machine (SVM) classification scheme to derive seven land cover classes from the APEX image of September 3, 2013. Forest, grassland and croplands (i.e. maize and sugar beet) were our target land cover classes for the GPP estimation, while three other classes (i.e., bare soil, recently germinated vegetation, and cast shadow) were excluded from further analyses. The class "recently germinated vegetation" is characterized by plants, which started to grow after agricultural fields were harvested. In the APEX data of September 3 these fields were clearly distinguishable. Although instantaneous GPP of the land cover class "recently germinated vegetation" could be estimated, it was not considered in the ES supply estimations, as we had no information on the land use at the beginning of the vegetation period.

We classified the seven land cover classes on the APEX image using a SVM classification with a radial basis function kernel and 10-fold cross validation (Mountrakis et al., 2011). Model calibration and validation were based on reference data obtained from a random sampling approach considering the entire image. The reference data contain training (i.e., 900 pixels per class) and independent validation data (i.e., 1000 pixels per class). The SVM classification was filtered using a three by three majority-moving window to remove scattered false classified pixels (Stuckens et al., 2000). Additionally, forest areas, grasslands, and croplands smaller than 5000 m^2 were excluded to account for misclassified pixels within these land cover classes.

Classification accuracy was determined with the independent validation data using kappa values, overall accuracy, as well as producer and user accuracies (Story and Congalton, 1986). We used the R package "caret" (Version 6.0-64) for SVM classification (Kuhn, 2015). The resulting classification was also used for APEX data of July 12, 2013, as all classified land cover classes were invariant between July and September 2013.

The resulting SVM classification differentiated seven land cover classes (Figure 2A.1.1). An accuracy assessment based on independent validation data yielded an overall kappa value of 0.8 and an overall accuracy of 83.6 %. Table 2A.1.1

shows class specific producer and user accuracies. Highest accuracies were achieved for shadow followed by maize. A moderate accuracy was determined for grassland, recently germinated vegetation and sugar beet. The relatively low producer accuracy for grassland can be explained by the heterogeneity in the grassland training data set due to different grassland types such as cut grassland, natural grassland and grassland used for grazing. Low user accuracy of sugar beet can be related to difficulties of the SVM to differentiate between sugar beet and other land cover classes such as grassland and recently germinated vegetation.

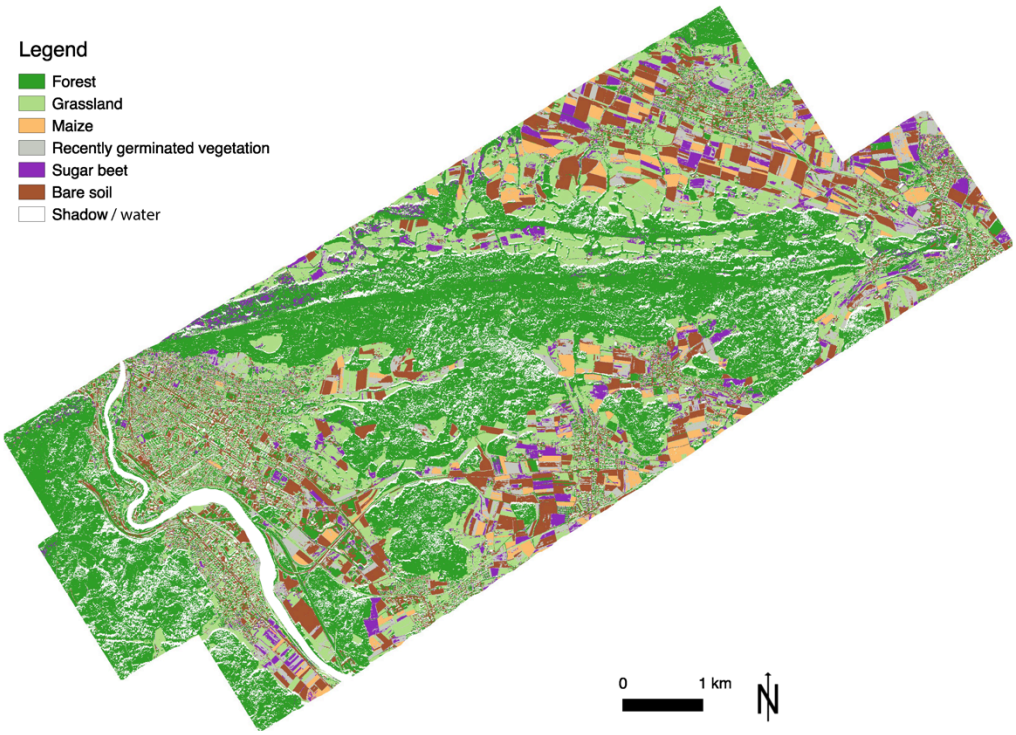

Figure 2A.1.1 Land cover map consisting of seven classes derived from supervised support vector machine classification.

Table 2A.1.1 Producer and user accuracies of SVM classification for the study area (crops divided into maize and sugar beet).

Class	Producer Accuracy (%)	User Accuracy (%)
Grassland	40	71
Forest	97	97
Bare soil	99	86
Recently germinated vegetation	75	87
Sugar beet	76	53
Maize	98	100
Shadow	99	100

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Appendix 2A.2 Biophysical parameters of ES models

Table 2A.2.1 Literature values of ecosystem specific carbon use efficiency.

Land cover class	Carbon use efficiency	Reference
Maize	0.59	Choudhury (2001)
Sugar beet	0.44	Aubinet et al. (2009)
Grassland	0.50	Gilmanov et al. (2007)
Forest	0.53	DeLucia et al. (2007)

Table 2A.2.2 *In situ* measured variables to estimate agricultural yield as food supply for maize and sugar beet.

Crop	Maize	Sugar beet
Harvest index [%]	0.50	0.14*
Root-to-shoot ratio [-]	0.18	2.30*
Moisture content [%]	0.46	-

* From Hoffmann (2006)

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Appendix 2A.3: Statistics of estimated ES

Table 2A.3.1 Mean and standard deviation (SD) of carbon dioxide regulation based on $R_{H\text{-}Raich}$ and $R_{H\text{-}EC}$ and food supply, and their respective reference data (EC flux tower or modeled NBP and measured yield) for selected land cover classes.

Land cover class	CO_2 regulation $R_{H\text{-}Raich}$ [g C $m^{-2}yr^{-1}$] mean ± SD	CO_2 regulation $R_{H\text{-}EC}$ [g C $m^{-2}yr^{-1}$] mean ± SD	EC flux tower or modeled NBP [g C $m^{-2}yr^{-1}$]	Food supply [t $ha^{-1}yr^{-1}$] mean ± SD	Measured yield[1] [t $ha^{-1}yr^{-1}$] mean ± SD
Maize	-656.17 ± 96.96	-651.56 ± 94.78	-794.19	9.46 ± 0.97	12.30 ± 2.87
Sugar beet	-845.90 ± 85.97	-748.43 ± 83.81	-686.60	109.87 ± 12.00	90.94 ± 24.46
Grassland	383.87 ± 358.46	-65.98 ± 358.39	-243.38	-	-
Forest	97.45 ± 281.50	104.94 ± 280.73	236.41	-	-

[1] Measured yield data of maize and sugar beet were from 2015.

Chapter 3

Spatio-temporal trends and trade-offs in ecosystem services: An Earth observation based assessment for Switzerland between 2004 and 2014

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Petchey, O. L. & Schaepman, M. E.

This chapter is based on a peer-reviewed article accepted by Ecological Indicators and has been modified to list all cited references in the Bibliography chapter.

Abstract

Understanding and monitoring pressures on ecosystems and their consequences for ecosystem services (ES) is essential for management decisions and verification of progress towards national and international policies (e.g. Aichi Biodiversity Targets, Sustainable Development Goals). Remote sensing (RS) offers a unique capability to assess ES systematically and regularly across spatial and temporal scales. We aim to evaluate the benefit of RS to monitor spatio-temporal variations of ES by assessing several ES in Switzerland between 2004 and 2014. We coupled mechanistic ES models and RS data to estimate time series of three regulating (i.e. carbon dioxide regulation (CO_2R), soil erosion prevention (SEP), and air quality regulation (AQR)) and one cultural ES (recreational hiking (RH)). The resulting ES were used to assess spatial and temporal changes, trade-offs and synergies of ES potential supply and flow in Switzerland between 2004 and 2014. Resulting ES trends showed diverse spatial patterns across Switzerland with largest changes in CO_2R and AQR. ES interactions revealed a scale and elevation dependency. We identified weak to strong synergies between all ES combinations except for trade-offs between CO_2R–AQR and AQR–RH at Swiss scale. Spatially, all ES interactions revealed a heterogeneous mix of synergies and trade-offs within Switzerland.

Our results demonstrate the strength of RS for systematic and regular spatio-temporal ES monitoring and contribute insights to the large potential of RS, which will be extended with future Earth observation missions. Derived spatially explicit ES information will facilitate decision-making in landscape planning and conservation and will allow examining progress towards environmental policies.

Authors' contributions (alphabetical order): DB, AD, MES designed the study and developed the methodology. DB collected the data and performed the analysis. All authors wrote the manuscript.

3.1 Introduction

Safe planetary boundaries of several Earth system processes, in particular biogeochemical flows of phosphorus and nitrogen, land use change, climate change, and freshwater use have already been crossed (Steffen et al., 2015). Associated and often irreversible environmental change influences ecosystems and the services they supply in multiple ways. These modifications emphasize the need and importance of monitoring and protecting ecosystems and ecosystem services (ES) (Millennium Ecosystem Assessment, 2005). Additionally, assessing changes in ecosystem services has become crucial for current environmental policies like IPBES (Díaz et al., 2015), the Aichi Biodiversity Targets (Convention on Biological Diversity, 2010), the European Union Biodiversity Strategy (European Commission, 2011) and the Sustainable Development Goals (United Nations, 2015) that aim to ensure sustainable development of socio-ecological systems. Such environmental policies particularly require regular and systematic monitoring of ES to improve management decisions and to verify progress towards environmental policies and effectiveness of conservation measures and payment schemes (Scullion et al., 2011). However, required ES information is often complex as not only a single ES is of interest, but knowledge of multiple ES and their interactions is needed (Bennett et al., 2009). Additionally, appropriate methods linking ES to processes in ecosystems are still lacking (Lavorel et al., 2017). As consequence, dedicated monitoring systems for regular and systematic monitoring of multiple ES are still missing.

One promising concept for assessing spatio-temporal changes in ecosystems and related ES to foster sustainable development is ecosystem accounting (European Commission et al., 2013). It is considered as an accounting framework that complements existing national accounts thereby acting as global comprehensive statistical standard for measuring economic activity. The development of ecosystem accounting as complementing framework to national accounts involves valuing the contribution of ES to human well-being in biophysical and economic terms. Ecosystem accounting relies on approaches to spatially measure and continuously monitor the conditions of ecosystems, their capacity to sustainably provide ES, and the effective flow of ES to society (Hein et al., 2015). Ensuring sustainable development of socio-ecological systems requires understanding and assessing the spatial and temporal changes in ES for improved management decisions and

verification of progress towards national and international policies (e.g. Aichi Biodiversity Targets, Sustainable Development Goals). To accomplish this, the guidelines of the System for Environmental Economic Accounts Experimental Ecosystem Accounting (SEEA EEA) particularly stress the need to capture spatial heterogeneity in ecosystems and related services (European Commission et al., 2013). The Ecosystem Accounting framework differentiates between ES flow, the amount of a service used by humans in a given time period, and potential supply defined as amount of a service generated by an ecosystem irrespective of its human use (Hein et al., 2016). Investigating both aspects is important to assess independently changes in ecosystem processes and in human use of ES.

Spatial ES modeling and mapping approaches have rapidly evolved in the past two decades covering a wide range of ES mapping approaches across various spatial scales (Burkhard et al., 2012; Maes et al., 2016; Malinga et al., 2015; Rabe et al., 2016; Schröter et al., 2014). Information of intra- and inter-ecosystem variability and related ES is essential to ensure sustainable development of socio-ecological systems, e.g. by improved management decisions and by the implementation of ecosystem accounting. However, there is a lack of accurate spatially explicit ES quantifications at larger scales (Lavorel et al., 2017). This justifies the request for new ES mapping approaches providing quantitative ES data compared to existing proxy indicators or models relying on land cover and expert judgments (Remme et al., 2014).

We argue that the use of remote sensing (RS) data offers new pathways for ecosystem accounting, particularly for the monitoring of status and trends in ecosystem conditions and ES across space and time. The large potential of RS for monitoring and mapping ecosystem services and biodiversity is already widely recognized (Cord et al., 2017, 2015; Pettorelli et al., 2016; Skidmore et al., 2015; Tallis et al., 2012). The integration of RS in ES models is, however, less elaborated compared to ES assessments without RS data. Three main gaps for ES monitoring can be identified: i) Land use and land cover are still the most common RS information used in ES models (e.g. InVEST (Sharp et al., 2016), ARIES (Villa et al., 2014)) (de Araujo Barbosa et al., 2015), often resulting in underestimated intra-class heterogeneity in ES supply due to the assumption of the same biophysical values per land cover class (Eigenbrod et al., 2010). Some recent studies started extracting the spatial explicitness of RS data to overcome this problem and consider spatial heterogeneity of ES in their mapping

approaches (Braun et al., 2017; Remme et al., 2014; Schröter et al., 2014; Strauch and Volk, 2013). Nevertheless, there remains a lack of quantitative approaches that link ES to ecosystem processes at larger scales (Lavorel et al., 2017). ii) Many studies neglect the advantage of large temporal coverage by RS products. Nearly 30 % of ES studies using Earth observation data, are based upon monotemporal imagery and 56 % of the studies covered only 10 years or less (de Araujo Barbosa et al., 2015). iii) A particular lack is related to RS based assessments of multiple ES across space and time.

With this study we make a contribution to a requested paradigm shift in ES assessment from purely mapping to spatially explicit monitoring of ES (Cord et al., 2017, 2015; Karp et al., 2015; Tallis et al., 2012). In this study, we aim at demonstrating the contribution of RS to quantitative and spatially explicit ES monitoring for sustainable development and natural resource management. This is demonstrated by investigating spatio-temporal trends in potential supply and flow of ES in Switzerland between 2004 and 2014 using the ecosystem accounting framework. We applied mechanistic models in combination with RS data to estimate three regulating services, i.e. CO_2 regulation (CO_2R), soil erosion prevention (SEP), air quality regulation (AQR), as well as the cultural service recreational hiking (RH). We use the obtained ES to assess spatio-temporal changes in ES potential supply and flow and to quantify ES trade-offs and synergies in Switzerland between 2004 and 2014.

3.2 Material and methods

3.2.1 Study area

Switzerland is located in the center of Europe with an area of approximately 4.1 million ha and an altitudinal range from 196 m to 4634m above sea level (a.s.l.). The country can be divided into four biogeographical regions: Jura, Swiss Midlands, Central Alps, and Southern Alps (Figure 3.1). Land use and land cover are characterized by urban and settled area covering 7.5 % of the surface area, agriculture (35.8 %), wooded areas (31.3%), surface water (4.3 %), and remaining natural environment (21.0 %) (Swiss Federal Statistical Office, 2016). During the last 25

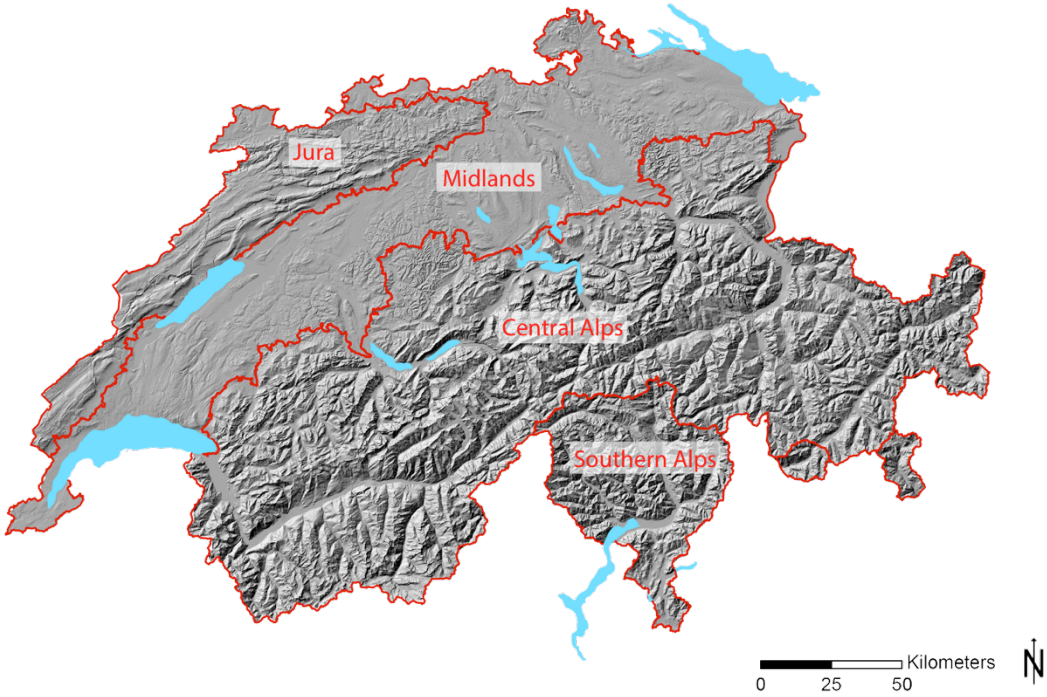

Figure 3.1 The study area Switzerland with its different geographical regions. A digital elevation model is used as background and spatial data of Swiss lakes (courtesy of Swisstopo).

years, Swiss landscape has constantly changed due to urban expansion particularly in the Swiss Midlands (Jaeger and Schwick, 2014), and due to land abandonment in Alpine areas followed by expansion of wooded areas (Gellrich et al., 2007).

3.2.2 Remote sensing data

Several products derived from data of the Moderate-resolution Imaging Spectroradiometer (MODIS) and the Defense Meteorological Satellite Program Operational Linescan System (DMSP-OLS) were used to assess the various ES and their spatio-temporal changes.

We utilized the MODIS products "Land cover" (MCD12Q1) (Friedl et al., 2010), "NDVI" (Normalized Difference Vegetation Index) (MOD13A1) (Didan, 2015), "LAI" (Leaf Area Index) (MOD15A2H) (Myneni et al., 2015), and "NPP" (Net Primary Production) (MOD17A3H) (Running and Zhao, 2015), all of them belonging

to product version #006 except for the “land cover” product that belongs to version #055. The data have a spatial resolution of 500 m, a temporal resolution of 16 days (16 day-composites) and cover the time period from 2004 until 2014

(cf., https://lpdaac.usgs.gov/dataset_discovery/modis/modis_products_table).

Additionally, we used a time series of the DMSP-OLS nighttime product “stable lights” (version #4) with a global spatial resolution of 0.0083° from 2005 to 2013 (https://www.ngdc.noaa.gov/eog/dmsp/downloadV4composites.html). The product contains light emissions from cities, towns, and other sites with persistent lighting. Temporary light emissions such as fires and background noise are excluded from the product. We resampled the data set to 500 m spatial resolution using bilinear interpolation.

3.2.3 Ecosystem services models

Following SEEA EEA (European Commission et al., 2013), ES are contributions of ecosystems to human benefits. The aim of ecosystem accounting is to measure ES in a manner that facilitates integration with national accounts (Hein et al., 2015). In this context, the SEEA EEA framework suggests the quantification of ecosystem assets to particularly determine ES potential supply and in combination with the use by people, ES flow (European Commission et al., 2013). Potential supply and flow differ for provisioning, regulating and cultural ES. While there is often a spatial match for most provisioning services between ES potential supply and flow related to resource extraction, potential supply of regulating services covers sometimes larger areas than ES flow. Soil erosion prevention of a meadow, for example, provides both ES potential supply and flow as it is used for farming. However, the same potential supply provided by an alpine grassland in an inaccessible area results in no ES flow, as the ES potential supply is not used by people. Similarly, cultural ecosystem services often provide a high potential supply in services, such as aesthetic beauty or recreation, but the, partly conscious, lack of infrastructure to access untouched nature results in low ES flow.

We selected four ES according to four criteria and mapped their potential supply and flow between 2004 and 2014. The selection criteria include that ES should (i) cover different Earth spheres (e.g. biosphere, atmosphere and pedosphere), (ii) be relevant

to the well-being of Swiss inhabitants, (iii) contain a RS input, which allows regular and spatially explicit monitoring of the respective ES, and (iv) require additional *in situ*, modeled or literature data that were available at the appropriate spatial and temporal resolution.

3.2.3.1 Carbon dioxide regulation (CO_2R)

The process of plant photosynthesis determines vegetated ecosystems as crucial for the global carbon cycle: Vegetation acts as sink of atmospheric CO_2 and allows mitigating trends and effects of rising atmospheric CO_2 by human activity (i.e. burn of fossil fuels, land cover change, deforestation). Here, we define carbon dioxide regulation (CO_2R) of ecosystems as net ecosystem production (NEP) of all vegetated areas (i.e. forest, agricultural and other ecosystems) to receive a spatially continuous ES map. We consider neither harvest losses due to a lack of temporally and spatially explicit harvest data nor carbon losses from fires, since they are very rare in Switzerland. NEP (g C m^{-2} yr^{-1}) was calculated as difference of annual net primary production (NPP) (g C m^{-2} yr^{-1}) and annual soil respiration (R_S) (g C m^{-2} yr^{-1}) as:

$$\mathrm{CO_2R} = NEP = NPP - R_S, \text{ with} \tag{3.1}$$

$$R_S = \sum 1.250 \cdot e^{(0.05452 \cdot T_a)} \cdot \left[\frac{P}{4.259+P}\right] \tag{3.2}$$

Annual NPP was derived from the annual MODIS NPP (MOD17A3H) product. R_S was estimated as yearly sum according to Raich et al. (2002) based on 1 km spatial mean monthly air temperature (T_a) (°C) and mean monthly precipitation (P) (mm) data from 2004 to 2014 (MeteoSwiss, 2013) and was downscaled to 500 m.

The actual use of this ES is global and relatively indirect, so we assume that the potential supply of carbon dioxide regulation directly translates into its ES flow, determining ES potential supply and flow as the same.

3.2.3.2 Air quality regulation (AQR)

Air pollution can severely affect human health in various ways (Künzli et al., 2010; Rückerl et al., 2011). One of the most important and well-documented atmospheric pollutants in Switzerland is particulate matter (PM_{10}) (Eeftens et al., 2015). PM_{10} is

defined as particulate matter with a diameter of less than 10 µm and is detrimental to human health already at low concentrations. Vegetation has the ability to remove PM_{10} from the atmosphere (Manes et al., 2016) and consequently decrease human health risks (Powe and Willis, 2004). We define the potential supply of air quality regulation (AQR) as yearly sum of vertical capture of PM_{10} per m^3 by an ecosystem as (adapted from Manes et al. (2016) and Nowak (1994)):

$$AQR = \sum C \cdot V_d \cdot LAI \cdot T \cdot 0.5, \quad (3.3)$$

where C is the PM_{10} concentration in the air ($\mu g\ m^{-3}$), V_d is the dry deposition velocity for PM_{10} ($m\ s^{-1}$), LAI is the leaf area index ($m^2\ m^{-2}$), T is the time step (s) corresponding to one year, and 0.5 is the suspension rate of deposited PM_{10} returning back to the atmosphere (Zinke, 1967).

We allocated different V_d values according to different land cover types following (Remme et al., 2014): 0.0080 $m\ s^{-1}$ for needle-leaved forest, 0.0032 $m\ s^{-1}$ for broad-leaved forest, 0.0010 $m\ s^{1}$ for grassland, cropland and other nature, and 0 $m\ s^{-1}$ for water and urban infrastructure.

The Swiss Federal Office for the Environment (2015) provided modeled annual PM_{10} concentrations for Switzerland based on 72 measurement stations resulting in a spatial resolution of 200 m. For the LAI, the MODIS LAI product (MOD15A2H) was used. Potential supply of air quality regulation is defined as removal of PM_{10} from the atmosphere due to vegetation. ES flow is the actual use of this service, so the utilization of filtered air by humans. Following Hein et al. (2016), potential ecosystem service supply registers strictly what the ecosystem does, in a given biophysical environment, without considering if people benefit from the service or not. Air filtration leads to an actual service supply (and a benefit), if there are people living in the area, where pollution levels are lower because of air filtration by vegetation. If air filtration leads to cleaner air in a place, where no-one is living or (virtually) no-one is ever visiting then it is a potential but not an actual service. Therefore, we calculated the flow of AQR as its potential supply but restricted to urban (i.e. residential) areas including a buffer area of 1 km following (Remme et al., 2014). In these areas people live and spend most of their time, consequently consuming the ES AQR. We assume that in other areas this ES flow is negligible small due to low duration of stay by people.

3.2.3.3 Soil erosion prevention (SEP)

Soil erosion by water is still a main environmental problem and causes strong economic losses in agriculture by the loss of fertile topsoil (CEC, 2006). In Switzerland, the problem was widely recognized but research mainly focused on soil erosion risk (Prasuhn et al., 2013). The ES of soil erosion prevention (SEP) has not been investigated so far in Switzerland. We define the potential supply of SEP (E_{pot}) in tonnes per hectare (t ha^{-1}) according to Guerra et al. (2016) as the difference between the structural impact Y (i.e. the total soil erosion impact in absence of SEP = potential soil erosion) and the remaining impact not mitigated by this ES β (i.e. the remaining soil erosion that was not regulated by SEP) as:

$$E_{pot} = \Upsilon - \beta = \Upsilon\,(1 - C_V), \qquad \text{with} \tag{3.4}$$

$$\Upsilon = R \cdot K \cdot LS \tag{3.5}$$

ϒ is the structural impact (tha^{-1}), C_V is the vegetation cover (-), R is the rainfall erosivity (MJ mm ha^{-1} h^{-1}), K is the soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}), and LS describes the effect of topography on soil erosion (-) with L representing the impact of slope length and S quantifying the effect of slope steepness. Vegetation cover C_V was estimated as function of NDVI as suggested by Guerra et al. (2016):

$$C_V = \exp\left[-2 \cdot \frac{NDVI}{(1-NDVI)}\right]. \tag{3.6}$$

Annual rainfall erosivity and soil erodibility of Switzerland were provided as spatial data with a resolution of 500 m by the Joint Research Center (JRC) European Soil Data Centre (ESDAC) (Panagos et al., 2015a; Panagos et al., 2014). We estimated the length and steepness factor according to Panagos et al. (2015b) by using a digital elevation model (DEM) of 25 m spatial resolution as input to the terrain analysis module of the SAGA (System for Automatic Geoscientific Analyses) software, which incorporates a multi flow algorithm to estimate LS (Pilesjö and Hasan, 2014). The ES flow of soil erosion prevention (E_{flow}) was equal to its potential supply in agricultural areas and forests. In these ecosystems humans use SEP and benefit from avoided soil loss on their agricultural fields and in forests. Therefore, the flow of this ES was estimated by clipping its potential supply to agricultural areas and forest using the MODIS land cover product (MCD12Q1).

3.2.3.4 Recreational hiking (RH)

The Swiss landscape is well known for its nature recreation potential. Particularly hiking is a popular outdoor activity, while about 44 % of 15 to 74 years old Swiss inhabitants regularly spend time on the approximately 65'000 km long network of hiking trails in Switzerland (Bundesamt für Strassen (ASTRA) und Schweizer Wanderwege, 2015).

We estimate the potential supply of recreational hiking (RH_{pot}) using a model based on a simple ratio of hiking path density as infrastructural parameter and RS derived nighttime stable lights (NSL), a parameter describing the natural- and remoteness of a landscape, as:

$$RH_{pot} = \frac{PD}{1 + NSL}, \tag{3.7}$$

where PD is the hiking path density as length of path per pixel (km). NSL (-) are derived from DMSP-OLS satellite data.

The flow of RH (RH_{flow}) was estimated based on its potential supply in combination with preferences of hikers derived from the Flickr data as:

$$RH_{flow} = \frac{PD \cdot (1 + PUD)}{1 + NSL}, \tag{3.8}$$

where PD was multiplied by PUD, the photo user days estimated per pixel (-). All variables of the equations were scaled between 0 and 1.

We estimated PUD by applying the recreation model of InVEST (Sharp et al., 2016), a widely used open source ecosystem service toolbox. The model calculates the distribution of photo-user-days (PUD) for recreation based on geo-tagged photographs posted on the photo-sharing website Flickr (www.flickr.com) (see Appendix 3A.1 Figure 3A.1.1). This results in a proxy for visitation frequency and preferences of hiking trails (Keeler et al., 2015; Wood et al., 2013). We defined April to October as core hiking season and summarized PUD during these months. We normalized PUD by the average PUD per year to account for the increasing use of Flickr data during our investigation period. As Flickr data were first available in 2005, RH was computed from 2005 to 2013.

Table 3.1 Overview of investigated ecosystem services (ES), their potential supply (= P) and flow (= F) as well as their remote sensing inputs. The transformations were applied to provide comparability for the trade-off and synergy analysis.

Section	ES	ES potential supply	ES flow	Remote sensing input	Transformation	
					P	F
Regulating	CO_2 regulation (CO_2 R)	Sequestered CO_2 [kgCm^{-2}yr^{-1}]	Same as potential supply	MODIS: Land cover (MCD12Q1), net primary production (MOD17A3H)	$x^{0.5}$	$x^{0.5}$
	Air quality regulation (AQR)	PM_{10} removal [µg m^{-2}]	PM_{10} removal [µg m^{-2}] in urban areas	MODIS: Leaf area index (LAI) (MOD15A2H)	$x^{0.25}$	$x^{0.25}$
	Soil erosion prevention (SEP)	Potential soil erosion [t ha^{-1}]	Potential soil erosion [t ha^{-1}] in agricultural areas	MODIS: NDVI (MOD13A1) Digital elevation model	$x^{0.2}$	$x^{0.2}$
Cultural	Recreational hiking (RH)	Density of hiking paths combined with nighttime stable lights as natural-/remoteness parameter [-]	Recreational hiking potential weighted with visitation preference [-]	DMSP-OLS: Nighttime stable lights	$x^{0.6}$	$x^{0.15}$

3.2.4 Spatial and temporal analysis of ecosystem services

Annual spatial maps of retrieved ES were used to quantify the temporal variability of each ES in Switzerland and its four regions (Figure 3.1). Therefore, we calculated the annual mean value of ES potential supply and flow per biogeographical region and represented results as time series.

Trends of ES between 2004 and 2014 were analyzed for all ES and spatially represented. We applied a linear regression analysis per pixel; slopes above zero indicated an increase of the respective ES over time and values below zero showed a decreasing ES trend. Results were represented in a map and were later utilized to calculate average ES trends for Switzerland and each Swiss region.

Additionally, annual total ES potential supply and flow of Switzerland were calculated and stratified by four ecosystem types, namely forest, grassland, agriculture and urban. The ecosystem types were selected from raster-based CORINE land cover 2006 (250 m) (Steinmeier, 2013) and resampled to 500 m spatial resolution based on majority count of the pixel and its nearest neighbours. We selected this classification to stratify the different ecosystem types during the investigation period, since land use change per year in Switzerland is negligibly small with approximately 1 % (Swiss Federal Statistical Office, 2016).

Annual spatial maps of ES per year were used to estimate ES relationships and their temporal change during the investigation period. We identified ES synergies and trade-offs among ES potential supply and flow for all pair-wise combinations of ES by calculating annual Pearson correlation coefficients. A synergy between two ES occurs if a high value of one ES correspond to a high value of another ES (Bennett et al., 2009). In this case, the Pearson correlation coefficient is larger than 0. A trade-off between two ES occurs if a certain ES value corresponds to an opposite value of another ES (Rodriguez et al., 2006), resulting in a Pearson correlation coefficient smaller than 0. Annual maps of Pearson correlation coefficients per ES pair combination were calculated at national and regional scale and visualized as time series.

Temporal ES synergies and trade-offs were analyzed for all ES pair combinations and spatially represented. A temporal synergy between two ES occurs if both ES fluctuate in synchrony (e.g. both increase) over time, while a temporal trade-off is defined by

opposite temporal trends of two ES (e.g. one increases while the other decreases). We computed temporal synergies and trade-offs using the Pearson correlation coefficient to describe the relationship between two ES time series per pixel. Positive correlations represented temporal synergies between two ES, while negative correlations represented temporal trade-offs. Preprocessing steps of the correlation analysis included a transformation (Table 3.1) and a standardization of each ES (i.e. mean of zero, standard deviation of one), since the variables must have a linear relationship and be normally distributed. The best transformation was selected by checking quantile-quantile plots. Resulting Pearson correlation coefficients per ES pair are presented in a map.

3.3 Results

3.3.1 Spatial and temporal trends in ecosystem services

The temporal analysis of ES in Switzerland revealed changes in yearly averages of both ES potential supply and flow (Figure 3.2). Temporal fluctuations were synchronous between the different Swiss regions. However, temporal variability within a region could be of similar magnitude to differences between regions, e.g. for CO_2R and AQR (Figure 3.2a, d, e). In contrast, temporal fluctuations of SEP and RH within a region were smaller compared to differences between regions (Figure 3.2b, c, f, g). ES potential supply and flow showed slightly differing changes during the investigated period.

CO_2R displayed ES values for Jura and Swiss Midlands above the national average, while Central and Southern Alps had lower values compared to the national average (Figure 3.2a). Temporal CO_2R pattern are consistent across all regions with a decrease in the service since 2011. Temporal variations in SEP potential supply and flow per region were low relative to differences among the regions (Figure 3.2b, c), tough with apparently larger temporal fluctuations in the Southern Alps, where SEP was highest, approximately threefold the Swiss SEP average. AQR showed fluctuating temporal trends in ES potential supply and flow for all investigated regions characterized by an overall decrease, but with two maxima in 2007 and in 2011 (Figure 3.2d, e). Highest average AQR potential supply was found in Jura,

followed by Swiss Midlands, Southern Alps, and Central Alps. AQR ES flow was nearly the same for all regions except for the Southern Alps, which was larger compared to the other regions. RH averages of potential supply and flow slightly decreased during in investigated period with an absolute minimum in 2010 (Figure 3.2f, g). ES potential supply of RH revealed the highest averages for the Central Alps and Southern Alps, followed by Jura and Swiss Midlands. In contrast, averages of ES flow were highest for the Southern Alps and remained relatively stable for Jura and Swiss Midlands within the investigation period.

Spatial patterns of temporal trends in ES potential supply and flow were rather diverse (Figure 3.3, significant trends see Appendix 3A.2). Nearly nationwide, CO_2R showed decreasing trends (Figure 3.3a). Highest negative trends were located in the Southern Alps, while partly increasing trends were detected in the western and eastern Central Alps. SEP potential supply and flow remained relatively stable in large areas of Switzerland between 2004 and 2014, with spatially scattered increasing trends in the

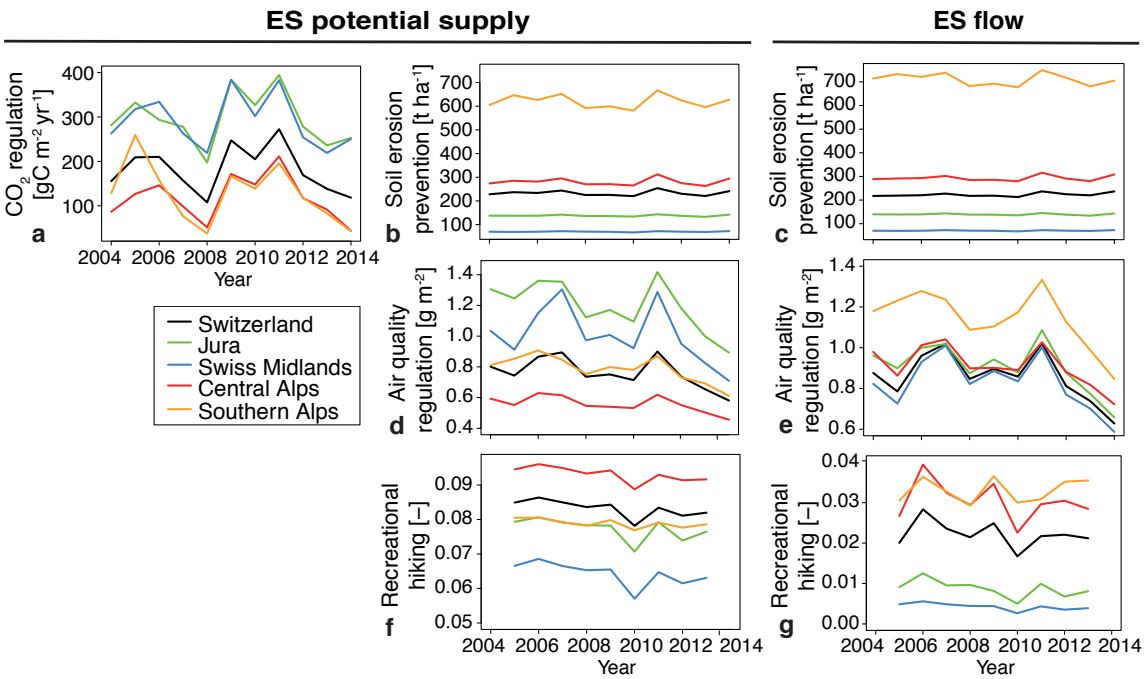

Figure 3.2 Average temporal change of ES potential supply (a, b, d, f) and flow (a, c, e, g) in Switzerland and its regions between 2004 and 2014: (a) carbon dioxide regulation, (b, c) soil erosion prevention, (d, e) air quality regulation, and (f, g) recreational hiking (only 2005 – 2013). ES potential supply and flow are the same for carbon dioxide regulation (a), but is only shown here once.

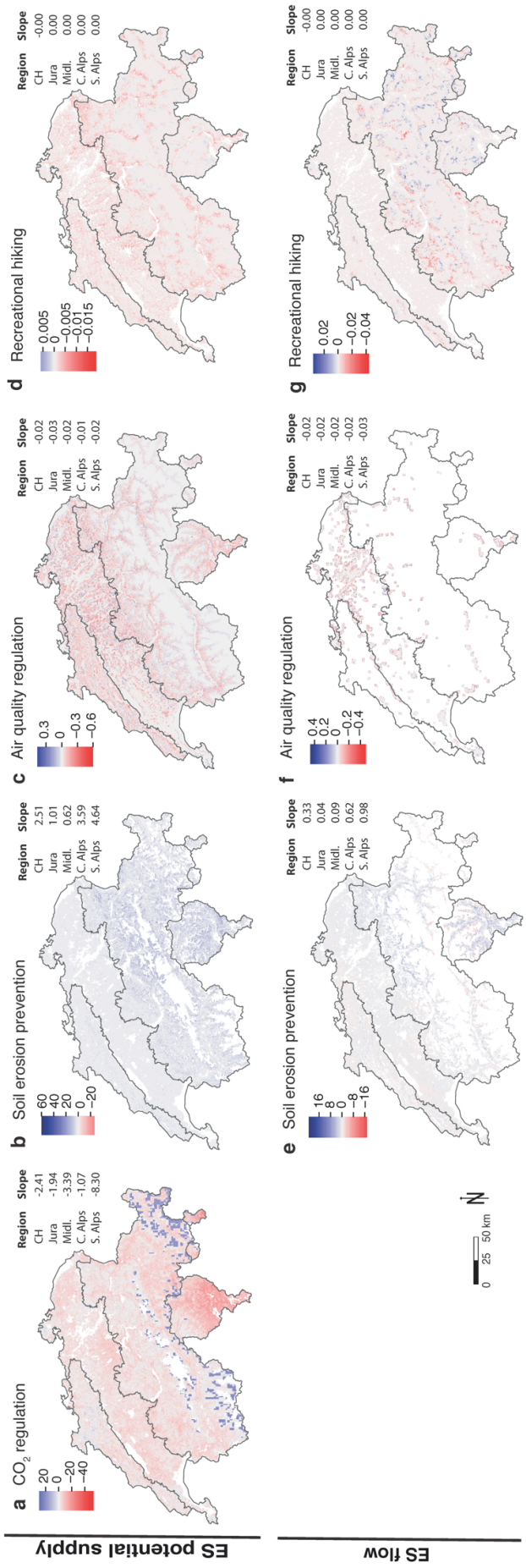

Figure 3.3 Temporal trends in ecosystem service potential supply (a-d) and flow (a, e-g) during 2004 and 2014: (a) carbon dioxide regulation, (b, e) soil erosion prevention, (c, f) air quality regulation, and (d, g) recreational hiking (only 2005 – 2013). ES potential supply and flow are the same for carbon dioxide regulation (a), but is only shown once here.

Central Alps and Southern Alps (Figure 3.3b, e).

Temporal trends of AQR revealed spatial heterogeneity with mainly decreasing trends for AQR potential supply in the Jura, Swiss Midlands and in valleys of the Central and Southern Alps (Figure 3.3c). AQR flow showed stable trends in urban areas with decreasing trends at their borders (Figure 3.3f). RH potential supply remained stable in most parts of Switzerland with some spatially scattered decreasing trends all-over Switzerland (Figure 3.3d). RH flow showed no trends in the Jura and the Swiss Midlands, but revealed some fluctuating spatial pattern of increasing and decreasing trends in the Central and Southern Alps (Figure 3.3g).

Swiss annual total ES potential supply and flow revealed the same relative importance of ecosystem types for all investigated ES (Figure 3.4). In general, forest was the most important ecosystem type to provide ES potential supply and flow, followed by agricultural areas, grasslands and urban areas. The strongest changes between ES

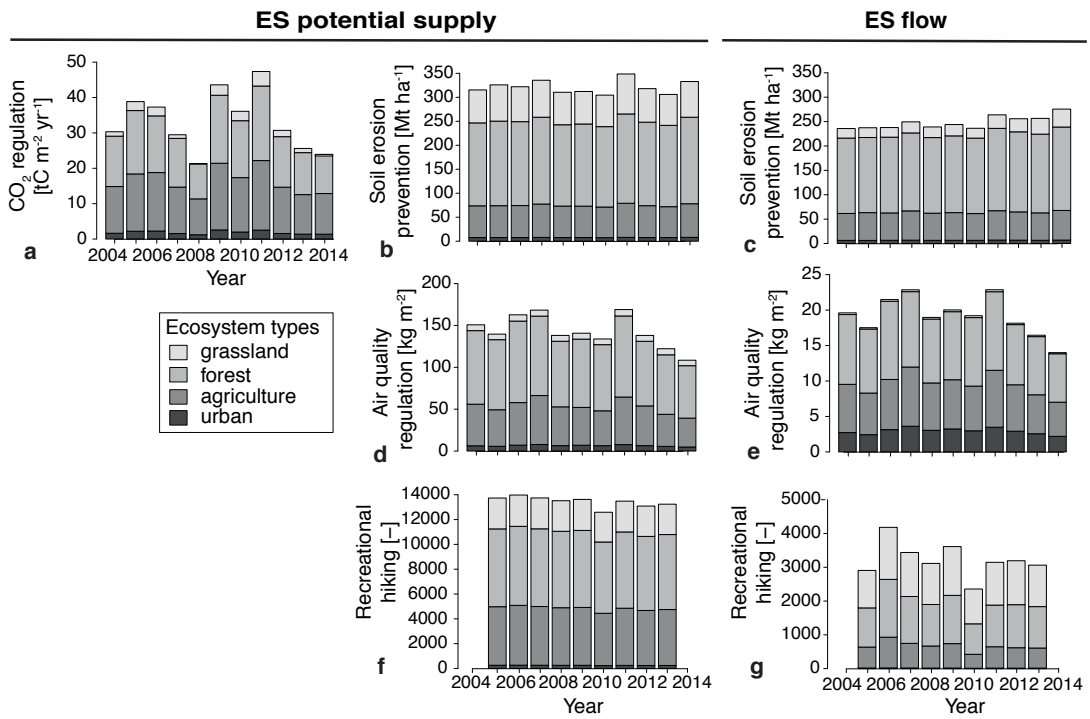

Figure 3.4 Annual total Swiss ES potential supply (a, b, d, f) and flow (a, c, e, g) stratified by ecosystem types between 2004 and 2014: (a) carbon dioxide regulation, (b, c) soil erosion prevention, (d, e) air quality regulation, and (f, g) recreational hiking (only 2005 – 2013). ES potential supply and flow are the same for carbon dioxide regulation (a), but is only shown here once.

potential supply and flow were found for grasslands with reduced contributions for SEP and AQR (Figure 3.4b-c, d-e) and urban areas with increased contributions for AQR (Figure 3.4d-e). The temporal variability in ES potential supply and flow revealed slight changes for all ecosystem types during the investigation period (Figure 3.5). Forest and agriculture were characterized by the strongest variability in ES potential supply and flow. Furthermore, forests displayed relatively strong temporal variability in CO_2R and RH flow and urban vegetation in potential supply and flow of AQR.

3.3.2 Spatial and temporal ecosystem service trade-offs and synergies

Retrieved annual spatial maps of ES were used to estimate at an annual base ES synergies and trade-offs for all pair-wise combinations of ES and their temporal changes during the investigation period (Figure 3.6). In general, identified patterns fluctuated over time and substantially differed across regions. Swiss-wide synergies were identified between ES potential supply of CO_2R-SEP (R_{mean} = 0.15), CO_2R-RH (R_{mean} = 0.05), SEP-AQR (R_{mean} = 0.02), SEP-RH (R_{mean} = 0.17), AQR-RH (R_{mean} = 0.02). Only one trade-off was found for CO_2R-AQR (R_{mean} = -0.70). ES flow showed the same patterns of synergies and trade-offs.

The analysis of ES relationships at national and regional scale revealed two patterns. Firstly, regional and altitudinal differences in synergies and trade-offs of both ES potential supply and flow became evident. Several ES combinations revealed a split in Jura and Swiss Midlands versus Central and Southern Alps and in low and high elevations, respectively (e.g. ES flow of CO_2R-RH and ES potential supply of CO_2R-AQR) (Figure 3.6). Secondly, ES relationships were depending on spatial scale. Pearson correlation coefficients were e.g. positive for ES potential supply of CO_2R and SEP at national scale representing a synergy, but they were negative at regional scale resulting in trade-offs. Similar patterns were identified for ES potential supply of CO_2R-RH and AQR-RH.

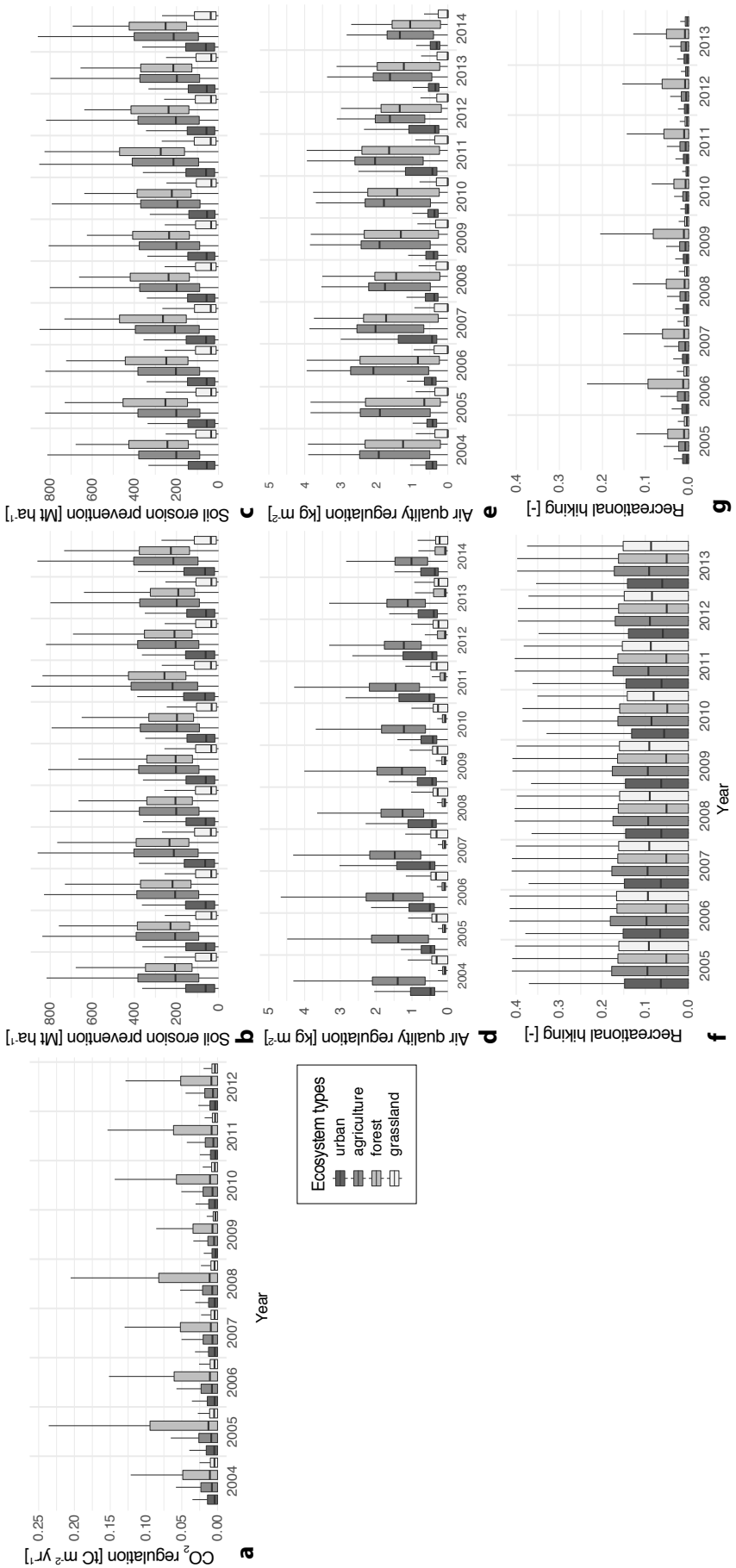

Figure 3.5 Temporal variability in ES potential supply (a, b, c, d, f) and flow (a, c, e, g) stratified by ecosystem types between 2004 and 2004: (a) carbon dioxide regulation, (b, c) soil erosion prevention, (d, e) air quality regulation, and (f, g) recreational hiking (only 2005 – 2013). ES potential supply and flow are the same for carbon dioxide regulation (a), but is only shown here once.

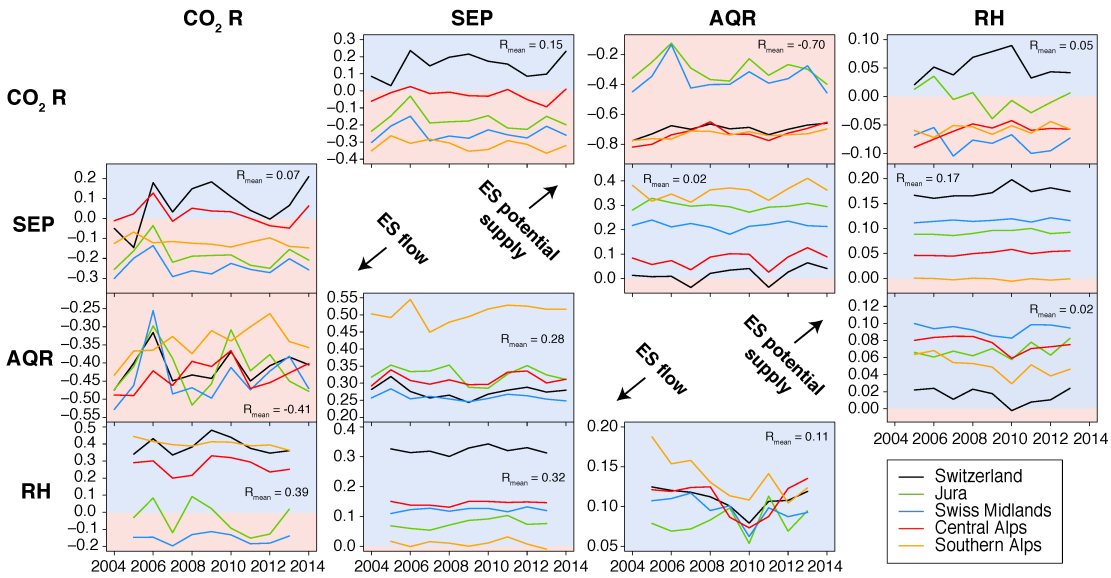

Figure 3.6 Annual Pearson correlation coefficients of ecosystem service potential supply (upper right corner) and flow (lower left corner) combinations between 2004 and 2014. R_{mean} is the Swiss mean correlation coefficient of two ecosystem services during the investigation period. Blue background colour indicates positive correlations, while red represents negative correlations (for interpretation of the references to colour in this figure caption, the reader is referred to the web version of this article).

Pixel-wise correlations of time series of ES pairs revealed a spatially heterogeneous mix of temporal synergies (positive correlation) and trade-offs (negative correlations) (Figure 3.7). Mainly temporal trade-offs were found for CO_2R-AQR potential supply and flow across Switzerland. CO_2R-SEP potential supply partly showed synergies in the Swiss Midlands and Southern Alps, but trade-offs in the Jura and the Central Alps. In contrast, ES flow displayed partly opposite patterns. A spatially heterogeneous mix of synergies and trade-offs characterized CO_2R-RH, SEP-RH, AQR-RH, and SEP-AQR all-over Switzerland for both ES potential supply and flow.

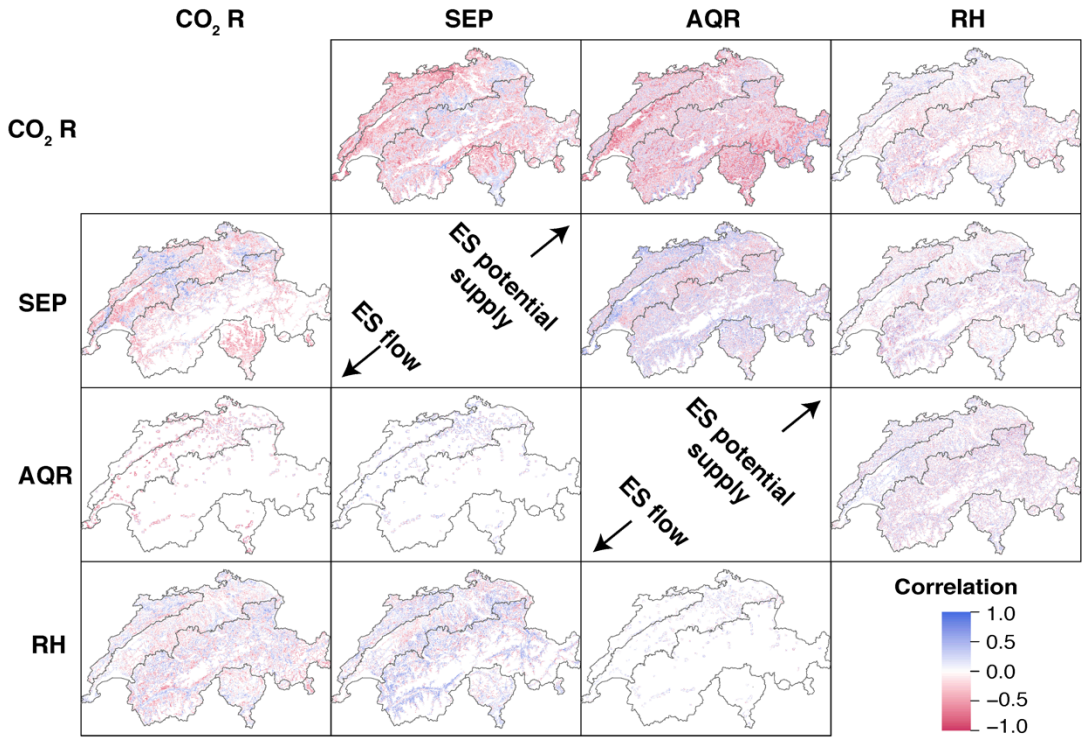

Figure 3.7 Pair-wise spatial correlations of ES time series from 2004 to 2014. Correlations between ES potential supplies are located in the upper right corner, between ES flow in the lower left corner. Correlations > 0 represent synergies and < 0 are trade-offs. Note that correlations with recreational hiking (RH) refer to the time period 2005 to 2013.

3.4 Discussion

3.4.1 Spatial and temporal pattern in potential supply and flow of ecosystem services

We observed a declining trend of CO_2R between 2004 and 2014 across regions in Switzerland. Considering the stimulating effect of increasing atmospheric CO_2 concentration for photosynthesis (Kirschbaum, 2011) and a general increase of biomass in Switzerland (de Jong et al., 2013), this result seems contradicting. We defined CO_2R as equivalent to NEP that was approximated by MODIS NPP and modeled soil respiration. MODIS NPP was lower in 2013 and 2014 compared to the years before, since we used a simple linear model to describe the trend, the two relatively low NPP values likely impact the calculated trend and partly explain the observed decline in CO_2R. The use of liner models to estimate ES trends could

influence the results of all ES, since these models are sensitive to extreme values, in particular when a relatively short time period of 10 years was investigated. Further, increasing temperatures between 2004 and 2014 also contributed to the decreasing trend of CO_2 net uptake, because of its enhancing effect on soil respiration (Bond-Lamberty and Thomson, 2010; Reichstein and Beer, 2008). Inaccuracies in modeled soil respiration could render another explanation eventually causing the decreasing trend of CO_2R: Schröter et al. (2014) applied a similar approach for mapping the annual carbon storage in a study area in Norway. The authors restricted their analysis to forested areas, while arguing that modeled soil respiration highly exceeded actual NPP in other ecosystems, where a neutral carbon balance can be assumed. This seems to indicate a problem to model soil respiration spatially explicit based on the approach of Raich et al. (2002). Some ecosystems, in particular with sparse vegetation such as grasslands in alpine areas, seem particularly sensitive to correct soil respiration estimations. In contrast to such underestimations, the influence of agricultural areas to CO_2R was partly overestimated, since carbon losses due to harvest removal were not considered. Apart of the general decline in CO_2R, we also observed increasing CO_2R trends in the Central Alps. These could originate from land abandonment in Alpine areas followed by expansion of wooded areas (Gellrich et al., 2007). Wooded areas enable higher CO_2 uptake (Bolliger et al., 2008), hence, increasing CO_2R during the investigated period. An alternative to our approach of estimating CO_2R is the measurement of change in carbon stocks, in particular in forests. This can be achieved e.g. by measuring over time the increment in standing biomass in a forest using light detection and ranging (LiDAR) (Babcock et al., 2016).

Swiss-wide SEP remained relatively stable during the investigation period. Only few areas in the Central and Southern Alps showed increasing trends of SEP. Compared to the study of Guerra et al. (2016), who developed and exemplified the SEP model for a Mediterranean test case, our estimates of SEP are lower. This difference can be explained by different climatic conditions, soil characteristics and agricultural practices. Furthermore, Switzerland has had several legal frameworks to protect soils, for example the Soil Pollution Ordinance from 1986 and the voluntary agri-environmental scheme as part of the agricultural legislation since 1993. 98% of Swiss farmers participate in this agri-environmental scheme and receive direct monetary support if ecological standards are fulfilled (Prasuhn et al., 2013). Therefore, the

contribution of agricultural ecosystems to annual total SEP was relatively high with approximately 20 % after forests as main contributor. Since the legal frameworks have already existed for approximately 25 years, ES trends in our investigation period were very small. Nevertheless, obtained results of SEP reveal the huge service ecosystems provide and confirm the success of Swiss legislation to effectively protect soils against soil erosion.

Analysis of AQR potential supply and flow showed a decreasing trend. This, however, does not imply something negative or a decreasing capability of ecosystems for AQR: The applied AQR model highly depends on actual PM_{10} concentrations. Since PM_{10} concentrations decreased during the investigation period, the actual service diminished as well, indicating a positive effect for human well-being. It is important to notice the inverse relationship between the trend in AQR and the effect for human well-being. This negative relationship is different compared to the other ES and could cause confusion when interpreting results or discussing further trajectories of environmental protection. Our average AQR values per year for Switzerland are in the range of other studies (Manes et al., 2016; Remme et al., 2014). The Swiss Midlands displayed the most heterogeneous picture with small-scale changes in AQR trends due to high particulate matter emissions by industry, traffic, heating systems of private households, and agriculture on the one hand and scattered green space in urban areas on the other hand. Urban vegetation had a higher contribution to AQR flow than to potential supply indicating the benefits of urban green space for aerosol removal and contributing to the small-scale spatial variability in ES potential supply and flow.

Trends in RH revealed stagnation in large parts of Switzerland. A decreasing trend of RH was found in some Alpine valleys and along lakes, suggesting decreasing recreational potential in these areas. We argue that this finding is mainly caused by the strategy and underlying data used to calculate RH, which does not lead to a clear indicator yet. Increasing nighttime light emissions, for example, represent less natural- and remoteness of the landscape compared to the years before. A decreasing amount of Flickr data at these locations suggests declining visitation rates, however the direct link of Flickr data to visitation rates is missing and requires additional research in the future. While the night-light data represent a reliable quantitative measure, the use of Flickr data is problematic. When using Flickr data, one has to

account for temporal and spatial biases. A temporally increasing bias is caused by the fact that the website was created in 2004 and has shown a strong increase in users since then. A spatial bias is due to an uneven and clustered distribution of social media data (Li et al., 2013; Wood et al., 2013). Additionally, only a limited amount of hikers take geotagged photographs and share them on Flickr, causing a preference in the used RH model towards a specific group of hikers using Flickr: Social media users are typically younger, better educated, and wealthier than average (Li et al., 2013). However, female hikers in Switzerland represent a nearly equal share between 15 to 74 years, while the proportion of male hikers increases with age (Bundesamt für Strassen (ASTRA) und Schweizer Wanderwege, 2015). Therefore, a large part of older hikers and their preferred hiking areas are likely not represented in the Flickr data, and hence in this analysis. Nevertheless, our results of RH indicated that even though the RH potential supply was the highest in forest and agricultural ecosystems, ES flow took mainly place in grasslands and forests. This represents a preference for alpine and natural landscapes instead of agricultural ecosystems and a realistic picture of hiking preferences in Switzerland (Bundesamt für Strassen (ASTRA) und Schweizer Wanderwege, 2015). In general, the use of social media data for analyzing people's preferences for recreation in combination with RS-based estimates of natural- and remoteness of a landscape offers many opportunities for more detailed investigations, in particular when neglecting the temporal component of Flickr data (Levin et al., 2015). Examples are the identification of nature elements that attract people or whether changes in ecosystems will modify visitation rates. In the future, estimations of recreation based on Flickr data can provide a more comprehensive picture by analyzing not only the photo user days, but as well the semantics attached to the geotagged photos and the content of the photo. This can provide a better inside about the executed activity (e.g. hiking, mountain biking, go for a walk etc.) and the landscape preferences (e.g. view, lake side, waterfall, alpine grassland etc.).

3.4.2 Spatio-temporal pattern in ecosystem service trade-offs and synergies

Identified synergies and trade-offs of different ES potential supply and flow combinations were relatively stable between 2004 and 2014. Except for the CO_2R-AQR combination, representing a trade-off in our analysis, our results are in agreement with findings of a European study by Jopke et al. (2015).

The presence of vegetation explains observed synergies between CO_2R-SEP, CO_2R-RH, and SEP-RH as it determines CO_2R due to CO_2 uptake by photosynthesis, enhances SEP due to mitigated soil erosion by vegetation cover, and is enjoyed while hiking. The trade-off between AQR-CO_2R was caused by a mismatch in areas of high PM_{10} concentrations, typically less covered with vegetation, and densely vegetated areas with potentially high rates of photosynthetic activity and thus CO_2 uptake. For other ES combinations including AQR (i.e. AQR-SEP, and AQR-RH) slight synergies were found but it is difficult to explain them mechanistically.

In general, research on ES trade-offs and synergies has particularly focused on trade-offs between provisioning and regulating ES (Maes et al., 2012; Rodriguez et al., 2006). Trade-offs within regulating or between regulating and cultural ES have not yet been extensively investigated so far. Observed temporal changes of ES trade-offs and synergies were relatively small, indicating robustness of the determined relationships over time. This is important for sustainable decision-making in agricultural management, landscape planning and conservation.

3.4.3 What can remote sensing contribute to spatio-temporal ecosystem service assessments?

RS-based assessments of spatio-temporal trends in ES potential supply and flow provide new insights and facilitate ES research and monitoring. We could demonstrate the added value of including RS data in ES assessments compared to commonly applied ES mapping approaches that use land cover information as key input in combination with expert knowledge (Burkhard et al., 2012; Jacobs et al., 2015) and ES models that assign the same biophysical values to each land cover class (Nelson and Daily, 2010). Such approaches often quantify only the change of an ES providing area (Eigenbrod et al., 2010), while RS allows detailed insights into the variability of ES potential supply and flows across spatial scales and time. This is particularly relevant since ES potential supply and flow are similarly important concerning natural capital accounting within SEEA. Further, the high spatio-temporal granularity of derived ES information allows relating changes in ES to environmental condition and supports the definition of appropriate spatial scales for monitoring purposes of ES, their trade-offs, and synergies.

Suggested RS-based approaches to assess ES avoid the use of less informative proxies or indicators of ES and rather rely on biophysical models to determine ES potential supply and flow. Trend analyses based on multi-temporal ES monitoring directly represent changes in ES, rendering another advantage compared to currently used ES indicators. Shepherd et al. (2016), for example, identified ES indicators for trend analyses at global scale. Only for 62 % of the investigated indicators they could find a strong ability to detect trends in relevant ES. Particularly for some regulating services (e.g. local climate and air quality, erosion prevention and soil fertility) and any cultural service, no suitable ES potential supply indictors could be identified at all. RS- based approaches can partly fill this gap for several ES.

Further, RS-based ES assessments foster a regular monitoring of ES at critical time intervals (i.e. monthly, half-yearly or yearly). Non-RS derived ES indicators or land cover-based ES maps are challenged with a regular monitoring in sufficiently short intervals to capture ES changes and facilitate decision-making with temporal ES assessments. Although a study by Maes et al. (2014) developed more than 300 ES indicators for a European ES mapping attempt, only a small fraction of these ES indicators (e.g. 15% for forests, 27% for agro-ecosystems, 13% for freshwater, 42% for marine systems) can be mapped regularly in short intervals. Further, some of the often used ES indicators rely on land-use data with a rather long update interval, i.e. the five yearly updated CORINE Land Cover product. Further, even though RS technology can provide land cover maps more regularly and accurate, direct estimates of ES potential supply and flow using biophysical models can avoid the use of rather static land cover classifications (Karp et al., 2015).

Detailed RS-based information of ES potential supply and flow, as well their trends, trade-offs, and synergies facilitate research and evidences of ES changes relevant for stakeholders and decision-makers. Regular spatially explicit monitoring of ES using RS data can enable the evaluation whether decisions in politics, economy, and conservation have a positive or negative influence on ES (Rose et al., 2015).

3.5 Conclusion

We conclude on the importance of RS and its versatile advantages for spatio-temporal monitoring of ES compared to indicator based mapping approaches. Particularly the analysis of spatial time series data enables the detection of ES trends and ES interactions in both time and space, eventually contributing to advance decision-making in landscape planning and conservation and to verify progress towards policy targets (e.g. Aichi Biodiversity Targets and Sustainable Development Goals). Spatial ES data can enable regular evaluations of e.g. the efficiency of agri-environmental schemes and of payments for ES.

We suggest exploiting the capability of RS data for ES assessments and implementing them at a regular base. Depending on the spatial and temporal scale as well as the spectral requirements of the ES assessment, a variety of RS sensor is available such as IKONOS, Landsat, Sentinel-2, AVHRR (Advanced Very High Resolution Radiometer), which partly enable either long (but spatially coarse) or dense (but temporally short) ES time series. The limited range of (partly) assessable ES with RS data (except for land cover), however, must be carefully taken into consideration. Nevertheless, the huge potential of RS has not yet been fully exploited, in particular, for regulating and cultural ES. Upcoming Earth observation missions will extend this potential in the future and will allow more holistic and regular monitoring of ES.

Acknowledgements

DB and AD were supported by a grant of the Swiss University Conference and ETH board in frame of the project Swiss Earth Observatory Network (SEON). The contributions of MES and OLP were supported by the University of Zurich Research Priority Program on 'Global Change and Biodiversity'. The authors' sequence is listed following the FLAE approach (doi: 10.1371/journal.pbio.0050018). We thank both anonymous reviewers for their helpful comments.

Appendix 3A.1: Time series of photo user days

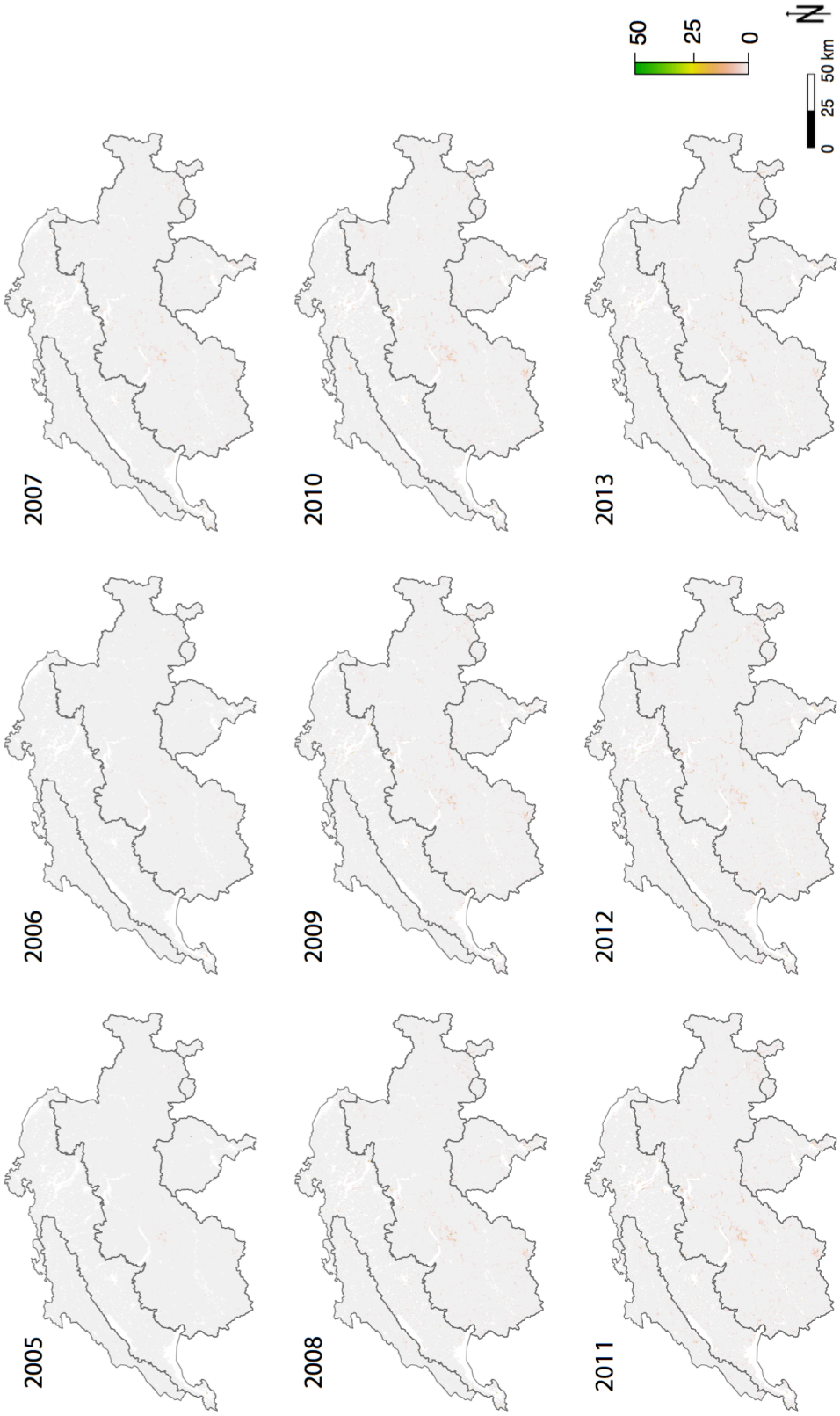

Figure 3A.1.1 Photo user days (PUD) per year from 2005 until 2013.

Appendix 3A.2: Significant ecosystem service trends

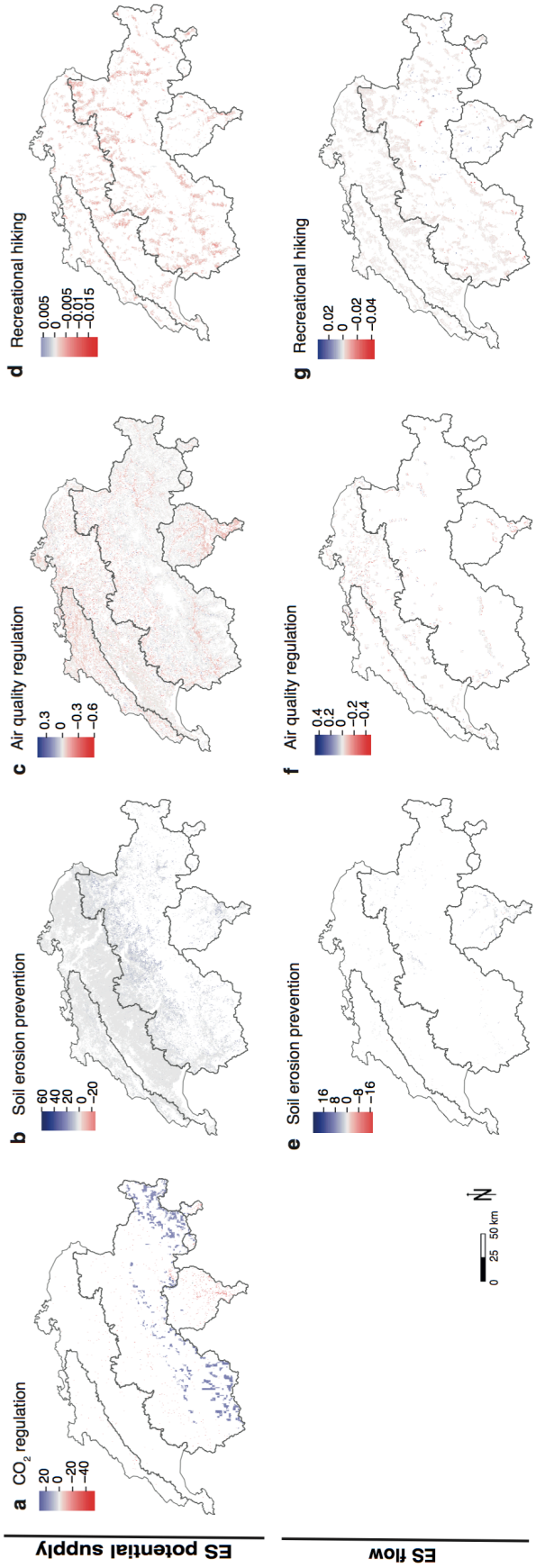

Figure 3A.2.1 Significant temporal trends in ecosystem service potential supply (a-d) and flow (a, e-g) during 2004 and 2014: (a) Carbon dioxide regulation, (b, e) soil erosion prevention, (c, f) air quality regulation, and (d, g) recreational hiking (only 2005 – 2013). ES potential supply and flow are the same for carbon dioxide regulation (a), but is only shown once here

Chapter 4

Drivers of ecosystem service change: The influence of climatological and non-climatological effects on ecosystem services in Switzerland between 2004 and 2014

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This chapter is based on an article submitted to Global Environmental Change and has been modified to list all cited references in the Bibliography chapter.

Abstract

Understanding the drivers of ecosystem change and their effects on ecosystem services (ES) is essential for management decisions and verification of progress towards national and international sustainability policies (e.g. Aichi Biodiversity Targets, Sustainable Development Goals).

We aim to test if and how the effects of climatological and non-climatological drivers on ES supply can be disentangled in a spatially explicit manner. To achieve this, we explored the time series of three ES in Switzerland between 2004 and 2014: carbon dioxide regulation (CO_2R), soil erosion prevention (SEP), and air quality regulation (AQR). We applied additive mixed-effects models to describe the spatial variation attributed to climatologies (i.e. temperature, precipitation and relative sunshine duration) and random effects representing other spatially structured processes that may affect ES change.

The obtained results indicated strong influences of climatologies on ES trends in Switzerland. We identified equal contributions of all three climatologies on trends in CO_2R and SEP, while AQR was stronger influenced by temperature. Additionally, the results showed that climatological and non-climatological drivers impacted ecosystem services both negatively and positively, varying across regions (in particular lower and higher altitudinal areas), drivers, and the service assessed.

Our findings highlight the varying effects of both climatological and non-climatological drivers on ES change and the importance of spatially explicit and regular ES monitoring that allows investigating the effects of climatological and non-climatological drivers on ES change. Such analyses should be extended to ES flow and demand in the future to complete ES assessments and to demonstrate and communicate more clearly the benefits of ES to human well-being.

Authors' contributions (alphabetical order): DB, RdJ, AD, MES designed the study and developed the methodology. DB collected the data. DB, RdJ, RF performed the analysis. All authors wrote the manuscript.

4.1 Introduction

Ecosystem services (ES) have been defined as the benefits that humans obtain from ecosystems (Millennium Ecosystem Assessment, 2005) as well as the contribution ecosystems make to these benefits (European Commission et al., 2013; TEEB, 2010). They are essential for human well-being by providing provisioning (e.g. food, timber, freshwater), regulating (e.g. climate regulation, water purification, soil erosion prevention) and cultural (e.g. recreation, aesthetic) services. Global change has threatened and affected many ecosystems causing severe modifications in ES and, consequently, in the benefits to human well-being (Karp et al., 2015; Mooney et al., 2009; Nelson et al., 2013; Runting et al., 2017). To ensure sustainable development of our planet in the future, it is not only important to monitor changes in ES, but also to identify the drivers of change and the spatial and temporal scales at which they operate (Rounsevell et al., 2010). A driver of change is defined as an ecological or human-induced factor that affects ecosystem processes, functioning and consequently ES directly or indirectly (Millennium Ecosystem Assessment, 2005). Direct drivers of change have an explicit effect on ecosystems, such as land use and land cover change, climate change, pollution, and invasive species. Indirect drivers are more diffuse, such as demographic, economic, sociopolitical, cultural as well as scientific and technological drivers. Both direct and indirect drivers can act at different spatial and temporal scales, which adds complexity to monitoring their impacts on ES.

Among the most investigated drivers of change is climate change and associated effects of changing temperature and precipitation, and sea level rise amongst others on the supply of ES (Runting et al., 2017; Scholes, 2016). Climate change impacts on ES are predominantly negative and vary across services, drivers and assessment methods with stronger negative impacts on regulating and cultural ES compared to provisioning services (Runting et al., 2017). In particular the spatial variability of climatic drivers may cause difficulties when assessing and managing their impacts on ES. The relationship between drivers and ES is often complex, with multiple drivers affecting ecosystem state or condition leading to changes in the capacity of ecosystems to generate ES. Changes in capacity, jointly with changes in ecosystem use (e.g. through adaptation) lead to changes in ES supply (e.g. Hein et al., 2015). Besides direct drivers such as climate change, there are indirect drivers of change caused by socioeconomic pressures such as population growth, demand for

agricultural commodities and their prices that act simultaneously. These can indirectly cause non-climatic drivers, such as land cover and land use change. Land cover and land use change have been one of the most studied non-climatic drivers with often negative impacts on ES (Egarter Vigl et al., 2017; Lawler et al., 2014; Polasky et al., 2011; Runting et al., 2017).

The influence of both climatic and non-climate drivers needs to be investigated simultaneously to understand their complex interactions and their relative and cumulative impacts on ES change. However, in spite of an increasing number of studies assessing the impacts of drivers of change on ES, the focus has mainly been set on the influence of selected individual drivers on ES supply and on predictions of ES under future climate and land use change scenarios (Runting et al., 2017).

Multiple drivers of ES change are often neglected in favor of detailed analysis of one selected driver and its effect on specific or several ES. However, it is known that the influences and interactions of several simultaneous drivers on ES are important and highly relevant to fully understand ES changes (Runting et al., 2017). Disentangling the contributions of all drivers with their individual and cumulative impacts on ES alterations is still an unsolved problem.

Furthermore, most determined causal relationships between ES and drivers of change are often based on future projections of ES change by land use and climate change scenarios (Lawler et al., 2014; Martinez-Harms et al., 2017; Nelson et al., 2010; Rosenzweig et al., 2014), while only few studies determined the effects of drivers of change on actually measured ES trends (Egarter Vigl et al., 2016; Guerra et al., 2016b; Nelson et al., 2013; Schirpke et al., 2013). However, in particular the identification of actual impacts of drivers of change on ES can improve decision-making in landscape planning and conservation, since it allows evaluating the effects and efficiency of past decisions. In general, environmental schemes that contain ES in their agenda require regular monitoring and consequent analysis of drivers of change, e.g. to monitor the efficiency of payments for provided ES and the progress towards political targets regarding ES restoration (e.g. EU Biodiversity Strategy 2020, the Aichi Biodiversity targets and the Sustainable development goals), and to reevaluate those measures if necessary. Therefore, we need ES monitoring approaches combined with investigations of how ES are affected by drivers of change.

The aim of this study is to test if and how the effects of climatological and non-climatological drivers on ES supply can be disentangled in a spatially explicit manner.

We explored time series of three ES in Switzerland between 2004 and 2014: CO_2 regulation, soil erosion prevention, and air quality regulation. An additive mixed-effects model was used to describe the spatial variation attributed to changes in climatologies (i.e. temperature, precipitation, and relative sunshine duration) and random effects representing other spatially structured processes that may affect ES change. The latter were investigated at regional scale to identify their underlying driving factors, e.g. land use change due to political, economic or environmental decision-making.

We realize that 10 years is a relatively short period to examine the effects of climate change on ES, with climate variability relatively strongly affecting trends in climatic factors at such a short time frame. Nevertheless, we believe our approach is useful for the study of the impacts of climatic trends on ES – even though these effects are likely to be more pronounced when studied at longer time frames.

4.2 Methods

4.2.1 Study area

Switzerland is located in central Europe with an area of approximately 4.1 million ha. The country covers altitudes ranging from 196 m to 4634 m above sea level (a.s.l.) in four biogeographical regions: Jura, a low mountain range in the northeast, the Swiss Midlands, which are lowlands crossing Switzerland from the west to the northwest, and Central and Southern Alps, a mountain range covering about 60 % of the country (Figure 4.1).

Land use and land cover are characterized by urban and settled areas covering 7.5 % of the surface area, agriculture 35.8 %, wooded areas 31.3%, surface water 4.3 %, and remaining natural environment 21.0 % (Swiss Federal Statistical Office, 2016). During the last 25 years Swiss landscape has constantly changed due to urban expansion, particularly in the Swiss Midlands (Jaeger and Schwick, 2014), and due to land abandonment in Alpine areas followed by expansion of wooded areas (Gellrich et al., 2007).

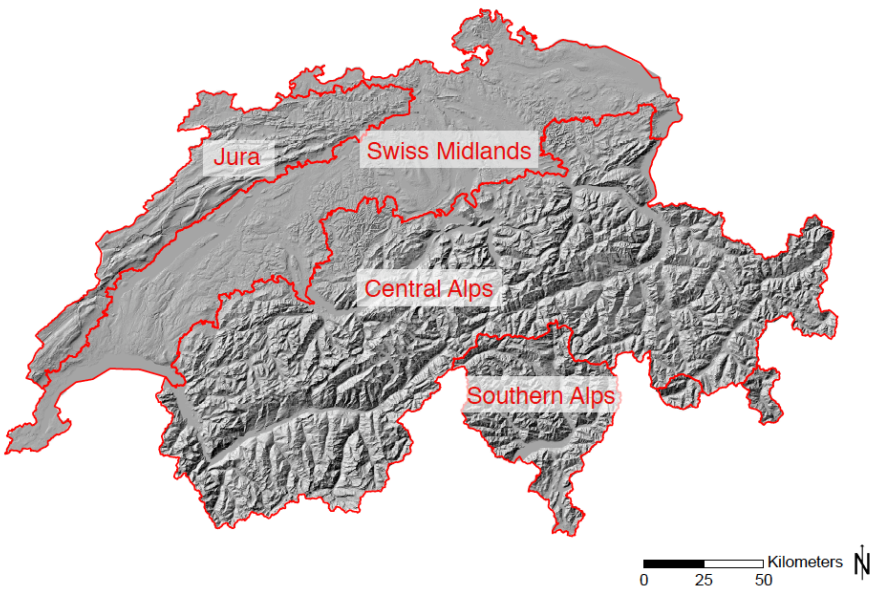

Figure 4.1 The study area Switzerland with its different geographical regions. A digital elevation model is used as background (courtesy of Swisstopo).

4.2.2 Data

4.2.2.1 Ecosystem service time series

We investigated time series of three ES, namely CO_2 regulation (CO_2R), soil erosion prevention (SEP), and air quality regulation (AQR) (Table 4.1) between 2004 and 2014 based on annual data. Details on the ES models and data used can be found in Braun et al. (2017).

CO_2 regulation (CO_2R)

CO_2R focused on the role of vegetation in mitigating climate change, and was defined as net ecosystem production (NEP) without consideration of harvest losses due to a lack of temporally and spatially explicit harvest data. NEP (g C m^{-2} yr^{-1}) was calculated as difference of annual net primary production (NPP) (g C m^{-2} yr^{-1}) and annual soil respiration (R_S) (g C m^{-2} yr^{-1}) as:

$$CO_2R = NEP = NPP - R_S, \text{ with} \quad (4.1)$$

$$R_S = \sum 1.250 \cdot e^{(0.05452 \cdot T_a)} \cdot \left(\frac{P}{4.259+P}\right). \quad (4.2)$$

Annual NPP was derived from the annual MODIS NPP (MOD17A3H) product (Running and Zhao, 2015). R_S was calculated as yearly sum (Σ) according to Raich et

al. (2002) based on 2 km spatial mean monthly air temperature (T_a) (°C) and mean monthly precipitation (P) (mm) data (MeteoSwiss, 2013).

Air quality regulation (AQR)

AQR described the ability of vegetation to filter aerosols from the atmosphere. It was calculated as yearly sum (Σ) integral of daily values of vertical capture of particulate matter of less than 10µm (PM_{10}) by ecosystems (Manes et al., 2016; Nowak, 1994) as:

$$AQR = \sum C \cdot V_d \cdot LAI \cdot T \cdot 0.5, \quad (4.3)$$

where C is the PM_{10} concentration in the air (µg m^{-3}) and was provided by the Swiss Federal Office for Environment (BAFU) (2015), V_d is the dry deposition velocity for PM_{10} (m s^{-1}), a constant reported in Remme et al. (2014), LAI is the leaf area index (m^2 m^{-2}) estimated by the MODIS LAI product (MOD15A2H) (Myneni et al., 2015), T is the time step (s) corresponding to one year, and 0.5 is the suspension rate of deposited PM_{10} returning back to the atmosphere (Zinke, 1967).

Soil erosion prevention (SEP)

We defined SEP according to Guerra et al. (2016) as the difference between the structural impact Υ (e.g. the total soil erosion impact in the absence of soil erosion prevention = potential soil erosion) and the remaining ES mitigated impact β (i.e. the remaining soil erosion that was not regulated by soil erosion prevention) as:

$$SEP = \Upsilon - \beta = \Upsilon\,(1 - C_V), \text{ with} \quad (4.4)$$

$$\Upsilon = R \cdot K \cdot LS, \quad (4.5)$$

where Υ is the structural impact (t ha^{-1}), C_V is the vegetation cover (-), R is the rainfall erosivity (MJ mm ha^{-1} h^{-1}) (Panagos et al., 2015), K is the soil erodibility (t ha h ha^{-1} MJ^{-1} mm^{-1}) (Panagos et al., 2014), and LS describes the effect of topography on soil erosion (-), estimated with the terrain analysis module of the SAGA (System for Automatic Geoscientific Analysis) software (Pilesjö and Hasan, 2014). Vegetation cover C_V was estimated as function of $NDVI$ as suggested by Guerra et al. (2016):

$$C_V = \exp\left[-2 \cdot \frac{NDVI}{(1-NDVI)}\right]. \quad (4.6)$$

The MODIS NDVI product (MOD13A1) (Didan, 2015) was used to calculate the vegetation cover C_V.

Table 4.1 Overview of investigated ecosystem services (ES) and their remote sensing (RS) inputs (cf. Braun et al. (2017) for details on the used methodology).

ES	ES potential supply	Remote sensing input
CO_2 regulation (CO_2R)	Sequestered CO_2 [g C m^{-2} yr^{-1}]	MODIS: Land cover (MCD12Q1), net primary production (MOD17A3H)
Air quality regulation (AQR)	PM_{10} removal [µg m^{-2}]	MODIS: Leaf area index (LAI) (MOD15A2H)
Soil erosion prevention (SEP)	Potential soil erosion [t ha^{-1}]	MODIS: NDVI (MOD13A1) Digital elevation model

4.2.2.2 Climatological data

MeteoSwiss provided annual mean temperature, precipitation, and relative sunshine duration data from 2004 to 2014 with a spatial resolution of 2 km (MeteoSwiss, 2013). The annual temperature (TabsY version 1.4) and precipitation data (RhiresY version 1.0) were obtained from spatial interpolation of in situ measurements of about 80 weather stations and 420 rain gauges, respectively in Switzerland (MeteoSwiss, 2013). Annual relative sunshine duration data (SrelY version 1.2) were based on a combination of in situ heliometer measurements of about 70 stations and high-resolution satellite imageries (MeteoSwiss, 2013). We calculated trends for all three climatological datasets by linear regression (Appendix 4A.1 Figure 4A.1.1).

These three climatologies were selected under the assumption that they may positively or negatively affect ES (Table 4.2). Changes in any of these climatic parameters might cause alterations in ecosystem properties, functioning and, consequently, ES. For instance, SEP is mainly negatively influenced by temperature, precipitation and relative sunshine duration since changes in these climatic factors have a negative effect on vegetation. AQR is highly dependent on weather and, thus, sensitive to climate change (Jacob and Winner, 2009). Increasing precipitation can have a positive effect on the air quality itself, because PM_{10} would be washed out, but

might have a negative effect on the ES AQR, AQR is dependent on the PM_{10} concentration, so less PM_{10} in the atmosphere, means less PM_{10} can be filtered by vegetation and less of the service AQR is provided. Consequently, decreasing precipitation might cause higher PM_{10} concentrations in the atmosphere with negative effects on air quality itself, but resulting in an increase in the ES AQR due to increased filtration of PM_{10}. The longer-term effects of changes in precipitation on vegetation and thereby on PM_{10} capture have not been considered in this study. The influence of causal relationships between climatic factors and CO_2R is not certain, since temperature, precipitation, and relative sunshine duration might have a positive or negative effect on CO_2R. For instance, less organic biomass will be decomposed due to less precipitation resulting in less CO_2 losses due to heterotrophic respiration and an increase in CO_2R. However, vegetation might reduce their photosynthetic activity and, thus, NPP due to limited water availability resulting in a decrease in CO_2R.

Table 4.2 The impact of climatologies on ES (based on Runting et al. (2017) and Jacob and Winner (2009)). The influence of climatologies on ES can be positive (+, blue colour), negative (-, red colour) or mixed (±, green colour), when studies found both positive and negative effects. The colour represents the certainty level of the causal relationships and was estimated from several studies (number of studies are added in brackets below the sign). The darker the colour the higher is the certainty level.

	CO_2 regulation	**Soil erosion prevention**	**Air quality regulation**
Temperature	- ± (16- 30)	- (6-15)	- (1)
Precipitation ↑	± (6-15)	- (6-15)	- (1)
Precipitation ↓	± (16-30)	- ± (6- 15)	+ (1)
Rel. sunshine duration	± (2-5)	- (1)	+ (1)

4.2.3 Determination of climatological and non-climatological effects

We used an additive mixed-effects model to describe the observed temporal changes in ES (*y*, response). The model contained a deterministic part *g(x)*, where *y* depends on a set of covariates *x* with their coefficients (fixed effects), a spatial process *h* representing all non-climatic factors that may influence ES and a residual component ε (de Jong et al., 2013):

$$y = g(x) + h + \varepsilon \quad (4.7)$$

We adapted this model by using a random forest (R package "randomForest", version 4.6 (Liaw and Wiener, 2002)) as deterministic part to model the influence of climatologies on ES change (fixed effects). We refer in the following to the deterministic part, the modeled fixed effects as *FE*. The relationship between ES and climatologies may depend on biogeographical regions because of their distinctly different nature, e.g. mountains versus midlands, which render simple linear models inadequate. For this, we split the data set into training (30 %) and test (70 %) parts and set the number of trees to 1000. The contribution of the individual climatological parameters to the overall modeled fixed effect of each ES was estimated by calculating the variable importance based on mean decrease in accuracy using the R package "caret" (Kuhn, 2015, version 6.071). The direction of the relationship between climatologies and the explained variation in ecosystem services was based on the sign of the respective correlation coefficients.

The spatial process (random effects) was estimated from the remaining spatial variance of observed changes in ES and the modeled fixed effects *y-FE* and may be attributed to non-climatological effects or to interactions that cannot be captured with the meteorological data. A transformation of *y-FE* per ES was necessary to meet the stationarity assumption of the random effects model (transformation CO_2R: log(*y-FE*); SEP: $(y\text{-}FE)^{0.25}$; AQR: $(y\text{-}FE)^{0.2}$). Before the transformation the absolute minimum value per ES was add to account for negative values.

The related spatial variation was modeled by a Gaussian process having zero mean and a parameterized spherical covariance function (de Jong et al., 2013): a range parameter that represents the length of the spatial dependence and a sill parameter that describes the marginal variance. We term the variance of the white noise component as the nugget. The initial sill and nugget were estimated by method-of-moments using *gstat* sample variograms (Pebesma and Wesseling, 1998) and we tested the range between 0.1 and 1.2 degrees (ca. 135 km) using maximum likelihood estimation to

assess the length of the spatial dependence. In order to account for the heterogeneous topography of Switzerland, we set the range to 1.2 degrees (ca. 135 km) for all ES. The residuals were calculated as the difference between the observations and the modeled effects (Eq. 4.7) after reversing the transformation.

The standardized ratio index between climatological and non-climatological effects (RI) indicates which type of driver was strongest in explaining the spatial variance in ES trends and was calculated as:

$$RI = -1 + \frac{2 \cdot |FE|}{|FE| + |h|} = \frac{|FE| - |h|}{|FE| + |h|} \quad (4.8)$$

Positive values of RI represented a stronger influence of climatological effects with 1 being only affected by climatologies, while negative RI values represented non-climatological influences with -1 being completely without climatological influence. RI equals zero indicated the same contributions of climatological and non-climatological effects to ES change. Additionally, the influence of climatological drivers on ES change was estimated by calculating the coefficient of determination (R^2) of the observed ES change and the fixed effects as quotient of the residual sum of squares and the total sum of squares:

$$R^2 = \frac{\sum(y_i - FE_i)^2}{\sum(y_i - \bar{y}_i)^2}, \quad (4.9)$$

where y_i are the values of observed ES change, FE_i are the predicted values of ES change due to climatological effects and $\bar{y}$ is the mean value of the observed ES change.

4.3 Results

4.3.1 Effects of climatologies on ES change (fixed effects *g(x)*)

The modeled fixed effects g(x) described the climatological effects on ES change. The spatial variation in ES change was well captured by the random forest fixed effect models (Figure 4.2b) indicated by R^2 values of 0.63 for SEP followed by 0.54 for AQR and 0.40 for CO_2R (Table 4.3). At regional scale, Jura and Swiss Midlands obtained the highest spatial variance explained by climatologies for all ES, while the ES changes in the Central Alps was less influenced by climatologies. All ES showed negative R^2 values in the Southern Alps and SEP in the Swiss Midlands indicating

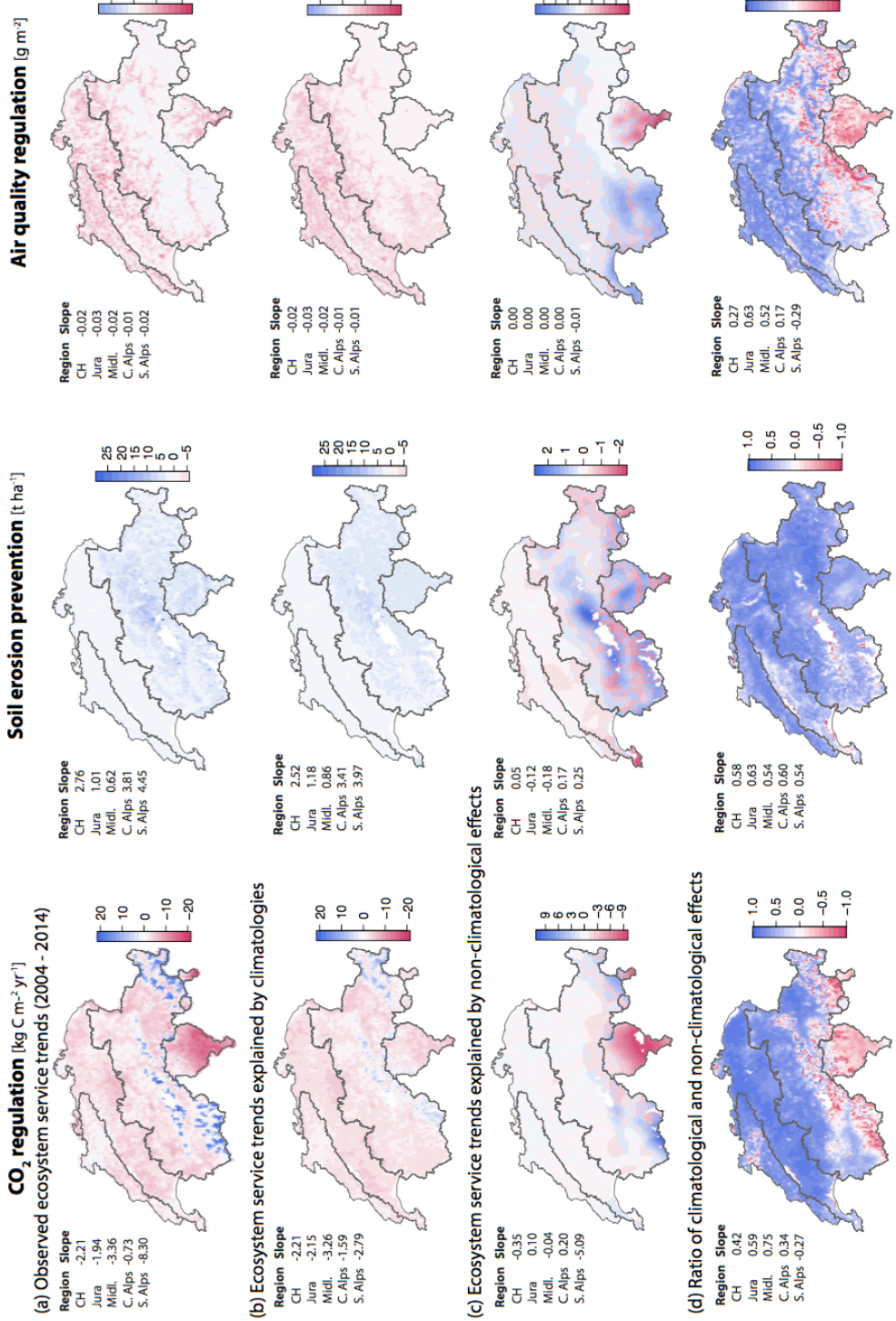

Figure 4.2 Observed ES trends between 2004 and 2014 (a), explained by climatological (b) and non-climatological effects (c). The standardized ratio of climatological and non-climatological effects (d) represents the influence of (b) (positive, blue) and (c) (negative, red) on ES change. The tables contain average trends for Switzerland and its regions. Note that for visualization purposes the scales of (c) differ from (a) and (b).

Table 4.3 Proportion of variance in ES trends explained by climatologies (fixed effects) (R^2) in Switzerland and its regions and variable importance of the individual climatologies (i.e. temperature, precipitation, rel. sunshine duration) to predict the fixed effects in Switzerland.

Ecosystem service		CO_2 regulation	Soil erosion prevention	Air quality regulation
R^2	Jura	0.67	0.62	0.71
	Swiss Midlands	0.68	-0.10	0.71
	Central Alps	0.39	0.48	0.51
	Southern Alps	-0.52	-0.17	-0.11
	Switzerland	0.40	0.63	0.54
Variable importance	Temperature	0.34	0.35	0.52
	Precipitation	0.31	0.31	0.32
	Rel. sunshine duration	0.34	0.34	0.16

that the deterministic model was not able to predict climatological effects in the these regions for these ES. The individual climatological parameters contributed diversely to the fixed effects of the different ES in Switzerland (Table 4.3). All three climatologies were nearly equally important for CO_2R and SEP. AQR was most affected by temperature that revealed the highest variable importance with 0.52. Furthermore, precipitation was approximately twice as important as relative sunshine duration for AQR.

The deterministic relationship between individual climatological parameters and fixed effects revealed several negative, neutral, and positive directions (Table 4.4). For instance, all climatological parameters indicated positive effects on SEP. In contrast, AQR revealed mainly negative effects of climatologies except for a neutral interaction of AQR with temperature. Furthermore, CO_2R was negatively influenced by precipitation, while temperature and relative sunshine duration revealed neutral effects.

Table 4.4 Direction of the relationship between climatologies and the explained variation in ecosystem services based on the sign of the respective correlation coefficients. Red colours represent positive and blue colours negative correlations. The darker the colour the stronger is the correlation.

Climatology	CO_2 regulation	Soil erosion prevention	Air quality regulation
Temperature	0.07	0.30	-0.09
Precipitation	-0.14	0.49	-0.28
Rel. sunshine duration	0.04	0.42	-0.22

4.3.2 Non-climatological effects on ES change (spatial process *h*)

The spatial term of the additive mixed-effects model described the spatial variation in ES trends that could not be attributed to climatologies (Figure 4.2c). These effects were smaller for all ES compared to the climatological effects. For all ES, the spatial variation of low negative and positive ES trends in the Jura and Swiss Midlands was attributed to the non-climatological component. Also, stronger negative and positive ES trends in the Central and Southern Alps were captured by this non-climatological component. This also holds true for decreasing trends in the Southern Alps in CO_2R and AQR, and increasing trends in SEP in the southwestern Central Alps.

4.3.3 Ratio of climatological and non-climatological effects

The standardized ratio of climatological and non-climatological effects indicated, the relative importance of both types of divers to explain spatial variations of ES change. The results showed that ES trends were mainly attributed to changes in climatologies in Switzerland (positive values in Figure 4.2d). In particular, trends in SEP were strongly affected by climatologies with an RI value of 0.58 for Switzerland followed by CO_2R (RI 0.42) and AQR (RI 0.27). Regional differences revealed that climatologies explained most of the ES trends in the Jura and Swiss Midlands for all ES and the changes in SEP in the Central and Southern Alps. Non-climatological effects dominated only locally changes in CO_2R and AQR in the Central and Southern Alps. However, these areas revealed also small-scale heterogeneity of climatological

and non-climatological equality. AQR showed, in particular a larger area with RI values around zero in the southwest of the Central Alps.

4.4 Discussion

4.4.1 Climatological causes of ES change

In our investigations, trends in all ES over the examined time period were mainly attributed to climatologies in Switzerland. In general, there has been relatively little land use change in Switzerland over the same period, which would have otherwise affected the relative importance of climatic variables (Swiss Federal Statistical Office, 2016). The explained climatological effects varied across ES and region with mainly negative, but also positive influences on ES. Swiss wide climatological impacts explained well the negative ES trends in CO_2R and AQR, which are in line with other studies (Runting et al., 2017). Temperature was both positively and negatively correlated to CO_2R in Switzerland with some spatially distributed positive effects on increasing CO_2R trends in parts of the Central Alps. This positive temperature effect on CO_2R in the Swiss Alps can be explained by forest expansion into higher altitudes (Gellrich et al., 2007). The widely distributed negative CO_2R trends across Switzerland could be equally attributed to changes in the three climatologies investigated due to their similar variable importance. This indicates that temperature might have both positive and negative effects on CO_2R trends, although spatially separated: positive in the Central Alps, where increasing temperature can have a strong effect on vegetation growth in higher altitudes, and negative in other parts of Switzerland, where increasing temperatures might be accompanied with a deficiency in water availability. Lu et al. (2013) detected such varying effects of temperature on carbon sequestration for different ecosystem types such as forest, grassland, shrubland, and tundra. This explanation can hold true for our analysis too, since Switzerland is characterized by small scale heterogeneity in land use and land cover with different ecosystem types, exposition and underlying environmental conditions.

Trends in SEP were, on average, positively and equally affected by climatologies in Switzerland. This contradicts current findings of ES studies, where individual climatologies (i.e. temperature, precipitation and rel. sunshine duration) mainly revealed negative impacts on SEP and soil fertility (Runting et al., 2017). However, it

seems that increasing temperatures (see Appendix 4A.1 Figure 4 A.1.1a) in Switzerland between 2004 and 2014 fostered vegetation growth and consequently an increase in SEP, even though precipitation had increased in the Southern Alps and parts of the Central Alps (Appendix 4A.1 Figure 4A.1.1b).

Furthermore, we attributed 54 % of the trends in AQR to climatologies with temperature contributing strongest to the change. In particular, AQR trends in the Jura and Swiss Midlands were strongly dominated by climatological effects, while climatological and non-climatological effects influenced the Central Alps equally. These climatological effects can be partly explained by decreasing precipitation in the Jura and the Swiss Midlands (Appendix 4A.1 Figure 4A.1.1b). The negative relationships of precipitation was confirmed by other studies that investigated the impacts of climate change attributes on PM driven air quality (Jacob and Winner, 2009). Temperature revealed a neutral effect and relative sunshine duration a negative effect on AQR changes, which contradicts Jacob and Winner (2009). However, these relationships are based on few studies and in particular the effect of temperature is uncertain, since temperature estimations are often too inaccurate.

The heterogeneous spatial patterns in ES trends were well captured in the fixed effect models, despite few local over- and underestimations. We acknowledge that our analysis regressed both climate change and inter-annual climate variability with ES supply. Our models convincingly showed that climate variability affects the supply of ES from on year to the next, and that there is major spatial variability in this process. Our research period of 11 years was relatively short compared to what is needed to capture the complex responses of ecosystems to climate change, and subsequent effects on ES. Hence, further research, over a longer time frame, is needed to confirm, how changes in ecosystems due to climate change affect the capacity of ecosystems to generate ES.

4.4.2 Non-climatological causes of ES change

Non-climatological effects explained ES trends in Switzerland to a smaller magnitude compared to climatological effects. The explained ES trends were partly spatially restricted to areas in the Central Alps and partly dominating over climatological effects, in particular for CO_2R and AQR. Such non-climatological influences on ES trends may be attributed to small-scale land cover and land use changes, which are

approximately 1 % per year in Switzerland and thus negligibly small (Swiss Federal Statistical Office, 2016), decreasing PM_{10} concentrations in the case of AQR and a decline in natural- and remoteness of landscapes. However, non-climatological influences might also be the effect of artifacts in the spatial component (h) of the model used compensating for possible overestimation of the fixed effect term (*g(x)*). The fixed effect component obtained higher trends in these areas compared to observed ES trends and represented an overestimation.

Decreasing PM_{10} concentrations were caused by less PM_{10} emissions in Switzerland between 2004 and 2014, in particular in the southern part of Switzerland (Appendix 4A.1 Figure 4A.1.2). Increasing trends of AQR in the southwest of the Swiss Midland and the Central Alps seemed to be caused by non-climatological effects. Since the trends of the fixed effect component exceeded in these areas the observed AQR trends, this represents an artifact caused by an overestimation of the fixed effect model.

Decreasing CO_2R was also caused by slightly dominating non-climatological effects in the Southern Alps. It is unclear, which non-climatological drivers caused these negative trends; potential explanations are land use changes, an underestimation of the fixed effects resulting in artifacts in the non-climatological explained variance, and further processes that are unknown.

Increasing CO_2R trends explained by non-climatological effects might be the result of an underestimation of the fixed effect model. Additionally, interactions between climatological and non-climatological effects, for instance, of climate and land use change, might as well be a reason for these ES trends.

Non-climatological effects impacted also SEP trends in the Central and Southern Alps explaining in particular spatial patches with relatively high SEP trends. Potential explanations for these trends could be land cover changes, e.g. due to afforestation and changes in management decisions of ecosystems that might have caused an increase in vegetation cover.

4.4.3 Reliability of the analysis

The used spatially explicit RS-based ES time series enabled detecting the influence of climatological and non-climatological drivers. The analysis was based on trends in ES and climatologies including climate variability, determined by linear regressions.

Obtained results might be affected by anomalous years, since the investigated time series were relatively short with only eleven years.

The fixed effect models estimating the climatological driven ES trends were based on random forest regressions. These accounted for non-linear effects in the causal relationships of ES and climatologies. Interpretation of these results at regional scale indicated that the FE of the Southern Alps could not be estimated correctly using random forest. Small-scale heterogeneity in climatological trends all-over Switzerland caused prediction errors, since these trends resulted in diverse ES trends that could not be predicted by the deterministic model. Training and predicting FE per region instead of at national scale could solve this problem (cf. Appendix 4A.2). Therefore, as always, interpreting the explained climatological and non-climatological effects should be done carefully.

In the fixed-effect model, we limited ourselves to climatologies but this component can be extended with other drivers. Land use changes is one of the most included non-climatological drivers in ES studies (Egarter Vigl et al., 2016; Lawler et al., 2014; Nelson et al., 2010; Runting et al., 2017). This might be due to the well-established importance of this driver, the availability of land use change models, the mainly negative impacts of land use change on ES (Foley et al., 2005; Runting et al., 2017). Furthermore, many ES mapping approaches are built on land cover data (Lavorel et al., 2017), which fostered research on future ES changes based on land cover scenarios (Lawler et al., 2014; Martinez-Harms et al., 2017; Nelson et al., 2010; Nelson and Daily, 2010). However, other drivers such as pollution might be neglected due to a too strong focus on land use change. So disentangling the impact of climatological from non-climatological effects without specific non-climatological drivers revealed their cumulative effect and not just the effect of land use change.

Analyzing climatological and non-climatological drivers and their effects is critical for understanding the complexities of the impacts on ecosystem services (Bryan, 2013; Carpenter et al., 2009). Our analysis contributes to this challenge by quantifying cumulative and individual climatological contributions to ES change, as well as the spatial structure of other drivers. Only few studies (Runting et al., 2017; Schirpke et al., 2017) combined climatological and non-climatological drivers in their analysis, while most focused exclusively on climate change (Nelson et al., 2013; Scholes, 2016) or land use change (Guerra et al., 2016b; Martinez-Harms et al., 2017; Nelson et al., 2010; Schirpke et al., 2013; Stürck et al., 2015). Therefore, this analysis

represents an enhancement and additional first approach in tackling this challenge of investigating both monitored climatological and non-climatological effects on ES. Furthermore, investigation of cumulative and individual impacts of climatologies on ES has been limited too (Runting et al., 2017), although it provides further insights and information that can improve management decision in landscape planning and conservation. Additionally, this information is also crucial for evaluating the progress towards policy targets such as the Aichi Biodiversity and the Sustainability Goals and to provide further guidelines for implementing the ES concept efficiently into practice (Cord et al., 2017).

4.5 Conclusions and outlook

We conclude that the proposed approach building upon an additive model offers new pathways to disentangle climatological and non-climatological effects on ES trends. Particularly, the application of a random forest model as FE could compensate limitations of linear models by accounting for spatially heterogeneous ES trends within Switzerland. However, the random forest model is scale dependent and needs to be adapted for subscale analysis.

Climatological and non-climatologial drivers affected ecosystem services both negatively and positively, varying across regions, drivers, and the service assessed. ES trends were mainly affected by climatological influences in the lower altitudinal areas (Jura and Swiss Midlands), while higher altitudes in the Central Alps revealed stronger impacts of non-climatological effects. This highlights the importance of conducting regional ecosystem service monitoring and a consequent analysis of the effects of drivers of change.

Future research should focus in impact assessments of drivers of change not only on the biophysical supply of ES, but should extend such investigations to ES flow and demand. Additionally, research investigating both climatological and non-climatological effects on ES should be fostered and improved to identify affected ES and their spatial location and extent.

Acknowledgments

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Appendix 4A.1: Trends in climatologies and particulate matter 10 (PM_{10})

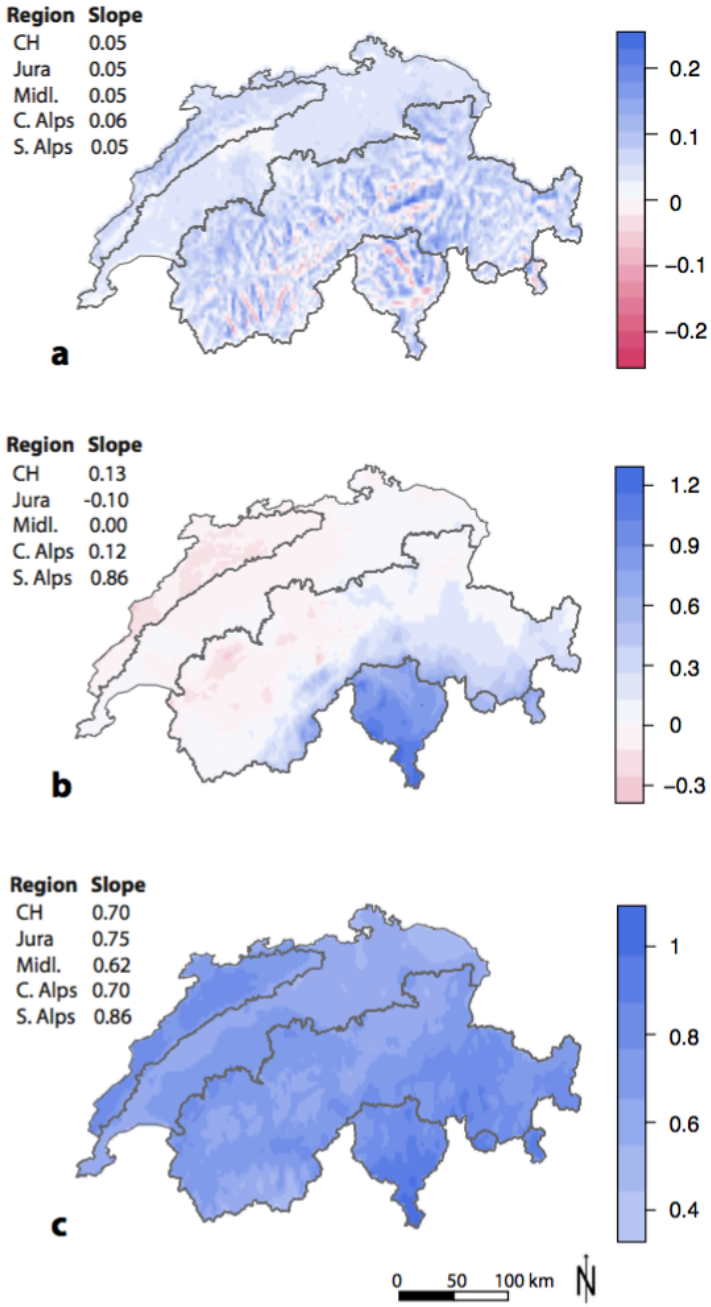

Figure 4A.1.1 Change in climatologies between 2004 and 2014: temperature (a), precipitation (b), relative sunshine duration (c) with average trends for Switzerland and its regions.

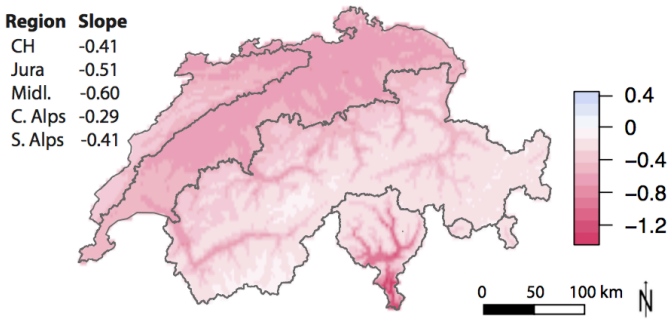

Figure 4A.1.2 Change in particulate matter 10 (PM_{10}) (µg m^{-3}) in Switzerland and its regions between 2004 and 2014.

Appendix 4A.2: Region specific deterministic models

Table 4A.2.1 Proportion of variance in ES trends explained by climatologies (fixed effects) (R^2) based on region specific deterministic models.

Ecosystem service		CO_2 regulation	Soil erosion prevention	Air quality regulation
R^2	Jura	0.18	0.41	0.32
	Swiss Midlands	0.51	0.49	0.46
	Central Alps	0.30	0.48	0.52
	Southern Alps	0.76	0.41	0.40

Chapter 5

Synthesis

5.1 Main findings

The research field of ES mapping has fast developed in the past two decades by providing a variety of different methods characterized by diverse data requirements, various spatial scales and different usefulness for spatial and temporal monitoring of ES. The importance of spatially explicit ES mapping and in particular of ES monitoring is widely recognized and a key challenge in ES research (Cord et al., 2017; Karp et al., 2015; Tallis et al., 2012).

Earth observation data play a crucial role as a tool to implement spatially explicit ES mapping and monitoring approaches. This thesis (i) provides an observation-based framework to map ES following the ES cascade, (ii) maps and monitors ES and its interactions at landscape and national scale using RS data and (iii) identifies the contributions of climatological and non-climatological drivers to the ES changes detected.

The previous Chapters addressed the three Research Questions (RQs) formulated in Chapter 1. In the following Chapter these RQs are discussed together with the four Hypotheses.

5.1.1 How can remote sensing be used as a tool to map ecosystem services and what is the spatial variability of ecosystem services?

Remote sensing enables the spatially explicit estimation of ecosystem properties, functioning and ES. Chapter 2 introduces an approach to map the components of the biophysical part of the ES cascade using Earth observations. In contrast to the theoretical framework of the ES cascade (Burkhard et al., 2014; Haines-Young and Potschin, 2010), this thesis further developed an observation-based ES cascade that presents guidelines how to use RS as valuable tool and basis to map ES spatially explicit. It suggests assessing ecosystem properties and functioning directly with RS data. Afterwards, these derived variables should be coupled with mechanistic ES models as well as with in situ and literature data to acquire spatially explicit and continuous ES maps.

Many studies have already used mechanistic models to assess and map ES, however these models were often only based on land cover maps providing discrete instead of

spatially explicit and continuous ES estimations (Jacobs et al., 2015; Schulp et al., 2014). Using Earth observation data for ES mapping allows the spatially explicit retrieval of ecosystem properties from spectral information and the consecutive estimation of ecosystem functioning and ES with modeling approaches. This is exemplified at landscape scale in Chapter 2 and at national scale in Chapter 3.

In Chapter 2, the spatial variability in ES was detected within individual and among different ecosystems at landscape scale by mapping the components of the proposed observation-based ES cascade. ES variability is often undetected and neglected by land cover based ES mapping approaches (Eigenbrod et al., 2010), but is a crucial information for environmental assessments and conservation measures (Hauck et al., 2013; Maes et al., 2012a) as well as for ecosystem accounting (Hein et al., 2015). In this thesis, higher spatial variability in ES supply was identified in more natural ecosystems such as forests and grasslands compared to highly managed systems like agricultural areas (Section 2.5.3). Even within ecosystems, spatial variability in ES was detected. For instance, parts of a forest sequestered more CO_2 per year than others.

At national scale, spatial variability in ES was characterized by regional differences that were influenced by altitude (Chapter 3), climatologies and non-climatological factors (Chapter 4). CO_2 regulation, for example, decreased with elevation, while recreational hiking revealed an increase. The findings of Chapter 2 and 3 confirm Hypothesis I and provide interesting insights into the spatial variability of ES, which are relevant for management decisions in landscape planning and conservation.

5.1.2 What are the spatial and temporal ecosystem service trends and how do the ecosystem services interact (e.g. trade-off and synergy) in Switzerland over time?

ES change can occur at intra- and inter-annual time scales. Within one year ES change is mostly driven by seasonal changes (Burkhard et al., 2014), for instance by vegetation phenology, agricultural practices and weather conditions that might influence cultural services such as recreational hiking. Individual RS imageries represent therefore a snapshot in time, which might not be representative for annual

ES estimations. The two airborne RS data sets used in Chapter 2 represent such snapshots within one year and make it necessary to account for the intra-annual changes in CO_2 regulation and food supply, since the policy relevant time scale for decision-making is at least one year. The temporal scaling approach developed, allowed the retrieval of annual ES estimates and made the results comparable to other studies.

Inter-annual ES change can be caused by multiple drivers, such as climate and land use change, which make ES monitoring over time necessary. Chapter 3 investigates inter-annual ES changes and trends over 11 years in Switzerland for four ES, namely CO_2 regulation, soil erosion prevention, air quality regulation and recreational hiking. While CO_2 regulation and air quality regulation decreased in Switzerland between 2004 and 2014, soil erosion prevention increased and recreational hiking remained stable. These findings confirm Hypothesis II except for soil erosion prevention for which the Hypothesis has to be rejected. Additionally, regional differences in ES change became evident. CO_2 regulation revealed an increasing trend in the western and eastern Central Alps due to expansion of wooded areas caused by agricultural land abandonment in higher altitudes (Gellrich et al., 2007) and a decreasing trend in the Southern Alps due to changes in climatologies and non- climatological effects (see Section 4.3.1). The Swiss Midlands were mainly characterized by declines, e.g. in air quality regulation, CO_2 regulation and recreational hiking that can be explained by decreasing PM_{10} concentrations, changes in crop rotations with longer time periods of bare soil fields, and an increase in nighttime stable lights, as proxy for remoteness and naturalness of landscapes.

Most of the existing ES monitoring approaches rely on models based on land cover maps and census data, and are therefore restricted to 5-year or even longer monitoring cycles (Karp et al., 2015; Schirpke et al., 2013; Stürck et al., 2015). The suggested RS based ES monitoring approach, however, allowed annual ES estimates, which provided detailed information about inter-annual changes in ES. Additionally, the estimated ES time series enabled the detection of ES interactions, which can provide valuable information for improved management decisions, land use planning and conservation. ES interactions revealed synergies for all ES pairs at national scale except for CO_2 regulation and air quality regulation, which were a trade-off. This is in agreement with ES interactions determined in other studies (Jopke et al., 2015; Lee

and Lautenbach, 2016; Maes et al., 2012b). All ES interactions fluctuated over time, but the type of interaction, i.e. trade-off and synergy, remained the same. These findings confirm Hypothesis III. In this thesis, it was observed that ES interactions changed from national to regional scale and that they differed across regions indicating altitudinal dependencies (Section 3.3.2). These results confirm the dependency of ES interactions on spatial scales, which was already found in other studies (Lee and Lautenbach, 2016; Raudsepp-Hearne and Peterson, 2016; Willemen et al., 2012). Additionally, these results support the use of RS-based approaches for ES monitoring, since they enable spatial scaling of the derived ES information to match the scale of decision-making. While the majority of studies has focused on interactions between provisioning and regulating services and has neglected the spatial scale dependency of ES interactions (Howe et al., 2014), this thesis focused on regulating and cultural services and their interactions at different spatial scales.

5.1.3 What are the effects of climatological and non-climatological drivers of change on ecosystem service trends in Switzerland?

The spatially explicit ES time series derived from RS data in Chapter 3 were used to identify the influences of climatological and non-climatological drivers on ES change with additive mixed-effect models. Obtained results revealed strong effects of climatologies on ES trends in Switzerland (i.e. above 50 %) except for recreational hiking, which showed the weakest influences with less than 10 %. Spatially, climatologies caused most ES change in the Jura and Swiss Midlands. In contrast, SEP was dominated by climatological effects throughout Switzerland. The climatological effects on CO_2 R and SEP were equally driven by temperature, precipitation and relative sunshine duration, while AQR and RH were stronger influenced by temperature.

The ES trends in Central and Southern Alps were more affected by non-climatological effects, in particular CO_2 R and AQR. Additionally, the analysis revealed that climatological and non-climatologial drivers impacted ES both negatively and positively, varying across regions (in particular between lower and higher altitudinal areas), drivers, and the service assessed. These findings can be

confirmed by other studies that identified varying effects of drivers on ES change (Runting et al., 2017). The results approve Hypothesis IV partly, since it has to be rejected for ES trends of CO_2 R and AQR in the Southern Alps, which were mainly caused by non-climatological effects.

While many ES studies focused on selected drivers of change and their impacts on ES (e.g. climate change (Nelson et al., 2013; Runting et al., 2017; Scholes, 2016) and land use change (Guerra et al., 2016; Martinez-Harms et al., 2017; Nelson et al., 2010; Polasky et al., 2011; Schirpke et al., 2013)), this thesis investigates both the individual and cumulative effects of climatological and non-climatological effects on ES. This information is crucial in understanding the complexity of impacts on ES (Bryan, 2013; Carpenter et al., 2009). Additionally, it can help combating the drivers of ES change by improving both the implementation of the ES concept in practice and the decision-making in landscape planning and conservation.

5.2 General contributions

Due to the current need for evaluating the progress of international and national regulations (e.g. Aichi Biodiversity Targets and Sustainable Development Goals) and conservation actions, this work investigates three important steps towards systematic ES monitoring: (i) spatially explicit mapping of ES, (ii) spatial and temporal monitoring of ES including ES interactions and (iii) investigation of ES change causes over time. All three aspects demonstrate that the use of RS is valuable to conduct ES mapping and monitoring.

The main contributions of this thesis are three-fold. First, the observation-based ES cascade building upon RS data represents progress towards spatially explicit ES mapping. This is a step forward towards standardization of detailed ES assessments that can be used across spatial and temporal scales. It is crucial for evaluating progress of policy regulations and a future operationalization of ES in practice. In this thesis, the consequent mapping of all components of the biophysical part of the ES cascade suggests and exemplifies possible classes for essential ES variables (eESVs) as ES counterpart to essential biodiversity variables (EBVs, Pereira et al., 2013). While many studies of ES mapping focused only on ES supply and demand, this thesis includes ecosystem properties and functioning into the analysis, since changes caused by global change will affect and become apparent in them first.

Second, this thesis moves from pure ES mapping to monitoring by providing a case study of ES monitoring at national scale over more than a decade. The focus of this work is not restricted to changes in a single ES service, but to changes in multiple services and how they interact over time. The approach proposed allows annual ES monitoring with RS data and derived consistent time series of four ES over 11 years. The ES information acquired can be used to measure progress towards national and international policy regulations as well as to evaluate the usefulness of conservation actions (e.g. agri-environmental payments).

Third, these ES time series allowed identifying the influence of climatological and non-climatological drivers on ES change. Key findings were strong effects of climatologies on ES in Switzerland. However, the impacts varied across regions, drivers, and the services assessed, indicating the importance of spatially explicit and regular ES monitoring. This confirms findings from other studies that identified

varying effects of drivers of change on ES change (Runting et al., 2017). Only systematic ES monitoring approaches allow investigating the effects of both climatological and non-climatological drivers of change on ES change. This information is crucial for the implementation of the ES concept in practice and improved decision-making in landscape planning and conservation, since it can improve combating drivers of ES change.

5.3 Final considerations and future directions

Besides the findings of this thesis and its main contributions, there remain open issues to discuss and next research directions to focus on.

5.3.1 Open issues

First, the ES mapping and monitoring studies revealed different scales of ES dynamics. The investigations covered both very fine spatial (2 m) and spectral resolutions by using airborne imageries for a landscape study and quite broad spatial resolutions of 500 and 2000 m using mainly MODIS satellite data products at national scale. Depending on the ES, the spectral requirements were different and the spatial variability altered from small-scale changes in forests to changes among agricultural fields and changes among regions. Based on these finding and the policy relevant scales of decision-making, ES mapping and monitoring approaches should select sensors depending on the spatial and temporal process length and the spectral requirements of the ES investigated. For instance, multispectral sensors with intermediate spatial resolution such as Landsat (30 m) and Sentinel-2 (10-60 m) could be used to map and monitor ES with low spectral requirements at different spatial and temporal scales. Such analyses would be useful to investigate changes at regional, national and even continental scale in more detail. Additionally, larger spatial coverage would allow comparisons of ES changes among countries and for instance, the efficiency of their different conservation measures. However, while Landsat has a longer temporal extent, since it goes back to the 1980s, its temporal coverage is limited to few cloud-free images per year, which impedes annual ES estimations. In contrast, the temporal coverage of Sentinel-2 is comparable to MODIS with a 5-day revisit time, but its temporal extent is limited, since it was only launched in 2015. Additionally, Sentinel2 possesses less spectral bands compared to MODIS. In general, a higher spatial resolution automatically increases the information content and the variability in the estimated surface properties substantially. However, this increased information content due to a high spatial resolution (e.g. 2 m) might often not be relevant for decision-making. Therefore, a wide range of different RS data is available and could be exploited for ES assessments, however, RS sensors should be selected

depending on the appropriate spatial and temporal scales, the spectral requirements of the ES assessed and the policy relevant scale of decision-making.

Second, given the presented advantages of RS for ES mapping and monitoring, more ES should be assessed based on this approach. Various other services have a direct or indirect link to RS such as freshwater and timber supply, water quality and aesthetic beauty and could be mapped based on this approach. However, this requires besides the use of hyper- and multispectral data also different types of RS data, for instance from active sensors like LiDAR (light detection and ranging) and SAR (synthetic aperture radar). The respective methods for the use of such data exist already, e.g. to estimate forest standing biomass using LiDAR (Babcock et al., 2016) and forest carbon stocks using SAR data (Dawson et al., 2016).

Furthermore, not only different RS data, but as well the combination of RS, social science and socioeconomic data derived from semi-structured interviews, household surveys, social media and citizen science engagement needs to be improved and extended (Cord et al., 2017). In particular, cultural ES, which are in general the least quantified of all ES (Daniel et al., 2012; Hernández-Morcillo et al., 2013), are often only indirectly related to RS data. They strongly rely on subjective evaluations that are assessed in social science with qualitative data derived from questionnaires and interviews (e.g. Fagerholm and Käyhkö, 2009). Therefore, further research should foster interdisciplinary approaches that combine different information sources such as RS, social science and social media data (e.g. Flickr), which can provide valuable progress towards a more comprehensive assessment of ES.

5.3.2 Operationalization of ecosystem service monitoring based on essential ecosystem service variables

Systematic monitoring of ES is highly needed for measuring progress towards international policy targets, for evaluating efficiency of conservation actions, agri-environmental payments and payments for ES, and for incorporating ecosystem accounting into national accounts (Cord et al., 2017; Hein et al., 2015; Karp et al., 2015; Shepherd et al., 2016; Tallis et al., 2012). This requires standardized approaches for ES mapping and consequently monitoring, where EBVs (Pereira et al., 2013) can be used as a blueprint. Such standardized methods would allow comparing

different ES assessments. Equivalent to EBVs, eESVs should be scalable, temporally sensitive, feasible and relevant capturing the different components of the ES cascade. Current approaches mainly focused on the mapping of ES supply, flow and demand (Burkhard et al., 2012; Egarter Vigl et al., 2017; Schulp et al., 2014; Stürck et al., 2014). However, changes affect firstly ecosystem properties and functioning before ES are finally altered – often with a delay. Therefore, in this thesis, the observation-based ES cascade suggests relevant classes for essential ES variables starting from ecosystem properties and functioning to biophysical ES supply and ES flow. These classes should be expanded further from the pure biophysical part to the social and even economic side of the cascade including ES demand, human benefits and economic value. Such complete ES assessments could help practitioners, stakeholders and politicians to acquire a solid information base for improved decision-making. The progress towards targets of international regulations, such as the Sustainable Development Goals and the Aichi Biodiversity Targets, could be assessed at national scale with such selected essential ES variables and the results compared among countries.

First approaches exist that proposed eESVs classes (Geijzendorffer et al., 2015) and the selection of eESVs and their classes is fostered under different initiatives such as GEO BON and IPBES. Besides ES monitoring, the selection of appropriate eESVs will allow as well the forecasting of ES change in the future.

5.3.3 Communicating ecosystem service research to policy and practice

Research on ES mapping and monitoring has provided many findings, however, concrete actions based on these results that might improve the decision-making of governmental actors, stakeholders and practitioners, are often not communicated (Hauck et al., 2013; McKenzie et al., 2014). International platforms such as IPBES strengthen the communication between research and governments by releasing thematic assessment reports (Díaz et al., 2015). However, governments are not committed to implement them, which would require translating the findings into national guidelines and actions. Therefore, communicating ES findings into local actions remains a key challenge (Ruckelshaus et al., 2015). Since ES research shifts from pure ES mapping towards monitoring, translating these findings into guidelines

for practitioners and stakeholders becomes even more important. ES changes are highly relevant e.g. for governmental actors working with agri-environmental schemes that could use ES time series data for evaluating the efficiency of payments to support more sustainable agricultural practices.

Future research might focus on stakeholder involvement in the ES mapping and monitoring process, which could increase the information flow of research findings into local measures. Involving local actors into place-based research on biodiversity knowledge has shown to increase the uptake of recommendations compared to top-down governance implementations (Danielsen et al., 2010). Incorporation of stakeholders in ES mapping and monitoring can as well trigger more applied research, which might enhance communication between scientists and practitioners. Such initiatives should not be restricted only to governmental actors, but rather contain local decision makers such as farmers, landscape planners and citizens as well as multinational companies that influence with their actions ecosystems and their services worldwide (Guerry et al., 2015). Involving stakeholders with different roles and acting at different scales is crucial to improve the communication of research findings and their implementation into practice for achieving a sustainable development of ES.

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Education

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2010 – 2013 M.Sc. in Global Change Ecology, Faculty of Biology, Chemistry and Earth Sciences, University of Bayreuth (D).

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Title: *Estimating the current and future human impact in Tanzania for wildlife corridor management.*

2007 – 2010 B.Sc. in Geoecology, Faculty of Biology, Chemistry and Earth Sciences, University of Bayreuth (D).

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Title: *Studies on silting-up of bed sediments in pearl mussel streams*

Teaching and supervision

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2015 – 2016 M.Sc. supervision, Tiziana Gees.
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Previous research and professional experience

2011 – 2013 Student research assistant, University of Bayreuth, Department of Hydrology (D).

2012 Internship, Zoological Society of London, London (UK).

Internship, German Aerospace Center (DLR), Earth Observation Center, Land Surface, Oberpfaffenhofen (D).

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Graduate courses and training

- PhD Seminars I & II
- Principles and Theory in Geography
- Graduate School Retreat Seminars (I & II)
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- Introduction to geoprocessing scripts using ArcPython

Peer-reviewed publications (published)

Braun, D. et al. (2017) From instantaneous to continuous: Using imaging spectroscopy and *in situ* data to map two productivity-related ecosystem services. *Ecological Indicators*, Vol. 82, pp. 409-419.

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Peer-reviewed publications (accepted)

Braun, D. et al. Spatio-temporal trends and trade-offs in ecosystem services: An Earth observation based assessment for Switzerland between 2004 and 2014. *Ecological Indicators*, accepted.

Peer-reviewed publications (submitted)

Braun, D. et al. (submitted to Global Environmental Change) Drivers of ecosystem service change: The influence of climatological and non-climatological effects on ecosystem services in Switzerland between 2004 and 2014.

Conference contributions (selected)

2017 Colloquium on Remote Sensing, University of Zurich (CH). *Using remote sensing to map ecosystem services.*

10th EARSeL Imaging Spectroscopy Workshop, Zurich (CH). *Spatio-temporal Trends and Trade-offs In Ecosystem Services: An Earth Observation Based Assessment For Switzerland Between 2004 and 2014.*

2016 GEO BON Open Science Conference & All Hands Meeting, Leipzig (D). *Remote sensing for spatially explicit ecosystem service mapping - from instantaneous data to continuous estimation.*

2015 IALE-D Annual Conference, Bonn (D). *Ecosystem service mapping using imaging spectroscopy: Where do we benefit from ecosystems?*

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